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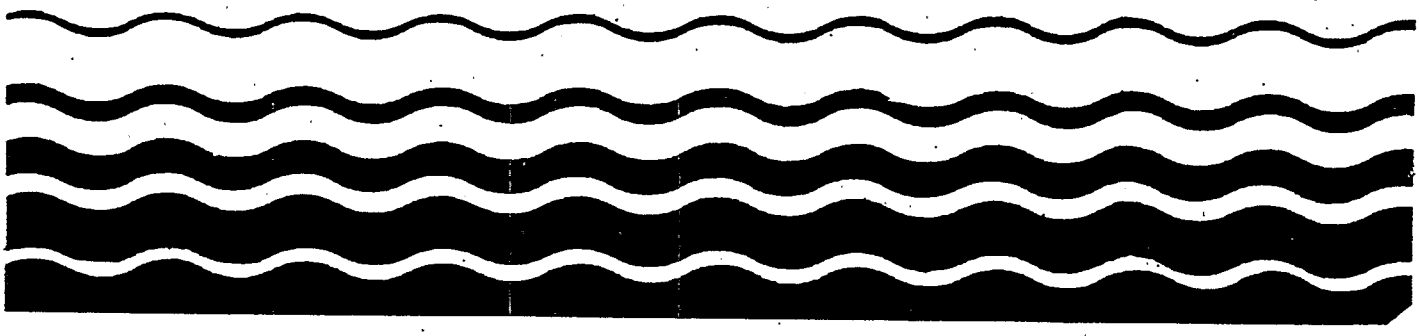
EPA Technical Support Document for the Round Two Sewage Sludge Pollutants



Technical Support Document for the Round Two Sewage Sludge Pollutants

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EXECUTIVE SUMMARY

Section 405(d) of the Clean Water Act requires the U.S. Environmental Protection Agency (EPA) to promulgate regulations, including numerical limitations for pollutants that may be present in sewage sludge, to protect public health and the environment from any reasonably anticipated adverse effects of the pollutants. The statute requires EPA to promulgate regulations in two stages. When the Agency failed to meet the statutory deadlines for promulgation of these regulations, several groups sued EPA. The Agency agreed to promulgate the first stage of regulation by November 1992, and the second stage by December 2001. Under the agreement, filed in Federal district court, EPA also agreed to identify pollutants that it was considering for regulation in the second stage by November 1995 (*Gearhart et al. v. Browner*, Civ. No. 89-6266-HO, D. Oregon).

Implementation of the Clean Water Act (CWA) has resulted in higher levels of wastewater treatment, leading to both greater pollutant removal from the wastewater before discharge and generation of larger amounts of sewage sludge. Publicly Owned Treatment Works (POTWs) currently generate 5.3 million dry metric tons of sewage sludge per year, or approximately 47 pounds per person per year.

A POTW has a number of alternative practices for the use or disposal of sewage sludge. These practices include, but are not limited to, land application, surface disposal, and incineration. Approximately 33 percent of the sewage sludge generated by POTWs is used to condition the soil or to fertilize crops grown in the soil. Approximately ten percent of the sewage sludge is placed in surface disposal units, such as surface impoundments (into which liquid sewage sludge is placed), sludge-only landfills (monofills), and piles left on land surfaces. Approximately 16 percent of the sewage sludge generated by POTWs is combusted in sewage sludge incinerators. The requirements that have to be met for these practices are described in the Standards for the Use or Disposal of Sewage Sludge (40 CFR Part 503), which were published under the authority of section 405(d) of the CWA. These standards are known as the "Round One sewage sludge regulation." Most of the remaining sewage sludge generated by POTWs is co-disposed in landfills, which have requirements established by the Solid Waste Disposal Facility Criteria (40 CFR Part 258 and Part 503.4).

In November 1992, EPA promulgated numerical limits for ten pollutants in sewage sludge. At the same time, the Agency promulgated an operational standard for total hydrocarbons as a surrogate for limits on organic pollutants in the exit gas from sewage sludge incinerators. EPA is now required to make a determination as to whether it is necessary to propose and promulgate a regulation covering a second set of pollutants that may cause adverse effects to public health or the environment (Round Two). A candidate list of pollutants for the second round of the sewage sludge regulations was provided to the District Court in Oregon in May 1993. The final list of pollutants was provided to the District Court in Oregon in November 1995. The purpose of this Technical Support Document is to provide information on how both the candidate list and the final list of pollutants for the Round Two sewage sludge regulation were derived.

To select the candidate pollutants for Round Two, a Preliminary Hazard Identification study was conducted. First, pollutants that were detected frequently in sewage sludge were determined by using data from the 1988 National Sewage Sludge Survey (NSSS). Analytical results for the 411 pollutants for which samples were analyzed in the NSSS were reviewed to determine the frequency of detection for each pollutant. Pollutants with a frequency of detection of less than ten percent were deleted from further consideration for the Round Two pollutant list. There were 254 pollutants that were not detected, and another 69 pollutants that had a frequency of detection of greater than zero but less than ten percent. If a pollutant had a frequency of detection of ten percent or greater, and had not already been regulated in Round One, then scientific literature was reviewed to determine whether there were toxicity data for a pollutant. If no human health or ecological data were found for a pollutant, no further consideration was given to that pollutant for the Round Two list. Based on the results of the Preliminary Hazard Identification, a list of 31 Round Two pollutant candidates was submitted to the District Court in Oregon in May, 1993.

The 31 pollutant candidates identified in the Preliminary Hazard Identification study were then evaluated in a Comprehensive Hazard Identification study to determine the final list of pollutants for the Round Two sewage sludge regulation. In the Comprehensive Hazard Identification study, a quantitative risk assessment, including dose-response evaluation, exposure assessment, and risk characterization, was performed. The goal was to identify pollutants that may potentially cause human health or ecological risk for a Highly Exposed Individual (HEI). The risk to the HEI was estimated using a combination of high-end and average assumptions designed to give a plausible estimate of the individual risk at the upper end of the risk distribution (e.g., above the 90th percentile of the actual distribution). In general, high-end assumptions were used to characterize sewage sludge concentrations and certain exposure parameters, while average values were typically used to characterize use/disposal practices and soil and meteorological characteristics. Sewage sludge concentrations were based on the 95th percentile concentrations of pollutants obtained in the NSSS, with non-detects set equal to the minimum level (e.g., the minimum concentration of pollutant that could be measured).

For land application, risks were estimated for 15 exposure pathways. For surface disposal, risks were estimated for two exposure pathways, and for incineration, one exposure pathway. If risk values were greater than certain thresholds for a given pollutant and exposure pathway, that pathway was defined as "critical" for that pollutant. The threshold for carcinogens was an individual risk of 1×10^{-4} or higher; for non-carcinogens, a ratio of exposure to the Risk Reference Dose of one or greater; and for ecological risk, a Risk Quotient of one or greater. Critical pathways were identified for 12 pollutants. These 12 pollutants were then evaluated further. As a result of this evaluation, EPA reported in November 1995 to the District Court in Oregon that it only was considering proposing regulation of two pollutants, dioxins/dibenzofurans and coplanar polychlorinated biphenyls, in Round Two.

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LIST OF MATHEMATICAL SYMBOLS

α_i	=	intermediate diffusivity variable defined in Eq. 4-31 (m^2/sec),
ΔM_{ct}	=	change in total mass of pollutant in soil (kg),
Δt	=	one year,
μ_a	=	viscosity of air (g/cm-sec),
ρ_a	=	density of air (g air/cm^3 air),
ρ_{sl}	=	particle density of sewage sludge (kg sewage sludge/m^3 sewage sludge),
ρ_{ss}	=	particle density of sewage sludge-soil mixture (kg/m^3),
ρ_w	=	density of water (kg water/m^3 water),
σ_z	=	standard deviation of the vertical distribution of pollutant concentration in air (m),
θ_a	=	air-filled porosity (dimensionless),
θ_{ca}	=	air-filled porosity of cover layer of soil (dimensionless),
θ_{ct}	=	total porosity of cover layer of soil (dimensionless),
θ_e	=	effective porosity of soil (dimensionless),
θ_t	=	total soil porosity (dimensionless),
θ_w	=	water-filled porosity (dimensionless),
A	=	area of surface impoundment (m^2),
AA_{ij}	=	estimated ambient air concentration of pollutant j in grid cell i due to sewage sludge incineration (μg pollutant/m^3 air),
AA_j	=	average ambient air concentration of pollutant j due to sewage sludge incineration (μg pollutant/m^3 air),
A_{mono}	=	area of monofill (m^2),
A_{site}	=	area of land application site (ha),
A_{ws}	=	area of the watershed (ha),
AR	=	annual whole sludge application rate of sewage sludge to land (dry Mg sewage sludge/ha-yr),
AT	=	averaging time (yr),
b	=	model parameter that is equal to the integer part of $((N_{site}-1)/c)$, where N_{site} is the site life (yr),
BAF	=	pollutant-specific bioaccumulation factor (L water/kg fish),
$BACC_j$	=	bioaccumulation factor for pollutant j (mg pollutant/kg soil organisms per mg pollutant/kg sewage sludge-amended soil),
BCF	=	pollutant-specific bioconcentration factor (L water/kg fish),
BD_{mix}	=	bulk density of sewage sludge-amended soil (kg soil/m^3 total bulk soil volume),
BD_{sludge}	=	bulk density of sewage sludge (kg sewage sludge/m^3 sewage sludge),
BD_{soil}	=	bulk density of soil (kg soil/m^3 soil),
BS_j	=	background concentration (dry weight) of pollutant j in soil (mg pollutant/kg soil),
BW	=	body weight (kg),
BW_{infant}	=	average body weight of the infant during the exposure period (kg),
c	=	application interval (number of years between sewage sludge applications),
C_l	=	total concentration of pollutant (adsorbed and dissolved) in liquid layer of impoundment (kg pollutant/m^3 liquid layer),

C_2 = total concentration of pollutant in solids layer (kg pollutant/m^3 solids layer),
C_a = concentration of gaseous pollutant in air-filled pore space (kg pollutant/m^3 air),
$C_{air,j}$ = average concentration of pollutant j in ambient air at the downwind edge of the site (μg pollutant/m^3 air),
$C_{esoil,j}$ = dry weight concentration of pollutant j in eroded soil entering the stream (mg pollutant/kg eroded soil),
$C_{ff,j}$ = concentration of pollutant j in fish fillets (mg pollutant/kg fish fillet),
C_i = concentration of pollutant in inflow to the impoundment (kg pollutant/m^3 sewage sludge),
C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge),
C_{lec} = concentration of pollutant in water leaching from sewage sludge-amended soil (kg pollutant/m^3 porewater),
$C_{lec,j}$ = average concentration of pollutant j in water leaching from the sewage sludge-amended soil (mg pollutant/L water),
$C_{milkfat,j}$ = concentration of pollutant j in maternal milk (mg pollutant/kg milk fat),
C_s = concentration of sorbed pollutant on sewage sludge-amended soil particles (kg pollutant/kg soil),
$C_{sep,j}$ = average concentration of pollutant j in water seeping through the bottom of the impoundment (mg pollutant/L water),
$C_{site,j}$ = concentration of pollutant j in sewage sludge-amended soil eroded from the land application site (mg pollutant/kg sewage sludge-amended soil),
$C_{sw,j}$ = concentration of pollutant j in surface water (mg pollutant/L water),
C_t = total concentration of pollutant in bulk sewage sludge-amended soil (kg pollutant/m^3 total bulk soil volume),
C_w = concentration of dissolved pollutant in water-filled pore space (kg pollutant/m^3 porewater),
$C_{wel,j}$ = concentration of pollutant j in well water (mg pollutant/L water),
CA_{jk} = concentration of pollutant j in animal product k (mg pollutant/kg animal tissue),
$CD_{forage,j}$ = tissue concentration (dry weight) of pollutant j in forage/pasture (mg pollutant/kg forage/pasture),
CD_{ij} = tissue concentration (dry weight) of pollutant j in crop i (mg pollutant/kg crop tissue),
CD_j = tissue concentration (dry weight) of pollutant j in forage/pasture (mg pollutant/kg forage/pasture),
CI_j = incremental cancer risk from pollutant j for exposed individual (incremental risk of developing cancer per lifetime of exposure),
CT_j = concentration of pollutant j in sewage sludge-amended soil (mg pollutant/kg sewage sludge-amended soil),
d = depth of incorporation of sewage sludge (cm),
d_1 = depth of liquid layer (m),
d_2 = depth of solids layer (m),
d_a = depth of aquifer (m),
d_c = depth of soil cover (m),

d_e = average rate of soil loss due to erosion from sewage sludge-amended land each year (m/yr),

d_{mf} = depth of a monofill active sewage sludge unit (m),

d_{si} = total depth of impoundment (m),

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

D_{avg} = geometric mean of the maximum dispersion ratios for 172 incinerators (μg pollutant/m^3 air per g pollutant/sec),

D_{ca} = molecular diffusivity of pollutant in air (cm^2/sec),

D_{cw} = diffusivity of pollutant in water (cm^2/sec),

D_{ei} = intermediate diffusivity variable defined in Eq. 4-30 (m^2/sec),

D_{eth} = diffusivity of diethyl ether in water (cm^2/sec),

D_f = dilution factor (dimensionless),

D_{ip} = dispersion ratio for grid cell i impacted by incinerator p (μg pollutant/m^3 air per g pollutant/sec),

DA_k = daily dietary consumption of animal product k (g dry weight animal product/day),

DC_i = daily dietary consumption of crop i (g crop tissue/day),

de = effective diameter (or fetch) of surface impoundment (m),

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

DV = rate of change in the volume of the liquid layer (m^3 liquid layer/sec),

$E_{avg,\,j}$ = average emission rate for pollutant j (g pollutant/sec),

E_{jp} = emission rate for pollutant j at incinerator p (g pollutant/sec),

ED = exposure duration (yr),

$EXP_{f,\,j}$ = exposure to pollutant j through ingestion of fish (mg pollutant/kg body weight-day),

EXP_{ij} = exposure to pollutant j for individuals living in grid cell i (mg pollutant/kg body weight-day),

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

$EXP_{w,\,j}$ = exposure to pollutant j through direct ingestion of surface water (mg pollutant/kg body weight-day),

$EXPA_j$ = exposure of animal to pollutant j (mg pollutant/kg diet),

$EXPI_j$ = infant's average daily exposure to pollutant j (mg pollutant/kg body weight-day),

$EXPO_j$ = exposure of soil-dwelling organisms to pollutant j (mg pollutant/kg sewage sludge-amended soil),

$EXPT_j$ = exposure of tractor operator to pollutant j (mg pollutant/m^3 air),

$f_{1,\,j}$ = proportion of ingested pollutant j that is stored in fat (dimensionless),

f_2 = proportion of mother's weight that is fat (kg maternal fat/kg total body weight),

f_3 = proportion of fat in breast milk (dimensionless),

$f_{4,\,j}$ = proportion of ingested pollutant j that is absorbed (dimensionless),

f_{ac} = fraction of total pollutant lost during monofill's active lifetime (dimensionless),

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

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

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

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

f_{da} = fraction of total pollutant loss during monofill's active operation attributable to degradation (dimensionless),

f_{deg} = fraction of total pollutant loss caused by degradation (dimensionless),

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

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

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

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

f_{ero} = fraction of total pollutant loss caused by erosion (dimensionless),

f_{la} = fraction of total pollutant loss during monofill's active operation attributable to leaching (dimensionless),

f_{lec} = fraction of total pollutant loss caused by leaching (dimensionless),

f_{LS} = fraction of total cumulative loading lost in individual's lifetime to all four loss processes (dimensionless),

f_{oc} = fraction of organic carbon (dimensionless),

f_{out} = fraction of total pollutant lost from the impoundment through outflow (dimensionless),

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

f_{sep} = fraction of total pollutant lost from the impoundment through seepage (dimensionless),

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

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

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

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

f_{un} = fraction of monofill's active lifetime that a typical active sewage sludge unit contains sewage sludge without soil cover (dimensionless),

f_{va} = fraction of total pollutant loss during monofill's active operation attributable to volatilization (dimensionless),

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

f_{vls} = fraction of pollutant mass that volatilizes over a human lifetime (dimensionless),

f_{vol} = fraction of total pollutant loss caused by volatilization (dimensionless),

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

F_a = the volume of fluid passing through a vertical cross-section of the aquifer oriented perpendicular to the direction of flow (m^3/sec),

FA_k = fraction of dietary consumption of animal product k produced on sewage sludge-amended soil (dimensionless),

FA_{lec} = annual average flux of pollutant leached from sewage sludge-amended soil (kg pollutant/ha-yr),

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

$FA_{vol,\ j}$ = annual average flux of pollutant j volatilizing from the sewage sludge-amended soil (kg pollutant/ha-yr),

FC_i = fraction of dietary consumption of crop i grown in sewage sludge-amended soil (dimensionless),

FD = fraction of diet considered to be soil organisms (dimensionless, kg soil organisms/kg diet),

FM = pollutant-specific food chain multiplier (dimensionless),

FR = ratio of fetch to depth (dimensionless),

FS = fraction of animal's diet that is sewage sludge (dimensionless, kg sewage sludge/kg diet),

$\dot{H}$ = Henry's Law constant (dimensionless),

H = Henry's Law constant (atm-m^3/mol),

h_j = half-life of pollutant j in adults (days),

IA = inhalation rate (m^3 air/day),

IF = daily consumption of fish fillets (kg fish fillets/day),

IM = ingestion rate of breast milk (kg milk/day),

IS = sewage sludge ingestion rate (g sewage sludge/day),

IW = volume of water ingested daily (L water/day),

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

K_{deg} = loss rate due to abiotic or microbial degradation of the pollutant on sewage sludge-amended land (yr^{-1}),

K_{deg1} = anaerobic rate of pollutant degradation in liquid layer (sec^{-1}),

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

K_{ero} = loss rate due to erosion of the pollutant from sewage sludge-amended land (yr^{-1}),

K_g = mass transfer coefficient for the gas layer (m/sec),

K_l = mass transfer coefficient for the liquid layer (m/sec),

K_{lec} = loss rate due to leaching of the pollutant (yr^{-1}),

K_{oc} = organic carbon-water partition coefficient (mL water/g organic carbon),

K_{ow} = octanol-water partition coefficient (dimensionless, mg pollutant/L octanol per mg pollutant/L water),

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

K_{ti} = total loss rate of pollutant from inactive monofill (yr^{-1}),

K_{tot} = total loss rate for the pollutant from sewage sludge-amended land (yr^{-1}),

K_{tot1} = coefficient for the total rate at which pollutant is lost from the liquid layer of a surface impoundment (m^3/sec),

K_{tot2} = coefficient for the total rate at which pollutant is lost from or stored in the solids layer of a surface impoundment (m^3/sec),

K_{va} = loss rate of pollutant due to volatilization during monofill's active operation (yr^{-1}),

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

K_{vol} = loss rate due to volatilization of the pollutant from sewage sludge-amended land (yr^{-1}),

K_{voll} = rate of pollutant volatilization from liquid layer (m/sec),

LF = active lifetime of monofill (yr),

LS = average human lifetime (yr),

$m_{background,\ j}$ = maternal intake of pollutant j from sources other than sewage sludge (mg pollutant/kg body weight-day),

M_{avg} = geometric mean of sewage sludge feed rates for 172 incinerators (kg sewage sludge/yr),

M_{ca} = mass of gaseous pollutant (kg),

M_{cs} = mass of sorbed pollutant (kg),

M_{ct} = total mass of pollutant in soil (kg),

M_{cw} = mass of dissolved pollutant (kg),

ME_{site} = rate of soil loss for land treated with sewage sludge (kg sewage sludge-amended soil/ha-yr),

ME_{ws} = estimated rate of soil loss (erosion) for the watershed (kg soil/ha-yr),

M_{LF} = mass of pollutant in sewage sludge/soil at end of monofill's active lifetime (kg pollutant/ha),

M_{LS} = mass of pollutant in soil at end of a period equal to an individual lifetime (mg pollutant/ha),

M_p = mass of sewage sludge incinerated at incinerator p each year (kg sewage sludge/yr),

M_s = mass of soil (kg),

MS = mass of soil in mixing zone of one hectare of land (Mg soil/ha land),

$m_{sludge,\ j}$ = maternal intake of pollutant j from relevant sewage sludge exposure pathways (mg pollutant/kg body weight-day),

MW = molecular weight of pollutant (g/mol),

n = number of incinerators modeled,

N = total number of years sewage sludge is applied to land (yr),

N_{site} = site life (yr),

Na = total average emissions from the soil surface over time interval t_e (kg pollutant/m^2 soil),

Na_s = emissions from the soil surface in first second (kg pollutant/m^2),

Na_y = total average emissions from the soil surface in first year (kg pollutant/m^2),

NR = annual recharge to groundwater (m^3 recharge/m^2 area-yr, or m recharge/yr),

P_1 = fraction of solids (by mass) in liquid layer (kg solids/kg liquid layer),

P_2 = fraction of solids (by mass) in solids layer (kg solids/kg solids layer),

P_f = ratio of pollutant concentration in fillet to whole fish (dimensionless),

PIL_j = intake level of pollutant j in insectivorous mammal's diet (mg pollutant/kg diet),

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

q_{co}	=	pollutant flux from treated soil for covered period (kg pollutant/m^2-sec),
q_j^*	=	human cancer potency of pollutant j (mg pollutant/kg body weight-day)$^{-1}$,
q_{un}	=	pollutant flux from sewage sludge/soil for uncovered period (kg pollutant/m^2 unit-sec),
Q_i	=	rate at which sewage sludge enters the impoundment (m^3 sewage sludge/sec),
Q_o	=	rate at which outflow leaves the impoundment (m^3 sewage sludge/sec),
Q_{sep}	=	rate of seepage beneath the impoundment (m/sec),
r'	=	distance from center of the land application site to the downwind edge (m),
R	=	gas constant (L-atm/mol-K),
R_j	=	combined removal efficiency for pollutant j of furnace and control devices expressed as fraction of original pollutant mass retained by the furnace or pollution control devices (dimensionless),
R_{jp}	=	combined removal efficiency for pollutant j of furnace and control devices for incinerator p expressed as fraction of original pollutant mass retained by the furnace or pollution control devices (dimensionless),
RfD_j	=	Risk Reference Dose for pollutant j (mg pollutant/kg body weight-day),
RNC_j	=	ratio of the exposure to the RfD for pollutant j (dimensionless),
RQ_j	=	ecological risk quotient for pollutant j (dimensionless),
RTC_j	=	ration of the exposure to the TLV-TWA for pollutant j (dimensionless),
S_1	=	concentration of solids in liquid layer (kg/m^3),
S_2	=	concentration of solids in solids layer (kg/m^3),
Sc_G	=	the Schmidt number on the gas side (dimensionless),
S_{site}	=	sediment delivery ratio for the land application site (dimensionless),
S_{ws}	=	sediment delivery ratio for the watershed (dimensionless),
SC	=	estimated mass of sewage sludge contained in one hectare of completed monofill (kg/ha),
SRR	=	source-receptor ratio (sec/m),
t	=	time (yr),
t_e	=	duration of emissions (sec),
t_{un}	=	time that a typical active sewage sludge unit contains uncovered sewage sludge (yr),
T	=	temperature (Kelvin),
TDA	=	total exposure of tractor operator to soil dust (mg soil dust/m^3 air),
TF	=	estimated active lifetime of surface impoundment (sec),
$TLV\text{-}TWA_j$	=	Threshold Limit Value-Time Weighted Average for pollutant j (mg pollutant/m^3 air),
TP	=	duration of "square wave" for approximating the loading of pollutant into the unsaturated soil zone (yr),
TP_N	=	total mass of pollutant available at a site after the final year of application (mg pollutant/ha),
TRV_j	=	toxicological reference value for pollutant j for an animal (mg pollutant/kg diet),
TSS	=	concentration of total suspended solids in the stream (mg solids/L water),
u	=	average wind speed (m/sec),
u_{10}	=	average wind speed 10 m above surface (m/sec),

$UC_{forage,j}$	=	rate of uptake of pollutant j into tissue of forage/pasture (mg pollutant/kg dry weight forage/pasture per mg pollutant/kg soil),
UC_{ij}	=	rate of uptake of pollutant j into tissue of crop i (μg pollutant/g dry weight crop tissue per μg pollutant/g sewage sludge-amended soil),
U_{jk}	=	rate of uptake of pollutant j into animal product k (mg pollutant/kg dry weight animal tissue per mg pollutant/kg dry weight diet),
v	=	vertical term for dispersion of pollutant in air (dimensionless),
v_h	=	regional velocity of horizontal groundwater flow (m/sec),
v_i	=	superimposed radial velocity from water seeping from impoundment (m/sec),
v_v	=	vertical velocity due to the source (m/sec),
V_a	=	volume of air in soil (m^3),
V_s	=	volume of solids in soil (m^3),
V_t	=	total bulk volume of soil (m^3),
V_w	=	volume of water in soil (m^3),
x	=	distance from the center of the land application site to the downwind edge (km),
x_y	=	lateral virtual distance to land application site (m).

1. INTRODUCTION

Under section 405(d) of the Clean Water Act, the U.S. Environmental Protection Agency (EPA) is required to promulgate regulations to protect public health and the environment from any reasonably anticipated adverse effects of pollutants in sewage sludge. In November 1992, limits were promulgated for ten pollutants in sewage sludge. At the same time, an operational standard was promulgated for total hydrocarbons as a surrogate for organic pollutants in the exit gas from sewage sludge incinerators. EPA is now required to propose and promulgate a regulation covering a second set of pollutants that may cause adverse effects to public health or the environment.

1.1 BACKGROUND

Congress adopted the Clean Water Act (CWA) to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (section 101(a), 33 U.S.C. 1251(a)). To achieve this goal, the CWA prohibits the discharge of pollutants into navigable waters except in compliance with the statute. The CWA directs EPA to promulgate regulations that establish limits on the types and amounts of pollutants discharged from various industrial, commercial, and public sources of wastewaters. In addition, the CWA requires EPA to promulgate regulations limiting pollutant discharges to sewers flowing to Publicly Owned Treatment Works (POTWs).

POTWs provide treatment to domestic sewage and other wastewaters; the effluent is generally discharged to surface waters and the residual material (i.e., sewage sludge) may be used as a fertilizer or soil nutrient or disposed by a number of practices. Sewage sludge is mostly water (approximately 90 percent), but also contains solids and dissolved substances. The chemical and biological constituents in sewage sludge depend on the composition of the wastewater that enters the treatment works and the processes used to treat the wastewater. Typically, constituents of sewage sludge include organic chemicals, organic solids, nutrients, inorganic chemicals, and disease-causing organisms or pathogens (e.g., bacteria, viruses, and helminth ova).

Implementation of the CWA has resulted in higher levels of wastewater treatment, leading to both greater pollutant removal from the wastewater before discharge and generation of larger amounts of sewage sludge. In the United States, the amount of sewage sludge generated has almost doubled since enactment of the Water Pollution Control Act Amendments in 1972. POTWs currently generate 5.3 million dry metric tons of sewage sludge per year, or approximately 47 pounds per person per year. Proper management of the use or disposal of the sewage sludge is important.

A POTW has a number of alternative practices for the use or disposal of sewage sludge. These practices include, but are not limited to, land application, surface disposal, placement in a municipal solid waste landfill, and incineration. The requirements that have to be met for all of these practices are described in the Standards for the Use or Disposal of Sewage Sludge (40 CFR Part 503), which were published under the authority of section

405(d) of the CWA. These standards are known as the "Round One sewage sludge regulation."

The Federal Water Pollution Control Act Amendments of 1972 regulated sewage sludge use or disposal in only one limited circumstance: when the use or disposal poses a threat to navigable waters. Section 405(a) of the CWA prohibited the disposal of sewage sludge if it would result in any pollutant from the sewage sludge entering navigable wasters unless in accordance with a permit issued by the EPA Administrator. In 1977, Congress amended section 405 to add a new subsection 405(d) that required EPA to develop regulations for the use or disposal of sewage sludge. The regulations had to establish guidelines that: (1) identify uses for sewage sludge, including disposal; (2) specify factors to be taken into account in determining the methods and practices applicable to each use or disposal practice; and (3) identify concentrations of pollutants that interfere with each use or disposal practice.

In 1987, Congress amended section 405 again and for the first time set forth a comprehensive program to reduce the potential public health and environmental risks from sewage sludge and to maximize the beneficial use of sewage sludge. Amended section 405(d) established a timetable for the development of the sewage sludge use or disposal guidelines. The basis for the program Congress mandated to protect public health and the environment is the development of technical requirements or standards for sewage sludge use or disposal and the implementation of the standards, in part through a permit program.

Under the current section 405(d), EPA must first identify, based on available information, toxic pollutants that may be present in sewage sludge in concentrations that may adversely affect public health and the environment. Next, for each identified use or disposal practice, EPA must promulgate regulations that specify acceptable management practices and numerical limits for sewage sludge that contains these pollutants. The management practices and numerical limits must be "adequate to protect public health and the environment from any reasonably anticipated adverse effects of each pollutant." Section 405(d) requires that EPA promulgate the sewage sludge regulations in two rounds and review the regulations periodically to identify additional toxic pollutants for regulation.

On February 19, 1993, EPA published the Round One sewage sludge regulation in the Federal Register (58 FR 9248). Subsequently, the regulation was amended on February 24, 1994 (59 FR 9095) and on October 25, 1995 (60 FR 54764).

A candidate list of pollutants for the second round of the sewage sludge regulations (i.e., Round Two) was developed in May 1993 in accordance with an agreement filed with the District Court in Oregon (*Gearhart et al. v. Browner*, Civ. No. 89-6266-HO, D. Oregon). The final list of pollutants was provided to the District Court in Oregon in November 1995, and the Round Two regulation is scheduled for proposal in December 1999 and for publication in December 2001. The Round Two regulation will be an amendment to the Round One regulation.

1.2 PURPOSE

The purpose of this document is to provide information on the list of pollutants for the Round Two sewage sludge regulation. It discusses the Round One regulation (Chapter 2), describes the methodology used to evaluate pollutants for inclusion on the candidate list of Round Two pollutants and presents the candidate list of Round Two pollutants (Chapter 3), describes the methodology used to develop the final list of Round Two pollutants (Chapters 4 and 5), and presents the final list (Chapter 6).

2. ROUND ONE SEWAGE SLUDGE REGULATION

2.1 DESCRIPTION OF A PART 503 STANDARD

The Round One sewage sludge regulation established standards for the use or disposal of sewage sludge when the sewage sludge is applied to the land (including sewage sludge sold or given-away in a bag or other container for application to the land), placed in a surface disposal site, placed in a municipal solid waste landfill, or incinerated. In general, the rule does not apply to the processing of sewage sludge before use or final disposal.

A Part 503 standard contains seven elements: general requirements, pollutant limits, management practices, operational standards, and frequency of monitoring, recordkeeping, and reporting requirements. These elements are designed to protect public health and the environment (only public health was evaluated in the case of incineration) from the reasonably anticipated adverse effects of pollutants in sewage sludge.

The general requirements in a Part 503 standard contain what are often called "administrative requirements." For example, the general requirements for land application require the preparer of the sewage sludge to notify the permitting authority about the interstate transfer of sewage sludge that is land applied. Without the general requirements, a Part 503 standard is incomplete. Thus, it is important that the regulated community understand and follow the Part 503 general requirements. Note that the general requirements are different for land application, surface disposal, and incineration.

Specific pollutant limits are established separately for each regulated use or disposal practice. EPA developed these limits based on the results of a risk assessment. For example, in the case of land application, up to 14 exposure pathways were evaluated in the risk assessment for each pollutant. The most stringent value for all of the exposure pathways for a pollutant was used to determine the limit for that pollutant in the Part 503 land application subpart.

Management practices contain requirements that have to be met at a use or disposal site, or, in the case of incineration, requirements for the incinerator. For example, one of the surface disposal management practices indicates that to protect public health, food crops cannot be grown on a surface disposal site unless otherwise authorized by the permitting authority.

Operational standards are technology-based requirements that in the judgement of the EPA Administrator protect public health and the environment. Section 405(d)(3) of the CWA indicates that when risk-based pollutant limits and management practices cannot be developed, requirements such as an operational standard can be utilized. For land application and surface disposal, the pathogen and vector attraction reduction requirements are operational standards. For incineration, the allowable concentration of total hydrocarbons in the exit gas from the incinerator stack is an operational standard.

The final three elements of a Part 503 standard help make the Round One regulation self-implementing. This means that the Part 503 requirements have to be met even in the absence of a federal permit. The frequency of monitoring requirements establish the frequency at which sewage sludge samples have to collected and analyzed, while the recordkeeping requirements indicate who has to keep what records and for how long. Finally, the reporting requirements indicate who has to report information to the permitting authority annually.

2.2 SEWAGE SLUDGE USE OR DISPOSAL PRACTICES

This section discusses the use or disposal practices for which requirements are included in the Round One sewage sludge regulation. These practices include land application, surface disposal, and incineration. These same use or disposal practices were evaluated during the development of the final list of pollutants for the Round Two regulation, as discussed in more detail in Chapter 4. Note that the placement of sewage sludge in a municipal solid waste landfill is not considered under Round Two, because the requirements have already been established by the Solid Waste Disposal Facility Criteria (40 CFR Part 258 and Part 503.4).

2.2.1 Land Application

Approximately 33 percent of the sewage sludge generated by POTWs is used to condition the soil or to fertilize crops grown in the soil. Approximately 22 percent of the sewage sludge generated by POTWs is applied to agricultural land (including pasture and range land). Approximately 11 percent of the sewage sludge generated by POTWs is applied to forests, public contact sites (e.g., parks, ball fields, and highway median strips), and reclamation sites, or is sold or given away in a bag or other container for home gardens. The method of applying sewage sludge to the land depends on the physical characteristics of the sewage sludge and the soil, and on the types of crops or vegetation grown. Liquid sewage sludge may be applied to the land surface or may be injected below the land surface. Dewatered sewage sludge is usually applied to the soil surface and either is left on the soil surface or incorporated into the soil by plowing or disking.

Ten or more States have undertaken sewage sludge application to forests, at least on an experimental field-scale level. The most extensive experience with this practice is in the Pacific Northwest. Usually, partially dewatered sewage sludge is sprayed onto established forest stands using mobile equipment.

Sewage sludge that is sold or given away in a bag or other container for application to the land often is composted with another material or heat-dried. The sewage sludge most often is used to fertilize lawns and home gardens.

2.2.2 Surface Disposal

Surface disposal is the term used to describe the placement of sewage sludge on land for final disposal. Surface disposal is used to describe a number of different practices,

including disposal in surface impoundments (into which liquid sewage sludge is placed), disposal in sludge-only landfills (monofills), and disposal in piles left on land surfaces. These practices account for the disposal of approximately ten percent of the sewage sludge generated by POTWs.

2.2.3 Incineration

Incineration is a practice that reduces the volume of sewage sludge through combustion. A sewage sludge incinerator is an enclosed device in which sewage sludge and auxiliary fuel are fired at high temperatures. During the incineration process, emissions are discharged to the atmosphere. The residual ash that is generated is not sewage sludge, by definition.

Approximately 16 percent of the sewage sludge generated by POTWs (i.e., 865,000 dry metric tons) is fired in a sewage sludge incinerator. Most of the sewage sludge incinerators employ the multiple hearth technology. Other types of sewage sludge incinerators include multiple hearth incinerators with a secondary combustion chamber, fluidized bed incinerators, and electric incinerators.

2.3 DATA GATHERING STUDIES

The results of several EPA-sponsored data gathering studies were used to support development of the Round One sewage sludge regulation. These include the "40 City Study" (U.S. EPA, 1982), Environmental Profiles and Hazard Indices (U.S. EPA, 1985), sewage sludge incinerator field studies, and the National Sewage Sludge Survey (U.S. EPA, 1990a). These studies are described below.

2.3.1 The 40 City Study

During 1979 and 1980, EPA conducted a study, known as the "40 City Study" (U.S. EPA, 1982), to determine the fate of priority toxic pollutants in POTWs. Wastewater samples were collected at 40 POTWs throughout the United States and analyzed for priority toxic pollutants. Analytical data from the study were used to determine whether priority pollutants were in the influent to POTWs and, if they were, what happened to them as they moved through the POTW.

At several of the POTWs where wastewater samples were collected, sewage sludge samples also were collected and analyzed for priority toxic pollutants. Analytical data for the sewage sludge samples were used to determine the concentrations of 40 pollutants (12 metals, six base neutral organic compounds, six volatile organic compounds, nine pesticides, and seven polychlorinated biphenyls) in sewage sludge. These concentrations were used to develop the Environmental Profiles and Hazard Indices discussed below.

2.3.2 Environmental Profiles and Hazard Indices

In the spring of 1984, EPA enlisted the assistance of Federal, State, academic, and private sector experts to determine which pollutants likely to be found in sewage sludge should be examined closely as possible candidates for Part 503 pollutant limits. These experts screened a list of 185 pollutants in sewage sludge that, if used or disposed improperly, could cause adverse public health or environmental effects. The experts were then requested to revise the list, adding or deleting pollutants, based on the potential risk to public health and the environment when sewage sludge containing a particular pollutant was applied to the land, placed in a surface disposal site, or incinerated. The Agency also requested that the experts identify the most likely route that a pollutant would travel to reach target organisms, whether human, plant, or animal. The experts recommended that the Agency gather additional information on approximately 50 pollutants. During 1984 and 1985, the Agency collected data and information from published scientific reports on the toxicity, persistence, means of transport, and environmental fate of the 50 pollutants.

EPA made an assessment of the likelihood that each pollutant would adversely affect public health or the environment by using data from the 40 City Study on the frequency of detection and concentration of pollutants in sewage sludge and preliminary information on pollutants' toxicity and persistence, the pathways by which the pollutants travel through the environment to a receptor organism (human, animal, or plant), the mechanisms that transport or bind the pollutants in the pathway, and the effects of pollutants on the target organism. For these analyses, EPA relied on simple screening models and calculations to predict the exposure to each pollutant that would occur if the pollutant were present in surface or ground water, soil, air, or food. EPA then developed a hazard index for each pollutant by comparing the predicted exposure to an Agency human health criterion, such as a drinking water standard promulgated under the Safe Drinking Water Act, or an animal or plant toxicity number, to determine whether the pollutant could be expected to have an adverse effect on public health or the environment. For purposes of this initial screening, EPA assumed: (1) conditions that would maximize the exposure of an individual human, animal, or plant to the pollutant and (2) the worst possible toxicity effects of the pollutant.

Using the hazard indices, EPA scored each pollutant and ranked them for more rigorous analysis. Two categories of pollutants were excluded from further evaluation. First, EPA excluded pollutants that had a hazard index of less than one (i.e., presented no risk to public health or the environment at the highest concentration that the Agency found in the "40 City Study" or in other available databases). Second, EPA deferred consideration of pollutants for which EPA lacked human health criteria or sufficient data. An environmental profile for each pollutant was then developed by combining environmental fate, transport, and toxicity data on each pollutant, the results of the simple screening models and calculations, and the hazard indices for that pollutant (U.S. EPA, 1985). The environmental profiles were used to determine the pollutants for which limits would be developed through a detailed risk assessment of exposure pathways.

2.3.3 Sewage Sludge Incinerator Field Studies

In 1987, the Agency initiated a series of field studies at ten sewage sludge incinerators to support the Part 503 rulemaking effort. The purposes of the on-site tests were to obtain: (1) information about the percentage of hexavalent chromium in the total chromium in the exit gas from a sewage sludge incinerator, (2) information on the percentage of nickel subsulfide in the total nickel in the exit gas from a sewage sludge incinerator, (3) total hydrocarbon (THC) emissions data for the sewage sludge incinerators, and (4) information about organic compounds in the exit gas from a sewage sludge incinerator.

Data from the studies on the exit gas THC concentrations were used as the basis for the THC operational standard in the Round One sewage sludge regulation. This standard is technology-based in that it is based on performance data from sewage sludge incinerators. Test data indicated that there is a significant correlation between THC and organic compound concentrations in the exit gas, which is important because sampling and analysis techniques are not available to identify or quantify all potential organic compounds emitted from sewage sludge incinerators, nor are toxicity data available for all compounds.

2.3.4 National Sewage Sludge Survey

In 1988, EPA conducted the National Sewage Sludge Survey (NSSS) to collect information and data necessary to produce national estimates of: (1) concentrations of toxic pollutants in sewage sludge, (2) sewage sludge generation and treatment processes, (3) sewage sludge use or disposal practices, and (4) sewage sludge treatment and use or disposal costs.

The NSSS consisted of a questionnaire survey and an analytical survey. The sample for the questionnaire survey was selected from the 11,407 POTWs in the United States, Puerto Rico, and the District of Columbia, identified in the 1986 Needs Survey as having at least secondary treatment. Secondary treatment was defined as primary clarification followed by biological treatment and secondary clarification. The sample for the analytical survey was a subset of the sample for the questionnaire survey.

POTWs in the two samples were selected using a stratified probability design based on 24 mutually exclusive groups. Membership in these groups was based on four categories of wastewater flow rates and six use or disposal practices. The flow rates and use or disposal categories were as follows:

POTW average daily flow rate categories:

- Flow rate less than or equal to one million gallons per day (MGD)
- Flow rate greater than one MGD, but less than or equal to 10 MGD
- Flow rate greater than 10 MGD, but less than or equal to 100 MGD
- Flow rate greater than 100 MGD

POTW sewage sludge use or disposal practice groups:

- Land application
- Distribution and marketing (now called sale or give-away in a bag or other container for application to the land)
- Incineration
- Monofill (sewage sludge-only landfill)
- Co-disposal landfill
- Ocean disposal

For the questionnaire survey, a 50 page questionnaire was mailed to 479 POTWs. Information collected through the questionnaire included service area, POTW operating data, sewage sludge use or disposal practices, pretreatment activities, wastewater and sewage sludge testing frequencies, and POTW financial information. In addition to the six use or disposal practices used for the stratified sampling, information was collected on co-incineration and surface disposal.

POTWs in the analytical survey were restricted to POTWs in the contiguous States and the District of Columbia. As mentioned above, the POTWs were a subset of the POTWs in the questionnaire sample. Sewage sludge samples were collected at 208 POTWs from the four flow rate categories and analyzed for 411 analytes. (Note that in 58 FR 9269, it is stated that samples were analyzed for a total of 412 analytes. In Appendix C-List of NSSS Analytes in the NSSS Data Element Dictionary for the Questionnaire and Analytical Data Bases, (U.S. EPA, 1990a), a total of 413 analytes are listed. However, the pollutants crotoxyphos and phosphorus were listed twice in Appendix C (crotoxyphos under both semivolatile organics and pesticides/herbicides; phosphorus under both metals and classicals). Therefore samples were analyzed for a total of 411 analytes.) Results of the NSSS are discussed in Chapter 3.

2.4 ROUND ONE POLLUTANTS

The Round One sewage sludge regulation included limits (or equations for calculating these limits) for 11 pollutants for one or more use or disposal practices. Not every pollutant is regulated for each practice. When the rule was proposed, limits were developed for 28 pollutants for one or more use or disposal practices. The final Round One regulation contains limits for the following pollutants:

Land application:	arsenic, cadmium, chromium (deleted from the land application subpart on October 25, 1995), copper, lead, mercury, molybdenum, nickel, selenium, and zinc
Surface disposal:	arsenic, chromium, and nickel
Incineration:	arsenic, cadmium, chromium, lead, nickel, and total hydrocarbons. Beryllium and mercury are regulated through a National Emission Standard

3.CANDIDATE LIST OF ROUND TWO POLLUTANTS

3.1 SELECTION PROCESS

The Clean Water Act requires EPA to: (1) identify pollutants that may be present in sewage sludge in concentrations that may adversely affect public health and the environment, and (2) promulgate standards, including numerical limits for the identified pollutants. To select the pollutants on the candidate list of Round Two pollutants, a Preliminary Hazard Identification study was conducted. The first step in the study was to determine the pollutants not regulated in the Round One sewage sludge regulation that are detected frequently in sewage sludge. This was done using data from the 1988 NSSS. Analytical results for the 411 pollutants for which samples were analyzed in the NSSS were reviewed to determine the frequency of detection for each pollutant. Pollutants with a frequency of detection of less than ten percent were deleted from further consideration for the Round Two pollutant list, if the results of a toxicity analysis did not indicate potential for adverse public health effects (see Appendix A). If a pollutant had a frequency of detection of ten percent or greater, the second step in the Preliminary Hazard Identification study was conducted.

In the second step, scientific literature was reviewed to determine whether there were toxicity data for a pollutant. If no human health or ecological data were found for a pollutant, no further consideration was given to that pollutant for the Round Two list.

Pollutants for which sewage sludge samples were analyzed in the NSSS are presented in Section 3.2. Section 3.3 presents the results of the Preliminary Hazard Identification study for pollutants with a frequency of detection of ten percent or greater and for which toxicity data were available.

3.2 NATIONAL SEWAGE SLUDGE SURVEY POLLUTANTS

Sewage sludge samples collected during the NSSS were analyzed for 411 analytes, as shown in Exhibit 3-1 at the end of this section. These analytes included inorganics as well as every organic, pesticide, dibenzofuran, dioxin, and polychlorinated biphenyl (PCB) for which EPA had gas chromatography and mass spectrometry (GC/MS) standards. The pollutants also were selected in consideration of the CWA priority toxic pollutants; toxic pollutants identified in the Domestic Sewage Study (U.S. EPA, 1986a); and Resource Conservation and Recovery Act (RCRA, Pub. L. 94-580) appendix VIII pollutants.

Three categories of POTWs were excluded from calculations of national estimates of pollutant concentrations in sewage sludge: Primary Samples Only, No Sludge Sampled, and Out of Business/Ineligible. In addition, data for one POTW were not entered into the NSSS database. As a result, data from only 176 of the 208 POTWs sampled were used to estimate pollutant concentrations. The rationale for excluding the three categories is described below:

Primary Samples Only - Samples of sewage sludge generated during less than secondary treatment were collected from some POTWs; for two POTWs, they were the only type of sample collected. These data were not suitable to use in determining the national estimates for pollutant concentrations in sewage sludge generated during secondary or more-advanced treatment.

No Sludge Sampled - No sewage sludge samples were collected at 24 of the selected POTWs. For 18 of those POTWs, pre-sampling phone contacts indicated that sewage sludge was not being used or disposed at these POTWs during the time of the NSSS. These 18 POTWs claimed to treat their wastewater in stabilization ponds; sewage sludge generated in a stabilization pond remains in the pond until it is removed for final use or disposal through some other practice.

Out of Business/Ineligible - Five of the selected POTWs were either closed or could not be contacted by telephone. Therefore, no data were available for those POTWs because no samples could be collected.

Sewage sludge sampling, preservation, and analytical protocols were developed specifically for the NSSS. Analytical methods 1624 and 1625 were developed for volatile and semi-volatile organics, respectively, in the sewage sludge matrix; these methods use gel permeation chromatography sample clean-up followed by isotope dilution GC/MS analyte identification and quantification. Pesticides and PCBs, and dibenzofurans/dioxins, were analyzed using analytical methods 1618 and 1613, respectively. Metals, other inorganics, and classicals were analyzed using standard EPA methods. The analytical methods were developed or adapted specifically for the sewage sludge matrix to give the most reliable, accurate, and precise measurements of the 411 analytes.

Each NSSS sample was tested by EPA contract laboratories for 411 pollutants. A minimum level, one kind of "detection limit" used by the Agency, was identified for each pollutant in the protocol of the analytical method. That minimum level, as applied to the determination of pollutants by GC/MS, is defined by the EPA's Engineering and Analysis Division in the Office of Water as the level at which the entire analytical system shall give recognizable mass spectra and acceptable calibration points. In the NSSS, the minimum level is equivalent to the minimum concentration of pollutant that could be measured.

Pollutant concentrations and minimum levels were reported in dry weight units due to differences in the percent solids in sewage sludge samples; percent solids ranged from less than one percent to 100 percent in NSSS samples. The use of dry weight units allows all sewage sludges to be evaluated on an equivalent basis. Implicit in this form of reporting is that pollutants are associated with the solid phase of sewage sludge.

For any given pollutant, the minimum levels varied due to varying volumes or masses of samples tested, dilution of the sample or extract, and matrix effects or interference. All analytical protocols specified the volume or amount of sewage sludge to be tested. When matrix interferences prevented accurate determination of pollutant concentrations, however, samples were diluted with reagent water and then reanalyzed. The minimum level for a diluted sample was then raised by the dilution factor. For example, if a sample was diluted

by a factor of 10, then the minimum level was raised by a factor of 10. Analytical protocols provided explicit guidance as to the limits of dilution.

For each pollutant, EPA statisticians calculated the frequency of occurrence as well as the median and other percentile concentrations. Non-detected values were treated in two ways to capture the full range of possible concentrations: nondetects were set to both the minimum level and to zero. These national pollutant concentration estimates reflect the distribution of pollutant concentrations in dry weight sewage sludge that is generated by secondary or more advanced wastewater treatment at POTWs. Appendix B provides details on the statistical analyses performed on the NSSS data.

EXHIBIT 3-1
Pollutants Analyzed in the National Sewage Sludge Survey

Pollutant[1]	Pollutant[1]
Acenaphthene	Biphenyl, 4-nitro
Acenaphthylene	Bismuth
Acetic acid, 2,4-dichlorophenoxy	Bis(2-chloroethoxy)methane
Acetophenone	Bis(2-chloroethyl)ether
Acrylonitrile	Bis(2-chloroisopropyl)ether
Aldrin	Bis(2-ethylhexyl)phthalate
Aluminum	Boron
Aminobiphenyl, 4-	Bromodichloromethane
Aniline	Bromomethane
Aniline, 2,4,5-trimethyl-	Butadiene, 2-chloro-1,3-
Anisidine, o-	Butanone, 2-
Anthracene	Butene, trans-1,4-dichloro-2-
Antimony	Butyl benzyl phthalate
Aramite	Cadmium
Arsenic	Calcium
Azinphos ethyl	Captafol
Azinphos methyl	Captan
Barium	Carbazole
Benzanthrone	Carbon disulfide
Benzene	Carbophenothion (Trithion)
Benzenethiol	Cerium
Benzidine	Chlordane
Benzofluorene, 2,3-	Chloroacetonitrile
Benzoic acid	Chloroaniline, p-
Benzonitrile, 3,5-dibromo-4-hydroxy-	Chlorobenzene
Benzoquinone, 2,6-di-tert-butyl-p-	Chlorobenzene, 1-bromo-2-
Benzothiazole, 2-(methylthio)	Chlorobenzene, 1-bromo-3-
Benzo(a)pyrene	Chlorobenzilate
Benzo(b)fluoranthene	Chloroethane
Benzo(ghi)perylene	Chloroethylvinyl ether, 2-
Benzo(k)fluoranthene	Chlorofenvinphos
Benzyl alcohol	Chloroform
Benz(a)anthracene	Chloromethane
Beryllium	Chloronaphthalene, 2-
BHC, alpha-	Chlorophenol, 2-
BHC, beta-	Chloropropene, 3-
BHC, delta-	Chlorpyrifos
Biphenyl	Chromium

Pollutant[1]	Pollutant[1]
Chrysene	Dichloropropene, cis-1,3-
Ciodrin	Dichloropropene, trans-1,3-
Cobalt	Dichlorvos
Copper	Dicrotophos (Bidrin)
Coumaphos	Dieldrin
Cresol, o-	Diepoxybutane, 1,2:3,4-
Cresol, p-	Diethyl ether
Crotonaldehyde	Diethyl phthalate
Crotoxyphos	Dimethoate
Cyanides (soluble salts and complexes)	Dimethoxybenzidine, 3,3'-
Cymene, p-	Dimethyl phthalate
DDD	Dimethyl sulfone
DDE	Dimethylaminoazobenzene, p-
DDT	Dimethylbenz(a)anthracene, 7,12-
Decane, n-	Dimethylphenanthrene, 3,6-
Demeton	Dimethylphenol, 2,4-
Diallate	Dinitrobenzene, 1,4-
Diazinon	Dinitrophenol, 2,4-
Dibenzofuran	Dinitrophenol, 2-sec-butyl-4,6- (Dinoseb)
Dibenzothiophene	Dinitrotoluene, 2,4-
Dibenz(a,h)anthracene	Dinitrotoluene, 2,6-
Dibromochloromethane	Dioxane, 1,4-
Dibromoethane, 1,2-	Dioxathion
Dibromomethane	Diphenyl ether
Dichloroaniline, 2,3-	Diphenylamine
Dichlorobenzene, 1,2-	Diphenyldisulfide
Dichlorobenzene, 1,3-	Diphenylhydrazine, 1,2-
Dichlorobenzene, 1,4-	Disulfoton
Dichlorobenzidine, 3,3'-	Di-n-butyl phthalate
Dichloroethane, 1,1-	Di-n-octyl phthalate
Dichloroethane, 1,2-	Di-n-propylnitrosamine
Dichloroethene, 1,1-	Docosane, n-
Dichloroethene, trans-1,2-	Dodecane, n-
Dichloronitrobenzene, 2,3-	Dysprosium
Dichlorophenol, 2,4-	Eicosane, n-
Dichlorophenol, 2,6-	Endosulfan sulfate
Dichloropropane, 1,2-	Endosulfan-I
Dichloropropane, 1,3-	Endosulfan-II

EXHIBIT 3-1
Pollutants Analyzed in the National Sewage Sludge Survey (cont'd.)

Pollutant[1]	Pollutant[1]
Endrin	Hexachlorodibenzo-p-dioxin, 1,2,3,7,8,9-
Endrin aldehyde	Hexachloroethane
Endrin ketone	Hexachloropropene
Erbium	Hexacosane, n-
Ether, 4-bromophenylphenyl	Hexadecane, n-
Ether, 4-chlorophenylphenyl	Hexanoic acid
Ethion	Hexanone, 2-
Ethyl cyanide	Holmium
Ethyl methacrylate	Indeno(1,2,3-cd)pyrene
Ethyl methanesulfonate	Indium
Ethylbenzene	Iodine
Ethylenethiourea	Iodomethane
Europium	Iridium
Famphur	Iron
Fensulfothion	Isobutyl alcohol
Fenthion	Isodrin
Fluoranthene	Isophorone
Fluorene	Isopropylnaphthalene, 2-
Fluoride	Isosafrole
Gadolinium	Lanthanum
Gallium	Lead
Germanium	Leptophos
Gold	Lindane
Hafnium	Lithium
Heptachlor	Longifolene
Heptachlor epoxide	Lutetium
Heptachlorodibenzofuran, 1,2,3,4,6,7,8-	Magnesium
Heptachlorodibenzofuran, 1,2,3,4,7,8,9-	Malachite green
Heptachlorodibenzo-p-dioxin, 1,2,3,4,6,7,8-	Malathion
Hexachlorobenzene	Manganese
Hexachlorobutadiene	Mercury
Hexachlorocyclopentadiene	Mestranol
Hexachlorodibenzofuran, 1,2,3,4,7,8-	Methapyrilene
Hexachlorodibenzofuran, 1,2,3,6,7,8-	Methoxychlor
Hexachlorodibenzofuran, 1,2,3,7,8,9-	Methyl methacrylate
Hexachlorodibenzofuran, 2,3,4,6,7,8-	Methyl methanesulfonate
Hexachlorodibenzo-p-dioxin, 1,2,3,4,7,8-	Methyl parathion
Hexachlorodibenzo-p-dioxin, 1,2,3,6,7,8-	Methylbenzothioazole, 2-

EXHIBIT 3-1
Pollutants Analyzed in the National Sewage Sludge Survey (cont'd.)

Pollutant[1]	Pollutant[1]
Methylcholanthrene, 3-	N-nitrosodi-N-butylamine
Methylene chloride	N-nitrosomethylethylamine
Methylene phenanthrene, 4,5-	N-nitrosomethylphenylamine
Methylene bis(2-chloroaniline), 4,4'-	N-nitrosomorpholine
Methylfluorene, 1-	N-nitrosopiperidine
Methylnaphthalene, 2-	Octachlorodibenzofuran
Methylphenanthrene, 1-	Octachlorodibenzo-p-dioxin
Methylphenol, 4-chloro-3-	Octacosane, n-
Mevinphos (Phosdrin)	Octadecane, n-
Mirex	Osmium
Molybdenum	Palladium
Monocrotophos	Parathion
N,N-dimethylformamide	PCB-1016
Naled (Dibrom)	PCB-1221
Naphthalene	PCB-1232
Naphthalenediamine, 1,5-	PCB-1242
Naphthoquinone, 1,4-	PCB-1248
Naphthoquinone, 2,3-dichloro-1,4-	PCB-1254
Naphthylamine, 1-	PCB-1260
Naphthylamine, beta-	Pentachlorobenzene
Neodymium	Pentachlorodibenzofuran, 1,2,3,7,8-
Nickel	Pentachlorodibenzofuran, 2,3,4,7,8-
Niobium	Pentachlorodibenzo-p-dioxin, 1,2,3,7,8-
Nitrate	Pentachloroethane
Nitrite	Pentachloronitrobenzene
Nitroaniline, 2,6-dichloro-4-	Pentachlorophenol
Nitroaniline, 2-	Pentamethylbenzene
Nitroaniline, 3-	Pentanone, 4-methyl-2-
Nitroaniline, 4-chloro-2-	Perylene
Nitroaniline, p-	Phenacetin
Nitrobenzene	Phenanthrene
Nitrobenzene, 1-chloro-3-	Phenol
Nitrofen (TOK)	Phenol, 2-methyl-4,6-dinitro-
Nitrophenol, 2-	Phenothiazine
Nitrophenol, 4-	Phenylnaphthalene, 1-
N-nitrosodiethylamine	Phenylnaphthalene, 2-
N-nitrosodimethylamine	Phorate
N-nitrosodiphenylamine	Phosmet

EXHIBIT 3-1
Pollutants Analyzed in the National Sewage Sludge Survey (cont'd.)

Pollutant[1]	Pollutant[1]
Phosphamidon	Terbufos
Phosphoric acid, trimethyl ester	Terpineol, alpha-
Phosphoric acid, tri-o-tolyl ester	Tetrachlorobenzene, 1,2,4,5-
Phosphoric triamide, hexamethyl-	Tetrachlorodibenzofuran, 2,3,7,8-
Phosphorus	Tetrachlorodibenzo-p-dioxin, 2,3,7,8-
Picoline, 2-	Tetrachloroethane, 1,1,1,2-
Platinum	Tetrachloroethane, 1,1,2,2-
Potassium	Tetrachloroethene
Praseodymium	Tetrachloromethane
Pronamide	Tetrachlorophenol, 2,3,4,6-
Propane, 1,2-dibromo-3-chloro-	Tetrachlorvinphos
Propanol, 1,3-dichloro-2-	Tetracosane, n-
Propanone, 2-	Tetradecane, n-
Propenal, 2-	Tetraethyldithiopyrophosphate
Propenenitrile, 2-methyl-2-	Tetraethylpyrophosphate
Propen-1-ol, 2-	Thallium
Propionic acid, 2-(2,4,5-trichlorophenoxy) (Silvex)	Thianaphthene
Pyrene	Thioacetamide
Pyridine	Thioxanthe-9-one
Resorcinol	Thorium
Rhenium	Thulium
Rhodium	Tin
Ruthenium	Titanium
Safrole	Toluene
Samarium	Toluene, 2,4-diamino-
Santox (EPN)	Toluidine, 5-chloro-o-
Scandium	Toluidine, 5-nitro-o-
Selenium	Toluidine, o-
Silicon	Total heptachlorodibenzofurans
Silver	Total heptachlorodibenzo-p-dioxins
Sodium	Total hexachlorodibenzofurans
Squalene	Total hexachlorodibenzo-p-dioxins
Strontium	Total kjeldahl nitrogen
Styrene	Total pentachlorodibenzofurans
Sulfur	Total pentachlorodibenzo-p-dioxins
Tantalum	Total residue
Tellurium	Total tetrachlorodibenzofurans
Terbium	Total tetrachlorodibenzo-p-dioxins

EXHIBIT 3-1
Pollutants Analyzed in the National Sewage Sludge Survey (cont'd.)

Pollutant[1]	Pollutant[1]
Toxaphene	Trimethoxybenzene, 1,2,3-
Triacontane, n-	Triphenylene
Tribromomethane	Tripropyleneglycol methyl ether
Trichlorobenzene, 1,2,3-	Trithiane, 1,3,5-
Trichlorobenzene, 1,2,4-	Tungsten
Trichloroethane, 1,1,1-	Uranium
Trichloroethane, 1,1,2-	Vanadium
Trichloroethene	Vinyl acetate
Trichlorofluoromethane	Vinyl chloride
Trichlorofon	Xylene, m-
Trichlorophenol, 2,3,6-	Xylene, o- and p-
Trichlorophenol, 2,4,5-	Ytterbium
Trichlorophenol, 2,4,6-	Yttrium
Trichlorophenoxyacetic acid, 2,4,5-	Zinc
Trichloropropane, 1,2,3-	Zirconium
Trifluralin (Treflan)	

[1] This exhibit presents the 411 specific analytes in the NSSS. In 58 FR 9269, it is stated that samples were analyzed for a total of 412 analytes. However, in Appendix C-List of NSSS Analytes in U.S. EPA, 1990a, there are 411 unique analytes out of 413 listed (crotoxyphos and phosphorus are listed twice).

3.3 RESULTS OF PRELIMINARY HAZARD IDENTIFICATION STUDY

The purpose of the Preliminary Hazard Identification study was to identify a list of Round Two pollutant candidates that would be further evaluated in the Comprehensive Hazard Identification study. Pollutants analyzed in the NSSS formed the initial set of candidates for Round Two regulation. This section discusses the methods used to reduce the initial set of candidates to the list of Round Two pollutant candidates submitted to the Court in May, 1993. These methods included removing pollutants already regulated and pollutants with analytical problems; combining pollutant congeners into classes; removing pollutants based on frequency of detection data; and ascertaining the availability of human and ecological toxicity data.

3.3.1 Pollutants Removed from Further Consideration

Several NSSS pollutants were removed from further consideration for Round Two. Total residue was deleted because it is the inert material left after all the pollutants have been extracted from sewage sludge. Total kjeldahl nitrogen, the sum of the concentrations of ammonia (NH_3) and organic nitrogen compounds in the trinegative oxidation state (N(-III)), was deleted because the concentrations of the individual compounds cannot be discerned and because nitrogen is already controlled through management practices in Part 503. The herbicide Dinoseb (2-sec-butyl-4,6-dinitrophenol) could not be analyzed in the NSSS. Because all U.S. uses of this herbicide have been cancelled, Dinoseb was eliminated from further consideration under Round Two. Lastly, the ten individual pollutants regulated under Round One, shown in Exhibit 3-2, were not considered to be pollutant candidates for Round Two because they were already regulated. This does not preclude possible re-evaluation of the Round One pollutants in the Round Two regulation, however.

3.3.2 Individual Pollutants Combined Into Classes

Two classes of pollutants, polychlorinated biphenyls (PCBs) and dioxins/dibenzofurans, were formed by aggregating individual pollutants (Exhibit 3-3). For PCBs, seven Aroclor mixtures were combined into one category. For dioxins and dibenzofurans, 25 chlorinated dioxins and dibenzofurans were combined by multiplying each congener's concentration by its corresponding toxicity equivalency factor (U.S. EPA, 1989b) and adding the resultant values over all congeners for each POTW.

3.3.3 Frequency of Detection of Pollutants

The Agency removed from further consideration all pollutants that had a frequency of detection of zero percent in sewage sludge sampled in the NSSS. A total of 254 pollutants, shown in Exhibit 3-4, fell into that category.

The remaining 114 pollutants in the NSSS were then analyzed further. The Agency used non-zero frequency of occurrence data to evaluate which of the remaining 114 pollutants should be candidates for Round Two regulation. For each pollutant in the NSSS, except dioxins/dibenzofurans, there was only one frequency of occurrence value. For dioxins/dibenzofurans, however, frequency of occurrence was calculated in two ways: (1)

dioxins/dibenzofurans were considered to be detected if at least one of the individual congeners was detected above the minimum level; (2) dioxins/dibenzofurans were considered to be detected if all of the individual congeners were detected above the minimum level. Because all of the individual congeners were not detected above the minimum level at any given POTW, the second method resulted in zero percent detection. At the other extreme, the first method resulted in 100 percent detection.

Of the 114 pollutants, the Agency deleted from further consideration those pollutants that were detected less than ten percent of the time if the results of a toxicity analysis (see Appendix A) indicated that a pollutant did not have the potential to affect public health adversely. EPA concluded that if a pollutant was not present in more than ten percent of the sewage sludge samples and is not highly toxic, the potential to adversely affect public health and the environment is low. Of the 69 pollutants deleted for this reason, 35 had a frequency of detection of two percent or less; 22 had a frequency of detection greater than two percent, but equal to or less than five percent; and 12 had a frequency of detection greater than five percent, but less than ten percent. After deleting those 69 pollutants, 45 pollutants remained as candidates for the Round Two regulation, as shown in Exhibit 3-6.

3.3.4 Available Human Health and Ecological Toxicity Data

For the 45 candidate pollutants that had frequencies of occurrence of ten percent or greater, EPA obtained available human and ecological toxicity data. For human toxicity data, EPA predominantly relied on IRIS (Integrated Risk Information System) and HEAST (Health Effects Assessment Summary Tables). EPA also attempted to obtain human toxicity data from the following sources in descending order of preference:

- Reportable Quantities/Potency Factors,
- California Environmental Protection Agency data,
- HSDB (Hazardous Substances Data Base),
- RTECS (Registry of Toxic Effects of Chemical Substances), and
- Office of Pesticide Programs data sources.

For carcinogenic effects, EPA obtained both the slope potency factor, q_1^*, in units of $(mg/kg\text{-}day)^{-1}$ for the oral and inhalation pathways, and the weight of evidence factors as classified by the Cancer Risk Assessment Guidelines (U.S. EPA, 1986b). For non-carcinogenic effects, the Risk Reference Doses (RfDs) for the oral pathway and the Risk Reference Concentrations (RfCs) for the inhalation pathway, in units of (mg/kg-day) and (mg/m^3), respectively, were obtained. RfCs were converted to units of (mg/kg-day) by assuming a person weighs 70 kg and breathes 20 m^3 of air per day. For those chemicals with only noncarcinogenic effects, it was also noted whether any positive mutagenicity studies had been reported. These human health toxicity data are shown in Exhibit 3-7.

EPA's Health and Ecological Criteria Division provided several sources of ecological toxicity data. These sources included work in support of the Great Lakes Water Quality Initiative (U.S. EPA, 1993b), "Implementation of a Chemical Ranking System" (U.S. EPA, 1990b), Oneliner in PIRANHA (U.S. EPA, 1991b), "Screening Study for Wildlife Criteria Development" (U.S. EPA, 1989f), Devillers and Exbrayat (1992), Harfenist et al. (1989),

and 40 CFR Part 131 (Federal Register, Part II, December 22, 1992). These ecological toxicity data are shown in Exhibit 3-8.

For 14 pollutants, shown in Exhibit 3-9, no human or ecological toxicity data were available; these pollutants were not considered further. In addition, there were no human toxicity data from IRIS or HEAST for the pollutant n-decane; the only available data were from the least preferable source. Therefore n-decane also was not considered further. Although human or ecological toxicity data were not available for aluminum, some evidence exists that aluminum may be a phytotoxin. Florida farmers have reported that high background concentrations and high loadings of aluminum have led to phytotoxicity and decreased crop yields. Therefore, aluminum was kept and evaluated in the Comprehensive Hazard Identification study.

3.3.5 Pollutant Candidate List for Round Two Regulation

Of the remaining 30 pollutants, seven pollutants were evaluated for some use or disposal practices under Round One; they are being re-evaluated in Round Two because they were not evaluated for all use or disposal practices (Exhibit 3-10). Beryllium was initially modeled only under the incineration pathway for the Round One risk assessment due to a lack of data; it is considered as a pollutant candidate for Round Two for the land application and surface disposal pathways. Bis(2-ethylhexyl)phthalate was not modeled under the land application pathways in Round One due to a lack of data, and therefore is considered as a pollutant candidate in Round Two. Although the PCB Aroclor mixtures were evaluated for Round One, coplanar congeners of PCBs were not evaluated; coplanar PCBs are pollutant candidates for Round Two because they have characteristics similar to those of dioxins. Four compounds considered in the Environmental Profiles ranking system (discussed in Section 2.3.2), cyanide, fluoride, methylene chloride, and phenol, are considered again in Round Two, because previously they were not fully evaluated for all use or disposal practices.

Even though it is not known if asbestos is present in sewage sludge, asbestos was raised as a possible pollutant candidate after the public comment period for the Round One rule was closed. NSSS sewage sludge samples were not analyzed for asbestos; friable asbestos, however, is reported as being released under the Toxics Release Inventory. If asbestos were present in sewage sludge, it could potentially be released and become airborne as the sewage sludge matrix breaks down. It could then undergo secondary deposition on soil and water, as well as on food and feed crops. The secondary deposition pathway was never evaluated during Round One. After including asbestos as a potential candidate, a total of 31 pollutant candidates resulted, as shown in Exhibit 3-11. Exhibit 3-12 summarizes the rationale for the number of pollutants selected as candidate pollutants for the Round Two sewage sludge regulation.

EXHIBIT 3-2
Pollutants Regulated in Round One Sewage Sludge Regulation

Pollutant
Arsenic
Beryllium[1]
Cadmium
Chromium
Copper
Lead
Mercury[1]
Molybdenum
Nickel
Selenium
Zinc
Total Hydrocarbons[2]

[1] Regulated for incineration through a National Emission Standard, which is referenced in Part 503.

[2] Note that "Total Hydrocarbons" is not a pollutant explicitly listed in the 411 analytes in the NSSS. Total Hydrocarbons encompass all organic compounds in the stack gas exiting an incinerator.

EXHIBIT 3-3
Pollutants Combined Into Classes or Removed from Further Consideration

Number of Pollutants Deleted	Explanation
6	7 Aroclor mixtures combined into one composite called "PCBs".
24	25 chlorinated dioxins and furans combined using Toxic Equivalency Factors into one surrogate called "dioxins".
1	Total residue was removed from further consideration because it is not a pollutant; it is the inert material left after all the pollutants have been extracted from sewage sludge.
1	Total Kjeldahl Nitrogen, the sum of the concentrations of ammonia (NH_3) and organic nitrogen compounds in the trinegative oxidation state (N(-III)), was removed from further consideration because the concentrations of the individual compounds cannot be discerned and because nitrogen is controlled through management practices in Part 503.
1	Dinitrophenol, 2-sec-butyl-4,6- (also called Dinoseb) was dropped from the NSSS due to analytical problems.
33	Total number of pollutants from the NSSS analytical survey removed from further consideration.

EXHIBIT 3-4
Pollutants With a Frequency of Detection of Zero Percent in the National Sewage Sludge Survey

Pollutant	Pollutant
Acenaphthene	Chloroethane
Acenaphthylene	Chloroethylvinyl ether, 2-
Acrylonitrile	Chlorofenvinphos
Aminobiphenyl, 4-	Chloromethane
Aniline	Chlorophenol, 2-
Aniline, 2,4,5-trimethyl-	Chloropropene, 3-
Anisidine, o-	Ciodrin
Aramite	Coumaphos
Azinphos ethyl	Crotonaldehyde
Benzanthrone	Crotoxyphos
Benzene	DDD
Benzenethiol	Demeton
Benzidine	Di-n-propylnitrosamine
Benzo(ghi)perylene	Diallate
Benzofluorene, 2,3-	Dibenz(a,h)anthracene
Benzonitrile, 3,5-dibromo-4-hydroxy-	Dibenzothiophene
Benzoquinone, 2,6-di-tert-butyl-p-	Dibromochloromethane
Benzothiazole, 2-(methylthio)	Dibromoethane, 1,2-
Biphenyl, 4-nitro	Dibromomethane
Bis(2-chloroethoxy) methane	Dichloroaniline, 2,3-
Bis(2-chloroethyl) ether	Dichlorobenzene, 1,2-
Bis(2-chloroisopropyl) ether	Dichlorobenzene, 1,3-
Bismuth	Dichlorobenzidine, 3,3'-
Bromodichloromethane	Dichloroethane, 1,1-
Bromomethane	Dichloroethane, 1,2,-
Butadiene, 2-chloro-1,3-	Dichloroethene, 1,1-
Butene, trans-1,4-dichloro-2-	Dichloronitrobenzene, 2,3-
Captafol	Dichlorophenol, 2,4-
Captan	Dichlorophenol, 2,6-
Carbazole	Dichloropropane, 1,2-
Carbophenothion (Trithion)	Dichloropropane, 1,3-
Cerium	Dichloropropene, cis-1,3-
Chlordane	Dichloropropene, trans-1,3-
Chloroacetonitrile	Dichlorvos
Chlorobenzene, 1-bromo-2-	Dicrotophos (Bidrin)
Chlorobenzene, 1-bromo-3-	Diethyl ether

EXHIBIT 3-4
Pollutants With a Frequency of Detection of Zero Percent in the National Sewage Sludge Survey (cont'd.)

