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NCBC GULFPORT
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LETTER OF TRANSMITTAL FOR THE 19 JANUARY 1990 LETTER FROM U S EPA REGION
IV REGARDING INCINERATOR ASH AT THE HERBICIDE ORANGE SITE AND 22 MARCH
1990 LETTER REGARDING U S AIR FORCE RESPONSE TO U S EPA REGION IV
COMMENTS ON THE DIOXIN RISK ASSESSMENT WITH A
11/3/1990
DEPARTMENT OF THE AIR FORCE



DEPARTMENT OF THE AIR FORCE
HEADQUARTERS UNITED STATES AIR FORCE
WASHINGTON, DC 20330

NCBC Gulfport Administrative Record
Document Index Number

39501-SITE 8 INCINERATION
19.01.08.0012

19 JAN 1990

Mr. James H. Scarborough
Chief, RCRA Branch
Waste Management Division
U.S. EPA, Region IV
345 Courtland St., N.E.
Atlanta GA 30365

19.1 8.12

Dear Mr. Scarborough

1. References:

a. Your 19 Jan 90 ltr to Lt Col Lubozynski, Tyndall AFB, FL regarding the incinerator ash and Herbicide Orange site at NCBC, Gulfport, MS.

b. Our 22 Mar 90 response annotating actions to clarify your faxed comments on the dioxin risk assessment in the decision document for the NCBC site and creation of a sampling and analysis strategy for remaining HO contaminants.

2. Attachment is the final reply to your comments on the dioxin risk assessment in the draft decision document. It was prepared by A. S. Rood and D. J. Thorne, of EG&G Idaho. Please contact Mr. Jeff Short at (202) 767-0275 or Mr. Karl Kneeling at (202) 767-4151 if you have any questions. Mr. Tom Sarros of NCBC, Gulfport, MS (601) 865-2243 is the point of contact regarding site work at the HO site.

PETER WALSH, COL, USAF
Chief, Environmental Quality Division
Directorate of Engr & Svcs

1 Atch
Reply to EPA IV Questions

cc: HQ AFESC/RDV
NCBC, Gulfport, Code 470.2
SOUTH DIV NAVFACENCOM
Code 1151
MS DEQ (Mr Mabry)
EG&G Idaho (Mr Cook)

INTEROFFICE CORRESPONDENCE

DATE: August 15, 1990

TO: J. Cook

FROM: A. S. Rood *asr*

SUBJECT: REPLY TO EPA REGION IV COMMENTS ON TCDD RISK ASSESSMENT - ASR-05-90

The attached document contains the draft response to EPA Region IV Comments on the site closure risk assessment for the Naval Construction Battalion Center (NCBC) in Gulfport Mississippi.

ASR:lra

cc: N. E. Stanley *NES*
G. B. Wiersma
Central Files
Letter File
Project File



DRAFT

Reply to EPA Region IV Comments
on the Ground Water Model for NCBC

Arthur S. Ross

David J. Thorne

August 14, 1990



SUMMARY

A reply to comments submitted by EPA Region IV concerning groundwater modeling and risk assessment for 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) at the former herbicide orange storage area at the Naval Construction Battalion Center, Gulfport, Mississippi was prepared by A. S. Rood of the Environmental Assessment and Modeling Unit of EG&G Idaho and D. J. Thorne of Oak Ridge National Laboratory, Grand Junction Office (formally at EG&G Idaho). Many of the comments had little if no effect on the modeling results or conclusions on the risk due to ingestion of drinking water.

One comment however, had significant impact on the estimated drinking water risk. This comment concerned the degradation rate of TCDD in the groundwater. The degradation of TCDD in the groundwater was based on degradation rates of TCDD in the soil and surface water. Soil and surface water degradation rates are reported in the EPA Superfund Public Health Evaluation Manual. A literature investigation revealed that TCDD degradation was primarily a photolytic process. Thus, decay of TCDD in the groundwater would not be possible since groundwater is not exposed to sunlight.

New calculations were performed which did not take credit for TCDD decay in groundwater however credit was taken for TCDD decay in the surface soil. In addition, the aquifer modeled was changed from the Miocene aquifer (the primary drinking water source) to the surficial aquifer. This was done for the following reasons:

1. More data was available on the surficial aquifer.
2. Estimated contaminant concentrations in the surficial aquifer would provide a bounding case for the maximum concentration expected in the Miocene aquifer since the Miocene aquifer lies below the surficial aquifer



and the additional contaminant travel distance would result in greater dispersion of the contaminant.

7. Previous investigators consider transport of TCDD to the Miocene aquifer extremely unlikely.

A carcinogenic risk of $1.1E-06$ was calculated for the drinking water ingestion pathway. This risk was based on an individual who ingests 2 L/d of water from a well located 152 m down-gradient from the NCBC site. The TCDD concentration in the well water was assumed to be the model predicted maximum concentration for the duration of exposure. A carcinogenic risk of $1.0E-06$ is the point of departure for acceptability. The range of $1.0E-04$ to $1.0E-07$ may be considered acceptable depending on the circumstances surrounding the exposure and assumptions used in the risk assessment. A risk of $1.0E-07$ is generally considered acceptable. The risk calculated using no decay of TCDD in the groundwater was $2.6E-04$ which was slightly above the acceptable cutoff of $1.0E-04$.

The sensitivity of aquifer retardation (which depends directly on the fraction of organic carbon in the aquifer) on the carcinogenic risk was investigated. This investigation revealed an inverse power relationship between the drinking water carcinogenic risk and aquifer retardation. The greater the retardation, the lower the carcinogenic risk from drinking water. The sensitivity of the retardation factor (RF) indicates that the release and transport model for TCDD is highly sensitive to the fraction of organic carbon assumed to be in the aquifer and soil. Therefore, if the analysis is to be further refined to determine risk acceptability, it is suggested that measurements of the fraction of organic carbon in the surficial aquifer and contaminated soil be made.

The data analyzed in this report indicates that several parameters in combination may be the key to determining the acceptability of the proposed TCDD action at the NCBC. The source term should be more clearly defined, the fraction of organic carbon in the surficial aquifer should be investigated,



and the actual down gradient well distance of interest should be defined as either hypothetical or actual.

In addition, the pathway of interest may no longer be drinking water. The migration of TCDD from the surficial aquifer to the Miocene aquifer is considered highly unlikely. This then indicates that the risk of drinking water from the Miocene aquifer would be considerably less than that reported for the surficial aquifer. Therefore, if the surficial aquifer is not used as a potable water source, other pathways such as irrigation, food chain, and immersion should be analyzed to determine the total carcinogenic risk.



1.0 SPECIFIC REPLY TO COMMENTS

1.1 Reply to Comment 1:

The comment concerned the degradation rate of TCDD in the groundwater and the assumption of zero concentration initial condition.

Degradation rates for TCDD in soil and surface water were reported in the Superfund Exposure Assessment Manual draft version printed in 1988. Subsequent versions of this document have omitted the Physical and Chemical data for Contaminants. Schaum (1984) reported half lives for TCDD in soil of 10 to 12 years. This process was attributed to photolytic (sunlight) degradation, thus degradation of TCDD in the groundwater would not occur by this process. In the original risk assessment, degradation in the groundwater was assumed to occur at some rate between the soil and surface water rates. The authors were not aware of the mechanism by which TCDD degrades at the time the original transport calculations were performed, thus new transport calculations were performed that ignore degradation in the groundwater. Degradation in the soil was taken into account since TCDD on the surface is susceptible to degradation from sunlight. These calculations are presented in Section 2 of this document.

The zero concentration initial condition was assumed because of lack of any surficial aquifer field sampling data. This assumption was not necessarily conservative and may not be valid. Field sampling data however is needed for a starting point and this data is not available at this time.

1.2 Reply to Comment 2, 4, 7, 8:

The comments concerned the following: a) effect of a fluctuating water table, b) where the drinking water aquifer is recharged, c) verification c



dispersivity equations and d) justification for the lateral and transverse dispersivity, pore velocity and aquifer thickness. The reply to these comments are somewhat related, thus they will all be addressed together in this section.

The model used for the groundwater transport calculations was not designed to handle a fluctuating water table explicitly. The source and transport model used in the analysis ignored the unsaturated zone. That is, transport from the soil surface to the upper surface of the aquifer was ignored and the contaminant was assumed to enter the aquifer after it had been leached from the soil. This assumption is very conservative since it does not account for retardation, dilution and dispersion of the contaminant in the unsaturated zone. Thus, a deeper or shallower water table would not effect the predicted release to the groundwater.

In general, a fluctuating water table effects the aquifer thickness and gradient. Lower gradients result in lower pore velocity and somewhat higher contaminant concentrations due to lower hydrodynamic dispersion. A thinner aquifer results in higher concentrations due to a decrease in the vertical mixing area. Since the model assumes uniform mixing in the vertical (z) direction, a smaller aquifer thickness will result in somewhat higher concentrations. The sensitivity of the model to aquifer thickness and pore velocity was examined and the results are presented in Section 3.

In the original analysis, the contaminant was assumed to enter the Miocene aquifer. This aquifer is the source for domestic and industrial water needs in the Gulfport area (Barracough, 1986). Producing zones are variable but generally are at depths ranging from 649 ft to 1196 ft. The main recharge area for the Miocene aquifer is several miles north of Gulfport. Recharge occurs by infiltration of rain that falls on sandy outcrops. This aquifer maintains sufficient artesian head pressure to support flowing wells in the Gulfport area. Beds which contain the Miocene aquifer include the Citronelle and Graham Ferry Formation. These



beds strike east-southeast and dip south-southwest at about 90 feet per mile near Gulfport.

The permeable portion of the near surface layers has been called the surficial aquifer. From the information presented by Barraclough, downward migration of dioxin to the Miocene aquifer is considered to be a remote possibility. In addition, under the geochemical and hydrological conditions present in the Gulfport area, significant movement of dioxin down to the principal pumping zones is not considered to be possible.

The surficial aquifer has the highest potential for contaminants to be transported according to Barraclough. The surficial aquifer flows generally south, toward the Gulf of Mexico. Little use is made of water in the surficial aquifer down gradient from the MCBG. Some lawn or garden irrigation may occur. The nearest irrigation well is estimated to be about 1/2 mile (800 m) from the Herbiside Storage Area.

Since the drinking water pathway was to be assessed in the original analysis, modeling was performed in the Miocene aquifer. For conservatism, dioxin was assumed to leach from the soil and enter the Miocene aquifer. The original analysis ignored the effects of the surficial aquifer and any unsaturated media between the ground surface and the Miocene aquifer. These simplifications were conservative since migration in the surficial aquifer and unsaturated zone only served to further dilute and retard the contaminant.

Groundwater flow parameters in the original analysis for the Miocene aquifer could not be located in the time allotted, thus surficial aquifer parameters were used with the exception of pore velocity. Pore velocity (1.6E-05 m/s) was estimated from hydraulic gradients of the Miocene aquifer and typical hydraulic conductivity values for fine grained sandstone. The porosity of the surficial aquifer was between 0.20 and 0.30. A mean value of 0.25 was used. Lateral (30.9 m) and transverse (9.0 m) dispersivity values were selected from a table of typical



dispersivity values in Till and Meyer (1983) for alluvial sediments and distances around 305 m. Other investigators (Weeks, 1989) used these same values for a herbicide risk assessment in Florida. The geohydrologic setting for this risk assessment was very similar to the geohydrologic setting at Gulfport thus these values appear to be appropriate for use of this assessment model. In the revised transport calculations, all parameters used in the groundwater model are for the surficial aquifer.

The equations which describe dispersivity in the X and Y directions were taken from Till and Meyer (1983).

1.3 Reply to Comment 10:

The equation for the leach rate constant was obtained from the Prototype Safety Analysis Report for Below Ground Vault Low Level Waste Radioactive Waste Disposal Facility by Rogers and Associates. The equation and units of the parameters have been checked and it agrees with what is stated in the text. The equation given in the comments is difficult to compare because the terms and units have not been defined. The units of the leach rate constant are time^{-1} . This is not a ratio (which would imply no units) or a rate (which would imply units of mass/time).

1.4 Other Comments:

Comment 3 refers to a discrepancy between the aquifer description in Figure 6 and its description in the text. The Micocene aquifer has been ignored in the new calculations thus the figure and Micocene aquifer description will be eliminated.



2.0 REVISED TRANSPORT CALCULATIONS

These revised transport calculations model transport of TCDD in the surficial aquifer only. Using the surficial aquifer to evaluate risk resulting from the drinking water consumption is conservative since migration to the actual drinking water aquifer (Miocene aquifer) is a remote possibility and if migration occurred, the resulting groundwater concentrations would be lower due to additional dilution and dispersion. Transport parameters for the surficial aquifer are summarized in Table 1.

2.1 Source Term:

The source term model was revised because the available literature (Schumm, 1984) indicates TCDD degradation occurs by photolytic reaction. Surface contamination would be most susceptible to this reaction and the effectiveness of this process would decrease with soil depth. Schumm recommends using a half life of 10-12 years for TCDD contamination at or near the surface and a half life greater than 100 years for deeper contamination.