Pollutant	Pollutant
Diethyl phthalate	Gold
Dimethoxybenzidine, 3,3'-	Hafnium
Dimethyl sulfone	Heptachlor
Dimethylaminoazobenzene, p-	Hexachlorobenzene
Dimethylbenz(a)anthracene, 7,12-	Hexachlorobutadiene
Dimethylphenanthrene, 3,6-	Hexachlorocyclopentadiene
Dimethylphenol, 2,4-	Hexachloroethane
Dinitrobenzene, 1,4-	Hexachloropropene
Dinitrophenol, 2,4-	Holmium
Dinitrotoluene, 2,4-	Indeno(1,2,3-cd)pyrene
Dinitrotoluene, 2,6-	Indium
Dioxathion	Iodine
Diphenyl ether	Iodomethane
Diphenylamine	Iridium
Diphenyldisulfide	Isodrin
Diphenylhydrazine, 1,2-	Isophorone
Disulfoton	Isopropylnaphthalene, 2-
Dysprosium	Isosafrole
Endosulfan sulfate	Lanthanum
Endrin aldehyde	Leptophos
Endrin ketone	Lindane
Erbium	Lithium
Ether, 4-bromophenylphenyl	Longifolene
Ether, 4-chlorophenylphenyl	Lutetium
Ethion	Malachite green
Ethyl cyanide	Malathion
Ethyl methacrylate	Mestranol
Ethyl methanesulfonate	Methapyrilene
Ethylenethiourea	Methoxychlor
Europium	Methyl methacrylate
Famphur	Methyl methanesulfonate
Fensulfothion	Methyl parathion
Fenthion	Methylbenzothioazole, 2-
Fluorene	Methylcholanthrene, 3-
Gadolinium	Methylene bis(2-chloroaniline), 4,4'-
Gallium	Methylene phenanthrene, 4,5-
Germanium	Methylfluorene, 1-

EXHIBIT 3-4
Pollutants With a Frequency of Detection of Zero Percent in the National Sewage Sludge Survey (cont'd.)

Pollutant	Pollutant
Methylphenanthrene, 1-	Phenacetin
Methylphenol, 4-chloro-3-	Phenol, 2-methyl-4,6-dinitro-
Mevinphos (Phosdrin)	Phenothiazine
Mirex	Phenylnaphthalene, 1-
Monocrotophos	Phenylnaphthalene, 2-
N-nitrosodi-n-butylamine	Phorate
N-nitrosodiethylamine	Phosmet
N-nitrosodimethylamine	Phosphoric acid, trimethyl ester
N-nitrosomethylethylamine	Phosphoric triamide, hexamethyl-
N-nitrosomethylphenylamine	Phosphorus
N-nitrosopiperidine	Platinum
N-nitrosomorpholine	Potassium
Naphthalenediamine, 1,5-	Praseodymium
Naphthoquinone, 1,4-	Pronamide
Naphthoquinone, 2,3-dichloro-1,4-	Propane, 1,2-dibromo-3-chloro-
Naphthylamine, 1-	Propanol, 1,3-dichloro-2-
Naphthylamine, beta-	Propen-1-ol, 2-
Neodymium	Propenal, 2-
Niobium	Propenenitrile, 2-methyl-2-
Nitroaniline, 2-	Pyridine
Nitroaniline, 2,6-dichloro-4-	Resorcinol
Nitroaniline, 3-	Rhenium
Nitroaniline, 4-chloro-2-	Rhodium
Nitroaniline, p-	Ruthenium
Nitrobenzene	Safrole
Nitrobenzene, 1-chloro-3-	Samarium
Nitrophenol, 2-	Scandium
Nitrophenol, 4-	Silicon
N,N-dimethylformamide	Squalene
Osmium	Strontium
Palladium	Sulfur
Parathion	Tantalum
Pentachlorobenzene	Tellurium
Pentachloroethane	Terbium
Pentachlorophenol	Terbufos
Pentamethylbenzene	Tetrachlorobenzene, 1,2,4,5-
Perylene	Tetrachloroethane, 1,1,1,2-

EXHIBIT 3-4
Pollutants With a Frequency of Detection of Zero Percent in the National Sewage Sludge Survey (cont'd.)

Pollutant	Pollutant
Tetrachloroethane, 1,1,2,2-	Trichloroethane, 1,1,1-
Tetrachlorophenol, 2,3,4,6-	Trichloroethane, 1,1,2-
Tetrachlorvinphos	Trichlorofon
Tetraethyldithiopyrophosphate	Trichlorophenol, 2,3,6-
Thianaphthene	Trichlorophenol, 2,4,5-
Thioacetamide	Trichlorophenol, 2,4,6-
Thioxanthe-9-one	Trichloropropane, 1,2,3-
Thorium	Trimethoxybenzene, 1,2,3-
Thulium	Triphenylene
Toluene, 2,4-diamino-	Tripropyleneglycol methyl ether
Toluidine, 5-chloro-o-	Trithiane, 1,3,5-
Toluidine, 5-nitro-o-	Tungsten
Toluidine, o-	Uranium
Toxaphene	Vinyl acetate
Tribromomethane	Vinyl chloride
Trichlorobenzene, 1,2,3-	Ytterbium
Trichlorobenzene, 1,2,4-	Zirconium

EXHIBIT 3-5
Pollutants With A Frequency of Detection of Less Than Ten Percent in the National Sewage Sludge Survey

Pollutant	Frequency of Detection (%)
Acetophenone	2
Aldrin	3
Anthracene	2
Azinphos methyl	2
Benz(a)anthracene	4
Benzo(a)pyrene	3
Benzo(b)fluoranthene	7
Benzo(k)fluoranthene	5
Benzoic acid	6
Benzyl alcohol	1
BHC, alpha-	2
BHC, beta-	6
BHC, delta-	2
Biphenyl	1
Butyl benzyl phthalate	9
Chloroaniline, p-	5
Chlorobenzene	2
Chlorobenzilate	7
Chloroform	1
Chloronaphthalene, 2-	1
Chlorpyrifos	3
Chrysene	5
Cobalt	9
Cresol, o-	6
Cymene, p-	7
DDE	1
DDT	2
Di-n-butyl phthalate	5
Di-n-octyl phthalate	1
Diazinon	2
Dibenzofuran	1
Dichlorobenzene, 1,4-	2
Dichloroethene, trans-1,2-	1
Dieldrin	5
Diepoxybutane, 1,2:3,4-	2

EXHIBIT 3-5
Pollutants With A Frequency of Detection of Less Than Ten Percent in the National Sewage Sludge Survey (cont'd.)

Pollutant	Frequency of Detection (%)
Dimethoate	1
Dimethyl phthalate	1
Dioxane, 1,4-	2
Docosane, n-	8
Endosulfan-I	1
Endrin	6
Ethylbenzene	4
Fluoranthene	6
Heptachlor epoxide	4
Hexanone, 2-	5
Isobutyl alcohol	3
Methylnaphthalene, 2-	1
N-nitrosodiphenylamine	1
Naled (Dibrom)	2
Naphthalene	2
Nitrofen (TOK)	3
Octadecane, n-	8
Pentanone, 4-methyl-2-	2
Phenanthrene	2
Phosphamidon	1
Phosphoric acid, tri-o-tolyl ester	2
Picoline, 2-	1
Pyrene	5
Santox (EPN)	2
Styrene	3
Terpineol, alpha-	2
Tetrachloroethene	3
Tetrachloromethane	4
Tetraethylpyrophosphate	2
Trichloroethene	1
Trichlorofluoromethane	5
Trifluralin (Treflan)	3
Xylene, m-	4
Xylene, o- and p-	3

EXHIBIT 3-6
Pollutants With A Frequency of Detection of Ten Percent or Greater in the National Sewage Sludge Survey

Pollutant	Frequency of Detection (%)
Acetic acid (2,4-dichlorophenoxy)	16
Aluminum	100
Antimony	38
Barium	100
Beryllium	22
Bis(2-ethylhexyl)phthalate	62
Boron	48
Butanone, 2-	34
Calcium	100
Carbon disulfide	10
Cresol, p-	43
Cyanides (Soluble salts and complexes)	37
Decane, n-	10
Dioxins	100
Dodecane, n-	14
Eicosane, n-	13
Endosulfan-II	12
Fluoride	63
Hexacosane, n-	15
Hexadecane, n-	12
Hexanoic acid	40
Iron	100
Magnesium	100
Manganese	100
Methylene chloride	42
Nitrate	95
Nitrite	83
Octacosane, n-	13
PCBs	19
Pentachloronitrobenzene	10
Phenol	34
Propanone, 2-	58
Propionic acid, 2-(2,4,5-trichlorophenoxy)	15
Silver	84
Sodium	100
Tetracosane, n-	15
Tetradecane, n-	14

EXHIBIT 3-6
Pollutants With A Frequency of Detection of Ten Percent or Greater in the National Sewage Sludge Survey (cont'd.)

Pollutant	Frequency of Detection (%)
Thallium	15
Tin	84
Titanium	98
Toluene	61
Triacontane, n-	14
Trichlorophenoxyacetic acid, 2,4,5-	29
Vanadium	62
Yttrium	61

EXHIBIT 3-7
Available Human Toxicity Data[1]

CAS #	Pollutant	Percent Detect	Risk Reference Doses[2] (mg/kg-d)			Cancer Potency Slopes (Risk/mg/kg-d)		Weight of Evidence	
			Inhalation	Oral	Mut.[3]	Inhalation	Oral	Inhalation	Oral
7429905	Aluminum	100							
7440393	Barium	100	$1.0x10^{-4}$[5]	$7.0x10^{-2}$	N				
828002	Dioxins and Dibenzofurans[4]	100					$1.6x10^{5}$[6]		
7439965	Manganese	100	$1.1x10^{-4}$	$1.0x10^{-1}$	N			D	
7440326	Titanium	98		$4.0x10^{-6}$	Y				
14797558	Nitrate	95		1.6					
7440315	Tin	84		$6.0x10^{-1}$[5]	N				
7440224	Silver	84		$5.0x10^{-3}$	N				
14797650	Nitrite	83		$1.0x10^{-1}$	Y				
7782414	Fluoride	63		$6.0x10^{-2}$	N				
117817	Bis(2-ethylhexyl)phthalate	62		$2.0x10^{-2}$			$1.4x10^{-2}$		B2
7440622	Vanadium	62		$7.0x10^{-3}$[5]	N				
108883	Toluene	61	$1.1x10^{-1}$	$2.0x10^{-1}$	Y				
67641	Propanone, 2-	58		$1.0x10^{-1}$	Y				
7440428	Boron	48	$5.7x10^{-3}$[5]	$9.0x10^{-2}$	N				
106445	Cresol, p-	43		$5.0x10^{-2}$[5]	Y			C	C
75092	Methylene chloride	42	$9.0x10^{-1}$[5]	$6.0x10^{-2}$		$1.5x10^{-3}$	$7.5x10^{-3}$[5]	B2	B2[5]
7440360	Antimony	38		$4.0x10^{-4}$	N				

EXHIBIT 3-7: Available Human Toxicity Data (cont'd.)[1]

CAS #	Pollutant	Percent Detect	Risk Reference Doses[2] (mg/kg-d)			Cancer Potency Slopes (Risk/mg/kg-d)		Weight of Evidence	
			Inhalation	Oral	Mut.[3]	Inhalation	Oral	Inhalation	Oral
57125	Cyanides (soluble salts and complexes)	37		$2.0x10^{-2}$	N				
108952	Phenol	34		$6.0x10^{-1}$	Y				
78933	Butanone, 2-	34	$8.6x10^{-2(5)}$	$5.0x10^{-2(5)}$	Y				
93765	Trichlorophenoxyacetic acid, 2,4,5-	29		$1.0x10^{-2}$	Y				
7440417	Beryllium	22		$5.0x10^{-3}$		8.4	4.3		B2
1336363	PCBs [7]	19					7.7		
94757	Acetic acid (2,4-dichlorophenoxy)	16		$1.0x10^{-2}$			$1.9x10^{-2}$		C
7440280	Thallium	15		$7.0x10^{-5(5)}$	N				
93721	Propionic acid, 2-(2,4,5-trichlorophenoxy)	15		$8.0x10^{-3}$					
33213659	Endosulfan-II	12		$5.0x10^{-5}$	N				
75150	Carbon disulfide	10	$2.9x10^{-3(5)}$	$1.0x10^{-1}$	Y				
124185	Decane, n-	10	$9.0x10^{-2(8)}$						
82688	Pentachloronitrobenzene	10		$3.0x10^{-3}$			$2.6x10^{-1(5)}$		C[5]

[1] Toxicity data from IRIS (1992) unless otherwise noted.
[2] Inhalation Risk Reference Concentrations converted to units of (mg/kg-d) by assuming a person weighs 70 kg and breathes 20 m^3 air per day.
[3] Mutagenic activity as reported in RTECS (April 1993); Y = yes, N = no.
[4] 25 chlorinated dioxins and furans have been combined by using toxicity equivalency factors. This composite was considered a detect if at least one congener was measured above the minimum level.
[5] HEAST (data from 1992 and 1991 Tables were used, with 1992 data used preferentially over 1991 data).
[6] U.S. EPA, 1991a.
[7] A composite PCB concentration was estimated by combining the concentrations of seven Aroclor mixtures measured in the NSSS.
[8] Office of Pesticide Programs RfD Tracking Report (27 January 1992).

EXHIBIT 3-8
Available Ecological Toxicity Data[1]

Pollutant	Bioaccumulation Factor (BAF)	Bioconcentration Factor (BCF)	Aquatic Life µg/L(source)	Birds mg/L(source)	Mammals mg/L(source)	Amphibians mg/L
Acetic acid (2,4-dichlorophenoxy)		52[2]				8.05[3]
Aluminum			0.087[4]	0.54(ch)[4]		0.627[3]
Antimony	0[5]				2.2(ch)[4]	0.3[3]
Barium			26,000,000[2]		32.1(ch)[4]	
Beryllium	19[5]		5.3(loel)[6]		0.274(ch)[4]	3.15[3]
Bis(2-ethylhexyl)phthalate		72[2]				
Boron					8.62(ac)[4]	
Butanone, 2-						
Carbon disulfide			21,000,000[2]			
Cresol, p-		19[2]	2,570,000[2]	96(ac/kg)[2]		
Cyanides (soluble salts and complexes)			5.2(cfwqc)[6]		68(ch)[4]	
Decane, n-						
Dioxins and dibenzofurans[7]			0.00001(loel)[6]			
Endosulfan-II			0.056(cfwqc)[6]			0.001[3]
Fluoride					3.45(ac)[4]	
Manganese			31,000,000[2]			1.42[3]
Methylene chloride	2.5[5]		1.1x10[8(2)]		4.75(ac)[4]	48[3]
Nitrate						
Nitrite						1.09[3]
PCBs[8]	1,776,860[5]		0.014(cfwqc)[6]	2000(ac/kg)[2]		

EXHIBIT 3-8
Available Ecological Toxicity Data (cont'd.)[1]

Pollutant	Bioaccumulation Factor (BAF)	Bioconcentration Factor (BCF)	Aquatic Life µg/L(source)	Birds mg/L(source)	Mammals mg/L(source)	Amphibians mg/L
Pentachloronitrobenzene		1043[9]	22(ch)[2]	2000(ac/kg)[2]	0.00975(ch)[4]	
Phenol	1728[5]		2560(loel)[6]	113(ac/kg)[2]		0.04[3]
Propanone, 2-		1[2]				19,200[3]
Propionic acid, 2-(2,4,5-trichlorophenoxy)						
Silver	0[5]		0.12(cfwqc)[6]			0.004[3]
Thallium	67[5]		40(loel)[6]		0.0589(ch)[4]	0.11[3]
Tin						0.09[3]
Titanium						
Toluene	1547[5]		5000(cmwqc)[6]			0.39[3]
Trichlorophenoxyacetic acid, 2,4,5-						
Vanadium			5x10[8](ch)[2]			

[1] All data provided by EPA's Health and Ecological Criteria Division.
[2] Implementation of a Chemical Ranking System, Appendix B (U.S. EPA, 1990b). For BCF values, it is not entirely clear whether these values represent BAFs or BCFs. (ac) refers to acute values, (ch) refers to chronic values. /kg means that the units for the value are (mg/kg body weight).
[3] Amphibian toxicological data from two databases: Devillers and Exbrayat, 1992 and Harfenist et al., 1989.
[4] Screening Study for Wildlife Criteria Development (U.S. EPA, 1989f). (ac) refers to acute values, (ch) refers to chronic values.
[5] Document in support of the Great Lakes Water Quality Initiative. (U.S. EPA, 1993b). The BAFs presented were derived as human health criteria, and were calculated at 5% lipid.
[6] For references ending in "wqc," the source is Federal Register Part II, 40 CFR Part 131, December 22, 1992. A "cf" prefix indicates that the value is a chronic freshwater value. A "cm" prefix indicates that the value is a chronic marine value. A "loel" indicates the value is a Lowest Observed Effect Level, but not a criterion.
[7] Dioxins and furans have been combined using toxicity equivalency factors.
[8] A composite PCB concentration was estimated by combining the concentrations of seven Aroclor mixtures measured in the NSSS.
[9] Oneliner, in PIRANHA (U.S. EPA, 1991b).

EXHIBIT 3-9
Pollutants With No Human Health or Ecological Toxicity Data Available

Pollutant
Calcium
Decane, n-[1]
Dodecane, n-
Eicosane, n-
Hexacosane, n-
Hexadecane, n-
Hexanoic acid
Iron
Magnesium
Octacosane, n-
Sodium
Tetracosane, n-
Tetradecane, n-
Triacontane, n-
Yttrium

[1] Available data from the Office of Pesticide Programs RfD Tracking Report (27 January 1992) were not considered to be appropriate for the assessment of the Round Two list of pollutants.

EXHIBIT 3-10
Round One Pollutants Included as Potential Candidates For Round Two

Pollutant	Rationale
PCBs	Aroclor mixtures of PCBs were evaluated in Round One. Coplanar PCBs are being considered for Round Two because of their chemical and toxicological similarities to dioxins.
Beryllium	Not evaluated for all exposure pathways and use/disposal practices in Round One.
Phenol	Not evaluated for all exposure pathways and use/disposal practices in Round One.
Methylene chloride	Not evaluated for all exposure pathways and use/disposal practices in Round One.
Bis(2-ethylhexyl)phthalate	Not evaluated for all exposure pathways and use/disposal practices in Round One.
Cyanide	Not evaluated for all exposure pathways and use/disposal practices in Round One.
Fluoride	Not evaluated for all exposure pathways and use/disposal practices in Round One.

EXHIBIT 3-11
31 Pollutant Candidates For Round Two Regulation[1]

Pollutant
Acetic acid (2,4-dichlorophenoxy)
Aluminum[2]
Antimony
Asbestos[3]
Barium
Beryllium
Bis(2-ethylhexyl)phthalate
Boron
Butanone, 2-
Carbon disulfide
Cresol, p-
Cyanides (soluble salts and complexes)
Dioxins and dibenzofurans
Endosulfan-II
Fluoride
Manganese
Methylene chloride
Nitrate
Nitrite
Pentachloronitrobenzene
Phenol
Polychlorinated biphenyls -- coplanar
Propanone, 2-
Propionic acid, 2-(2,4,5-trichlorophenoxy)
Silver
Thallium
Tin
Titanium
Toluene
Trichlorophenoxyacetic acid, 2,4,5-
Vanadium

[1] Pollutants detected at a frequency of at least ten percent with human health and/or ecological toxicity data available.
[2] Aluminum does not have human health or ecological toxicity data available, but is included due to its potential for phytotoxicity.
[3] Asbestos was not tested in the NSSS, but is toxic, persistent, and may be in sewage sludge.

EXHIBIT 3-12
Rationale for the Number of Pollutants Selected as Candidates for the Round Two Sewage Sludge Regulation

Rationale	Number of Pollutants
Pollutants for which NSSS sewage sludge samples were analyzed (Exhibit 3-1).	411
Pollutants regulated in the Round One sewage sludge regulation (Exhibit 3-2).	10
Pollutants combined into classes of congeners or removed from further consideration (Exhibit 3-3).	33
Pollutants with a frequency of detection of zero percent (Exhibit 3-4).	254
Pollutants deleted because they did not have a non-zero frequency of detection of ten percent or greater (Exhibit 3-5).	69
Pollutants detected at a frequency of ten percent or greater (Exhibit 3-6).	45
Pollutants detected at a frequency of ten percent or greater, but with insufficient human health or ecological toxicity data available (Exhibit 3-9).	15
Pollutants detected at a frequency of ten percent or greater, with human health and/or ecological toxicity data available.	30
Asbestos added.	1
Number of candidate pollutants for Round Two sewage sludge regulation (Exhibit 3-11).	31

4. FINAL LIST OF ROUND TWO POLLUTANTS

This section presents the methods and the results of the Comprehensive Hazard Identification study, in which a quantitative risk assessment for the Highly Exposed Individual, including dose-response evaluation, exposure assessment, and risk characterization, is performed. The 31 pollutant candidates for the Round Two sewage sludge regulation, identified in the Preliminary Hazard Identification, are evaluated in this Comprehensive Hazard Identification. Note that in the Technical Support Documents for Round One, the calculations begin with an acceptable level of risk, and work backwards to determine what pollutant concentrations in the sewage sludge are acceptable for that use/disposal practice, thereby calculating pollutant limits. In this study, levels of risk that might be associated with a given pollutant under a given use/disposal practice are estimated, based on sewage sludge pollutant concentrations from the NSSS. Those pollutants/practices with high risk estimates are candidates for the final list of Round Two pollutants. The results of this Comprehensive Hazard Identification indicate that only a subset of the 31 pollutants should be considered for regulation in Round Two.

4.1 GENERAL APPROACH FOR THE COMPREHENSIVE HAZARD IDENTIFICATION

The purpose of the Comprehensive Hazard Identification study is to identify pollutants that warrant further consideration for the final list of Round Two pollutants. Analyses are performed to identify pollutants that may potentially cause human health or ecological risk for a Highly Exposed Individual (HEI). Consistent with the EPA Guidelines for Exposure Assessment (57 FR 22888, May 29, 1992), the risk to the HEI is estimated using a combination of high-end and average assumptions designed to give a plausible estimate of the individual risk at the upper end of the risk distribution (e.g., above the 90th percentile of the actual distribution).

In general for this study, high-end assumptions are used to characterize sewage sludge concentrations and certain exposure parameters, while average values are typically used to characterize use/disposal practices and soil and meteorological characteristics. Specifically, sewage sludge concentrations are based on the 95th percentile concentrations of pollutants obtained in the NSSS, with non-detects set equal to the minimum level (e.g., the minimum concentration of pollutant that could be measured) (see Exhibit 4-1). For each sewage sludge use/disposal practice, the HEI is defined as "an individual who remains for an extended period of time at or adjacent to the site where the maximum exposure occurs" (U.S. EPA, 1992a). Numerous exposure assumptions are specific to each exposure pathway and are given in the subsequent sections.[1]

This chapter describes the methods used for each sewage sludge use or disposal practice and exposure pathway (Section 4.2 (land application), 4.3 (surface disposal), and 4.4 (incineration)). Next, methods for estimating human health and ecological risk are presented

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[1] More detailed explanations for the derivations of the specific values used in this study are provided in the Technical Support Documents for Round One (e.g., U.S. EPA, 1992a).

(Section 4.5). For those pollutant-exposure pathway combinations for which an exposure can be estimated, the calculated risk associated with that exposure is then presented. Based on these risk estimates, the pollutants that warrant further consideration for inclusion on the final list of Round Two pollutants are presented in Chapter 5.

EXHIBIT 4-1
95th Percentile Concentrations for Round Two Candidate Pollutants

Pollutant	95th Percentile Sewage Sludge Concentrations[1] (mg/kg dry weight)
Acetic acid (2,4-dichlorophenoxy)	0.030
Aluminum	36,400
Antimony	24
Asbestos[2]	Not Available
Barium	1,730
Beryllium	8
Bis(2-ethylhexyl)phthalate	191
Boron	182
Butanone, 2-	69.3
Carbon disulfide	3.13
Cresol, p-	306
Cyanides (soluble salts and complexes)	130
Dioxins and dibenzofurans	3.11×10^{-4}
Endosulfan-II	0.0667
Fluoride	411
Manganese	1,620
Methylene chloride	31.3
Nitrate	5,020
Nitrite	462
Pentachloronitrobenzene	0.0793
Phenol	57.5
Polychlorinated biphenyls -- coplanar[3]	5.4
Propanone, 2-	116
Propionic acid, 2-(2,4,5-trichlorophenoxy)	0.040
Silver	128

Pollutant	95th Percentile Sewage Sludge Concentrations[1] (mg/kg dry weight)
Thallium	10.6
Tin	138
Titanium	363
Toluene	238
Trichlorophenoxyacetic acid, 2,4,5-	0.0505
Vanadium	64.1

[1] Non-detects set equal to the Minimum Level. Concentrations from the NSSS.
[2] Asbestos was not tested in the NSSS, but is toxic, persistent, and may be in sewage sludge.
[3] Sewage sludge samples were not analyzed for coplanar PCBs in the NSSS. A composite PCB concentration was estimated by combining the concentrations of the seven Aroclor mixtures measured in the NSSS.

4.2 LAND APPLICATION PATHWAY EXPOSURE METHODOLOGIES

Methods were developed in Round One to evaluate risk from sewage sludge that is land-applied to agricultural and non-agricultural sites. These methods include evaluating both human health and ecological risks associated with exposure to sewage sludge through 14 different pathways. In Round Two, an additional exposure pathway, breastfeeding, is also considered. These 15 exposure pathways and the corresponding HEIs are summarized in Exhibit 4-2.

To estimate exposure to pollutants in sewage sludge that is land-applied, several non-pollutant-specific types of data are required, including information on application practices and soil characteristics. The average values used for these parameters are shown in Exhibit 4-3. The way in which these data were combined with pollutant-specific data to estimate exposure is described in Sections 4.2.1 through 4.2.15.

For both agricultural and non-agricultural land application sites, it is necessary to estimate the number of years that a site is used and the rate at which sewage sludge is applied to that site. An estimate of the depth to which sewage sludge is incorporated at the site also must be made. The values used for these parameters in this analysis are presented in Exhibit 4-3. Note that for the forest and public contact sites, depth of incorporation is assumed to be zero, and therefore the "soil concentration" of a pollutant is the same as the sewage sludge concentration.

For the soil at agricultural land and reclaimed sites, an average bulk density of 1600 kg/m^3 is assumed (U.S. EPA, 1992a). For the bulk density of sewage sludge-amended soil, an average value of 1400 kg/m^3 is assumed (U.S. EPA, 1992a). For natural "background" soil concentrations of inorganics, 90th percentile concentrations of inorganics in soil throughout the United States were used (U.S. Geological Survey, 1992). To ascertain

whether sewage sludge pollutant concentrations were greater than natural soil concentrations, the 90th percentile soil concentrations were compared to the 95th percentile pollutant concentrations in sewage sludge. As shown in Exhibit 4-4, for three inorganics the 95th percentile sewage sludge concentrations are less than (aluminum and titanium) or approximately equal to (vanadium) the 90th percentile natural soil concentrations. This suggests that natural concentrations of these inorganics contribute significantly to the overall exposure of an HEI. For antimony and tin, sewage sludge concentrations are much greater than natural soil concentrations, indicating that sewage sludge loadings are the major determinant of an HEI's exposure to those inorganic pollutants. In this analysis, background concentrations of nitrate and nitrite are not applicable due to the cycling of nitrogen in soil systems. The background concentration of asbestos also is not applicable because asbestos would not be taken up by crops. Cyanide was the only applicable inorganic pollutant for which background soil concentration data were not available. For the organic pollutants, background concentration data are not required because it is assumed that there are not natural concentrations of organic pollutants in soil.

EXHIBIT 4-2
Definitions of Exposure Pathways and Highly Exposed Individuals (HEIs) for Land Application

Pathway	Agricultural HEI	Non-Agricultural HEI		
		Forest	Reclamation	Public Contact
1. Sewage Sludge → Soil → Plant → Human	Human ingesting crops grown on sewage sludge-amended agricultural land.	A person who regularly harvests wild plants from forest land amended with sewage sludge.	A person who regularly harvests wild plants from reclamation sites amended with sewage sludge.	A person who regularly harvests wild plants from public contact sites amended with sewage sludge.
2. Sewage Sludge → Soil → Plant → Human	Residential home gardener.	NA	NA	NA
3. Sewage Sludge → Human	Child between the ages of 1 and 6 who ingests sewage sludge from storage piles or the soil surface of agricultural land daily for 5 years.	Child between the ages of 4 and 6 who ingests sewage sludge from the soil surface of forest land daily for 2 years.	Child between the ages of 4 and 6 who ingests sewage sludge from the soil surface of a reclamation site daily for 2 years.	Child between the ages of 1 and 6 who ingests sewage sludge from the soil surface of a public contact site daily for 5 years.
4. Sewage Sludge → Soil → Plant → Animal → Human	Farm households producing a major portion of the animal products they consume. It is assumed that the animals eat plants grown in soil amended with sewage sludge.	A hunter of deer and elk that feed in forests. The hunter preserves meat for consumption throughout the year.	A hunter of deer and elk that feed on reclamation sites. The hunter preserves meat for consumption throughout the year.	NA
5. Sewage Sludge → Animal → Human	Farm households producing a major portion of the animal products they consume. It is assumed that livestock ingest sewage sludge while grazing.	Consumer of livestock that ingest sewage sludge in sewage sludge-amended forests.	Consumer of livestock that ingest sewage sludge in sewage sludge-amended reclamation sites.	NA

EXHIBIT 4-2
Definitions of Exposure Pathways and Highly Exposed Individuals (HEIs) for Land Application (cont'd.)

Pathway	Agricultural HEI	Non-Agricultural HEI		
		Forest	Reclamation	Public Contact
6. Sewage Sludge → Soil → Plant → Animal	Livestock ingesting crops grown on sewage sludge-amended agricultural land.	A small herbivore that lives its entire life in a sewage sludge-amended forest feeding on seeds and small plants; or, livestock that graze on grasses growing in a sewage-sludge amended forest.	Livestock that graze on grasses growing on sewage sludge-amended reclamation sites.	A small herbivore that lives its entire life in a sewage sludge-amended area feeding on seeds and small plants close to the sewage sludge-soil layer in a public contact site.
7. Sewage Sludge → Animal	Grazing livestock that ingest sewage sludge applied to agricultural land.	Grazing livestock that ingest sewage sludge applied to forest sites.	Grazing livestock that ingest sewage sludge applied to reclamation sites.	NA
8. Sewage Sludge → Soil → Plant	Plants grown on sewage sludge-amended agricultural land.	Plants grown in sewage sludge-amended soil in forests.	NA	Plants grown in sewage sludge-amended soil on public contact sites.
9. Sewage Sludge → Soil → Soil Organism	Earthworms living in sewage sludge-amended agricultural soil.	Earthworms living in sewage sludge-amended soil in forests.	Earthworms living in sewage sludge-amended soil on reclamation sites.	Earthworms living in sewage sludge-amended soil on public contact sites.
10. Sewage Sludge → Soil → Soil Organism → Soil Organism Predator	Insectivorous small mammals eating soil organisms living in sewage sludge-amended agricultural soil.	Insectivorous small mammals eating soil organisms that live in sewage sludge-amended soil in forests.	Insectivorous small mammals eating soil organisms that live in sewage sludge-amended soil on reclamation sites.	Insectivorous small mammals eating soil organisms that live in sewage sludge-amended soil on public contact sites.
11. Sewage Sludge → Soil → Airborne Dust → Human	Tractor operator exposed to dust while plowing large areas of sewage sludge-amended soil.	NA	NA	NA

EXHIBIT 4-2
Definitions of Exposure Pathways and Highly Exposed Individuals (HEIs) for Land Application (cont'd.)

Pathway	Agricultural HEI	Non-Agricultural HEI		
		Forest	Reclamation	Public Contact
12. Sewage Sludge → Soil → Surface Water → Human	Human who consumes both fish and water from a stream that receives eroded sewage sludge-amended soil from agricultural land.	Human who consumes both fish and water from a stream that receives eroded sewage sludge-amended soil from a forest.	Human who consumes both fish and water from a stream that receives eroded sewage sludge-amended soil from a reclamation site.	Human who consumes both fish and water from a stream that receives eroded sewage sludge-amended soil from a public contact site.
13. Sewage Sludge → Soil → Air → Human	Human breathing volatile pollutants from sewage sludge-amended agricultural land.	Human breathing volatile pollutants from sewage sludge-amended soil in a forest.	Human breathing volatile pollutants from sewage sludge-amended soil on a reclamation site.	Human breathing volatile pollutants from sewage sludge-amended soil on a public contact site.
14. Sewage Sludge → Soil → Ground Water → Human	Human drinking water from wells that contain pollutants leached from sewage sludge-amended agricultural land to ground water.	Human drinking water from wells that contain pollutants leached from sewage sludge-amended soil in a forest to ground water.	Human drinking water from wells that contain pollutants leached from sewage sludge-amended soil on a reclamation site to ground water.	Human drinking water from wells that contain pollutants leached from sewage sludge-amended soil on a public contact site to ground water.
15. Exposure to mother from pathways 1, 2, 4, 5, 12, 13, and 14 → Nursing Infant	Nursing infant of exposed mother.	Nursing infant of exposed mother.	Nursing infant of exposed mother.	Nursing infant of exposed mother.

NA - Not Applicable

EXHIBIT 4-3
Average Values for Sewage Sludge Land Application Parameters

Parameter	Definition	Value	Notes
N, agricultural land	Number of years of application to agricultural land	20	assumed applied once every year for 20 year site life (U.S. EPA, 1993a)
N, forest	Number of years of application to forest land	7	assumed applied once every three years for 20 year site life (U.S. EPA, 1984)
N, reclamation sites	Number of years of application to reclamation sites	1	assumed one-time application
N, public contact	Number of years of application to public contact sites	10	assumed applied once every two years for 20 year site life (U.S. EPA, 1984)
N_{site}, agricultural land, forest, public contact	Site life, i.e., period of time during which sewage sludge may be applied	20yr	
N_{site}, reclamation site	Site life, i.e., period of time during which sewage sludge may be applied	1yr	
d, agricultural land	Depth of incorporation on agricultural land	15 cm	U.S. EPA, 1992a
d, forest	Depth of incorporation on forest land	0 cm	no incorporation assumed
d, reclamation sites	Depth of incorporation on reclamation sites	10 cm	U.S. EPA, 1984
d, public contact	Depth of incorporation on public contact sites	0 cm	no incorporation assumed
AR, agricultural land	Annual whole sludge application rate for agricultural land	7 dry Mg/ha-yr[1]	U.S. EPA, 1992a
AR, forest	Annual whole sludge application rate for forest land	26 dry Mg/ha-yr[1]	U.S. EPA, 1992a
AR, reclamation sites	Annual whole sludge application rate for reclamation sites	74 dry Mg/ha-yr[1]	U.S. EPA, 1992a
AR, public contact	Annual whole sludge application rate for public contact sites	18 dry Mg/ha-yr[1]	U.S. EPA, 1992a
BD_{soil}	Bulk density of soil at agricultural land and reclamation sites	1600 kg/m^3	U.S. EPA, 1992a
BD_{mix}	Bulk density of sewage sludge-amended soil	1400 kg/m^3	U.S. EPA, 1992a

[1] Note that 1 Mg = 1 megagram = 1 metric ton = 1000 kg.

EXHIBIT 4-4
Background Concentrations of Pollutants in Soil

Pollutant	90th Percentile Background Concentration in Soil[1] (mg/kg dry weight)	Ratio of Sewage Sludge (95th) to Soil Concentrations (dimensionless)
Aluminum	68,000	0.54
Antimony	0.51	47
Barium	452	3.8
Beryllium	0.65	12
Boron	27	6.7
Cyanide (soluble salts and complexes)	NA	NA
Fluoride	220	1.9
Manganese	342	4.7
Silver	0[2]	NA
Thallium	0[3]	NA
Tin	0.94	150
Titanium	3400	0.11
Vanadium	60	1.1

NA means Not Available.
[1] Concentration obtained by calculating 90th percentile value, based on geometric means and standard deviations obtained from U.S. Geological Survey, 1992.
[2] Silver was measured too infrequently for a reliable mean concentration to be calculated, as discussed in U.S. Geological Survey, 1992.
[3] Thallium was analyzed for in all samples but was never found, as discussed in U.S. Geological Survey, 1992.

4.2.1 Pathway 1 - Ingestion of Crops Grown on Sewage Sludge-Amended Soil

Pathway 1 evaluates human ingestion of plants that have taken up pollutants from sewage sludge-amended agricultural and non-agricultural lands. Non-agricultural lands include forests, reclaimed land, and public contact sites.

Mass Balance Equations

To be conservative in these analyses, the entire mass of a pollutant applied in sewage sludge over the life of a land application site is assumed to be available for plant uptake. Unlike Pathway 1 in the Technical Support Document for Land Application of Sewage Sludge, developed for the Round One regulation (U.S. EPA, 1992a), this analysis does not assume that organic pollutants either degrade or volatilize each year between sewage sludge applications. This conservative approach is used for this level of assessment because: (a) plant uptake of most pollutants is low and (b) the dissipation of many of the pollutants being considered in Round Two is slow. If this pathway yielded high risk for a particular pollutant, the assumptions would be refined.

In calculating total pollutant concentration in the soil (CT_j) for the agricultural and reclaimed land scenarios, the following equation is used for both inorganic and organic pollutants:

$$CT_j = \frac{(BS_j \cdot MS) + (N \cdot C_j \cdot AR)}{(N \cdot AR) + MS} \qquad (4\text{-}1)$$

where:

CT_j = concentration of pollutant j in sewage sludge-amended soil (mg pollutant/kg sewage sludge-amended soil),
BS_j = background concentration (dry weight) of pollutant j in soil (mg pollutant/kg soil),
MS = mass of soil in mixing zone of one hectare of land (Mg soil/ha land),
N = total number of years sewage sludge is applied to land (yr),
C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge), and
AR = annual whole sludge application rate of sewage sludge to land (dry Mg sewage sludge/ha-yr).

The mass of soil in the mixing zone of one hectare of land is calculated as:

$$MS = BD_{soil} \cdot d \cdot 10^{-1} \qquad (4\text{-}2)$$

where:

BD_{soil} = bulk density of soil (kg soil/m^3 soil),
d = depth of incorporation (cm), and
10^{-1} = constant to convert (kg · cm/m^3) to (Mg/ha).

For forests and public contact sites, it is assumed that there is no incorporation of land-applied sewage sludge. Therefore, the concentration of each pollutant in the "soil" is set equal to its concentration in the sewage sludge.

The concentrations of pollutants in crops grown on sewage sludge-amended soil are calculated as:

$$CD_{ij} = CT_j \cdot UC_{ij} \quad (4\text{-}3)$$

where:

CD_{ij} = tissue concentration (dry weight) of pollutant j in crop i (mg pollutant/kg crop tissue), and

UC_{ij} = rate of uptake of pollutant j into tissue of crop i (μg pollutant/g dry weight crop tissue per μg pollutant/g sewage sludge-amended soil).

Exposure Equation

Once the various concentrations of a pollutant in crop tissues are estimated, they are combined with data on the fraction of crops grown on sewage sludge-amended soil and the daily dietary consumption of crops to estimate human exposure:

$$EXP_j = \frac{10^{-3}}{BW} \sum_i CD_{ij}\, FC_i\, DC_i = \frac{10^{-3}\, CT_j}{BW} \sum_i UC_{ij}\, FC_i\, DC_i \quad (4\text{-}4)$$

where:

EXP_j = exposure to pollutant j from crops produced on sewage sludge-amended soil (mg pollutant/kg body weight-day),

10^{-3} = constant to convert units from (g) to (kg),

BW = body weight (kg), assumed to be 70 kg,

FC_i = fraction of dietary consumption of crop i grown in sewage sludge-amended soil (dimensionless), and

DC_i = daily dietary consumption of crop i (g crop tissue/day).

Data Inputs

There are three types of data inputs specific to this pathway: daily dietary consumption of various crops, fraction of consumption derived from sewage sludge-amended soil, and pollutant-specific plant uptake rates. Values for the daily dietary consumption of crops and the fraction of consumption derived from sewage sludge-amended soil are presented in Exhibit 4-5.

EXHIBIT 4-5
Dietary Assumptions for Pathway 1

Crop	Daily Dietary Consumption of Crop (g dry weight/day)[1]	Fraction of Consumption Derived from Sewage Sludge-Amended Soil[2]
Garden Fruits	4.15	0.025
Grains and Cereals	90.7	0.025
Leafy Vegetables	1.97	0.025
Legumes	8.75	0.025
Peanuts	2.25	0.025
Potatoes	15.6	0.025
Root Vegetables	1.6	0.025
Berries	2.60	0.25[4]
Mushrooms	0.6[3]	0.25[4]

[1] Values represent the estimated lifetime average daily food intakes for the crops, in dry weight, unless otherwise noted. U.S. EPA, 1992a.
[2] Fractions represent reasonable estimates, unless otherwise noted. U.S. EPA, 1992a.
[3] Due to the potential to gather large quantities of mushrooms from forest lands, a daily dietary consumption value for a Highly Exposed Individual, not an average individual, was used. U.S. EPA, 1992a.
[4] Fractions represent reasonable worst-case assumptions. U.S. EPA, 1992a.

Plant uptake slopes are needed for the seven agricultural crop categories that represent major human dietary intakes: garden fruits (e.g., tomatoes); grains and cereals (e.g., barley, wheat); leafy vegetables (e.g., swiss chard, cabbage, lettuce); dry and fresh legumes (e.g., beans, peas); peanuts; potatoes; and, root vegetables (e.g., carrots, beets) (U.S. EPA, 1992a). In addition, plant uptake rates for berries and mushrooms are needed for non-agricultural lands. Data on plant uptake slopes, however, did not exist for all pollutant candidates and for all crops. When no data were available for a particular crop, the following extrapolations were made between crops for a given pollutant:

- grain and cereal uptake slopes and forage/pasture uptake slopes (used in animal exposure pathways) were considered interchangeable;
- potato uptake slopes and root vegetable uptake slopes were considered interchangeable;

- peanut uptake slopes and legume uptake slopes were considered interchangeable; and

- any vegetative or leafy growth uptake slopes identified in a study (e.g., soybean leaves) were used for leafy vegetable uptake slopes if no leafy vegetable studies could be identified.

When multiple data points were available for a particular pollutant and crop from a variety of studies, the average of the data from the most appropriate studies was used. The appropriateness of a given study was determined from the study hierarchy established in Round One: data from sludge-amended field studies were preferred over data from sludge-amended pot studies, which in turn were preferred over data from metal-salt-amended field or pot studies. If uptake slope data existed for a particular pollutant in a particular crop category from more than one study of the same hierarchical level, they were averaged.

For the agricultural pathway, if uptake slope data were not available or could not be estimated using the above extrapolations for all seven crops for a particular pollutant, then exposure to that pollutant was not estimated. Available plant uptake slopes for agricultural land crops are presented in Exhibit 4-6. Note that only 14 pollutant candidates had available data on plant uptake slopes for at least one crop; only three pollutant candidates had uptake slope data available for all seven crops. Therefore only three pollutants could be evaluated for this exposure pathway.

The non-agricultural Pathway 1 models human consumption of plants grown in forests, on reclaimed lands, or on public contact sites that have been amended with sewage sludge. Humans are assumed to be potentially exposed to wild berries or mushrooms. For this pathway, garden fruits were used as a surrogate for berries if actual data did not exist for organic and inorganic pollutant uptake. For uptake of organic and inorganic pollutants into mushrooms, data were not available. In the Technical Support Document for Land Application of Sewage Sludge (U.S. EPA, 1992a), it was noted that mushrooms have potential to bioaccumulate both mercury and cadmium. As shown in Exhibit 4-7, information is not available on whether mushrooms can bioaccumulate any of the Round Two pollutant candidates. Note that only four pollutant candidates, those with available wild berry uptake data, are included in the exhibit. No pollutants could be evaluated for the non-agricultural pathway due to the lack of uptake data for mushrooms.

EXHIBIT 4-6
Available Plant Uptake Slopes for Agricultural Pathway 1

Pollutant	Plant Uptake Slopes (μg/g crop tissue per μg/g soil)						
	Garden Fruits	Grains and Cereals	Leafy Vegetables	Fresh and Dry Legumes	Peanuts	Potatoes	Root Vegetables
Aluminum		3.13[1]	0.0026[2]				
Antimony			0.103[3]				
Barium			0.0695[4]				
Beryllium		0.31[5]	12.2[6]			0.49[7]	0.49[7]
Boron	2.29[8]	3.9[9]	3.115[10]	109.075[11]	109.075[11]		
Dioxins and Dibenzofurans	0.000036[12]	0.0043[13,14]	0.009[13]	0.0043[13]	0.0043[13]	0.00145[12]	0.00145[12]
Fluoride	0.35[15]	0.4367[16]	1.875[17]	2.24[18]	2.24[18]	0.2475[19]	0.2475[19]
Manganese		9.98[20]	1.5565[21]			8.4[22]	8.4[22]
Polychlorinated biphenyls (coplanar)		2[23]					
Silver		0.021[24]					
Thallium			22.47[25]	18.73[26]	18.73[26]		
Tin			0.3352[27]				
Titanium	0.136[28]	0.0325[29]	0.028[30]	0.7288[31]	1.22[31]	0.13[32]	0.13[32]
Vanadium			0.0062[33]				

Key to Study Type:
A: Based on sewage sludge field study.
B: Based on non-field sewage sludge study.
C: Based on non-sewage sludge study.

Footnotes:
[1] Muchovej et al., 1986 (ryegrass: 3.13): C.
[2] Chaney et al., 1978 (swiss chard: 0.00301): A; Babish et al., 1979 (cabbage: 0.0021): A.
[3] Chaney et al., 1978 (swiss chard: 0.194): A; Babish et al., 1979 (cabbage: 0.012): A.
[4] Chaney et al., 1978 (swiss chard: 0.097): A; Babish, et al., 1979 (cabbage: 0.042): A.
[5] Bohn and Seekamp, 1979 (oats: 0.31): C.
[6] Kosak-Channing, 1986 (tobacco leaves: 12.2): C.
[7] Bohn and Seekamp, 1979 (potatoes: 0.49): C.
[8] Soon and Bates, 1985 (corn kernels: 2.29): A.
[9] Soon and Bates, 1985 (bromegrass: 3.9): A.
[10] Chaney et al., 1978 (swiss chard: 3.43): A; Babish et al., 1979 (cabbage: 2.8): A.
[11] Agbenin et al., 1991 (cowpea: 134.4): C; Stark and Redente, 1990 (alfalfa: 83.75): C.
[12] U.S. EPA, 1992d. Estimated from model. Assumed dry weight basis.
[13] U.S. EPA, 1994a. Fresh weight basis.
[14] Assumed to be the same as legumes.
[15] Doss et al., 1977 (tomatoes: 0.35): C.
[16] Davis, 1980 (first cut rye: 0.18); Singh et al., 1979b (wheat: 0.8): C; Singh et al., 1979a (rice: 0.33): C.
[17] Bar-Yosef and Rosenberg, 1988 (corn shoot: 3.4, tomato shoot: 3.8): C; Doss et al., 1977 (lettuce: 0.5, petunia leaves: 1.0, potato leaves: 0.675): C.
[18] Stark and Redente, 1990 (alfalfa: 2.24): C.
[19] Doss et al., 1977 (potato: 0.2475): C.
[20] Kalbasi, 1988 (sorghum: 9.98): C.
[21] Chaney et al., 1978 (swiss chard: 3.09): A; Babish et al., 1979 (cabbage: 0.023): A.
[22] Denduluri, 1993 (okra roots: 8.4): C.
[23] Webber et al., 1983 (whole oat: 2.0): A.
[24] Romney et al., 1977 (barley: 0.021): C.
[25] Kaplan et al., 1990 (bean leaves: 22.47): C.
[26] Kaplan et al., 1990 (bean pods: 18.73): C.
[27] Chaney et al., 1978 (swiss chard: 0.6604): A; Babish et al., 1979 (cabbage: 0.01): A.
[28] Tonkonozhenko and Khlyupina, 1974 (corn kernels: 0.136): C.
[29] Tonkonozhenko and Khlyupina, 1974 (winter wheat grains: 0.035, rice grains: 0.03): C.
[30] Babish et al., 1979 (cabbage: 0.028): A.
[31] Tonkonozhenko and Khlyupina, 1974 (peas: 0.2375, beans: 1.22): C.
[32] Tonkonozhenko and Khlyupina, 1974 (sugar beets: 0.13): C.
[33] Chaney et al., 1978 (swiss chard: 0.00645): A; Babish et al., 1979 (cabbage: 0.006): A.

EXHIBIT 4-7
Available Plant Uptake Slopes for Non-Agricultural Pathway 1

Pollutant	Plant Uptake Slopes (μg/g crop tissue per μg/g soil)	
	Berries	Mushrooms
Boron	2.29[1]	
Dioxins and Dibenzofurans	0.000036[2]	
Fluoride	0.35[3]	
Titanium	0.136[4]	

Key to Study Type:
A: Studies conducted in the field with sewage sludge.
B: All other studies conducted with sewage sludge.
C: All other studies.

Footnotes:
[1] Soon and Bates, 1985 (corn kernels: 2.29): A.
[2] U.S. EPA, 1992b. Estimated from model. Assumed dry weight basis.
[3] Doss et al., 1977 (tomatoes: 0.35): C.
[4] Tonkonozhenko and Khlyupina, 1974 (corn kernels: 0.136): C.

Example Exposure Calculation for Pathway 1

The following example presents the calculations for estimating human exposure to fluoride in sewage sludge applied to agricultural land. Fluoride is one of three pollutants with plant uptake slopes for all seven crops. The example uses the equations and input data presented above for Pathway 1.

First, the mass of soil in the mixing zone, *MS*, is estimated for agricultural land by using Eq. 4-2:

$$MS = \frac{1600 kg}{m^3} \cdot 15 cm \cdot \frac{10^{-1} Mg/ha}{kg \cdot cm/m^3} = \frac{2400 Mg}{ha}$$

where:

1600 = BD_{soil} (bulk density of soil) from Exhibit 4-3,
15 = d (depth of incorporation for agricultural land) from Exhibit 4-3, and
10^{-1} = constant to convert (kg · cm/m^3) to (Mg/ha).

Then, the concentration of fluoride in agricultural soil is calculated using Eq. 4-1:

$$CT_{fluoride} = \frac{\left(\frac{220mg}{kg} \cdot \frac{2400Mg}{ha}\right) + \left(20yr \cdot \frac{411mg}{kg} \cdot \frac{7Mg}{ha\text{-}yr}\right)}{\left(20yr \cdot \frac{7Mg}{ha\text{-}yr}\right) + \frac{2400Mg}{ha}}$$

$$= 230 mg/kg$$

where:

220	=	BS_j (background concentration of fluoride in soil) from Exhibit 4-4,
2400	=	MS (mass of soil in agricultural mixing zone), estimated above,
20	=	N (total number of years sewage sludge is applied to agricultural land) from Exhibit 4-3,
411	=	C_j (fluoride concentration in sewage sludge) from Exhibit 4-1, and
7	=	AR (application rate of sewage sludge to agricultural land) from Exhibit 4-3.

Total dietary exposure is then determined using Eq. 4-4:

$$EXP_{fluoride} = \left[\frac{10^{-3}\ kg/g \cdot 230\ mg/kg}{70\ kg}\right] \cdot [(0.35g/g \cdot 0.025 \cdot 4.15g/day)$$

$$+ (0.44g/g \cdot 0.025 \cdot 90.7g/day) + (1.9g/g \cdot 0.025 \cdot 1.97g/day)$$

$$+ (2.2g/g \cdot 0.025 \cdot 8.75g/day) + (2.2g/g \cdot 0.025 \cdot 2.25g/day)$$

$$+ (0.25g/g \cdot 0.025 \cdot 15.6g/day) + (0.25g/g \cdot 0.025 \cdot 1.6g/day)]$$

$$= 0.0061 mg/kg\text{-}day$$

where:

10^{-3}	=	constant to convert units from (g) to (kg),
230	=	CT_j (concentration of fluoride in agricultural soil), estimated above,
70	=	BW (body weight), assumed to be 70 kg,
0.35 · 0.025 · 4.15	=	contribution to dietary exposure from garden fruits,
0.44 · 0.025 · 90.7	=	contribution to dietary exposure from grains and cereals,
1.9 · 0.025 · 1.97	=	contribution to dietary exposure from leafy vegetables,
2.2 · 0.025 · 8.75	=	contribution to dietary exposure from legumes,
2.2 · 0.025 · 2.25	=	contribution to dietary exposure from peanuts,
0.25 · 0.025 · 15.6	=	contribution to dietary exposure from potatoes, and
0.25 · 0.025 · 1.6	=	contribution to dietary exposure from root vegetables.

Contribution to dietary exposure is derived as the product of UC_{ij} (uptake slope of fluoride into crop) from Exhibit 4-6, FC_i (fraction of dietary consumption of crop grown in sewage sludge-amended soil) from Exhibit 4-5, and DC_i (daily dietary consumption of crop) from Exhibit 4-5.

4.2.2 Pathway 2 - Ingestion of Crops Grown in Sewage Sludge-Amended Home Gardens

Pathway 2 evaluates human ingestion of plants that have taken up pollutants from sewage sludge-amended home gardens. The mass balance and exposure equations used are identical to those for Pathway 1 (Section 4.2.1).

Data Inputs

There are three types of data inputs specific to this pathway: the daily dietary consumption of specific crops, the fraction of that daily consumption that comes from sewage sludge-amended home gardens, and pollutant-specific plant uptake slopes. Values for the daily dietary consumption of crops and the fraction of consumption derived from sewage sludge-amended soil are presented in Exhibit 4-8.

EXHIBIT 4-8
Dietary Assumptions for Pathway 2

Crop	Daily Dietary Consumption of Crop (g/day)[1]	Fraction of Consumption Derived from Sewage Sludge-Amended Soil[2]
Garden Fruits	4.15	0.58
Grains and Cereals	89.1	0.0043
Leafy Vegetables	1.97	0.58
Fresh Legumes	3.22	0.58
Potatoes	15.6	0.37
Root Vegetables	1.60	0.58
Sweet Corn	1.60	0.58

[1] Values represent the estimated lifetime average daily food intakes for the crops. U.S. EPA, 1992a.
[2] Values are for a Highly Exposed Individual. U.S. EPA, 1992a.

Plant uptake slopes are needed for the seven crops that represent major human dietary intakes and are commonly grown by home gardeners: garden fruits (e.g., tomatoes); grains and cereals (e.g., barley, wheat); leafy vegetables (e.g., swiss chard, cabbage, lettuce); fresh

legumes (e.g., beans, peas); potatoes; root vegetables (e.g., carrots, beets); and sweet corn (U.S. EPA, 1992a). Note that there are three differences between these crops and the agricultural crops used to evaluate exposure in Pathway 1. First, only fresh legumes are considered for Pathway 2, not both fresh and dry, because home gardeners do not usually grow the dried legumes they consume. Second, peanuts are not considered in Pathway 2, also because home gardeners do not usually grow the peanuts they consume. Third, sweet corn is separated out as a food group for home gardeners because so many gardeners grow sweet corn. In Pathway 1, sweet corn is included in the category of cereals and grains. In Pathway 2, the percent of sweet corn that is homegrown differs from the percent of grains and cereals that are homegrown, and thus the two food categories are separated. The non-agricultural crops of Pathway 1, berries and mushrooms, which are grown on forest land, reclaimed sites, and public contact sites, are not considered relevant for home gardens.

Data on plant uptake slopes did not exist for all pollutant candidates and for all crops. When no data were available for a particular crop, the following extrapolations were made between crops for a given pollutant:

- grain and cereal uptake slopes and forage/pasture uptake slopes (used in animal exposure pathways) were considered interchangeable;
- potato uptake slopes and root vegetable uptake slopes were considered interchangeable;
- peanut uptake slopes and legume uptake slopes were considered interchangeable; and
- any vegetative or leafy growth uptake slopes identified in a study (e.g., soybean leaves) were used for leafy vegetable uptake slopes if no leafy vegetable studies could be identified.

When multiple data points were available for a particular pollutant and crop from a variety of studies, the average of the data from the most appropriate studies was used. The appropriateness of a given study was determined from the study hierarchy established in Round One: data from sludge-amended field studies were preferred over data from sludge-amended pot studies, which in turn were preferred over data from metal-salt-amended field or pot studies. If uptake slope data existed for a particular pollutant in a particular crop from more than one study of the same hierarchical level, they were averaged.

Even after all of the above extrapolations were made, many pollutants still had data gaps. If uptake slope data were not available or could not be estimated using the above extrapolations for all seven crops for a particular pollutant, then exposure to that pollutant was not estimated. Available plant uptake slopes are presented in Exhibit 4-9. Note that the exhibit includes only the 14 pollutant candidates for which there were uptake data for at least one crop. Only three pollutant candidates could be evaluated for this exposure pathway because only three pollutant candidates had uptake slope data available for all seven crops.

EXHIBIT 4-9
Available Plant Uptake Slopes for Agricultural Pathway 2

Pollutant	Plant Uptake Slopes (μg/g crop tissue per μg/g soil)						
	Garden Fruits	Grains and Cereals	Leafy Vegetables	Fresh Legumes	Potatoes	Root Vegetables	Sweet Corn
Aluminum		3.13[1]	0.0026[2]				
Antimony			0.103[3]				
Barium			0.0695[4]				
Beryllium		0.31[5]	12.2[6]		0.49[7]	0.49[7]	
Boron	2.29[8]	3.9[9]	3.115[10]	109.075[11]			2.29[8]
Dioxins and Dibenzofurans	0.000036[12]	0.0043[13,14]	0.009[13]	0.0043[13]	0.00145[12]	0.00145[12]	0.000036[12]
Fluoride	0.35[15]	0.4367[16]	1.875[17]	2.24[18]	0.2475[19]	0.2475[19]	0.35[15]
Manganese		9.98[20]	1.5565[21]		8.4[22]	8.4[22]	
Polychlorinated biphenyls (coplanar)		2[23]					
Silver		0.021[24]					
Thallium			22.47[25]	18.73[26]			
Tin			0.3352[27]				
Titanium	0.136[28]	0.0325[29]	0.028[30]	0.7288[31]	0.13[32]	0.13[32]	0.136[28]
Vanadium			0.0062[33]				

Key to Study Type:
A: Based on sewage sludge field study.
B: Based on non-field sewage sludge study.
C: Based on non-sewage sludge study.

Footnotes:
[1] Muchovej et al., 1986 (ryegrass: 3.13): C.
[2] Chaney et al., 1978 (swiss chard: 0.00301): A; Babish et al., 1979 (cabbage: 0.0021): A.
[3] Chaney et al., 1978 (swiss chard: 0.194): A; Babish et al., 1979 (cabbage: 0.012): A.
[4] Chaney et al., 1978 (swiss chard: 0.097): A; Babish, et al., 1979 (cabbage: 0.042): A.
[5] Bohn and Seekamp, 1979 (oats: 0.31): C.
[6] Kosak-Channing, 1986 (tobacco leaves: 12.2): C.
[7] Bohn and Seekamp, 1979 (potatoes: 0.49): C.
[8] Soon and Bates, 1985 (corn kernels: 2.29): A.
[9] Soon and Bates, 1985 (bromegrass: 3.9): A.
[10] Chaney et al., 1978 (swiss chard: 3.43): A; Babish et al., 1979 (cabbage: 2.8): A.
[11] Agbenin et al., 1991 (cowpea: 134.4): C; Stark and Redente, 1990 (alfalfa: 83.75): C.
[12] U.S. EPA, 1992d. Estimated from model. Assumed dry weight basis.
[13] U.S. EPA, 1994a. Fresh weight basis.
[14] Assumed to be the same as legumes.
[15] Doss et al., 1977 (tomatoes: 0.35): C.
[16] Davis, 1980 (first cut rye: 0.18); Singh et al., 1979b (wheat: 0.8): C; Singh et al., 1979a (rice: 0.33): C.
[17] Bar-Yosef and Rosenberg, 1988 (corn shoot: 3.4, tomato shoot: 3.8): C; Doss et al., 1977 (lettuce: 0.5, petunia leaves: 1.0, potato leaves: 0.675): C.
[18] Stark and Redente, 1990 (alfalfa: 2.24): C.
[19] Doss et al., 1977 (potato: 0.2475): C.
[20] Kalbasi, 1988 (sorghum: 9.98): C.
[21] Chaney et al., 1978 (swiss chard: 3.09): A; Babish et al., 1979 (cabbage: 0.023): A.
[22] Denduluri, 1993 (okra roots: 8.4): C.
[23] Webber et al., 1983 (whole oat: 2.0): A.
[24] Romney et al., 1977 (barley: 0.021): C.
[25] Kaplan et al., 1990 (bean leaves: 22.47): C.
[26] Kaplan et al., 1990 (bean pods: 18.73): C.
[27] Chaney et al., 1978 (swiss chard: 0.6604): A; Babish et al., 1979 (cabbage: 0.01): A.
[28] Tonkonozhenko and Khlyupina, 1974 (corn kernels: 0.136): C.
[29] Tonkonozhenko and Khlyupina, 1974 (winter wheat grains: 0.035, rice grains: 0.03): C.
[30] Babish et al., 1979 (cabbage: 0.028): A.
[31] Tonkonozhenko and Khlyupina, 1974 (peas: 0.2375, beans: 1.22): C.
[32] Tonkonozhenko and Khlyupina, 1974 (sugar beets: 0.13): C.
[33] Chaney et al., 1978 (swiss chard: 0.00645): A; Babish et al., 1979 (cabbage: 0.006): A.

Example Exposure Calculation for Pathway 2

The following example presents the calculations for estimating human exposure to fluoride in sewage sludge applied to a home garden. The example uses the equations and input data presented above for Pathway 2.

First, the mass of soil in the mixing zone, *MS*, is estimated for a home garden by using Eq. 4-2:

$$MS = \frac{1600kg}{m^3} \cdot 15cm \cdot \frac{10^{-1}Mg/ha}{kg \cdot cm/m^3} = \frac{2400Mg}{ha}$$

where:

1600 = BD_{soil} (bulk density of soil) from Exhibit 4-3,
15 = d (depth of incorporation for agricultural land) from Exhibit 4-3, and
10^{-1} = constant to convert (kg · cm/m^3) to (Mg/ha).

Then, the concentration of fluoride in soil is calculated using Eq. 4-1:

$$CT_{fluoride} = \frac{\left(\frac{220mg}{kg} \cdot \frac{2400Mg}{ha}\right) + \left(20yr \cdot \frac{411mg}{kg} \cdot \frac{7Mg}{ha-yr}\right)}{\left(20yr \cdot \frac{7Mg}{ha-yr}\right) + \frac{2400Mg}{ha}}$$

$$= 230mg/kg$$

where:

220 = BS_j (background concentration of fluoride in soil) from Exhibit 4-4,
2400 = MS (mass of soil in home garden mixing zone), estimated above,
20 = N (total number of years sewage sludge is applied to home garden) from Exhibit 4-3,
411 = C_j (fluoride concentration in sewage sludge) from Exhibit 4-1, and
7 = AR (application rate of sewage sludge to home garden) from Exhibit 4-3.

Total dietary exposure is then determined using Eq. 4-4:

$$EXP_{fluoride} = \left[\frac{10^{-3}\ kg/g \cdot 230\ mg/kg}{70\ kg}\right] \cdot [(0.35g/g \cdot 0.58 \cdot 4.15g/day) + (0.44g/g \cdot 0.0043 \cdot 89.1g/day) + (1.9g/g \cdot 0.58 \cdot 1.97g/day) + (2.2g/g \cdot 0.58 \cdot 3.22g/day) + (0.25g/g \cdot 0.37 \cdot 15.6g/day) + (0.25g/g \cdot 0.58 \cdot 1.6g/day) + (0.35g/g \cdot 0.58 \cdot 1.6g/day)]$$

$$= 0.031 mg/kg\text{-}day$$

where:

10^{-3}	=	constant to convert units from (g) to (kg),
230	=	CT_j (concentration of fluoride in agricultural soil), estimated above,
70	=	BW (body weight), assumed to be 70 kg,
$0.35 \cdot 0.58 \cdot 4.15$	=	contribution to dietary exposure from garden fruits,
$0.44 \cdot 0.0043 \cdot 89.1$	=	contribution to dietary exposure from grains and cereals,
$1.9 \cdot 0.58 \cdot 1.97$	=	contribution to dietary exposure from leafy vegetables,
$2.2 \cdot 0.58 \cdot 3.22$	=	contribution to dietary exposure from fresh legumes,
$0.25 \cdot 0.37 \cdot 15.6$	=	contribution to dietary exposure from potatoes,
$0.25 \cdot 0.58 \cdot 1.6$	=	contribution to dietary exposure from root vegetables, and
$0.35 \cdot 0.58 \cdot 1.6$	=	contribution to dietary exposure from sweet corn.

Contribution to dietary exposure is derived as the product of UC_{ij} (uptake slope of fluoride into crop) from Exhibit 4-9, FC_i (fraction of dietary consumption of crop grown in sewage sludge-amended soil) from Exhibit 4-8, and DC_i (daily dietary consumption of crop) from Exhibit 4-8.

4.2.3 Pathway 3 - Direct Ingestion of Sewage Sludge by Children

Pathway 3 evaluates children's exposure to pollutants from direct ingestion of sewage sludge applied to land. The agricultural and non-agricultural scenarios are different in their assumptions regarding the age of children who ingest sewage sludge. At agricultural and public contact sites, children ages 1 to 6 are assumed to be exposed, whereas for forest and reclaimed sites, only older children ages 4 to 6 are assumed to have the opportunity for exposure. For all scenarios, children are assumed to be exposed directly to sewage sludge from storage piles or from the soil surface, not to a sewage sludge/soil mixture.

Data Inputs and Exposure Equation

As in Round One, children (ages 1 to 6) exposed to agricultural land and public contact sites are assumed to ingest 0.2 g soil (dry weight) per day, and weigh 16 kg. Older children (ages 4 to 6) exposed to forest and reclaimed sites are assumed to ingest 0.2 g soil per day, and weigh 19 kg. The ingestion rate of 0.2 g soil per day is a high-end value, but does not represent a PICA child. The body weights are average values. Exposure is calculated as:

$$EXP_j = \frac{I_s \cdot 10^{-3} \cdot C_j}{BW} \qquad (4\text{-}5)$$

where:

EXP_j = exposure to pollutant j in sewage sludge (mg pollutant/kg body weight-day),
IS = sewage sludge ingestion rate (g sewage sludge/day),
10^{-3} = constant to convert units from (g/day) to (kg/day),
C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge), and
BW = body weight (kg).

Example Exposure Calculation for Pathway 3

The exposure of a child to fluoride from directly ingesting sewage sludge applied to agricultural land can be estimated from Eq. 4-5:

$$EXP_{fluoride} = \frac{\frac{0.2g}{day} \cdot \frac{10^{-3}kg}{g} \cdot \frac{411mg}{kg}}{16kg}$$

$$= 0.0051\ mg/kg\text{-}day$$

where:

0.2 = IS (sewage sludge ingestion rate for agricultural land), from Data Inputs section above,

10^{-3} = constant to convert (g) to (kg),
411 = C_j (concentration of fluoride in sewage sludge) from Exhibit 4-1, and
16 = BW (body weight of child assumed to be exposed to agricultural land), from Methods section above.

4.2.4 Pathway 4 - Ingestion of Animal Products Produced From Animals Consuming Forage/Pasture Grown on Sewage Sludge-Amended Soil

Pathway 4 calculates human exposure to pollutants through consumption of animals that ingest forage/pasture grown on sewage sludge-amended soils. In the agricultural Pathway 4, animals ingest forage and pasture produced on sewage sludge-amended soil. Humans then ingest animal products, such as beef, pork, lamb, poultry, dairy products, and eggs. The non-agricultural Pathway 4 examines human exposure to pollutants through consumption of deer and elk that forage on sewage sludge-amended forest land and reclaimed land.

Methods

As in Pathways 1 and 2, to be conservative, the entire mass of pollutant applied in sewage sludge over the life of a land application site is assumed to be available for plant uptake (see Section 4.2.1). In calculating total pollutant concentration in the soil for the agricultural and reclaimed land scenarios, Eq. 4-1 from Section 4.2.1 is used for inorganics and organics. The expected concentrations of pollutants in forage/pasture grown on sewage sludge-amended soil are then estimated using Eq. 4-3 from Section 4.2.1.

To estimate concentrations of pollutants in animal tissues, the forage/pasture pollutant concentrations are combined with animal uptake rates. In this calculation it is assumed that forage is 100 percent of the animal's diet:

$$CA_{jk} = CD_{forage,\,j} \cdot U_{jk} \qquad (4\text{-}6)$$

where:

CA_{jk} = concentration of pollutant j in animal product k (mg pollutant/kg animal tissue),
$CD_{forage,\,j}$ = tissue concentration (dry weight) of pollutant j in forage/pasture (mg pollutant/kg forage/pasture), and
U_{jk} = rate of uptake of pollutant j into animal product k (mg pollutant/kg dry weight animal tissue per mg pollutant/kg dry weight diet).

Once the concentrations of pollutants in animal tissues have been estimated, they are combined with data on the daily dietary consumption of animal products and on the fraction of these animal products that are produced on sewage sludge-amended soil to estimate human exposure:

$$EXP_j = \frac{10^{-3}}{BW} \sum_k CA_{jk}\, FA_k\, DA_k$$

$$= \frac{10^{-3}\, CD_{forage,\, j}}{BW} \sum_k U_{jk}\, FA_k\, DA_k \qquad (4\text{-}7)$$

where:

EXP_j = exposure to pollutant j from animal products produced on sewage sludge-amended soil (mg pollutant/kg body weight-day),

10^{-3} = constant to convert units from (g) to (kg),

BW = body weight (kg), assumed to be 70 kg,

FA_k = fraction of dietary consumption of animal product k produced on sewage sludge-amended soil (dimensionless), and

DA_k = daily dietary consumption of animal product k (g dry weight animal product/day).

Data Inputs

This pathway requires data on pollutant uptake rates into forage/pasture and animal products as well as data on daily dietary consumption of specific animal products and the fraction of that daily consumption that comes from animals feeding on forage/pasture produced on sewage sludge-amended soil. Values used for the latter two parameters are shown in Exhibit 4-10.