In order to account for the different degradation rates of TCDD in the soil, the source term model was broken up into two compartments: a surface compartment and a subsurface compartment. The surface compartment represented the first 6 cm of soil and was characterized by a 12 year half life. The second compartment represented soil depths from 6 to 61 cm and was characterized by a TCDD half life of 100 years. The leach rate constant (as defined in the original document) operated on both compartments as shown in Figure 1.

The differential equations which describe the time rate of change of source mass in both compartments are

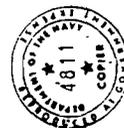


Table 1. Transport parameter values for the surficial aquifer model.

Parameter	Value	Units	Source
Hydrologic Conductivity	150	ft/day	Barracough, 1986
Hydrologic Gradient	3 - 4	ft/mile	Barracough, 1986
Thickness	24	ft	Barracough, 1986
Porosity (Mean Value)	0.20 - 0.30 0.25	---	Barracough, 1986
Pore Velocity (Mean Value)	0.3 - 0.6 0.45	ft/day ft/day	Barracough, 1986
Lateral Dispersivity	30.9	m	Till and Meyer, 1983, Weeks, 1989
Transverse Dispersivity	9.0	m	Till and Meyer, 1983, Weeks, 1989
Bulk Density	2.0	g/cc	PSI, 1967
Organic Sorption Coef	3.3E+06	ml/g	EPA, 1986
Fraction of Org Carbon	5.0E-04		Pettijohn, 1973
Retardation	1.32E04	---	calculated
Source Area	220 x 220	m	original report
Distance to Receptor	152	m	original report

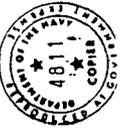
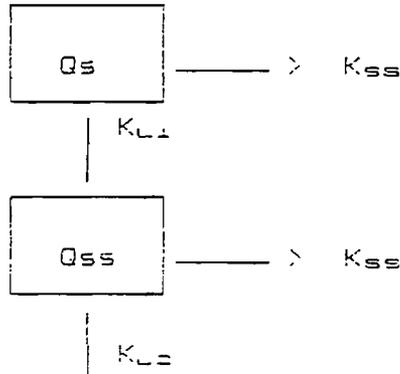


Figure 1. Revised source release model for the TCDD release from the NCBC Herbicide storage area.





$$\frac{dQ_s}{dt} = -(K_{L1} + K_s)Q_s \quad (1)$$

$$\frac{dQ_{ss}}{dt} = K_{L1}Q_s - (K_{L2} + K_{ss})Q_{ss} \quad (2)$$

where Q_s = the mass in the surface soil compartment (g)
 Q_{ss} = the mass in the subsurface soil compartment (g)
 K_{L1} = the leach rate constant for compartment Q_{ss} (s^{-1})
 K_{L2} = the leach rate constant for compartment Q_s (s^{-1})
 K_s = the decay rate constant for contamination 0-6 cm (s^{-1})
 K_{ss} = the decay rate constant for contamination 6-61 cm (s^{-1})

The solution to (1) with initial conditions $Q_s = Q_{s0}$ at $t = 0$ is

$$Q_s(t) = Q_{s0} \text{EXP}[-(K_{L1} + K_s)t] \quad (3)$$

The solution to (2) with initial conditions $Q_{ss} = Q_{ss0}$ at $t = 0$ is

$$Q_{ss}(t) = \frac{Q_{s0} K_{L1}}{(K_x - K_T)} \left[\text{EXP}[-K_T t] - \text{EXP}[-K_x t] \right] + Q_{ss0} \text{EXP}[-K_x t] \quad (4)$$

where $K_x = K_{L2} + K_{ss}$
 $K_T = K_{L1} + K_s$

The release rate (R) to the groundwater is given by

$$R = Q_{ss}(t) K_{L2} \quad (5)$$

The total mass released to the groundwater is given by

$$T = \int_0^{\infty} R dt \quad (6)$$

$$T = \frac{K_{L2}}{K_x} Q_{ss0} - \frac{K_{L1} K_{L2} Q_{s0}}{(K_x - K_T)} \left[\frac{1}{K_x} - \frac{1}{K_T} \right] \quad (7)$$

Using the values for the variables found in Table 2, the total amount released from the source to the groundwater was 0.387 g.

The amount of TCDD in each compartment at $t = 0$ was calculated by multiplying the total TCDD mass (79 g) by the ratio of the soil