EXHIBIT 4-10
Dietary Assumptions for Pathway 4

Animal Product	Daily Dietary Consumption of Animal Product (g/day)[1]	Fraction of Consumption Derived from Sewage Sludge-Amended Soil[2]
Beef (lean)	19.3	9.7×10^{-2}
Beef Fat	15.5	9.7×10^{-2}
Beef Liver (lean and fat)	1.1	9.7×10^{-2}
Dairy (non-fat)	28.9	3.1×10^{-2}
Dairy Fat	18.1	3.1×10^{-2}
Eggs	8.3	7.9×10^{-2}
Lamb (lean)	0.20	9.7×10^{-2}
Lamb Fat	0.21	9.7×10^{-2}

EXHIBIT 4-10
Dietary Assumptions for Pathway 4 (cont'd.)

Animal Product	Daily Dietary Consumption of Animal Product (g/day)[1]	Fraction of Consumption Derived from Sewage Sludge-Amended Soil[2]
Poultry (lean)	6.7	1.1×10^{-1}
Poultry Fat	1.3	1.1×10^{-1}
Pork (lean)	9.0	9.7×10^{-2}
Pork Fat	12.7	9.7×10^{-2}
Deer (lean)	15.3[3]	1
Deer Fat	5.1[3]	1
Deer Liver (total)	0.38[3]	1
Elk (lean)	30.6[3]	0.5
Elk Fat	10.2[3]	0.5
Elk Liver (total)	0.76[3]	0.5

[1] Values represent the estimated lifetime average daily food intakes for the animal products unless otherwise noted. U.S. EPA, 1992a.
[2] Fractions represent reasonable estimates. U.S. EPA, 1992a.
[3] It was assumed that total consumption of deer and elk meat and fat constitutes 50 percent of the HEI's consumption of agricultural animal products. U.S. EPA, 1992a.

If no data were available on pollutant uptake slopes into forage/pasture, it was assumed that grain and cereal uptake slopes and forage/pasture uptake slopes were interchangeable. When multiple data points were available from a variety of studies for the uptake of a particular pollutant into forage/pasture crops, the average of the data from the most appropriate studies was used. The appropriateness of a given study was determined from the study hierarchy established in Round One: data from sewage sludge-amended field studies were preferred over data from sewage sludge-amended pot studies, that in turn were preferred over data from metal-salt-amended field or pot studies. If uptake slope data existed for a particular pollutant from more than one study of the same hierarchical level, they were averaged.

Available plant uptake data are presented in Exhibit 4-11. Note that if forage/pasture uptake data were not available for a particular pollutant, that pollutant is not included in the exhibit.

EXHIBIT 4-11
Forage/Pasture Uptake Slopes for Agricultural Pathway 4

Pollutant	Forage/Pasture Uptake Slopes (μg/g plant per μg/g soil)
Aluminum	3.13[1]
Beryllium	0.31[2]
Boron	3.9[3]
Dioxins and Dibenzofurans	0.024[4]
Fluoride	0.6775[5]
Manganese	8.22[6]
Polychlorinated biphenyls (coplanar)	2.0[7]
Silver	0.021[8]
Titanium	0.0543[9]

Key to Study Type:
A: Based on sewage sludge field study.
B: Based on non-field sewage sludge study.
C: Based on non-sewage sludge study.

Footnotes:
[1] Muchovej et al., 1986 (ryegrass: 3.13): C.
[2] Bohn and Seekamp, 1979 (oats: 0.31): C.
[3] Soon and Bates, 1985 (bromegrass: 1.64): A.
[4] U.S. EPA, 1992d. Estimated from model.
[5] Stark and Redente, 1990 (grasses: 0.58): C; Singh et al., 1979b (wheat: 0.8): C; Singh et al., 1979a (rice: 0.33): C; Doss et al., 1977 (petunia leaves: 1.0): C.
[6] Denduluri, 1993 (okra stems: 11.3, okra leaves: 12.6): C; El-Kherbawy and Sanders, 1984 (clover: 0.76): C.
[7] Webber et al., 1983 (whole oat: 2.0): A.
[8] Romney et al., 1977 (barley: 0.021): C.
[9] Tonkonozhenko and Khlyupina, 1974 (winterwheat leaves+stem: 0.07, clover leaves+stems: 0.063, rice: 0.03): C.

Ideally, for animal uptake slopes, uptake data for each pollutant into each animal product tissue would be available. A literature search revealed information on animal uptake slopes for only four pollutants: aluminum, barium, boron, and manganese. Unfortunately, uptake data were not available for all agricultural animal products of interest. Although uptake data into beef, beef liver, lamb, pork, and non-fat dairy products were considered interchangeable with available uptake data for sheep and goats, extrapolations could not be made to poultry or eggs. For non-agricultural animal products, however, tissue values for sheep, goats, deer, and elk were considered interchangeable in the lean and liver tissue categories. Therefore, risk estimates could be made for forests and reclaimed land. For coplanar PCBs, uptake values were taken directly from Round One values for PCBs.

Exhibit 4-12 presents the available animal uptake slopes for the agricultural animal products of interest; Exhibit 4-13 presents the available uptake slopes for non-agricultural animals. As before, note that only those pollutant candidates with data for at least one animal uptake slope are included in the exhibits.

EXHIBIT 4-12
Animal Uptake Slopes for Agricultural Pathway 4

Pollutant	Animal Uptake Slopes (μg/g animal per μg/g feed)											
	Beef Fat	Beef Liver	Beef (Lean)	Lamb Fat	Lamb (Lean)	Pork Fat	Pork (Lean)	Poultry Fat	Poultry (Lean)	Dairy Fat	Dairy (Non-Fat)	Eggs
Aluminum	NA	0.007[1]	0.021[1]	NA	0.021[1]	NA	0.021[1]	NA		NA	0.021[1]	
Barium	NA	0.0535[3]	0.088[3]	NA	0.088[3]	NA	0.088[3]	NA		NA	0.088[3]	
Boron	NA	0.0915[1]	0.091[1]	NA	0.091[1]	NA	0.091[1]	NA		NA	0.091[1]	
Manganese	NA	0.00925[5]	0.0005[4]	NA	0.00925[5]	NA	0.00925[5]	NA		NA	0.0005[4]	
Polychlorinated biphenyls (coplanar)	4.215[2]	6.664[2]	NA	6.664[2]	NA	6.664[2]	NA	6.664[2]	NA	10.536[2]	NA	10.536[2]

NA means Not Applicable; it is assumed that inorganic pollutants are not taken up into fat, and organic pollutants are not taken up into lean.
[1] Bray et al., 1985 (goat).
[2] U.S. EPA, 1992a.
[3] Whelan, 1993 (sheep).
[4] Voigt et al., 1988 (beef).
[5] Bray et al., 1985; Voigt et al., 1988.

EXHIBIT 4-13
Animal Uptake Slopes for Non-Agricultural Pathway 4

Pollutant	Animal Uptake Slopes (μg/g animal per μg/g feed)					
	Deer (Lean)	Deer Fat	Deer Liver	Elk (Lean)	Elk Fat	Elk Liver
Aluminum	0.021[1]	NA	0.007[1]	0.021[1]	NA	0.007[1]
Barium	0.088[3]	NA	0.0535[3]	0.088[3]	NA	0.0535[3]
Boron	0.091[1]	NA	0.0915[1]	0.091[1]	NA	0.0915[1]
Manganese	0.00925[4]	NA	0.00925[4]	0.00925[4]	NA	0.00925[4]
Polychlorinated biphenyls (coplanar)	NA	4.215[2]	6.664[2]	NA	4.215[2]	6.664[2]

NA means Not Applicable.
[1] Bray et al., 1985 (goat).
[2] U.S. EPA, 1992a.
[3] Whelan, 1993 (sheep:muscle).
[4] Bray et al., 1985 (goat); Voigt et al., 1988 (beef).

Example Exposure Calculation for Pathway 4

The following example presents the calculations for estimating human exposure to boron from consumption of deer and elk which have eaten forage plants on forest lands amended with sewage sludge. Because it is assumed that there is no incorporation of sewage sludge into forest soils, the concentration of boron in sewage sludge is used as the relevant concentration for uptake into the forage. The concentration of boron in forage is calculated using Eq. 4-3:

$$CD_{forage,\ boron} = 182\ \frac{mg}{kg} \cdot 3.9\ \frac{\mu g/g}{\mu g/g}$$

$$= 710\ mg/kg$$

where:

182 = CT_j (concentration of boron in forest soil, equivalent to the concentration of boron in sewage sludge) from Exhibit 4-1, and

3.9 = UC_{ij} (uptake slope of boron into forage/pasture) from Exhibit 4-11.

Total dietary exposure to boron from wild animals eating plants in sewage sludge-amended forests is then determined using Eq. 4-7:

$$EXP_{boron} = [\frac{10^{-3}\ kg/g \cdot 710\ mg/kg}{70\ kg}] \cdot [(0.091 g/g \cdot 1 \cdot 15.3 g/day) + (0.092 g/g \cdot 1 \cdot 0.38 g/day) + (0.091 g/g \cdot 0.5 \cdot 30.6 g/day) + (0.092 g/g \cdot 0.5 \cdot 0.76 g/day)]$$

$$= 0.029 mg/kg\text{-}day$$

where:

10^{-3}	=	constant to convert units from (g) to (kg),
710	=	CT_j (concentration of boron in forest soil), estimated above,
70	=	BW (body weight), assumed to be 70 kg,
$0.091 \cdot 1 \cdot 15.3$	=	contribution to dietary exposure from lean deer,
$0.092 \cdot 1 \cdot 038$	=	contribution to dietary exposure from deer liver,
$0.091 \cdot 0.5 \cdot 30.6$	=	contribution to dietary exposure from lean elk, and
$0.092 \cdot 0.5 \cdot 0.76$	=	contribution to dietary exposure from elk liver.

Contribution to dietary exposure is derived as the product of UC_{jk} (uptake slope of boron into animal product) from Exhibit 4-13, FA_k (fraction of dietary consumption of animal product derived from sewage sludge-amended soil) from Exhibit 4-10, and DA_k (daily dietary consumption of animal product) from Exhibit 4-10.

4.2.5 Pathway 5 - Consumption of Animal Products Produced From Animals That Ingest Sewage Sludge

Pathway 5 involves the application of sewage sludge to land, the direct ingestion of sewage sludge by animals, and finally, the consumption of animal products by humans. Agricultural Pathway 5 considers only the direct ingestion of sewage sludge by livestock, following the surface application of sewage sludge to pasture crops. Non-agricultural Pathway 5 considers the direct ingestion of sewage sludge by livestock that graze on grasses growing on forest land or reclaimed land; the animals are then ingested by humans. The pathway does not consider the grazing of livestock on public contact sites because it is assumed such grazing would be controlled. The pathway also does not consider wild animals in forest land because deer do not graze on plants close to the ground and would not inadvertently ingest sewage sludge. Furthermore, other wild herbivorous game animals are assumed to graze over large territories.

Methods

To estimate concentrations of pollutants in animal tissues, sewage sludge pollutant concentrations are combined with percent of animal ingestion that is sewage sludge and tissue uptake rates. In this calculation the sewage sludge pollutant concentration multiplied by the

animal's percentage sewage sludge consumption and pollutant uptake rate is assumed to equal the resulting animal tissue concentration of the pollutant:

$$CA_{jk} = C_j \cdot FS \cdot U_{jk} \qquad (4\text{-}8)$$

where:

CA_{jk} = concentration of pollutant j in animal product k (mg pollutant/kg animal tissue),

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

FS = fraction of animal's diet that is sewage sludge (unitless, kg sewage sludge/kg diet), and

U_{jk} = rate of uptake of pollutant j into animal product k (mg pollutant/kg dry weight animal tissue per mg pollutant/kg dry weight diet).

Once the concentrations of pollutants in animal tissues have been estimated, they are combined with data on the daily dietary consumption of animal products and on the fraction of those animal products that are produced on sewage sludge-amended soil to estimate human exposure:

$$EXP_j = \frac{10^{-3}}{BW} \sum_k CA_{jk}\, FA_k\, DA_k$$

$$= \frac{10^{-3} \cdot C_j \cdot FS}{BW} \sum_k U_{jk}\, FA_k\, DA_k \qquad (4\text{-}9)$$

where:

EXP_j = exposure to pollutant j from animal products produced on sewage sludge-amended soil (mg pollutant/kg body weight-day),

10^{-3} = constant to convert units from (g) to (kg),

BW = body weight (kg), assumed to be 70 kg,

FA_k = fraction of dietary consumption of animal product k produced on sewage sludge-amended soil (dimensionless), and

DA_k = daily dietary consumption of animal product k (g dry weight animal product/day).

Data Inputs

For this pathway, there are four data inputs required to calculate human exposure from consumption of animal products produced from animals that ingest sewage sludge: the percentage of animal diet that consists of sewage sludge, the daily dietary consumption of specific animal

products, the fraction of that daily consumption which comes from animals that ingest sewage sludge, and animal uptake rates of pollutants.

The percentage of a grazing cattle's diet that is sewage sludge, averaged over a season, is estimated to be 2.5 percent (Chaney et al., 1987; Bertrand et al., 1981). However, given that in any one year, the maximum percentage of a farm treated with sewage sludge is approximately 33 percent, and assuming that livestock are rotated among several pasture fields, the actual percentage of the diet that is sewage sludge is assumed to be lower than 2.5 percent. The diet for cattle grazing on land treated with sewage sludge that was applied the previous growing season has been shown to be approximately 1.0 percent sewage sludge (Decker et al., 1980). When a weighted average is calculated from these two values of sewage sludge ingestion, the long-term average percentage of diet that is sewage sludge is 1.5 percent: $\frac{1}{3}(2.5\%) + \frac{2}{3}(1.0\%) = 1.5\%$ (Chaney et al., 1991).

Values for the daily dietary human consumption of specific animal products and the fraction of consumption from sewage sludge-amended soil are shown in Exhibit 4-14. For animal uptake slopes for beef, dairy, and lamb products, see Exhibit 4-12 in Section 4.2.4.

EXHIBIT 4-14
Dietary Assumptions for Pathway 5

Animal Product	Daily Dietary Consumption of Animal Product (g/day)[1]	Fraction of Consumption Derived from Sewage Sludge-Amended Soil[2]
Beef (Lean)	19.3	9.7×10^{-2}
Beef Fat	15.5	9.7×10^{-2}
Beef Liver	1.1	9.7×10^{-2}
Dairy (Non-Fat)	28.9	3.1×10^{-2}
Dairy Fat	18.1	3.1×10^{-2}
Lamb (Lean)	0.20	9.7×10^{-2}
Lamb Fat	0.21	9.7×10^{-2}

[1] Values represent the estimated lifetime average daily food intakes for the animal products unless otherwise noted. U.S. EPA, 1992a.
[2] Fractions represent reasonable estimates. U.S. EPA, 1992a.

Example Exposure Calculation for Pathway 5

The following example presents the calculation for estimating human exposure to manganese from consumption of livestock that ingest sewage sludge on sewage sludge-amended forest land. Using Eq. 4-9:

$$EXP_{manganese} = \left[\frac{10^{-3}\frac{kg}{g} \cdot 1620\frac{mg}{kg} \cdot 0.015}{70kg}\right] \left[\left(0.00925\frac{g}{g} \cdot 9.7x10^{-2} \cdot 1.1\frac{g}{day}\right) + \left(0.0005\frac{g}{g} \cdot 9.7x10^{-2} \cdot 19.3\frac{g}{day}\right) + \left(0.00925\frac{g}{g} \cdot 9.7x10^{-2} \cdot 0.20\frac{g}{day}\right) + \left(0.0005\frac{g}{g} \cdot 3.1x10^{-2} \cdot 28.9\frac{g}{day}\right)\right]$$

$$= 8.9x10^{-7}\frac{mg}{kg-day}$$

where:

10^{-3}	=	constant to convert units from (g) to (kg),
1620	=	C_j (concentration of manganese in sewage sludge) from Exhibit 4-1,
0.015	=	*FS* (fraction of animal's diet that is sewage sludge), discussed above,
70	=	*BW* (body weight), assumed to be 70 kg,
$(0.00925 \cdot 9.7 \times 10^{-2} \cdot 1.1)$	=	contribution to dietary exposure from beef liver,
$(0.0005 \cdot 9.7 \times 10^{-2} \cdot 19.3)$	=	contribution to dietary exposure from lean beef,
$(0.00925 \cdot 9.7 \times 10^{-2} \cdot 0.20)$	=	contribution to dietary exposure from lean lamb, and
$(0.0005 \cdot 3.1 \times 10^{-2} \cdot 28.9)$	=	contribution to dietary exposure from non-fat dairy.

Contribution to dietary exposure is derived as the product of U_{jk} (uptake of manganese into animal product) from Exhibit 4-12, FA_k (fraction of dietary consumption of animal product derived from sewage sludge-amended land) from Exhibit 4-14, and DA_k (daily dietary consumption of animal product) from Exhibit 4-14.

4.2.6 Pathway 6 - Animal Toxicity From Plant Consumption

Pathway 6 calculates herbivorous animal toxicity caused by the consumption of plants that are grown on sewage sludge-amended soil on both agricultural and non-agricultural lands. Non-agricultural lands include forests, reclaimed land, and public contact sites. In the agricultural pathway, the HEI is the most sensitive herbivorous livestock that consumes plants grown on sewage sludge-amended soil. For the non-agricultural forest land Pathway 6, two exposure scenarios are possible. In one, the HEI is a small herbivorous mammal that spends its entire life in a sewage sludge-amended area feeding on seeds and small plants close to the sewage sludge/soil layer. In the second scenario, the HEI is an herbivorous livestock that grazes on the

grasses growing on sewage sludge-amended forest land. The HEI for reclaimed land is livestock; the HEI for public contact sites is a small herbivorous mammal.

Methods

For the agricultural Pathway 6, the animals of interest are herbivorous livestock. For the non-agricultural Pathway 6, the animals of interest are small herbivorous mammals as well as herbivorous livestock. Eq. 4-1 from Section 4.2.1 is used to calculate total pollutant concentration in the soil (CT_j) for the agricultural and reclaimed land scenarios. Exposure is reported in terms of milligrams pollutant per kilogram of diet, assuming that the herbivore receives its total diet from forage/pasture grown on sewage sludge-amended land. Therefore, the dietary exposure for the HEI can be expressed as:

$$EXPA_j = CD_j = CT_j \cdot UC_{forage,\,j} \qquad (4\text{-}10)$$

where:

$EXPA_j$	=	exposure of animal to pollutant j (mg pollutant/kg diet),
CD_j	=	tissue concentration (dry weight) of pollutant j in forage/pasture (mg pollutant/kg forage/pasture),
CT_j	=	concentration of pollutant j in sewage sludge-amended soil (mg pollutant/kg sewage sludge-amended soil), and
$UC_{forage,j}$	=	rate of uptake of pollutant j into tissue of forage/pasture (mg pollutant/kg dry weight forage/pasture per mg pollutant/kg soil).

Data Inputs

For this pathway, data are needed on pollutant uptake rates into forage/pasture. For available uptake rates into forage/pasture, see Exhibit 4-11 in Section 4.2.4.

Example Exposure Calculation for Pathway 6

The following example presents the calculations for estimating exposure of herbivorous livestock to fluoride in sewage sludge applied to agricultural land.

First, the mass of soil in the mixing zone, MS, is estimated for agricultural land by using Eq. 4-2:

$$MS = \frac{1600kg}{m^3} \cdot 15cm \cdot \frac{10^{-1}Mg/ha}{kg \cdot cm/m^3} = \frac{2400Mg}{ha}$$

where:

1600	=	BD_{soil} (bulk density of soil) from Exhibit 4-3,
15	=	d (depth of incorporation for agricultural land) from Exhibit 4-3, and
10^{-1}	=	constant to convert (kg · cm/m^3) to (Mg/ha).

Then, the concentration of fluoride in agricultural soil is calculated using Eq. 4-1:

$$CT_{fluoride} = \frac{\left(\frac{220mg}{kg} \cdot \frac{2400Mg}{ha}\right) + \left(20yr \cdot \frac{411mg}{kg} \cdot \frac{7Mg}{ha\text{-}yr}\right)}{\left(20yr \cdot \frac{7Mg}{ha\text{-}yr}\right) + \frac{2400Mg}{ha}}$$

$$= 230\,mg/kg$$

where:

220	=	BS_j (background concentration of fluoride in soil) from Exhibit 4-4,
2400	=	MS (mass of soil in agricultural mixing zone), estimated above,
20	=	N (total number of years sewage sludge is applied to agricultural land) from Exhibit 4-3,
411	=	C_j (fluoride concentration in sewage sludge) from Exhibit 4-1, and
7	=	AR (application rate of sewage sludge to agricultural land) from Exhibit 4-3.

The expected exposure of livestock to fluoride in forage/pasture is calculated using Eq. 4-10:

$$EXP_{fluoride} = \frac{230mg}{kg} \cdot \frac{0.68g}{g}$$

$$= 160\ mg/kg$$

where:

230	=	CT_j (concentration of fluoride in agricultural soil), estimated above, and
0.68	=	$UC_{forage,j}$ (uptake slope of fluoride into forage/pasture) from Exhibit 4-11.

4.2.7 Pathway 7 - Animal Toxicity From Direct Ingestion of Sewage Sludge

This pathway calculates herbivorous animal toxicity caused by the direct ingestion of sewage sludge on both agricultural and non-agricultural lands. Non-agricultural lands include forests and reclaimed land. For both the agricultural and non-agricultural pathways, the HEI is the most sensitive or most exposed herbivorous livestock that directly ingests sewage sludge from sewage sludge-amended soil. This pathway does not consider exposure to sewage sludge on public contact sites because livestock usually are not grazed there.

Methods

For both the agricultural and non-agricultural Pathway 7, the HEI is herbivorous livestock. Exposure is reported in terms of milligrams pollutant per kilogram of diet, assuming

that the herbivore receives its total diet from forage/pasture grown on sewage sludge-amended land. Therefore, the dietary exposure for the HEI can be expressed as:

$$EXPA_j = C_j \cdot FS \qquad (4\text{-}11)$$

where:

$EXPA_j$ = exposure of animal to pollutant j (mg pollutant/kg diet),
C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge), and
FS = fraction of animal's diet that is sewage sludge (unitless, kg sewage sludge/kg diet).

Data Inputs

For this pathway, data are required on the percentage of animal diet that consists of sewage sludge. As described for Pathway 5, the percentage of a grazing cattle's diet that is sewage sludge, averaged over a season, is estimated to be 2.5 percent (Chaney et al., 1987; Bertrand et al., 1981). However, given that in any one year, the maximum percentage of a farm treated with sewage sludge is approximately 33 percent, and assuming that livestock are rotated among several pasture fields, the actual percentage of the diet that is sewage sludge is assumed to be lower than 2.5 percent. The diet for cattle grazing on land treated with sewage sludge that was applied the previous growing season has been shown to be approximately 1.0 percent sewage sludge (Decker et al., 1980). When a weighted average is calculated from these two values of sewage sludge ingestion, the long-term average percentage of diet that is sewage sludge is 1.5 percent: $\frac{1}{3}(2.5\%) + \frac{2}{3}(1.0\%) = 1.5\%$ (Chaney et al., 1991).

Example Exposure Calculation for Pathway 7

The exposure of herbivorous livestock to manganese from direct ingestion of sewage sludge is calculated from Eq. 4-11:

$$EXP_{manganese} = 1620\frac{mg}{kg} \cdot 0.015$$
$$= 24\frac{mg}{kg}$$

where:

1620 = C_j (concentration of manganese in sewage sludge) from Exhibit 4-1, and
0.015 = FS (fraction of animal's diet that is sewage sludge), discussed above.

4.2.8 Pathway 8 - Toxicity to Plants

This pathway could not be evaluated due to a lack of data on the phytotoxicity effects of the candidate Round Two pollutants.

4.2.9 Pathway 9 - Toxicity to Soil-Dwelling Organisms

Pathway 9 evaluates toxicity to soil-dwelling organisms due to the presence of pollutants in sewage sludge that is land-applied to agricultural and non-agricultural lands. Non-agricultural lands include forests, reclaimed land, and public contact sites. The soil-dwelling organisms considered are earthworms. There is no evidence that earthworms are the most sensitive species; however, because of a lack of data for other soil-dwelling species, earthworms are considered the HEI for this pathway.

Data Inputs and Exposure Equation

For each pollutant and type of land application site, the concentration of pollutant in the soil had to be calculated. For agricultural and reclaimed land, the pollutant concentration was calculated using Eq. 4-1 in Section 4.2.1. For forests and public contact sites, it was assumed that there would be no incorporation of land-applied sewage sludge. This implies that the soil layer to which the soil-dwelling organisms are exposed is pure sewage sludge. Therefore, the concentration of each pollutant in the exposure layer ("soil") was set equal to its concentration in the sewage sludge. For this pathway, exposure to pollutants in soil by earthworms is measured by the concentration of the pollutants in the sewage sludge/soil:

$$EXPO_j = CT_j \qquad (4\text{-}12)$$

where:

$EXPO_j$ = exposure of soil-dwelling organisms to pollutant j (mg pollutant/kg sewage sludge-amended soil), and

CT_j = concentration of pollutant j in sewage sludge-amended soil (mg pollutant/kg sewage sludge-amended soil).

Example Exposure Calculation for Pathway 9

The following example estimates exposure of earthworms to manganese in agricultural soil.

For agricultural land, the mass of soil in the mixing zone, MS, must first be estimated by using Eq. 4-2:

$$MS = \frac{1600kg}{m^3} \cdot 15cm \cdot \frac{10^{-1}Mg/ha}{kg \cdot cm/m^3} = \frac{2400Mg}{ha}$$

where:

1600	=	BD_{soil} (bulk density of soil) from Exhibit 4-3,
15	=	d (depth of incorporation for agricultural land) from Exhibit 4-3, and
10^{-1}	=	constant to convert (kg · cm/m^3) to (Mg/ha).

Then, the concentration of manganese in the soil, and thus the earthworm's exposure to manganese, is calculated using Eqs. 4-1 and 4-10:

$$CT_{manganese} = EXPO_{manganese} = \frac{\left(\frac{342mg}{kg} \cdot \frac{2400Mg}{ha}\right) + \left(20yr \cdot \frac{1620mg}{kg} \cdot \frac{7Mg}{ha-yr}\right)}{\left(20yr \cdot \frac{7Mg}{ha-yr}\right) + \frac{2400Mg}{ha}}$$

$$= 410mg/kg$$

where:

342	=	BS_j (background concentration of manganese in soil) from Exhibit 4-4,
2400	=	MS (mass of soil in agricultural mixing zone), estimated above,
20	=	N (total number of years sewage sludge is applied to agricultural land) from Exhibit 4-3,
1620	=	C_j (manganese concentration in sewage sludge) from Exhibit 4-1, and
7	=	AR (application rate of sewage sludge to agricultural land) from Exhibit 4-3.

4.2.10 Pathway 10 - Toxicity to Predators of Soil-Dwelling Organisms

Pathway 10 evaluates toxicity to animals feeding on soil-dwelling organisms living in sewage sludge-amended soils on both agricultural and non-agricultural lands. Non-agricultural lands include forests, reclaimed land, and public contact sites. For both the agricultural and non-agricultural Pathway 10, the HEI is a small insectivorous mammal (e.g., shrew or mole), that consumes soil-dwelling organisms. The soil-dwelling organisms considered in this pathway are earthworms.

Methods

For exposure to occur through this pathway, soil-dwelling organisms must bioconcentrate or bioaccumulate pollutants from the sewage sludge-amended soil. For agricultural and reclaimed land, the concentration of pollutant in the soil is calculated using Eq. 4-1 in Section 4.2.1. For forests and public contact sites, it is assumed that there would be no incorporation

of land-applied sewage sludge. This implies that the soil layer to which the soil-dwelling organisms are exposed is pure sewage sludge. Therefore, the concentration of each pollutant in the exposure layer ("soil") is set equal to its concentration in the sewage sludge.

Exposure for Pathway 10 is reported in terms of milligrams pollutant per kilogram of diet for the insectivorous mammal, assuming that the only source of the pollutant in the mammal's diet is from ingestion of contaminated soil-dwelling organisms (earthworms). This dietary concentration is referred to as the "pollutant intake level" and is represented by PIL_j in the following equation:

$$EXPA_j = PIL_j = CT_j \cdot BACC_j \cdot FD \qquad (4\text{-}13)$$

where:

$EXPA_j$	=	exposure of insectivorous mammal to pollutant j (mg pollutant/kg diet),
PIL_j	=	intake level of pollutant j in insectivorous mammal's diet (mg pollutant/kg diet),
CT_j	=	concentration of pollutant j in sewage sludge-amended soil (mg pollutant/kg sewage sludge-amended soil),
$BACC_j$	=	bioaccumulation factor for pollutant j (mg pollutant/kg soil organisms per mg pollutant/kg sewage sludge-amended soil), and
FD	=	fraction of diet considered to be soil organisms (unitless, kg soil organisms/kg diet).

First, the concentration of the pollutant in the soil-dwelling prey is calculated by multiplying the pollutant concentration in the soil (CT_j) by a bioaccumulation factor ($BACC_j$). To adjust the pollutant concentration in soil-dwelling organisms to the analogous concentration in the entire diet of the insectivorous mammal, the concentration in the soil-dwelling organisms is then multiplied by the fraction of the insectivorous mammal's diet that consists of soil-dwelling organisms.

Data Inputs

There are two data inputs required for this pathway: fraction of diet considered to be soil organisms and bioaccumulation factors. As in Round One, it is assumed that the fraction of the HEI's diet that is composed of soil-dwelling organisms is one-third, based on a consideration of maximum chronic consumption of earthworms by wildlife (U.S. EPA, 1992a).

Because earthworms are generally the most conspicuous prey item of the soil biota and are considered a potential vector for the transfer of sewage sludge pollutants up the food chain, a number of studies have measured bioaccumulation in earthworms. Most of the studies, however, have been focused on a select set of metals (e.g., cadmium, lead, mercury, selenium, zinc, copper, nickel, and chromium) or persistent chlorinated hydrocarbons, such as DDT (see Beyer, 1990; Gillett, 1994). Very few data were found on bioaccumulation of Round Two candidate pollutants by earthworms from soil.

Bioaccumulation of Organic Pollutants. No empirical data were found on bioaccumulation in soil organisms for any of the candidate organic pollutants for Round Two except dioxins and dibenzofurans. A predictive equation that describes bioaccumulation of all organic compounds in earthworms inhabiting contaminated soils was found (Menzie et al., 1992). This equation is based on a relationship between the fraction of organic carbon in the soil and the lipid content of earthworms:

$$BACC = \frac{Y_1}{0.66 \cdot f_{oc}}$$

where:

$BACC$	=	bioaccumulation factor for earthworms (dimensionless),
Y_1	=	lipid content of earthworms (fraction),
0.66	=	constant derived by Menzie et al., 1992, and
f_{oc}	=	the fraction of organic carbon in the mixing zone of the sewage sludge-amended soil.

The lipid content of earthworms can be assumed to be two percent, as reported in Menzie et al. (1992) for the earthworm *Lumbricus terrestris*. Assuming the fraction of organic carbon in the mixing zone of the sewage sludge-amended soil to be 0.01 (U.S. EPA, 1992a), the bioaccumulation factor in earthworms for all organic pollutants would be 3.0. This value was not used in Round Two, however, because it is not pollutant-specific.

Bioaccumulation of Inorganic Pollutants. Very few studies were found from which bioaccumulation factors for the inorganic Round Two candidate pollutants could be determined. Walton (1987) investigated sodium fluoride accumulation in earthworms (primarily *Lumbricus terrestris*). From the results of one of the experiments, the bioaccumulation factor for worms (including gut contents) in soil with a high level of fluoride was calculated to be 0.670. This value was used for the *BACC* for fluoride.

Helmke et al. (1979) investigated effects of land-applied sewage sludge on the concentration of many different elements in earthworms (*Aporrectodea tuberculata*). Four of the elements measured were candidate Round Two pollutants: antimony, barium, manganese, and thallium. By taking the ratio of the reported concentrations of these metals in earthworms living in the control soil to the concentrations of the metals in the control soil, *BACCs* were calculated for antimony, barium, manganese, and thallium. The authors noted that the earthworms may not have truly accumulated these metals into their tissues. Instead, the concentrations measured were probably due to the metals in the casts (soil in the gut of the worms). However, because predators of earthworms eat entire earthworms, including casts, these bioaccumulation values were considered appropriate for this analysis. Exhibit 4-15 displays the bioaccumulation factors used for the candidate Round Two pollutants. Note that if pollutants do not have data on bioaccumulation, they are not included in the exhibit.

EXHIBIT 4-15
Bioaccumulation Factors for Soil-Dwelling Organisms

Pollutant	Bioaccumulation Factor	Reference
Antimony	0.13	Helmke et al. (1979)
Barium	0.062	Helmke et al. (1979)
Dioxins and Dibenzofurans	10	U.S. EPA (1994a)
Fluoride	0.67	Walton (1987)
Manganese	0.073	Helmke et al. (1979)
Thallium	0.062	Helmke et al. (1979)

Example Exposure Calculation for Pathway 10

The following example estimates exposure of predators of soil-dwelling organisms to manganese in agricultural soil.

First, the mass of soil in the mixing zone, *MS*, is estimated for agricultural land by using Eq. 4-2:

$$MS = \frac{1600kg}{m^3} \cdot 15cm \cdot \frac{10^{-1}Mg/ha}{kg \cdot cm/m^3} = \frac{2400Mg}{ha}$$

where:

1600 = BD_{soil} (bulk density of soil) from Exhibit 4-3,
15 = d (depth of incorporation for agricultural land) from Exhibit 4-3, and
10^{-1} = constant to convert (kg · cm/m^3) to (Mg/ha).

The concentration of manganese in the soil is then calculated using Eq. 4-1:

$$CT_{manganese} = \frac{\left(\frac{342mg}{kg} \cdot \frac{2400Mg}{ha}\right) + \left(20yr \cdot \frac{1620mg}{kg} \cdot \frac{7Mg}{ha\text{-}yr}\right)}{\left(20yr \cdot \frac{7Mg}{ha\text{-}yr}\right) + \frac{2400Mg}{ha}}$$

$$= 410 mg/kg$$

where:

342	=	BS_j (background concentration of manganese in soil) from Exhibit 4-4,
2400	=	MS (mass of soil in agricultural mixing zone), estimated above,
20	=	N (total number of years sewage sludge is applied to agricultural land) from Exhibit 4-3,
1620	=	C_j (manganese concentration in sewage sludge) from Exhibit 4-1, and
7	=	AR (application rate of sewage sludge to agricultural land) from Exhibit 4-3.

The "pollutant intake level" (*PIL*), and thus exposure of the HEI, is then calculated using Eq. 4-13:

$$EXPA_{manganese} = PIL_{manganese} = 410\frac{mg}{kg} \cdot 0.073 \cdot \frac{1}{3}$$
$$= 10 mg/kg$$

where:

410	=	CT_j (concentration of manganese in agricultural soil), estimated above,
0.073	=	$BACC_j$ (bioaccumulation factor for manganese for soil-dwelling organisms) from Exhibit 4-15, and
⅓	=	FD (fraction of diet that is soil-dwelling organisms), discussed above.

4.2.11 Pathway 11 - Human Toxicity Through Inhalation of Particulates Resuspended by Tilling Sewage Sludge

Pathway 11 evaluates human (tractor operator) exposure to particles that have been resuspended by the tilling of dewatered sewage sludge into the soil. Because this type of exposure is associated only with farming, exposure to sewage sludge applied to non-agricultural land is not evaluated for Pathway 11.

Methods

To calculate the total pollutant concentration in the soil for both organics and inorganics, Eq. 4-1 in Section 4.2.1 is used. To calculate the exposure of the tractor operator to pollutants, the following equation is used:

$$EXPT_j = CT_j \cdot TDA \cdot 10^{-6} \quad (4\text{-}14)$$

where:

$EXPT_j$ = exposure of tractor operator to pollutant j (mg pollutant/m^3 air),
CT_j = concentration of pollutant j in sewage sludge-amended soil (mg pollutant/kg sewage sludge-amended soil),
TDA = total exposure of tractor operator to soil dust (mg soil dust/m^3 air), and
10^{-6} = constant to convert (kg) to (mg).

Data Inputs

As in Round One, the total dust exposure of the tractor operator is assumed to be the total dust standard of 10 mg/m^3 established by the American Conference of Governmental Industrial Hygienists (ACGIH).

Example Exposure Calculation for Pathway 11

The following example estimates exposure of a tractor operator to manganese.

First, the mass of soil in the mixing zone, *MS*, is estimated for agricultural land by using Eq. 4-2:

$$MS = \frac{1600kg}{m^3} \cdot 15cm \cdot \frac{10^{-1}Mg/ha}{kg \cdot cm/m^3} = \frac{2400Mg}{ha}$$

where:

1600 = BD_{soil} (bulk density of soil) from Exhibit 4-3,
15 = d (depth of incorporation for agricultural land) from Exhibit 4-3, and
10^{-1} = constant to convert (kg · cm/m^3) to (Mg/ha).

The concentration of manganese in the soil is then calculated using Eq. 4-1:

$$CT_{manganese} = \frac{\left(\frac{342mg}{kg} \cdot \frac{2400Mg}{ha}\right) + \left(20yr \cdot \frac{1620mg}{kg} \cdot \frac{7Mg}{ha\text{-}yr}\right)}{\left(20yr \cdot \frac{7Mg}{ha\text{-}yr}\right) + \frac{2400Mg}{ha}}$$

$$= 410mg/kg$$

where:

342 = BS_j (background concentration of manganese in soil) from Exhibit 4-4,
2400 = MS (mass of soil in agricultural mixing zone), estimated above,

20 = N (total number of years sewage sludge is applied to agricultural land) from Exhibit 4-3,
1620 = C_j (manganese concentration in sewage sludge) from Exhibit 4-1, and
7 = AR (application rate of sewage sludge to agricultural land) from Exhibit 4-3.

Using Eq. 4-14, exposure of the tractor operator to manganese is then estimated as:

$$EXPT_{manganese} = 410\frac{mg}{kg} \cdot 10\frac{mg}{m^3} \cdot 10^{-6}\frac{kg}{mg}$$

$$= 4.1x10^{-3}\ mg/m^3$$

where:

410 = CT_j (concentration of manganese in agricultural soil), estimated above,
10 = TDA (total exposure of tractor operator to soil dust), discussed above, and
10^{-6} = constant to convert (kg) to (mg).

4.2.12 Pathway 12 - Ingestion of Fish and Water from Surface Water that Receives Eroded Soil

Pathway 12 evaluates human ingestion of fish and water from surface water that receives eroded soil from sewage sludge-amended agricultural and non-agricultural lands. Non-agricultural lands include forests, reclamation sites, and public contact sites.

To estimate exposure for this pathway, a mass balance analysis is required. This mass balance analysis accounts for the partitioning of pollutants into different soil phases (solids, air, and water) and the subsequent losses of pollutants from the land application site. Pollutants are lost from a land application site by: erosion of soil particles, which releases sorbed pollutants into surface waters; volatilization of pollutants into air; leaching of pollutants into groundwater; and degradation. A mass balance for a pollutant must be maintained, given these four competing loss processes of erosion, volatilization, leaching, and degradation. Once mass balances for pollutants have been established, exposures to pollutants that have eroded, volatilized, or leached are calculated under three separate pathways: surface water (Pathway 12), air (Pathway 13), and groundwater (Pathway 14). It is assumed that if pollutants degrade, they degrade into chemicals that do not pose unacceptable risks to public health or the environment.

The methods for performing the mass balance calculation for Pathways 12, 13, and 14 are discussed below. Subsequent to this discussion, the equations particular to Pathway 12, for estimating the maximum amount of pollutant available for erosion at a site and the transport of that pollutant mass to a surface water stream, are presented.

Method for Mass Balance (Pathways 12, 13, and 14)

There are two major steps involved in the mass balance calculation. First, a pollutant is partitioned among the three phases present in a soil: solids, air, and water. Second, the rates at which the four loss processes occur (erosion, volatilization, leaching, and degradation) are estimated. These two steps are described in the next two subsections.

Step 1: Partitioning of Pollutant Among Solids, Air, and Water in Sewage Sludge-Amended Soil. This section describes the methods for partitioning a pollutant in the sewage sludge-amended soil at a land application site, assuming that equilibrium is maintained among the pollutant concentration sorbed onto soil particles (which erodes into surface water), the pollutant concentration in the air-filled pore space (which volatilizes), and the pollutant concentration in the porewater (which leaches to groundwater).

Equilibrium partitioning between sorbed and dissolved phases is described by soil-water partition coefficients; partitioning between dissolved and gaseous phases is described by Henry's Law constants. From these assumptions and the definitions of concentrations in different phases presented below, equations are derived to describe pollutant partitioning among all of the phases.

Mathematically, pollutant concentrations in different phases can be expressed as:

$$C_s = \frac{M_{cs}}{M_s} \qquad C_a = \frac{M_{ca}}{V_a} \qquad C_w = \frac{M_{cw}}{V_w} \tag{4-15}$$

and:

$$C_t = \frac{M_{ct}}{V_t} = \frac{M_{cs} + M_{ca} + M_{cw}}{V_s + V_a + V_w} \tag{4-16}$$

where:

C_s = concentration of sorbed pollutant on sewage sludge-amended soil particles (kg pollutant/kg soil),
M_{cs} = mass of sorbed pollutant (kg),
M_s = mass of soil (kg),
V_s = volume of solids in soil (m^3),
C_a = concentration of gaseous pollutant in air-filled pore space of sewage sludge-amended soil (kg pollutant/m^3 air),
M_{ca} = mass of gaseous pollutant (kg),
V_a = volume of air in soil (m^3),
C_w = concentration of dissolved pollutant in water-filled pore space of sewage sludge-amended soil (kg pollutant/m^3 porewater),
M_{cw} = mass of dissolved pollutant (kg),
V_w = volume of water in soil (m^3),
C_t = total concentration of pollutant in bulk sewage sludge-amended soil (kg pollutant/m^3 total bulk soil volume),
M_{ct} = total mass of pollutant in soil (kg), and
V_t = total bulk volume of soil (m^3).

The definitions of equilibrium partition coefficients and soil characteristics, such as bulk density and porosity, are used in conjunction with Eqs. 4-15 and 4-16 to estimate the pollutant concentrations in each soil phase (solids, air, and water). The equilibrium partition or distribution coefficient (K_d), describing the partitioning of a pollutant between pollutant sorbed on solids and pollutant dissolved in porewater, can be defined as:

$$K_d = \frac{[M_{cs}/M_s]}{[M_{cw}/V_w]} \cdot 10^3 = \frac{M_{cs}\ V_w}{M_s\ M_{cw}} \cdot 10^3 \tag{4-17}$$

where:

K_d = soil-water partition coefficient (L water/kg soil), and
10^3 = constant to convert (m^3) to (L).

The dimensionless Henry's Law constant, which describes the partitioning of a pollutant between gaseous and dissolved phases, is defined as:

$$\dot{H} = \frac{[M_{ca}/V_a]}{[M_{cw}/V_w]} = \frac{M_{ca}\ V_w}{M_{cw}\ V_a} \qquad (4\text{-}18)$$

where:

$\dot{H}$ = Henry's Law constant (dimensionless).

The bulk density of soil is defined as:

$$BD_{mix} = M_s\ /\ V_t \qquad (4\text{-}19)$$

where:

BD_{mix} = bulk density of sewage sludge-amended soil (kg soil/m^3 total bulk soil volume).

The air-filled porosity of soil is defined as:

$$\theta_a = V_a\ /\ V_t \qquad (4\text{-}20)$$

where:

θ_a = air-filled porosity (dimensionless).

The water-filled porosity is defined as:

$$\theta_w = V_w\ /\ V_t \qquad (4\text{-}21)$$

where:

θ_w = water-filled porosity (dimensionless).

And, the total porosity of soil is defined as:

$$\theta_t = (V_t - V_s)\ /\ V_t = \theta_a + \theta_w \qquad (4\text{-}22)$$

where:

θ_t = total soil porosity (dimensionless).

By combining Eqs. 4-15 through 4-22, equations that describe the pollutant concentrations in the air and water phases in terms of the total pollutant concentration can be derived:

$$C_a = \frac{C_t}{\frac{K_d\ BD_{mix}}{\dot{H}} \cdot 10^{-3} + \frac{\theta_w}{\dot{H}} + \theta_a} \qquad (4\text{-}23)$$

and:

$$C_w = \frac{C_t}{BD_{mix}\ K_d \cdot 10^{-3} + \theta_w + \dot{H}\ \theta_a} \qquad (4\text{-}24)$$

Equations 4-23 and 4-24 are used to estimate the first-order rate constants for volatilization and leaching in the second step of the mass balance calculation.

It is important to note that in these derivations, C_t, the total pollutant concentration in sewage sludge-amended soil, is expressed as mass per volume. Recall that in Pathways 1 through 11, the total pollutant concentration is expressed as mass per mass of soil (CT_j). These two quantities can be related through the following equation:

$$C_t = CT_j \cdot BD_{mix} \cdot 10^{-6} \qquad (4\text{-}25)$$

where:

C_t	=	total concentration of pollutant in bulk sewage sludge-amended soil (kg pollutant/m^3 total bulk soil volume),
CT_j	=	concentration of pollutant j in sewage sludge-amended soil (mg pollutant/kg sewage sludge-amended soil), and
10^{-6}	=	constant to convert (mg) to (kg).

Step 2: Estimating Rates at Which Pollutants Are Lost From Sewage Sludge-Amended Soil. There are four major processes by which a pollutant can be lost from sewage sludge-amended soil: erosion of soil to which the pollutant is sorbed; volatilization of the pollutant; leaching of the pollutant to groundwater; and degradation of the pollutant. These four loss processes are all modeled as being first-order, i.e., the rate at which a pollutant erodes, volatilizes, leaches, or degrades is proportional to the remaining pollutant concentration in the soil. The loss rate coefficients for the four loss processes can be combined to yield a coefficient for the total rate at which the pollutant is lost from sewage sludge-amended soil:

$$K_{tot} = K_{ero} + K_{vol} + K_{lec} + K_{deg} \quad (4\text{-}26)$$

where:

K_{tot} = total loss rate for the pollutant from sewage sludge-amended land (yr^{-1}),
K_{ero} = loss rate due to erosion of the pollutant from sewage sludge-amended land (yr^{-1}),
K_{vol} = loss rate due to volatilization of the pollutant from sewage sludge-amended land (yr^{-1}),
K_{lec} = loss rate due to leaching of the pollutant from sewage sludge-amended land (yr^{-1}), and
K_{deg} = loss rate due to abiotic or microbial degradation of the pollutant on sewage sludge-amended land (yr^{-1}).

The ratio of each individual loss coefficient to the total then describes the fraction of pollutant loss caused by each individual process:

$$f_{ero} = \frac{K_{ero}}{K_{tot}} \qquad f_{vol} = \frac{K_{vol}}{K_{tot}}$$

$$f_{lec} = \frac{K_{lec}}{K_{tot}} \qquad f_{deg} = \frac{K_{deg}}{K_{tot}} \quad (4\text{-}27)$$

where:

f_{ero} = fraction of total pollutant loss caused by erosion (dimensionless),
f_{vol} = fraction of total pollutant loss caused by volatilization (dimensionless),
f_{lec} = fraction of total pollutant loss caused by leaching (dimensionless), and
f_{deg} = fraction of total pollutant loss caused by degradation (dimensionless).

These fractions are used with the pollutant concentrations in various soil phases to maintain a mass balance of pollutant. In the next four subsections, the calculations for estimating the loss rate coefficients for each individual loss process (erosion, volatilization, leaching, and degradation) are presented.

First-Order Loss Rate for Erosion. Pollutant losses to erosion are approximated by taking the ratio of the average rate of soil loss for the land application site to the depth to which sewage sludge is incorporated, assuming a pollutant is evenly incorporated into the soil. Therefore, the first-order loss rate coefficient for erosion is:

$$K_{ero} \approx \frac{d_e}{d \cdot 10^{-2}} \qquad (4\text{-}28)$$

where:

d_e = average rate of soil loss due to erosion from sewage sludge-amended land each year (m/yr),
d = depth of incorporation of sewage sludge (cm), and
10^{-2} = constant to convert (cm) to (m).

Note that this assumes that the loss to erosion includes pollutant mass in all three phases (solids, air, and water); therefore, the first-order loss coefficient is not pollutant-specific. That is, given the assumption of even incorporation, if one-tenth of the sewage sludge-soil mixture is removed by erosion, one-tenth of the mass of pollutant is also removed.

First-Order Loss Rate for Volatilization. Estimates of volatile emissions from uncovered soil are used in conjunction with estimates of pollutant concentrations in the air-filled pore space of soil to estimate the loss rate coefficient for volatilization, K_{vol}.

Estimates of volatile emissions are based on equations provided by Hwang and Falco (1986) for contaminated soil with no cover:

$$Na = \frac{2 t_e \theta_e D_{ei} C_a}{\sqrt{\pi \alpha_i t_e}} \qquad (4\text{-}29)$$

where:

Na = total average emissions from the soil surface over time interval t_e (kg pollutant/m^2 soil),
t_e = duration of emissions (sec),
θ_e = effective porosity of soil (dimensionless),
D_{ei} = intermediate diffusivity variable (defined in Eq. 4-30) (m^2/sec),
C_a = concentration of gaseous pollutant in air-filled pore space of sewage sludge-amended soil (kg pollutant/m^3 air), and
α_i = intermediate diffusivity variable (defined in Eq. 4-31) (m^2/sec).

In Hwang and Falco (1986), C_a is estimated from the concentration of sorbed pollutant (C_s). However, this analysis requires the relationship between the <u>total</u> concentration of pollutant in sewage sludge-amended soil (in sorbed, gaseous, and dissolved phases) and the concentration in gaseous phase within the soil's air-filled pore space. Therefore, Eq. 4-23 is used to estimate C_a.

The intermediate diffusivity variables required in Eq. 4-29 are obtained from the following relationships (Hwang and Falco, 1986):

$$D_{ei} = D_{ca}\, 10^{-4}\, \theta_e^{1/3} \qquad (4\text{-}30)$$

and:

$$\alpha_i = \frac{D_{ei}\, \theta_e}{\theta_e + \rho_{ss} \cdot (1-\theta_e) \cdot K_d \cdot 10^{-3}/\dot{H}} \qquad (4\text{-}31)$$

where:

$$\rho_{ss} = \frac{BD_{mix}}{(1-\theta_t)} \qquad (4\text{-}32)$$

and:

$$\dot{H} = \frac{H}{R \cdot T \cdot 10^{-3}} \qquad (4\text{-}33)$$

and where:

D_{ca}	=	the molecular diffusivity of pollutant in air (cm^2/sec),
10^{-4}	=	constant to convert units from (cm^2) to (m^2),
θ_e	=	effective porosity of soil (dimensionless),
ρ_{ss}	=	particle density of sewage sludge-soil mixture (kg/m^3),
K_d	=	soil-water partition coefficient (L water/kg soil),
$\dot{H}$	=	Henry's Law constant (dimensionless),
BD_{mix}	=	bulk density of sewage sludge-amended soil (kg soil/m^3 total bulk soil volume),
θ_t	=	total soil porosity (dimensionless),
H	=	Henry's Law constant (atm-m^3/mol),
R	=	gas constant (L-atm/mol-K),
T	=	temperature (Kelvin), and
10^{-3}	=	constant to convert (L) to (m^3).

Equation 4-29 provides an estimate of total average emissions from an uncovered layer of soil as a function of both time and the initial concentration of pollutant. For consistency with methods used to estimate losses for other pathways, Eq. 4-29 is evaluated for t_e equal to 1 year (t_e=3.2x10^7 sec), and results are used to estimate an annual loss coefficient. Losses predicted for the first year (Na_y) are divided by the total mass of pollutant in soil to estimate the approximate fraction of available pollutant lost per unit of time. For a unit concentration (1 kg/m^3) of the pollutant in soil (i.e., C_i), the mass of pollutant beneath one square meter of soil surface (in kg/m^2) is equal to the volume of treated soil beneath a square meter of surface (m^3

per m^2), that is, equal to the depth of incorporation (m). The estimated loss rate (in kg/m^2-yr) is approximated as a comparable first-order loss coefficient (in yr^{-1}) as:

$$K_{vol} \approx -\ln\left(1-\frac{Na_y}{d \cdot 10^{-2}}\right) \qquad (4\text{-}34)$$

where:

K_{vol} = loss rate due to volatilization (yr^{-1}),
Na_y = total average emissions from the soil surface in first year (kg pollutant/m^2), estimated using Eq. 4-29,
d = depth of incorporation of sewage sludge (cm. When converted to (m), equivalent to kg/m^2 for a unit concentration of pollutant in sewage sludge-amended soil), and
10^{-2} = constant to convert (cm) to (m).

Because Eq. 4-29 was derived by assuming the column of soil is of infinite depth, it can predict greater than 100 percent loss within a year for a relatively shallow layer of treated soil and a relatively volatile pollutant. For such cases, Eq. 4-34 cannot be evaluated and the loss rate coefficient is instead estimated from predicted emissions in the first second (t_e=1 sec). The loss rate coefficient estimated from the first second of emissions is then converted to an annual loss rate coefficient:

$$K_{vol} \approx -3.2\times10^{7} \ln\left(1-\frac{Na_s}{d \cdot 10^{-2}}\right) \qquad (4\text{-}35)$$

where:

Na_s = emissions from the soil surface in first second (kg pollutant/m^2), estimated using Eq. 4-29,
$3.2\text{x}10^{7}$ = constant to convert (sec^{-1}) to (yr^{-1}), and
10^{-2} = constant to convert (cm) to (m).

Equation 4-34 was used to evaluate carbon disulfide, dioxins and dibenzofurans, endosulfan, pentachloronitrobenzene, PCBs, and 2-(2,4,5-trichlorophenoxy) propionic acid. However, for the other Round Two pollutants that volatilize, Eq. 4-35 was used.

First-Order Loss Rate for Leaching. To estimate pollutant loss to leaching, the following equation, which computes a first-order loss rate for a pollutant leaching from treated soil (U.S. EPA, 1989f), was modified to take into account that leaching is only one of four competing pollutant loss processes:

$$K_{lec} = \frac{NR}{BD_{mix} \cdot K_d \cdot 10^{-3} \cdot d \cdot 10^{-2}} \quad (4\text{-}36)$$

where:

K_{lec} = loss rate due to leaching of the pollutant from sewage sludge-amended land (yr^{-1}),
NR = annual recharge to groundwater beneath the treated soil (m^3 recharge/m^2 area-yr, or m recharge/yr),
BD_{mix} = bulk density of sewage sludge-amended soil (kg soil/m^3 total bulk soil volume),
K_d = soil-water partition coefficient (L water/kg soil),
d = depth of incorporation of sewage sludge (cm),
10^{-3} = constant to convert (L) to (m^3), and
10^{-2} = constant to convert (cm) to (m).

To derive a coefficient for first-order loss to leaching while maintaining a mass balance, the mass of pollutant expected to be lost each year is estimated and divided by the available mass of pollutant. The mass of pollutant that will be lost to leaching in any interval of time per unit area (i.e., the flux of pollutant) can be described by the volume of water percolating through the treated soil multiplied by the average concentration of pollutant in that water:

$$FA_{lec} = NR \cdot C_{lec} \cdot 10{,}000 \quad (4\text{-}37)$$

where:

FA_{lec} = annual average flux of pollutant leached from sewage sludge-amended soil (kg pollutant/ha-yr),
C_{lec} = concentration of pollutant in water leaching from sewage sludge-amended soil (kg pollutant/m^3 porewater), and
10,000 = constant to convert units from (kg/m^2-yr) to (kg/ha-yr).

Assuming that all the porewater forms leachate, Eq. 4-24 can be used to estimate the pollutant concentration in the leachate:

$$C_w = C_{lec} = \frac{C_t}{\left[BD_{mix} \cdot K_d \cdot 10^{-3} + \theta_w + H\,\theta_a\right]} \quad (4\text{-}38)$$

where:

C_w = concentration of dissolved pollutant in water-filled pore space of sewage sludge-amended soil (kg pollutant/m^3 porewater),
C_t = total concentration of pollutant in bulk sewage sludge-amended soil (kg pollutant/m^3 total bulk soil volume),
θ_w = water-filled porosity (dimensionless),

H	=	Henry's Law constant (dimensionless),
θ_a	=	air-filled porosity (dimensionless), and
10^{-3}	=	constant to convert (L) to (m^3).

Given that first-order loss rates are being assumed, the total concentration of pollutant in soil decreases due to leaching according to the following equation:

$$\frac{dC_t}{dt} = -K_{lec} C_t \qquad (4\text{-}39)$$

K_{lec} can be estimated with the discrete approximation:

$$K_{lec} = \frac{\left[\frac{dC_t}{dt}\right]}{C_t} \approx \frac{\left[\frac{\Delta C_t}{\Delta t}\right]}{C_t} \approx \frac{\left[\frac{\Delta M_{ct}}{\Delta t}\right]}{M_{ct}} \qquad (4\text{-}40)$$

where:

t	=	time (yr).

The change in the total pollutant mass in the soil with time can be expressed in terms of the pollutant flux leaching to the groundwater:

$$\frac{\Delta M_{ct}}{\Delta t} = FA_{lec} \cdot A \cdot 10^{-4} \qquad (4\text{-}41)$$

where:

$\Delta M_{ct}/\Delta t$	=	change in total mass of pollutant in soil over time interval of one year (kg/yr),
A	=	area of land application site (m^2), and
10^{-4}	=	constant to convert units from (m^2) to (ha).

Combining Eqs. 4-40 and 4-41:

$$K_{lec} \approx \frac{FA_{lec} \cdot A \cdot 10^{-4}}{M_{ct}} \qquad (4\text{-}42)$$

Given that the total bulk volume of sewage sludge-amended soil equals the area of the land application site multiplied by the depth of incorporation:

$$V_t = A \cdot d \cdot 10^{-2}$$

where:

V_t = total bulk volume of soil (m^3),

and rearranging Eq. 4-16:

$$C_t \cdot V_t = M_{ct}$$

the following equation can be written:

$$\frac{A}{M_{ct}} = \frac{1}{C_t\, d \cdot 10^{-2}} \qquad (4\text{-}43)$$

Using Eqs. 4-37, 4-38, and 4-43, Eq. 4-42 can be rewritten as:

$$K_{lec} = \frac{NR\ C_{lec}}{C_t\, d \cdot 10^{-2}} = \frac{NR}{\left[BD_{mix} \cdot K_d + \theta_w + \dot{H}\ \theta_a\right] d \cdot 10^{-2}} \qquad (4\text{-}44)$$

Equation 4-44 is used to predict the rate of pollutant loss to leaching.

First-Order Loss Rate for Degradation. Values of K_{deg} are obtained from the literature for each pollutant. For K_{deg}, rates of abiotic reactions such as hydrolysis are preferentially used because they can be more reliably measured; for pollutants only degraded by microbes, the lowest aerobic rates measured under environmental conditions are used. For inorganics and some persistent organic pollutants such as dioxins and dibenzofurans, K_{deg} is zero.

Methods Specific to Pathway 12

Overview of Methods Specific to Pathway 12. To calculate the average concentration of a pollutant in eroding soil, estimates of the maximum mass of pollutant available for erosion from a land application site are combined with estimates of the rate at which the pollutant is lost from the site due to the four loss processes over a human lifetime. The average concentration of pollutant in eroded, sewage sludge-amended soil is then "diluted" by the erosion of clean soil from the remainder of the watershed. Estimates of the concentration of pollutants in the stream receiving the eroded soil are then made, derived from the mass of pollutant on eroded soil and

on soil-water partition coefficients. Pollutants then partition into fish inhabiting the stream. Humans are exposed to pollutants through both direct ingestion of surface water and ingestion of fish. For this analysis, organics as well as inorganics are allowed to build up in the soil over the active life of the land application site.

Maximum Pollutant Mass Available for Erosion. The maximum mass of pollutant available for erosion at a land application site occurs after the final application of sewage sludge to the site. This maximum mass is estimated as:

$$TP_N = AR \cdot C_j \cdot \left(1 + e^{-c \cdot K_{tot}} + e^{-2c \cdot K_{tot}} + \ldots + e^{-b \cdot c \cdot K_{tot}}\right) \cdot 1000 \cdot 1 \qquad (4\text{-}45)$$

where:

TP_N = total mass of pollutant available at a site after the final year of application (mg pollutant/ha),
AR = annual whole sludge application rate of sewage sludge to land (dry Mg sewage sludge/ha-yr),
C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge),
c = application interval (number of years between applications),
K_{tot} = total loss rate for the pollutant from sewage sludge-amended land (yr^{-1}),
b = model parameter that is equal to the integer part of $((N_{site}-1)/c)$, where N_{site} is the site life (yr),
1000 = constant to convert (Mg) to (kg), and
1 = interval of time over which pollutant loss is evaluated (yr).

Losses to Erosion Averaged Over a Human Lifetime. Because human exposure is assumed to continue for the duration of an individual's lifetime, concentrations of pollutants in surface water are calculated based on losses of pollutant through surface erosion for a period equal to the human life expectancy. Therefore, loss to erosion during the period between the end of land application and the end of an individual's lifetime must be estimated. To do so, the mass of pollutant left at the end of an individual lifetime is first calculated. After the last application of sewage sludge to a site, a pollutant continues to be depleted according to the following equation:

$$M_{LS} = TP_N \cdot e^{-K_{tot}(LS - N_{site})} \qquad (4\text{-}46)$$

where:

M_{LS} = mass of pollutant in soil at end of a period equal to an individual lifetime (mg pollutant/ha),
LS = average human lifetime (yr), and
N_{site} = site life (yr).

The fraction of total, cumulative loading lost to all four loss processes in the human lifetime (over both the application and post-application periods) can then be calculated as:

$$f_{LS} = \frac{(N \cdot C_j \cdot AR \cdot 1000) - M_{LS}}{N \cdot C_j \cdot AR \cdot 1000} \quad (4\text{-}47)$$

where:

f_{LS} = fraction of total cumulative loading lost in individual's lifetime to all four loss processes (dimensionless).

This fraction is used to estimate the average pollutant concentration in eroded soil for both organic and inorganic pollutants. The estimated total loading of pollutant is multiplied by the fraction expected to be lost to erosion in the human lifetime ($f_{ero} \cdot f_{LS}$), and divided by the total mass of eroded soil lost during that period to calculate the expected average concentration of each pollutant in eroded soil:

$$C_{site,\,j} = \frac{N \cdot C_j \cdot AR \cdot 1000 \cdot f_{ero} \cdot f_{LS}}{ME_{site} \cdot LS} \quad (4\text{-}48)$$

where:

$C_{site,\,j}$ = the concentration of pollutant j in sewage sludge-amended soil eroded from the land application site (mg pollutant/kg sewage sludge-amended soil),

ME_{site} = rate of soil loss for land treated with sewage sludge (kg sewage sludge-amended soil/ha-yr), and

$$ME_{site} = d_e \cdot BD_{mix} \cdot 10{,}000 \quad (4\text{-}49)$$

where:

d_e = average rate of soil loss due to erosion from sewage sludge-amended land each year (m/yr),

BD_{mix} = bulk density of sewage sludge-amended soil (kg soil/m^3 total bulk soil volume), and

10,000 = constant to convert (m^2) to (ha).

Note that Eq. 4-48 assumes that the same mass of pollutant leaves the site by erosion every year.

Dilution of Eroded, Sewage Sludge-Amended Soil with Non-Sewage Sludge-Amended Soil. The extent to which eroded soil from the land application site is "diluted" by soil from the untreated remainder of the watershed also needs to be estimated. A "dilution factor"

describes the fraction of the total eroded soil in the watershed originating in the land application site:

$$D_f = \frac{A_{site}\ ME_{site}\ S_{site}}{\left[A_{site}\ ME_{site}\ S_{site}\right] + \left[(A_{ws} - A_{site})ME_{ws}\right]\ S_{ws}} \qquad (4\text{-}50)$$

where:

D_f	=	dilution factor (dimensionless),
A_{site}	=	area of land application site (ha),
A_{ws}	=	area of the watershed (ha),
S_{site}	=	sediment delivery ratio for the land application site (dimensionless),
S_{ws}	=	sediment delivery ratio for the watershed (dimensionless), and
ME_{ws}	=	estimated rate of soil loss (erosion) for the watershed (kg soil/ha-yr).

If the rates of soil erosion from the land application site and the remainder of the watershed are assumed to be the same, ME_{site} and ME_{ws} cancel from Eq. 4-50, and the dilution factor can be calculated by:

$$D_f = \frac{A_{site}\ S_{site}}{A_{site}\ S_{site} + \left[(A_{ws} - A_{site})S_{ws}\right]} \qquad (4\text{-}51)$$

The sediment delivery ratios for the land application site and the watershed are calculated with the following empirical relationship (Vanoni, 1975):

$$S = 0.872\ A^{-0.125}$$

Thus, the sediment delivery ratio for the site and for the watershed are:

$$S_{site} = 0.872\ [A_{site}]^{-0.125} \qquad (4\text{-}52)$$

and

$$S_{ws} = 0.872\ [A_{ws}]^{-0.125} \qquad (4\text{-}53)$$

If all of a pollutant entering the stream on eroded soil is assumed to originate from the land application site, the dilution factor, D_f, also describes the ratio between the average concentration of pollutant in soil entering the stream and the average concentration in soil eroding from the land application site:

$$C_{esoil,\,j} = D_f \cdot C_{site,\,j} \qquad (4\text{-}54)$$

where:

$C_{esoil,j}$ = dry weight concentration of pollutant j in eroded soil entering the stream (mg pollutant/kg eroded soil).

Pollutant Concentration in the Stream. The estimated concentration of pollutant in the eroded soil is used as an input to calculate the expected concentration of pollutant in the stream. Once the eroded soil enters the stream, the pollutant is assumed to partition between the solid and liquid phases of the stream according to equilibrium conditions. The total amount of pollutant available to partition depends on the amount of eroded soil in the stream. Assuming that all of the total suspended solids in the stream are from eroded soil particles, Eq. 4-55 partitions the total mass of pollutant between dissolved and sorbed phases. The lefthand term represents the total mass of pollutant (per liter of water) entering the stream in eroded soil; the middle term represents the dissolved pollutant; and, the right-most term represents the mass of sorbed pollutant (per liter of water):

$$C_{esoil,j} \cdot TSS \cdot 10^{-6} = C_{sw,j} + \left(C_{sw,j} \cdot K_d \cdot TSS \cdot 10^{-6}\right) \qquad (4\text{-}55)$$

where:

TSS = concentration of total suspended solids in the stream (mg solids/L water),
$C_{sw,j}$ = concentration of pollutant j in surface water (mg pollutant/L water),
K_d = soil-water partition coefficient (L water/kg soil), and
10^{-6} = constant to convert (mg) to (kg).

Solving Eq. 4-55 for the concentration of pollutant dissolved in the stream:

$$C_{sw,j} = \frac{C_{esoil,j} \cdot TSS \cdot 10^{-6}}{1 + K_d \cdot TSS \cdot 10^{-6}} \qquad (4\text{-}56)$$

Partition Coefficients for Organic Pollutants. For organic pollutants, a partition coefficient between water and suspended solids (K_d) is estimated from each pollutant's organic carbon-water partition coefficient (K_{oc}) and an assumption about the fraction of organic carbon (f_{oc}) on suspended solids in streams:

$$K_d = K_{oc} \cdot f_{oc} \quad (4\text{-}57)$$

where:

K_{oc} = organic carbon-water partition coefficient (mL water/g organic carbon), and

f_{oc} = fraction of organic carbon in suspended solids (dimensionless, g organic carbon/g suspended solids).

A value of 0.01 is used for the f_{oc} of suspended solids, to correspond to the f_{oc} of the mixing zone from which the suspended solids are assumed to have eroded (U.S. EPA, 1993a).

To estimate an organic pollutant's K_{oc} value, an empirical regression equation presented by Hasset et al. (1983) is used:

$$\log_{10}(K_{oc}) = 0.088 + 0.909 \log_{10}(K_{ow}) \quad (4\text{-}58)$$

where:

K_{ow} = octanol-water partition coefficient (dimensionless, mg pollutant/L octanol per mg pollutant/L water).

Partition Coefficients for Inorganic Pollutants. For inorganic pollutants, it is much more difficult to predict a "typical" K_d value. Metals can sorb onto soils through the processes of ion exchange, specific adsorption, co-precipitation with hydrous oxides, and incorporation into cationic lattice sites in crystalline sediments (Bodek et al., 1988). Therefore, clay minerals, organic matter, and manganese and iron oxides are all important sorbents of metals in soil (Bodek et al., 1988). The pH of the system also affects metal sorption, with most metals tending to sorb more at higher pHs. Therefore, measured K_d values reported in the literature for conditions similar to those being modeled are used.

Estimating Human Exposure

Humans can be exposed to surface water contaminated by sewage sludge through two pathways: ingestion of water and ingestion of fish. Potential exposure through ingestion of contaminated surface water is calculated as:

$$EXP_{w,j} = \frac{C_{sw,j} \cdot IW}{BW} \quad (4\text{-}59)$$

where:

$EXP_{w,j}$ = exposure to pollutant j through direct ingestion of surface water (mg pollutant/kg body weight-day),

$C_{sw,j}$ = concentration of pollutant j in surface water (mg pollutant/L water),
IW = quantity of water ingested daily (L water/day), and
BW = body weight (kg), assumed to be 70 kg.

Exposure through ingestion of fish is calculated based on estimates of the bioaccumulation of a pollutant in fish and the assumed rate of fish ingestion. Bioaccumulation is the process by which aquatic organisms accumulate pollutants, from both water and food, at concentrations higher than the ambient concentration. The process by which a pollutant is absorbed from water through gill membranes or other external body surfaces is called bioconcentration, and the measure of a pollutant's tendency to bioconcentrate is described by the bioconcentration factor. For organic pollutants, a regression equation based on $\log(K_{ow})$ values is used to estimate bioconcentration factors (*BCF*s). The equation was developed for a three percent lipid content of fish (U.S. EPA, 1990):

$$\log_{10}(BCF) = 0.79 \log_{10}(K_{ow}) - 0.80 \quad (4\text{-}60)$$

where:
BCF = pollutant-specific bioconcentration factor (L water/kg fish).

For inorganic pollutants, available literature values were used for *BCF*s.

Biomagnification denotes the process by which the concentration of a pollutant increases in different organisms occupying successive trophic levels. The combined accumulation from bioconcentration and biomagnification is represented by the bioaccumulation factor, which is calculated as the product of the bioconcentration factor and a food chain multiplier:

$$BAF = BCF \cdot FM \quad (4\text{-}61)$$

where:
BAF = pollutant-specific bioaccumulation factor (L water/kg fish), and
FM = pollutant-specific food chain multiplier (dimensionless).

Assuming that humans only ingest fish fillets, and not the whole fish, the pollutant concentration in fish fillets can be expressed as:

$$C_{ff,j} = C_{sw,j} \cdot BAF \cdot P_f \quad (4\text{-}62)$$

where:
$C_{ff,j}$ = concentration of pollutant j in fish fillets (mg pollutant/kg fish fillet), and

P_f = ratio of pollutant concentration in fillet to whole fish (dimensionless).

Thus, human exposure through ingestion of contaminated fish can be expressed as:

$$EXP_{f,j} = \frac{C_{ff,j} \cdot IF}{BW} \quad (4\text{-}63)$$

where:

$EXP_{f,j}$ = exposure to pollutant j through ingestion of fish (mg pollutant/kg body weight-day), and

IF = daily consumption of fish fillets (kg fish fillets/day).

For this analysis of the surface water pathway, exposures through drinking water consumption and fish ingestion are combined:

$$EXP_j = EXP_{w,j} + EXP_{f,j} \quad (4\text{-}64)$$

where:

EXP_j = exposure to pollutant j through consumption of both surface water and fish combined (mg pollutant/kg body weight-day).

Data Inputs

Both non-pollutant-specific and pollutant-specific inputs are required for the exposure equations described above. Values for non-pollutant-specific inputs, such as the area of a watershed, its hydrogeological characteristics, and the daily consumption of fish and drinking water, are presented in Exhibit 4-16. Note that all parameters necessary for Pathways 12, 13, and 14 are presented in this Exhibit.

There are several pollutant-specific fate and transport parameters required to maintain the mass balance of a pollutant among the four loss processes and to estimate the rates at that those four loss processes occur. In Exhibit 4-17, all of the fate and transport parameters are presented.

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

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

In the absence of pollutant-specific data for the ratio of pollutant concentration in fillet to the concentration in whole fish, it is assumed that these concentrations are the same ($P_f = 1$) for all pollutants except PCBs and dioxins. PCBs are assumed to behave similarly to dioxins, for which a ratio of 0.5 has been estimated (Branson et al., 1985).

For *BCF* values for inorganic pollutants, the Ambient Aquatic Life Water Quality Criteria documents were reviewed. Only three Round Two inorganics had such documents available: aluminum, antimony, and silver. For aluminum, bioconcentration factors for young brook trout were reported to range from 50 to 231 (U.S. EPA, 1988a). The geometric mean (107) was used in this analysis. For antimony, one study on bioconcentration in bluegill found no significant accumulation above controls (U.S. EPA, 1988b). AQUIRE (Aquatic Toxicity Information Retrieval) was then searched for *BCF* data on antimony. The AQUIRE run turned up values for one saltwater fish and one fish that may or may not be saltwater. The *BCF*s for the shanny (*Blennius pholis*) and for the two-spot goby (*Gobiusculus flavescens*) are 0.40 and 0.15, respectively. The geometric mean of these two values (0.24) was used in this analysis. For silver, the Ambient Aquatic Life Water Quality Criteria for Silver document (U.S. EPA, 1987) had *BCF* information for two freshwater fish species, bass and bluegill. The geometric mean of the bass *BCF*s (11 and 19) is 14; the geometric mean of the bluegill *BCF*s (15 and 150) is 47. The geometric mean of the two species' *BCF*s is 26; this value was used in this analysis.

AQUIRE was then searched for all the remaining inorganics (Ba, Be, B, F, Mn, Th, Sn, Ti, and V). No data on *BCF* values for freshwater fish were found.

For *FM* values, if an organic pollutant had a $\log(K_{ow})$ value less than or equal to five, a value of one was used; otherwise a value of ten was used for *FM* (U.S. EPA, 1990). This relationship is applicable to a species on a trophic level of three. For inorganic pollutants, an *FM* value of one was used.

EXHIBIT 4-16
Non-Pollutant-Specific Parameters for Pathways 12, 13, and 14

Parameter	Definition	Value	Reference
θ_t	total soil porosity	0.4 (dimensionless)	U.S. EPA, 1992a
θ_e	effective soil porosity	0.4 (dimensionless)	U.S. EPA, 1992a
θ_a	air-filled porosity	0.2 (dimensionless)	U.S. EPA, 1992a
θ_w	water-filled porosity	0.2 (dimensionless)	U.S. EPA, 1992a
A_{site}	area of land treated with sewage sludge	1,074 ha	U.S. EPA, 1992a
A_{ws}	area of watershed	440,300 ha	U.S. EPA, 1992a
c	application interval	varies	See N in Exhibit 4-3
d_e	yearly depth of soil eroded	6×10^{-4} m/yr	USDA, 1987
NR	annual recharge to groundwater	0.5 m/yr	U.S. EPA, 1992a
ρ_w	density of water	1 kg/L	approximate density of water under environmental conditions
TSS	total suspended solids in surface water	16 mg/L	U.S. EPA, 1992a
θ	angle subtended by the land application site's width	22.5°	U.S. EPA, 1992a
u	average wind velocity	4.5 m/sec	U.S. EPA, 1992a
v	vertical dispersion of pollutant in air	1 (dimensionless)	U.S. EPA, 1992a
T	average air temperature	288 Kelvin	U.S. EPA, 1992a
IF	daily consumption of fish fillets	0.04 kg/day	U.S. EPA, 1992a
IA	daily inhalation rate	20 m^3/day	U.S. EPA, 1992a
IW	daily ingestion of water	2 L/day	U.S. EPA, 1992a
N_{site}	site life	20 yr ag., forest, pub., 1 yr recl.	
R	gas constant	0.082 L-atm/mol-Kelvin	constant

EXHIBIT 4-17
Environmental Fate and Transport Parameters

Pollutant	log K_{ow}	K_d (L/kg)	Henry's Law Constant[1] (atm-m^3/mol)	K_{deg} (yr^{-1})[2]	Diffusivity in Air (D_{ca})[1] (cm^2/sec)	Diffusivity in Water (D_{cw})[1] (cm^2/sec)	BCF (L/kg)	FM
Acetic acid (2,4-dichlorophenoxy)	2.81[3]	4.4[4]	1.4×10^{-10} [5]	36[6]	5.5×10^{-2}	4.9×10^{-6}	26[7]	1
Aluminum	NA	4[8]	NA	NA	NA	NA	79[9]	1
Antimony	NA	6[10]	NA	NA	NA	NA	0.25[11]	1
Barium	NA	6[10]	NA	NA	NA	NA		1
Beryllium	NA	43[10]	NA	NA	NA	NA		1
Bis(2-ethylhexyl) phthalate	5.11[12]	540[4]	8.8×10^{-6} [13]	11[14]	3.4×10^{-2}	2.9×10^{-6}	1700[7]	10
Boron	NA		NA	NA	NA	NA		1
Butanone, 2-	0.29[16]	0.022[4]	5.8×10^{-5} [17]	25[18]	9.2×10^{-2}	7.8×10^{-6}	0.27[7]	1
Carbon disulfide	2.00[19]	0.81[4]	1.4×10^{-2} [20]	0.006[21]	1.1×10^{-1}	9.7×10^{-6}	6.0[7]	1
Cresol, p-	1.94[3]	0.71[4]	7.9×10^{-7} [22]	380[14]	7.4×10^{-2}	6.6×10^{-6}	5.4[7]	1
Cyanides (soluble salts and complexes)	-1.04[23]	0.0014[4]	1.2×10^{-4} [24]	13[24]	1.9×10^{-1}	1.8×10^{-5}	0.02[7]	1
Dioxins and Dibenzofurans	6.64[25]	13,000[4]	6.8×10^{-5} [25]	0[25]	4.4×10^{-2}	4.1×10^{-6}	27,900[7]	10

EXHIBIT 4-17
Environmental Fate and Transport Parameters (cont'd.)

Pollutant	log K_{ow}	K_d (L/kg)	Henry's Law Constant[1] (atm-m^3/mol)	K_{deg} (yr^{-1})[2]	Diffusivity in Air (D_{ca})[1] (cm^2/sec)	Diffusivity in Water (D_{cw})[1] (cm^2/sec)	BCF (L/kg)	FM
Endosulfan-II	3.52[26]	19[4]	1.9×10^{-5} [27]	6.7[28]	4.3×10^{-2}	3.6×10^{-6}	96[7]	1
Fluoride	NA	36[8]	NA	NA	NA	NA		1
Manganese	NA	0.9[10]	NA	NA	NA	NA		1
Methylene chloride	1.3[29]	0.19[4]	2.0×10^{-3} [27]	2.6[30]	1.0×10^{-1}	9.2×10^{-6}	1.7[7]	1
Pentachloronitrobenzene	4.64[3]	200[4]	9.9×10^{-5} [31]	1.1[32]	5.2×10^{-2}	4.9×10^{-6}	730[7]	1
Phenol	1.46[33]	0.26[4]	4.0×10^{-7} [22]	72[34]	8.2×10^{-2}	7.4×10^{-6}	2.3[7]	1
Polychlorinated biphenyls (coplanar)	6.7[35]	15,000[4]	2.0×10^{-4} [36]	0.063[37]	4.5×10^{-2}	4.1×10^{-6}	31,000[7]	10
Propanone, 2-	-0.24[3]	0.0074[4]	3.7×10^{-5} [38]	36[14]	1.1×10^{-1}	9.1×10^{-6}	0.10[7]	1
Propionic acid, 2-(2,4,5-trichlorophenoxy)	3.41[3]	15[4]	1.3×10^{-8} [39]	17[40]	4.9×10^{-2}	4.4×10^{-6}	78[7]	1
Silver	NA	290[10]	NA	NA	NA	NA	25[41]	1
Thallium	NA		NA	NA	NA	NA		1
Tin	NA		NA	NA	NA	NA		1
Titanium	NA		NA	NA	NA	NA		1

EXHIBIT 4-17
Environmental Fate and Transport Parameters (cont'd.)

Pollutant	log K_{ow}	K_d (L/kg)	Henry's Law Constant[1] (atm-m^3/mol)	K_{deg} (yr^{-1})[2]	Diffusivity in Air (D_{ca})[1] (cm^2/sec)	Diffusivity in Water (D_{cw})[1] (cm^2/sec)	BCF (L/kg)	FM
Toluene	2.79[27]	4.2[4]	6.7×10^{-3} [27]	120[42]	8.0×10^{-2}	6.9×10^{-6}	25[7]	1
Trichlorophenoxy acetic acid, 2,4,5-	3.13[3]	8.6[4]	9.4×10^{-11} [43]	13[15]	5.4×10^{-2}	4.6×10^{-6}	47[7]	1
Vanadium	NA	39[10]	NA	NA	NA	NA		1

Notes:
NA means Not Applicable
[1] Discussed in Sections 4.2.13 and 4.2.14.
[2] Aerobic degradation rate.
[3] Hansch and Leo (1985a).
[4] Estimated using Eqs. 4-57 and 4-58.
[5] Jury et al. (1987).
[6] Ou (1984).
[7] Estimated using Eq. 4-60.
[8] Based on Langmuir isotherm data in Bodek et al. (1988).
[9] U.S. EPA (1988a).
[10] Low value from K_d range measured for that inorganic in sand in Gerritse et al. (1982). See text for details.
[11] U.S. EPA (1988b).
[12] CHEMEST procedure of GEMS, U.S. EPA (1989c).
[13] Howard et al. (1985); Montgomery and Welkom (1990).
[14] Howard et al. (1991).
[15] Smith (1978, 1979); McCall et al. (1981); Alton and Stritzke (1973).
[16] Hansch and Leo (1981).
[17] Rathburn and Tai (1987).
[18] Dojlido (1979); Price et al. (1974); Bridie et al. (1979); Dose et al. (1975); Heukelekian and Rand (1955).
[19] Verschueren (1983).

[20] Elliot (1989).
[21] Peyton et al. (1976).
[22] Hine and Mookerjee (1975).
[23] Estimated from Lyman et al. (1990) for hydrogen cyanide.
[24] Bodek et al. (1988).
[25] U.S. EPA (1994b).
[26] OHM/TADS (1989).
[27] Mabey et al. (1982).
[28] Greve and Wit (1971).
[29] Hansch and Leo (1979).
[30] Davis and Madsen (1991).
[31] Calculated from water solubility (Eckert, 1962) and vapor pressure (Gile and Gillett, 1979).
[32] Wang and Broadbent (1972).
[33] Hansch and Leo (1985b).
[34] Baker and Mayfield (1980).
[35] Based on Aroclor 1254 from O'Connor (1992) and Anderson and Parker (1990).
[36] Murphy et al. (1987).
[37] Fries (1982).
[38] Snider and Dawson (1985).
[39] Howard, 1991.
[40] Smith (1978); Alton and Stritzke (1973).
[41] U.S. EPA (1987).
[42] Wilson et al. (1983).
[43] Based on water solubility (Klopffer et al., 1982) and vapor pressure (Que Hee et al., 1981).

Example Exposure Calculations for Pathway 12

The following example calculates exposure of humans to dioxins and dibenzofurans through ingestion of water and fish from surface water receiving eroded sewage sludge-amended soil from agricultural land.

Step 1: Partitioning of Pollutant

In Step 1 of the mass balance calculation, relationships among pollutant concentrations in the bulk sewage sludge-amended soil (C_t), in the air-filled pore space (C_a), and in the water-filled pore space (C_w) are derived. In this example calculation, these relationships are used to estimate K_{vol} and K_{lec} in Step 2.

Step 2: Estimation of K_{ero}

Equation 4-28 can be used to calculate the loss rate coefficient for erosion (K_{ero}):

$$K_{ero} = \frac{6x10^{-4}\ m/yr}{15\ cm \cdot 10^{-2}\ m/cm} = 4x10^{-3}\ yr^{-1}$$

where:

$6x10^{-4}$ = d_e (average rate of soil loss due to erosion from sewage sludge-amended land each year) from Exhibit 4-16,
15 = d (depth of incorporation for sewage sludge on agricultural land) from Exhibit 4-3, and
10^{-2} = constant to convert (cm) to (m).

Step 2: Estimation of K_{vol}

Several equations are used to calculate the loss rate coefficient for volatilization (K_{vol}). First, the intermediate diffusivity variable D_{ei} is calculated from Eq. 4-30:

$$D_{ei} = 4.4x10^{-2}\ cm^2/sec \cdot 10^{-4}\ m/cm^2 \cdot (0.4)^{1/3} = 3.2x10^{-6}\ m^2/sec$$

where:

$4.4x10^{-2}$ = D_{ca} (diffusivity of pollutant in air) from Exhibit 4-17,
10^{-4} = factor to convert (cm^2) to (m^2), and
0.4 = θ_e (effective porosity of soil) from Exhibit 4-16.

Second, the particle density of the sewage sludge-soil mixture (ρ_{ss}) is calculated using Eq. 4-32:

$$\rho_{ss} = \frac{1400\ kg/m^3}{(1 - 0.4)} = 2.3x10^3 kg/m^3$$

where:

1400 = BD_{mix} (bulk density of sewage sludge-amended soil) from Exhibit 4-3, and
0.4 = θ_t (total porosity of soil) from Exhibit 4-16.

Third, the dimensionless Henry's Law constant ($\dot{H}$) is calculated using Eq. 4-33:

$$\dot{H} = \frac{6.8x10^{-5} atm\text{-}m^3/mol}{(0.082\ L\text{-}atm/mol\text{-}K) \cdot 288K \cdot 0.001\ m^3/L} = 2.9x10^{-3}$$

where:

$6.8x10^{-5}$ = H (Henry's Law constant) from Exhibit 4-17,
0.082 = R (universal gas constant) from Exhibit 4-16,
288 = T (average air temperature in Kelvin) from Exhibit 4-16, and
0.001 = factor to convert (L) to (m^3).

Fourth, the intermediate diffusivity variable α_i is calculated from Eq. 4-31:

$$\alpha_i = \frac{3.2x10^{-6}\ m^2/sec \cdot 0.4}{0.4 + \frac{2.3x10^3\ kg/m^3 \cdot (1\text{-}0.4) \cdot 13{,}000\ L/kg}{2.9x10^{-3} \cdot 1000\ L/m^3}} = 2.0x10^{-13} m^2/sec$$

where:

$3.2x10^{-6}$ = D_{ei} (calculated above),
0.4 = θ_e (effective porosity of soil) from Exhibit 4-16,
$2.3x10^3$ = ρ_{ss} (calculated above),
13,000 = K_d (partition coefficient for dioxins between water and soil) from Exhibit 4-17,
$2.9x10^{-3}$ = $\dot{H}$ (calculated above), and
1000 = constant to convert (m^3) to (L).

Next, to calculate C_a, Eq. 4-23 is used with a unit concentration of 1kg/m^3 for C_t:

$$C_a = \frac{1\ kg/m^3}{\frac{[13{,}000\ L/kg \cdot 1400\ kg/m^3 \cdot 0.001\ m^3/L]}{2.9x10^{-3}} + \frac{0.2}{2.9x10^{-3}} + 0.2}$$

$$= 1.6x10^{-7}\ kg/m^3$$

where:

1	=	C_t (unit concentration of pollutant in bulk sewage sludge-amended soil),
13000	=	K_d (soil-water partition coefficient) from Exhibit 4-17,
1400	=	BD_{mix} (bulk density of sewage sludge-amended soil) from Exhibit 4-3,
0.001	=	constant to convert (L) to (m^3),
2.9x10^{-3}	=	H (dimensionless Henry's Law constant) calculated above,
0.2	=	θ_w (water-filled porosity) from Exhibit 4-16, and
0.2	=	θ_a (air-filled porosity) from Exhibit 4-16.

Total average emissions from the soil surface in one year are then calculated using Eq. 4-29:

$$Na_y = \frac{2 \cdot 31{,}536{,}000\ sec \cdot 0.4 \cdot 3.2x10^{-6}\ m^2/sec \cdot 1.6x10^{-7}\ kg/m^3}{\sqrt{\pi \cdot 2.0x10^{-13}\ m^2/sec \cdot 31{,}536{,}000\ sec}}$$

$$= 2.9x10^{-3} kg/m^2$$

where:

31,536,000	=	t_e (duration of emissions), corresponding to one year,
0.4	=	θ_e (effective porosity of soil) from Exhibit 4-16,
3.2x10^{-6}	=	D_{ei} (intermediate diffusivity variable) calculated above,
1.6x10^{-7}	=	C_a (concentration of dioxins in air-filled pore space) calculated above, and
2.0x10^{-13}	=	α_i (intermediate diffusivity variable) calculated above.

Finally, K_{vol} can be calculated using Eq. 4-34:

$$K_{vol} \approx -\ln\left[1 - \frac{2.9x10^{-3}}{0.15\ kg/m^2}\right] \approx 1.9x10^{-2}yr^{-1}$$

where:

$2.9x10^{-3}$	=	Na_y (total average emissions from the soil surface in the first year), calculated above, and
0.15	=	d (depth of incorporation of sewage sludge) from Exhibit 4-3; see text for explanation of why this may be expressed as a mass per area.

Step 2: Estimation of K_{lec}

Equation 4-44 can be used to approximate the loss rate coefficient for leaching (K_{lec}):

$$K_{lec} \approx \frac{0.5\ m/yr}{[1400\ kg/m^3 \cdot 13{,}000\ L/kg \cdot 0.001\ m^3/L + 0.2 + 2.9x10^{-3} \cdot 0.2] \cdot 0.15\ m}$$

$$\approx 1.8x10^{-4}\ yr^{-1}$$

where:

0.5	=	NR (annual recharge to ground water) from Exhibit 4-16,
1400	=	BD_{mix} (bulk density of sewage sludge-amended soil) from Exhibit 4-3,
13,000	=	K_d (soil-water partition coefficient) from Exhibit 4-17,
0.001	=	constant to convert (L) to (m^3),
0.2	=	θ_w (water-filled porosity) from Exhibit 4-16,
0.2	=	θ_a (air-filled porosity) from Exhibit 4-16,
$2.9x10^{-3}$	=	H (dimensionless Henry's law constant) calculated above, and
0.15	=	d (depth of incorporation of sewage sludge) from Exhibit 4-3.

Step 2: Estimation of K_{tot}

The total loss rate for dioxins from the sewage sludge-amended agricultural land is calculated from Eq. 4-26:

$$K_{tot} = 4x10^{-3}\ yr^{-1} + 1.9x10^{-2}\ yr^{-1} + 1.8x10^{-4}\ yr^{-1} + 0\ yr^{-1}$$

$$= 2.3x10^{-2}\ yr^{-1}$$

where:

$4x10^{-3}$	=	K_{ero} (loss coefficient for erosion) calculated above,
$1.9x10^{-2}$	=	K_{vol} (loss coefficient for volatilization) calculated above,
$1.8x10^{-4}$	=	K_{lec} (loss coefficient for leaching) calculated above, and
0	=	K_{deg} (loss coefficient for degradation) from Exhibit 4-17.

Maximum Pollutant Mass Available for Erosion

Once K_{tot} is calculated, it is then used in Eq. 4-45 to calculate the maximum mass of pollutant onsite:

$$TP_N = \frac{7\ Mg}{ha\text{-}yr} \cdot 3.11x10^{-4}\ \frac{mg}{kg} \cdot \left(1 + e^{-1 \cdot 2.3x10^{-2}} + \ldots + e^{-19 \cdot 1 \cdot 2.3x10^{-2}}\right) \cdot 1000\ \frac{kg}{Mg} \cdot 1yr$$

$$= 35\ mg/ha$$

where:

7	=	AR (application rate) from Exhibit 4-3,
$3.11x10^{-4}$	=	C_j (concentration of dioxins in sewage sludge) from Exhibit 4-1,
1	=	c (application interval) from Exhibit 4-3,
$2.3x10^{-2}$	=	K_{tot} (total loss rate for dioxins) calculated above,
19	=	b (equal to integer part of $(N\text{-}1)/c$, where N is the site life), from Exhibit 4-3,
1000	=	constant to convert (Mg) to (kg), and
1	=	interval of time over which pollution loss is evaluated.

To calculate the mass of pollutant left in the sewage sludge-amended soil at the end of an individual's lifetime, Eq. 4-46 is used:

$$M_{LS} = 35\ mg/ha \cdot e^{-(2.3x10^{-2}yr^{-1}) \cdot (70yr - 20yr)}$$

$$= 11\ mg/ha$$

where:

35	=	TP_N (total mass of dioxins available at a site after the final year of application) calculated above,

$2.3x10^{-2}$	=	K_{tot} (total loss rate for dioxins) calculated above,
70	=	LS (average human lifetime), assumed to be 70 yr, and
20	=	N_{site} (site life) from Exhibit 4-3.

Then, f_{LS} can be calculated using Eq. 4-47:

$$f_{LS} = \frac{\left(20\ yr \cdot 3.11x10^{-4}\ \frac{mg}{kg} \cdot \frac{7\ Mg}{ha\text{-}yr} \cdot 1000\ \frac{kg}{Mg}\right) - 11\frac{mg}{ha}}{20\ yr \cdot 3.11x10^{-4}\ \frac{mg}{kg} \cdot \frac{7\ Mg}{ha\text{-}yr} \cdot 1000\ \frac{kg}{Mg}}$$

$$= 0.75$$

where:

20	=	N (number of years sewage sludge is applied to land) from Exhibit 4-3,
$3.11x10^{-4}$	=	C_j (concentration of dioxins in sewage sludge) from Exhibit 4-1,
7	=	AR (application rate) from Exhibit 4-3,
1000	=	constant to convert (Mg) to (kg), and
11	=	M_{LS} (mass of dioxins in soil at end of period equal to an individual's lifetime) calculated above.

Pollutant Concentration on Eroded Soil

The expected average concentration of pollutant on soil eroding from the land application site ($C_{site,\ j}$) requires two additional parameters to be calculated. First, the fraction of total pollutant loss caused by erosion is calculated by Eq. 4-27:

$$f_{ero} = \frac{4x10^{-3}\ yr^{-1}}{2.3x10^{-2}\ yr^{-1}} = 0.17$$

where:

$4x10^{-3}$	=	K_{ero} (loss coefficient for erosion) calculated above, and
$2.3x10^{-2}$	=	K_{tot} (total loss rate for dioxins) calculated above.

Second, the calculated rate of soil loss for a land application site is calculated using Eq. 4-49:

$$ME_{site} = 6x10^{-4}\ m/yr \cdot 1400\ kg/m^3 \cdot 10{,}000\ m^2/ha$$

$$= 8400\ kg/ha\text{-}yr$$

where:

$6x10^{-4}$	=	d_e (average rate of soil loss due to erosion from sewage sludge-amended land each year) from Exhibit 4-16,
1400	=	BD_{mix} (bulk density of sewage sludge-amended soil) from Exhibit 4-3, and
10,000	=	constant to convert (ha) to (m^2).

Then, $C_{site,j}$ can be calculated using Eq. 4-48:

$$C_{site,dioxins} = \frac{20\ yr \cdot 3.11x10^{-4}\ \frac{mg}{kg} \cdot 7\ \frac{Mg}{ha\text{-}yr} \cdot 1000\ \frac{kg}{Mg} \cdot 0.17 \cdot 0.75}{8400\ \frac{kg}{ha\text{-}yr} \cdot 70\ yr}$$

$$= 9.5x10^{-6}\ \frac{mg}{kg}$$

where:

20	=	N (number of years sewage sludge is applied to land) from Exhibit 4-3,
$3.11x10^{-4}$	=	C_j (concentration of dioxins in sewage sludge) from Exhibit 4-1,
7	=	AR (application rate) from Exhibit 4-3,
1000	=	constant to convert (Mg) to (kg),
0.17	=	f_{ero} (fraction of total pollutant loss caused by erosion) calculated above,
0.75	=	f_{LS} (fraction of total cumulative loading lost in individual's lifetime to all four loss processes) calculated above,
8400	=	ME_{site} (rate of soil loss) calculated above, and
70	=	LS (lifetime of an individual), assumed to be 70 years.

A dilution factor, to represent the extent to which eroded soil from the land application site is "diluted" by soil from the untreated remainder of the watershed, is calculated using Eq. 4-51. First, the sediment delivery ratios for the land application site and watershed are calculated using Eqs. 4-52 and 4-53:

$$S_{site} = 0.872 \cdot (1074\ ha)^{-0.125} = 0.36$$
$$S_{ws} = 0.872 \cdot (440{,}300\ ha)^{-0.125} = 0.17$$

where:

1074 = A_{site} (area of land application site treated with sewage sludge) from Exhibit 4-16, and

440,300 = A_{ws} (area of the watershed) from Exhibit 4-16.

Then the dilution factor is calculated using Eq. 4-51:

$$D_f = \frac{1074\ ha \cdot 0.36}{(1074\ ha \cdot 0.36) + [(440{,}300\ ha - 1074\ ha) \cdot 0.17]}$$

$$= 5.2x10^{-3}$$

where:

0.36 = S_{site} (sediment delivery ratio for land treated with sludge) calculated above, and

0.17 = S_{ws} (sediment delivery ratio for the watershed) calculated above.

The dry weight concentration of pollutant in eroded soil ($C_{esoil,j}$) can be calculated using Eq. 4-54:

$$C_{esoil,dioxins} = 5.2x10^{-3} \cdot 9.5x10^{-6} \frac{mg}{kg}$$

$$= 4.9x10^{-8}\ \frac{mg}{kg}$$

where:

$5.2x10^{-3}$ = D_f (dilution factor) calculated above, and

$9.5x10^{-6}$ = $C_{site,j}$ (concentration of dioxins in sewage sludge-amended soil eroded from the land application site) calculated above.

Equation 4-56 is then used to calculate the concentration of pollutant dissolved in the surface water:

$$C_{sw,dioxins} = \frac{4.9x10^{-8}\ mg/kg \cdot 16\ mg/L \cdot 10^{-6}\ kg/mg}{1 + 13{,}000\ L/kg \cdot 16\ mg/L \cdot 10^{-6}\ kg/mg}$$

$$= 6.5x10^{-13}\ mg/L$$

where:

$4.9x10^{-8}$	=	$C_{esoil,\ j}$ (dry weight concentration of dioxins in eroded soil) calculated above,
16	=	*TSS* (concentration of total suspended solids in the surface water) from Exhibit 4-16,
13,000	=	K_d (soil-water partition coefficient for dioxins in the stream) from Exhibit 4-17, and
10^{-6}	=	constant to convert (mg) to (kg).

Exposure Calculations

Potential human exposure to dioxins through direct ingestion of surface water is calculated using Eq. 4-59:

$$EXP_{w,dioxins} = \frac{6.5x10^{-13}\ mg/L \cdot 2\ L/day}{70\ kg}$$

$$= 1.8x10^{-14}\ mg/kg\text{-}day$$

where:

$6.5x10^{-13}$	=	$C_{sw,\ j}$ (concentration of dioxins in surface water) calculated above,
2	=	*IW* (ingestion rate of water) from Exhibit 4-16, and
70	=	*BW* (body weight), assumed to be 70 kg.

Potential human exposure to dioxins through consumption of contaminated fish is calculated using Eqs. 4-60 through 4-63. First, the *BCF* for dioxins is calculated using Eq. 4-60:

$$\log_{10}(BCF) = 0.79 \cdot 6.64 - 0.80 = 4.4$$

where:

6.64	=	$\log_{10}(K_{ow})$ from Exhibit 4-17.

The BAF is calculated using Eq. 4-61:

$$BAF = 28{,}000 \cdot 10 = 2.8x10^{5}$$

where:

28,000	=	*BCF* (bioconcentration factor for dioxins) calculated above by exponentiating $\log_{10}(BCF)$, and
10	=	*FM* (food chain multiplier for dioxins) from Exhibit 4-17.

The concentration of dioxins in fish fillets is then calculated from Eq. 4-62:

$$C_{ff,j} = 6.5x10^{-13}\ mg/L \cdot 2.8x10^{5}\ L/kg \cdot 0.5$$

$$= 9.0x10^{-8}\ mg/kg$$

where:

$6.5x10^{-13}$	=	$C_{sw,j}$ (concentration of dioxins in surface water) calculated above,
$2.8x10^{5}$	=	*BAF* (bioaccumulation factor for dioxins) calculated above, and
0.5	=	P_f (ratio of pollutant concentration in fillet to whole fish) from Data Inputs.

Human exposure through ingestion of fish fillets is then calculated using Eq. 4-63:

$$EXP_{f,dioxins} = \frac{9.0x10^{-8}\ mg/kg \cdot 0.04\ kg/day}{70\ kg}$$

$$= 5.2x10^{-11}\ mg/kg\text{-}day$$

where:

$9.0x10^{-8}$	=	$C_{ff,j}$ (concentration of dioxins in fish fillets) calculated above,
0.04	=	*IF* (daily ingestion of fish) from Exhibit 4-16, and
70	=	*BW* (body weight), assumed to be 70 kg.

Total exposure to dioxins in surface water is the sum of the exposures to dioxins in water and fish, as shown in Eq. 4-64:

$$EXP_{dioxins} = 1.8x10^{-14}\ mg/kg-day + 5.2x10^{-11} mg/kg-day$$

$$= 5.2x10^{-11}\ mg/kg-day$$

where:

$1.8x10^{-14}$ = $EXP_{w,j}$ (exposure to dioxins through ingestion of surface water), calculated above, and

$5.2x10^{-11}$ = $EXP_{f,j}$ (exposure to dioxins through ingestion of fish) calculated above.

4.2.13 Pathway 13 - Inhalation of Pollutants Volatilized from Land-Applied Sewage Sludge

Pathway 13 evaluates human exposure to pollutants volatilizing from both agricultural and non-agricultural lands to which sewage sludge has been applied. Non-agricultural lands include forests, reclamation sites, and public contact sites.

To estimate exposure for this pathway, a mass balance analysis is required. This mass balance analysis accounts for the partitioning of pollutants into different soil phases (solids, air, and water) and the subsequent losses of pollutants from the land application site. Pollutants are lost from a land application site by: erosion of contaminated soil particles, which releases pollutants into surface waters; volatilization of pollutants into air; leaching of pollutants into groundwater; and degradation. A mass balance for a pollutant must be maintained, given these four competing loss processes of erosion, volatilization, leaching, and degradation. Once mass balances for pollutants have been established, exposures to pollutants that have eroded, volatilized, or leached are calculated under three separate pathways: surface water (Pathway 12), air (Pathway 13), and groundwater (Pathway 14). Pollutants which have degraded are assumed to have degraded into chemicals that do not pose unacceptable risks to public health or the environment.

The methods for performing the mass balance calculation for Pathways 12, 13, and 14 are discussed in Section 4.2.12. In this section, the equations particular to Pathway 13, for estimating the pollutant mass expected to volatilize and its transport to the downwind edge of the land application site, are presented. Note that volatilization is assumed to occur within a one-year period; any contribution to volatilization from sewage sludge applied in prior years is considered negligible.

Methods Specific to Pathway 13

There are two major steps required to estimate the concentration of a volatilized pollutant in air at the downwind edge of the land application site:

1) Using the mass balance calculations presented in Section 4.2.12, the mass of pollutant expected to volatilize from the land application site within a period equivalent to a human lifespan is estimated.

2) Using a simplified version of the Industrial Source Complex Long Term Model (ISCLT), the transport and dispersion of pollutant in ambient air at the downwind edge of the land application site are modeled.

In the first step, the rate at which a pollutant volatilizes from the site is estimated, based on the assumption that equilibrium has been achieved between annual pollutant loadings and total losses:

$$FA_{vol,j} = 0.001 \cdot AR \cdot C_j \cdot f_{vol} \qquad (4\text{-}65)$$

where:

$FA_{vol,j}$ = annual average flux of pollutant j volatilizing from the sewage sludge-amended soil (kg pollutant/ha-yr),
0.001 = constant to convert units from (Mg-mg/kg) to (kg),
AR = annual whole sludge application rate of sewage sludge to land (dry Mg sewage sludge/ha-yr),
C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge), and
f_{vol} = fraction of total pollutant loss caused by volatilization (dimensionless).

The fraction of total pollutant loss caused by volatilization is obtained from the mass balance calculation presented in Section 4.2.12.

In the second step, pollutant concentrations in ambient air at the downwind edge of the land application site are estimated, based on pollutant fluxes from the site. The model used to simulate transport of pollutants from treated land is described by U.S. EPA (1986d) and is based on equations provided by Environmental Science and Engineering (1985). These equations are simplifications of equations used in ISCLT.

The exposed individual is assumed to live within 1 km of the land application site and to be exposed to concentrations present at the downwind edge of the land application site. A source-receptor ratio is calculated to relate the concentration of pollutant in ambient air at that individual's location (g/m^3) to the rate at which that pollutant is emitted from the treated soil (g/m^2-sec):

$$SRR = 2.032 \frac{A_{site} \cdot v \cdot 10{,}000}{(r' + x_y) \cdot u \cdot \sigma_z} \quad (4\text{-}66)$$

where:

SRR	=	source-receptor ratio (sec/m),
2.032	=	empirical constant,
A_{site}	=	area of land application site (ha),
v	=	vertical term for dispersion of pollutant in air (dimensionless),
10,000	=	constant to convert (ha) to (m^2),
r'	=	distance from center of the land application site to the downwind edge (m),
x_y	=	lateral virtual distance to land application site (m),
u	=	average wind speed (m/sec), and
σ_z	=	standard deviation of the vertical distribution of pollutant concentration in air (m).

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

$$x_y = \sqrt{\frac{A_{site} \cdot 10{,}000}{\pi}} \cot \frac{\theta}{2} \quad (4\text{-}67)$$

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

$$r' = \frac{\sqrt{A_{site} \cdot 10{,}000}}{2} \quad (4\text{-}68)$$

The standard deviation of the vertical distribution of concentration (σ_z) is defined by an atmospheric stability class and the distance from the center of the site to the downwind edge. Exhibit 4-18 provides values for two parameters, a and b, for a range of distances under stable

atmospheric conditions. Based on values from this table, an appropriate value of σ_z is calculated as:

$$\sigma_z = a\, x^b \qquad (4\text{-}69)$$

where:

$$x = 10^{-3} \cdot r' \qquad (4\text{-}70)$$

and:

x = distance from the center of the land application site to the downwind edge (km), and

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

EXHIBIT 4-18
Parameters Used to Calculate σ_z Under Stable Conditions[1,2]

x (km)	a	b
0.10 - 0.20	15.209	0.81558
0.21 - 0.70	14.457	0.78407
0.71 - 1.00	13.953	0.68465
1.01 - 2.00	13.953	0.63227
2.01 - 3.00	14.823	0.54503
3.01 - 7.00	16.187	0.46490
7.01 - 15.00	17.836	0.41507
15.01 - 30.00	22.651	0.32681
30.01 - 60.00	27.084	0.27436
> 60.00	34.219	0.21716

[1] Environmental Science and Engineering, 1985.
[2] σ_z calculated as $\sigma_z = ax^b$, where x is distance in km.

Once the source-receptor ratio has been estimated, it is combined with the estimated average flux of pollutant to predict the average concentration of pollutant in ambient air at the downwind edge of the site:

$$C_{air,j} = FA_{vol,j} \cdot SRR \cdot 0.00317 \qquad (4\text{-}71)$$

where:

$C_{air,j}$ = average concentration of pollutant j in ambient air at the downwind edge of the site (μg pollutant/m^3 air), and

0.00317 = constant to convert (kg/ha-yr) to (μg/m^2-sec).

Estimating Human Exposure

The estimated concentrations of pollutants in air are converted to estimates of human exposure based on assumptions about the rate at which the average individual inhales air:

$$EXP_j = \frac{10^{-3} \cdot C_{air,j} \cdot IA}{BW} \qquad (4\text{-}72)$$

where:

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

10^{-3} = constant to convert (μg) to (mg),

$C_{air,j}$ = average concentration of pollutant j in ambient air (μg pollutant/m^3 air),

IA = inhalation rate (m^3 air/day), and

BW = body weight (kg).

Data Inputs

All the non-pollutant-specific data inputs required for this pathway are presented in Exhibit 4-16 and all the pollutant-specific parameters are presented in Exhibit 4-17, both in Section 4.2.12. As shown in Exhibit 4-16, the daily inhalation rate for humans is assumed to be 20 m^3/day, the average wind velocity is assumed to be 4.5 m/sec, and the average air temperature is assumed to be 288 K.

Example Exposure Calculations for Pathway 13

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

$$f_{vol} = \frac{1.9x10^{-2}/yr}{2.3x10^{-2}/yr} = 0.82$$

where:

1.9×10^{-2}	=	K_{vol} (loss rate due to volatilization of dioxins from sewage sludge-amended land) calculated in Pathway 12, and
2.3×10^{-2}	=	K_{tot} (total loss rate for dioxins from sewage sludge-amended land) calculated in Pathway 12.

Equation 4-65 is used to estimate the annual average flux of dioxins volatilizing from sewage sludge-amended land:

$$FA_{vol,dioxins} = 0.001 \cdot 7 Mg/ha\text{-}yr \cdot 3.11x10^{-4} mg/kg \cdot 0.82$$

$$= 1.8x10^{-6} kg/ha\text{-}yr$$

where:

0.001	=	constant to convert (Mg-mg/kg) to (kg),
7	=	AR (annual whole sludge application rate of sewage sludge to land) from Exhibit 4-3,
$3.11x10^{-4}$	=	C_j (concentration of dioxins in sewage sludge) from Exhibit 4-1, and
0.82	=	f_{vol} (fraction of total pollutant loss caused by volatilization) calculated above.

The source-receptor ratio is then calculated by Eq. 4-66. Three variables, x_y, r', and σ_z, must first be estimated. The lateral virtual distance to the land application site, x_y, is calculated using Eq. 4-67:

$$x_y = \sqrt{\frac{1074\,ha \cdot 10{,}000 m^2/ha}{\pi}} \cdot \cot\frac{22.5}{2}$$
$$= 9295\ m$$

where:

1074	=	A_{site} (area of land application site treated with sewage sludge) from Exhibit 4-16,
10,000	=	constant to convert (ha) to (m^2), and
22.5	=	θ (the angle subtended by the site's width) from Exhibit 4-16.

The standard deviation of the vertical distribution of concentration (σ_z) is then calculated using Eqs. 4-68 through 4-70. The distance from the center of the land application site to the downwind edge is calculated using Eq. 4-68:

$$r' = \frac{\sqrt{1074\,ha \cdot 10{,}000}}{2} = 1639 m$$

where:

1074	=	A_{site} (area of land application site treated with sewage sludge) from Exhibit 4-16, and
10000	=	constant to convert (ha) to (m^2).

The distance from the center of the land application site to the downwind edge is then converted to kilometers using Eq. 4-70:

$$x = 10^{-3}\ km/m \cdot 1639 m = 1.6\ km$$

where:

10^{-3}	=	constant to convert (m) to (km), and
1639	=	r' (distance from center of the land application site to the downwind edge) calculated above.

Then the standard deviation of the vertical distribution is calculated using Eq. 4-69:

$$\sigma_z = 13.953 \cdot 1.6^{0.63227} = 19m$$

where:

13.953	=	a (corresponding to x = 1.6 km) from Exhibit 4-18, and
0.63227	=	b (corresponding to x = 1.6 km) from Exhibit 4-18.

The source-receptor ratio can then be calculated using Eq. 4-66:

$$SRR = 2.032 \cdot \frac{1074ha \cdot 1 \cdot 10{,}000}{(1639m + 9295m) \cdot 4.5m/\text{sec} \cdot 19m}$$
$$= 23\text{sec}/m$$

where:

2.032	=	empirical constant,
1074	=	A_{site} (area of land application site treated with sewage sludge) from Exhibit 4-16,
1	=	v (vertical dispersion term) from Exhibit 4-16,
10000	=	constant to convert (ha) to (m^2),
1639	=	r' (distance from center of the land application site to the downwind edge) calculated above,
9295	=	x_y (lateral virtual distance to land application site) calculated above,
4.5	=	u (wind speed) from Exhibit 4-16, and
19	=	σ_z (standard deviation of the vertical distribution of concentration in air) calculated above.

The concentration of dioxins in air ($C_{air,j}$) at the downwind edge of the site can then be calculated using Eq. 4-71:

$$C_{air,dioxins} = 1.8x10^{-6}kg/ha\text{-}yr \cdot 23\text{sec}/m \cdot 0.00317$$
$$= 1.3x10^{-7}\mu g/m^3$$

where:

$1.8x10^{-6}$	=	$FA_{vol,j}$ (annual average flux of dioxins volatilizing from the treated land) calculated above,
23	=	SRR (source-receptor ratio) calculated above, and
0.00317	=	constant to convert (kg/ha-yr) to ($\mu g/m^2$-sec).

Potential human exposure to dioxins through inhalation of dioxins volatilizing from sewage sludge-amended land is calculated from Eq. 4-72:

$$EXP_{dioxins} = \frac{10^{-3} \cdot 1.3x10^{-7}\mu g/m^3 \cdot 20m^3/day}{70kg}$$

$$= 3.8x10^{-11} mg/kg\text{-}day$$

where:

10^{-3}	=	constant to convert (μg) to (kg),
$1.3x10^{-7}$	=	$C_{air,j}$ (average concentration of dioxins in ambient air) calculated above,
20	=	*IA* (daily inhalation volume) from Exhibit 4-16, and
70	=	*BW* (body weight), assumed to be 70 kg.

4.2.14 Pathway 14 - Ingestion of Groundwater Containing Leached Pollutants

Pathway 14 evaluates human exposure to pollutants through ingestion of groundwater that receives leachate from agricultural and non-agricultural lands to which sewage sludge has been applied. Non-agricultural lands include forests, reclamation sites, and public contact sites.

To estimate exposure for this pathway, a mass balance analysis is required. This mass balance analysis accounts for the partitioning of pollutants into different soil phases (solids, air, and water) and the subsequent losses of pollutants from the land application site. Pollutants are lost from a land application site by: erosion of contaminated soil particles, which releases pollutants into surface waters; volatilization of pollutants into air; leaching of pollutants into groundwater; and degradation. A mass balance for a pollutant must be maintained, given these four competing loss processes of erosion, volatilization, leaching, and degradation. Once mass balances for pollutants have been established, exposures to pollutants that have eroded, volatilized, or leached are calculated under three separate pathways: surface water (Pathway 12), air (Pathway 13), and groundwater (Pathway 14). Pollutants which have degraded are assumed to have degraded into chemicals that do not pose unacceptable risks to public health or the environment.

The methods for performing the mass balance calculation for Pathways 12, 13, and 14 are discussed in Section 4.2.12. In this section, the equations particular to Pathway 14, for estimating the concentration of pollutant in leachate from a site and modeling the transport of that pollutant to the groundwater, are presented.

Methods Specific to Pathway 14

There are two steps required to estimate the concentration of each pollutant in groundwater near the land application site:

1) Determine the concentration of pollutant in water leaching through the treated soil.

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

The maximum mass of pollutant available for leaching from a site is estimated first. For all organic pollutants, except dioxins and dibenzofurans and coplanar PCBs, it is assumed that pollutant concentrations gradually increase in the soil until the rates of annual loss equal the rates of annual loading, and steady-state is achieved. At steady-state, the rate at which the organic pollutant leaches from the site can be determined from the annual loading (which equals total annual losses) and the fraction of total losses attributable to leaching:

$$FA_{lec,j} = AR \cdot C_j \cdot f_{lec} \cdot 0.001 \qquad (4\text{-}73)$$

where:

$FA_{lec,j}$ = annual average flux of pollutant j leaching from sewage sludge-amended soil (kg pollutant/ha-yr),
AR = annual whole sludge application rate of sewage sludge to land (dry Mg sewage sludge/ha-yr),
C_j = concentration of pollutant j in sewage sludge (mg pollutant/kg sewage sludge),
f_{lec} = fraction of total pollutant loss caused by leaching (dimensionless), and
0.001 = constant for converting units from (Mg-mg/kg) to (kg).

For inorganic pollutants, dioxins and dibenzofurans, and coplanar PCBs, sewage sludge is assumed to be applied over a 20 year period, followed by an inactive period. During the inactive period, pollutant is depleted from the treated soil by leaching and erosion, and for the two classes of organic pollutants, volatilization and degradation as well. To simulate potential contamination of groundwater, the loading of pollutant into the unsaturated zone is "linearized" into a pulse of constant magnitude to represent the maximum annual loss of pollutant (in kg/ha-yr) occurring over the 300-year simulation period modeled. The duration of that pulse is calculated so that pollutant mass is conserved. For land application sites, the maximum rate of loss is expected in the year immediately following the last application of sewage sludge, because the concentration of pollutant at the site reaches its peak at that time. As explained in Appendix C, this peak loss rate could be maintained for a maximum length of time described by:

$$TP = \frac{N}{[1 - e^{(-K_{tot}N)}]} \quad (4\text{-}74)$$

where:

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

N = total number of years sewage sludge is applied to land (yr).

This result is combined with an estimate of the fraction of total pollutant loss to leaching for a conservative estimate of the average flux of pollutant leaching from the land application site:

$$FA_{lec,j} = \frac{N \cdot AR \cdot C_j \cdot f_{lec} \cdot 0.001}{TP} \quad (4\text{-}75)$$

The fraction of total pollutant loss caused by leaching is obtained from the mass balance calculation presented in Section 4.2.12.

For both organic and inorganic pollutants, the estimated flux from either Eq. 4-73 or Eq. 4-75 can be combined with the assumed rate of net recharge to groundwater at the land application site to derive an estimate of the average concentration of pollutant in the leachate:

$$C_{lec,j} = \frac{0.1 \; FA_{lec,j}}{NR} \quad (4\text{-}76)$$

where:

$C_{lec,j}$ = average concentration of pollutant j in water leaching from the sewage sludge-amended soil (mg pollutant/L water),

0.1 = constant to convert units from (kg/ha-m) to (mg/L), and

NR = annual recharge to groundwater beneath the treated soil (m recharge/yr).

Next, the leachate concentration is used to estimate the concentration of pollutant in drinking water wells near the site. Two mathematical models are combined to calculate an expected ratio between these two concentrations. The Vadose Zone Flow and Transport finite element module (VADOFT) from the RUSTIC model (U.S. EPA, 1989d,g) is used to estimate flow and transport through the unsaturated zone, and the AT123D analytical model (Yeh, 1981) is used to estimate pollutant transport through the saturated zone.

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

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

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

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

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

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

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

Estimating Human Exposure

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

$$EXP_j = \frac{C_{wel,j} \cdot IW}{BW} \qquad (4\text{-}77)$$

where:

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

Data Inputs

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

Example Exposure Calculations for Pathway 14

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

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

where:

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

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

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

where:

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

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

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

where:

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

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

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

where:

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

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

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

where:

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

4.2.15 Pathway 15 - Infant Exposure to Pollutants Through Breastfeeding

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

Methods

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

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

where:

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

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

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

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

where:

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

BW_{infant} = average body weight of the infant during the exposure period (kg), and

AT = averaging time (yr).

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

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

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

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

Data Inputs

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Example Exposure Calculations for Pathway 15

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

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

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

where:

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

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

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

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

where:

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

4.3 SURFACE DISPOSAL EXPOSURE METHODOLOGIES

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

4.3.1 Definitions of a Monofill and a Surface Impoundment

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

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

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

4.3.2 Methods for the Monofill Prototype

Method for Mass Balance

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

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

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

where:

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

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

$$\dot{H} = \frac{H}{R \cdot T \cdot 10^{-3}} \qquad (4\text{-}81)$$

where:

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

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

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

where:

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

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

$$f_{un} = \frac{t_{un}}{LF} \quad (4\text{-}83)$$

where:

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

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

$$f_{co} = \frac{1}{2} - f_{un} \quad (4\text{-}84)$$

where:

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

½ = fraction of monofill's active lifetime that typical active sewage sludge unit contains sewage sludge (dimensionless).

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

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

where:

q_{un} = pollutant flux from sewage sludge/soil for uncovered period (kg pollutant/m^2 unit-sec),

u = average wind speed (m/sec),

T = temperature (Kelvin),

C_a = concentration of gaseous pollutant in air-filled pore space of sewage sludge/soil (kg pollutant/m^3 air), and

MW = molecular weight of pollutant (g/mol).

For a sewage sludge unit with soil cover:

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

where:

q_{co} = pollutant flux from treated soil for covered period (kg/m^2-sec),

θ_{ca} = air-filled porosity of cover layer of soil (dimensionless),

θ_{ct} = total porosity of cover layer of soil (dimensionless), and

d_c = depth of soil cover (m).

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

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

where:

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

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

$$q_{ac} = q_{un}\, f_{un} + q_{co}\, f_{co} \qquad (4\text{-}88)$$

where:

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

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

$$K_{va} = \frac{q_{ac}\ 3.16\times10^{7}}{d_{mf}} \qquad (4\text{-}89)$$

where:

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

$3.16x10^{7}$ = constant to convert units from (sec^{-1}) to (yr^{-1}).

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

$$K_{ta} = K_{lec} + K_{va} + K_{deg} \qquad (4\text{-}90)$$

where:

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

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

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

where:

f_{la} = fraction of total pollutant loss during monofill's active operation attributable to leaching (dimensionless),

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

f_{da} = fraction of total pollutant loss during monofill's active operation attributable to degradation (dimensionless).

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

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

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

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

where:

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

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

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

where:

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

The fraction of loss attributable to volatilization is calculated as:

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

where:

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

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

Method for Groundwater Pathway

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

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

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

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

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

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

where:

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

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

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

where:

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

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

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

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

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

$$SC = d_{mf} f_{sl} BD_{sludge} 10^4 \quad (4\text{-}98)$$

where:

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

and:

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

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

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

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

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

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

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

$$C_{lec,j} = \frac{0.1 \; FA_{lec,j}}{NR} \quad (4\text{-}100)$$

where:

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

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

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

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

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

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

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

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

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

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

Method for Volatilization Pathway

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

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

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

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

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

where:

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

LS = average human lifetime (yr).

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

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

where:

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

10^{-6} = constant to convert units from (mg/ha-yr) to (kg/ha-yr), and

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

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

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

where:

SRR = source-receptor ratio (sec/m),

2.032 = empirical constant,

A_{mono} = area of monofill (m^2),

v = vertical term for dispersion of pollutant in air (dimensionless),

r' = distance from the monofill's center to the downwind edge (m),

x_y = lateral virtual distance to the monofill (m),

u = average wind speed (m/sec), and

σ_z = standard deviation of the vertical distribution of pollutant concentration in air (m).

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

$$x_y = \sqrt{\frac{A_{mono}}{\pi}} \cot \frac{\theta}{2} \qquad (4\text{-}104)$$

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

$$r' = \frac{\sqrt{A_{mono}}}{2} \qquad (4\text{-}105)$$

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

$$\sigma_z = a\, x^b \qquad (4\text{-}106)$$

where:

$$x = 10^{-3} \cdot r' \qquad (4\text{-}107)$$

and:

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

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

$$C_{air,j} = FA_{vol,j} \cdot SRR \cdot 0.00317 \qquad (4\text{-}108)$$

where:

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

4.3.3 Methods for the Surface Impoundment Prototype

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

Method for Mass Balance

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

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

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

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

4) volatilization from the liquid surface of the impoundment.

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

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

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

Two additional simplifying assumptions are made:

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

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

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

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

$$Q_i C_i = Q_o f_{dl} C_1 + K_{degl} A d_1 C_1 + K_{voll} f_{dl} A C_1 + Q_{sep} f_{dl} A C_1 + DV C_1 \qquad (4\text{-}109)$$

where:

Q_i = rate at which sewage sludge enters the impoundment (m^3 sewage sludge/sec),

C_i = concentration of pollutant in inflow to the impoundment (kg pollutant/m^3 sewage sludge),

Q_o = rate at which outflow leaves the impoundment, possibly for further treatment (m^3 sewage sludge/sec),

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

C_l = total concentration of pollutant (adsorbed and dissolved) in liquid layer of impoundment (kg pollutant/m^3 liquid layer),

K_{degl} = anaerobic rate of pollutant degradation in liquid layer (sec^{-1}),

A = area of surface impoundment (m^2),

d_l = depth of liquid layer (m),

K_{voll} = rate of pollutant volatilization from liquid layer (m/sec),

Q_{sep} = rate of seepage beneath the impoundment (m/sec), and

DV = rate of change in the volume of the liquid layer (m^3 liquid layer/sec).

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

$$DV = \frac{d_{si}A}{TF} \qquad (4\text{-}110)$$

where:

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

The active lifetime of the surface impoundment is calculated as:

$$TF = \frac{d_{si}AS_2}{Q_iS_1} \qquad (4\text{-}111)$$

where:

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

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

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

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

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

where:

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

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

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

$$f_{dl} = \frac{1}{1 + K_d \, 10^{-3} \, S_1} \qquad (4\text{-}114)$$

where:

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

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

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

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

where:

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

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

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

$$de = 2\sqrt{A/\pi}$$

$$FR = \frac{de}{d_1} \qquad (4\text{-}116)$$

where:

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

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

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

where:

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

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

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

where:

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

$$Sc_G = \frac{\mu_a}{\rho_a D_{ca}} \qquad (4\text{-}119)$$

where:

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

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

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

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

$$K_{totl} = Q_o f_{dl} + K_{degl} d_l A + K_{voll} f_{dl} A + Q_{sep} f_{dl} A + DV \qquad (4\text{-}120)$$

so that:

$$Q_i C_i = K_{totl} C_1 \qquad (4\text{-}121)$$

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

$$f_{outl} = \frac{Q_o f_{dl}}{K_{totl}} \qquad f_{degl} = \frac{K_{degl} d_l A}{K_{totl}} \qquad f_{voll} = \frac{K_{voll} f_{dl} A}{K_{totl}}$$

$$f_{sepl} = \frac{Q_{sep} f_{dl} A}{K_{totl}} \qquad f_{dell} = \frac{DV}{K_{totl}} \qquad (4\text{-}122)$$

where:

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

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

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

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

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

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

$$Q_{sep} f_{dl} A C_1 + DVC_1 = K_{deg2} d_2 A C_2 + Q_{sep} f_{d2} A C_2 + DVC_2 \qquad (4\text{-}123)$$

where:

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

d_2 = depth of solids layer (m),

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

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

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

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

where:

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

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

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

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

$$K_{tot2} = K_{deg2} d_2 A + Q_{sep} f_{d2} A + DV \qquad (4\text{-}125)$$

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

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

where:

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

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

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

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

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

where:

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

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

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

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

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

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

where:

f_{sep} = fraction of total pollutant lost from the impoundment through seepage (dimensionless),

f_{vol} = fraction of total pollutant lost from the impoundment through volatilization (dimensionless),

f_{deg} = fraction of total pollutant lost from the impoundment through degradation (dimensionless), and

f_{out} = fraction of total pollutant lost from the impoundment through outflow (dimensionless).

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

Method for Groundwater Pathway

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

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

where:

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

TF = estimated active lifetime of surface impoundment (sec), and

31,536,000 = constant to convert (sec) to (yr).

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

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

where:

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

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

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

where:

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

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

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

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

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

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

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

$$D_f = \frac{F_a}{A Q_{sep} + F_a} \qquad (4\text{-}132)$$

where:

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

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

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

$$v_i = \frac{Q_{sep}\, de}{4 d_a} \qquad (4\text{-}133)$$

where:

v_i = superimposed radial velocity from water seeping from impoundment (m/sec), and
d_a = depth of aquifer (m).

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

$$D_{af} = \frac{v_v + v_h}{v_h} \qquad (4\text{-}134)$$

where:

D_{af} = anti-dilution factor (dimensionless),
v_v = vertical velocity due to the source (m/sec), and
v_h = regional velocity of horizontal groundwater flow (m/sec).

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

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

Method for Volatilization Pathway

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

$$f_{vls} = f_{act} \cdot f_{voll} \qquad (4\text{-}135)$$

where:

f_{vls} = fraction of pollutant mass that volatilizes over a human lifetime (dimensionless),

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

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

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

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

where:

$FA_{vol,j}$ = annual average flux of pollutant j volatilizing from the surface impoundment (kg/ha-yr),

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

f_{vls} = fraction of pollutant mass that volatilizes over a human lifetime (dimensionless),

0.01 = constant to convert (mg/m^2) to (kg/ha),

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

d_{si} = total depth of impoundment (m), and

LS = average human lifetime (yr).

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

$$C_{air,j} = \frac{FA_{vol,j} \cdot SRR}{315.36} \quad (4\text{-}137)$$

where:

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

4.3.4 Estimating Human Exposure

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

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

where:

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

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

$$EXP_j = \frac{C_{wel,j} \cdot IW}{BW} \quad (4\text{-}139)$$

where:

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

4.3.5 Data Inputs

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

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

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

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

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

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

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

EXHIBIT 4-23
Environmental Fate and Transport Parameters

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

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

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

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

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

Notes: NA means Not Applicable

[1] Abiotic hydrolysis rates preferred over biodegradation rates.
[2] Estimated from Lyman et al. (1990), using the Wilke and Lee method.
[3] Estimated from Lyman et al. (1990), using the Hayduk and Laudie method.
[4] Hansch and Leo (1985a).
[5] Estimated using Eq. 4-57 and 4-58.
[6] Jury et al. (1987).
[7] Ou (1984).
[8] Based on Langmuir isotherm data in Bodek et al. (1988).
[9] Smith (1978, 1979); McCall et al. (1981); Alton and Stritzke (1973).
[10] Median value from K_d range in sand measured for that inorganic in Gerritse et al. (1982).
[11] Based on water solubility (Klopffer et al., 1982) and vapor pressure (Que Hee et al., 1981).
[12] CHEMEST procedure of GEMS, U.S. EPA (1989c).
[13] Howard et al. (1985); Montgomery and Welkom (1990).
[14] Howard et al. (1991).
[15] Sattar and Paasivirta, 1980.
[16] Hansch and Leo (1981).
[17] Rathburn and Tai (1987).
[18] Dojlido (1979); Price et al. (1974); Bridie et al. (1979); Dose et al. (1975); Heukelekian and Rand (1955).
[19] Verschueren (1983).
[20] Elliot (1989).
[21] Peyton et al. (1976).

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

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

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

4.3.6 Modeling of Surface Impoundments in the Comprehensive Hazard Identification

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

4.3.7 Example Exposure Calculations for Surface Disposal

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

Groundwater Exposure Pathway for Surface Impoundment

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

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

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

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

where:

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

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

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

$$= 1.5$$

where:

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

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

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

where:

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

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

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

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

where:

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

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

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

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

where:

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

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

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

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

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

and:

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

where:

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

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

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

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

where:

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

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

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

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

where:

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

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

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

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

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

where:

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

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

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

where:

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

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

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

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

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

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

where:

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

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

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

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

where:

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

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

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

where:

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

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

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

where:

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

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

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

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

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

where:

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

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

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

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

where:

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

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

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

$$= 0.86$$

where:

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

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

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

$$= 0.12$$

where:

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

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

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

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

where:

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

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

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

$$= 0.89$$

where:

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

2	=	d_2 (depth of solids layer) from Exhibit 4-22,
20.236	=	A (area of surface impoundment) from Exhibit 4-22, and
$1.7x10^{-2}$	=	K_{tot2} (coefficient for the total rate at which methylene chloride is lost from or stored in the solids layer) calculated above.

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

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

where:

0.86	=	f_{vol1} (fraction of total methylene chloride lost from liquid layer that is lost to volatilization) calculated above,
0.12	=	f_{deg1} (fraction of total methylene chloride lost from liquid layer that is lost to degradation) calculated above,
$2.1x10^{-3}$	=	f_{out1} (fraction of total methylene chloride lost from liquid layer that is lost in outflow from the impoundment) calculated above,
$1.3x10^{-2}$	=	f_{sep1} (fraction of total methylene chloride lost from liquid layer that is lost to seepage) calculated above,
$2.5x10^{-3}$	=	f_{del1} (fraction of total methylene chloride lost from the liquid layer as a result of the diminishing volume of the liquid layer) calculated above,
0.89	=	f_{deg2} (fraction of methylene chloride reaching the solids layer that is lost to degradation) calculated above, and
$8.9x10^{-2}$	=	f_{sep2} (fraction of methylene chloride reaching the solids layer that is lost to seepage) calculated above.

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

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

where:

$2.5x10^{8}$	=	TF (estimated active lifetime of surface impoundment) calculated above,
1.0	=	f_{act} (fraction of each year's loading of methylene chloride lost during each year of the surface impoundment's active phase) calculated above, and
31,536,000	=	constant to convert (sec) to (yr).

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

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

where:

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

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

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

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

where:

0.01	=	constant to convert (mg/m^2) to (kg/ha),
$1.4x10^{-3}$	=	f_{sep} (fraction of total methylene chloride lost from impoundment through seepage) calculated above,
206	=	S_2 (concentration of solids in solids layer) calculated above,
4	=	d_{s1} (total depth of surface impoundment) from Exhibit 4-22,
31.3	=	C_j (concentration of methylene chloride in sewage sludge) from Exhibit 4-1, and
8.0	=	TP (duration of "square wave" for approximating the loading of methylene chloride into the unsaturated soil zone) calculated above.

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

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

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

where:

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

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

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

where:

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

Volatilization Exposure Pathway for Surface Impoundment

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

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

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

where:

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

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

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

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

where:

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

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

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

$$= 403\,m$$

where:

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

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

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

where:

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

From Eq. 107:

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

where:

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

From Eq. 106:

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

where:

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

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

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

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

where:

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

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

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

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

where:

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

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

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

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

where:

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

4.4 INCINERATION EXPOSURE METHODOLOGIES

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

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

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

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

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

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

4.4.1 Estimating Emissions of Pollutants from Incinerators

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

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

where:

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

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

4.4.2 Modeling the Dispersion of Pollutants in Air

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

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

4.4.3 Mapping Dispersion and Pollutant Concentrations Onto a Unified Grid

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

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

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

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

where:

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

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

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

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

where:

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

4.4.4 Estimating Human Exposure

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

$$EXP_j = \frac{AA_j \; IA \; 10^{-3}}{BW} \qquad (4\text{-}143)$$

where:

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

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

4.4.5 Data Inputs

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

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

4.4.6 Example Calculations for Incineration

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

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

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

where:

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

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

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

where:

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

4.5 RISK CALCULATIONS

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

4.5.1 Human Health Risk Calculations

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

$$CI_j = EXP_j \cdot q_j^* \qquad (4\text{-}144)$$

where:

CI_j = incremental cancer risk from pollutant j for exposed individual (incremental risk of developing cancer per lifetime of exposure),
EXP_j = exposure to pollutant j (mg pollutant/kg body weight-day), and
q_j^* = human cancer potency of pollutant j (mg pollutant/kg body weight-day)$^{-1}$.

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

$$CI_j = EXP_j \cdot DE \cdot q_j^* \qquad (4\text{-}145)$$

where:

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

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

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

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

$$RNC_j = \frac{EXP_j}{RfD_j} \qquad (4\text{-}146)$$

where:

RNC_j = ratio of the exposure to the RfD for pollutant j (dimensionless), and
RfD_j = risk reference dose for pollutant j (mg pollutant/kg body weight-day).

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

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

EXHIBIT 4-24
Human Health Toxicity Numbers[1]

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

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

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

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

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

$$RTC_j = \frac{EXP_j}{TLV\text{-}TWA_j} \qquad (4\text{-}147)$$

where:

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

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

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

EXHIBIT 4-25
Threshold Limit Values for Pollutants

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

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

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

Example Calculations for Human Risk

For carcinogens, this example estimates the risk associated with the application of sewage sludge containing dioxins and dibenzofurans to agricultural land. The exposure pathway is Pathway 12, ingestion of fish and water from surface water that receives eroded soil. In the example calculation for exposure presented in Section 4.2.12, the sum of the exposures to dioxins in water and fish was estimated to be 5×10^{-11} mg/kg-day. To estimate the individual risk from this exposure to these carcinogens, Eq. 4-144 is used:

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

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

where:

5.2×10^{-11}	=	EXP_j (exposure to dioxins) calculated in Section 4.2.12, and
3.0×10^{6}	=	q^*_j (human cancer potency of dioxins) from Exhibit 4-24.

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

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

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

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

where:

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

Then using Eq. 4-145:

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

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

where:

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

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

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

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

where:

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

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

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

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

where:

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

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

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

4.5.2 Ecological Risk Calculations

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

$$RQ_j = \frac{EXPA_j}{TRV_j} \qquad (4\text{-}148)$$

where:

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

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

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

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

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

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

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

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

EXHIBIT 4-26
Toxicological Reference Values for Mammals

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

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

Pollutant	Toxicological Reference Value (mg pollutant/kg diet)	Reference (Species)
Bis(2-ethylhexyl) phthalate[1]	130	Ganning et al., 1991 (rat)^N
Boron[1]	190	Weir and Fisher, 1972 (dog)^N
Butanone, 2-[1]	2300	Ralston et al., 1985 (rat)^N
Carbon disulfide[1]	80	Jones-Price et al., 1984 (rabbit)^L
Cresol, p-[1]	410	Hornshaw et al., 1986 (mink)^N
Cyanides (soluble salts and complexes)[1]	10	Gerhart, 1987 (rat)^N
Dioxins and Dibenzofurans	$1.3x10^{-5}$	Kociba et al., 1978 (rat)^N; Murray et al., 1979 (rat)^L
Fluoride[2]	11	HSDB (mice)^N
Manganese[1]	17	Laskey et al., 1982 (rat)^L
Methylene chloride[1]	78	Serota et al., 1986 (rat)^N
Pentachloronitrobenzene[3]	40	HSDB (dog)^L
Phenol[1]	8200	NCI, 1980 (rat)^N
Polychlorinated biphenyls (coplanar)[1]	0.13	Barsotti and Van Miller, 1984 (monkey)^N
Propionic acid, 2-(2,4,5-trichlorophenoxy)	30	HSDB (dog)^N
Silver[1]	9.9	Rungby and Danscher, 1984 (mouse)^L
Thallium[1]	5.2	Downs et al., 1960 (rat)^N
Tin[1]	0.91	Schroeder et al., 1968 (rat)^L
Titanium[1]	27	Schroeder et al., 1964 (mouse)^N
Toluene[1]	4000	NTP, 1990 (rat)^N
Trichlorophenoxy acetic acid, 2,4,5-[1]	39	HSDB (rat)^N
Vanadium[1]	3.9	Domingo et al., 1985 (rat)^N

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

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

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

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

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

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

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

Example Calculation for Ecological Risk

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

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

where:

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

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

4.5.3 Human Health and Ecological Risk Results

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

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

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

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

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

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

Pollutant	Individual Cancer Risk				Exposure as a Fraction of RfD			
	Agricultural Land	Forest	Reclamation Site	Public Contact Sites	Agricultural Land	Forest	Reclamation Site	Public Contact Sites
Acetic acid (2,4-dichlorophenoxy)	5×10^{-10}	2×10^{-10}	2×10^{-10}	5×10^{-10}	<0.01	<0.01	<0.01	<0.01
Antimony					0.8	0.6	0.6	0.8
Barium					0.3	0.3	0.3	0.3
Beryllium	$3x10^{-5}$	$1x10^{-5}$	$1x10^{-5}$	$3x10^{-5}$	0.02	0.02	0.02	0.02
Bis(2-ethylhexyl) phthalate	$2x10^{-6}$	$8x10^{-7}$	$8x10^{-7}$	$2x10^{-6}$	0.1	0.1	0.1	0.1
Boron					0.03	0.02	0.02	0.03
Butanone, 2-					<0.01	<0.01	<0.01	<0.01
Carbon disulfide					<0.01	<0.01	<0.01	<0.01
Cyanides (soluble salts and complexes)					0.08	0.07	0.07	0.08
Dioxins and Dibenzofurans	8×10^{-4}	3×10^{-4}	3×10^{-4}	8×10^{-4}				
Endosulfan-II					0.02	0.01	0.01	0.02
Fluoride					0.09	0.07	0.07	0.09
Manganese					4	3	3	4
Methylene chloride	$2x10^{-7}$	$7x10^{-8}$	$7x10^{-8}$	$2x10^{-7}$	<0.01	<0.01	<0.01	<0.01
Pentachloronitrobenzene	$2x10^{-8}$	$6x10^{-9}$	$6x10^{-9}$	$2x10^{-8}$	<0.01	<0.01	<0.01	<0.01
Phenol					<0.01	<0.01	<0.01	<0.01

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

5. FURTHER ANALYSES OF ROUND TWO POLLUTANTS

5.1 INTRODUCTION

Chapter 4 presented the risk assessments used in the Comprehensive Hazard Identification to evaluate the list of candidate pollutants for the Part 503 Round Two regulation. In that Chapter, results are presented only for those pollutant-exposure pathway combinations for which all pollutant-specific data are available. Examples of pollutant-specific data are plant-uptake slopes for different crops; animal uptake slopes for livestock, poultry, etc.; distribution coefficients (K_ds); and human and ecological toxicity values. In this chapter, the candidate pollutants that warrant further consideration for the final list are presented. For each pollutant, the critical pathways, defined as exposure pathways for which the carcinogenic risk is 1×10^{-4} or higher, the ratio of exposure to the Risk Reference Dose (RfD) is one or greater, or the ecological risk quotient (RQ) is one or greater, are summarized.

5.2 POLLUTANTS THAT WARRANT FURTHER CONSIDERATION

Based on the results of the risk assessments of the Comprehensive Hazard Identification, 12 pollutant candidates have critical pathways for land application and five pollutant candidates have critical pathways for surface disposal. These pollutant candidates and their critical pathways are summarized below in Exhibits 5-1 and 5-2, respectively. None of the inorganic pollutants evaluated had a critical pathway for incineration.

EXHIBIT 5-1
Pollutants with Critical Land Application Pathways

Pollutant	Critical Agricultural Pathways	Critical Non-Agricultural Pathways
Aluminum	6	6 (for., rec., pub.)
Antimony	7, 14	7 (for., rec.); 10 (for., pub.); 14 (for., rec., pub.)
Barium	7, 10, 14	7 (for., rec.); 10 (for., rec., pub.); 14 (for., rec., pub.)
Beryllium	14	14 (for., rec., pub.)
Boron		6 (for., pub.)
Dioxins and Dibenzofurans	2, 3, 10, 12, 13, 15	3 (for., rec., pub.); 10 (for., rec., pub.); 12 (for., rec., pub.); 13 (for., rec., pub.); 15 (for., rec., pub.)
Fluoride	6, 10	6 (for., rec., pub.); 10 (for., rec., pub.)
Manganese	3, 6, 7, 14	3 (for., rec., pub.); 4 (for., rec.); 6 (for., rec., pub.); 7 (for., rec.); 10 (for., pub.); 14 (for., rec., pub.)
PCBs-coplanar	3, 4, 5, 6, 15	3 (for., rec., pub.); 4 (for., rec.); 5 (for., rec.); 6 (for., rec., pub.); 13 (for., rec.); 15 (for., rec., pub.)
Thallium	3	3 (for., rec., pub.)
Tin	7	7 (for., rec.)
Titanium	6	6 (rec.)