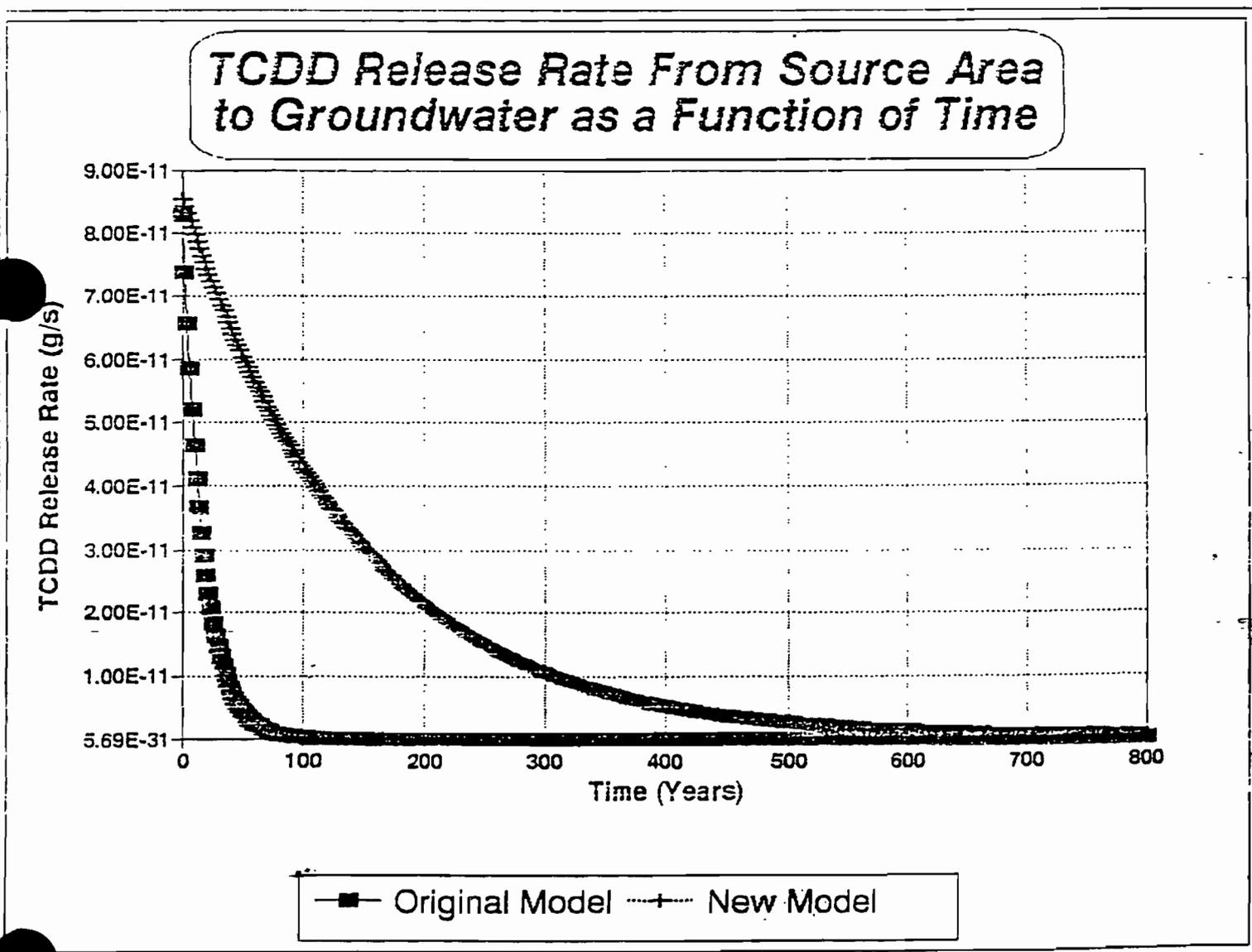


Table 2. Values for the source term parameters

Parameter	Value	Units	Source
Leach Rate (K_{L1})	3.5E-04	y^{-1}	Calculated
Leach Rate (K_{L2})	3.8E-05	y^{-1}	Calculated
TCDD 1/2 life, Surface	12	y	Schaum, 1984
TCDD 1/2 life, Buried	100	y	Schaum, 1984
Organic Sorption Coeff	3.3E-06	ml/g	EPA, 1986
Fraction of Org Carbon	4.4E-03		IT Corp., 1987
K_s	5.77E-02	y	calculated
K_{ss}	6.93E-03	y	calculated Surface
Surface Soil Thickness	6	cm	assumed
Buried Soil Thickness	55	cm	assumed
TCDD Mass	79.227	g	original report
TCDD in Surface Soil	7.793	g	calculated
TCDD in Buried Soil	71.434	g	calculated



Figure 2. Release rate as a function of time from the Herbicide Storage Area to the surficial aquifer for the original model and the revised model.





compartment thickness to the total contamination thickness (61 cm).

The organic sorption coefficient (f_{oc}) of TCDD is relatively high ($5.3E06$ ml/g). This results in a high retardation factor for TCDD transport in groundwater. The retardation factor is given by

$$RD = 1 + K_d \rho / n_e \quad (6)$$

where K_d = the linear sorption coefficient (ml/g)

ρ = the density of the medium (g/cm^3)

n_e = the effective porosity (total porosity is assumed to equal effective porosity)

The K_d value is derived from the K_{oc} by the relationship

$$K_d = (K_{oc})(f_{oc}) \quad (7)$$

where f_{oc} = the fraction of organic carbon in the media.

For high K_{oc} values, the R_d factor is directly proportional to the f_{oc} . A f_{oc} value of $4.4 E-03$ was reported for soils in the NCBC area. This value was used in the source term calculations and for the groundwater calculations in the original report. No direct measurements were made of the f_{oc} in the surficial aquifer, thus using the f_{oc} value for overlying soils may not be appropriate for groundwater simulation in the surficial aquifer since sandstone and alluvial gravels typical contain less than 0.1 % organic carbon (Pettijohn, 1973). The f_{oc} value for surface soil may also be inappropriate since this value is based on only one measurement. In these calculations, the fraction of organic carbon was assumed to be $4.4E-03$ for surface soils in the source term model and $5.0E-04$ for the surficial aquifer. The R_d factor for the surficial aquifer is



$$R_0 = 1 + \frac{(5.0E-04)(3.3E06 \text{ ml/g})(2 \text{ g/cc})}{0.25} = 1.33E04$$

and the leach rate constant for the first 6 cm of soil is

$$K_{L1} = \frac{58 \text{ cm/y}}{(6 \text{ cm})(0.5) + (2.0)(4.4E-03)(3.3E06 \text{ ml/g})(6 \text{ cm})} = 3.5E-04 \text{ y}^{-1}$$

2.2 Estimated TCDD Concentrations and Carcinogenic Risk:

The release rate as a function of time (Figure 2) shows that the effective release time is approximately 800 years. It was found that this time was insignificant when compared to the groundwater concentration peak time of 10,000 y (The time the groundwater concentration reaches its maximum value). Thus, release of the contaminant could be treated as a single pulse event released at time equal = 0. Simulations used in this analysis are all single pulse release events and do not use any of the numeric integration routines and methods described in the original document. The recalculated peak TCDD concentration was 2.61E-10 mg/L which occurred 10,2 years after release from the source area.

The Chronic Daily Intake (CDI) of TCDD from the consumption of drinking water was calculated according to the Human Health Evaluation Manual, Part 2 (EPA, 1980). The CDI was determined by the following equation

$$CDI = \frac{CW \cdot IR \cdot EF \cdot ED}{BW \cdot AT} \quad (10)$$

where CDI = Chronic Daily Intake (mg/kg/d)

CW = groundwater concentration (assumed to be the maximum predicted concentration 2.61E-10 mg/l)

IR = standard drinking water intake (2 l/d)

EF = the exposure frequency (365 days/year)

ED = exposure duration (70 years)



SW = Body Weight (70 kg)

AT = period over which exposure is averaged (70 y • 365 d/y)

The potential carcinogenic risk was calculated as follows:

$$PCR = CDI \cdot PF \quad (11)$$

where PCR = potential carcinogenic risk

PF = the carcinogenic potency factor ($1.5E05 \text{ mg/kg/d}^{-1}$)

The EPA no longer uses the term "Potency Factor". Instead, the term "Slope Factor" has been adopted and is used to estimate an upper bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a potential carcinogen. The slope factor for TCDD was not listed in the March, 1990 edition of the Integrated Risk Information System (IRIS) database but was listed in the July 1990 Health Effect Summary Tables (HEST). The slope factor reported in the HEST tables was the same as the potency factor reported in the outdated Superfund Public Health Evaluation Manual ($1.5E+05 \text{ mg/kg/d}^{-1}$). The slope factor for TCDD is currently under review by EPA and may be revised in the future. The guidance given by EPA has been to use the slope factor reported in the HEST document until further data is available.