Notes:
Pathway 2 = residential home gardener
Pathway 3 = child ingesting sewage sludge
Pathway 4 = human ingesting animal products
Pathway 6 = livestock ingesting forage/pasture
Pathway 7 = livestock ingesting sewage sludge
Pathway 10 = soil organism predators ingesting soil organisms
Pathway 12 = humans ingesting surface water and fish
Pathway 13 = humans inhaling volatilized pollutants
Pathway 14 = humans ingesting groundwater
Pathway 15 = breastfeeding infant
for. = forest land
rec. = reclamation site
pub. = public contact site

EXHIBIT 5-2
Pollutants with Critical Surface Disposal Pathways

Pollutant	Surface Impoundments
Antimony	Groundwater
Barium	Groundwater
Beryllium	Groundwater
Dioxins and Dibenzofurans	Air
Manganese	Groundwater

From Exhibits 5-1 and 5-2, it is evident that the organic candidate pollutants dioxins and dibenzofurans as well as coplanar PCBs have more critical pathways than the inorganic candidate pollutants, except for manganese, which has the same number of critical pathways. These two organic pollutant candidates are recommended to be included on the list of pollutants for the Round Two regulation. The Agency has decided not to recommend including any of the inorganic pollutants on the list for the Round Two regulation, however. The justifications for that decision are presented in Appendix D on a pollutant by pollutant basis.

6. LIST OF POLLUTANTS FOR THE ROUND TWO REGULATION SUBMITTED TO THE COURT

In May, 1993, the Agency submitted a list of 31 pollutant candidates for the Part 503 Round Two regulation to the District Court in Oregon. A copy of the court notice is presented in Appendix D1. On November 30, 1995, EPA submitted the final list of pollutants for the Part 503 Round Two regulation to the court. A copy of that court notice is presented in Appendix D2.

After considering the results of the Comprehensive Hazard Identification, the analysis of pollutants that warranted further consideration, and information received from others, EPA concluded that two pollutants should be on the list for each use or disposal practice. They are: dioxins/dibenzofurans (all monochloro to octachloro congeners) and polychlorinated biphenyls (coplanar). The court notice indicates that EPA may, in the exercise of its discretion, determine to add or delete other pollutants to or from this list at the time the Round Two regulation is proposed.

In addition to the list of pollutants submitted to the court, EPA may change a limit for any of the pollutants in the Round One regulation during development of the Round Two regulation. For this reason, the Round One pollutants also are considered pollutants for the Round Two regulation.

Including the pollutants from Round One regulation, the list of pollutants for the Part 503 Round Two regulation by use or disposal practice is:

Land application

arsenic, cadmium, copper, lead, mercury, molybdenum, nickel, selenium, zinc, dioxins/dibenzofurans, and coplanar polychlorinated biphenyls

Surface disposal

arsenic, chromium, nickel, dioxins/dibenzofurans, and coplanar polychlorinated biphenyls

Sewage sludge incineration

arsenic, beryllium, cadmium, chromium, lead, mercury, nickel, dioxins/ dibenzofurans, and coplanar polychlorinated biphenyls

Dioxins/dibenzofurans and coplanar polychlorinated biphenyls were included on the list of pollutants for sewage sludge incineration even though they were regulated under the Total Hydrocarbons operational standard in Round One. EPA currently is conducting a reassessment of dioxins/dibenzofurans. Because the results of this assessment are unknown, dioxins/dibenzofurans were included on the Round Two list of pollutants for all use or disposal practices. At the completion of the dioxin reassessment, EPA may decide not to regulate

dioxins/dibenzofurans for a particular use or disposal practice or may decide to regulate dioxins/dibenzofurans on an accelerated schedule.

7. REFERENCES

Abt Associates Inc. 1989. Characterization of Surface Disposal for Wastewater Sludge. Memorandum to Alan Rubin, U.S. EPA Office of Water Regulations and Standards. March 27.

Agbenin, J.O., G. Lombin, and J.J. Owunubi. 1991. Direct and Interactive Effect of Boron and Nitrogen on Selected Agronomic Parameters and Nutrient Uptake by Cowpea (Vigna unguicula) Under Glasshouse Conditions. Trop. Agric. (Trinidad). 68(4):356-362.

Agency for Toxic Substances and Disease Registry. 1989. Toxicological Profile for Phenol. Prepared by Syracuse Research Corporation under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-89/20.

Agency for Toxic Substances and Disease Registry. 1990. Toxicological Profile for Silver. Prepared by Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-90/24.

Agency for Toxic Substances and Disease Registry. 1992a. Toxicological Profile for Aluminum and Compounds. Prepared by Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/01.

Agency for Toxic Substances and Disease Registry. 1992b. Toxicological Profile for Antimony and Compounds. Prepared by Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/02.

Agency for Toxic Substances and Disease Registry. 1992c. Toxicological Profile for Barium and Compounds. Prepared by Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/03.

Agency for Toxic Substances and Disease Registry. 1992d. Toxicological Profile for Boron and Compounds. Prepared by Life Systems under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/05.

Agency for Toxic Substances and Disease Registry. 1992e. Toxicological Profile for 2-Butanone. Prepared by Syracuse Research Corporation under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/08.

Agency for Toxic Substances and Disease Registry. 1992f. Toxicological Profile for Carbon Disulfide. Prepared by Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/09.

Agency for Toxic Substances and Disease Registry. 1992g. Toxicological Profile for Cresols. Prepared by Syracuse Research Corporation under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/11.

Agency for Toxic Substances and Disease Registry. 1992h. Toxicological Profile for Manganese and Compounds. Prepared by Life Systems under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/19.

Agency for Toxic Substances and Disease Registry. 1992i. Toxicological Profile for Thallium. Prepared by Life Systems under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/26.

Agency for Toxic Substances and Disease Registry. 1992j. Toxicological Profile for Tin and Compounds. Prepared by Life Systems under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/27.

Agency for Toxic Substances and Disease Registry. 1992k. Draft: Toxicological Profile for Toluene. Prepared by Life Systems under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service.

Agency for Toxic Substances and Disease Registry. 1992l. Toxicological Profile for Vanadium and Compounds. Prepared by Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/29.

Agency for Toxic Substances and Disease Registry. 1993a. Draft: Toxicological Profile for Asbestos (Update). Prepared by Life Systems under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/02.

Agency for Toxic Substances and Disease Registry. 1993b. Toxicological Profile for Di(2-Ethylhexyl)Phthalate (DEHP). Prepared by Life Systems, Inc. under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-92/05.

Agency for Toxic Substances and Disease Registry. 1993c. Toxicological Profile for Beryllium. Prepared by Syracuse Research Corporation under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-92/04.

Agency for Toxic Substances and Disease Registry. 1993d. Toxicological Profile for Cyanide. Prepared by Syracuse Research Corporation under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-92/09.

Agency for Toxic Substances and Disease Registry. 1993e. Toxicological Profile for Polychlorinated Biphenyls. Prepared by Syracuse Research Corporation under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-92/16.

Agency for Toxic Substances and Disease Registry. 1993f. Toxicological Profile for Methylene Chloride. Prepared by Life Systems, Inc. under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-92/13.

Agency for Toxic Substances and Disease Registry. 1993g. Toxicological Profile for Endosulfan. Prepared by Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/16.

Allen, J.R., D.H. Norback, and I.C. Hsu. 1974. Tissue Modifications in Monkeys as Related to Absorption, Distribution, and Excretion of Polychlorinated Biphenyls. Arch. Environ. Contam. Toxicol. 2:86-95.

Alton, J.D. and J.F. Stritzke. 1973. Weed Science. 21:556-60. [Cited in Howard, 1991.]

American Conference of Governmental Industrial Hygienists (ACGIH). 1994. Threshold Limit Values for Chemical Substances and Physical Agents and Biological Exposure Indices for 1994-1995. Cincinnati, OH.

Anderson, H. and J. Amrhein. 1993. Protocol for a Uniform Great Lakes Sport Fish Consumption Advisory. Prepared for the Great Lakes Fish Advisory Task Force. May.

Anderson, M.A. and J.C. Parker. 1990. Sensitivity of Organic Contaminant Transport and Persistence Models to Henry's Law Constants: Case of Polychlorinated Biphenyls. Water, Air, and Soil Pollution. 5:1-18.

AQUIRE (Aquatic Toxicity Information Retrieval). U.S. Environmental Protection Agency. Environmental Research Laboratories. Duluth, MN. August 1994 and February 1995 runs.

Babish, J.G., G.S. Stoewsand, A.K. Furr, T.F. Parkinson, C.A. Bache, W.H. Gutenmann, P.C. Wszolek, and D.J. Lisk. 1979. Elemental and PCB Content of Tissues and Intestinal Aryl Hydrocarbon Hydroxylase Activity of Guinea Pigs Fed Cabbage Grown on Sewage Sludge. J. Agric. Food. Chem. 27(2):399-402.

Baker, M.D. and C.I. Mayfield. 1980. Water, Air, and Soil Pollution. 13:411. [Cited in HSDB, 1994.]

Bar-Yosef, B. and R. Rosenberg. 1988. Response of Corn and Tomato Plants to Fluorine Concentration in Solution Culture. Agronomy Journal. 80:173-177.

Barnes, D.G. and M. Dourson. 1988. Reference Dose (RfD): Description and Use in Health Risk Assessments. Reg. Tox. and Pharm. 8:471-486.

Barsotti, D.A. and J.P. Van Miller. 1984. Accumulation of a Commercial Polychlorinated Biphenyl Mixture (Aroclor 1016) in Adult Rhesus Monkeys and their Nursing Infants. Toxicology. 30:31-44. [Cited in ATSDR, 1993e].

Bertrand, J.E., M.C. Lutrick, G.T. Edds, and R.L. West. 1981. Metal Residues in Tissues, Animal Performance and Carcass Quality with Beef Steers Grazing Pensacola Bahiagrass Pastures Treated with Liquid Digested Sludge. J. Anim. Sci. 53:146-153.

Beyer, W.N. 1990. Evaluating Soil Contamination. U.S. Fish Wild. Serv., Biological Report 90(2). 25 pp.

Bleavins, M.R., W.J. Breslin, R.J. Aulerich, et al. 1984. Placental and Mammary Transfer of a Polychlorinated Biphenyl Mixture (Aroclor 1254) in the European Ferret (*Mustela putorius furo*). Environ. Toxicol. Chem. 3:637-644.

Bodek, I., W.J. Lyman, W.F. Reehl, and D.H. Rosenblatt. 1988. Environmental Inorganic Chemistry. Pergamon Press Inc., New York.

Bohn, H.L. and G. Seekamp. 1979. Beryllium Effects on Potatoes and Oats in Acid Soils. Water, Air, and Soil Pollution. 11:319-322.

Bowers, J.F., et al. 1980. Industrial Source Complex (ISC) Dispersion Model User's Guide (Vol. 1). PB80-133044. U.S. EPA. Research Triangle Park, NC.

Boyd, S.A., D.R. Shelton, D. Berry, and J.M. Tiedje. 1983. Anaerobic Biodegradation of Phenolic Compounds in Digested Sludge. Applied Environmental Microbiology. 46:50-54. [Cited in HSDB, 1994.]

Branson, D.R., I.T. Takahashi, W.M. Parker, and G.E. Blau. 1985. Bioconcentration Kinetics of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin in Rainbow Trout. Environmental Toxicology and Chemistry. 4:779-788.

Bray, B.J., R.H. Dowdy, R.D. Goodrich, and D.E. Pamp. 1985. Metal Accumulation in Tissues of Goats Fed Silage Produced on Sewage-Sludge Amended Soil. J. Environ. Qual. 14:114-118.

Bridie, A.L., C.J.M. Wolff, and M. Winter. 1979. BOD and COD of Some Petrochemicals. Water Res. 13:627-30. [Cited in HSDB, 1994.]

Brown, J.F. and R.W. Lawton. 1984. Polychlorinated Biphenyl Partitioning Between Adipose Tissue and Serum. Bull. Environ. Contamin. Toxicol. 33:277-280.

Butte, N.F., C. Garza, J.E. Stuff, E.O. Smith, and B.L. Nichols. 1984. Effect of Maternal Diet and Body Composition on Lactational Performance. American Journal of Clinical Nutrition. 39:296-306. [Cited in Smith, 1987.]

Callahan, C.A., M.A. Shirazi, and E.F. Neuhauser. 1994. Comparative Toxicity of Chemicals to Earthworms. Environmental Toxicology and Chemistry. 13(2):291-298.

Chaney, R. 1992. U.S. Department of Agriculture. Personal Communication.

Chaney, R.L., G.S. Stoewsand, A.K. Furr, C.A. Bache, and D.J. Lisk. 1978. Elemental Content of Tissues of Guinea Pigs Fed Swiss Chard Grown on Municipal Sewage Sludge Amended Soil. J. Agric. Food Chem. 26(4):994-997.

Chaney, R.L., J.A. Ryan, and G.A. O'Connor. 1991. Risk Assessment for Organic Micropollutants: U.S. Point of View. In: P. L'Hermite et al. (eds). Proc. EEC Symp. Treatment and Use of Sewage Sludge and Liquid Agricultural Wastes. Athens, Greece, Sept. 1990.

Chaney, R.L., R.J.F. Bruins, D.E. Baker, R.F. Korcak, J.E. Smith, Jr., and D.W. Cole. 1987. Transfer of Sludge-Applied Trace Elements to the Food Chain. pp. 67-99. In: A.L. Page, T.J. Logan, and J.A. Ryan (eds.). Land Application of Sludge-Food Chain Implications. Lewis Publishers Inc., Chelsea, MI.

Chou, W.L., et al. 1979. Bioengineering Symposium. 8:391-414. [Cited in HSDB, 1994.]

Davis, J.W. and S.S. Madsen. 1991. The Biodegradation of Methylene Chloride in Soils. Environmental Toxicology and Chemistry. 10:463-474.

Davis, R.D. 1980. Uptake of Fluoride by Ryegrass Grown in Soil Treated with Sewage Sludge. Environ. Pollut. 1:277-284.

Decker, A.M., R.L. Chaney, J.P. Davidson, T.S. Rumsey, S.B. Mohanty and R.C. Hammond. 1980. Animal Performance on Pastures Topdressed with Liquid Sewage Sludge and Sludge Compost. pp. 37-41. In: Proc. Nat'l Conf. Municipal and Industrial Sludge Utilization and Disposal. Information Transfer, Inc., Silver Spring, MD.

Denduluri, S. 1993. Reduction of Manganese Accumulation by EDTA and NTA in Okra (*Abelmoschus esculentus L.*) Grown in Sewage-Irrigated Soil. Bull. Environ. Contam. Toxicol. 52:438-443.

Devillers, J. and J.M. Exbrayat. 1992. Ecotoxicity of Chemicals to Amphibians. Gordon and Breach Science Publishers, Philadelphia, PA. 351 pp.

Dojlido, J.R. 1979. Investigation of Biodegradability and Toxicity of Organic Compounds: Final Report 1975-79. USEPA-600/2-79-163. [Cited in HSDB, 1994.]

Domingo, J.L., J.M. Llobet, J.M. Tomas, et al. 1985. Short-Term Toxicity Studies of Vanadium in Rats. J. Appl. Toxicol. 5:418-421. [Cited in ATSDR, 1992l].

Domingo, J.L., J.M. Llobet, M. Gomez, et al. 1987. Nutritional and Toxicological Effects of Short-Term Ingestion of Aluminum by the Rat. Res. Commun. Chem. Pathol. Pharmacol. 56:409-419. [Cited in ATSDR, 1992a].

Dose, M., et al. 1975. Trib. Cebedeau. 28:3-11. [Cited in HSDB, 1994.]

Doss, G.J., L.E. St. John, Jr., and D.J. Lisk. 1977. Studies of Fluoride Absorption by Plants Grown in Perlite. Bulletin of Env. Contam. and Tox. 18(3):366-369.

Dowdy, R.H. and W.E. Larson. 1975. The Availability of Sludge Borne Metals to Various Vegetable Crops. J. Environ. Quality. 4:278-82.

Downs, W.L., J.K. Scott, L.T. Steadman, et al. 1960. Acute and Sub-Acute Toxicity Studies of Thallium Compounds. Am. Ind. Hyg. Assoc. J. 21:399-406. [Cited in ATSDR, 1992i.]

Eckert, J.W. 1962. Phytopathology. 52:642-649. [Cited in HSDB, 1994.]

Edwards, C.A. and P.J. Bohlen. 1992. The Effects of Toxic Chemicals on Earthworms. Reviews of Environmental Contamination and Toxicology. 125:23-99.

El-Kherbawy, M. and J. Sanders. 1984. Effects of pH and Phosphate Status of a Silty Clay Loam on Mn, Zn, and Cu Concentrations in Soil Fractions and in Clover. J. Sci. Food. Agric. 35:733-739.

Elliot, S. 1989. Atmos. Environ. 23:1977-80. [Cited in HSDB, 1994.]

Environmental Science and Engineering. 1985. Exposure to Airborne Contaminants Released from Land Disposal Facilities--A Proposed Methodology. Prepared by Environmental Science and Engineering, Gainesville, FL for the U.S. EPA Office of Solid Wastes, Washington, DC.

Fries, G.F. 1982. Potential Polychlorinated Biphenyl Residues in Animal Products From Application of Contaminated Sewage Sludge to Land. Journal of Environmental Quality. 11(1):14-20.

Ganning, A.E., M.J. Olsson, U. Brunk, et al. 1991. Effects of Prolonged Treatment with Phthalate Ester on Rat Liver. Pharmacol. Toxicol. 68:392-401. [Cited in ATSDR, 1993b.]

Gerhart, J.M. 1987. Ninety-Day Oral Toxicity Study of Potassium Silver Cyanide ($[KAg(CN)_2]$ in Sprague-Dawley Rats. Prepared for The Dynamac Corporation, Rockville, MD by IIT Research Institute, Chicago, IL. IITRI Project No. L06183, Study No. 4. [Cited in ATSDR, 1993d.]

Gerritse, R.G., R. Vriesema, J.W. Dalenberg, and H.P. De Roos. 1982. Effect of Sewage Sludge on Trace Element Mobility in Soils. Journal of Environmental Quality. 11(3):359-364.

Gile, J.D. and J.W. Gillett. 1979. J. Agric. Chem. 27:1159-1164. [Cited in HSDB, 1994.]

Gillett, J.W. 1994. Cornell University, Ithaca, New York. Phone conversation with Michael Wise, Abt Associates Inc. August 22.

Greve, P.A. and S.L. Wit. 1971. Journal of the Water Pollution Control Federation. 42:2338-48. [Cited in HSDB, 1994.]

Hansch, C. and A.J. Leo. 1979. Substituent Constants for Correlation Analysis in Chemistry and Biology. John Wiley and Sons, New York, NY. [Cited in ATSDR, 1993f.]

Hansch, C. and A.J. Leo. 1981. Medchem Project. Pomona College, Claremont, CA. Issue No. 19. [Cited in HSDB, 1994.]

Hansch, C. and A.J. Leo. 1985a. Medchem Project. Pomona College, Claremont, CA. Issue No. 26. [Cited in Howard, 1991, in ATSDR, 1992g, and in HSDB, 1994.]

Hansch, C. and A.J. Leo. 1985b. Substituent Constants for Correlation Analysis in Chemistry and Biology. John Wiley and Sons, Inc. New York. [Cited in ATSDR, 1989.]

Harfenist, A., T. Power, K.L. Clark, and D.B. Peakall. 1989. A Review and Evaluation of the Amphibian Toxicological Literature. Technical Report Series No. 61. Canadian Wildlife Service Headquarters. 222 pp.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. Effects of Heavy Metal and Other Elemental Additives to Activated Sludge on Growth of *Eisenia foetida*. Journal of Environmental Quality. 10(3):372-376.

Hasset, J.J, W.L. Banwart, and R.A. Griffin. 1983. Environment and Solid Wastes: Characterization, Treatment, and Disposal. Edited by Francis, C.W., S.I. Auerbach, and V.A. Jacobs. Butterworth Publishers, Woburn, MA. pp. 161-175.

Hawley's Condensed Chemical Dictionary. 1993. R.J. Lewis, Sr. (ed.). 12th ed. Van Nostrand Reinhold, New York.

Hazardous Substances Data Base (HSDB). 1994. Chemical files on-line from Toxnet.

Healy, J.B. and L.Y. Young. 1978. Catechol and Phenol Degradation by a Methanogenic Population of Bacteria. Food Microbiology and Toxicology. 35:216-8. [Cited in HSDB, 1994.]

Helmke, P.A., W.P. Robarge, R.L. Korotev, and P.J. Schomberg. 1979. Effects of Soil-Applied Sewage Sludge on Concentrations of Elements in Earthworms. Journal of Environmental Quality. 8(3):322-327.

Henry, J.G. and G.W. Heinke. 1989. Environmental Science and Engineering. Prentice Hall. Englewood Cliffs, N.J.

Heukelekian, H. and M.C. Rand. 1955. Journal of Water Pollution Control Association. 29:1040-53. [Cited in HSDB, 1994.].

Hine, J. and P.K. Mookerjee. 1975. The Intrinsic Hydrophilic Character of Organic Compounds. Correlations in Terms of Structural Contributions. J. Org. Chem. 40(3):292-298.

Hornshaw, T.C., R.J. Aulerich, and R.K. Ringer. 1986. Toxicity of *o*-Cresol to Mink and European Ferrets. Environ. Toxicol. Chem. 5(8):713-720. [Cited in ATSDR, 1992g.]

Horowitz, A., D.R. Shelton, C.P. Cornell, and J.M. Tiedje. 1982. Anaerobic Degradation of Aromatic Compounds in Sediment and Digested Sludge. Dev. Ind. Microbiology. 23:435-444. [Cited in Howard et al., 1991.]

Howard, P.H. (ed.). 1991. Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Lewis Publishers, Inc., Chelsea, Michigan.

Howard, P.H., S. Banerjee, and K.H. Robillard. 1985. Measurement of Water Solubilities, Octanol/Water Partition Coefficients and Vapor Pressures of Commercial Phthalate Esters. Environ. Toxicol. Chem. 4:653-661. [Cited in ATSDR, 1993b.]

Howard, P.H., R.S. Boethling, W.F. Jarvis, W.M. Meylan, and E.M. Michalenko. 1991. Handbook of Environmental Degradation Rates. Lewis Publishers, Inc., Chelsea, Michigan.

Hue, N.V. 1988. A Possible Mechanism for Manganese Phytotoxicity in Hawaii Soils Amended with a Low-Manganese Sewage Sludge. J. Environ. Quality. 17:473-79.

Hwang, S.T. 1982. Toxic Emissions From Land Disposal Facilities. Environmental Progress. 1(1):46-52.

Hwang, S.T. 1985. Model Prediction of Volatile Emissions. Environmental Progress. 4(2):141-144.

Hwang, S.T. and J.W. Falco. 1986. Estimation of Multimedia Exposures Related to Hazardous Waste Facilities. In: Y. Cohen (ed.). Pollutants in a Multimedia Environment. Plenum Publishing Co., New York.

Johnson, B.T. and W. Lulves. 1975. Biodegradation of Di-n-Butyl Phthalate and Di-2-Ethylhexyl Phthalate in Freshwater Hydrosoil. Journal of Fish. Res. Board of Canada. 32:333-339. [Cited in HSDB, 1994.]

Jones-Price, C., R. Tyl-Wolkowski, M.C. Marr, et al. 1984. Teratologic Evaluation of Carbon Disulfide (CAS No. 75-15-0) Administered to New Zealand White Rabbits on Gestational Days 6 through 19. Research Triangle Park, NC: National Center for Toxicological Research, Division of Teratogenesis Research. NCTR 222-80-2031(C); NTIS PB84-0192350. [Cited in ATSDR, 1992f.]

Jury, W.A., et al. 1987. Rev. Environ. Contam. Toxicol. 99:119-164. [Cited in Howard, 1991.]

Kalbasi, M., F. Filsoof, and y. Rezai-Nejad. 1988. Effect of Sulfur Treatments on Yield and Uptake of Fe, Zn, and Mn by Corn, Sorghum, and Soybeans. Journal of Plant Nutrition. 11(6-11):1353-1360.

Kaplan, D.I., D.C. Adriano, and K.S. Sajwan. 1990. Thallium Toxicity in Beans. Journal of Environmental Quality. 19(3):359-365.

Khattak, R.A. and W.M. Jarrell. 1989. Effect of Saline Irrigation Waters on Soil Manganese Leaching and Bioavailability to Sugar Beets. Soil Sci. Soc. Am. J. 53:142-146.

Klopffer, W., et al. 1982. Ecotox Environ. Safety. 6:294-301. [Cited in HSDB, 1994.]

Kociba, R.J., D.G. Keyes, J.E. Beyer, R.M. Carreon, C.E. Wade, D.A. Dittenber, R.P. Kalnins, L. Frauson, C.N. Park, S.D. Barnard, R.A. Hummell, and G.C. Humiston. 1978. Results of a Two-Year Chronic Toxicity and Oncogenicity Study of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) in Rats. Tox. and Appl. Pharm. 46:279-303.

Kosak-Channing, L. 1986. Beryllium Distribution in Hydroponically-Grown Tobacco Plants. Plant Science. 46:175-180.

Laskey, J.W., G.L. Rehnberg, J.F. Hein, et al. 1982. Effects of Chronic Manganese (Mn_3O_4) Exposure on Selected Reproductive Parameters in Rats. J. Toxicol. Environ. Health. 9:677-687. [Cited in ATSDR, 1992h.]

Linehan, D.J. 1984. Micronutrient Cation Sorption by Roots and Uptake by Plants. Journal of Experimental Botany. 35:1571-1574.

Liu, D., W.M.J. Strachan, K. Thomson, and K. Kwasniewska. 1981. Determination of the Biodegradability of Organic Compounds. Environmental Science and Technology. 15(7):788-93.

Lyman, W.J., W.F. Reehl, and D.H. Rosenblatt. 1990. Handbook of Chemical Property Estimation Methods. American Chemical Society, Washington, DC.

Mabey, W.R., J.H. Smith, R.T. Podoll, et al. 1982. Aquatic Fate Process Data for Organic Priority Pollutants. Washington, DC: U.S. EPA, Office of Water Regulations and Standards. EPA-440/4-81-014. PB87-169090. [Cited in ATSDR, 1992k, 1993f, 1993g.]

Mackay, D. and P.J. Leinonen. 1975. Rate of Evaporation of Low-Solubility Contaminants From Water Bodies to Atmosphere. Environmental Science and Technology. 9(13):1178-1180.

Mackay, D. and A. Yeun. 1983. Mass Transfer Coefficient Correlations for Volatilization of Organic Solutes from Water. Environmental Science and Technology. 17(4):423-429.

McCall, P.J., et al. 1981. Journal of Agriculture and Food Chemistry. 29:100-7. [Cited in Howard, 1991.]

Menzie, C.A., D.E. Burmaster, J.S. Freshman, and C.A. Callahan. 1992. Assessment of Methods for Estimating Ecological Risk in the Terrestrial Component: A Case Study at the Baird & McGuire Superfund Site in Holbrook, Massachusetts. Environmental Toxicology and Chemistry. 11:245-260.

Merck Index. 1989. S. Budavari, M.J. O'Neil, A. Smith, and P.E. Heckelman (eds.). 11th ed. Merck & Co., Inc., Rahway, NJ.

Montgomery, J.H. and L.M. Welkom. 1990. Groundwater Chemicals Desk Reference. Lewis Publishers, Inc., Chelsea, Michigan. [Cited in ATSDR, 1993b.]

Muchovej, R.M.C., V.G. Allen, D.C. Martens, L.W. Zelazny, and D.R. Notter. 1986. Aluminum, Citric Acid, Nitrotriacetic Acid, and Soil Moisture Effects on Aluminum and Iron Concentrations in Ryegrass. Agron. J. 78:138-145.

Murphy, T.J., et al. 1987. Environmental Science and Technology. 21:155-162. [Cited in HSDB, 1994.]

Murray, F.J., F.A. Smith, K.D. Nitschke, C.G. Humiston, R.J. Kociba, and B.A. Schwetz. 1979. Three-Generation Reproduction Study of Rats Given 2,3,7,8-TCDD in the Diet. Tox. and Appl. Pharm. 50:241-252.

NCI. 1980. Bioassay of Phenol for Possible Carcinogenicity. Bethesda, MD: U.S. Department of Health and Human Services, National Cancer Institute. NCI-CG-TR-203. [Cited in ATSDR, 1989.]

Neuhauser, E.F. and C.A. Callahan. 1990. Growth and Reproduction of the Earthworm *Eisenia fetida* Exposed to Sublethal Concentrations of Organic Chemicals. Soil Biochemistry. 22(2):175-179.

NTP. 1990. National Toxicology Program Technical Report Series Toxicology and Carcinogenesis Studies of Toluene (CAS No. 108-88-3) in F344/N Rats and 86C3F Mice (Inhalation Studies). Research Triangle Park, NC: U.S. Environmental Protection Agency, Department of Health and Human Services, No. 371. PB90-256371. [Cited in ATSDR, 1992k.]

O'Connor, G. 1992. Professor and Chairman, Soil and Water Science Dept., University of Florida. Personal Communication.

OHM/TADS. 1989. Oil and Hazardous Materials/Technical Assistance Data System. Chemical Information Systems. September 14, 1989. [Cited in ATSDR, 1993g.]

Ou, Li-Tse. 1984. 2,4-D Degradation and 2,4-D Degrading Microorganisms in Soils. Soil Science. 137(2):100-7.

Perry, H.M., Jr., S.J. Kopp, M.W. Erlanger, et al. 1983. Cardiovascular Effects of Chronic Barium Ingestion. Trace Subst. Environ. Health. 17:155-164. [Cited in ATSDR, 1992c.]

Perry, H.M., Jr., S.J. Kopp, E.F. Perry, et al. 1989. Hypertension and Associated Cardiovascular Abnormalities Induced by Chronic Barium Feeding. J. Toxicol. Environ. Health. 28:373-388. [Cited in ATSDR, 1992c.]

Perry, H.M., Jr., E.F. Perry, M.W. Erlanger, et al. 1985. Barium-Induced Hypertension. Adv. Mod. Environ. Toxicol., Inorg. Drinking Water Cardio. Vasc. Dis. 9:221-229. [Cited in ATSDR, 1992c.]

Peyton, T.O., et al. 1976. Carbon Disulfide, Carbonyl Sulfide Literature Review and Environmental Assessment. USEPA-600/9-78-009. [Cited in HSDB, 1994.]

Price, K.S., et al. 1974. Journal of Water Pollution Control Federation. 46:63-77. [Cited in HSDB, 1994.]

Que Hee, S.S., et al. 1981. The Phenoxyalkanoic Herbicides. Vol. 1. Chem. Anal. Environ. Pollution. CRC Press Inc., Boca Raton, FL. [Cited in HSDB, 1994.]

Ralston, W., R. Hilderbrand, D. Uddin, et al. 1985. Potential of 2,5-Hexanedione Neurotoxicity by Methyl Ethyl Ketone. Toxicol. Appl. Pharmacol. 81:319-327. [Cited in ATSDR, 1992e.]

Rathburn, R.E. and D.Y. Tai. 1987. Vapor Pressures and Gas-Film Coefficients for Ketones. Chemosphere. 16:69-78. [Cited in ATSDR, 1992e.]

Rhee, G.Y., et al. 1989. Water Res. 23(8):957-64. [Cited in HSDB, 1994.]

Roberts, B.L. and H.W. Dorough. 1984. Relative Toxicities of Chemicals to the Earthworm *Eisenia foetida*. Environmental Toxicology and Chemistry. 3:67-78.

Roberts, B.L. and H.W. Dorough. 1985. Hazards of Chemicals to Earthworms. Environmental Toxicology and Chemistry. 4:307-323.

Romney, E.M., A. Wallace, R. Wood, A.M. El-Gazzar, J.D. Childress, and G.V. Alexander. 1977. Role of Organic Matter in a Desert Soil on Plant Response to Ag, W, Co, and Pb. Commun. in Soil Science and Plant Analysis. 8(9):719-725.

Rungby, J. and G. Danscher. 1984. Hypoactivity in Silver Exposed Mice. Acta Pharamcol. Toxicol. 55:398-401. [Cited in ATSDR, 1990.]

Sattar, M.A. and J. Paasivirta. 1980. Chemosphere. 9:745-752. [Cited in HSDB, 1994.]

Schmid, P. and C. Schlatter. 1985. Excretion and Metabolism of Di(2-Ethylhexyl)Phthalate in Man. Xenobiotica. 15(3):251-256.

Schroeder, H.A., J.J. Balassa, and W.H. Vinton, Jr. 1964. Chromium, Lead, Cadmium, Nickel, and Titanium in Mice: Effect on Mortality, Tumors, and Tissue Levels. J. Nutrit. 83:239-250. [Cited in WHO, 1982.]

Schroeder, H.A., M. Kanisawa, D.V. Frost, et al. 1968. Germanium, Tin, and Arsenic in Rats: Effects on Growth, Survival, Pathological Lesions, and Life Span. J. Nutr. 96:37-45. [Cited in ATSDR, 1992j.]

Schroeder, H.A. and M. Mitchener. 1975. Life-term Studies in Rats: Effects of Aluminum, Barium, Beryllium, and Tungsten. J. Nutr. 105:421-427. [Cited in ATSDR, 1992c.]

Schroeder, H.A., M. Mitchener, and A.P. Nason. 1970. Zirconium, Niobium, Antimony, Vanadium, and Lead in Rats: Life-Time Studies. J. Nutr. 100:59-68. [Cited in ATSDR, 1992b.]

Schwartz, H.E., et al. 1979. International Journal of Environmental Analytical Chemistry. 6:133-144. [Cited in HSDB, 1994.]

Schwarzenbach, R.P., P.M. Gschwend, and D.M. Imboden. 1993. Environmental Organic Chemistry. John Wiley & Sons, Inc. New York.

Serota, D.G., A.K. Thakur, B.M. Ulland, et al. 1986. A Two-Year Drinking-Water Study of Dichloromethane in Rodents. I. Rats. Food Chem. Toxicol. 24:951-958. [Cited in ATSDR, 1993f.]

Shelton, D.R., S.A. Boyd, and J.M. Tiedje. 1984. Anaerobic Biodegradation of Phthalic Acid Esters in Sludge. Environmental Science and Technology. 18:93-97. [Cited in Howard et al., 1991.]

Shen, T.T. 1982. Estimation of Organic Compound Emissions From Waste Lagoons. Journal of the Air Pollution Control Association. 32(1):79-82.

Singh, A., R. Chhabra, and I.P. Abrol. 1979a. Effect of Fluorine and Phosphorous on the Yield and Chemical Composition of Rice Grown in Soils of Two Sodicities. Soil Science. 127(2):86-93.

Singh, A., R. Chhabra, and I.P. Abrol. 1979b. Effect of Fluorine and Phosphorous Applied to a Sodic Soil on Their Availability and on Yield and Chemical Composition of Wheat. Soil Science. 128(2):90-97.

Sjoberg, et al. 1985. [Cited in ATSDR, 1993b.]

Smith, A.E. 1978. Relative Persistence of Di- and Tri-Chlorophenoxyalkanoic Acid Herbicides in Saskatchewan Soils. Weed Res. 18:275-9. [Cited in Howard et al., 1991, and HSDB, 1994.]

Smith. A.E. 1979. Soil Persistence Experiments with $^{(14C)}$2,4-D in Herbicidal Mixtures and Field Persistence Studies with Tri-Allate and Trifluralin Both Singly and Combined. Weed Res. 19:165-170. [Cited in Howard et al., 1991.]

Smith, A.H. 1987. Infant Exposure Assessment for Breast Milk Dioxins and Furans Derived From Waste Incineration Emissions. Risk Analysis. 7(3):347-353.

Snider, J.R. and G.A. Dawson. 1985. Journal of Geophys. Res. (Atmosphere). 90(D):3797-805. [Cited in HSDB, 1994.]

Soon, Y.K. and T.E. Bates. 1985. Molybdenum, Cobalt, and Boron Uptake from Sewage Sludge Amended Soils. Can. J. Soil Sci. 65:507-17.

Springer, C., P.D. Lunney, and K.T. Valsaraj. 1984. Emission of Hazardous Chemicals From Surface and Near Surface Impoundments to Air. U.S. EPA, Solid and Hazardous Waste Research Division, Cincinnati, OH. Project Number 808161-02.

Stark, J.M. and E.F. Redente. 1990. Plant Uptake and Cycling of Trace Elements on Retorted Oil Shale Disposal Piles. J. Environ. Qual. 19:495-501.

Thomann, R.J. and J.A. Mueller. 1987. Principles of Surface Water Quality Modeling and Control. Harper and Row, New York.

Timson, B.F. and J.L. Coffman. 1984. Body Composition by Hydrostatic Weighing at Total Lung Capacity and Residual Volume. Medicine and Science in Sports Exercise. 16:411-414. [Cited in Smith, 1987.]

Tonkonozhenko, Y.V. and M.I. Khlyupina. 1974. Titanium in the Soils and Plants of Krasnodar Kray. Soviet Soil Science (Pochvovedeniye). 3:38-45.

U.S. Department of Agriculture (USDA). 1987. Summary Report: National Resources Inventory. Statistical Bulletin Number 790. Soil Conservation Service.

U.S. EPA. 1978. Process Design Manual for Municipal Sludge Landfills. Office of Solid Waste. EPA-625/1-78-010/SW-705.

U.S. EPA. 1982. Fate of Priority Pollutants in Publicly-Owned Treatment Works. Vol. I. Effluent Guidelines Division, Washington, DC. EPA 440/1-82-303.

U.S. EPA. 1984. Environmental Regulations and Technology: Use and Disposal of Municipal Wastewater Sludge. Prepared by the U.S. EPA Intra-Agency Sludge Task Force. EPA 625/10-84-003. September.

U.S. EPA. 1985. Summary of Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge: Methods and Results. Washington, DC. Office of Water.

U.S. EPA. 1986a. Report to Congress on the Discharge of Hazardous Wastes to Publicly Owned Treatment Works. EPA/530-SW-86-004.

U.S. EPA. 1986b. Cancer Risk Assessment Guidelines. CFR 51(185):33992-34003. September 24.

U.S. EPA. 1986c. Industrial Source Complex (ISC) Dispersion Model User's Guide. Second Edition. U.S. EPA, EPA 450/4-86-005a and 005b. Research Triangle Park, NC.

U.S. EPA. 1986d. Research and Development: Development of Risk Assessment Methodology for Municipal Sludge Landfilling. Prepared by Environmental Criteria and Assessment Office, Cincinnati, OH for the Office of Water Regulations and Standards. ECAO-CIN-485.

U.S. EPA. 1987a. Ambient Aquatic Life Water Quality Criteria for Silver. Draft. Environmental Research Laboratories, Duluth, MN and Narragansett, RI. September.

U.S. EPA. 1987b. Hazardous Waste Treatment, Storage, and Disposal Facilities (TSDF)--Air Emissions Models. Office of Air Quality Planning and Standards, Research Triangle Park, NC. EPA-450/3-87-026.

U.S. EPA. 1988a. Ambient Water Quality Criteria for Aluminum--1988. EPA 440/5-86-008. August.

U.S. EPA. 1988b. Ambient Aquatic Life Water Quality Criteria for Antimony (III). Draft. Environmental Research Laboratories, Duluth, MN and Narragansett, RI. August.

U.S. EPA. 1988c. Recommendations for and Documentation of Biological Values for Use in Risk Assessment. Environmental Criteria and Assessment Office, Office of Health and Environmental Assessment, Office of Research and Development. EPA/600/6-87/008. February.

U.S. EPA. 1989a. 1988 National Sewage Sludge Survey. Office of Water.

U.S. EPA. 1989b. Development of Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge. Office of Research and Development, Cincinnati, OH. EPA/600/6-89/001.

U.S. EPA. 1989c. PC-GEMS Database. User's Guide, Release 1.0. Prepared by General Sciences Corporation for the Office of Pesticides and Toxic Substances. Contract No. 68024281.

U.S. EPA. 1989d. Risk of Unsaturated/Saturated Transport and Transformation of Chemical Concentrations (RUSTIC), Volume II: User's Guide. Environmental Research Laboratory, Athens, GA. EPA/600/3-89/048b.

U.S. EPA. 1989e. Background Document for the Surface Impoundment Modeling System (SIMS). Control Technology Center. Research Triangle Park, NC. EPA/600-6-89-001. NTIS PB90-135740/A5.

U.S. EPA. 1989f. Screening Study for Wildlife Criteria Development. Office of Water, Office of Water Regulations and Standards.

U.S. EPA. 1989g. Risk of Unsaturated/Saturated Transport and Transformation Interactions for Chemical Concentrations (RUSTIC), Volume I: Theory and Code Verification. Prepared by Woodward Clyde Consultants, HydroGeologic, and AQUA TERRA Consultants for the Office of Research and Development, Environmental Research Laboratory, Athens, GA. Contract No. 68-03-6304.

U.S. EPA. 1989h. Technical Support Document: Incineration of Sewage Sludge (Proposal). Office of Water.

U.S. EPA. 1989i. Interim Procedures for Estimating Risks Associated with Exposures to Mixtures of Chlorinated Dibenzo-p-Dioxins and Dibenzofurans (CDDs and CDFs). A 1989 Update. U.S. EPA Risk Assessment Forum, Washington, DC. EPA 6253-89-016.

U.S. EPA. 1990a. Development of Risk Assessment Methodology for Surface Disposal of Municipal Sludge. Prepared by Abt Associates Inc. for the Environmental Criteria Assessment Office, Office of Research and Development, Cincinnati, OH. ECAO-CIN-750.

U.S. EPA. 1990b. Guidance On: Assessment and Control of Bioconcentratable Contaminants in Surface Waters. DRAFT.

U.S. EPA. 1990c. National Sewage Sludge Survey; Availability of Information and Data, and Anticipated Impacts on Proposed Regulations; Proposed Rule. 40 CFR Part 503, Washington, DC.

U.S. EPA. 1990d. Implementation of a Chemical Ranking System. Draft Final Report. Criteria and Standards Division. EPA Contract #68-03-3534. May 24.

U.S. EPA. 1991a. Human Health Risk Assessment for Dioxin in Pulp and Paper Sludge: Technical Support Document for the Proposed Land Application Rule. April.

U.S. EPA. 1991b. PIRANHA. Version 2.0. Environmental Research Laboratory, Office of Research and Development.

U.S. EPA. 1992a. Technical Support Document for Land Application of Sewage Sludge. Office of Water, Office of Science and Technology. EPA 822/R-93-001a. November.

U.S. EPA. 1992b. Technical Support Document for Sewage Sludge Incineration. Office of Water. EPA 822/R-93-003. November.

U.S. EPA. 1992c. Technical Support Document for the Surface Disposal of Sewage Sludge. Office of Water. EPA 822/R-93-002. November.

U.S. EPA. 1992d. Estimating Exposure to Dioxin-Like Compounds. Review Draft. Office of Research and Development. EPA/600/6-88/005B. August.

U.S. EPA. 1993a. Human Health Risk Assessment for the Use and Disposal of Sewage Sludge: Benefits of Regulation. Prepared by Abt Associates Inc. under contract no. 68-C0-0093 for the Office of Water. January.

U.S. EPA. 1993b. Comparison and Rank of Proposed Human Health Bioaccumulation Factors for the Great Lakes Initiative. EPA-822-R-93-010. Office of Water. August.

U.S. EPA. 1994a. Revision of Assessment of Risks to Terrestrial Wildlife from TCDD and TCDF in Pulp and Paper Sludge. Prepared by Abt Associates Inc. under contract no. 68-C0-0093 for the Office of Pollution Prevention and Toxics. December.

U.S. EPA. 1994b. Estimating Exposure to Dioxin-Like Compounds. Volume II: Properties, Sources, Occurrence and Background Exposures. Office of Health and Environmental Assessment. June. EPA/600/6-88/005Cb. External Review Draft.

U.S. EPA. 1994c. Health Assessment Document for 2,3,7,8 - Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds. Volume II. EPA/600/BP-92/001b. External Review Draft. June.

U.S. Geological Survey. 1992. Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States. H.T. Shacklette and J.G. Boerngen. U.S. Geological Survey Professional Paper 1270. Second printing.

Vanoni, Vita A. (ed.). 1975. Sedimentation Engineering. Prepared by the ASCE Task Committee for the Preparation of the Manual on Sedimentation of the Sedimentation Committee of the Hydraulics Division, New York, NY.

Verschueren, K. 1983. Handbook of Environmental Data on Organic Chemicals. Van Nostrand Reinhold Co., New York, NY. 2nd ed. [Cited in ATSDR, 1992f, 1993b.]

Voight, G., K. Henrichs, G. Prohl, and H.G. Paretzke. 1988. Measurements of Transfer Coefficients for 137Cs, 60 Co, 54 Mn, 22 Na, 131I and 95mTc from Feed into Milk and Beef. Radiation and Environmental Biophysics. 27:153-164.

Walton, K.C. 1987. Effects of Treatment with Sodium Fluoride and Subsequent Starvation on Fluoride Content of Earthworms. Bulletin of Environmental Contamination and Toxicology. 38:163-170.

Wang, C.H. and F.E. Broadbent. 1972. Kinetics of Losses of PCNB and DCNA in Three California Soils. Soil Sci. Soc. Amer. Proc. 36:742-745.

Weast, R.C. (ed.). 1990. CRC Handbook of Chemistry and Physics. 70th ed. CRC Press, Inc., Boca Raton, FL.

Webber, M.D., H.D. Monteith, and D.G.M. Corneau. 1983. Assessment of Heavy Metals and PCBs at Sludge Application Sites. Journal of the Water Poll. Control Fed. 55(2):187-195.

Weir, R.J., Jr. and R.S. Fisher. 1972. Toxicologic Studies on Borax and Boric Acid. Toxicol. Appl. Pharmacol. 23:351-364. [Cited in ATSDR, 1992d.]

Whelan, B.R. 1993. Effect of Barium Selenate Fertilizer on the Concentration of Barium in Pasture and Sheep Tissues. J. Agric. Food Chem. 41:768-770.

Whitehead, R.G. and A.A. Paul. 1981. Infant Growth and Human Milk Requirements. Lancet. 2:161-163. [Cited in Smith, 1987.]

WHO. 1982. World Health Organization. Environmental Health Criteria 24: Titanium. Geneva.

Wilson, J.T., J.F. McNabb, D.L. Balkwill, and W.C. Ghiorse. 1983. Enumeration and Characterization of Bacteria Indigenous to a Shallow Water-Table Aquifer. Ground Water. 21(2):134-142.

Yakushiji, T., I. Watanabe, K. Kuwabura, et al. 1978. Long-Term Studies of the Excretion of Polychlorinated Biphenyls (PCBs) Through the Mother's Milk of an Occupationally Exposed Worker. Arch. Environ. Contam. Toxicol. 7:493-504. [Cited in ATSDR, 1993e.]

Yeh, G.T. 1981. AT123D: Analytical Transport One-, Two-, and Three-Dimensional Simulation of Waste Transport in the Aquifer System. Oak Ridge National Laboratory, Environmental Sciences Division. Publication No. 1439. March.

APPENDIX A

ANALYSIS OF POLLUTANTS DETECTED LESS THAN TEN PERCENT OF THE TIME

Introduction

EPA conducted two screening analyses to determine if any of the 69 pollutants detected less than ten percent of the time in sewage sludge might still pose an unacceptable risk to human health. For the first screening analysis, EPA used the algorithms from agricultural Pathway 3. This pathway tends to result in high risk because small children are directly ingesting sewage sludge, without any of the mitigating influences of degradation, dilution, etc. found in other pathways. For the second screening analysis, EPA evaluated other pathways for pollutants with relatively large cancer potency slopes, or q_1* values.

To conduct these analyses, human toxicity data were needed. Exhibit A-1 presents the available human toxicity data for the 69 pollutants as well as each pollutant's frequency of detection, as measured in the 1988 National Sewage Sludge Survey (U.S. EPA, 1989a).

Screening Analysis Based on Pathway 3

To calculate exposure from agricultural Pathway 3, the only pollutant-specific data required is the pollutant's concentration in sewage sludge, as described in Section 4.2.3. EPA chose to use 98th percentile pollutant concentrations with non-detects set equal to the minimum detection level. The Agency did not use 99th percentile concentrations because such estimates are not as statistically meaningful when pollutants are only detected a few percent of the time. For the non-pollutant-specific data required for this analysis, a sewage sludge ingestion rate of 0.2 g/day, a body weight of 16 kg, and an exposure duration (for cancer) of 5/70 were used.

To calculate risk, either an oral Risk Reference Dose (RfD) or an oral q_1* value was needed. Of the 69 pollutants detected less than ten percent of the time, 49 had at least one of these estimates of toxicity. Six of these pollutants had already been evaluated for Pathway 3 in Round One, and so were not considered further: aldrin, dieldrin, benzo(a)pyrene, DDT, DDE, and trichloroethene. For the remaining 43 pollutants, EPA estimated risk. For those pollutants with an oral RfD value, the ratio of exposure to RfD was calculated. For those pollutants with an oral q_1^* value, the risk of cancer was estimated. These results are presented in Exhibit A-2.

As shown in Exhibit A-2, for all but one of the 43 pollutants analyzed, the ratio of exposure to RfD was below one and the cancer risk was below one in 100,000. For 2-picoline, the ratio of exposure to RfD was five. EPA chose to not evaluate 2-picoline further, however, because it was only detected one percent of the time in the 1988 NSSS.

EXHIBIT A–1

Available Human Toxicity Data for 72 Chemicals Detected Less Than 10 Percent of the Time[a]

CAS	Chemical Name	Percent Detect	REFERENCE DOSES VALUES mg/kg–d[b] INHALATION		ORAL		Mut.?[c]	CANCER POTENCY VALUES Risk/mg/kg–d INHALATION		ORAL		WEIGHT OF EVIDENCE INHAL		ORAL	
85687	BUTYL BENZYL PHTHALATE	9	ND		2.0E–01		N							C	
7440484	COBALT	9													
629970	DOCOSANE, n–	8													
593453	OCTADECANE, n–	8													
205992	BENZO(B)FLUORANTHENE	7													
510156	CHLOROBENZILATE	7			2.0E–02		Y								
25155151	CYMENE, p–	7													
65850	BENZOIC ACID	6			4.0E+00		Y					D		D	
319857	BHC, beta–	6													
95487	CRESOL, o–	6			5.0E–02		Y					C		C	
72208	ENDRIN	6			3.0E–04		Y					D		D	
206440	FLUORANTHENE	6			4.0E–02		Y					D		D	
207089	BENZO(K)FLUORANTHENE	5													
106478	CHLOROANILINE, p–	5			4.0E–03		Y								
218019	CHRYSENE	5													
60571	DIELDRIN	5			3.00E–05			1.60E+01		1.70E+01				B2	
84742	DI–N–BUTYL PHTHALATE	5	ND		1.0E–01										
591786	HEXANONE, 2–	5	1.4E–03	d											
129000	PYRENE	5			3.0E–02		Y								
75694	TRICHLOROFLUOROMETHANE	5	2.0E–01	e	3.0E–01		N					D		D	
56553	BENZ(A)ANTHRACENE	4													
100414	ETHYLBENZENE	4	2.9E–01		1.0E–01		Y								
1024573	HEPTACHLOR EPOXIDE	4			1.3E–05			9.1E+00		9.1E+00		B2		B2	
56235	TETRACHLOROMETHANE	4			7.0E–04			5.2E–02		1.3E–01		B2		B2	
1330207	XYLENE, m–	4			2.0E+00	e	N					D		D	
309002	ALDRIN	3			3.00E–05			1.60E+01		1.70E+01				B2	
50328	BENZO(A)PYRENE	3						6.10E+00		7.30E+00				B2	
2921882	CHLORPYRIFOS	3			3.0E–03		Y								
78831	ISOBUTYL ALCOHOL	3			3.0E–01		Y								
1836755	NITROFEN (TOK)	3													
100425	STYRENE	3	ND		2.0E–01			2.0E–03	e	3.0E–02	e	B2	e	B2	e
127184	TETRACHLOROETHENE	3			1.0E–02			5.2E–07	e	5.1E–02	e	B2	e	B2	e
1582098	TRIFLURALIN (TREFLAN)	3	ND		7.5E–03					7.7E–03		C		C	
1330207	XYLENE, o– and p–	3			2.0E+00		N					D		D	
98862	ACETOPHENONE	2	5.7E–06	e	1.0E–01		Y					D		D	
120127	ANTHRACENE	2	ND		3.0E–01		Y								
86500	AZINPHOS METHYL	2			7.5E–03	d									
319846	BHC, alpha–	2													
319868	BHC, delta–	2													
108907	CHLOROBENZENE	2	5.7E–03	e	2.0E–02		Y			1.7E+00	d	B2			

EXHIBIT A-1 (cont'd)

Available Human Toxicity Data for 72 Chemicals Detected Less Than 10 Percent of the Time[a]

CAS	Chemical Name	Percent Detect	REFERENCE DOSES VALUES mg/kg-d[b] INHALATION		ORAL		Mut.?[c]	CANCER POTENCY VALUES Risk/mg/kg-d INHALATION	ORAL	WEIGHT OF EVIDENCE INHAL		ORAL	
50293	DDT	2			5.00E-04			3.40E-01	3.40E-01				
333415	DIAZINON	2			9.0E-04	e	Y						
106467	DICHLOROBENZENE, 1,4-	2	2.0E-01	e	7.0E-01	e			2.4E-02	e	C	e	C	e
1464535	DIEPOXYBUTANE, 1,2:3,4-	2												
123911	DIOXANE, 1,4-	2												
300765	NALED (DIBROM)	2							1.1E-02		B2		B2	
91203	NAPHTHALENE	2			2.0E-03		Y							
108101	PENTANONE, 4-METHYL-2-	2	ND		4.0E-02	e	Y							
85018	PHENANTHRENE	2	2.0E-02	e			N							
	PHOSPHORIC ACID, TRI-O-TOLYL ESTER	2									D		D	
2104645	SANTOX (EPN)	2												
98555	TERPINEOL, alpha-	2			1.0E-05	d								
107493	TETRAETHYLPYROPHOSPHATE	2												
100516	BENZYL ALCOHOL	2												
92524	BIPHENYL	1			3.0E-01	e	Y							
67663	CHLOROFORM	1	ND		5.0E-02		Y							
91587	CHLORONAPHTHALENE, 2-	1	ND		1.0E-02			8.1E-02	6.1E-03		B2		B2	
	DDE	1			8.0E-02		N							
132649	DIBENZOFURAN	1			5.00E-04			3.40E-01	3.40E-01					
540590	DICHLOROETHENE, trans-1,2-	1									D		D	
60515	DIMETHOATE	1			2.0E-02		Y							
131113	DIMETHYL PHTHALATE	1			2.0E-04		Y							
117817	DI-N-OCTYL PHTHALATE	1	ND		1.0E+01	e	Y							
959988	ENDOSULFAN-I	1			2.0E-02				1.4E-02				B2	
91576	METHYLNAPHTHALENE, 2-	1			5.0E-05		N							
86306	N-NITROSODIPHENYLAMINE	1												
13171216	PHOSPHAMIDON	1							4.9E-03		B2	e	B2	
109068	PICOLINE, 2-	1			2.0E-04	d								
79016	TRICHLOROETHENE	1			5.0E-04	d	Y							
								1.70E-02	1.10E-02				B2	

[a] Toxicity data from IRIS (1992) unless otherwise noted.

[b] Inhalation Risk Reference Concentrations converted to units of (mg/kg-d) by assuming a person weighs 70 kg and breathes 20 m^3 air per day.

[c] Mutagenic activity as reported in RTECS (April 1993); Y = yes, N = no.

[d] Office of Pesticide Programs RfD Tracking Report (27 January 1992), and List of Chemicals Evaluated for Carcinogenic Potential (Health Effects Division, 27 February 1992).

[e] HEAST (data from 1992 and 1991 Tables were used, with 1992 data used preferentially over 1991 data.)

EXHIBIT A–2

Agricultural Pathway 3 Analysis for 43 Chemicals with Oral RfD or q1* Values

CAS	Chemical Name	98th Conc. ND = ML	Percent Detect	REFERENCE DOSES VALUES mg/kg–d[b] INHALATION	REFERENCE DOSES VALUES mg/kg–d[b] ORAL	CANCER POTENCY VALUES Risk/mg/kg–d INHALATION	CANCER POTENCY VALUES Risk/mg/kg–d ORAL	EXPOSURE (mg/kg–day)	EXP/RfD (fraction)	Individual Cancer Risk
85687	BUTYL BENZYL PHTHALATE	42801.58	9	ND	2.0E–01			5E–04	< 0.01	
510156	CHLOROBENZILATE	110.58	7		2.0E–02			1E–06	< 0.01	
65850	BENZOIC ACID	263587.8	6		4.0E+00			3E–03	< 0.01	
95487	CRESOL, o–	271592.5	6		5.0E–02			3E–03	0.07	
72208	ENDRIN	84.95	6		3.0E–04			1E–06	< 0.01	
206440	FLUORANTHENE	42801.58	6		4.0E–02			5E–04	0.01	
106478	CHLOROANILINE, p–	42801.58	5		4.0E–03			5E–04	0.1	
84742	DI–N–BUTYL PHTHALATE	68172.03	5	ND	1.0E–01			9E–04	< 0.01	
129000	PYRENE	42801.58	5		3.0E–02			5E–04	0.02	
75694	TRICHLOROFLUOROMETHANE	5000.62	5	2.0E–01	3.0E–01			6E–05	< 0.01	
100414	ETHYLBENZENE	5012.94	4	2.9E–01	1.0E–01			6E–05	< 0.01	
1024573	HEPTACHLOR EPOXIDE	108.71	4		1.3E–05	9.1E+00	9.1E+00	1E–06	0.1	9E–07
56235	TETRACHLOROMETHANE	5000.62	4		7.0E–04	5.2E–02	1.3E–01	6E–05	0.09	6E–07
1330207	XYLENE, m–	5064.36	4		2.0E+00			6E–05	< 0.01	
2921882	CHLORPYRIFOS	343.87	3		3.0E–03			4E–06	< 0.01	
78831	ISOBUTYL ALCOHOL	5000.62	3		3.0E–01			6E–05	< 0.01	
100425	STYRENE	54538.92	3	ND	2.0E–01	2.0E–03	3.0E–02	7E–04	< 0.01	1E–06
127184	TETRACHLOROETHENE	5000.62	3		1.0E–02	5.2E–07	5.1E–02	6E–05	< 0.01	2E–07
1582098	TRIFLURALIN (TREFLAN)	271.62	3	ND	7.5E–03		7.7E–03	3E–06	< 0.01	2E–09
1330207	XYLENE, o– and p–	5000.62	3		2.0E+00			6E–05	< 0.01	
98862	ACETOPHENONE	42801.58	2	5.7E–06	1.0E–01			5E–04	< 0.01	
120127	ANTHRACENE	42801.58	2	ND	3.0E–01			5E–04	< 0.01	
86500	AZINPHOS METHYL	645.97	2		7.5E–03			8E–06	< 0.01	
319868	BHC, delta–	136.65	2				1.7E+00	2E–06		2E–07
108907	CHLOROBENZENE	5012.94	2	5.7E–03	2.0E–02			6E–05	< 0.01	
333415	DIAZINON	258.39	2		9.0E–04			3E–06	< 0.01	
106467	DICHLOROBENZENE, 1,4–	42801.58	2	2.0E–01	7.0E–01		2.4E–02	5E–04	< 0.01	9E–07
123911	DIOXANE, 1,4–	5064.36	2				1.1E–02	6E–05		5E–08
300765	NALED (DIBROM)	1756.88	2		2.0E–03			2E–05	0.01	
91203	NAPHTHALENE	42801.58	2	ND	4.0E–02			5E–04	0.01	
2104645	SANTOX (EPN)	258.81	2		1.0E–05			3E–06	0.3	
100516	BENZYL ALCOHOL	42809.81	1		3.0E–01			5E–04	< 0.01	
92524	BIPHENYL	43256.26	1	ND	5.0E–02			5E–04	0.01	
67663	CHLOROFORM	5052.05	1	ND	1.0E–02	8.1E–02	6.1E–03	6E–05	< 0.01	3E–08
91587	CHLORONAPHTHALENE, 2–	42801.58	1		8.0E–02			5E–04	< 0.01	
540590	DICHLOROETHENE, trans–1,2–	5052.05	1		2.0E–02			6E–05	< 0.01	
60515	DIMETHOATE	260.16	1		2.0E–04			3E–06	0.02	
131113	DIMETHYL PHTHALATE	42801.58	1	ND	1.0E+01			5E–04	< 0.01	
117817	DI–N–OCTYL PHTHALATE	42801.58	1		2.0E–02		1.4E–02	5E–04	0.03	5E–07
959988	ENDOSULFAN–I	271.62	1		5.0E–05			3E–06	0.07	
86306	N–NITROSODIPHENYLAMINE	85603.17	1				4.9E–03	1E–03		4E–07
13171216	PHOSPHAMIDON	1550.13	1		2.0E–04			2E–05	0.10	
109068	PICOLINE, 2–	214049.0	1		5.0E–04			3E–03	5	

Screening Analysis Based on Cancer Potency Slopes

The second screening analysis EPA conducted consisted of identifying those pollutants with relatively high cancer potency slopes. As shown in Exhibit A-1, four pollutants, aldrin, dieldrin, heptachlor epoxide, and benzo(a)pyrene, have relatively large q_1^* values. These pollutants were evaluated further.

Although aldrin and dieldrin are both insecticides, they are often evaluated together, as aldrin/dieldrin, because dieldrin is an environmental degradation product of aldrin. In addition, aldrin and dieldrin have the same human health toxicity values. In Round One, aldrin/dieldrin were evaluated for Pathways 1 through 11, but not 12, 13, or 14.

Given the $\log(K_{ow})$ value for dieldrin is greater than five, aldrin/dieldrin might pose an unacceptable risk by sorbing to particles that subsequently erode and enter a stream. Aldrin/dieldrin is not expected to leach significantly to groundwater, given the high $\log(K_{ow})$ value. However, aldrin/dieldrin might also pose an unacceptable risk through volatilization. Therefore, EPA evaluated risks from Pathway 12 and Pathway 13 for aldrin/dieldrin using the assumptions and equations presented in Sections 4.2.12 and 4.2.13, respectively.

To correspond to the methods used in the Comprehensive Hazard Identification exercise, the 95th percentile pollutant concentrations with the non-detect values set equal to the minimum detection level were used. The pollutant-specific data for both pathways are presented in Exhibit A-3.

EXHIBIT A-3
Pollutant-Specific Data Required for Pathways 12 and 13

Parameter	Aldrin/Dieldrin
95th percentile concentration (mg/kg)	0.0486[1]
log (K_{ow})	5.48[2]
K_d (L/kg)	1173[3]
Henry's Law constant (atm-m^3/mol)	1.1×10^{-5} (2)
K_{deg} (yr^{-1})	0[4]
Diffusivity in Air (cm^2/sec)	4×10^{-2} (3)
BCF (L/kg)	3400[3]
FM (dimensionless)	10

[1] Composite aldrin/dieldrin concentration from 1988 NSSS.
[2] Schwarzenbach et al., 1993.
[3] Calculated using equations in Section 4.2.12.
[4] Howard, 1991.

Results of the analysis are presented in Exhibit A-4 for Pathway 12 and Exhibit A-5 for Pathway 13. For Pathway 12, the individual cancer risks range from $7x10^{-9}$ for reclamation sites to $2x10^{-8}$ for other land application sites. For Pathway 13, individual cancer risks range from $9x10^{-8}$ for agricultural land to $1x10^{-6}$ for reclamation sites.

EXHIBIT A-4
Individual Cancer Risks for Aldrin/Dieldrin from Pathway 12

Agricultural Land	Forest	Reclamation Site	Public Contact Site
$2x10^{-8}$	$2x10^{-8}$	$7x10^{-9}$	$2x10^{-8}$

EXHIBIT A-5
Individual Cancer Risks for Aldrin/Dieldrin from Pathway 13

Agricultural Land	Forest	Reclamation Site	Public Contact Site
$9x10^{-8}$	$4x10^{-7}$	$1x10^{-6}$	$2x10^{-7}$

For heptachlor epoxide, the individual risk for a child directly ingesting sewage sludge (Pathway 3) was calculated above to be 9×10^{-7} (Exhibit A-2). Given the low magnitude of the risk, this pollutant was not evaluated further.

Benzo(a)pyrene was fully evaluated for all land application pathways in Round One except Pathway 11 (tractor driver). Benzo(a)pyrene cannot be considered further in Round Two for Pathway 11, however, because there is not a Threshold Limit Value for this pollutant to be evaluated under Pathway 11.

APPENDIX B

STATISTICAL ANALYSES OF THE NATIONAL SEWAGE SLUDGE SURVEY DATA

Final Report:

Percentile Estimates Used to Develop the List of Pollutants for Round Two of the Part 503 Regulation

Submitted to:

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Office of Science and Technology
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401 M Street, SW. (4303)
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EPA Contract No. 68-C4-0046
SAIC Project No. 01-0813-07-5046-010

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I. INTRODUCTION

In February, 1993, the Environmental Protection Agency (EPA) promulgated limits for nine toxic pollutants in sewage sludge. These limits which were issued by EPA under the authority of section 405(d) Clean Water Act, as amended, are referred to as the "Round One" sewage sludge regulation. In May, 1993, the EPA submitted to the court a list of 31 candidate pollutants for "Round Two" regulations. This report presents percentile estimates used to develop the list of pollutants for Round Two of the Part 503 Regulation. All elements, compounds, or solids physically measured will be referred to in this report as pollutants. The term **pollutant** is used here to mean only that a substance, in certain quantities, could cause harm to human health or the environment; not that it will cause harm to human health or the environment.

Data analyzed to produce these pollutant concentration percentile estimates are from the EPA's 1988 National Sewage Sludge Survey (NSSS). Section II briefly describes the NSSS. Data conventions are presented in Section III. Section IV provides the statistical methodology employed to produce the percentile estimates. And finally, Section V presents tabulated percentile estimates.

II. EPA's 1988 NATIONAL SEWAGE SLUDGE SURVEY

To support Round One and Two regulatory development efforts, the EPA's 1988 NSSS collected sewage sludge quality and pollutant occurrence data from a national probability sample of Publicly Owned Treatment Works (POTWs) practicing at least secondary treatment of wastewater. Operationally, secondary treatment was defined as a primary clarifier process followed by biological treatment and secondary clarification. In 1988, 11,407 POTWs in the 50 States, Puerto Rico, and the District of Columbia met this criteria.

A statistical probability sample of 208 POTWs in the contiguous states and the District of Columbia comprised the analytical component of the 1988 NSSS. These POTWs were randomly drawn from secondary or higher treatment POTWs which were categorized into one of four strata based on their average daily flow rate. These strata are defined as follows:

1) Flow greater than 100 million gallons per day (MGD)
2) Flow more than 10 MGD but less than or equal to 100 MGD
3) Flow more than 1 MGD but less than or equal to 10 MGD
4) Flow less than or equal to 1 MGD.

EPA contract personnel collected sewage sludge samples from 180 POTWs in the analytical component of the NSSS. Samples were collected just prior to use or disposal of the sewage sludge. All sample collection and preservation was conducted according to protocol. Contract laboratories analyzed each sewage sludge sample for 412 analytes. EPA adapted analytical methods 1624 and 1625 to allow volatile and semi-volatile organic analytes to be quantified from the sewage sludge matrix. Pesticides and polychlorinated biphenyls (PCBs) were quantified

according to EPA method 1618; method 1613 measured dibenzofurans and dioxins; metals, other inorganics, and classicals were quantified according to standard EPA methods. All chemical analysis methods were either developed, chosen, or adapted to allow for the most reliable and accurate measurement of the 412 analytes in the sewage sludge matrix.

A more detailed discussion the NSSS sampling plan, POTWs, and data is included in a November, 1992 final report entitled "Statistical Support Documentation for the 40 CFR, Part 503 Final Standards for the Use or Disposal of Sewage Sludge."

III. DATA CONVENTIONS

A total of 208 POTWs were selected for sampling as part of the analytical component of the 1988 NSSS. However, 32 POTWs were excluded from the statistical analyses because sewage sludge samples were not obtained after the completion of secondary treatment of wastewater. POTWs that were selected for the NSSS but excluded from the statistical analyses are listed on Table 1. The EPISODE number listed on Table 1 designates the POTW's identification number in the analytical survey. An episode number of "0" indicates that the POTW was selected for sampling as part of the analytical probability sample but samples of sewage sludge were not collected.

The reported national pollutant concentration estimates were calculated from a sample of 176 POTWs. These estimates apply to a population of 7,750 POTWs that practiced at least secondary treatment of wastewater during 1988. Pesticides were not quantified for SurveyID 35-38-348 (Episode=1565.) Therefore, estimates for pesticides reported on the tables result from a sample of 175 POTWs and are projected to a population of 7,720 POTWs in the Nation. Sewage sludge samples from SurveyIDs 23-07-036 (Episode=1554) and 35-05-012 (Episode=1561) were not analyzed for the dioxin/furans. Therefore the dioxin estimates, generated from a sample of 174 POTWs, apply to a population of 7,714 POTWs. Adjusted stratum weights for each sample size are tabulated below.

ADJUSTED WEIGHTS for STRATA (w_i) by Sample Size

STRATUM	Sample size = 174	Sample size = 175	Sample size = 176
1	27/7,714	27/7,720	27/7,750
2	301/7,714	307/7,720	307/7,750
3	1,838/7,714	1,838/7,720	1,868/7,750
4	5,548/7,714	5,548/7,720	5,548/7,750

In the NSSS, if a pollutant was measured above the Minimum Level, as adjusted for interferences, then the measure is considered a detection. In the August, 1989 document titled "Analytical Methods for the National Sewage Sludge Survey," the EPA's Industrial Technology Division defines a Minimum Level for pollutants quantified by gas chromatography combined with mass spectrometry (GCMS) as the level at which " the entire analytical system shall give recognizable mass spectra and acceptable calibration points." For elemental pollutants, the Minimum Level is defined as "the minimum concentration of substance that can be measured and reported in 99% confidence that the value is above zero." The final report for Round One Part 503 regulations refers to the Minimum Level as "roughly equivalent to the minimum concentration or amount of pollutant that could be measured."

If a pollutant was not measured above the Minimum Level, then estimates were generated using two substitution methods. One set of estimate were produced using the value of the Minimum Level for those samples for which the pollutant was considered to be an non-detect. The second set of estimates substituted zero for pollutant concentration value for those samples from which a pollutant was not quantified above the Minimum Level. Tabulated results identify the substitution method employed for the reported set of estimates.

Prior to calculating the estimates, pollutant concentrations were aggregated on a POTW basis to form one concentration value per POTW for each pollutant. Field duplicate samples were averaged together. For POTWs with multiple treatment trains, sample measurements on pollutant concentrations were averaged together, using a weighted average based on the dry weight of sewage sludge disposed by the treatment train associated with each sample. Primary samples were dropped from analysis. The quality of these primary sludges is different than that associated with secondary treatment. Because the percent solids in sampled sewage sludge ranged from less than one percent to 100 percent, reported pollutant concentrations were transformed to dry weight units as a function of the sample's percent solids. This transformation allows a standardized basis for evaluating pollutant loads. Implicit is the assumption that pollutants are associated with the solid phase of sewage sludge.

The dioxins and furans are reported individually and in aggregate. Aggregates were mathematically created using the following method. After each congener was aggregated on a POTW basis, the concentration of the congener for each POTW was multiplied by the congener's corresponding 1989 NATO Toxicity Equivalence Factor (TEF.) The TEF's which are based on the toxicity of 2,3,7,8 TCDD are listed in Table 2 . The resulting TEF adjusted values were then summed over all congeners for each POTW to create the POTW composite dioxin concentration. Two conventions were used to determine if composite dioxin was a detect for a POTW. For the first convention, the composite dioxin was considered a detect if all of the individual congeners were detected above the Minimum Level. The estimate for this method is designated on the tables as "dioxin[a]." For the second method of determining a detection for the composite dioxin, the composite dioxin was considered a detect if at least one of the individual congeners was detected above the Minimum Level. This estimate is designated "dioxin[b]." TEF adjusted estimates of the individual congeners appear in Section V.