The carcinogenic risk using the HEST published slope factor and the best estimate concentration ($2.6E-10 \text{ mg/L}$) was $1.1E-06$.



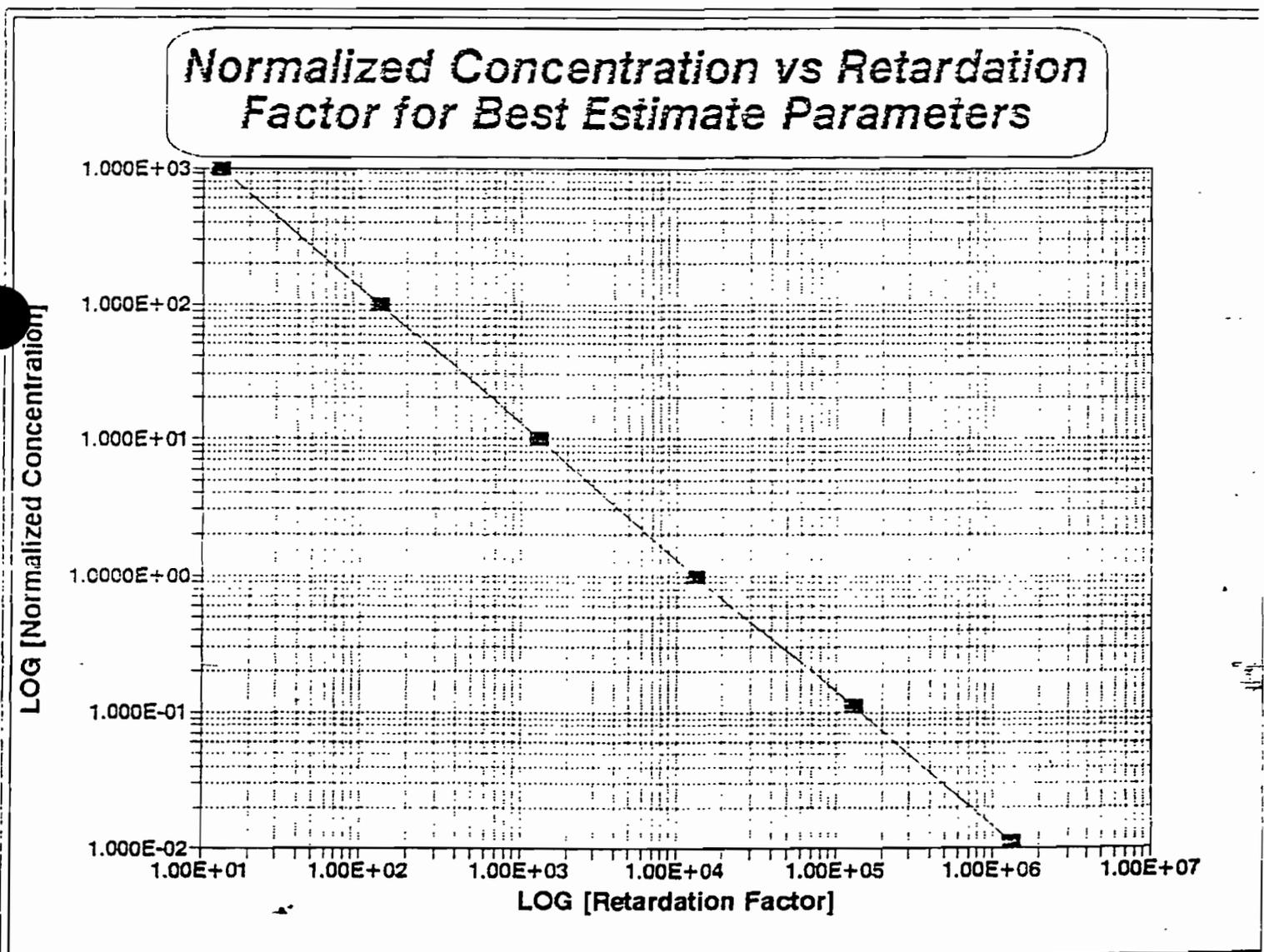
3.0 DISCUSSION OF RISK INTERPRETATION AND SENSITIVE PARAMETERS

A carcinogenic risk of $1.0E-06$ is the point of departure for acceptability. The range of $1.0E-04$ to $1.0E-07$ may be considered acceptable depending on the circumstances surrounding the exposure and assumptions used in the risk assessment. A risk of $1.0E-07$ is generally considered acceptable. The carcinogenic risk using the best estimate surficial aquifer parameters (RD of $1.32E04$ and pore velocity of $1.6E-06$) was $1.1E-06$. This risk was above the point of departure for acceptability but may be considered acceptable depending on the assumptions used.

In order to assess the degree of conservatism in the risk estimate, a parametric sensitivity analysis was performed on the model. The concentration in the following graphs (Figure 3 thru 7) has been normalized to the peak concentration calculated with the best estimate parameter values (Table 1).

The effect retardation has on peak concentration is shown in Figure 3. The retardation factor is highly dependent on the fraction of organic carbon in the aquifer. Since very little data was available on organic carbon fractions, the retardation factor was allowed a large degree of variance in order to account for all possible cases. The retardation factor shows an inverse power function relationship to peak concentration. The peak concentration time was linearly related to the retardation factor (Figure 4). Single pulse releases were used in all computer simulations shown in Figure 3. For low Rd values, the single pulse release over predicts the peak concentration because the release time becomes a significant portion of the contaminant travel time. For example, using an Rd of 1.0, the peak concentration for a single pulse release of 0.3375 g was $3.0E-06$ mg/L. The peak time was 2.5 years after the release. Using the source model described previously and the time variant release rate (Figure 2.), the peak concentration was $1.0E-07$ mg/L and occurred 12 years after the release began.

Figure 3. Peak concentration (normalized to the peak concentration using the best estimate parameters) vs retardation factor. All transport parameters (except retardation) were held constant at their best estimate value.