PCBs were also mathematically aggregated. These aggregates were generated as described above with the exception that the individual PCB's were not multiplied by a toxicity equivalence factor.

IV. STATISTICAL METHODS

Percentile estimates were calculated using the nonparametric, weighted cumulative distribution function (CDF) technique. Denote the dry weight concentration of a given pollutant in the sampled sewage sludge from the j^{th} POTW in the i^{th} survey flow stratum as X_{ij}. The values of the variable X_{ij} were then sorted in order of increasing concentration. The values of the adjusted survey weights (w_i) associated with the ordered values of X_{ij} are then summed until the first occurrence of p.

If X_p is defined as the concentration of the p^{th} percentile then,

$$X_p = F(X) \geq p \text{ where } F(X) = \sum_{i=1}^{4} w_i F_i(X)$$

with

$$F_i = \sum_{j=1}^{n_{i1}} \frac{I(X_{ij} \leq x)}{n_{ij}}$$

and $I(X_{ij} \leq x) = 1$ if $X_{ij} \leq x$ for $x \geq 0$
$= 0$ otherwise.

To determine the pollutant concentration associated with the p^{th} percentile, an inverse function was applied to the cumulative distribution function. Define the p^{th} percentile as $F_X(x_p)$ such that $P[X \leq x_p] \geq p/100$. The inverse of this function $F^{-1}(p)$, is the smallest value of x satisfying $F_X(x) \geq p$ where p is the desired percentile point (P) divided by 100.

Because the cumulative distribution created by application of the formula in the previous section is empirical, integer valued percentile points are not always realized in the data. The convention applied to determine the concentration associated with the p^{th} integer percentile from the empirical distribution function was to determine the smallest concentration value x such that $F_X(x) > p$. This value was denoted x_{p+} and is the q^{th} ordered concentration. The next smallest concentration from x_{p+}, or the concentration associated with the (q-1)st ordered concentration was then defined. The concentration value for the p^{th} percentile was obtained using linear interpolation between the q^{th} and (q-1) values.

Nonparametric estimates of pollutant concentration means and standard deviations are also reported in the tables. Retaining the definition of X_{ij} as the dry weight concentration of a given

pollutant in the sampled sewage sludge from the j^{th} POTW in the i^{th} survey stratum and w_i as the adjusted survey weight for the i^{th} stratum, then the mean pollutant concentration was estimated as listed on the next page.

$$E(X) = \sum_{i=1}^{4} w_i \frac{\sum_{j=1}^{n_{ij}} X_{ij}}{n_{ij}}.$$

The pollutant concentration standard deviation was estimated as the square root of the method of moments estimator of the variance. That is:

$$V(X)^{1/2} = \left[\sum_{i=1}^{4} w_i \sum_{j=1}^{n_{ij}} \frac{X_{ij}^{2}}{n_{ij}} - [E(X)]^2\right]^{1/2}.$$

V. POLLUTANT CONCENTRATION PERCENTILE ESTIMATES

Tables 3 and 4 present pollutant concentration percentile estimates for pollutants from the 1988 National Sewage Sludge Survey (NSSS.) Taking into account the individual dioxin and furan congeners and the PCB aroclors, Tables 3 and 4 present concentration estimates for 353 pollutants. The listing of pollutants is ordered by percent detection. The ordering is from highest to lowest detection rates in the nation. Excluded from this listing are the metals regulated in Round One, and the 42 semiquantitative metals listed on Table 5. Of the 42 semiquantitative metals, 36 had no quantitative measurements recorded in the NSSS database. Of the remaining six, potassium and iodine had one recorded measure while silicon, strontium, and sulfur had measurements recorded for two samples. All other samples were missing measurements. This precluded estimation of pollutant concentrations. Estimates of phosphorus concentrations were generated from data collected using colorimetric method 365.2 as reported in EPA's August, 1989 "Analytical Method for the National Sewage Sludge Survey."

For each pollutant, the tables report the following: pollutant type, unit of measure, sample size, an estimate of the national percent detect, mean, standard deviation, the observed maximum, and the 99^{th}, 98^{th}, 95^{th}, 90^{th} and median percentiles estimated from empirical national, cumulative distributions of pollutant concentrations. The column labeled "Sample size" records the number of POTWs in the NSSS from which data were used to generate the reported estimates.

Table 3 is subtitled "Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level." The nonparametric estimation procedure is that described in Section IV.

The substitution of nondetects set to Minimum Levels indicates that Minimum Level of a pollutant was used in the estimation procedure for those samples that were not quantified above the pollutant's Minimum Level of detection. Estimates on Table 4 were generated using the value zero for samples from which a pollutant was not quantified above the Minimum Level.

Tables 3 and 4 indicate that there are 45 tested pollutants detected at an estimated national rate of ten percent or higher from sewage sludge resulting from secondary or higher treatment of wastewater in 1988. EPA used this list of pollutants in conjunction with human health and ecological toxicity data to select the 31 candidate pollutants for Round Two regulation.

TABLE 1.

LISTING OF POTWS EXCLUDED FROM PERCENTILE ESTIMATION

SURVEYID	EPISODE	REASON	FLOW STRATUM
12-49-455	0	Ineligible/Out of Business	1
21-25-234	0	Not sampled	2
25-38-345	0	Ineligible/Out of Business	2
25-50-472	1386	Only primary sludge sampled	2
31-18-140	1477	Data not entered into database	3
31-23-206	1398	Only primary sludge sampled	3
41-24-215	0	Not sampled	4
41-36-312	0	Not sampled	4
45-02-005	0	Wastewater Stabilization pond (WWSP)	4
45-11-064	0	WWSP	4
45-13-083	0	WWSP	4
45-13-089	0	WWSP	4
45-14-092	0	WWSP	4
45-15-112	0	WWSP	4
45-16-130	0	WWSP	4
45-17-131	0	WWSP	4
45-19-154	0	WWSP	4
45-23-208	0	WWSP	4
45-24-220	0	WWSP	4
45-25-229	0	Ineligible/Out of Business	4
45-25-231	0	WWSP	4
45-26-237	0	WWSP	4

45-28-246	0	WWSP	4
45-29-248	0	WWSP	4
45-30-253	0	WWSP	4
45-37-339	0	Not sampled	4
45-42-387	0	Ineligible/Out of Business	4
45-42-392	1488	Ineligible/Out of Business	4
45-45-415	0	WWSP	4
45-45-423	0	Not sampled	4
45-50-463	0	Not sampled	4
45-50-474	0	WWSP	4

TABLE 2
1989 NATO TOXICITY EQUIVALENCY FACTORS

CONGENER	TEF
Octachlorodibenzo-p-dioxin	0.001
Octachlorodibenzofuran	0.001
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	0.010
1,2,3,4,6,7,8-Heptachlorodibenzofuran	0.010
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	0.100
1,2,3,4,7,8-Hexachlorodibenzofuran	0.100
1,2,3,4,7,8,9-Heptachlorodibenzofuran	0.010
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	0.100
1,2,3,6,7,8-Hexachlorodibenzofuran	0.100
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	0.500
1,2,3,7,8-Pentachlorodibenzofuran	0.050
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	0.100
1,2,3,7,8,9-Hexachlorodibenzofuran	0.100
2,3,4,6,7,8-Hexachlorodibenzofuran	0.100
2,3,4,7,8-Pentachlorodibenzofuran	0.500
2,3,7,8-Tetrachlorodibenzo-p-dioxin	1.000
2,3,7,8,-Tetrachlorodibenzofuran	0.100

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CALCIUM	Metals	MG/KG	176	7750	100	35,300.00	46,500.00	333,000.00	313,000.00	151,000.00	84,600.00	66,600.00	26,200.00
IRON	Metals	MG/KG	176	7750	100	18,000.00	21,100.00	229,000.00	81,400.00	79,900.00	59,600.00	41,700.00	12,400.00
MAGNESIUM	Metals	MG/KG	176	7750	100	7,460.00	17,600.00	134,000.00	75,500.00	30,700.00	17,500.00	9,750.00	4,270.00
PHOSPHORUS	Metals	MG/KG	176	7750	100	4,580.00	9,870.00	71,700.00	42,600.00	17,600.00	13,400.00	12,100.00	1,330.00
SODIUM	Metals	MG/KG	176	7750	100	7,970.00	19,600.00	141,000.00	103,000.00	49,100.00	21,400.00	17,200.00	2,070.00
ALUMINUM	Metals	MG/KG	176	7750	100	11,200.00	14,600.00	92,700.00	85,300.00	70,200.00	36,400.00	19,400.00	7,550.00
BARIUM	Metals	MG/KG	176	7750	100	673.00	640.00	5,570.00	3,000.00	2,370.00	1,730.00	1,230.00	499.00
dioxins b	Dioxins	NG/KG	174	7714	100	90.40	172.00	1,820.00	556.00	532.00	311.00	218.00	50.40
MANGANESE	Metals	MG/KG	176	7750	100	538.00	1,040.00	13,200.00	4,060.00	3,720.00	1,620.00	929.00	276.00
OCTACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	100	11.70	59.20	905.00	127.00	116.00	33.70	15.90	3.32
TITANIUM	Metals	MG/KG	176	7750	98	109.00	140.00	2,360.00	570.00	416.00	363.00	237.00	68.50
1,2,3,4,6,7,8-HEPTACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	98	9.52	37.00	525.00	103.00	73.70	27.90	14.00	3.35
NITRATE	Classicals	MG/KG	176	7750	95	1,420.00	5,040.00	35,300.00	26,500.00	15,500.00	5,020.00	1,690.00	96.50
SILVER	Metals	MG/KG	176	7750	84	48.20	112.00	852.00	546.00	218.00	128.00	75.80	25.50
TIN	Metals	MG/KG	176	7750	84	108.00	278.00	2,100.00	1,370.00	715.00	138.00	130.00	51.60
NITRITE	Classicals	MG/KG	176	7750	83	201.00	1,210.00	17,700.00	2,920.00	2,910.00	462.00	215.00	12.90
OCTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	80	0.67	4.38	69.50	5.04	4.75	3.31	0.55	0.11
1,2,3,4,6,7,8-HEPTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	71	2.31	7.14	71.00	31.50	27.60	5.72	3.09	0.57
2,3,7,8-TETRACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	65	2.50	3.32	33.70	14.80	14.70	7.69	4.70	1.70
FLUORIDE	Classicals	MG/KG	176	7750	63	126.00	233.00	1,510.00	1,260.00	1,040.00	411.00	319.00	44.20

(b) Composites considered a detect if at least one congener is measured above the minimum level.

* Dioxins and Furans Reported as Toxic Equivalents

Note: Three significant figures are reported

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
BIS(2-ETHYLHEXYL) PHTHALATE	Organics	UG/KG	176	7750	62	55,800.00	106,000.00	891,000.00	525,000.00	459,000.00	191,000.00	148,000.00	17,200.00
VANADIUM	Metals	MG/KG	176	7750	62	25.30	43.70	570.00	181.00	130.00	64.10	54.20	14.90
YTTRIUM	Metals	MG/KG	176	7750	61	6.62	5.77	52.50	21.40	21.00	20.00	14.10	5.20
TOLUENE	Organics	UG/KG	176	7750	61	41,300.00	116,000.00	1,180,000.00	575,000.00	439,000.00	238,000.00	93,400.00	1,660.00
2-PROPANONE	Organics	UG/KG	176	7750	58	64,300.00	323,000.00	2,430,000.00	1,460,000.00	736,000.00	116,000.00	65,200.00	4,710.00
1,2,3,6,7,8-HEXACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	49	5.20	10.60	73.70	58.50	47.70	13.10	9.43	2.73
BORON	Metals	MG/KG	176	7750	48	34.10	65.30	310.00	304.00	293.00	182.00	68.70	13.20
P-CRESOL	Organics	UG/KG	176	7750	43	52,300.00	110,000.00	1,160,000.00	542,000.00	430,000.00	306,000.00	202,000.00	11,000.00
1,2,3,4,7,8-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	43	7.08	20.90	150.00	109.00	73.20	15.40	11.00	2.80
METHYLENE CHLORIDE	Organics	UG/KG	176	7750	42	8,560.00	37,900.00	262,000.00	184,000.00	114,000.00	31,300.00	3,060.00	553.00
1,2,3,7,8,9-HEXACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	39	4.12	6.56	73.70	22.20	19.50	13.50	8.58	2.80
ANTIMONY	Metals	MG/KG	176	7750	38	6.47	20.00	660.00	27.30	25.50	24.00	13.80	3.20
CYANIDES (SOLUBLE SALTS AND COMPLEXES)	Classicals	MG/KG	176	7750	37	35.20	58.00	372.00	295.00	155.00	130.00	66.40	11.50
HEXANOIC ACID	Organics	UG/KG	176	7750	35	48,000.00	93,600.00	1,960,000.00	406,000.00	280,000.00	202,000.00	125,000.00	13,100.00
1,2,3,6,7,8-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	35	3.43	5.84	73.70	16.90	15.00	11.70	7.71	1.80
2-BUTANONE	Organics	UG/KG	176	7750	34	25,500.00	96,900.00	1,540,000.00	553,000.00	544,000.00	69,300.00	38,400.00	2,550.00
PHENOL	Organics	UG/KG	176	7750	34	19,700.00	40,400.00	920,000.00	177,000.00	158,000.00	57,500.00	37,600.00	10,200.00
2,4,5-TRICHLOROPHENOXYACETIC ACID	Pesticides	UG/KG	176	7750	29	21.00	134.00	2,170.00	73.90	67.10	50.50	28.30	6.88
2,3,4,6,7,8-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	27	3.30	5.60	73.70	15.70	14.00	9.69	6.94	1.80
1,2,3,4,7,8,9-HEPTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	26	0.39	0.73	6.42	2.14	1.48	1.21	0.78	0.23

Note: Three significant figures are reported

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2,3,4,7,8-PENTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	26	11.30	27.40	368.00	74.50	68.00	46.50	37.90	5.22
1,2,3,4,7,8-HEXACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	25	3.73	6.07	73.70	19.00	19.00	13.60	7.87	2.25
BERYLLIUM	Metals	MG/KG	176	7750	22	1.84	2.43	21.00	8.56	8.33	6.00	5.00	0.86
1,2,3,7,8-PENTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	22	1.04	2.63	36.80	7.15	6.06	4.22	1.65	0.48
PCB b	Pesticides	UG/KG	175	7720	19	2,120.00	2,460.00	14,700.00	13,400.00	12,300.00	5,400.00	3,050.00	1,490.00
1,2,3,7,8-PENTACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	18	9.97	25.70	368.00	66.70	60.10	39.00	16.30	4.92
ACETIC ACID (2,4-DICHLOROPHENOXY)	Pesticides	UG/KG	176	7750	16	10.40	17.60	422.00	66.40	65.60	30.00	16.00	6.06
1,2,3,7,8,9-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	16	3.56	8.69	126.00	24.10	16.90	9.16	6.92	1.60
2,3,7,8-TETRACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	16	10.60	12.60	116.00	42.30	33.60	26.70	24.00	6.66
N-HEXACOSANE	Organics	UG/KG	176	7750	15	11,600.00	32,600.00	498,000.00	58,600.00	43,300.00	33,300.00	29,600.00	4,760.00
N-TETRACOSANE	Organics	UG/KG	176	7750	15	14,900.00	44,200.00	650,000.00	101,000.00	87,200.00	42,600.00	31,200.00	5,260.00
2,4,5-TRICHLOROPHENOXYPROPIONIC ACID	Pesticides	UG/KG	176	7750	15	10.90	14.50	121.00	66.20	60.10	40.00	17.30	6.26
THALLIUM	Metals	MG/KG	176	7750	15	5.20	16.90	210.00	62.80	29.50	10.60	6.54	2.00
N-DODECANE	Organics	UG/KG	176	7750	14	19,600.00	42,600.00	525,000.00	221,000.00	201,000.00	88,700.00	35,600.00	5,870.00
N-TETRADECANE	Organics	UG/KG	176	7750	14	14,100.00	28,600.00	352,000.00	94,500.00	93,500.00	60,300.00	32,900.00	5,260.00
N-TRIACONTANE	Organics	UG/KG	176	7750	14	14,600.00	42,600.00	597,000.00	156,000.00	154,000.00	37,000.00	31,100.00	4,930.00
N-EICOSANE	Organics	UG/KG	176	7750	13	12,400.00	19,400.00	198,000.00	93,300.00	52,700.00	43,100.00	31,300.00	5,260.00
N-HEXADECANE	Organics	UG/KG	176	7750	12	14,600.00	50,000.00	758,000.00	121,000.00	46,600.00	33,900.00	31,200.00	5,260.00
N-OCTACOSANE	Organics	UG/KG	176	7750	12	12,600.00	28,600.00	413,000.00	66,100.00	43,400.00	33,300.00	31,600.00	5,260.00
ENDOSULFAN-II	Pesticides	UG/KG	175	7720	12	19.00	29.20	184.00	175.00	128.00	66.70	41.10	10.00

(b) Composites considered a detect if at least one PCB aroclor is measured above the minimum level.

Note: Three significant figures are reported

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CARBON DISULFIDE	Organics	UG/KG	176	7750	10	1,080.00	3,260.00	31,800.00	19,200.00	5,400.00	3,130.00	2,080.00	406.00
PENTACHLORONITROBENZENE	Pesticides	UG/KG	175	7720	10	181.00	1,160.00	8,630.00	3,970.00	221.00	79.30	43.30	20.60
N-DECANE	Organics	UG/KG	176	7750	10	12,700.00	23,900.00	199,000.00	151,000.00	81,300.00	33,300.00	30,400.00	4,760.00
PCB-1260	Pesticides	UG/KG	175	7720	10	337.00	561.00	4,010.00	2,920.00	1,870.00	827.00	465.00	209.00
COBALT	Metals	MG/KG	176	7750	9	23.50	30.80	262.00	107.00	104.00	100.00	57.10	11.60
BUTYL BENZYL PHTHALATE	Organics	UG/KG	176	7750	9	9,860.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
PCB-1248	Pesticides	UG/KG	175	7720	9	275.00	311.00	5,200.00	1,660.00	1,500.00	673.00	426.00	209.00
N-DOCOSANE	Organics	UG/KG	176	7750	8	10,300.00	13,800.00	154,000.00	58,500.00	43,200.00	33,300.00	29,300.00	5,610.00
N-OCTADECANE	Organics	UG/KG	176	7750	8	11,300.00	16,500.00	154,000.00	93,100.00	51,600.00	33,300.00	30,300.00	5,260.00
PCB-1254	Pesticides	UG/KG	175	7720	8	501.00	1,260.00	9,350.00	5,490.00	2,520.00	1,570.00	826.00	209.00
CHLOROBENZILATE	Pesticides	UG/KG	175	7720	7	29.50	30.80	211.00	149.00	109.00	86.70	54.20	20.00
P-CYMENE	Organics	UG/KG	176	7750	7	10,900.00	14,600.00	154,000.00	66,100.00	43,200.00	33,600.00	30,600.00	5,260.00
BENZO(B)FLUORANTHENE	Organics	UG/KG	176	7750	6	9,830.00	13,500.00	154,000.00	46,700.00	43,000.00	33,000.00	29,000.00	4,760.00
BETA-BHC	Pesticides	UG/KG	175	7720	6	14.30	15.90	163.00	83.20	71.20	41.50	22.20	10.00
ENDRIN	Pesticides	UG/KG	175	7720	6	14.70	16.80	123.00	88.20	84.90	41.50	22.30	10.20
O-CRESOL	Organics	UG/KG	176	7750	6	16,500.00	46,300.00	329,000.00	327,000.00	257,000.00	42,600.00	29,300.00	4,760.00
FLUORANTHENE	Organics	UG/KG	176	7750	5	9,950.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
P-CHLOROANILINE	Organics	UG/KG	176	7750	5	9,640.00	13,600.00	154,000.00	46,700.00	43,000.00	33,300.00	28,800.00	4,760.00
PYRENE	Organics	UG/KG	176	7750	5	9,950.00	13,400.00	154,000.00	46,700.00	43,000.00	33,000.00	28,000.00	4,760.00
TRICHLOROFLUOROMETHANE	Organics	UG/KG	176	7750	5	1,000.00	2,940.00	31,800.00	7,210.00	5,110.00	3,470.00	2,060.00	357.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2-HEXANONE	Organics	UG/KG	176	7750	5	4,850.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
BENZO(A)ANTHRACENE	Organics	UG/KG	176	7750	4	9,760.00	13,400.00	154,000.00	46,700.00	43,000.00	33,000.00	28,000.00	4,760.00
BENZO(K)FLUORANTHENE	Organics	UG/KG	176	7750	4	9,790.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	28,000.00	4,760.00
BENZOIC ACID	Organics	UG/KG	176	7750	4	53,100.00	76,200.00	835,000.00	314,000.00	313,000.00	167,000.00	151,000.00	23,800.00
CHRYSENE	Organics	UG/KG	176	7750	4	9,750.00	13,400.00	154,000.00	46,700.00	43,000.00	33,100.00	28,000.00	4,760.00
DI-N-BUTYL PHTHALATE	Organics	UG/KG	176	7750	4	11,200.00	17,600.00	331,000.00	72,300.00	68,100.00	33,300.00	30,200.00	4,950.00
DIELDRIN	Pesticides	UG/KG	175	7720	4	13.40	13.70	105.00	74.20	53.60	33.30	20.80	10.20
ETHYLBENZENE	Organics	UG/KG	176	7750	4	995.00	2,930.00	31,800.00	7,290.00	5,120.00	3,130.00	2,060.00	370.00
HEPTACHLOR EPOXIDE	Pesticides	UG/KG	175	7720	4	25.50	26.70	211.00	149.00	107.00	45.70	38.10	19.70
M-XYLENE	Organics	UG/KG	176	7750	4	985.00	2,920.00	31,800.00	7,430.00	5,170.00	3,050.00	2,060.00	360.00
TETRACHLOROMETHANE	Organics	UG/KG	176	7750	4	969.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,060.00	357.00
ALDRIN	Pesticides	UG/KG	175	7720	3	13.00	13.40	105.00	74.20	53.60	25.20	20.80	10.00
BENZO(A)PYRENE	Organics	UG/KG	176	7750	3	9,740.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
CHLORPYRIFOS	Pesticides	UG/KG	176	7750	3	65.30	73.60	529.00	426.00	335.00	157.00	105.00	51.70
ISOBUTYL ALCOHOL	Organics	UG/KG	176	7750	3	967.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,060.00	357.00
NITROFEN (TOK)	Pesticides	UG/KG	175	7720	3	26.50	26.80	211.00	164.00	123.00	66.50	40.90	20.00
O+P XYLENE	Organics	UG/KG	176	7750	3	971.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,060.00	357.00
STYRENE	Organics	UG/KG	176	7750	3	12,100.00	24,000.00	310,000.00	67,300.00	63,800.00	37,300.00	30,400.00	4,760.00
TETRACHLOROETHENE	Organics	UG/KG	176	7750	3	994.00	2,930.00	31,800.00	7,210.00	5,110.00	3,130.00	2,060.00	357.00
TRIFLURALIN (TREFLAN)	Pesticides	UG/KG	175	7720	3	65.90	68.80	526.00	371.00	269.00	155.00	107.00	50.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY

Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
AZINPHOS METHYL	Pesticides	UG/KG	176	7750	2	158.00	165.00	1,210.00	893.00	643.00	311.00	269.00	130.00
CHLOROBENZENE	Organics	UG/KG	176	7750	2	995.00	2,920.00	31,800.00	7,290.00	5,120.00	3,130.00	2,080.00	357.00
DELTA-BHC	Pesticides	UG/KG	175	7720	2	32.00	33.70	263.00	186.00	135.00	62.30	51.70	24.90
DIAZINON	Pesticides	UG/KG	176	7750	2	64.00	66.70	483.00	357.00	257.00	150.00	107.00	51.90
DIBENZOFURAN	Organics	UG/KG	176	7750	2	9,930.00	13,600.00	154,000.00	59,600.00	43,200.00	33,300.00	28,600.00	4,760.00
NALED (DIBROM)	Pesticides	UG/KG	176	7750	2	424.00	446.00	3,280.00	2,430.00	1,750.00	840.00	707.00	340.00
PHENANTHRENE	Organics	UG/KG	176	7750	2	9,930.00	13,500.00	154,000.00	46,700.00	43,000.00	33,300.00	28,600.00	4,760.00
PHOSPHORIC ACID, TRI-O-TOLYL ESTER	Pesticides	UG/KG	176	7750	2	742.00	976.00	7,080.00	5,160.00	3,640.00	1,530.00	1,310.00	570.00
SANTOX (EPN)	Pesticides	UG/KG	176	7750	2	62.60	66.10	545.00	359.00	258.00	124.00	104.00	51.90
TETRAETHYLPYROPHOSPHATE	Pesticides	UG/KG	176	7750	2	1,270.00	1,100.00	20,000.00	3,920.00	3,100.00	2,950.00	2,500.00	1,090.00
1,2:3,4-DIEPOXYBUTANE	Organics	UG/KG	176	7750	2	19,600.00	28,900.00	308,000.00	93,300.00	66,000.00	66,000.00	56,000.00	9,520.00
1,4-DICHLOROBENZENE	Organics	UG/KG	176	7750	2	9,720.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1,4-DIOXANE	Organics	UG/KG	176	7750	2	987.00	2,960.00	36,000.00	7,540.00	5,170.00	3,130.00	2,060.00	357.00
4-METHYL-2-PENTANONE	Organics	UG/KG	176	7750	2	10,200.00	96,800.00	1,560,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
4,4'-DDT	Pesticides	UG/KG	175	7720	2	25.50	26.90	211.00	155.00	125.00	46.40	41.50	19.70
ACETOPHENONE	Organics	UG/KG	176	7750	2	9,720.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
ALPHA-BHC	Pesticides	UG/KG	175	7720	2	12.80	13.40	105.00	74.20	53.60	22.80	20.10	10.00
ALPHA-TERPINEOL	Organics	UG/KG	176	7750	2	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
ANTHRACENE	Organics	UG/KG	176	7750	2	9,740.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
BIPHENYL	Organics	UG/KG	176	7750	1	68,700.00	847,000.00	15,300,000.00	78,800.00	43,300.00	33,300.00	28,000.00	4,760.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CHLOROFORM	Organics	UG/KG	176	7750	1	986.00	2,950.00	31,600.00	7,530.00	5,160.00	3,130.00	2,080.00	357.00
DI-N-OCTYL PHTHALATE	Organics	UG/KG	176	7750	1	9,700.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
DIMETHOATE	Pesticides	UG/KG	176	7750	1	62.40	66.30	483.00	412.00	259.00	126.00	104.00	50.10
DIMETHYL PHTHALATE	Organics	UG/KG	176	7750	1	9,600.00	13,500.00	154,000.00	46,700.00	43,000.00	33,100.00	28,600.00	4,760.00
ENDOSULFAN-I	Pesticides	UG/KG	175	7720	1	63.30	67.00	526.00	371.00	269.00	115.00	95.90	49.20
N-NITROSODIPHENYLAMINE	Organics	UG/KG	176	7750	1	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
NAPHTHALENE	Organics	UG/KG	176	7750	1	9,690.00	13,500.00	154,000.00	46,700.00	43,000.00	32,900.00	28,000.00	4,760.00
TRANS-1,2-DICHLOROETHENE	Organics	UG/KG	176	7750	1	975.00	2,930.00	31,600.00	7,530.00	5,160.00	2,940.00	2,080.00	357.00
TRICHLOROETHENE	Organics	UG/KG	176	7750	1	979.00	2,920.00	31,600.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
2-CHLORONAPHTHALENE	Organics	UG/KG	176	7750	1	9,720.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2-METHYLNAPHTHALENE	Organics	UG/KG	176	7750	1	10,100.00	15,500.00	154,000.00	66,100.00	43,200.00	33,300.00	28,600.00	4,760.00
2-PICOLINE	Organics	UG/KG	176	7750	1	48,900.00	67,400.00	769,000.00	236,000.00	215,000.00	165,000.00	139,000.00	23,800.00
4,4'-DDE	Pesticides	UG/KG	175	7720	1	63.80	67.10	526.00	371.00	269.00	124.00	99.10	49.30
ANILINE, 2,4,5-TRIMETHYL-	Organics	UG/KG	176	7750	0	19,400.00	27,000.00	308,000.00	100,000.00	86,100.00	66,000.00	56,000.00	9,520.00
ARAMITE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
AZINPHOS ETHYL	Pesticides	UG/KG	176	7750	0	155.00	165.00	1,210.00	693.00	643.00	309.00	260.00	125.00
BENZANTHRONE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
BENZENE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,600.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
BENZENETHIOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
BENZIDINE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
BENZO(GHI)PERYLENE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
BENZONITRILE, 3,5-DIBROMO-4-HYDROXY-	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
BENZYL ALCOHOL	Organics	UG/KG	176	7750	0	9,720.00	13,600.00	156,000.00	47,600.00	43,000.00	32,900.00	27,800.00	4,760.00
BETA-NAPHTHYLAMINE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
BIPHENYL, 4-NITRO	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
BIS(2-CHLOROETHOXY)METHANE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
BIS(2-CHLOROETHYL) ETHER	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
BIS(2-CHLOROISOPROPYL) ETHER	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,060.00	357.00
BROMODICHLOROMETHANE	Organics	UG/KG	176	7750	0	4,830.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
BROMOMETHANE	Pesticides	UG/KG	175	7720	0	632.00	670.00	5,260.00	3,710.00	2,660.00	1,140.00	951.00	492.00
CAPTAFOL	Pesticides	UG/KG	175	7720	0	127.00	134.00	1,050.00	747.00	541.00	228.00	194.00	98.50
CAPTAN	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
CARBAZOLE	Pesticides	UG/KG	175	7720	0	632.00	670.00	5,260.00	3,710.00	2,660.00	1,140.00	951.00	492.00
CARBOPHENOTHION (TRITHION)	Pesticides	UG/KG	175	7720	0	317.00	335.00	2,630.00	1,860.00	1,340.00	568.00	504.00	246.00
CHLORDANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,060.00	357.00
CHLOROACETONITRILE	Pesticides	UG/KG	176	7750	0	4,830.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
CHLOROETHANE	Organics	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
CHLOROFENVINPHOS	Pesticides	UG/KG	176	7750	0	4,830.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
CHLOROMETHANE	Organics	UG/KG	176	7750	0	126.00	133.00	970.00	723.00	517.00	249.00	206.00	102.00
CIODRIN	Pesticides	UG/KG	175	7614	0								

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CIS-1,3-DICHLOROPROPENE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
COUMAPHOS	Pesticides	UG/KG	176	7750	0	124.00	132.00	970.00	716.00	514.00	247.00	208.00	100.00
CROTONALDEHYDE	Organics	UG/KG	176	7750	0	4,630.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
CROTOXYPHOS	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
DEMETON	Pesticides	UG/KG	176	7750	0	124.00	132.00	970.00	716.00	514.00	247.00	208.00	100.00
DI-N-PROPYLNITROSAMINE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	66,000.00	65,800.00	55,500.00	9,520.00
DIALLATE	Pesticides	UG/KG	175	7720	0	128.00	135.00	1,050.00	743.00	537.00	251.00	199.00	98.40
DIBENZO(A,H)ANTHRACENE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	66,000.00	65,800.00	55,500.00	9,520.00
DIBENZOTHIOPHENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
DIBROMOCHLOROMETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
DIBROMOMETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
DICHLORVOS	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
DICROTOPHOS (BIDRIN)	Pesticides	UG/KG	176	7750	0	358.00	380.00	2,790.00	2,060.00	1,480.00	712.00	598.00	289.00
DIETHYL ETHER	Organics	UG/KG	176	7750	0	4,630.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
DIETHYL PHTHALATE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
DIMETHYL SULFONE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
DIOXATHION	Pesticides	UG/KG	176	7750	0	497.00	526.00	3,870.00	2,660.00	2,060.00	968.00	831.00	400.00
DIPHENYL ETHER	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
DIPHENYLAMINE	Organics	UG/KG	176	7750	0	9,800.00	13,500.00	154,000.00	46,700.00	43,000.00	33,200.00	28,800.00	4,760.00
DIPHENYLDISULFIDE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	66,000.00	65,800.00	55,500.00	9,520.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
DISULFOTON	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
ENDOSULFAN SULFATE	Pesticides	UG/KG	175	7720	0	63.30	67.00	526.00	371.00	269.00	114.00	95.20	49.20
ENDRIN ALDEHYDE	Pesticides	UG/KG	175	7720	0	31.80	33.50	263.00	186.00	135.00	57.10	47.70	24.90
ENDRIN KETONE	Pesticides	UG/KG	175	7720	0	63.30	67.00	526.00	371.00	269.00	114.00	95.20	49.20
ETHANE, PENTACHLORO-	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	66,000.00	65,800.00	55,500.00	9,520.00
ETHION	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
ETHYL CYANIDE	Organics	UG/KG	176	7750	0	1,010.00	3,400.00	64,700.00	7,530.00	5,160.00	3,130.00	2,080.00	357.00
ETHYL METHACRYLATE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
ETHYL METHANESULFONATE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	66,000.00	65,800.00	55,500.00	9,520.00
ETHYLENETHIOUREA	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	66,000.00	65,800.00	55,500.00	9,520.00
FAMPHUR	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
FENSULFOTHION	Pesticides	UG/KG	176	7750	0	154.00	163.00	1,200.00	887.00	638.00	306.00	258.00	124.00
FENTHION	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
FLUORENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
HEPTACHLOR	Pesticides	UG/KG	175	7720	0	25.30	26.80	211.00	149.00	107.00	45.70	38.10	19.70
HEXACHLOROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
HEXACHLOROBUTADIENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
HEXACHLOROCYCLOPENTADIENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
HEXACHLOROETHANE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
HEXACHLOROPROPENE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	66,000.00	65,800.00	55,500.00	9,520.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
INDENO(1,2,3-CD)PYRENE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
IODOMETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
ISODRIN	Pesticides	UG/KG	175	7720	0	63.30	67.00	526.00	371.00	269.00	114.00	95.20	49.20
ISOPHORONE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
ISOSAFROLE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
LEPTOPHOS	Pesticides	UG/KG	176	7750	0	75.20	79.60	583.00	432.00	315.00	149.00	125.00	60.40
LINDANE (GAMMA-BHC)	Pesticides	UG/KG	175	7720	0	31.90	33.50	263.00	186.00	135.00	57.10	51.90	24.90
LONGIFOLENE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
MALACHITE GREEN	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
MALATHION	Pesticides	UG/KG	176	7750	0	62.10	65.60	483.00	357.00	257.00	124.00	104.00	50.00
MESTRANOL	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
METHAPYRILENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
METHOXYCHLOR	Pesticides	UG/KG	175	7720	0	63.30	67.00	526.00	371.00	269.00	114.00	95.20	49.20
METHYL METHACRYLATE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
METHYL METHANESULFONATE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
METHYL PARATHION	Pesticides	UG/KG	176	7750	0	62.10	65.60	483.00	357.00	257.00	124.00	104.00	50.00
MEVINPHOS (PHOSDRIN)	Pesticides	UG/KG	176	7750	0	62.10	65.60	483.00	357.00	257.00	125.00	104.00	50.00
MIREX	Pesticides	UG/KG	175	7720	0	63.30	67.00	526.00	371.00	269.00	114.00	95.20	49.20
MONOCROTOPHOS	Pesticides	UG/KG	176	7750	0	1,860.00	1,960.00	14,500.00	10,700.00	7,710.00	3,700.00	3,120.00	1,500.00
N-NITROSODI-N-BUTYLAMINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
N-NITROSODIETHYLAMINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
N-NITROSODIMETHYLAMINE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
N-NITROSOMETHYLETHYLAMINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
N-NITROSOMETHYLPHENYLAMINE	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
N-NITROSOMORPHOLINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
N-NITROSOPIPERIDINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
N,N-DIMETHYLFORMAMIDE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
NITROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
O-ANISIDINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
O-TOLUIDINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
O-TOLUIDINE, 5-CHLORO-	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
P-DIMETHYLAMINOAZOBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
P-NITROANILINE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
PARATHION	Pesticides	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
PENTACHLOROBENZENE	Organics	UG/KG	176	7750	0	124.00	132.00	970.00	716.00	514.00	247.00	206.00	100.00
PENTACHLOROPHENOL	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
PENTAMETHYLBENZENE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
PERYLENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
PHENACETIN	Organics	UG/KG	176	7750	0	9,720.00	13,500.00	154,000.00	50,400.00	43,000.00	33,000.00	28,000.00	4,760.00
PHENOL, 2-METHYL-4,6-DINITRO-	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00

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Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
PHENOTHIAZINE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
PHORATE	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
PHOSMET	Pesticides	UG/KG	176	7750	0	124.00	132.00	970.00	716.00	514.00	247.00	208.00	100.00
PHOSPHAMIDON	Pesticides	UG/KG	176	7750	0	373.00	396.00	2,900.00	2,150.00	1,540.00	741.00	623.00	300.00
PHOSPHORIC ACID, TRIMETHYL ESTER	Pesticides	UG/KG	176	7750	0	87.10	92.40	678.00	501.00	360.00	173.00	145.00	70.00
PHOSPHORIC TRIAMIDE, HEXAMETHYL-	Pesticides	UG/KG	176	7750	0	311.00	330.00	2,420.00	1,790.00	1,280.00	617.00	520.00	250.00
PRONAMIDE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
PYRIDINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
RESORCINOL	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
SAFROLE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
SQUALENE	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
TERBUFOS	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
TETRACHLORVINPHOS	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
TETRAETHYLDITHIOPYROPHOSPHATE	Pesticides	UG/KG	176	7750	0	62.10	65.80	483.00	357.00	257.00	124.00	104.00	50.00
THIANAPHTHENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
THIOACETAMIDE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
THIOXANTHE-9-ONE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
TOLUENE, 2,4-DIAMINO-	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
TOXAPHENE	Pesticides	UG/KG	175	7720	0	1,260.00	1,340.00	10,500.00	7,430.00	5,370.00	2,270.00	1,900.00	964.00
TRANS-1,3-DICHLOROPROPENE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
TRANS-1,4-DICHLORO-2-BUTENE	Organics	UG/KG	176	7750	0	4,880.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
TRIBROMOMETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
TRICHLOROFON	Pesticides	UG/KG	176	7750	0	661.00	701.00	5,130.00	3,790.00	2,730.00	1,320.00	1,100.00	535.00
TRIPHENYLENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
TRIPROPYLENEGLYCOL METHYL ETHER	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
VINYL ACETATE	Organics	UG/KG	176	7750	0	4,830.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
VINYL CHLORIDE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1-BROMO-2-CHLOROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1-BROMO-3-CHLOROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1-CHLORO-3-NITROBENZENE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
1-METHYLFLUORENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1-METHYLPHENANTHRENE	Organics	UG/KG	176	7750	0	9,730.00	13,500.00	154,000.00	46,700.00	43,000.00	33,300.00	28,800.00	4,760.00
1-NAPHTHYLAMINE	Organics	UG/KG	176	7750	0	9,690.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1-PHENYLNAPHTHALENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1,1-DICHLOROETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,1-DICHLOROETHENE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,1,1-TRICHLOROETHANE	Organics	UG/KG	176	7750	0	1,030.00	3,210.00	31,800.00	16,000.00	5,200.00	3,130.00	2,080.00	357.00
1,1,1,2-TETRACHLOROETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,1,2-TRICHLOROETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,1,2,2-TETRACHLOROETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
1,2-DIBROMO-3-CHLOROPROPANE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
1,2-DIBROMOETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,2-DICHLOROBENZENE	Organics	UG/KG	176	7750	0	9,700.00	13,500.00	154,000.00	47,600.00	43,000.00	32,900.00	27,800.00	4,760.00
1,2-DICHLOROETHANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,2-DICHLOROPROPANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,2-DIPHENYLHYDRAZINE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
1,2,3-TRICHLOROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,500.00	154,000.00	46,700.00	43,400.00	32,900.00	27,800.00	4,760.00
1,2,3-TRICHLOROPROPANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,2,3-TRIMETHOXYBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1,2,4-TRICHLOROBENZENE	Organics	UG/KG	176	7750	0	9,710.00	13,700.00	164,000.00	47,600.00	43,000.00	32,900.00	27,800.00	4,760.00
1,2,4,5-TETRACHLOROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1,3-BUTADIENE, 2-CHLORO	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,3-DICHLORO-2-PROPANOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
1,3-DICHLOROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,800.00	4,760.00
1,3-DICHLOROPROPANE	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
1,3,5-TRITHIANE	Organics	UG/KG	176	7750	0	46,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
1,4-DINITROBENZENE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
1,4-NAPHTHOQUINONE	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
1,4-NAPHTHOQUINONE, 2,3-DICHLORO-	Pesticides	UG/KG	175	7720	0	401.00	370.00	2,630.00	2,030.00	1,430.00	964.00	624.00	280.00
1,5-NAPHTHALENEDIAMINE	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2-(METHYLTHIO)BENZOTHIAZOLE	Organics	UG/KG	176	7750	0	9,710.00	13,500.00	154,000.00	50,400.00	43,000.00	33,000.00	28,000.00	4,760.00
2-CHLOROETHYLVINYL ETHER	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
2-CHLOROPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2-ISOPROPYLNAPHTHALENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2-METHYLBENZOTHIOAZOLE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2-NITROANILINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2-NITROPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2-PHENYLNAPHTHALENE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
2-PROPEN-1-OL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2-PROPENAL	Organics	UG/KG	176	7750	0	966.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
2-PROPENENITRILE, 2-METHYL-	Organics	UG/KG	176	7750	0	4,830.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
2,3-BENZOFLUORENE	Organics	UG/KG	176	7750	0	1,130.00	6,610.00	218,000.00	7,530.00	5,160.00	3,130.00	2,080.00	357.00
2,3-DICHLOROANILINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,3-DICHLORONITROBENZENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,3,4,6-TETRACHLOROPHENOL	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
2,3,6-TRICHLOROPHENOL	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
2,4-DICHLOROPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,4-DIMETHYLPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,4-DINITROPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,4-DINITROTOLUENE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2,4,5-TRICHLOROPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,4,6-TRICHLOROPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,6-DI-TERT-BUTYL-P-BENZOQUINONE	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
2,6-DICHLORO-4-NITROANILINE	Organics	UG/KG	176	7750	0	95,800.00	133,000.00	1,520,000.00	462,000.00	426,000.00	326,000.00	275,000.00	47,100.00
2,6-DICHLOROPHENOL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
2,6-DINITROTOLUENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
3-CHLOROPROPENE	Organics	UG/KG	176	7750	0	967.00	2,920.00	31,800.00	7,210.00	5,110.00	3,130.00	2,080.00	357.00
3-METHYLCHOLANTHRENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
3-NITROANILINE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
3,3'-DICHLOROBENZIDINE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
3,3'-DIMETHOXYBENZIDINE	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
3,6-DIMETHYLPHENANTHRENE	Organics	UG/KG	176	7750	0	9,750.00	13,500.00	154,000.00	46,700.00	43,000.00	33,300.00	28,800.00	4,760.00
4-AMINOBIPHENYL	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
4-BROMOPHENYL PHENYL ETHER	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
4-CHLORO-2-NITROANILINE	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00
4-CHLORO-3-METHYLPHENOL	Organics	UG/KG	176	7750	0	9,760.00	13,900.00	154,000.00	50,400.00	43,000.00	33,000.00	28,000.00	4,760.00
4-CHLOROPHENYLPHENYL ETHER	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
4-NITROPHENOL	Organics	UG/KG	176	7750	0	48,400.00	67,200.00	769,000.00	233,000.00	215,000.00	164,000.00	139,000.00	23,800.00
4,4'-DDD	Pesticides	UG/KG	175	7720	0	63.30	67.10	526.00	419.00	394.00	114.00	95.30	49.20
4,4'-METHYLENEBIS(2-CHLOROANILINE)	Organics	UG/KG	176	7750	0	19,400.00	26,900.00	308,000.00	93,300.00	86,000.00	65,800.00	55,500.00	9,520.00

TABLE 3. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to the Minimum Level

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
4,5-METHYLENE PHENANTHRENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
5-NITRO-O-TOLUIDINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
7,12-DIMETHYLBENZ(A)ANTHRACENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
ACENAPHTHENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
ACENAPHTHYLENE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
ACRYLONITRILE	Organics	UG/KG	176	7750	0	4,830.00	14,600.00	159,000.00	36,100.00	25,500.00	15,600.00	10,400.00	1,790.00
ANILINE	Organics	UG/KG	176	7750	0	9,680.00	13,400.00	154,000.00	46,700.00	43,000.00	32,900.00	27,700.00	4,760.00
dioxins a	Dioxins	NG/KG	174	7714	0	90.40	172.00	1,820.00	556.00	532.00	311.00	218.00	50.40
PCB a	Pesticides	UG/KG	175	7720	0	2,120.00	2,460.00	14,700.00	13,400.00	12,300.00	5,400.00	3,050.00	1,490.00
PCB-1016	Pesticides	UG/KG	175	7720	0	253.00	268.00	2,110.00	1,490.00	1,070.00	454.00	381.00	197.00
PCB-1221	Pesticides	UG/KG	175	7720	0	253.00	268.00	2,110.00	1,490.00	1,070.00	454.00	381.00	197.00
PCB-1232	Pesticides	UG/KG	175	7720	0	253.00	268.00	2,110.00	1,490.00	1,070.00	454.00	381.00	197.00
PCB-1242	Pesticides	UG/KG	175	7720	0	253.00	268.00	2,110.00	1,490.00	1,070.00	454.00	381.00	197.00

(a) Composites considered a detect if all individual congeners or PCBs are measured above the minimum level.

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CALCIUM	Metals	MG/KG	176	7750	100	35,300.00	46,500.00	333,000.00	313,000.00	151,000.00	84,600.00	66,600.00	26,200.00
IRON	Metals	MG/KG	176	7750	100	18,000.00	21,100.00	229,000.00	81,400.00	79,900.00	59,800.00	41,700.00	12,400.00
MAGNESIUM	Metals	MG/KG	176	7750	100	7,460.00	17,600.00	134,000.00	75,500.00	30,700.00	17,500.00	9,750.00	4,270.00
PHOSPHORUS	Metals	MG/KG	176	7750	100	4,580.00	9,870.00	71,700.00	42,600.00	17,600.00	13,400.00	12,100.00	1,330.00
SODIUM	Metals	MG/KG	176	7750	100	7,970.00	19,600.00	141,000.00	103,000.00	49,100.00	21,400.00	17,200.00	2,070.00
ALUMINUM	Metals	MG/KG	176	7750	100	11,200.00	14,800.00	92,700.00	85,300.00	70,200.00	36,400.00	19,400.00	7,550.00
BARIUM	Metals	MG/KG	176	7750	100	673.00	640.00	5,570.00	3,000.00	2,370.00	1,730.00	1,230.00	499.00
dioxins b	Dioxins	NG/KG	174	7714	100	54.10	168.00	1,700.00	547.00	516.00	250.00	84.90	11.20
MANGANESE	Metals	MG/KG	176	7750	100	538.00	1,040.00	13,200.00	4,060.00	3,720.00	1,620.00	929.00	276.00
OCTACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	100	11.70	59.20	905.00	127.00	116.00	33.70	15.90	3.32
TITANIUM	Metals	MG/KG	176	7750	98	109.00	140.00	2,380.00	570.00	416.00	363.00	237.00	68.50
1,2,3,4,6,7,8-HEPTACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	98	9.50	37.00	525.00	103.00	73.70	27.90	14.00	3.35
NITRATE	Classicals	MG/KG	176	7750	95	1,420.00	5,040.00	35,300.00	26,500.00	15,500.00	5,020.00	1,890.00	96.50
SILVER	Metals	MG/KG	176	7750	84	46.40	113.00	852.00	546.00	218.00	128.00	75.80	23.70
TIN	Metals	MG/KG	176	7750	84	96.60	280.00	2,100.00	1,370.00	715.00	136.00	129.00	45.00
NITRITE	Classicals	MG/KG	176	7750	83	196.00	1,210.00	17,700.00	2,920.00	2,910.00	462.00	190.00	9.46
OCTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	80	0.65	4.36	69.50	5.04	4.75	3.31	0.55	0.06
1,2,3,4,6,7,8-HEPTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	71	1.70	6.46	71.00	31.50	27.80	5.11	2.93	0.36
2,3,7,8-TETRACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	65	1.84	3.48	33.70	14.80	14.70	7.21	4.56	0.39
FLUORIDE	Classicals	MG/KG	176	7750	63	92.20	225.00	1,510.00	1,280.00	744.00	327.00	219.00	9.47

(b) Composites considered a detect if at least one congener is measured above the minimum level.

* Dioxins and Furans Reported as Toxic Equivalents

Note: Three significant figures are reported

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
BIS(2-ETHYLHEXYL) PHTHALATE	Organics	UG/KG	176	7750	62	50,500.00	108,000.00	891,000.00	525,000.00	459,000.00	459,000.00	191,000.00	148,000.00
VANADIUM	Metals	MG/KG	176	7750	62	16.00	43.80	570.00	181.00	93.50	93.50	56.40	30.10
YTTRIUM	Metals	MG/KG	176	7750	61	3.02	3.51	16.20	14.30	13.50	13.50	6.97	6.80
TOLUENE	Organics	UG/KG	176	7750	61	40,800.00	116,000.00	1,180,000.00	575,000.00	439,000.00	439,000.00	238,000.00	93,400.00
2-PROPANONE	Organics	UG/KG	176	7750	58	58,400.00	323,000.00	2,430,000.00	1,460,000.00	738,000.00	738,000.00	74,900.00	39,700.00
1,2,3,6,7,8-HEXACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	49	3.24	10.70	73.70	58.50	47.70	47.70	12.10	7.21
BORON	Metals	MG/KG	176	7750	48	28.10	66.90	310.00	304.00	293.00	293.00	182.00	64.40
P-CRESOL	Organics	UG/KG	176	7750	43	46,200.00	112,000.00	1,160,000.00	542,000.00	430,000.00	430,000.00	306,000.00	202,000.00
1,2,3,4,7,8-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	43	5.05	21.20	150.00	109.00	73.20	73.20	15.30	8.17
METHYLENE CHLORIDE	Organics	UG/KG	176	7750	42	7,820.00	37,900.00	262,000.00	184,000.00	114,000.00	114,000.00	13,600.00	1,930.00
1,2,3,7,8,9-HEXACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	39	2.09	6.60	73.70	19.60	17.90	12.70	3.92	0.00
ANTIMONY	Metals	MG/KG	176	7750	38	1.90	19.10	680.00	20.50	9.68	4.55	3.20	0.00
CYANIDES (SOLUBLE SALTS AND COMPLEXES)	Classicals	MG/KG	176	7750	37	14.30	54.00	372.00	295.00	147.00	85.30	11.90	0.00
HEXANOIC ACID	Organics	UG/KG	176	7750	35	41,200.00	95,300.00	1,960,000.00	406,000.00	280,000.00	202,000.00	127,000.00	0.00
1,2,3,6,7,8-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	35	1.20	5.55	73.70	15.00	15.00	9.00	0.60	0.00
2-BUTANONE	Organics	UG/KG	176	7750	34	18,300.00	96,400.00	1,540,000.00	541,000.00	510,000.00	43,300.00	5,510.00	0.00
PHENOL	Organics	UG/KG	176	7750	34	12,200.00	40,400.00	920,000.00	162,000.00	111,000.00	45,200.00	32,200.00	0.00
2,4,5-TRICHLOROPHENOXYACETIC ACID	Pesticides	UG/KG	176	7750	29	15.20	134.00	2,170.00	72.70	56.30	38.70	22.70	0.00
2,3,4,6,7,8-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	27	1.05	5.22	73.70	14.80	13.10	3.43	1.19	0.00
1,2,3,4,7,8,9-HEPTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	26	0.16	0.72	8.42	1.54	1.45	1.11	0.20	0.00

Note: Three significant figures are reported

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2,3,4,7,8-PENTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	26	6.27	27.50	368.00	74.50	66.00	45.50	5.33	0.00
1,2,3,4,7,8-HEXACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	25	1.48	5.91	73.70	17.90	16.20	13.40	2.29	0.00
BERYLLIUM	Metals	MG/KG	176	7750	22	0.14	0.38	3.90	1.17	0.96	0.70	0.50	0.00
1,2,3,7,8-PENTACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	22	0.53	2.62	36.80	7.15	6.06	3.63	0.49	0.00
PCB b	Pesticides	UG/KG	175	7720	19	393.00	1,790.00	13,400.00	8,700.00	2,470.00	1,320.00	683.00	0.00
1,2,3,7,8-PENTACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	18	4.56	25.50	368.00	66.70	60.10	27.50	3.33	0.00
ACETIC ACID (2,4-DICHLOROPHENOXY)	Pesticides	UG/KG	176	7750	16	3.62	16.50	422.00	42.00	30.90	23.70	11.30	0.00
1,2,3,7,8,9-HEXACHLORODIBENZOFURAN	Dioxins	NG/KG	174	7714	16	1.31	8.53	126.00	19.50	14.40	5.21	0.89	0.00
2,3,7,8-TETRACHLORODIBENZO-P-DIOXIN	Dioxins	NG/KG	174	7714	16	1.71	8.45	116.00	26.30	25.00	7.59	2.43	0.00
N-HEXACOSANE	Organics	UG/KG	176	7750	15	2,920.00	31,200.00	498,000.00	44,200.00	19,300.00	3,270.00	668.00	0.00
N-TETRACOSANE	Organics	UG/KG	176	7750	15	6,620.00	43,800.00	650,000.00	97,700.00	84,100.00	24,400.00	3,930.00	0.00
2,4,5-TRICHLOROPHENOXYPROPIONIC ACID	Pesticides	UG/KG	176	7750	15	4.23	13.40	121.00	53.50	43.50	31.90	11.80	0.00
THALLIUM	Metals	MG/KG	176	7750	15	0.25	0.97	10.10	4.42	3.10	1.36	0.40	0.00
N-DODECANE	Organics	UG/KG	176	7750	14	12,000.00	43,200.00	525,000.00	221,000.00	196,000.00	83,000.00	20,000.00	0.00
N-TETRADECANE	Organics	UG/KG	176	7750	14	6,460.00	28,000.00	352,000.00	94,300.00	93,300.00	46,600.00	6,640.00	0.00
N-TRIACONTANE	Organics	UG/KG	176	7750	14	6,220.00	42,200.00	597,000.00	134,000.00	96,900.00	13,100.00	2,210.00	0.00
N-EICOSANE	Organics	UG/KG	176	7750	13	3,990.00	15,600.00	196,000.00	63,600.00	50,300.00	21,600.00	6,630.00	0.00
N-HEXADECANE	Organics	UG/KG	176	7750	12	6,130.00	49,600.00	758,000.00	110,000.00	35,700.00	20,200.00	166.00	0.00
N-OCTACOSANE	Organics	UG/KG	176	7750	12	3,760.00	27,300.00	413,000.00	45,300.00	32,500.00	21,500.00	1,620.00	0.00
ENDOSULFAN-II	Pesticides	UG/KG	175	7720	12	7.79	29.00	184.00	175.00	126.00	44.80	12.50	0.00

(b) Composites considered a detect if at least one PCB aroclor is measured above the minimum level.

Note: Three significant figures are reported

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CARBON DISULFIDE	Organics	UG/KG	176	7750	10	127.00	1,520.00	23,500.00	496.00	413.00	76.80	10.20	0.00
PENTACHLORONITROBENZENE	Pesticides	UG/KG	175	7720	10	158.00	1,160.00	8,830.00	3,970.00	156.00	20.20	0.15	0.00
N-DECANE	Organics	UG/KG	176	7750	10	4,180.00	22,400.00	199,000.00	149,000.00	67,800.00	15,800.00	0.00	0.00
PCB-1260	Pesticides	UG/KG	175	7720	10	97.20	534.00	4,010.00	2,190.00	650.00	494.00	0.00	0.00
COBALT	Metals	MG/KG	176	7750	9	1.15	7.03	103.00	17.00	13.90	6.30	0.00	0.00
BUTYL BENZYL PHTHALATE	Organics	UG/KG	176	7750	9	475.00	2,010.00	13,400.00	10,100.00	6,610.00	2,720.00	0.00	0.00
PCB-1248	Pesticides	UG/KG	175	7720	9	33.60	193.00	5,200.00	600.00	376.00	231.00	0.00	0.00
N-DOCOSANE	Organics	UG/KG	176	7750	8	907.00	5,010.00	70,200.00	24,300.00	7,220.00	5,770.00	0.00	0.00
N-OCTADECANE	Organics	UG/KG	176	7750	8	2,270.00	11,800.00	130,000.00	58,400.00	33,400.00	13,600.00	0.00	0.00
PCB-1254	Pesticides	UG/KG	175	7720	8	262.00	1,280.00	9,350.00	5,490.00	2,430.00	1,240.00	0.00	0.00
CHLOROBENZILATE	Pesticides	UG/KG	175	7720	7	5.48	21.50	104.00	101.00	97.70	54.50	0.00	0.00
P-CYMENE	Organics	UG/KG	176	7750	7	1,390.00	7,410.00	64,300.00	35,100.00	33,300.00	4,690.00	0.00	0.00
BENZO(B)FLUORANTHENE	Organics	UG/KG	176	7750	6	181.00	1,750.00	34,200.00	1,330.00	1,160.00	232.00	0.00	0.00
BETA-BHC	Pesticides	UG/KG	175	7720	6	2.30	11.10	163.00	51.50	38.80	22.80	0.00	0.00
ENDRIN	Pesticides	UG/KG	175	7720	6	2.65	12.60	123.00	62.70	42.00	11.70	0.00	0.00
O-CRESOL	Organics	UG/KG	176	7750	6	7,500.00	47,800.00	328,000.00	327,000.00	257,000.00	496.00	0.00	0.00
FLUORANTHENE	Organics	UG/KG	176	7750	5	331.00	1,870.00	19,400.00	11,500.00	8,200.00	140.00	0.00	0.00
P-CHLOROANILINE	Organics	UG/KG	176	7750	5	397.00	2,930.00	40,200.00	5,660.00	4,140.00	257.00	0.00	0.00
PYRENE	Organics	UG/KG	176	7750	5	320.00	2,000.00	37,100.00	9,550.00	5,600.00	345.00	0.00	0.00
TRICHLOROFLUOROMETHANE	Organics	UG/KG	176	7750	5	71.50	520.00	3,970.00	1,600.00	116.00	4.66	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2-HEXANONE	Organics	UG/KG	176	7750	5	28.60	344.00	12,700.00	468.00	441.00	0.00	0.00	0.00
BENZO(A)ANTHRACENE	Organics	UG/KG	176	7750	4	104.00	1,250.00	40,500.00	1,160.00	998.00	0.00	0.00	0.00
BENZO(K)FLUORANTHENE	Organics	UG/KG	176	7750	4	136.00	1,310.00	31,200.00	1,390.00	1,060.00	0.00	0.00	0.00
BENZOIC ACID	Organics	UG/KG	176	7750	4	6,620.00	43,100.00	635,000.00	227,000.00	48,900.00	0.00	0.00	0.00
CHRYSENE	Organics	UG/KG	176	7750	4	102.00	1,110.00	32,400.00	1,130.00	1,090.00	0.00	0.00	0.00
DI-N-BUTYL PHTHALATE	Organics	UG/KG	176	7750	4	1,690.00	13,100.00	322,000.00	56,300.00	11,800.00	0.00	0.00	0.00
DIELDRIN	Pesticides	UG/KG	175	7720	4	1.05	5.54	47.50	29.50	23.40	0.00	0.00	0.00
ETHYLBENZENE	Organics	UG/KG	176	7750	4	24.80	321.00	20,200.00	520.00	322.00	0.00	0.00	0.00
HEPTACHLOR EPOXIDE	Pesticides	UG/KG	175	7720	4	0.60	3.14	17.90	16.90	13.80	0.00	0.00	0.00
M-XYLENE	Organics	UG/KG	176	7750	4	27.00	254.00	6,910.00	466.00	372.00	0.00	0.00	0.00
TETRACHLOROMETHANE	Organics	UG/KG	176	7750	4	4.05	26.10	339.00	90.90	67.90	0.00	0.00	0.00
ALDRIN	Pesticides	UG/KG	175	7720	3	0.64	3.70	38.80	19.40	13.70	0.00	0.00	0.00
BENZO(A)PYRENE	Organics	UG/KG	176	7750	3	78.80	1,140.00	24,700.00	968.00	749.00	0.00	0.00	0.00
CHLORPYRIFOS	Pesticides	UG/KG	176	7750	3	4.77	39.40	529.00	79.40	64.60	0.00	0.00	0.00
ISOBUTYL ALCOHOL	Organics	UG/KG	176	7750	3	0.94	10.60	165.00	6.37	2.44	0.00	0.00	0.00
NITROFEN (TOK)	Pesticides	UG/KG	175	7720	3	1.59	13.30	195.00	26.80	22.50	0.00	0.00	0.00
O+P XYLENE	Organics	UG/KG	176	7750	3	8.30	101.00	2,740.00	155.00	86.20	0.00	0.00	0.00
STYRENE	Organics	UG/KG	176	7750	3	2,490.00	21,000.00	310,000.00	51,200.00	28,200.00	0.00	0.00	0.00
TETRACHLOROETHENE	Organics	UG/KG	176	7750	3	40.50	300.00	4,050.00	1,270.00	526.00	0.00	0.00	0.00
TRIFLURALIN (TREFLAN)	Pesticides	UG/KG	175	7720	3	3.78	24.90	235.00	117.00	58.50	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
AZINPHOS METHYL	Pesticides	UG/KG	176	7750	2	4.91	36.10	279.00	127.00	0.00	0.00	0.00	0.00
CHLOROBENZENE	Organics	UG/KG	176	7750	2	31.20	263.00	8,920.00	1,420.00	196.00	0.00	0.00	0.00
DELTA-BHC	Pesticides	UG/KG	175	7720	2	1.28	8.88	81.80	56.80	21.00	0.00	0.00	0.00
DIAZINON	Pesticides	UG/KG	176	7750	2	2.63	19.70	151.00	64.30	0.00	0.00	0.00	0.00
DIBENZOFURAN	Organics	UG/KG	176	7750	2	275.00	3,690.00	59,300.00	3,940.00	0.00	0.00	0.00	0.00
NALED (DIBROM)	Pesticides	UG/KG	176	7750	2	7.16	47.30	484.00	299.00	207.00	0.00	0.00	0.00
PHENANTHRENE	Organics	UG/KG	176	7750	2	348.00	2,860.00	40,200.00	7,370.00	2,850.00	0.00	0.00	0.00
PHOSPHORIC ACID, TRI-O-TOLYL ESTER	Pesticides	UG/KG	176	7750	2	127.00	930.00	7,080.00	4,380.00	442.00	0.00	0.00	0.00
SANTOX (EPN)	Pesticides	UG/KG	176	7750	2	1.22	11.20	545.00	26.20	0.00	0.00	0.00	0.00
TETRAETHYLPYROPHOSPHATE	Pesticides	UG/KG	176	7750	2	53.80	622.00	20,000.00	1,720.00	0.00	0.00	0.00	0.00
1,2:3,4-DIEPOXYBUTANE	Organics	UG/KG	176	7750	2	229.00	2,720.00	73,900.00	2,180.00	0.00	0.00	0.00	0.00
1,4-DICHLOROBENZENE	Organics	UG/KG	176	7750	2	88.90	664.00	12,000.00	3,070.00	647.00	0.00	0.00	0.00
1,4-DIOXANE	Organics	UG/KG	176	7750	2	13.90	518.00	35,300.00	52.30	0.00	0.00	0.00	0.00
4-METHYL-2-PENTANONE	Organics	UG/KG	176	7750	2	2.68	27.00	567.00	32.20	9.65	0.00	0.00	0.00
4,4'-DDT	Pesticides	UG/KG	175	7720	2	0.51	4.42	121.00	13.30	6.35	0.00	0.00	0.00
ACETOPHENONE	Organics	UG/KG	176	7750	2	125.00	908.00	6,920.00	3,530.00	337.00	0.00	0.00	0.00
ALPHA-BHC	Pesticides	UG/KG	175	7720	2	0.27	1.93	46.90	11.70	3.45	0.00	0.00	0.00
ALPHA-TERPINEOL	Organics	UG/KG	176	7750	2	19.90	150.00	2,560.00	501.00	0.00	0.00	0.00	0.00
ANTHRACENE	Organics	UG/KG	176	7750	2	87.20	894.00	27,200.00	1,600.00	0.00	0.00	0.00	0.00
BIPHENYL	Organics	UG/KG	176	7750	1	59,000.00	947,000.00	15,300,000.00	5,640.00	0.00	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CHLOROFORM	Organics	UG/KG	176	7750	1	20.40	454.00	15,900.00	0.00	0.00	0.00	0.00	0.00
DI-N-OCTYL PHTHALATE	Organics	UG/KG	176	7750	1	31.70	588.00	18,300.00	0.00	0.00	0.00	0.00	0.00
DIMETHOATE	Pesticides	UG/KG	176	7750	1	0.61	10.50	340.00	0.00	0.00	0.00	0.00	0.00
DIMETHYL PHTHALATE	Organics	UG/KG	176	7750	1	135.00	2,010.00	32,400.00	0.00	0.00	0.00	0.00	0.00
ENDOSULFAN-I	Pesticides	UG/KG	175	7720	1	0.29	4.21	125.00	1.93	0.00	0.00	0.00	0.00
N-NITROSODIPHENYLAMINE	Organics	UG/KG	176	7750	1	101.00	1,290.00	19,700.00	0.00	0.00	0.00	0.00	0.00
NAPHTHALENE	Organics	UG/KG	176	7750	1	104.00	1,420.00	26,700.00	316.00	0.00	0.00	0.00	0.00
TRANS-1,2-DICHLOROETHENE	Organics	UG/KG	176	7750	1	12.50	303.00	10,700.00	0.00	0.00	0.00	0.00	0.00
TRICHLOROETHENE	Organics	UG/KG	176	7750	1	18.50	220.00	3,300.00	45.40	0.00	0.00	0.00	0.00
2-CHLORONAPHTHALENE	Organics	UG/KG	176	7750	1	54.80	714.00	11,100.00	0.00	0.00	0.00	0.00	0.00
2-METHYLNAPHTHALENE	Organics	UG/KG	176	7750	1	533.00	8,380.00	136,000.00	0.00	0.00	0.00	0.00	0.00
2-PICOLINE	Organics	UG/KG	176	7750	1	555.00	7,300.00	365,000.00	0.00	0.00	0.00	0.00	0.00
4,4'-DDE	Pesticides	UG/KG	175	7720	1	0.72	9.46	190.00	0.00	0.00	0.00	0.00	0.00
ANILINE, 2,4,5-TRIMETHYL-	Organics	UG/KG	176	7750	0	71.40	2,610.00	95,500.00	0.00	0.00	0.00	0.00	0.00
ARAMITE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
AZINPHOS ETHYL	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BENZANTHRONE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BENZENE	Organics	UG/KG	176	7750	0	0.07	3.50	220.00	0.00	0.00	0.00	0.00	0.00
BENZENETHIOL	Organics	UG/KG	176	7750	0	2.43	88.70	3,250.00	0.00	0.00	0.00	0.00	0.00
BENZIDINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
BENZO(GHI)PERYLENE	Organics	UG/KG	176	7750	0	13.80	383.00	12,900.00	0.00	0.00	0.00	0.00	0.00
BENZONITRILE, 3,5-DIBROMO-4-HYDROXY-	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BENZYL ALCOHOL	Organics	UG/KG	176	7750	0	82.50	2,270.00	156,000.00	0.00	0.00	0.00	0.00	0.00
BETA-NAPHTHYLAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BIPHENYL, 4-NITRO	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BIS(2-CHLOROETHOXY)METHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BIS(2-CHLOROETHYL) ETHER	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BIS(2-CHLOROISOPROPYL) ETHER	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BROMODICHLOROMETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BROMOMETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CAPTAFOL	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CAPTAN	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CARBAZOLE	Organics	UG/KG	176	7750	0	0.88	17.30	968.00	0.00	0.00	0.00	0.00	0.00
CARBOPHENOTHION (TRITHION)	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CHLORDANE	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CHLOROACETONITRILE	Organics	UG/KG	176	7750	0	1.68	30.20	489.00	0.00	0.00	0.00	0.00	0.00
CHLOROETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CHLOROFENVINPHOS	Pesticides	UG/KG	176	7750	0	0.06	2.05	75.10	0.00	0.00	0.00	0.00	0.00
CHLOROMETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CIODRIN	Pesticides	UG/KG	175	7614	0	0.35	5.71	93.00	0.00	0.00	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
CIS-1,3-DICHLOROPROPENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
COUMAPHOS	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CROTONALDEHYDE	Organics	UG/KG	176	7750	0	0.27	9.78	358.00	0.00	0.00	0.00	0.00	0.00
CROTOXYPHOS	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DEMETON	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DI-N-PROPYLNITROSAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DIALLATE	Pesticides	UG/KG	175	7720	0	1.69	25.20	394.00	0.00	0.00	0.00	0.00	0.00
DIBENZO(A,H)ANTHRACENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DIBENZOTHIOPHENE	Organics	UG/KG	176	7750	0	1.10	40.20	1,470.00	0.00	0.00	0.00	0.00	0.00
DIBROMOCHLOROMETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DIBROMOMETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DICHLORVOS	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DICROTOPHOS (BIDRIN)	Pesticides	UG/KG	176	7750	0	0.41	15.00	550.00	0.00	0.00	0.00	0.00	0.00
DIETHYL ETHER	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DIETHYL PHTHALATE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DIMETHYL SULFONE	Organics	UG/KG	176	7750	0	3.00	48.40	784.00	0.00	0.00	0.00	0.00	0.00
DIOXATHION	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DIPHENYL ETHER	Organics	UG/KG	176	7750	0	3.98	234.00	16,400.00	0.00	0.00	0.00	0.00	0.00
DIPHENYLAMINE	Organics	UG/KG	176	7750	0	125.00	2,010.00	32,600.00	0.00	0.00	0.00	0.00	0.00
DIPHENYLDISULFIDE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
DISULFOTON	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ENDOSULFAN SULFATE	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ENDRIN ALDEHYDE	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ENDRIN KETONE	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ETHANE, PENTACHLORO-	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ETHION	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ETHYL CYANIDE	Organics	UG/KG	176	7750	0	48.40	1,770.00	64,700.00	0.00	0.00	0.00	0.00	0.00
ETHYL METHACRYLATE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ETHYL METHANESULFONATE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ETHYLENETHIOUREA	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FAMPHUR	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FENSULFOTHION	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FENTHION	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FLUORENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HEPTACHLOR	Pesticides	UG/KG	175	7720	0	0.02	0.62	22.60	0.00	0.00	0.00	0.00	0.00
HEXACHLOROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HEXACHLOROBUTADIENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HEXACHLOROCYCLOPENTADIENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HEXACHLOROETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HEXACHLOROPROPENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

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Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
INDENO(1,2,3-CD)PYRENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
IODOMETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ISODRIN	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ISOPHORONE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ISOSAFROLE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LEPTOPHOS	Pesticides	UG/KG	176	7750	0	0.35	9.84	319.00	0.00	0.00	0.00	0.00	0.00
LINDANE (GAMMA-BHC)	Pesticides	UG/KG	175	7720	0	0.17	2.54	37.80	0.00	0.00	0.00	0.00	0.00
LONGIFOLENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MALACHITE GREEN	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MALATHION	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MESTRANOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
METHAPYRILENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
METHOXYCHLOR	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
METHYL METHACRYLATE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
METHYL METHANESULFONATE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
METHYL PARATHION	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MEVINPHOS (PHOSDRIN)	Pesticides	UG/KG	176	7750	0	0.13	4.08	148.00	0.00	0.00	0.00	0.00	0.00
MIREX	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MONOCROTOPHOS	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N-NITROSODI-N-BUTYLAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

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Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
N-NITROSODIETHYLAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N-NITROSODIMETHYLAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N-NITROSOMETHYLETHYLAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N-NITROSOMETHYLPHENYLAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N-NITROSOMORPHOLINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N-NITROSOPIPERIDINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N,N-DIMETHYLFORMAMIDE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
NITROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
O-ANISIDINE	Organics	UG/KG	176	7750	0	6.72	100.00	1,550.00	0.00	0.00	0.00	0.00	0.00
O-TOLUIDINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
O-TOLUIDINE, 5-CHLORO-	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
P-DIMETHYLAMINOAZOBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
P-NITROANILINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PARATHION	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PENTACHLOROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PENTACHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PENTAMETHYLBENZENE	Organics	UG/KG	176	7750	0	204.00	3,300.00	53,400.00	0.00	0.00	0.00	0.00	0.00
PERYLENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PHENACETIN	Organics	UG/KG	176	7750	0	51.80	1,690.00	69,300.00	0.00	0.00	0.00	0.00	0.00
PHENOL, 2-METHYL-4,6-DINITRO-	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
PHENOTHIAZINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PHORATE	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PHOSMET	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PHOSPHAMIDON	Pesticides	UG/KG	176	7750	0	1.05	15.20	232.00	0.00	0.00	0.00	0.00	0.00
PHOSPHORIC ACID, TRIMETHYL ESTER	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PHOSPHORIC TRIAMIDE, HEXAMETHYL-	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PRONAMIDE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PYRIDINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
RESORCINOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SAFROLE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SQUALENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TERBUFOS	Pesticides	UG/KG	176	7750	0	39.50	630.00	16,700.00	0.00	0.00	0.00	0.00	0.00
TETRACHLORVINPHOS	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TETRAETHYLDITHIOPYROPHOSPHATE	Pesticides	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
THIANAPHTHENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
THIOACETAMIDE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
THIOXANTHE-9-ONE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TOLUENE, 2,4-DIAMINO-	Organics	UG/KG	176	7750	0	14.70	536.00	19,600.00	0.00	0.00	0.00	0.00	0.00
TOXAPHENE	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TRANS-1,3-DICHLOROPROPENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
TRANS-1,4-DICHLORO-2-BUTENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TRIBROMOMETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TRICHLOROFON	Pesticides	UG/KG	176	7750	0	2.13	69.60	2,530.00	0.00	0.00	0.00	0.00	0.00
TRIPHENYLENE	Organics	UG/KG	176	7750	0	11.50	422.00	15,400.00	0.00	0.00	0.00	0.00	0.00
TRIPROPYLENEGLYCOL METHYL ETHER	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
VINYL ACETATE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
VINYL CHLORIDE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1-BROMO-2-CHLOROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1-BROMO-3-CHLOROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1-CHLORO-3-NITROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1-METHYLFLUORENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1-METHYLPHENANTHRENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1-NAPHTHYLAMINE	Organics	UG/KG	176	7750	0	127.00	2,050.00	33,100.00	0.00	0.00	0.00	0.00	0.00
1-PHENYLNAPHTHALENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,1-DICHLOROETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,1-DICHLOROETHENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,1,1-TRICHLOROETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,1,1,2-TETRACHLOROETHANE	Organics	UG/KG	176	7750	0	87.60	1,410.00	22,900.00	0.00	0.00	0.00	0.00	0.00
1,1,2-TRICHLOROETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,1,2,2-TETRACHLOROETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

TABLE 4. NATIONAL POLLUTANT CONCENTRATION PERCENTILE ESTIMATES FROM THE NATIONAL SEWAGE SLUDGE SURVEY
Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
1,2-DIBROMO-3-CHLOROPROPANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,2-DIBROMOETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,2-DICHLOROBENZENE	Organics	UG/KG	176	7750	0	21.50	1,590.00	117,000.00	0.00	0.00	0.00	0.00	0.00
1,2-DICHLOROETHANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,2-DICHLOROPROPANE	Organics	UG/KG	176	7750	0	0.04	3.11	230.00	0.00	0.00	0.00	0.00	0.00
1,2-DIPHENYLHYDRAZINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,2,3-TRICHLOROBENZENE	Organics	UG/KG	176	7750	0	7.95	587.00	43,400.00	0.00	0.00	0.00	0.00	0.00
1,2,3-TRICHLOROPROPANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,2,3-TRIMETHOXYBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,2,4-TRICHLOROBENZENE	Organics	UG/KG	176	7750	0	51.40	2,580.00	184,000.00	0.00	0.00	0.00	0.00	0.00
1,2,4,5-TETRACHLOROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,3-BUTADIENE, 2-CHLORO	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,3-DICHLORO-2-PROPANOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,3-DICHLOROBENZENE	Organics	UG/KG	176	7750	0	6.82	416.00	30,600.00	0.00	0.00	0.00	0.00	0.00
1,3-DICHLOROPROPANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,3,5-TRITHIANE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,4-DINITROBENZENE	Organics	UG/KG	176	7750	0	16.60	272.00	4,400.00	0.00	0.00	0.00	0.00	0.00
1,4-NAPHTHOQUINONE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,4-NAPHTHOQUINONE, 2,3-DICHLORO-	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1,5-NAPHTHALENEDIAMINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

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Candidate Pollutants for Round Two Regulations

Nonparametric Substitution Method Estimation Procedure - Nondetects Set to zero

Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2-(METHYLTHIO)BENZOTHIAZOLE	Organics	UG/KG	176	7750	0	48.20	1,760.00	64,400.00	0.00	0.00	0.00	0.00	0.00
2-CHLOROETHYLVINYL ETHER	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-CHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-ISOPROPYLNAPHTHALENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-METHYLBENZOTHIOAZOLE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-NITROANILINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-NITROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-PHENYLNAPHTHALENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-PROPEN-1-OL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2-PROPENAL	Organics	UG/KG	176	7750	0	0.12	1.95	31.20	0.00	0.00	0.00	0.00	0.00
2-PROPENENITRILE, 2-METHYL-	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,3-BENZOFLUORENE	Organics	UG/KG	176	7750	0	163.00	5,960.00	218,000.00	0.00	0.00	0.00	0.00	0.00
2,3-DICHLOROANILINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,3-DICHLORONITROBENZENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,3,4,6-TETRACHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,3,6-TRICHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,4-DICHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,4-DIMETHYLPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,4-DINITROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,4-DINITROTOLUENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

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Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
2,4,5-TRICHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,4,6-TRICHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,6-DI-TERT-BUTYL-P-BENZOQUINONE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,6-DICHLORO-4-NITROANILINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,6-DICHLOROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2,6-DINITROTOLUENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
3-CHLOROPROPENE	Organics	UG/KG	176	7750	0	9.44	152.00	2,470.00	0.00	0.00	0.00	0.00	0.00
3-METHYLCHOLANTHRENE	Organics	UG/KG	176	7750	0	1.49	21.90	367.00	0.00	0.00	0.00	0.00	0.00
3-NITROANILINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
3,3'-DICHLOROBENZIDINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
3,3'-DIMETHOXYBENZIDINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
3,6-DIMETHYLPHENANTHRENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4-AMINOBIPHENYL	Organics	UG/KG	176	7750	0	143.00	2,310.00	37,400.00	0.00	0.00	0.00	0.00	0.00
4-BROMOPHENYL PHENYL ETHER	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4-CHLORO-2-NITROANILINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4-CHLORO-3-METHYLPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4-CHLOROPHENYLPHENYL ETHER	Organics	UG/KG	176	7750	0	104.00	3,730.00	136,000.00	0.00	0.00	0.00	0.00	0.00
4-NITROPHENOL	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4,4'-DDD	Pesticides	UG/KG	175	7720	0	0.07	5.30	391.00	0.00	0.00	0.00	0.00	0.00
4,4'-METHYLENEBIS(2-CHLOROANILINE)	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

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Pollutant	Type	Unit	Sample Size	POTWs	Percent Detect	Mean	Standard Deviation	Observed Maximum	99th Percentile	98th Percentile	95th Percentile	90th Percentile	50th Percentile
4,5-METHYLENE PHENANTHRENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
5-NITRO-O-TOLUIDINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
7,12-DIMETHYLBENZ(A)ANTHRACENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ACENAPHTHENE	Organics	UG/KG	176	7750	0	1.27	46.60	1,700.00	0.00	0.00	0.00	0.00	0.00
ACENAPHTHYLENE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ACRYLONITRILE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ANILINE	Organics	UG/KG	176	7750	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
dioxins a	Dioxins	NG/KG	174	7714	0	54.10	168.00	1,700.00	547.00	518.00	250.00	84.90	11.20
PCB a	Pesticides	UG/KG	175	7720	0	393.00	1,790.00	13,400.00	8,700.00	2,470.00	1,320.00	683.00	0.00
PCB-1016	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PCB-1221	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PCB-1232	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PCB-1242	Pesticides	UG/KG	175	7720	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

(a) Composites considered a detect if all individual congeners or PCBs are measured above the minimum level.