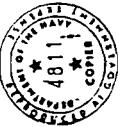
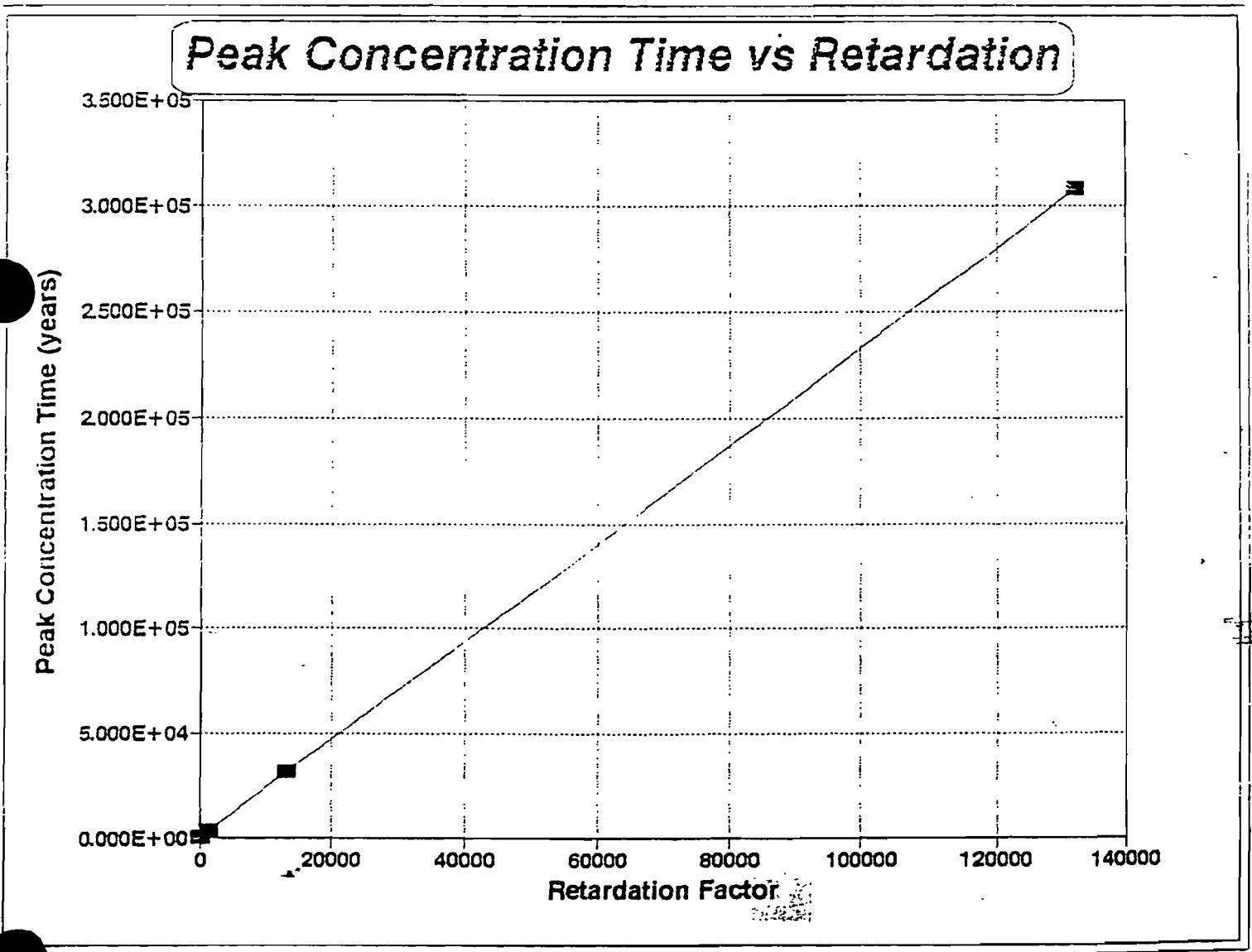


Figure 4. Peak concentration time vs retardation factor. All transport parameters (except retardation) were held constant at their best estimate value.





The fraction of organic carbon in the surface soil will also affect the release rate of TCDD to the groundwater. High concentrations of organic matter in the surface soil will tend to immobilize TCDD and if sunlight is present, photolytic degradation will decrease the contaminant mass. The response of the source model to percent organic carbon in the soil shows an exponential decrease in the integrated mass of TCDD released to the groundwater vs increasing organic carbon in the soil (Figure 5). For single pulse releases to the groundwater, the mass released is directly proportional to the potential carcinogenic risk. The relationship between carcinogenic risk and a time variant release of TCDD to the groundwater is more complex and is not discussed here.

The effect of the pore velocity was investigated to determine the model's sensitivity to this parameter. The pore velocities were varied from the low value reported in the literature of $1.1E-06$ to the high value of $2.1E-05$ m/s. The model does not appear to be highly sensitive to variability in pore velocity (Figure 6), at least in the range of reported pore velocity values since the peak concentration did not change for either velocity compared to that obtained for the best estimate velocity of $1.6E-06$ m/s.

The most permeable sections of the surficial aquifer averages approximately 7 m thick. For this analysis, it was assumed that the aquifer thickness would vary between 1 and 15 m. Aquifer thickness follows a functional relationship of $Y = 1/X$ when compared to peak concentration (Figure 7).

Distance to the receptor was also investigated. Concentrations at the nearest known well 0.5 mile (800 m) away were calculated. The peak concentration was $8.7E-11$ which was 66% less than the concentration using the 152 m receptor distance. The calculated potential carcinogenic risk due to drinking water ingestion was $3.9E-07$.



Figure 5. Mass of TCDD released to groundwater vs percent organic carbon in the surface soil for the revised source model.

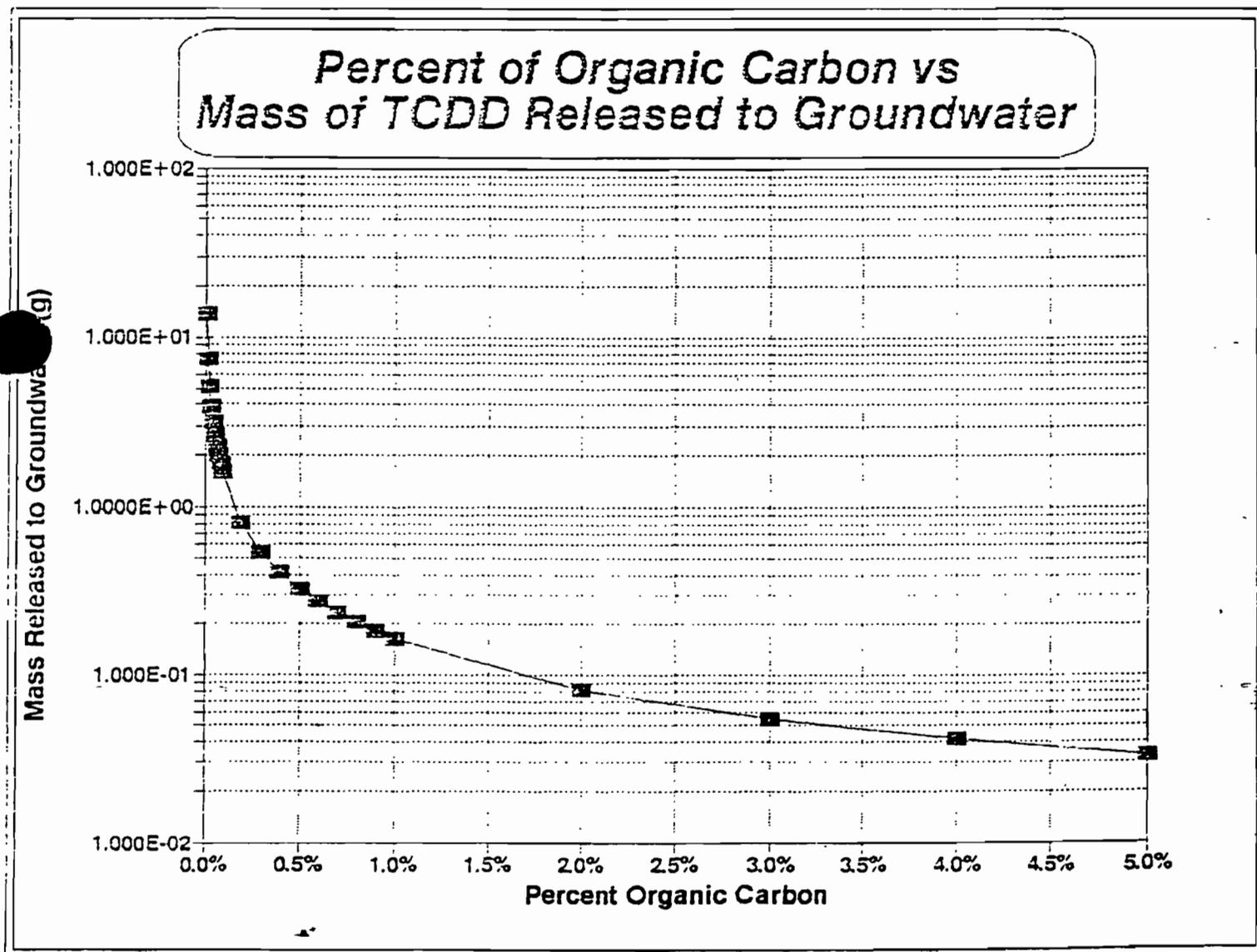




Figure 6. Peak concentration (normalized to peak concentration using the best estimate parameters) vs pore velocity. All parameters (except aquifer thickness) were held constant at their best estimate value.

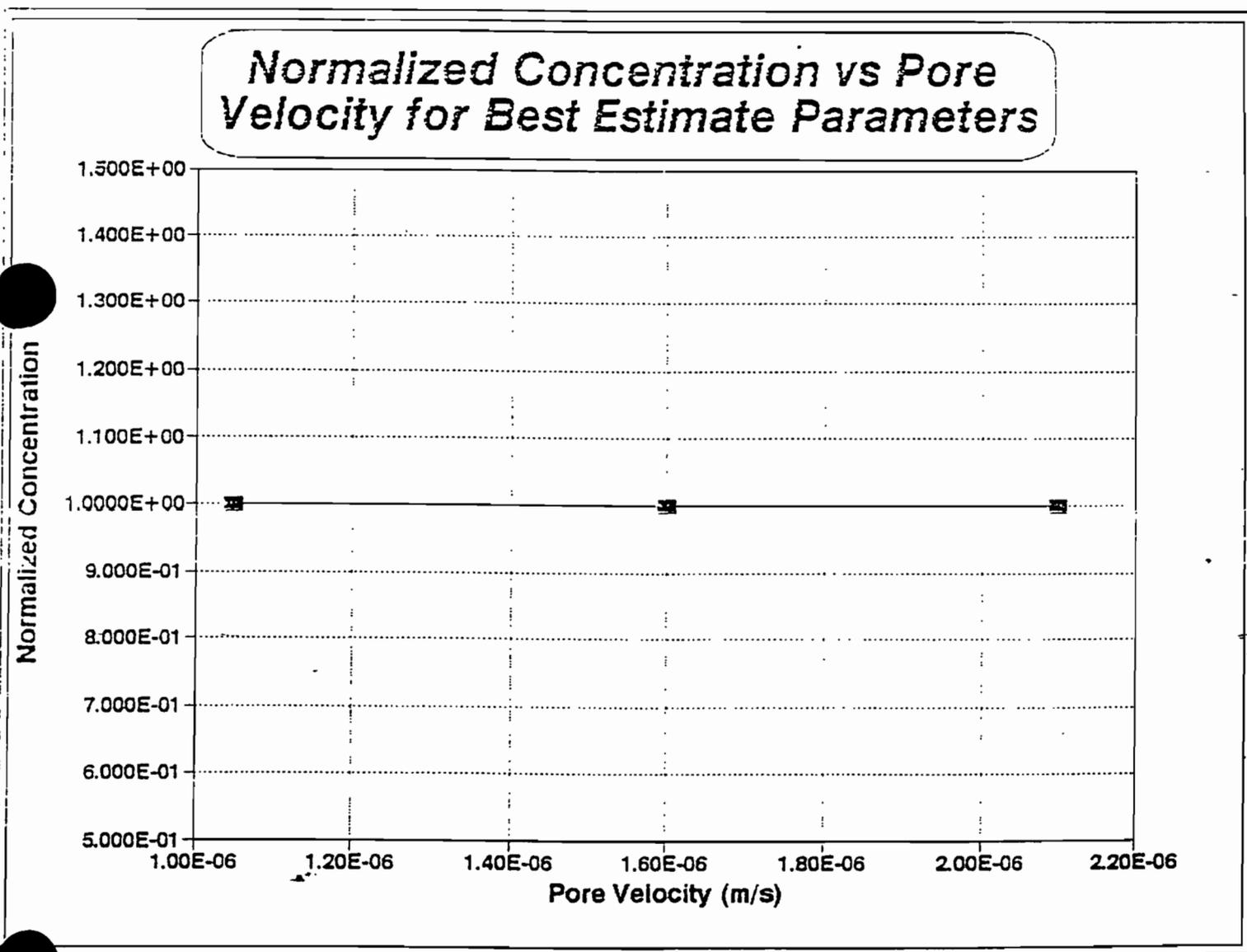
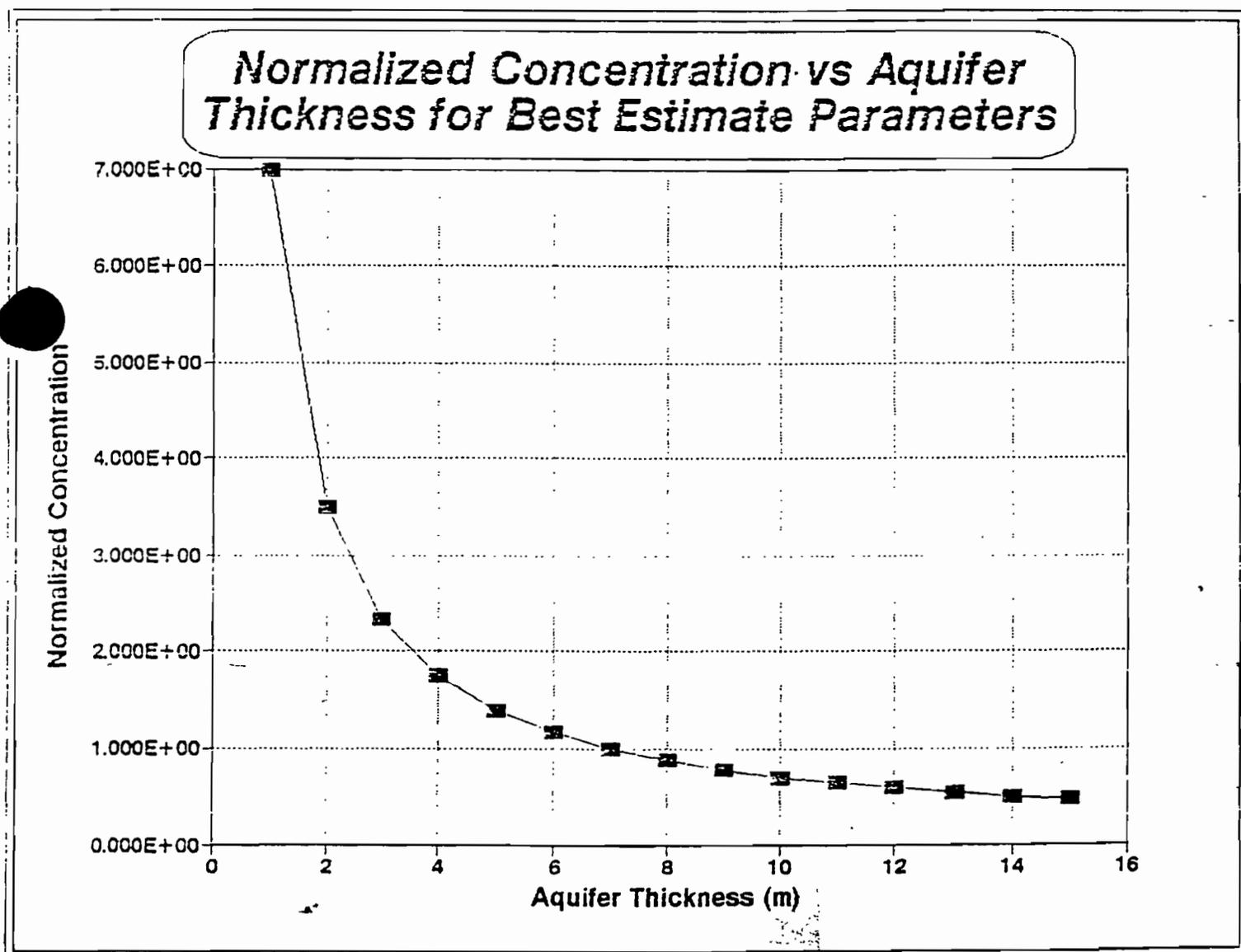