TABLE 5

SEMIQUANTITATIVE METALS in the NATIONAL SEWAGE SLUDGE SURVEY

CAS NUMBER	CHEMICAL NAME	CAS NUMBER	CHEMICAL NAME
7440699	BISMUTH	7440246	STRONTIUM
7440451	CERIUM	7704349	SULFUR
7429916	DYSPROSIUM	7440257	TANTALUM
7440520	ERBIUM	13494809	TELLURIUM
7440531	EUROPIUM	7440279	TERBIUM
7440542	GADOLINIUM	7440291	THORIUM
7440553	GALLIUM	7440304	THULIUM
7440564	GERMANIUM	7440337	TUNGSTEN
7440575	GOLD	7440611	URANIUM
7440064	PLATINUM	7440031	NIOBIUM
7440097	POTASSIUM	7440644	YTTERBIUM
7440100	PRASEODYMIUM	7440042	OSMIUM
7440155	RHENIUM	7440677	ZIRCONIUM
7440166	RHODIUM	7440053	PALLADIUM
7440188	RUTHENIUM	7723140	PHOSPHORUS
7440199	SAMARIUM	7553562	IODINE
7440202	SCANDIUM	7439885	IRIDIUM
7440213	SILICON	7439910	LANTHANUM
7440586	HAFNIUM	7439932	LITHIUM
7440600	HOLMIUM	7439943	LUTETIUM
7440746	INDIUM	7440008	NEODYMIUM

APPENDIX C

CALCULATION OF A "SQUARE WAVE" FOR THE GROUNDWATER PATHWAY

Potential human exposure and risk through the groundwater pathway are estimated for both land application and surface disposal of sewage sludge. To prepare input for the VADOFT model of pollutant transport through the unsaturated zone, it is conservatively assumed for both land application and surface disposal that the pollutant is consistently loaded into the top of the unsaturated zone at the maximum rate estimated by mass balance calculations. The duration of this constant pulse, or "square wave", is constrained so that the total mass of pollutant leaching or seeping from the site is conserved. Although the general approach is the same for both land application and surface disposal, details differ according to which management practice is being considered. This appendix provides a brief discussion of the methods for estimating the magnitude and duration of the "square wave" of pollutant loading for land application and both prototype facilities for surface disposal.

Land Application

Both inorganic and organic pollutants can accumulate in soil with repeated applications of sewage sludge. As described in Chapter 4, it is assumed that all competing pollutant loss processes for sewage sludge-amended soil can be approximated as first-order, and that coefficients describing the rate of loss to each process can be summed to yield a total or "lumped" coefficient for first-order loss. Losses at any time t can then be described as:

$$\frac{dM_t}{dt} = -K_{tot} M_t$$

where:

M_t = mass of pollutant in sewage sludge-amended soil at time t (kg) and
K_{tot} = total loss rate for the pollutant from sewage sludge-amended soil (yr^{-1}).

If pollutant loading to treated soil is approximated as a continuous process, the mass of pollutant in soil after t years of applications can be described by:

$$M_t = \int_0^t PA\ e^{-K_{tot}x}\ dx = \frac{PA}{K_{tot}} (1 - e^{-K_{tot}t})$$

where:

PA = total annual loading of pollutant to site (kg/yr).

As t approaches infinity, M_t therefore approaches $(PA)/K_{tot}$ and yearly loss approaches yearly loading.

For organic pollutants, it is assumed that sewage sludge has been applied repeatedly until steady-state is achieved. In other words, pollutant has accumulated in the soil until total yearly losses through erosion, degradation, leaching, and volatilization (which are assumed to be proportional to the concentration in soil) catch up with yearly loadings of pollutant to

soil. Estimates of risks from organic pollutants on land application sites are derived for this steady-state condition. The amplitude of the square wave pulse for the groundwater pathway model is therefore equal to the annual loading of pollutant multiplied by the fraction of annual loss attributable to leaching. The length of the square wave is equal to the length of the simulation (300 years).

For inorganic pollutants, this condition of steady-state is not necessarily reached. The leaching of inorganic pollutants from sewage sludge to groundwater depends not only on the cumulative loading of inorganic pollutants, but also on the period of time in which this cumulative loading takes place. It is assumed that after 20 years, applications are discontinued. To capture the risks associated with the peak rate at which inorganic pollutants leave the soil layer, the peak loss rate (calculated for the 20th year of application) is used for the calculations. The length of the square wave is calculated by dividing the total (cumulative) loading of pollutant by this maximum rate of loss:

$$TP = \frac{N\ PA}{PA\ (1-e^{-K_{tot}t})} = \frac{N}{(1-e^{-K_{tot}N})}$$

where:

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

Surface Disposal: Monofill Prototype

The modeling of the groundwater pathway for the monofill prototype of surface disposal is similar to that for land application. For both cases, it is assumed that the site receives repeated loadings of pollutant for the duration of its active lifetime. By analogy with the above discussion for land application, this maximum rate of loss from the facility can be described as a function of its yearly loading, yearly loss, and number of years of active operation:

$$K_{tot}\ M_{LF} = PA\ (1-e^{-K_{tot}\ LF})$$

where:

LF = active lifetime of monofill (yr),

M_{LF} = mass of pollutant in sewage sludge/soil at end of monofill's active lifetime (kg), and

PA = total annual loading of pollutant to monofill (kg/yr).

The length of time this maximum rate of loss could be maintained is then:

$$TP = \frac{LF\ PA}{PA\ (1-e^{-K_{tot}\ LF})} = \frac{LF}{1-e^{-K_{tot}\ LF}}$$

Surface Disposal: Surface Impoundment Prototype

For the surface impoundment prototype of surface disposal, calculations are based on the conservative assumption that steady-state is maintained for concentrations of pollutants within the liquid and sediment layers of the impoundment. It is also assumed that the flux of pollutant leaching from the impoundment is constant with respect to time, at least until the total mass of pollutant deposited in the impoundment has been depleted. For this prototype, the length of the square wave used for execution of the VADOFT model is therefore equal to the total mass of pollutant entering the impoundment each year, multiplied by the expected lifetime of the facility and divided by the amount lost each year:

$$TP = \frac{PA \cdot TF}{31{,}536{,}000 \cdot PA \cdot f_{act}} = \frac{TF}{31{,}536{,}000 \cdot f_{act}}$$

where:

PA	=	total annual loading of pollutant into the surface impoundment (kg/yr),
TF	=	estimated active lifetime of surface impoundment (sec),
31,536,000	=	constant to convert (sec) to (yr), and
f_{act}	=	fraction of each year's loading of pollutant lost during each year of the surface impoundment's active phase (dimensionless).

APPENDIX D

EVALUATION OF CANDIDATE POLLUTANTS FOR THE ROUND TWO SEWAGE SLUDGE REGULATION

EVALUATION OF CANDIDATE POLLUTANTS FOR THE ROUND TWO SEWAGE SLUDGE REGULATION

by

U.S. ENVIRONMENTAL PROTECTION AGENCY
401 M STREET, S.W.
WASHINGTON, D.C. 20460

AUGUST 1996

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1. INTRODUCTION

1.1 BACKGROUND

In 1987, Congress amended section 405 of the Clean Water Act (CWA) to require a comprehensive program to reduce the potential public health and environmental risks from the use or disposal of sewage sludge, which is solid, semi-solid, or liquid residue generated during the treatment of domestic sewage in a treatment works. Amended section 405(d) established a timetable for the development of the sewage sludge use or disposal regulations. The basis for the program Congress mandated to protect public health and the environment is the development of technical requirements or standards for sewage sludge use or disposal, and the implementation of the standards through a permit program.

Under the current section 405(d), EPA first had to identify toxic pollutants that may be present in sewage sludge in concentrations that may affect public health and the environment. Next, for each identified use or disposal practice, EPA had to publish regulations that specify management practices for sewage sludge that contains the toxic pollutants and establish numerical limits for the toxic pollutants. The management practices and numerical limits must be "adequate to protect public health and the environment from any reasonably anticipated adverse effect of each pollutant." Section 405(d) requires that EPA publish the sewage sludge regulations in two rounds and then review the regulations periodically to identify additional pollutants for regulation.

On February 19, 1993, EPA published the Round One sewage sludge regulation (i.e., the Standards for the Use or Disposal of Sewage Sludge - 40 CFR Part 503) in the *Federal Register* (58 *FR* 9248). It was amended subsequently on February 24, 1994 (59 *FR* 9095), and on October 25, 1995 (60 *FR* 54764).

A candidate list of pollutants for the second round of the sewage sludge regulations (i.e., Round Two) was provided to the District Court in Oregon in May 1993 (see Appendix D1). The final list of pollutants was submitted to the District Court in Oregon in November 1995 (see Appendix D2). The Round Two sewage sludge regulation is scheduled for proposal in December 1999 and for publication in December 2001.

To develop the final list of pollutants for the Round Two sewage sludge regulation, a Comprehensive Hazard Identification study was conducted by use or disposal practice for the 31 pollutants on the candidate list. Results of that study were used to determine the candidate pollutants that warrant further consideration for the Round Two list of pollutants.

1.2 PURPOSE

This paper reviews the candidate pollutants from the Comprehensive Hazard Identification study that warrant further consideration for the Round Two list of pollutants and presents the rationales for not including some of the pollutants on the final list. It also presents the pollutants on the final list of pollutants for the Round Two sewage sludge regulation.

1.3 POLICY DECISIONS

For the review of the candidate pollutants from the Comprehensive Hazard Identification study that warrant further consideration for the Round Two list, EPA made several policy decisions. They are:

- Uptake rates from non-sewage sludge studies (i.e., crops for which the uptake rates were obtained were not grown in sewage sludge-amended soil) are not appropriate for crops grown in sewage sludge-amended soils because sewage sludge is expected to "bind" pollutants and makes them less available for plant uptake

(Corey et al., 1987).

- Potential population effects are of greater concern than are individual effects for exposure pathways in which the Highly Exposed Individual (HEI) is a nonendangered animal.

- The route through which a pollutant is administered (e.g., in drinking water or food) in a toxicity study should be considered when determining the applicability of the study to an exposure pathway.

- A soil type for all land application sites and surface disposal sites of either sandy loam, shrinking clay, or sand is reasonable.

- A margin of safety that is smaller than the total uncertainty factor used for the Reference Dose (RfD) is reasonable in certain cases.

1.4 ADDITIONAL INFORMATION

Questions about the information in this paper should be addressed to:

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2. POLLUTANT EVALUATIONS

2.1 CANDIDATE POLLUTANTS THAT WARRANT CONSIDERATION

During the Comprehensive Hazard Identification study (U.S. EPA, 1996), 15 exposure pathways were evaluated for land-applied sewage sludge and two pathways were evaluated for sewage sludge placed on a surface disposal site. A pathway was considered "critical" for a pollutant if the risk level for a carcinogenic pollutant was 10^{-4} or higher; the ratio of exposure for a noncarcinogenic pollutant to its Reference Dose (RfD) was equal to or greater than one; or the risk quotient (RQ) for a pollutant for the ecological pathways was equal to or greater than one.

Based on the results of the Comprehensive Hazard Identification study, several of the candidate pollutants had critical pathways for land application and for surface disposal. The candidate pollutants and their critical pathways are presented in Table 2.1 for land application and Table 2.2 for surface disposal. The exposure pathway for incineration (i.e., inhalation) was not critical for any of the candidate inorganic pollutants. That pathway was not evaluated for the organic pollutants because organic pollutants are controlled by the allowable concentration of total hydrocarbons in the exit gas from a sewage sludge incinerator in the Part 503 regulation.

As indicated on Tables 2.1 and 2.2, dioxins, dibenzofurans, and coplanar polychlorinated biphenyls (PCBs) have several critical pathways. For this reason and because dioxins, dibenzofurans, and coplanar PCBs are bioaccumulative pollutants (i.e., they accumulate in human and animal tissues) with reproductive effects, EPA concluded that those pollutants should be on the final Round Two list of pollutants for land application and surface disposal.

TABLE 2.1 - POLLUTANTS WITH CRITICAL LAND APPLICATION PATHWAYS

Pollutant	Critical Ag Pathway	Critical Non-Ag Pathway
Aluminum	6	6(f,r,p)
Antimony	7,14	7(f,r); 10(f,p); 14(f,r,p,)
Barium	7,10,14	7(f,r); 10(f,r,p); 14(f,r,p)
Beryllium	14	14(f,r,p)
Boron		6(f,p)
Dioxins/furans	2,3,10,12,13,15	3(f,r,p); 10(f,r,p); 12(f,r,p); 13(f,r,p); 15(f,r,p)
Fluoride	6,10	6(f,r,p); 10(f,r,p)
Manganese	3,6,7,14	3(f,r,p); 4(f,r); 6(f,r,p); 7(f,r); 10(f,p); 14(f,r,p)
PCBs - coplanar	3,4,5,6,15	3(f,r,p); 4(f,r); 5(f,r); 6(f,r,p); 13(f,r); 15(f,r,p)
Thallium	3	3(f,r,p)
Tin	7	7(f,r)
Titanium	6	6(r)

Pathway 2 - residential home gardener
Pathway 3 - child ingesting sewage sludge
Pathway 4 - human ingesting animal products (foraging animals)
Pathway 5 - human ingesting animal products (grazing animals)
Pathway 6 - livestock ingesting forage/pasture
Pathway 7 - livestock ingesting sewage sludge
Pathway 10 - soil organism predators ingesting soil organisms
Pathway 12 - humans ingesting surface water and fish
Pathway 13 - humans inhaling volatilized pollutants
Pathway 14 - humans ingesting groundwater
Pathway 15 - breast-feeding infant
f - forest; r - reclamation site; p - public contact site; ag - agricultural land; non-ag - non-agricultural land

TABLE 2.2 - POLLUTANTS WITH CRITICAL SURFACE DISPOSAL PATHWAYS

Pollutants	Monofills	Surface Impoundments
Antimony	-	Ground water
Barium	-	Ground water
Beryllium	-	Ground water
Dioxins/furans	-	Air
Manganese	-	Ground water

EPA also concluded that the inorganic pollutants with critical pathways for land application and surface disposal should not be on the final list of pollutants for the Round Two regulation. The rationales for excluding those pollutants from the list are presented below.

2.2 INFORMATION USED TO DEVELOP RATIONALES TO EXCLUDE INORGANIC POLLUTANTS FROM FURTHER CONSIDERATION

The Comprehensive Hazard Identification study used to evaluate the candidate inorganic pollutants was, by design, conservative. After the critical pathways were identified for each pollutant, a detailed examination of each pathway was conducted by EPA to confirm that the pathway results supported inclusion of the pollutant on the final Round Two list of pollutants.

As part of the detailed examination for each critical pathway for a pollutant, three reviews were conducted. First, the assumptions made in conducting the pathway exposure assessment were reviewed. Next, the relevance of available toxicity data for a pathway to the Highly Exposed Individual (HEI) for the pathway was reviewed. Finally, the magnitude of the ratio of estimated exposure to the RfD for a noncarcinogenic pollutant in the non-ecological pathways or

the magnitude of the ratio of the estimated exposure to the toxicological reference value (TRV) for a pollutant in the ecological pathways was reviewed.

2.2.1 Land Application

The information in Tables 2.3, 2.4, 2.5, and 2.6 was used in the detailed examination of the critical land application pathways. Table 2.3 contains a summary of conservative assumptions for several of the critical pathways. Table 2.4 contains the Highly Exposed Individual (HEI) for each of the critical pathways, and Table 2.5 contains the measurement endpoint for each pollutant by critical pathway and the species used to develop the endpoint. Table 2.6 contains the results of the Comprehensive Hazard Identification study for each of the critical pathways.

TABLE 2.3 - SUMMARY OF CONSERVATIVE ASSUMPTIONS

Pathway	Conservative Assumption
3	One hundred percent of the material that the child ingests is sewage sludge, not a mixture of soil and sewage sludge.
4	Results from non-sewage-sludge studies can be used to develop pollutant uptake slopes into forage/pasture.
6	Herbivorous livestock or small herbivorous animals forage only on land on which sewage sludge has been applied; results from non-sewage-sludge studies can be used to develop pollutant uptake slopes into forage/pasture.
7	Herbivorous livestock graze only on land on which sewage sludge has been applied.
10	All of the soil organisms ingested by small mammals are exposed to sewage sludge-amended soil and, therefore, bioconcentrate pollutants.
14	The soil-water partition coefficient used is the lowest soil-water partition coefficient for sandy soil with a porewater pH of 5.

TABLE 2.4 - HIGHLY EXPOSED INDIVIDUALS FOR CRITICAL PATHWAYS

Pathway Number	Highly Exposed Individual (HEI)
3-agricultural	Child ingesting sewage sludge
3-non-agricultural	Child ingesting sewage sludge
4-non-agricultural	Human ingesting deer and elk
6-agricultural	Herbivorous livestock
6-non-agricultural	Herbivorous livestock (forest, reclamation site); small herbivorous mammal (forest, public contact site)
7-agricultural	Herbivorous livestock
7-non-agricultural	Herbivorous livestock
10-agricultural	Small insectivorous mammal ingesting soil organisms
10-non-agricultural	Small insectivorous mammal ingesting soil organisms
14-agricultural	Human ingesting ground water
14-non-agricultural	Human ingesting ground water

TABLE 2.5 - MEASUREMENT ENDPOINTS FOR CRITICAL PATHWAYS

Pollutant	Pathway Number	Endpoint/Species[1]
Aluminum	6	TRV/rat
Antimony	7 10 14	TRV/rat TRV/rat RfD/rat
Barium	7 10 14	TRV/rat TRV/rat RfD/human
Beryllium	3 14	CRL CRL
Boron	6	TRV/dog
Fluoride	6 10	TRV/mice TRV/mice
Manganese	3 4 6 7 10 14	RfD/human RfD/human TRV/rat TRV/rat TRV/rat RfD/human
Thallium	3	RfD/rat
Tin	7	TRV/rat
Titanium	6	TRV/mice

[1] CRL - cancer risk level
RfD - risk reference dose
TRV - toxicological reference value

TABLE 2.6 - RESULTS OF RISK ASSESSMENT FOR CRITICAL PATHWAYS

Pollutant	Pathway Number	RfD Ratio[1]	RQ[2]
Aluminum	6(ag,f,r,p)	-	80(f,p), 100(ag,r)
Antimony	7(ag,f,r) 10(f,p) 14(ag,f,r,p)	- - 20(ag),40(f),3(r),60(p)	1(ag,f,r) 3(f,p) -
Barium	7(ag,f,r) 10(ag,f,r,p) 14(ag,f,r,p)	- - 9(ag),20(f),1(r), 20(p)	40(ag,f,r) 10(ag,r), 50(f,p) -
Beryllium[1]	14(ag,f,r,p)	$7x10^{-4}$(ag),$9x10^{-4}$(f) $3x10^{-4}$(r),$1x10^{-3}$(p)	-
Boron	6(f,p)	-	4(f,p)
Fluoride	6(ag,f,r,p) 10(ag,f,r,p)	- -	10(ag,r),30(f,p) 5(ag,r),8(f,p)
Manganese	3(ag,f,r,p) 4(f,r) 6(ag,f,r,p) 7(ag,f,r) 10(f,p) 14(ag,f,r,p)	4(ag,p),3(f,r) 10(f), 40(r) - - - 700(ag),1000(f), 30(r),2000(p)	- 200(ag,r),800(f,p) 1(ag,f,r) 2(f,p) -
Thallium	3(ag,f,r,p)	2(ag,p),1(f,r)	-
Tin	7(ag,f,r)	-	2(ag,f,r)
Titanium	6(ag,r)	-	7(ag,r)

[1]Ratio of estimated exposure to Reference Dose (RfD). For beryllium, the value is a carcinogenic risk level.
[2]Risk Quotient - ratio of estimated exposure to Toxicological Reference Value (TRV).

ag - agricultural land; f - forest land; r - reclamation site; p - public contact site

2.2.2 Surface Disposal

During the Comprehensive Hazard Identification study for surface disposal, soil-water partition coefficients for sand with a porewater pH of 5 were used for the ground-water pathway. This is the same conservative assumption that was used in the groundwater pathway analyses for land application.

Results of the Comprehensive Hazard Identification study for the critical surface disposal pathways are presented in Table 2.7.

TABLE 2.7 - RESULTS FOR CRITICAL SURFACE DISPOSAL PATHWAYS

Pollutant	Pathway	Cancer Risk Level	RfD Ratio[1]
Antimony	Ground water	-	4
Barium	Ground water	-	1
Beryllium	Ground water	2×10^{-4}	-
Manganese	Ground water	-	90

[1] Ratio of estimated exposure to the Reference Dose (RfD).

2.3 RATIONALES FOR EXCLUDING INORGANIC POLLUTANTS FROM FURTHER CONSIDERATION

The rationales for excluding inorganic pollutants from the list of pollutants for the Round Two sewage sludge regulation for land application and surface disposal are presented below.

2.3.1 Land Application

Aluminum

The critical pathway for aluminum for land application is Pathway 6 (animal foraging) for both agricultural land and non-agricultural land (forest, reclamation sites, and public contact sites). As indicated in Table 2.3, the uptake slopes used in the Pathway 6 analyses were obtained from non-sewage sludge studies (i.e., crops from which the uptake slopes were obtained were not grown in sewage sludge-amended soil). EPA concluded it is not appropriate to use those uptake slopes to estimate the uptake of aluminum into forage grown in sewage sludge-amended soils (see Policy Decision on page 2). No other information was available on uptake slopes for aluminum.

Because aluminum is not a bioaccumulative pollutant (i.e., does not accumulate in human or animal tissue); because Pathway 6 was the only critical pathway for aluminum from the Comprehensive Hazard Identification study; and because after the detailed review of Pathway 6, it could not be evaluated using available information, EPA concluded that aluminum should not be on the list of pollutants for the Round Two regulation for land application.

Antimony

One of the critical pathways for antimony for land application is Pathway 7 (grazing animal that ingests sewage sludge directly). As indicated in Tables 2.3 and 2.5, the measurement endpoint (i.e., the toxicological reference value (TRV)) for this pathway for both agricultural and non-agricultural land is based on results of studies using laboratory animals (i.e., rats). This endpoint was extrapolated to the appropriate HEI (i.e., herbivorous animals) for the land application risk assessments.

The lowest observed adverse effect level (LOAEL) for antimony is 0.262 mg/kg-body weight/day, which is based on the results of a study in which antimony was fed to rats in water (Schroeder et al., 1970). This value was converted to a dietary value (i.e., 3.4 mg/kg-food) using a standard body weight of 0.4 kilograms for a rat and allometric equations (U.S. EPA, 1988). A decrease in survival and longevity for male and female rats was observed at this dose equivalent. The dietary value was divided by 10 to obtain the TRV for antimony.

There are two reasons why it is not appropriate to use the TRV for laboratory animals as the TRV for the HEI in the Pathway 7 exposure analyses for agricultural land, forests and reclamation sites. First, the study on which the LOAEL for antimony was based (Schroeder, 1970) indicates that the effect from exposure to antimony (a decrease in survival and longevity) occurs later in the life of a rat and growth was not affected. Thus, the potential for antimony to interfere with growth and reproduction (i.e., population effects) is unclear. Also, results of another study (Schroeder et al., 1968a) indicate a decrease in survival and longevity due to exposure to antimony was not observed in mice.

Second, the LOAEL on which the TRV is based was obtained from a study in which antimony was fed to rats in water. Gastrointestinal absorption of antimony in food is expected to be lower than the gastrointestinal absorption of antimony in drinking water. For example, results of other rat studies (Sunagawa, 1981; Smyth and Thompson, 1945) in which antimony was administered in food indicate that the no observed adverse effect level (NOAEL) for antimony can be as high as 200 mg/kg-day and not cause specific systemic effects (e.g., changes in blood pressure). This value, which did not result in population effects, is over two orders of magnitude higher than the LOAEL used to develop the TRV for antimony.

Because there is uncertainty in the animal studies about whether exposure to antimony causes population effects and because the LOAEL used to develop the TRV is based on the results of a study in which rats were fed antimony in water, EPA concluded that it is not appropriate to use the TRV in the Comprehensive Hazard Identification study in the Pathway 7 analyses. In those analyses, the HEI ingests sewage sludge while grazing on sewage sludge-amended soil. Other TRV values would likely be much higher based on other toxicity data. If the TRV is based on a NOAEL of 200 mg/kg-day (i.e., the NOAEL from rat studies in which rats were fed antimony in food), the risk quotient for the Pathway 7 analyses would be less than one. For these reasons, EPA concluded that antimony should not be on the Round Two list of pollutants based on exposure through Pathway 7.

Pathway 10 (predator of soil organism) also was critical for antimony for land application. EPA concluded that the TRV used in the Comprehensive Hazard Identification study is not appropriate for this pathway for the same reasons the TRV for Pathway 7 is not appropriate. Given that the RQ was 3 and that other TRV values would likely be much higher based on other toxicity data, EPA concluded antimony should not be included on the final Round Two list of pollutants for land application based on exposure through Pathway 10.

Pathway 14 (i.e., ground water) in the land application Comprehensive Hazard Identification study for agricultural land and non-agricultural land also was critical for antimony. One way to evaluate the RfD ratio for this pathway (i.e., the highest ratio is 60 for public contact sites) is to consider the uncertainty factor for the RfD with respect to the RfD ratio and the effect upon which the RfD is based.

The antimony RfD is based on an uncertainty factor of 1000 (IRIS, 1996). The highest

RfD ratio for this pathway is 60 for public contact sites. In this case, the margin of safety (that is, the ratio between the uncertainty factor and the RfD ratio) is approximately 17 (i.e., 1000/60). EPA concluded that a margin of safety of 17 is sufficiently protective for the HEI (i.e., human) in this case because the effect upon which the RfD is based (i.e., changes in cholesterol and glucose blood levels) is not severe and is likely reversible. EPA also concluded that the margins of safety for the other types of land application sites (i.e., 50 for agricultural land, 25 for forest, and 333 for reclamation sites) are protective for the HEIs for those types of land application sites.

The above information indicates that the critical pathways from the Comprehensive Hazard Identification study should not be used as the basis for including antimony on the list of pollutants for the Round Two sewage sludge regulation. For this reason, antimony was not included on the list for land application.

Barium

One of the critical pathways for barium for land application was Pathway 7 (grazing animal that ingests sewage sludge directly). The TRV for this pathway for agricultural land, forest, and reclamation sites is based on results of studies using laboratory animals (i.e., rats). This endpoint was extrapolated to the appropriate HEI (i.e., herbivorous animals) for the land application risk assessments.

Study results reported in the Agency for Toxic Substances and Disease Registry (ATSDR, 1992a) were used as the basis for the TRV for barium. In those studies (Perry et al., 1983, 1985, 1989), barium was fed to rats in drinking water. The NOAEL for barium was 0.056 mg/kg- body weight/day, which corresponds to a concentration in drinking water of 1 ppm. The dietary

equivalent is 0.7 mg/kg-food when the NOAEL is converted using allometric equations (U.S. EPA, 1988).

The RQ for barium for Pathway 7 was 40. Even if the LOAEL were used as the basis for the TRV, instead of the NOAEL, the RQ would be 4. This means that the estimated exposure for Pathway 7 could cause the LOAEL to be exceeded.

The effect for the LOAEL for barium is an increase in systolic blood pressure. This effect was not seen, however, until the eight month of a 16 month rat study. No other toxic effects were observed in the study, and growth was not impaired. The impact of slight increases in systolic blood pressure for cattle, other grazing animals, and small mammals is unclear, and population effects (i.e., growth, reproductive, and mortality) for those animals cannot be evaluated using the results of the rat study.

A 1975 study found reduced life span in male mice given 5 ppm barium in drinking water (Schroeder and Mitchener, 1975). The calculated LOAEL for this study was 0.95 mg/kg-body weight/day, which has a dietary equivalent of 4.8 mg/kg-food when converted using an allometric equation (U.S. EPA, 1988). During the study, longevity only was reduced slightly. Other studies in which cardiovascular and other systemic effects from exposure to barium were evaluated found NOAELs at an order of magnitude higher than in the NOAEL based on the results of the Perry et al. studies.

EPA concluded that it is not appropriate to use the above TRV as the TRV for the HEI in the Pathway 7 exposure analyses because the observed effects from exposure to barium, which is a non-bioaccumulative pollutant, were not population effects. In addition, the effects that were observed (i.e., increase in systolic blood pressure) occurred as a result of exposure to barium in

drinking water. The absorption of barium in drinking water is likely to be higher than absorption of barium in food or in sewage sludge. For these reasons, EPA concluded that barium should not be on the Round Two list of pollutants for land application based on exposure through Pathway 7.

Pathway 10 (predator of soil organism) also was critical for barium in the Comprehensive Hazard Identification study for agricultural land and non-agricultural land. EPA concluded that the TRV used in that study for Pathway 10 is not appropriate for the same reasons the TRV for Pathway 7 is not appropriate. Therefore, EPA concluded barium should not be on the Round Two list of pollutants based on exposure through Pathway 10.

Pathway 14 (i.e., ground water) also was critical for barium for agricultural and non-agricultural land application. Two conservative assumptions were made for this pathway in the Comprehensive Hazard Identification study. One was the type of soil at the land application sites and the other was the value for the soil-water partition coefficient (K_d).

The type of soil affects the ability of a pollutant to move vertically to an aquifer and laterally to a nearby well. Soil types in the unsaturated zone beneath a land application site in order of increasing pollution potential are: (1) nonshrinking clay, (2) clay loam, (3) silty loam, (4) loam, (5) sandy loam, (6) shrinking clay, (7) sand, (8) gravel, and (9) thin or absent soil (U.S. EPA, 1992). EPA concluded that it is reasonable to assume a soil type of either sandy loam, shrinking clay, or sand as the soil type for all land application sites. In the case of barium, the assumed soil type for the land application sites was sand.

The K_d value for sand with a porewater pH of 5 varies from 6 liters per kilogram to 174 liters per kilogram (Gerritse et al., 1982). In the Comprehensive Hazard Identification study,

Pathway 14 was critical for barium because the lower end of the K_d range (i.e., 6) was used to estimate exposure from barium. If the upper end of the K_d range (i.e., 174) is used, Pathway 14 is not critical (i.e., the RfD ratio is less than one) for barium.

EPA concluded that because there is an acceptable range of partition coefficients, it is appropriate to use the upper end of the range, particularly when the soil type for all land application sites is assumed to be sand. Because Pathway 14 is not critical when the upper end of the partition coefficient range is used, EPA concluded that barium should not be on the Round Two list of pollutants for land application based on exposure through Pathway 14.

The above information indicates that after the detailed examination of the critical pathways for barium (i.e., 7, 10, and 14) in the Comprehensive Hazard Identification study, none of the pathways are critical for both agricultural land and non-agricultural land. For this reason, barium was not included on the final list of pollutants for the Round Two regulation for land application.

Beryllium

Pathway 14 was critical for beryllium for both agricultural and non-agricultural land (forest, reclamation sites, and publication sites). As mentioned previously, the assumed soil type and the partition coefficient are important for this pathway.

In the case of beryllium, the assumed soil type for all land application sites is sand. This is a reasonable assumption, particularly for agricultural land. Loam soils (sandy loam, silty loam, silty clay loam) are predominant on agricultural land throughout the United States (sand and sandy loams predominate in the southeast). Of the loam soils, sandy loam has the highest pollution potential (U.S. EPA, 1992).

During the Comprehensive Hazard Identification study, the partition coefficient at the lower end of the range of partition coefficients for sand with a porewater pH of 5 was used. EPA concluded that because a reasonable soil type was used, it is appropriate to use any of the partition coefficients in the range of partition coefficients.

When the median partition coefficient value for sand with a porewater pH of 5 is used, Pathway 14 is not critical for beryllium (i.e., the cancer risk level is lower than 10^{-4}). For this reason, beryllium was not included on the final list of pollutants for the Round Two sewage sludge regulation for land application.

Boron

The critical pathway for boron for land application is Pathway 6 (animal foraging) for forest and reclamation sites. None of the pathways for agricultural land were critical for boron.

The uptake slopes used in the Pathway 6 analyses were obtained from non-sewage-sludge studies (i.e., crops for which the uptake slopes were obtained were not grown in sewage sludge-amended soil). EPA concluded that it is not appropriate to use those uptake slopes to estimate risks from boron in crops grown in sewage sludge-amended soils (see Policy Decision on page 2).

No other information is available on uptake slopes for boron. Because Pathway 6 was the only critical pathway for boron and because this pathway could not be evaluated using available information after the detailed examination of the critical pathways, EPA concluded that boron should not be on the list of pollutants for the Round Two regulation for land application.

Fluoride

Pathways 6 and 10 were critical for fluoride for both agricultural land and non-agricultural land (i.e., forest, reclamation sites, and public contact sites) in the Comprehensive Hazard Identification study. For Pathway 6 (animal foraging), the uptake slopes used in the analyses were obtained from non-sewage-sludge studies (i.e., crops from which the uptake slopes were obtained were not grown on sewage sludge-amended soils). EPA concluded it is not appropriate to use those uptake slopes to estimate risks from fluoride in forage grown in sewage sludge-amended soils (see Policy Decision on page 2).

No other information is available on uptake slopes for fluoride. Because Pathway 6 could not be evaluated using existing information after completion of the detailed examination of the critical pathways, EPA concluded that Pathway 6 is not critical. For this reason, fluoride was not included on the list of pollutants for the Round Two regulation for land application based on exposure through Pathway 6.

Pathway 10 (predator of soil organism) also was critical for fluoride for agricultural land and non-agricultural land. The TRV for this pathway was based on a NOAEL of 10 mg/L in drinking water administered to mice (Kanisawa and Schroeder, 1969). This was converted to a dietary equivalent value of 11 mg/kg-food using allometric equations (U.S. EPA, 1988). Results of other studies indicate that a dietary equivalent value for fluoride of 52 mg/kg-food resulted in changes in teeth and liver, and structural and functional changes in the kidney (Jankauskas, 1974; Lim et al., 1975; Roman et al., 1977, as cited in IARC, 1982).

The HEI for Pathway 10 is the predator of a soil organism (e.g., a shrew). The effect from the exposure in Pathway 10 is mild systemic changes (e.g., changes in teeth and liver).

Population effects from this exposure are unknown.

Because the effect for which the TRV is protective is mild systemic changes and not population effects, and because there is some evidence that fluoride is necessary for fertility in mice (Messer et al. 1973, as cited in IARC, 1982), EPA concluded that the TRV used in the Comprehensive Hazard Identification study was not appropriate. For this reason, and because no other relevant toxicological information on small mammals was available for Pathway 10, EPA concluded that Pathway 10 could not be evaluated for fluoride. Thus, Pathway 10 is not critical for fluoride.

The above information indicates that the critical pathways from the Comprehensive Hazard Identification study should not be the basis for including fluoride on the Round Two list of pollutants. For this reason, fluoride was not placed on the list of pollutants for the Round Two sewage sludge regulation for land application.

Manganese

Pathways 3, 6, 7, and 14 were critical for manganese for agricultural land. Pathways 3, 4, 6, 7, 10, and 14 were critical for manganese for non-agricultural land.

Pathway 3 is the child ingestion pathway. For agricultural land and public contact sites, a child between the ages of 1 and 6 is assumed to ingest 0.2 gram of sewage sludge (not the sewage sludge-soil mixture) daily. For forest and reclamation sites, a child between the ages of 4 and 6 is assumed to ingest 0.2 grams of sewage sludge daily.

The Reference Dose (RfD) for the Pathway 3 analyses in the Comprehensive Hazard Identification study was 0.005 milligrams of manganese per kilogram of body weight per day.

On May 1, 1996, the RfD for manganese in EPA's Integrated Risk Information System (IRIS) was changed. The current RfD for manganese in IRIS is 0.14 milligrams of manganese per kilogram of body weight per day for dietary exposure. As indicated in the Uncertainty and Modifying Factors section in IRIS, when assessing exposure to manganese from food, a modifying factor of one is used. When assessing exposure to manganese from drinking water or soil, a modifying factor of three is used. Because the HEI ingests sewage sludge, which is similar to soil, an uncertainty factor of three was applied to the RfD. The RfD for the Pathway 3 analyses should be 0.14 divided by 3, resulting in 0.05 milligrams of manganese per kilogram of body weight per day.

Using the current RfD for manganese, the RfD ratio for Pathway 3 is 0.4 for agricultural land and public contact sites, and 0.3 for forest and reclamation sites. Because these values are less than one, Pathway 3 is not critical for manganese. For this reason, EPA concluded that manganese should not be on the list of pollutants for the Round Two sewage sludge regulation based on exposure through Pathway 3.

The uptake slopes in Pathway 4, which was critical for forest and reclamation sites, were obtained using non-sewage-sludge studies (i.e., crops from which the uptake slopes were obtained were not grown in sewage sludge-amended soils). EPA concluded that it is not appropriate to use those uptake slopes for crops grown in sewage sludge-amended soils (see Policy Decision on page 2). Because there is no other information on manganese uptake slopes, manganese was not included on the Round Two list of pollutants based on exposure through Pathway 4.

Pathway 6 also was critical for manganese for agricultural land, forests, reclamation sites, and public contact sites. EPA concluded that manganese should not be included on the Round

Two list of pollutants based on exposure through this pathway because the uptake slopes used in the analyses were obtained from non-sewage-sludge studies. It is not appropriate to use those uptake slopes for crops grown in sewage sludge-amended soils (see Policy Decision on page 2).

The TRVs for Pathway 7, which was critical for manganese for agricultural land, forest, and reclamation sites, and for Pathway 10, which was critical for forest and public contact sites, are based on the results of studies using laboratory animals (i.e., rats). After reviewing the results in the original study (Laskey et al., 1982) used to develop the TRV, an error was found in the dietary value. The dietary value used to develop the TRV in the Comprehensive Hazard Identification study was 170 mg/kg-food. This value was divided by 10 to determine the TRV.

The dietary value in the Laskey study was 350 mg/kg-food. Thus, the TRV should have been 35 mg/kg-food instead of 17 mg/kg-food. When the revised TRV was used to calculate the RQs for Pathways 7 and 10, the RQ for Pathway 7 was 0.7 and the RQ for Pathway 10 was just 1. Therefore EPA concluded that manganese should not be included on the Round Two list of pollutants, because the RQ became less than one for one pathway, and just met the level of concern for the other pathway.

The final pathway that was critical for manganese is Pathway 14 (i.e., ground water). This pathway was critical for both agricultural land and non-agricultural land (i.e., forest, reclamation sites, and public contact sites).

Two of the important variables for this pathway are soil type and partition coefficient. As mentioned previously, EPA concluded that assuming a soil type of either sandy loam, shrinking clay, or sand is conservative. During the detailed examination of the critical pathways, the assumed soil type for Pathway 14 for manganese was sandy loam, not sand.

The partition coefficient for sandy loam with a porewater pH of 8 ranges from 8418 liter per kilogram to 15,774 liter per kilogram (Gerritse et al., 1982). Using any partition coefficient within that range is conservative. For the Pathway 14 analysis for manganese, when a value in the middle of the above range is used, Pathway 14 is not critical for manganese.

The above information indicates that after completion of the detailed examination of the critical pathways for manganese from the Comprehensive Hazard Identification study, none of the pathways are considered to be critical for agricultural land and non-agricultural land. For this reason, manganese was not included on the final list of pollutants for the Round Two sewage sludge regulation for land application.

Thallium

The critical pathway for thallium for agricultural land, forest, reclamation sites, and public contact sites was Pathway 3 - child ingestion of sewage sludge. In the Comprehensive Hazard Identification study, the ratio of exposure from Pathway 3 to the RfD for thallium was two.

The thallium RfD is based on the results of a 90-day study during which rats ingested soluble thallium salts in drinking water (IRIS, 1996). The uncertainty factor in the RfD is 3,000. In the case of the Pathway 3 analysis, the margin of safety is 1,500 (i.e., 3,000 divided by an RfD ratio of 2).

The absorption of metals like thallium in sewage sludge in the gastrointestinal tract after the sewage sludge is ingested by a child is expected to be lower than the absorption of soluble salts of thallium. For this reason and because the margin of safety for the Pathway 3 analysis is 1,500, EPA concluded that Pathway 3 was not critical for thallium. Thus, thallium was not

included on the final list of Round Two pollutants for land application for either agricultural land or non-agricultural land.

Tin

The critical pathway for tin for agricultural land, forest, and public contact sites was Pathway 7 (i.e., grazing animal that ingests sewage sludge directly). The TRV for tin was based on the results of a study in which female rats were fed 5 ppm tin in drinking water (Schroeder et al., 1968b). The observed effect in this study was decreased longevity.

The LOAEL reported in ATSDR (1992b) was 0.7 mg/kg/day, which is equivalent to a dietary value of 9 mg/kg-food. This value was divided by 10 to obtain a TRV for Pathway 7 of 0.9 mg/kg-food. When reviewing the original study on which the TRV is based, an error was found. The TRV should be 0.45 mg/kg-food, which means the RQ for tin for agricultural land, forest, and reclamation sites should have been four instead of two.

Studies other than the Schroeder et al. study (1968b) failed to find any effects in mice administered 5 ppm tin in drinking water (Schroeder and Balassa, 1967). In addition, other studies examining systemic effects in rats and mice found NOAELs an order of magnitude or more higher than the LOAEL from the Schroeder et al. study (1968b). Effects observed in these studies are not clear with respect to population effects from exposure to tin.

Because the LOAEL used to calculate the TRV for tin is from a study in which rats were administered tin in drinking water (absorption of tin in food or sewage sludge is likely to be lower than absorption of tin in drinking water); because results of other studies indicate that the NOAEL for tin is higher than the LOAEL from the Schroeder et al. study (1968b); and because

the population effects from exposure to tin are not known, EPA concluded that the TRV from the rat study should not be used as the TRV for the HEI in Pathway 7.

Because there is no other more appropriate information on the TRV for tin, Pathway 7 could not be evaluated for tin after completion of the detailed examination of the critical pathways. For this reason, Pathway 7 is not critical for tin, and tin was not included on the Round Two list of pollutants based on exposure through Pathway 7.

Titanium

The critical pathway for titanium for agricultural land and reclamation sites was Pathway 6 (i.e., animal foraging on sewage sludge-amended soils). The uptake slopes used in the Pathway 6 analyses were obtained from non-sewage-sludge studies (i.e., crops from which the uptake slopes were obtained were not grown in sewage sludge-amended soils). EPA concluded that it is not appropriate to use uptake slopes from non-sewage-sludge studies for forage grown in sewage sludge-amended soils (see Policy Decision on page 2).

No other information is available on uptake slopes for titanium. Because Pathway 6 could not be evaluated using available information, EPA concluded that Pathway 6 is not critical and that titanium should not be on the list of pollutants for the Round Two sewage sludge regulation for land application based on exposure through Pathway 6.

2.3.2 Surface Disposal

Antimony and Barium

The critical pathway for antimony and barium for surface disposal is the groundwater pathway. As mentioned in the above discussion of antimony for land application, one way to evaluate the RfD ratio (i.e., four for antimony and one for barium) for this pathway is to consider the uncertainty factor for the RfD with respect to the RfD ratio and the effect for which the RfD is protective.

The antimony and barium RfDs are based on an uncertainty factor of 1000. The margin of safety for a surface disposal site (i.e., surface impoundment) would be 250 (i.e., 1000 divided by 4) for antimony and 1000 (i.e., 1000 divided by one) for barium. EPA concluded that for antimony a margin of safety of 250 is sufficiently protective for the HEI (i.e., human) in this case because the effect upon which the RfD is based (i.e., changes in cholesterol and glucose blood levels) is not severe and is likely reversible. EPA also concluded that barium just met the critical pathway criteria. For these reasons, EPA concluded after completion of the detailed examination of the critical pathways that antimony and barium should not be on the Round Two list of pollutants for surface disposal based on exposure through the groundwater pathway.

Beryllium and Manganese

The groundwater pathway also was the critical pathway for beryllium and manganese for surface disposal. As mentioned previously during the discussion of the groundwater pathway for land application, two important parameters for the groundwater pathway are soil type and soil-water partition coefficient.

During the Comprehensive Hazard Identification study, the soil type for all surface disposal sites was assumed to be sand. EPA concluded that using a soil type of either sandy loam, shrinking clay, or sand is conservative.

A soil-water partition coefficient for sandy soil with a porewater pH of 5 was used in the Comprehensive Hazard Identification study for surface disposal. However, if the median partition coefficient for sandy loam with a porewater pH of 8 is used in the analysis, the groundwater pathway is no longer critical for beryllium and manganese for surface disposal.

EPA concluded that it is reasonable to use the sandy loam soil type in the surface disposal groundwater analysis. It is also reasonable to use the median value for partition coefficient in the range of partition coefficients for sandy loam soil in the analysis. When this value is used, the groundwater pathway is not critical for beryllium and manganese for surface disposal. For this reason, EPA concluded that those pollutants should not be on the final list of pollutants for the Round Two regulation for surface disposal based on exposure through the groundwater pathway.

2.3.3 Incineration

Results of the Comprehensive Hazard Identification study indicate that no pollutants warrant consideration for the list of pollutants for the Part 503 Round Two regulation for incineration. Dioxins/furans will be re-evaluated for the Part 503 use or disposal practices, including incineration, at the completion of EPA's dioxin reassessment.

2.4 POLLUTANTS RECOMMENDED BY OTHERS FOR THE ROUND TWO LIST OF POLLUTANTS

Prior to conducting the Comprehensive Hazard Identification study for the 31 candidate pollutants for the Round Two list of pollutants, EPA programs and experts from outside of EPA were contacted to obtain data (e.g., plant and animal uptake data) on the 31 candidate pollutants. Comments were received from Dr. George O'Connor from the University of Florida and Dr. Rufus Chaney from the U. S. Department of Agriculture (see Appendix D3).

Dr. O'Connor provided references on plant bioavailability for some of the candidate organic pollutants. Information from those references was used in the Comprehensive Hazard Identification study, where applicable.

Dr. Chaney also provided information on several of the candidate pollutants. He recommended that beryllium, boron, dioxins/furans, coplanar polychlorinated biphenyls, cobalt, fluoride, and iron be on the Round Two list of pollutants for land application.

With the exception of cobalt and iron, the pollutants that Dr. Chaney recommended for the Round Two list of pollutants for land application were evaluated in the Comprehensive Hazard Identification study. The results of the detailed examination of the critical pathways for those pollutants are presented in other sections of the Technical Support Document (U.S. EPA, 1996).

Both cobalt and iron were evaluated for the list of pollutants for the Part 503 Round One regulation for land application. Neither pollutant was include on the Round One list of pollutants.

Cobalt was not included on the Round One list of pollutants because the hazard index (estimated exposure divided by the reference dose) was less than one. Dr. Chaney stated that

results of cobalt feeding trials indicate that a cobalt concentration between 5 and 10 milligrams per gram of diet may be injurious to sheep and cattle. Cobalt was detected in nine percent of the samples from the National Sewage Sludge Survey. Using the 98th percentile concentration for cobalt from the NSSS (i.e., 104 mg/kg with non-detected values set equal to the minimum level) and the fraction of the animal's diet that is sewage sludge used in the Round One risk assessments (i.e., 1.5 percent), the 5-10 milligram per kilogram diet concentration for cobalt is not expected to be reached in an animal's diet from ingestion of sewage sludge. In addition, none of the updated information submitted by Dr. Chaney suggests that the original hazard index for cobalt would change. For these reasons, EPA concluded that cobalt should not be on the list of pollutants for the Part 503 Round Two regulation for land application.

Iron was not included on the Round One list of pollutants even though the hazard index for grazing animals that ingest the sewage sludge/soil mixture (i.e., Pathway 7) was 2.1. The rationale for not including iron on the Round One list was that the grazing animal index was based on a worst worst-case sewage sludge iron concentration and the assumption that five percent of the animal's diet is sewage sludge. If sewage sludge with a "typical" iron concentration (i.e., 28,000 mg/kg (U.S. EPA, 1985)) is used in the analysis, the hazard index for grazing animals is less than one. The hazard index for iron also is expected to be less than one if the fraction of the animal's diet from the risk assessment for the Round One regulation (i.e., 1.5 percent) and the 90th percentile concentration for iron from the NSSS (i.e., 41,800 mg/kg) are used to develop the index. For these reasons, EPA concluded that iron should not be on the list of pollutants for the Part 503 Round Two regulation for land application.

3. LIST OF POLLUTANTS FOR THE ROUND TWO SEWAGE SLUDGE REGULATION

On November 30, 1995, EPA submitted the list of pollutants for the Round Two sewage sludge regulation to the District Court in Oregon. The court notice is presented in Appendix D2.

After considering information from the Comprehensive Hazard Identification study; the rationales for deleting inorganic pollutants from the list of pollutants that warranted further consideration; and information received from others, EPA concluded that two pollutants should be on the list for each use or disposal practice. They are: dioxins/furans (all monochloro to octachloro congeners) and polychlorinated biphenyls (coplanar). The court notice indicates that EPA may, in the exercise of its discretion, determine to add or delete other pollutants to or from this list at the time the Round Two regulation is proposed.

In addition to the list of pollutants submitted to the court, EPA may change a limit for the pollutants in the Round One regulation during development of the Round Two regulation. For this reason, the Round One pollutants also are considered pollutants for the Round Two sewage sludge regulation.

Including the pollutants from the Round One regulation, the list of pollutants for the Round Two sewage sludge regulation by use or disposal practice is:

Land application

arsenic, cadmium, copper, lead, mercury, molybdenum, nickel, selenium, zinc, dioxins/furans, and coplanar polychlorinated biphenyls.

Surface disposal

arsenic, chromium, nickel, dioxins/furans, and coplanar polychlorinated biphenyls

Sewage sludge incineration

arsenic, beryllium, cadmium, chromium, lead, mercury, nickel, total hydrocarbons (or carbon monoxide), dioxins/furans, and coplanar polychlorinated biphenyls

Dioxins/furans were included on the list of pollutants for sewage sludge incineration even though results of the screening risk assessments indicate that no pollutant warrants consideration for the Round Two list of pollutants for incineration. EPA currently is conducting a reassessment of dioxins/furans. Because the results of this assessment are unknown, dioxins/furans were included on the Round Two list of pollutants for all use or disposal practices. At the completion of the dioxin reassessment, EPA may decide not to regulate dioxins/furans for a particular use or disposal practice or may decide to regulate dioxins/furans on an accelerated schedule.

4. REFERENCES

Agency for Toxic Substances and Disease Registry. 1992a. Toxicological Profile for Barium and Compounds. Prepared by Clement International Corporation under contract no. 205-88-0608. U:S. Public Health Service. ATSDR/TP-91/03.

Agency for Toxic Substances and Disease Registry. 1992b. Toxicological Profile for Tin and Compounds. Prepared by Life Systems under subcontract to Clement International Corporation under contract no. 205-88-0608. U.S. Public Health Service. ATSDR/TP-91/27.

Corey, R.B., L.D. King, C. Lue-Hing, D.S. Fanning, J.J. Street, and J.M. Walker. 1987. Effects of Sludge Properties on Accumulation of Trace Elements by Crops. In: Land Application of Sludge. A.L. Page, T.J. Logan, and J.A. Ryan. Lewis Publishers, Inc. Chelsea, MI.

Gerritse, R.G., R. Vriesema, J.W. Dalenberg, and H.P. De Roos. 1982. Effect of Sewage Sludge on Trace Element Mobility in Soils. Journal of Environmental Quality. 11(3):359-364.

IARC (International Agency for Research on Cancer). 1982. IARC Monographs on the Evaluation of the Carcinogenic Risk of Chemicals to Humans. Some Aromatic Amines, Anthraquinones and Nitroso Compounds, and Inorganic Fluorides Used in Drinking-water and Dental Preparations. Vol. 27.

IRIS. 1996. Integrated Risk Information System. June.

Jankauskas, J. 1974. Effects of Fluoride on the Kidney (A Review). Fluoride. 7:93-105. [As cited in IARC, 1982].

Kanisawa, M. and H.A. Schroeder. 1969. Life Term Studies on the Effect of Trace Elements on Spontaneous Tumors in Mice and Rats. Cancer Res. 29:892-895.

Laskey, J.W., G.L. Rehnberg, J.F. Hein, and S.D. Carter. 1982. Effects of Chronic Manganese (Mn_3O_4) Exposure on Selected Reproductive Parameters in Rats. J. Toxicol. Environ. Health. 9:677-687.

Lim. J.K.J., G.K. Jensen, and O.H. King, Jr. 1975. Some Toxicological Aspects of Stannous Fluoride After Ingestion as a Clear, Precipitate Free Solution Compared to Sodium Fluoride. J. Dent. Res. 54:615-625. [As cited in IARC, 1982].

Messer, H.H., W.D. Armstrong, and L. Singer. 1973. Influence of Fluoride Intake on Reproduction in Mice. J. Nutr. 103:1319-1327. [As cited in IARC, 1982].

Perry, H.M., Jr., S.J. Kopp, M.W. Erlanger, and E.F. Perry. 1983. Cardiovascular Effects of

Chronic Barium Ingestion. Trace Subst. Environ. Health. 17:155-164.

Perry, H.M., Jr., E.F. Perry, M.W. Erlanger, and S.J. Kopp. 1985. Barium-Induced Hypertension. In: Advances in Modern Environmental Toxicology, Vol. IX, Inorganics in Drinking Water and Cardiovascular Disease. E.J. Calabrese, R.W. Tuthill, and L. Condie, eds. Princeton Scientific Publishing Co., Inc. Princeton, N.J. pp. 221-229.

Perry, H.M., Jr., S.J. Kopp, E.F. Perry, and M.W. Erlanger. 1989. Hypertension and Associated Cardiovascular Abnormalities Induced by Chronic Barium Feeding. J. Toxicol. Environ. Health. 28:373-388.

Roman, R.J., J.R. Carter, W.C. North, and M.L. Kauker. 1977. Renal Tubular Site of Action of Fluoride in Fischer 344 Rats. Anesthesiology. 46:260-264. [As cited in IARC, 1982].

Schroeder, H.A. and J.J. Balassa. 1967. Arsenic, Germanium, Tin and Vanadium in Mice: Effects on Growth, Survival and Tissue Levels. J. Nutr. 92:245-252.

Schroeder, H.A., M. Mitchener, J.J. Balassa, M. Kanisawa, and A.P. Nason. 1968a. Zirconium, Niobium, Antimony and Fluorine in Mice: Effects on Growth, Survival and Tissue Levels. J. Nutr. 95:95-101.

Schroeder, H.A., M. Kanisawa, D.V. Frost, and M. Mitchener. 1968b. Germanium, Tin and Arsenic in Rats: Effects on Growth, Survival, Pathological Lesions and Life Span. J. Nutr. 96:37-45.

Schroeder, H.A., M. Mitchener, and A.P. Nason. 1970. Zirconium, Niobium, Antimony, Vanadium and Lead in Rats: Life Term Studies. J. Nutr. 100:59-68.

Schroeder, H.A. and M. Mitchener. 1975. Life-term Effects of Mercury, Methyl Mercury, and Nine Other Trace Metals on Mice. J. Nutr. 105:452-458.

Smyth, H.F., Jr. and W.L. Thompson. 1945. The Single Dose and Subacute-Toxicity of Antimony Oxide (Sb_2O_3). Melon Institute of Industrial Research, University of Pittsburgh. OTS 206062. [As cited in ATSDR, 1992b].

Sunagawa, S. 1981. Experimental Studies on Antimony Poisoning. Igaku kenkyu. 51:129-142. [As cited in ATSDR, 1992b].

U.S. EPA. 1985. Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge: iron. Office of Water, Regulations and Standards. June.

U.S. EPA. 1988. Recommendations for and Documentation of Biological Values for Use in Risk Assessment. Environmental Criteria and Assessment Office, Office of Health and Environmental Assessment, Office of Research and Development. EPA/600/6-87/008.

February.

U.S. EPA. 1992. Technical Support Document for Land Application of Sewage Sludge. Appendix J. Office of Water, Office of Science and Technology. EPA 822/R-93-001a. November.

U.S. EPA. 1996. Technical Support Document for the Round Two Sewage Sludge Pollutants. Health and Ecological Criteria Division, Office of Science and Technology, Office of Water. EPA-822-R-96-003. August.

APPENDIX D1

LIST OF 31 CANDIDATE POLLUTANTS FOR THE ROUND TWO SEWAGE SLUDGE REGULATION SUBMITTED TO THE DISTRICT COURT IN OREGON

IN THE UNITED STATES DISTRICT COURT
FOR THE DISTRICT OF OREGON

FRANK GEARHART, CITIZENS INTERESTED IN BULL RUN, INC., An Oregon Corporation, KATHY WILLIAMS, AND FRANCES PRICE COOK, Plaintiffs, NATURAL RESOURCES DEFENSE COUNCIL, INC., Intervenor Plaintiffs, ASSOCIATION OF METROPOLITAN SEWERAGE AGENCIES, Intervenor Plaintiffs, v. CAROL M. BROWNER Administrator, United States Environmental Protection Agency, Defendant.	Civil No. 89-6266-HO

NOTICE OF POLLUTANTS

Pursuant to paragraph 2 of the Consent Decree entered in this proceeding on September 5, 1990, as modified by this Court's September 14, 1992 order, the U.S. Environmental Protection Agency ("EPA") hereby gives notice that, based on available information reviewed to date, EPA presently intends to propose for regulation under section 405(d)(2)(B)(i) of the Clean Water Act, 33 U.S.C. § 1345(d)(2)(B)(i), the following pollutants:[1]

Acetic acid (2, 4, -dichlorophenoxy), aluminum, antimony, asbestos, barium, beryllium, boron, butanone (2-), carbon

[1] Based on information available at the time of proposal, EPA retains the discretion to either add or delete pollutants from the list of those that it currently intends to propose for regulation.

disulfide, cresol (p-), cyanides (soluble salts and complexes), dioxins/dibenzofurans (all monochloro to octochloro congeners), endsulfan-II, fluoride, manganese, methylene chloride, nitrate, nitrite, pentachloronitrobenzene, phenol, phthalate (bis-2-ethylhexyl), polychlorinated biphenyls (co-planar), propanone (2-), silver, thallium, tin, titanium, toluene, trichlorophenoxyacetic acid (2, 4, 5-), trichlorophenoxypropionic acid ([2 - (2,4, 5-)], and vanadium.

Respectfully submitted,

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Dated: May 21, 1993

APPENDIX D2

FINAL LIST OF POLLUTANTS FOR THE ROUND TWO SEWAGE SLUDGE REGULATION SUBMITTED TO THE DISTRICT COURT IN OREGON

RECEIVED

95 NOV 29 AM 11:05

CLERK, U.S. DISTRICT COURT
DISTRICT OF OREGON
EUGENE, OREGON

BY_____

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IN THE UNITED STATES DISTRICT COURT
FOR THE DISTRICT OF OREGON

FRANK GEARHART, CITIZENS INTERESTED IN BULL RUN, INC., An Oregon Corporation, KATHY WILLIAMS, AND FRANCES PRICE COOK,

Plaintiffs,

NATURAL RESOURCES DEFENSE COUNCIL, INC.,

Intervenor Plaintiffs,

ASSOCIATION OF METROPOLITAN SEWERAGE AGENCIES,

Intervenor Plaintiffs,

v.

CAROL M. BROWNER
Administrator, United States Environmental Protection Agency,

Defendant.

Civil No. 89-6266-HO

REVISED NOTICE OF POLLUTANTS

On May 24, 1993, pursuant to Paragraph 2 of the Consent Decree entered in this proceeding on September 5, 1990, as subsequently modified by this Court's orders, the U.S. Environmental Protection Agency ("EPA") submitted a Notice of Pollutants ("Notice"). The Notice stated that the Agency was considering proposing 31 pollutants for regulation under section 405(d)(2)(B)(i) of the Clean Water Act, 33 U.S.C. § 1345(d)(2)(B)(i). Paragraph 9d of the Consent Decree provides that the Agency may revise this list of pollutants if it concludes that regulations are not needed for some or all of the 31 pollutants. Based on current information, EPA has concluded that 29 of the listed pollutants need not be regulated: acetic acid (2, 4, -dichlorophenoxy), aluminum, antimony, asbestos, barium, beryllium, boron, butanone (2-), carbon disulfide, cresol (p-), cyanides (soluble salts and complexes), endsulfan-II, fluoride, manganese, methylene chloride, nitrate, nitrite, pentachloronitrobenzene, phenol, phthalate (bis-2-ethylhexyl), propanone (2-), silver, thallium, tin, titanium, toluene, trichlorophenoxyacetic acid (2, 4, 5-), trichlorophenoxypropionic acid ([2 - (2,4, 5-)], and vanadium.

Thus, EPA has concluded that only two of the listed pollutants warrant further consideration for regulation: dioxins/dibenzofurans (all monochloro to octochloro congeners) and polychlorinated biphenyls (co-planar). EPA may, in the exercise of its discretion, determine to add or delete other pollutants from this list at the time of proposal.

Respectfully submitted,

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Dated: November 28, 1995

CERTIFICATE OF SERVICE

I hereby certify that on this November 28, 1995 I caused a copy of the foregoing Revised Notice of Pollutants to be served by first class mail, postage prepaid, on the following counsel:

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Annette Bucco

APPENDIX D3

RESPONSES TO REQUESTS FOR DATA ON THE ROUND TWO CANDIDATE POLLUTANTS

United States Department of Agriculture | Agricultural Research Service | Beltsville Area Beltsville Agricultural Research Center | Beltsville, Maryland 20705

May 10, 1995

SUBJECT: Round 2 contaminants.

TO: Alan B. Hais, Chief, Multimedia Risk Assessment Branch.
Yogi Patel, Multimedia Risk Assessment Branch.

FROM: R.L. Chaney, USDA-ARS, Environmental Chemistry Lab.

I am responding to your letter of April 18, 1995 requesting information on plant uptake of these compounds or metals. I have written about the risks of most of these metals, and some of the organics over the last 10 years. I have huge amounts of literature on these elements, and several you appear to left our of consideration. Where uptake by plants is known to occur to any significant level from sludge-amended soils, these lesser-studied elements have often been examined by pot and field studies of Dr. Don Lisk and his collaborators (including me); they examined the sludges, soils, plants, and animal tissues using neutron activation (and atomic absorption or ICP) to analyze over 40 elements in numerous experiments.

I would hope that demonstrated iron toxicity to cattle and horses from high Fe sludges would put Fe on the list. Similarly, Co is a significant possibility based on food-chain injury to cattle and sheep. Fluoride is also a demonstrated risk from sludges, although mostly in the livestock grazing on surface-applied sludges. I had brought up these omissions in Round 1, so I am a little surprised that Fe and Co were not on the list. Even more surprised when Tl, Sn, and some of the others on your preliminary list were being considered when papers I have given EPA clearly show the lack of risk under any route of exposure to sludges. It would seem to me that your list partially came from the water people, and they base their concern on toxicity of water soluble salts in distilled water, or even on injected water soluble salts (e.g., Ag, Tl, Sn, etc.).

If there is a message to this letter, it is my concern about the need to have iron and cobalt on the thorough evaluation list. Comments below will provide a summary of the literature related to Pathway Analysis of Risk, and useful references.