Figure 7. Peak concentration (normalized to peak concentration using best estimate parameters) vs surficial aquifer thickness. All parameters (except aquifer thickness) were held constant at their best estimate value.





4.0 CONCLUSIONS AND RECOMMENDATIONS

The potential carcinogenic risk of $1.1E-06$ was technically above the EPA's "point of departure" for acceptable risk however some consideration as to the assumptions used in the model should be made. The assumptions used to estimate the source mass were conservative. Field data to support using a more refined estimate however was lacking. The distance to the receptor was chosen to be the nearest site boundary from the center of the remediated area. This location was not in the direction of the hydrologic gradient thus transport to this hypothetical receptor would be impossible. This distance was selected for the purpose of conservatism. An actual well was presumably located a half mile downgradient from the storage facility. Identification and verification of an actual downgradient receptor location would allow credit to be taken for this parameter. In addition, some credit should be allowed for modeling the surficial aquifer as a bounding estimate of the Miocene aquifer, the primary source of drinking water in the Gulfport area. Migration to the Miocene aquifer is considered highly unlikely (Barraclough, 1984), and even if there was migration, the estimated concentrations would be lower than the surficial aquifer concentrations due to additional dispersion. There are currently no known drinking water wells in the surficial aquifer.

Decay of TCDD in the groundwater was eliminated. This parameter had a largest impact on the predicted downgradient groundwater concentrations and potential carcinogenic risk in the revised calculations. Risk estimates taking no credit for decay were increased by as much as 12 orders of magnitude over the previous results. Credit was taken for decay of TCDD in the surface soil. The model was changed to reflect the decay mechanism reported by Schaum. These changes resulted in 760% increase in the TCDD mass released from the facility to the groundwater. This value however was dependent on the amount of organic carbon in the surface soil. At present, only one soil sample has been analyzed for organic carbon. Because the release and subsequent migration of TCDD is very sensitive to



this parameter, it is suggested that future sampling efforts focus on characterizing the amount and distribution of organic carbon in the soil and surficial aquifer.

The most conservative analysis where the entire estimated inventory was released to the groundwater as a pulse and no retardation or decay of the source mass was considered, the risk was unacceptable. When the source term model was used to estimate releases, retardation and decay were not considered and the groundwater receptor was assumed to be at a well 800 m downgradient from the facility, then the risk was $2.6E-04$ which is slightly above the unacceptable cutoff range of $1.0E-04$.

It is likely that the aquifer contains some organic material thus the preceding risk estimate was overly conservative. Also the surface soil will likely contain greater fractions of organic carbon than what has been measured in the remediated soil. More organic material in the surface soil will increase the TCDD transit time from the soil to the groundwater. This will increase the amount of TCDD decayed in the soil and decrease the mass of TCDD released to the groundwater.

Greater knowledge of the source term, fraction of organic matter in the soil and aquifer and actual down gradient well distance of interest will result in more certainty in the risk estimate. While the current risk estimate was within the bounds for acceptability, there was still reasonable doubt and uncertainty as to the actual values of the sensitive parameters identified in this report. Agreement with the regulatory agencies as to appropriate values to be used in the analysis will add credibility and defensibility to the risk estimates.

In addition, the pathway of interest may no longer be drinking water. The migration of TCDD from the surficial aquifer to the Miocene aquifer is considered highly unlikely due to the hydrologic properties of the Miocene aquifer. This then indicates that the risk of drinking water from the Miocene aquifer would be considerably less than that reported for the

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