If you want to reach me regarding these comments on the Round 2 List of Contaminants, I will be at my lab (301-504-8324) May 10 and 11, leaving

for England in mid-afternoon. I will return the evening of May 18, and be in the lab on May 19.

Aluminum: Al is severely phytotoxic to plants when soils remain at pH lower than about 5.2 for a number of years. Clays dissolve and Al^{3+} enters the cation exchange complex in soil. The water soluble Al^{3+} injures root initials, reduces root grown and reduces yield. Toxic Al in subsoils prevents plants from using water stored in subsoils. Al phytotoxicity is a common problem on agricultural and forest land. Addition of inorganic Al salts would allow development of Al phytotoxicity soon after acidification since precipitated $Al(OH)_3$ is present when the soil pH is over 5.2-5.5.

Little Al is absorbed and translocated into plant shoots, and even less into fruits and grains. Most plant Al is soil contamination from wind-blown dust in the field. Soil Al has lower bioavailability than do soluble salts of Al. Other than phytotoxicity, we know of no Pathway in which sludge-borne Al in soils will cause risk compared to unsludged soils. Al should be deleted from the list.

Antimony: In the 1970s and 1980s, Dr. Lisk and his collaborators used neutral activation to measure many elements in plants, sludges, and soils., in pot and field experiments. There were some limitations in these studies. However, the results with antimony were useful to your need. The normal chemical form of antimony (Sb^{3+}) in soils is quite insoluble at normal soil pH levels. Plant leaves, fruits, or grains had unchanged Sb concentration even when soil Sb was significantly increased by applied sludges; and animals did not accumulate Sb from sludge grown crops of Chaney et al. (1978). Sb has little toxicity to animals or plants. It is used some medications. I believe Sb should be deleted from your list.

Barium: In normal soils, which have adequate amounts of Ca and Mg even when sludges are utilized on land, Ba is an exchangeable cation which is pretty insoluble when sulfate is at the levels in soil required to produce high yielding crops. Plant shoots have little response to added sludge Ba, again from the data of Lisk et al. (including the Chaney et al., 1978b) paper on chard fed to Guinea pigs) show no risk of injury or residue transfer to livestock or wildlife. Barium occurs at unusual levels in a few crop species, including Brazil nut, but Lisk and other researchers have not shown significant increase in crop Ba on sludge-amended soils.

Beryllium: Added to soils as a soluble salt, Be has low phytoavailability. Lisk found little evidence that sludge Be moved into plants. And no evidence that Be accumulated in animal tissues when sludge grown crops were

fed to test animals. Be may require full evaluation because of known possible uptake and important industrial toxicology information. However, only Lisk may have measured Be in sludge research studies, and I'm not sure even he did. My comments are based on basic studies in which Be salts were added to soils for plant studies, and the NRC (1980) book on livestock.

Boron: Boron is important in agriculture and the environment because it is phytotoxic. High water soluble B in soils is accumulated by most plants, and they suffer phytotoxicity at foliar B levels which are not high enough to be toxic to livestock chronically fed the crops suffering B toxicity. There is reasonably good evidence that B is required by animals, and that dietary B is generally low. I can perceive no risk except phytotoxicity from sludge B; Lisk et al. provided good evidence of lack of B toxicity or food-chain accumulation of boron.

Only a few studies of sludge or effluent use on cropland or forests has shown B phytotoxicity. In one, a sensitive crop received spray-applied effluent with over 1 mg B/L. In a sludge study, a sensitive crop suffered B phytotoxicity when a sludge containing glass fiber wastes was land applied. Slow dissolution of B from the glass fibers caused excessive B uptake. More B tolerant crops would not have been expected to suffer any effects of biosolids-applied B in that study. I summarized sludge and compost B data in the Chaney and Ryan (1993) paper from the Ohio Composting Conference (see at end of reference section). The appropriate analysis of sludge boron risk will require extraction of "hot water soluble" boron. Based on substantial animal tolerance of B (NRC, 1980), only the phytotoxicity pathway will require risk assessment.

Fluoride: A few sludges contain very high levels of F, resulting from computer chip manufacturing wastewaters (HF is used to leach Si from marked surfaces of the chip), and from aluminum smelting processes. One sludge containing about 5% F was studied by Davis, 1980. He found this sludge could induce F phytotoxicity in ryegrass from soil applied high-fluoride sludge. Generally, foliar exposure of plants to HF causes high accumulation of F in the plants, which in turn poisons livestock. It is widely shown that animals are at much greater fluoride risk from sludge of soil ingestion than from plant uptake.

In the Denver sludge feeding studies (Kienholz et al. and Baxter et al.), CaF (the solid phase F compound in sludges) could be dissolved in the digestive system of cattle, and it could cause bones to become brittle and teeth to break. Analysis of sludges, using some selected

concentration below which no harm is expected to plants or livestock, will provide the protection needed for humans, livestock, and wildlife. Only highly contaminated soils will have phytoavailable F.

Manganese: Few sludges contain high levels of Mn (>1500 ppm DW). In fact, the principle problem regarding sludges is the induction of Mn deficiency when lime-treated sludges are used on coastal plain soils (historically depleted of total soil Mn, so they are more susceptible to lime-induced-Mn deficiency). I reviewed Mn in the Chaney and Ryan (1993a) paper at the Ohio Composting Conference.

We have been testing use of Mn amendments to sludges to prevent induced-Mn deficiency from lime-treated sludges, and have found no evidence of plant toxicity when limed sludge was enriched in Mn by about 6,000 ppm. Al Rubin heard our seminar on May 3 at the Maryland Department of the Environment.

When high Mn soils are strongly acidified (pH $\leq$ 5.4), Mn^{2+} accumulates among the exchangeable cations, and can cause phytotoxicity to sensitive crops. However, except for rare Mn hyperaccumulator species, plants suffer phytotoxicity and leaves remain low in Mn such that they do not comprise chronic toxicity risk to livestock or wildlife. Farmers are forced to add limestone to raise soil pH to prevent Mn phytotoxicity in strongly acidic high Mn soils. I believe that the added risk from sludge-borne Mn is trivial.

Silver: Silver is toxic to animals when injected, but not when ingested with a complete diet; AgCl precipitate is formed in the gut, and Ag is not toxic. When Ag is added to soils, it is strongly precipitated and adsorbed by the soils. Plants accumulate only traces of Ag, and no evidence of plant uptake which might comprise a chronic ingestion risk has been found. Most environmental concern about Ag results from toxicologists testing soluble Ag salts in purified waters. Never from sludge. Even when sludge was fed to livestock, sludge Ag was not toxic nor accumulated. Silver should be deleted from the list.

Thallium: Although Tl appears to comprise a risk to plants or the food-chain from deposition of aerosols on plants, there is little evidence that sludge-applied Tl is moved into edible plant tissues. Again, the studies of Lisk et al. using neutron activation provide adequate evidence that sludge Tl has not been found to comprise risk. Tl can be emitted from incinerators, and cement manufacturers commonly emit Tl and cause local enrichment of soils.

Tin: Sn is normally Sn^{4+} in the soil environment, and very insoluble. Like Ti and Cr, Sn is a good label for non-absorbed soil in the diet. Sludges seldom have really high levels of Sn, and no evidence of plant uptake of Sn from sludge-amended soil has been reported. Lisk included Sn in his studies by neutron activation. Actually, sludge Sn is not a risk to livestock which ingest sludge, in strong contrast with sludge Fe and F. Tin should be deleted from the list.

Titanium: Ti is usually Ti^{4+} in soils, and is very insoluble as TiO_2. Soil Ti is not found inside plants, only as soil or dust contamination on the plants. Soil/Sludge Ti is so insoluble that it does not comprise risk even when ingested by livestock. Titanium should be deleted from the list.

Vanadium: In nutrient solutions, certain unstable V salts can be accumulated by plants, and vanadate interferes with ion uptake by ATPase enzymes in the roots. Little V is translocated to edible crop tissues. The Lisk work usually showed that V was not accumulated by crops, nor in animal tissues. Vanadium should likely be deleted from the list.

Iron: I am a little concerned that no one in your team chose to enter Fe (iron) or cobalt (Co) into the Round 2 review. In 1976-1979, a cooperative study in Maryland allowed us to characterize Fe toxicity to cattle fed high Fe (11%) and low Cu sludges on pastures. When a sludge or compost with only about 4% Fe was surface-applied on pastures or added to feeds in a feeding study with cattle, they did not cause the Fe toxicity, but some accumulation of Fe in the spleen, liver, and duodenum was observed. Several other controlled feeding studies in the US did not find evidence of Fe toxicity from ingested sludges with 1-2% Fe, and seldom found Fe accumulation is tissues. The usual action of excessive Fe intake is to induce chronic Cu deficiency which causes joint disease. Because Fe has poisoned livestock in several sludge experiments, and if high Fe sludges are found by monitoring, the sludge can be required to be injected or incorporated rather than left on the surface, avoidance of sludge Fe risk is comparatively easy. When the ferrous Fe in anaerobic sludges becomes oxidized in the soil, or during composting, the ferric Fe has much lower solubility or toxicity to cattle. So the method of sludge processing and the concentration of Fe in the final product are important in prediction of animal risk. Humans seldom ingest sludges which are freshly anaerobic, and no evidence of human risk from sludge Fe has been identified.

In the Oklahoma miniature horses case, the horses were alleged to have suffered Fe toxicity, but the soil appears to have been the major source

of the Fe exposure. Historic observations of induced Cu deficiency on lateritic or other high Fe soils has been reported in cattle from many locations. Another case in Virginia may have comprised Fe poisoning, but the details of the source of excess soluble Fe remain unclear. One common symptom of Fe toxicity is red coloration of the duodenum from ferritin accumulation. Tissues (liver, kidney, spleen, blood/serum) have increased Fe concentrations when higher Fe sludges are ingested by livestock.

Cobalt: Because sludges normally do not contain high Co concentrations without unusual industrial discharge, no Co problems have been observed in sludge research. However, my analysis of the "Soil-Plant Barrier" indicated that plants could tolerate higher Co concentrations than can be tolerated by ruminant livestock. Apparently vitamin B_{12} is formed in the rumen, and this form of Co causes toxicity in the livestock. Co feeding trials (see NRC, 1980) have shown that 5-10 ppm Co in diets injures sheep and cattle. I have done a substantial risk assessment on Co for a compost to be made from wastewater treatment biosolids at a manufacturing plant of DuPont, and this could be made available to you upon request. Thus, although no adverse effects of sludge-applied Co have been reported to date, it is at least possible to poison ruminants by Co in forage plants. Analysis can identify the very few high Co sludges and require practices to prevent adverse effects.

So, of all the elements you have listed, Fe and F are the only ones with sludge research showing a toxic environmental effect from sludges utilized on land. Please add Fe and Co to the list now. And delete Al, Sb, Ba, Mn, Ag, Sn, Ti, and V.

Organics with substantial vapor pressure (toluene; 2-butanone; methylene chloride; phenol; 2-propanone; ~~toluene~~) are expected to be volatilized or biodegraded during activated sludge treatment of the wastewater, and trace residues will collect in the sludge. Each of these compounds is readily metabolized by soils, with short half-lives. These should be deleted because Round 1 consideration of other volatile compounds showed that no residue reached humans or livestock.

The 2,4-D, 2,4,5-T, and 2-(2,4,5)-TP are residues of pesticides which have lower use today because of their adverse effects in Agent Orange which was contaminated with dioxins produced as byproducts. These compounds are usually sprayed on the plant, and metabolized fairly rapidly by tolerant plants, but slowly by sensitive plants. These reactions are well reported in pesticide applications at EPA. Because these are not very lipophilic, they are usually

biodegraded rather than persistent in soils.

Cyanides can accumulate in sludges after precipitation of ferricyanide by other metals. Soluble cyanide is present only at very low concentrations. Sludges have low levels of total CN, and essentially all sludge/soil cyanide is found to be bound to Fe.

I know little about CS_2. But these is little evidence it would survive aerobic treatment of the water.

Co-planar PCBs, PCNB, Dioxins, Dibenzofurans, and endosulfan are persistent halogenated hydrocarbons. Detailed evaluation will be required for these compounds. But the toxicity endpoints for the halogenated hydrocarbons are seldom reached from these compounds in land-applied sludges. The Madison, WI, studies showed that no significant transfer of sludge-applied PCBs was observed in above ground plant biomass = forages. Direct ingestion of sludges allows digestion of these compounds from the sludge. Accumulation of dioxins in earthworm-food-webs is expected, but not yet shown to induce toxicity to animals.

Nitrate accumulates in fields with aerobic soils after sludge has been incorporated. Some plants accumulate excessive levels of plant nitrate (spinach, beet), and comprise nitrate-poisoning risk to infants. Further, excessive nitrate accumulation in some forage crops can poison livestock. Nitrite seldom accumulates unless some toxic factor inhibits nitrification of the nitrite. Because sludge application rate is limited to the fertilizer requirement of the crop, nitrate and nitrite so not require regulation.

I heard a story about tungsten toxicity in a field study in the UK, but no papers were prepared from the thesis and report to the funding agency. I hope to visit the University of Sheffield and obtain a copy of the thesis on May 12. Dr. Steven McGrath hypothesized that tungstate interfered with use of molybdate in plants by competition as a co-factor for an enzyme involved in N-fixation or nitrate reduction by the plants.

Thus, several elements on the list are of potential importance because of their phytotoxicity rather than food-chain-transfer. These include Al, B and F. Some comprise food-chain risk to livestock which graze the fields (F; and possibly Be, Ba, and Be). Some are not dangerous to livestock even when ingested (Ti, Sn, Sb, and probably Sb). As noted above, Fe and Co also comprise risk until sludge analysis provides the management information needed to prevent risk.

As shown by the Round 1 contaminants, Se and Mo are accumulated by forage crops such that they comprise risk to the livestock rather than to humans consuming garden crops in Pathway 3. Mo needs to be finalized. Several scientists have been conducting studies on Mo uptake by crops on sludge-amended soils. High sludge Fe reduces Mo phytoavailability as noted in my 1991/1992 comments on 503 Mo limits in which I clarified errors in the database on Mo uptake from sludges. The potential role of sludge Fe (and Al) in binding sludge F can sharply reduce fluoride risk assessment for sludges.

Based on widely accepted data about the trace elements on this list, I believe that the following should be deleted from Round 2 now (Al, Sb, Ba, Mn, Ag, Tl, Sn, Tl, and V, and the volatile organics). Others are only a risk in sludge is ingested (Fe, F), and some are sufficiently phytotoxic (based on field studies with sludge) that they might be regulated to avoid phytotoxicity): Al, B, Mn. And Fe should be added to include a well characterized sludge risk from anaerobic treatment conditions. Cobalt is theoretically toxic to ruminant livestock after it is accumulated in forage plants.

Please feel free to call or write me for further information if needed. I enclose several references which cover the Lisk/Furr papers, and have several databases on the sludge-trace element literature in WordPerfect 5.1 which contain references on these rarer elements in sludges.

PLEASE CONFIRM RECEIPT OF THIS MEMORANDUM.

References cited in letter to Hais:

Boyer, K.W., J.W. Jones, D. Linscott, S.K. Wright, W. Stroube and W. Cunningham. 1981. Trace element levels in tissues from cattle fed a sewage sludge-amended diet. J. Toxicol. Environ. Health. 8:281-295.

√REF-VER/Copy [Sewage Sludge—CO: Baxter et al.] "The levels of 20 elements (Al, Ca, Cd, Cl, Co, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, P, Pb, Rb, Sb, Se, V, and Zn are reported for kidney, liver, muscle, spleen, and brain tissues taken from two groups of 6 steers per group in a feeding study conducted at Colorado State University. The control group was fed a normal feedlot cattle ration and the test group was fed the same ration amended with 12% (by weight) air-dried municipal sewage sludge. elemental levels are also reported for the control and test diets, control and test feces, and sewage sludge added to the diet. All samples were analyzed by ICP-plasma emission spectroscopy and neutron activation analysis. Brief descriptions of the analytical methods are included. the levels of all metals determined were elevated in the test diet (as much as 19-fold for Cd) compared with the control diet. The levels of Pb and Cd in kidney and of Pb, Cd, and Cu in liver in the test animals were high enough to cause concern from a toxicological standpoint if these tissues were consumed regularly by humans. None of the levels of any of the other elements in the control and test animals tissues were high enough to cause similar concern with respect to human consumption."

Samples from the 2nd study, with Ft. Collins sludge when it was still high in Cd and Cu. Wet ashed samples. For higher conen metals, ran on ICP directly. For lower conen metals, adjusted to pH near 5 and used chelex resin to collect metals from a larger aliquot, and then acid stripped the

metals into small volume for analysis. Co was by NAA. Co in kidney of (Control/Test) were 0.020/0.041 μg/g FW; liver Co: 0.047/0.077; Muscle Co: 0.07/0.01?; Spleen Co: 0.02/0.02; Brain: 0.009/0.019. Diet contained 0.09/0.43 ppm Co, sludge, 2.5 ppm; and feces:0.43/1.60 ppm DW. ✓RLC•Ω

Capar, S.G., J.T. Tanner, M.H. Friedman and K.W. Boyer. 1978. Multielement analysis of animal feed, animal wastes, and sewage sludge. Environ. Sci. Technol. 12:785-790.

✓REF-VER/Copy [Sewage Sludge—CO: Baxter et al.] "Animal excreta and sewage sludge are currently being used as animal feed ingredients on an experimental basis. The levels of 30 elements are reported for a typical cattle feedlot diet, two dried cattle manures, a commercial cattle waste product, tow dried poultry manures, and a metropolitan sewage sludge. The analyses are conducted using neutron activation analysis, induction coupled plasma spectroscopy, atomic absorption spectroscopy, and anodic stripping voltammetry. The levels of most inorganic elements are considerably higher in animal wastes and sewage sludge than in traditional animal feeds. For most elements the levels determined by several techniques are in good agreement. Problems of loss of lead with precipitate formation, accurate quantitation of elements present in high levels, and obtaining homogeneous samples for analysis are discussed."

Worried about element contamination of sludge and manure if these are used as feed ingredients. thus analyzed many elements using newer techniques (at that time). The feedlot diet contained 0.10 ppm Co, while manures contained 1.1-2.2 ppm Co, and Denver sludge, 7.1 ppm Co. Also analyzed As, Ba, Be, Br, Cd, Cr, Cu, Eu, Hg, La, Mn, Mo, Pb, Rb, Sb, Sc, Se, Sn, Ti, V, Zn, Al, Ca, Cl, Fe, K, Mg, Na, and P. Found considerable contamination of samples with residues of a homogenizer (for Co, Cr, and Ni from stainless steel). Note need for studies of risk and health of animals which consume these contaminated materials. ✓RLC•Ω

Chaney, R.L. and J.A. Ryan. 1993. Heavy metals and toxic organic pollutants in MSW-composts: Research results on phytoavailability, bioavailability, etc. pp. 451-506. *In* H.A.J. Hoitink and H.M. Keener (eds.). Science and Engineering of Composting: Design, Environmental, Microbiological and Utilization Aspects. Ohio State University, Columbus, OH.

Chaney, R.L., G.S. Stoewsand, A.K. Furr, C.A. Bache and D.J. Lisk. 1978b. Elemental content of tissues of guinea pigs fed Swiss chard grown on municipal sewage sludge-amended soil. J. Agr. Food Chem. 26:994-997.

✓ [Sewage Sludge-USDA: Chaney et al.—FEEDING] ✓REF-VER/Copy [(Co in Soil/Plant: Misc. Auth.) Sewage Sludge—USDA: Chaney et al.—Bioavailability] Because we used neutron activation to analyze Co, data are available. "Swiss chard was grown on soil amended with municipal sewage sludges from Baltimore and Washington, DC. The harvested crops were fed at 20 or 28% of diet to guinea pigs for 80 days. Samples of soil, sludges, plant, and animal tissues were analyzed for up to 43 elements. The elements Br, Ca, Co, Eu, Fe, Ni, and Sr were found at higher concentrations in tissues of animals fed the chard cultured on sludge-amended soil than in control animals. Composting sludge prior to amending the soil appeared to render certain elements such as Cd, Cu, Ni, and Zn less available to Swiss chard subsequently grown."

COBALT SUMMARY: Chard was grown on plots of Woodstown silt loam amended with 56 Mg/ha of Baltimore digested sludge, 112 Mg/ha of Blue Plains digested sludge, and 224 Mg/ha of composted digested Blue Plains sludge, and on control. Because the BP compost included some serpentine rock chips, compost and chard were higher in Co than the other sludges: Soil = 9.1 ppm; Balto = 9.4 ppm; BP Dig = 8.0 ppm and BP Compost = 15 ppm DW. The chard (harvested at maturity, washed, rinsed, freeze-dried and ground): Control = 0.4; Balto = 0.8; BP Dig = 2.2; and BP Comp = 1.1 mg Co/kg DW. These results follow the pH of the plots rather than the Co content of the "sludge" or the amended soils. pH at harvest was 6.6, 5.0, 5.7, and 6.7 indicating that compost acted as a liming agent in contrast with sludge. Kidney of one of the 4 replicate

guinea pigs was analyzed for many elements, and all kidney and liver samples were analyzed for Ni, Pb, and Cd. Kidney Co was: Control = 0.6; Balto = 0.7; BP Dig = 1.0; BP Comp = not reported. No significance test was possible on the Co data. Ni was increased in Baltimore chard and kidney/liver of the guinea pigs. Although all sludge grown chard was higher in Cd than the control, no increase was found in kidney or liver! Attribute this to presence of Zn in same tissue. The guinea pigs did equally well on all sources of chard, growing 450 g in the 80 days. ✓RLC•Ω

Davis, R.D. 1980. Uptake of fluoride by ryegrass grown in soil treated with sewage sludge. Environ. Pollut. B1:277-284.

Decker, A.M., R.L. Chaney, J.P. Davidson, T.S. Rumsey, S.B. Mohanty and R.C. Hammond. 1980. Animal performance on pastures topdressed with liquid sewage sludge and sludge compost. pp 37-41. In Proc. Nat. Conf. Municipal and Industrial Sludge Utilization and Disposal. Information Transfer, Inc., Silver Spring, MD.
* RLC•Ω

Francois, L.E. 1986. Effect of excess boron on broccoli, cauliflower, and radish. J. Am. Soc. Hort. Scl. 111:494-498.

Francois, L.E. and R.A. Clark. 1979. Boron tolerance of twenty-five ornamental shrub species. J. Am. Soc. Hort. Sci. 104:319-322.

Furr, A.K., W.C. Kelly, C.A. Bache, W.H. Gutenmann, and D.J. Lisk. 1976. Multi-element absorption by crops grown on Ithaca sludge-amended soil. Bull. Environ. Contam. Toxicol. 16:756-763.
✓ RLC•Ω

Furr, A.K., T.F. Parkinson, D.C. Elfving et al. 1981. Element content of vegetable and apple trees grown on Syracuse sludge-amended soils. J. Agric. Food Chem. 29:156-160.
✓ RLC•Ω

Hogue, D.E., J.J. Parrish, R.H. Foote, J.R. Stouffer, J.L. Anderson, G.S. Stoewsand, J.N. Telford, C.A. Bache, W.H. Gutenmann and D.J. Lisk. 1984. Toxicologic studies with male sheep grazing on municipal sludge-amended soil. J. Toxicol. Environ. Health 14:153-161.

✓REF-VER/Copy [Heavy Metals in Soil/Plants: Lisk et al.—SLUDGE] "Growing sheep were grazed for 152 days on grass-legume forage growing on soil that had been amended with municipal sewage sludge from Syracuse, NY, at 224 metric tons/ha. Cd was higher, but not significantly ($P > 0.05$), in tissues of sheep fed the sludge-grown forage as compared to controls. No significant differences between the sludge or control treatments were found in weight of the complete or cauda epididymis or in % progressive motility of cauda epididymal sperm. The sludge-treatment group had significantly larger testes ($P<0.025$) when expressed as a percentage of body weight, and higher blood uric acid values ($P<0.05$). There were no observable changes in tissue ultrastructure of liver, kidney, muscle, or testes as examined by electron microscopy in either of the treatment groups. There were no significant differences for rate of animal weight gain, carcass weight, dressing percentage, or quality or yield grade of the carcass between the treatment groups."

Syracuse sludge. April 1980, applied weathered (1 yr) sludge to subsoil of Chenango gravelly loam, pH 7.1. Amended soil was pH 6.7. Collected grass-legume hay for feeding studies in 1980 and 1981. In 1982, used for grazing study. Had been planted with alfalfa, birdsfoot trefoil,

timothy, and bromegrass. 3 month old 'Morlam' sheep used to graze the pastures for 152 days. Each animal was also fed 250 g feed concentrate daily, and ad lib water. Composite soil from field Sludge contained 83 ppm Cd; forage 0.09 vs. 1.14 ppmDW Cd. Feed concentrate contained 0.21 ppm Cd. Kidney contained 0.55±0.14 ppmDW Cd vs. 0.83±0.17 NS; liver contained 0.22±0.04 vs. 0.40±0.08 NS; muscle contained 0.03±0.01 vs. 0.09 ±0.04 ppmDW Cd NS. Rate of gain was higher for sludge than control animals NS. ✓RLC•Ω

Kienholz, E.W., G.M. Ward, D.E. Johnson, J. Baxter, G. Braude and G. Stern. 1979. Metropolitan Denver sewage sludge fed to feedlot steers. J. Anim. Sci. 48:735-741.

✓REF-VER/Copy [Sewage Sludge—CO: Baxter, Kienholz, et al.] "Feedlot steers received 0, 4, or 12% Metropolitan Denver sewage sludge on a dry weight intake basis for a 94-day finishing period. The sludge was anaerobically digested primary sludge that had been treated with polyelectrolyte to aid in dewatering during vacuum filtering. It was then dried to 35% water prior to mixing into the pelleted diet given the steers. Cattle (6 on each treatment) were slaughtered and kidney, liver, muscle, bone, brain, blood, lung, spleen, and fat were analyzed for As, Cd, Cu, Hg, Mo, Ni, Pb, Se, and Zn.

"Growth of the sludge animals was less than controls ($P < 0.025$) because sludge, apparently, provided no energy. Sludge ingestion caused no pathology. All 10 inorganic elements except Ni were increased in one or more body tissues following the 94 day sludge ingestion. Percentage whole carcass retentions of ingested minerals were estimated as follows: 0.2% As, 0.04% Cd, 0.3% Cu, 0.07% Hg, 0.2% Mo, <0.006% Ni, 0.6% Pb, 1.3% Se, 0.2% Zn, and 32% F. Steers retained low amounts of the toxic heavy metals from sludge ingestion."

Sludge contained (ppmDW): 1.3 As, 21 Cd(diets 0.025, 0.65, and 1.9 ppm), 710 Cu(diets 3.2, 31, and 86 ppm), 11 Hg, 40 Mo, 125 Ni, 780 Pb(diets 0.6, 26, and 77 ppm), 5.4 Se, 1500 Zn, and 200 F. Diet was pelletted corn+cottonseed mean+molasses+limestone+NaCl. corn silage ad lib. Bone samples were taken from the proximal half of the tarsal bone. Samples digested with low metal acids. For many elements (not kidney or liver), sample metals were extracted by APDC, crystals collected, and filtered. Taken into small volume for analysis. Carbon rod used for some samples. Good QA/QC program. At 12% sludge, As was increased in liver, Cd in liver and kidney, Cu increased in liver, Hg increased in liver, kidney and muscle, Mo increased in bone and liver, Pb increased in liver, kidney, bone, and blood; Se increased in blood; Zn increased only in liver. At both rates, F increased in bone. Ni did not increase in any tissues.

Pb in tissues: Liver 0.2a 3.3b, 4.6c ppmDW for 0/4/12% sludge; kidney: 0.9a 12.2 b 15.8 b; Muscle: 0.2 . 0.2; bone:1a, 4b, 11c; blood: 0.12a, ., 0.82b; fat:0.16, ., 0.16. Cd in tissues: liver: 0.2a 0.5b 0.4b; kidney: 1.1a 2.5b 2.4b; muscle: <0.01, ., <0.01. Hg: Liver:0.01a 0.06b 0.14c; kidney 0.1a, 0.45b, 0.9c. Cu: liver: 124a, 260b 240b. ✓RLC•Ω

Neary, D.G., G. Schneider, and D.P. White. 1975. Boron toxicity in red pine following municipal wastewater irrigation. Soil Sci. Soc. am. Proc. 39:981-982.

NRC (National Research Council). 1980. Mineral Tolerance of Domestic Animals. National Academy of Sciences, Washington, D.C. 577pp.

• RLC•Ω

Rea, R.E. 1979. A rapid method for the determination of fluoride in sewage sludges. Water Pollut. Contr. 78:139-142.

Sanson, D.W., D.M. Hallford and G.S. Smith. 1984. Effects of long-term consumption of sewage solids on blood, milk and tissue elemental composition of breeding ewes. J. Anim. Sci. 59:416-424.

✓REF-VER/Copy [Sewage Sludge-NM: Smith et al.] "Fine-wool ewes received for 2 yr a complete pelleted diet (11% protein) or the basal diet fortified with 3.5% cottonseed meal (CSM,

12% protein) or gamma-irradiated (1 megarad) dried solids (SS, 12% protein) from primary (undigested) sewage (Las Cruces, NM municipal sewage). Five ewes fed each diet were sampled to determine Ag, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Pb, and Zn in blood, milk and tissues. Tissues and blood were sampled at slaughter 40 days after weaning of lambs. Mean whole blood mineral concentrations were similar ($P > 0.05$) among treatments 3 d postpartum; however, at 42 days after lambing both basal and sewage fed ewes had elevated blood Ca compared with ewes fed CSM. No biologically important differences were detected in the concentrations of elements in milk. Ewes fed SS had lower ($P<0.05$) flood Fe than animals in the other groups. Sewage-fed ewes also had higher ($P<0.05$) liver Fe (1092 ppmDw) than basal-fed ewes (626 ppm) whereas Fe in CSM-fed ewes (873 ppm) was similar to both. Basal-fed animals had 1.1-1.3 times more ($P<0.05$) liver Mg and 2-to-3-fold higher liver Na than CSM or SS. Livers from SS-fed ewes had higher concentrations ($P<0.05$) of Cd (1.5-1.5-fold) and Pb (1.4-1.9-fold) than livers from CSM or basal-fed ewes. Cd was 1.2-1.5-times higher ($P<0.05$) in kidneys from ewes receiving the sewage product, but all values were less than 1 $\mu g/g$. Lead was not detectably increased in kidney samples. Spleen Ca, Fe, and Ni and muscle Ca were increased due to feeding SS; however, no elements in bone samples from SS-fed animals were elevated above the control groups. Consumption of a diet containing 7% SS throughout a 2-yr period had only minor effects on element contents of blood, milk, or tissues."

Sludge contained: 20 Ag, 3.2 Cd, 361 Cr, 405 Cu, 5280 Fe, 99 Mn, 11 Ni, 150 Pb, and 606 ppmDW Zn. Blood Cu or Zn not really changed by SS ingestion. Milk Cu dropped with SS at the later sampling time. Liver Fe was (basal/CSM/SS) 626/873/1092; Co, 0.9/0.9/0.9; Cu 32/30/37;Zn 39/41/35; Cd 0.5b/0.6b/0.8c; Pb 2.1b/2.9b/4.0c. Kidney: Cd 2.4b/3.1b/3.7c; Pb 3.6b/1.9c/3.7b. Bone Pb: 21/15/18 NS. ✓RLC•Ω

Sanson, D.W., D.M. Hallford and G.S. Smith. 1984. Effects of dietary sewage solids on feedlot performance, carcass characteristics, serum constituents, and tissue elements of growing lambs. J. Anim. Sci. 59:425-431.

✓REV-VER/Copy [Sewage Sludge-NM: Smith et al.] "20 fine-wool wether lambs were allotted randomly at weaning to be fed a conventional diet (CD) formulated for growing lambs, or the same basal mixture plus dried gamma-irradiated solids from primary sewage sludge (SS) incorporated as 7% of the dry matter. Feedlot performance was chemical elements in whole blood were monitored during 90 days, after which lambs were slaughtered and carcass data were collected. Concn. of Ag, Ca, Cd, Co, Cr, Cu, fe, K, Mg, Mn, Na, Ni, P, Pb, and Zn were measured in livers, kidneys, bones, spleens and muscles. SS in the diet did not affect performance appreciably, although dressing percentages from lambs fed SS were 4% lower than Cd-fed animals ($P<0.05$). rib eye areas, fat thickness quality and yield grades were similar ($P>0.05$). element concentrations in whole blood at weaning, after 56 days of the feeding trial and at slaughter did not differ ($P>0.05$) between dietary groups. Serum chemistry determinations showed no biologically meaningful patterns related to diets. Lambs fed SS had higher ($P<0.05$) concn. of Cu in livers (51.1 vs. 34.3 $\mu g/g$) and Pb in kidneys (4.0 vs. 2.2 ± 0.3 $\mu g/g$ and lower Mg in kidneys. None of the elements in spleen and muscle tissue differed ($P>0.05$) between diet groups. Lambs fed SS had elevated ($P<0.05$) bone Co, Cu, Fe, K, and Na compared with those of CD. Lead concn. in bone were increased ($P<0.05$) by Ss over CD (30.5 vs. 26.3), but Cd and Zn did not differ. A feedlot diet with 7% SS did not appear to adversely affect growth or carcass characteristics of lambs. Serum clinical profiles and chemical elements in blood and tissues were affected negligibly by SS as 7% of the diet."

Sludge composition averaged: 3 Cd, 470 Cu, 9233 Fe, 110 Mn, 9 diet consumption, and lower gain rates. Blood Cu not affected by sludge ingestion. Liver contained: Cd <0.07/0.07 ppmDW; Cu 34.3/51.1; Fe 179/190; Pb 2.5/3.5; Kidney: Cd <0.07/<0.07; Pb 2.2a/4.0b; bone: Pb 26.3a/30.5b. ✓RLC•Ω

Smith, G.S., D.M. Hallford and J.B. Watkins, III. 1985. Toxicological effects of gamma-irradiated sewage solids fed as seven percent of diet to sheep for four years. J. Anim. Sci. 61:931-941.

√REF-VER/Copy [Sewage Sludge-NM: Smith et al.] "Breeding ewes in drylot were fed pelleted complete diets with 3% cottonseed meal (CSM) or 7% dried, gamma-irradiated sewage solids (DGSS) for 4 yr. Cytochrome P-450 content and enzyme activities for xenobiotics biotransformations were assayed in livers after 3 yr and in livers, kidneys and ileal tissue after 4 yr. Dietary DGSS caused no increase in P-450 and few changes in activities of oxidative, hydrolative, and conjugative biotransformational enzymes. Consumption of DGSS for 4 yr caused slight enlargement of spleens (1.1-fold) and ovaries (1.3-fold, $P<0.10$), but no change in size of livers, kidneys, hearts, adrenals and thyroids ($P>0.10$), nor liver vitamin A levels ($P>0.10$). Of 22 refractory lipophilic residues assayed in abdominal adipose tissue, few were detected and of those detected DGSS caused none to exceed normal levels. Dietary DGSS increased ($P<0.01$) Fe in livers 1.5-fold and in spleens 5.6-fold, and increased Cu in livers 1.3-fold ($P<0.01$) and in kidneys 1.2-fold. Dietary DGSS increased Cd levels in livers but not in kidneys or spleens ($P>0.10$); yet all Cd levels were within ranges for livestock fed conventional feed. Dietary DGSS caused no increase ($P>0.10$) in levels of Ag, Ca, Cr, Hg, K, Mg, Mn, Na, Ni, P, Pb, or Zn in livers, kidneys or spleens. There were no histopathological lesions of toxicosis except mild hemosideresis of spleens. Consumption of a diet with 7% DGSS throughout 4 yr caused no hazardous accumulation of toxic elements and little, if any, evidence of toxicity."

Undigested sewage solids (primary and activated) from Las Cruces, NM. Dried and irradiated. Contained: 0.58% Fe; 606 ppm Zn; 405 ppm Cu; 361 ppm Cr; 150 ppm Pb; 99 ppm Pb; 11 ppm Ni; <5 ppm Hg; <1 ppm Se. 41.5% ash. Liver Fe was increased, 849±387 (SD) vs. 1303±291 ppm DW. Liver Cu was raised: 597±308 vs. 761±259 ppmDW. Liver Cd [<0.03 vs. 1.47±0.30 ppmDW] was raised, but kidney was not [2.8±0.8 vs. 3.5±0.6 ppmDW]. Pb was unchanged and at very low levels in liver, kidney, and spleen (<0.10 ppm DW). p,p'DDE was increased in fat, but PCB and other chlorinated hydrocarbons were not increased. The animals were mature, fine-wool ewes of Rambouillet breeding. ✓RLC•Ω

Vimmerstedt, J.P. and T.N. Glover. 1984. Boron toxicity to sycamore on minesoil mixed with sewage sludge containing glass fibers. Soil Sci. Soc. Am. J. 48:389-393.

TABLE 5. Maximum tolerable levels of dietary minerals for domestic livestock in comparison with levels in forages.

Element	Soil-Plant Barrier[a]	Level in Plant Foliage[a]	Maximum Levels Chronically Tolerated[b]			
			Normal	Phytotoxic	Cattle	Sheep	Swine	Chicken
		--mg/kg dry foliage--	----------mg/kg dry diet--------					
As, inorg.	yes	0.01-1	3-10	50.	50.	50.	50.
B	yes	7-75	75	150.	(150.)	(150.)	(150.)
Cd[c]	Fails	0.1-1	5-700	0.5	0.5	0.5	0.5
Cr^{3+}	yes	0.1-1	20	(3000.)	(3000.)	(3000.)	3000.
Co	Fail?	0.01-0.3	25-100	10.	10.	10.	10.
Cu	yes	3-20	25-40	100.	25.	250.	300.
F	yes?	1-5	-	40.	60.	150.	200.
Fe	yes	30-300	-	1000.	500.	3000.	1000.
Mn	?	15-150	400-2000	1000.	1000.	400.	2000.
Mo	Fails	0.1-3.0	100	10.	10.	20.	100.
Ni	yes	0.1-5	50-100	50.	(50.)	(100.)	(300.)
Pb[c]	yes	2-5	-	30.	30.	30.	30.
Se	Fails	0.1-2	100	(2.)	(2.)	2.	2.
V	yes?	0.1-1	10	50.	50.	(10.)	10.
Zn	yes	15-150	500-1500	500.	300.	1000.	1000.

a/ Based on literature summarized in Chaney et al. (1982).

b/ Based on NRC (1980). Continuous long-term feeding of minerals at the maximum tolerable levels may cause adverse effects. Levels in parentheses were estimated (by NRC) by extrapolating between animal species.

c/ Maximum levels tolerated were based on Cd or Pb in liver, kidney, and bone in foods for humans rather than simple tolerance by the animals.

From: Chaney and Ryan, 1993.

Boron Phytotoxicity: In contrast with municipal sewage sludge, MSW-compost contains substantial levels of soluble boron (B). B toxicity from sewage sludge application was reported only for an unusual case of a sensitive tree species growing in soils amended with a sludge containing lots of glass fibers (Vimmerstedt and Glover, 1984; see also Neary et al., 1975, regarding high B levels in phosphate-free detergents). The glass fibers contained borosilicate and release of B caused phytotoxicity. Research has shown that much of the soluble B in MSW-compost comes from glues (Volk, 1976). It has long been known that plant samples placed in paper bags can become contaminated from B from glue used to

hold the bag together. El Bassam and Thorman (1979) and Gray and Biddlestone (1980) noted that the B level in MSW-composts was quite variable as might be expected if composts are not well mixed.

In general, B phytotoxicity has occurred when high application rates were used, and B-sensitive crops were grown. However, when MSW-compost is used at fertilizer rates in normal fields, the B might be important as a fertilizer rather than as a potential phytotoxicity problem.

Boric acid and most borates are quite water soluble, although B can be adsorbed on clays and by organic matter. Low soil pH facilitates B uptake by plants because the H_3BO_3 molecule (predominant form at lower soil pH) is absorbed by roots rather than anionic borates (Oertli and Grgurevic, 1975). Although most B toxicity has been reported on alkaline soils, this is due to the lack of leaching for most of these soils. Excess applications of soluble B are much more phytotoxic in acidic soils, and liming can correct B phytotoxicity. The usual liming action of compost should help prevent this problem.

There are large differences among crop species in tolerance of excessive soil B. Some crops are very sensitive, and these are the species which have suffered phytotoxicity from compost-applied B (bean, wheat, and mum). Francois has summarized the significant differences among several groups of crops (Francois and Clark, 1979; Gupta, 1979; Francois, 1986). Ornamental horticultural species have been examined to some extent (information on individual species can be found by literature searching), but many horticultural crops have not been studied. This is one research need related to practical microelement phytotoxicity from compost.

Perhaps the first report on B toxicity from MSW-compost is that of Purves (1972) who noted B phytotoxicity to beans on field plots which received high rates of MSW-compost. The full description of the compost experiment is reported in Purves and Mackenzie (1973), and a careful examination to prove B phytotoxicity was reported by Purves and Mackenzie (1974). Bean (but not potato or other species examined) suffered severe yield reduction at high compost rates; this yield reduction was proportional to rate of compost application. Bean is known to be especially sensitive to B phytotoxicity. Gray and Biddlestone (1980) also found B phytotoxicity in sensitive species grown in field plots with high rates of MSW-compost.

Gogue and Sanderson (1975) reported B phytotoxicity to chrysanthemums in potting media containing MSW-compost. Foliar analysis clearly supported the conclusion that B was toxic and that Mn, Cu, Zn, and other elements were not at toxic levels. They conducted a calibration experiment to determine the sensitivity of chrysanthemums (Gogue and Sanderson, 1973), and the levels found in the mums grown on the test media were in the phytotoxic range. In their research, they adjusted the pH of the media to 6 using sulfur, rather than allowing the MSW-compost to raise the pH of the media. This probably contributed to the severity of B phytotoxicity observed. Some other horticultural species also suffered B phytotoxicity in compost-containing media (Gilliam and Watson, 1981). Sanderson (1980) reviewed B toxicity in compost amended potting media. In contrast to MSW-compost, sewage sludge composts with wood chips have not been found to cause B phytotoxicity (Chaney, Munns, and Cathey, 1980). Only a few acid-loving

species require acidification of media to do well on neutral compost-amended media.

Interestingly, because the B which causes phytotoxicity is water soluble, the B phytotoxicity problem from MSW-compost is short-lived. Purves and Mackenzie (1973) noted that pre-leaching MSW-compost prevented B phytotoxicity. Other studies noted that the B-phytotoxicity occurred only during the year of application, and that soluble B was leached out of the root zone over winter (Volk, 1976) or by leaching potting media with normal horticultural watering practices. Sanderson (1980) noted that perlite also adds B to potting media, and that use of both may cause B toxicity when either perlite or MSW-compost alone might not have done so. Lumis and Johnson (1982) studied leaching of B in relation to toxicity of salts and B to *Forsythia* and *Thuja*. They reported that a simple leaching treatment removed excess soluble salts, but was unable to remove enough B to prevent phytotoxicity (the compost they studied contained 225 mg B/kg, higher than most reports). Nogales et al. (1987) also found compost-applied B leached quickly such that crop B was reduced in each successive ryegrass crop.

B phytotoxicity is significantly more severe when plants are N-deficient (Gogue and Sanderson, 1973; Nogales et al., 1987; Gupta et al., 1973). This makes the B in MSW-compost which is not properly cured (to avoid N immobilization) potentially more phytotoxic than in well cured composts. Further, B flows with the transpiration stream and accumulates in older leaves. In environments with low humidity, more transpiration occurs (e.g., greenhouses), and B toxicity is more severe. B and salt toxicity are easily confused; both are first observed in leaf tips or margins of older leaves. Diagnosis of B phytotoxicity requires a knowledge of relative plant tolerance of B, or analysis of the leaves bearing symptoms.

Thus, in general use, compost application at a reasonable fertilizer rate would simply add enough B to serve as a fertilizer for B-deficiency susceptible crops such as alfalfa or cole crops. However, use of MSW-compost at high rates in soils or potting media could cause phytotoxicity if high soluble B were present. The B phytotoxicity would not be persistent because soluble B would leach from the root zone with normal rainfall or irrigation. Compost-applied B would be more phytotoxic in N-deficient soils, which might result from application of improperly cured compost. Water soluble B should be one chemical which is regularly monitored in MSW-composts so that the need for warning about rates of application and use with sensitive crops can be identified. Deliberate use of MSW-compost as a B fertilizer for high B-requiring crops such as the cole crops (cabbage family) might become a regular agronomic practice. Sources of soluble B in modern MSW-compost should be evaluated, and alternative to B use identified.

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Plant Uptake of Pentachlorophenol from Sludge-Amended Soils

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ABSTRACT

A greenhouse study was conducted to determine the effects of sludge on plant uptake of ^{14}C-pentachlorophenol (PCP). Plants included tall fescue (*Festuca arundinacea* Schreb.), lettuce (*Latuca sativa* L.), carrot (*Daucus carota* L.), and chile pepper (*Capsicum annum* L.). Minimal intact PCP was detected in the fescue and lettuce by gas chromatography/mass spectrometry (GC/MS) analysis. No intact PCP was detected in the carrot tissue extracts. Chile pepper was not analyzed for intact PCP because methylene chloride extracts contained minimal ^{14}C. The GC/MS analysis of soil extracts at harvest suggests a half-life of PCP of about 10 d independent of sludge rate or PCP loading rate. Rapid degradation of PCP in the soil apparently limited PCP availability to the plant. Bioconcentration factors (dry plant wt./initial soil PCP concentration) based on intact PCP were <0.01 for all crops, suggesting little PCP uptake. Thus, food-chain crop PCP uptake in these alkaline soils should not limit land application of sludge.

LAND APPLICATION of sewage sludge is an inexpensive and convenient method of sludge disposal that provides plant nutrients and improves soil structure. The potential bioavailability of toxic organics that can contaminate some sludges (e.g., polychlorinated biphenyls) may limit sludge utilization in agricultural settings, however. The large number of toxic organics that can contaminate sludges, complex analysis, and unlimited environmental conditions makes it virtually impossible to study each compound in every setting. Therefore, a compound is selected based on physical and chemical properties to represent a group of compounds.

Chlorophenols are a group of ionizable organic compounds of environmental concern. Chlorophenols are not generally detected in sludges nationwide. However, 156 of 223 industrial and municipal sewage sludges from Michigan contained pentachlorophenol (PCP) at concentrations ranging from 0.2 to 8 490 mg kg^{-1} dry weight (Jacobs et al., 1987). The median concentration of the PCP contaminated sludges was 5 mg kg^{-1}.

The widespread use of PCP as a wood preservative and general biocide has lead to contamination of air, food, sediment (Bevenue and Beckman, 1967), water, and municipal sewage sludge (Buhler et al., 1973). Pentachlorophenol is categorized as very toxic [oral LD_{50} = 146 mg kg^{-1}, rat (*Rattus norvegicus*); Crosby, 1980] and is mutagenic in MP-1 yeast strain (Fahrig et al., 1978). The generally low PCP concentration in contaminated sludges (5 mg kg^{-1}) would be reduced 50 to 100 times during land application of sludge at normal agronomic rates (22.5–45 Mg ha^{-1}). Accumulation of PCP in food-chain crops would not likely adversely affect humans at the resulting PCP soil-sludge concentrations.

Bioavailability of nonpolar compounds such as PCP depends on the extent of sorption by the soil organic fraction and other transformation processes (Hamaker and Thompson, 1972). Sludge additions increase the soil organic fraction and thus possibly alter PCP bioavailability. The chemical characteristics also contribute to the complexity of PCP sorption in the soil-sludge system. Pentachlorophenol is a weak acid (pK_a = 4.74; USEPA, 1979) and sorption increases with decreasing soil pH (Lagas, 1988a; Banerji et al., 1986) possibly altering the bioavailability of PCP in soils.

Uptake of ^{14}C by crops grown in ^{14}C-PCP treated soils has been reported; however, actual PCP and metabolites were not verified (Kloskowski et al., 1981; Topp et al., 1986). Topp et al. (1986) reported significant ^{14}C uptake for barley (*Hordeum vulgare* L.) after 1 wk in a soil with a pH of 6.4. However, ^{14}C in the plant was attributed to rapid degradation and mineralization of PCP followed by assimilation of $^{14}CO_2$. On the contrary, Casterline et al. (1985) reported intact PCP uptake by spinach plants (*Spinacia oleracea* L.) and soybean plants [*Glycine max* (L.) Merr.] from an acid soil. Methoxytetrachlorophenol, 2,3,4,6-tetrachlorophenol, pentachloroanisol, and 2,3,4,6-tetrachloroanisol were the primary metabolites detected in the plant tissue. The purpose of this experiment was to determine the effects of sludge on PCP uptake by plants grown in alkaline soils in the greenhouse.

MATERIALS AND METHODS

The soils used in this study were a Glendale clay loam [fine-silty, mixed (calcareous), thermic Typic Torrifluvent] and a Bluepoint sandy loam (mixed, thermic Typic Torripsamment), with pH values (water paste) of 7.8 and 8.3, respectively. The soils were air-dried and sieved (6.25 mm) prior to the experiment.

A municipal sewage sludge obtained from Albuquerque, NM, was anaerobically digested, air-dried, and gamma irradiated (^{137}Cs 10 kGy) to reduce pathogens. The sludge was ground to <2 mm and amended to the soils at rates of 0, 22.5, and 45 Mg ha^{-1} (0, 10, and 20 g kg^{-1}). The sludge was PCP-free, <1 mg kg^{-1} as analyzed by gas chromatography with a flame ionization detector (USEPA method 625).

Reagent-grade PCP (Chemical Dynamics Corp., South Plainfield, NJ) and ^{14}C-PCP (Sigma Chemical Co., St. Louis, MO) (specific activity 455.1 MBq $mmol^{-1}$, uniformly ring-labeled) were used to obtain desired PCP rates (0, 0.1, 0.6, 1.1, and 5.1 mg kg^{-1}).

Fescue (*Festuca arundinacea* Schreb., 'Ky 31') and three foodchain crops: lettuce (*Latuca sativa* L., 'Great Lakes'), carrot (*Daucus carota* L., 'Nantes Scarlet'), and chile pepper (*Capsicum annum* L., 'Espanola Improved') were grown in a greenhouse. The foodchain crops were chosen to represent a leafy crop, a root crop, and a fruit crop, respectively. Treatments were duplicated.

Greenhouse Procedure

Soil, sludge, and fertilizer (phosphate fertilizer 920 kg P_2O_5 ha^{-1}) totaling 4 kg were mixed in a twin shell blender (18 kg

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capacity) for 5 min. This soil-sludge mixture was placed in two pots (2 kg pot^{-1}) and leached to remove excess salts. After drying, soil-sludge mixtures were spiked with ^{14}C-PCP solutions (20 mL of 0.1 *M* NaCl containing 740 MBq ^{14}C). The soil was mixed in a twin shell blender for 3 min and weighed into 18-cm diam. pots (2 kg pot^{-1}). Drainage holes in the pots were covered with fiberglass mesh. A 100-g sample of the soil-sludge mixture was retained for verification of ^{14}C-PCP application rate.

Seeds were placed on the soil surfaces and covered with 1 cm of soil previously removed from the pot. The pots were then watered by a drip irrigation system to 80% pot water holding capacity and maintained at this moisture content gravimetrically. A total of 16 h daylight was maintained (additional light from eight 400-W sodium lamps). Temperatures ranged from 15 to 40 °C. Liquid calcium nitrate fertilizer (equivalent to 360 kg N ha^{-1}) was added to the Bluepoint soil to compensate for N available in the sludge treatments. The experimental design was a randomized complete block for the lettuce, carrot, and chile and a completely random design for the fescue.

Plant and Soil Sampling

Fescue was grown in the Bluepoint soil and the Glendale soil for 34 and 42 d, respectively. Lettuce was grown for 58 d, carrot for 79 d, and chile pepper for 115 d.

Plants were cut 2 cm above the soil surface to avoid contamination from the soil. Chile fruits were cut from the foliage and carrot roots were removed from the soil. The carrot roots were washed in an ultrasonic bath with deionized water for 5 min.

Fresh weights of each fraction (combustion and extraction fractions) were recorded for all plant parts. The fraction for combustion was oven-dried at 50 °C, reweighed, and ground in a Wiley mill with a 20-mesh screen. Extraction samples were stored in Ziploc bags in a freezer at −10 °C.

Soil samples, consisting of four 2-cm diam. cores to the depth of the pot (100 g), were taken from each pot at each harvest. The soil samples were divided into two fractions: one for combustion and one for extraction. The samples for combustion were air-dried and ground with a mortar and pestle. The extraction samples were stored in Ziploc bags in a refrigerator at 5 °C.

Analytical Methods

Combustion. Ground, air-dried, 300-mg soil samples and ground, oven-dried, 50-mg plant samples were replaced in boats, covered with activated alumina/cupric oxide (5:1 w/w), and combusted at 1000 °C (Lindberg furnace) for 8 min under a stream of O_2 (150 mL min^{-1}). Evolved $^{14}CO_2$ was trapped in a solution of 8 mL ethoxyethanolethanolamine (3:2 v/v) and 10 mL Ready Gel cocktail (Beckman). Samples were counted by liquid scintillation (LS) with a Beckman LS1800 counter. Blanks and standards were combusted and counted to determine background counts (40 disintegrations per minute, dpm), oxidation efficiency (about 0.95), and counting efficiency (0.65). The mean radioactivity was 0.184 ± 0.018 MBq kg^{-1} soil-sludge mixture in all PCP-containing treatments.

Extraction Procedure. Soil and plant samples containing the highest ^{14}C concentrations were extracted and analyzed by LS counting and gas chromatography with mass spectrometer detector (GC/MS). Samples were extracted with a procedure modified from Casterline et al. (1985). The original procedure extracted PCP, lower chlorinated phenols, and anisols individually from plant and soil samples (93.2% total plant extraction efficiency). This procedure was simplified because individual PCP metabolite identification was not attempted.

Fine roots were removed from soil samples and 10 g of soil and 10 mL concentrated HCl were combined in glass centrifuge bottles, and heated in an oven at 60 °C overnight. After cooling, the acidified soils were extracted with 50 mL methylene chloride. The bottles were stoppered, shaken for 2 min, and centrifuged for 5 min at 2000 rpm (750 × *g*). The methylene chloride was removed and collected in a beaker. The soil mixture was extracted three more times for a total methylene chloride volume of 125 mL.

Frozen plant samples were homogenized in a Waring blender with 100 mL, 0.2 *M* HCl for 5 min. The mixture was transferred into a 200-mL glass centrifuge bottle, 100 mL methylene chloride added, and the bottle stoppered. After shaking for 2 min, the mixture was centrifuged for 4 min at 1000 rpm (300 × *g*). The methylene chloride was removed and collected into a beaker. The plant and acid mixtures were extracted three more times with 50, 25, and 25 mL methylene chloride. All extracts were combined.

Extract Cleanup. Soil and plant methylene chloride extracts were evaporated to 25 mL. A 2-mL fraction of the soil extract was evaporated to dryness in a LS vial, dissolved with scintillation cocktail, and counted by LS. A 2-mL fraction of the plant extract was evaporated to dryness in a LS vial. Bleach (0.75 mL of 5.25% sodium hypochlorite) was added and the suspension heated at 50 °C for 1 hr. A mixture of acetic acid and Ready Gel cocktail (18 mL of 3:400 v/v) was then added to the bleached plant samples, and the resulting mixture counted by LS.

The remaining 23-mL fractions of methylene chloride were placed into glass centrifuge tubes with 10 mL NaOH (pH 9), shaken for 2 min, and centrifuged for 4 min at 1000 rpm (300×*g*). The methylene chloride was discarded. The NaOH was acidified with 2 *M* HCl to pH≤2 and extracted with 2 mL methylene chloride. The acid fractions were extracted two more times with 2 mL methylene chloride. All extracts were combined in one vial and evaporated to dryness with N_2 gas.

Extract residues were derivatized for GC/MS (MS Model HP5970, connected to a GC Model HP5890) analysis by heating at 70 °C for 15 min with 20 μL *N,O*-bis(trimethylsilyl)-trifluoroacetamide (BSTFA) (Poole, 1978). Aliquots were injected on to a 0.32 mm i.d. by 25 m Ultra 2 column (Hewlett Packard, Boulder, CO) with a 0.52 μm film (Cross-linked 5% Phenol Methyl Silicone) at 150 °C. The oven temperature was ramped at 25 °C min^{-1} to 280 °C, with a final hold time of 4 min. The detector temperature was 270 °C. The helium carrier gas flow rate was 1.5 mL min^{-1}.

The mass spectra were recorded with the electron multiplier at 1800 V and the ionization energy was preset at 70 eV. Selective ion monitoring (SIM) at ion masses of 321, 323, 336, and 338 at 100 ms dwell times was used to analyze the extracts.

Calculations

Total ^{14}C estimates of plant uptake of PCP were based on ^{14}C of dry combusted plant material. This ^{14}C represents the maximum amount of PCP possible in the plant as ^{14}C-PCP, ^{14}C-labeled metabolites, or $^{14}CO_2$. Carbon-14 was detected in the control plants grown in soil containing no ^{14}C-PCP. The ^{14}C in the control plants probably represented foliar assimilation of the $^{14}CO_2$ released from pots containing ^{14}C-treated soils. This contamination was assumed uniform across all treatments. Thus, ^{14}C contents for crops were recorded as net ^{14}C (gross ^{14}C dpm g^{-1} in each treatment minus ^{14}C dpm g^{-1} in the controls, mean = 300 dpm g^{-1}). Bioconcentration factors (BCF) were calculated by dividing the net ^{14}C (dpm g^{-1}) in the dry plant material by the initial ^{14}C (dpm g^{-1}) in the dry soil.

RESULTS AND DISCUSSION

Carbon-14 Concentration in Plants

Analysis of plant tissue by combustion (total ^{14}C) suggested uptake of ^{14}C-PCP (Bellin, 1989). Confirmation of intact PCP in the plant tissue, however, was necessary due to possible ^{14}C-PCP degradation and detection of ^{14}C-labeled metabolites. The PCP was extracted with methylene chloride from 50 samples with the highest combustion ^{14}C contents. The ^{14}C in the methylene chloride extract represents PCP, lower chlorinated phenols, and anisols (Casterline et al., 1985).

Apparent PCP uptake based on ^{14}C in the methylene chloride extracts (Bellin, 1989) was significantly less than that based on total ^{14}C (combustion) analysis even after correcting for an extraction efficiency of 93%. Some carrot peel methylene chloride samples still contained high amounts of ^{14}C, but they were significantly less than ^{14}C in the combusted samples. Apparently, most of the ^{14}C in the plant tissue was not PCP or its immediate degradation products.

Pentachlorophenol Concentration in Plants

Selected methylene chloride extracts, with the highest ^{14}C content, were analyzed by GC/MS (SIM) to identify only intact PCP. Trace amounts of intact PCP were detected in the fescue and lettuce. No PCP was detected in the carrot (foliage and peel) tissue extracts. Chile was not analyzed for intact PCP because of minimal ^{14}C in the methylene chloride extracts. These results confirm that most of the ^{14}C in the plants is not PCP, or its expected immediate degradation products.

In these experiments, ^{14}C (dpm g^{-1}) can be misleading as an indicator of possible PCP accumulation. The same amount of radioactivity (0.184 ± 0.018 MBq kg^{-1}) was added to each treatment. The amount of PCP per unit of radioactivity (specific activity) increased with increasing PCP rate, however. Therefore, direct comparison of ^{14}C g^{-1} is not an accurate indicator of the actual PCP present. Measurement of intact PCP is required to determine the actual PCP contamination.

Low PCP concentrations in the plant tissues do not suggest whether PCP was degraded in the soil or metabolized by the plant. A degradation study (Bellin et al., 1990) suggested a half-life of 8 and 15 d in the unamended and amended Bluepoint soil, respectively. Analysis of ^{14}C in methylene chloride extracts from the Glendale soil (Bellin, 1989) suggested a half-life of approximately 9 d. Lagas (1988b) reported a half-life of PCP of 10 to 15 d in a low organic matter, high pH (8) soil. Rapid soil degradation would leave little PCP available for plant uptake.

Plants germinated 4 to 12 d after application. Thus, at least 25% of the PCP degraded by the time (4–12 d) the plants could begin translocating PCP to aboveground tissue. The concentration of ^{14}C-PCP in soil decreases with time, suggesting that uptake should follow the same trend. Therefore, we could expect the greatest uptake for fescue followed by lettuce, carrot, and chile. Even if all the uptake occurred early in the crop's growth, longer times to harvest would allow greater time for plant metabolism of PCP (Scheel et al., 1984; Schäfer and Sandermann, 1988) or growth dilution. In any case, PCP concentrations would be expected to decrease in the order of fescue > lettuce > carrot > chile.

Bioconcentration Factors

Bioconcentration factors (BCFs) were used to quantify plant contamination with PCP. Figure 1 shows various measures of lettuce contamination (as BCFs) as a function of soil organic C content. The BCFs based on total combustion ^{14}C decreased with increasing organic C content. This agrees with increased PCP sorption (reduced availability) with increased soil organic C content (Bellin et al., 1990). More important was

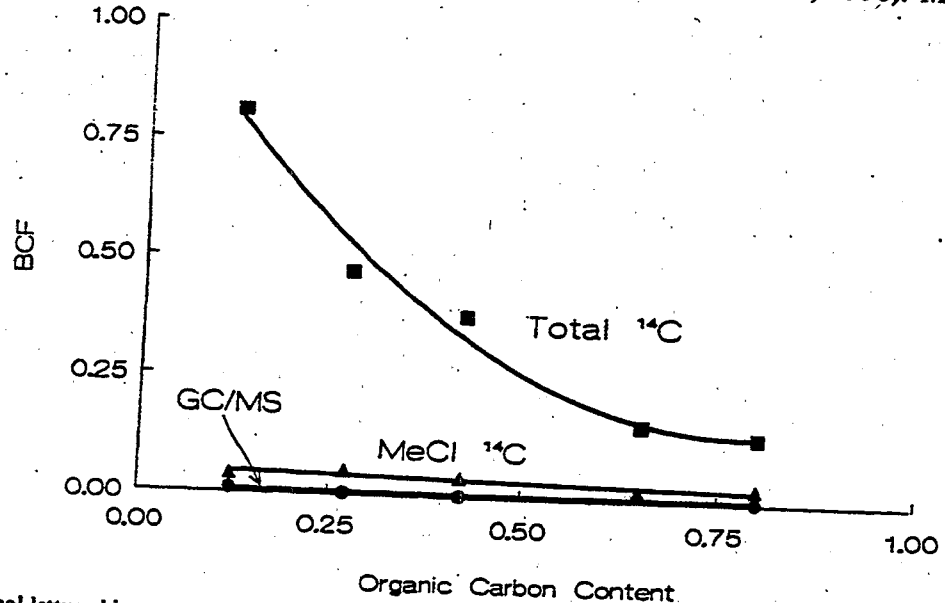

Fig. 1. Pentachlorophenol lettuce bioconcentration factors based on total ^{14}C (■), ^{14}C in methylene chloride extracts (▲), and actual PCP by GC/MS (●) as a function of soil organic C content.

the change in BCF values, calculated from ^{14}C in methylene chloride extracts. This effect was even more apparent when BCFs were calculated from actual PCP concentration, as determined by GC/MS SIM analysis of the methylene chloride extracts.

The BCFs for fescue, carrot, and chile (Table 1) were calculated from total ^{14}C determined by combustion, ^{14}C in the methylene chloride extracts, and GC/MS measurements. The BCFs calculated from total ^{14}C determined by combustion were maximum BCF values representing ^{14}C as PCP, PCP-metabolites, and any other compound containing ^{14}C. The BCFs from ^{14}C in the methylene chloride extracts (representing PCP, lower chlorinated phenols, and anisols) were substantially lower. The BCFs based on GC/MS analysis (actual PCP) were less than 0.01 for all crops. The PCP was not detected in the carrot peels or tops. The extracted radioactivity in chile samples was only 5% of the total (combustion) radioactivity. Thus, due to already low BCFs based on ^{14}C in the methylene chloride extracts, chile (foliage and fruit) was not analyzed by GC/MS.

The BCFs based on actual PCP suggest minimal plant uptake in any sludge or PCP rate treatment. This was likely caused by rapid degradation of PCP in the soils.

Topp et al. (1986) reported PCP concentration factors (fresh plant weight/air dry soil) for barley (*Hordeum vulgare* L.) of about 7 after 1 wk in a soil with a pH 6.4. This concentration factor, however, based on ^{14}C, likely overestimated actual PCP accumulation. Topp et al. (1986) attributed the large bioconcentration factors to rapid degradation and mineralization of PCP, followed by plant uptake of $^{14}CO_2$.

Casterline et al. (1985) reported BCF values (fresh plant weight and air dry soil) of $\gg 1.0$ for spinach plants, soybean plants, and soybean roots, based on analysis by gas chromatography with an electron capture detector. Thus, BCFs suggested significant PCP uptake and much greater uptake than measured here. Several factors may have contributed to the greater bioconcentration factors. Soil analysis revealed a half-life of PCP of about 25 d (whereas the PCP half-life in our soils was about 10 d). The relative average plant exposure increases with increasing half-life. Based on a 25- and 10-d PCP half-life the relative average exposure would be 0.3 and 0.15, respectively, for a 100-d growing season (Ryan et al., 1988). Thus, greater uptake would be expected with a longer PCP half-life. Casterline et al. (1985) sterilized their soil to intentionally delay degradation. The low soil pH (estimated to be <5.5) may also have reduced the activity of microorganism responsible for PCP degradation. DeLaune et al. (1983) reported maximal degradation at pH 8, and decreasing degradation with increasing or decreasing pH. Additionally, PCP sorption (Bellin et al., 1990) was greater in an acid soil than two alkaline soils. Adsorbed PCP may be less available for degradation than solution PCP. Speitel et al. (1989) reported slower degradation when PCP was adsorbed to granular activated carbon than when PCP remained in solution. Therefore, slower degradation in the acid soil due to increased sorption allowing longer PCP availability and thus, greater opportunity for plant uptake.

SUMMARY AND CONCLUSIONS

Detection of intact PCP by GC/MS in fescue and lettuce revealed minimal plant uptake of intact PCP. The BCF values (plant dry wt./initial soil concentrations) were <0.01 for fescue and lettuce. On a fresh-weight basis, BCF values were <0.001. Intact PCP was not detected in the carrot (foliage, peel, or pulp) and chile plants (foliage or pods).

The GC/MS analysis of extracts of soil samples taken after each crop harvest suggested PCP degraded so rapidly in these soils that minimal plant contami-

Table 1. Fescue, carrot, and chile bioconcentration factors based on initial soil concentration and dry plant weights.

Sludge rate	PCP rate	Fescue			Carrot†				Chile‡	
					Plant		Peels			
		Total§	MeCl¶	GC/MS#	Total	MeCl	Total	MeCl	Plant Total	Fruit Total
Mg/ha	mg/kg									
					Bluepoint soil					
0	0.6	0.77	0.08	—††	0.06	—	1.22	0.46	0.24	0.08
	5.1	1.54	0.14	0.007 2	1.09	0.06	0.48	—	0.40	0.13
22.5	0.6	1.18	0.08	—	0.04	—	1.34	—	0.23	0.06
	5.1	2.25	0.08	0.000 1	0.23	0.02	0.35	—	0.29	0.15
45	0.06	0.84	0.06	—	0.03	—	1.32	—	0.17	0.06
	5.1	2.18	0.10	—	0.28	0.04	0.33	—	0.19	0.09
					Glendale soil					
0	0.6	1.32	—	—	0.41	—	1.71	0.13	0.15	0.05
	5.1	0.96	0.12	—	0.22	0.03	0.27	0.08	0.16	0.00
22.5	0.6	0.61	0.06	—	0.45	—	2.09	0.16	0.09	—
	5.1	0.60	0.06	<0.000 04	0.10	0.01	0.37	0.07	0.07	0.03
45	0.6	0.24	0.02	—	0.20	—	1.63	0.09	0.17	0.03
	5.1	0.86	—	—	0.13	0.01	0.42	0.09	0.27	0.07

† GC/MS bioconcentration values for carrot peels were < 0.000 1.
‡ MeCl bioconcentration factors for chile plant and fruit were <0.01.
§ Total = ^{14}C by combustion.
¶ MeCl = ^{14}C in methlyene chloride extracts.
# GC/MS = actual PCP by GC/MS.
†† Plant samples not analyzed.

nation could occur in the field. Thus, given normal application rates of sludge with normal (or even abnormally high) PCP concentrations, concerns about food chain plant uptake of PCP should not limit land application of sludge in these soils. This conclusion is likely appropriate to other high (>6.5) pH soils. However, further study is necessary to determine the bioavailability of PCP in sludge-amended, low-pH soils, particularly those with high organic C contents where PCP half-lives are reportedly much longer (Bellin et al., 1990).

REFERENCES

Banerji, S.K., K. Piotek, and J.T. O'Connor. 1986. Pentachlorophenol adsorption on soils and its potential for migration into groundwater. Hazardous and industrial solid waste testing and disposal. Vol. 6. ASTM STP 933. p. 120–139. Am. Soc. for Testing and Materials, Philadelphia, PA.

Bellin, C.A. 1989. Plant uptake of pentachlorophenol in sludge amended soils. M.S. thesis. New Mexico State Univ., Las Cruces, NM.

Bellin, C.A., G.A. O'Connor, and Y. Jin. 1990. Sorption and degradation of pentachlorophenol in sludge-amended soils. J. Environ. Qual. 19:603–608 (this issue).

Bevenue, A., and H. Beckman. 1967. Pentachlorophenol: A discussion of its properties and its occurrence as a residue in human and animal tissues. Res. Rev. 19:83–134.

Buhler, D.R., M.E. Rassmusson, and H.S. Nakaue. 1973. Occurrence of hexachlorobenzene and pentachlorophenol in sewage sludge and water. Environ. Sci. Technol. 7:929–934.

Casterline, J.L., N.M. Barnett, and Y. Ku. 1985. Uptake, translocation, and transformation of pentachlorophenol in soybean and spinach plants. Environ. Res. 37:101–118.

Crosby, D.G. 1980. Environmental chemistry of pentachlorophenol: A special report on pentachlorophenol in the environment. p. 1052–1080. *In* Commission on pesticide chemistry. Dep. of Environmental Toxicology, Univ. of California, Davis, CA.

DeLaune, R.D., R.P. Gambrell, and K.S. Reddy. 1983. Fate of pentachlorophenol in estuarine sediment. Environ. Pollut. Ser. B 6:297–308.

Fahrig, R., C.A. Nilsson, and C. Rappe. 1978. Genetic activity of chlorophenols and chlorophenol impurities. p. 325–338. *In* [illegible] Rao (ed.) Pentachlorophenol. Plenum, New York.

Hamaker, J.W., and J.M. Thompson. 1972. Adsorption. p. 49–143. *In* C.A.I. Goring and J.W. Hamaker (ed.) Organic chemicals in the soil environment. Marcel Dekker, New York.

Jacobs, L.W., G.A. O'Connor, M.R. Overcash, M.J. Zabek, and P. Rygwiecz. 1987. Effect of trace organics in sewage sludges on soil-plant systems and assessing their risk to humans. p. 101–143. *In* A.L. Page et al. (ed.) Land application of sludge. Lewis Publishers, Chelsea, MI.

Kloskowski, R., I. Schuenert, W. Klein, and F. Korte. 1981. Laboratory screening of distribution, conversion and mineralization of chemicals in the soil-plant system and comparison to outdoor experimental data. Chemosphere 10:1089–1100.

Lagas, P. 1988a. Sorption of chlorophenols in the soil. Chemosphere 17:205–216.

Lagas, P. 1988b. Behavior of chlorophenols in soil. p. 264–266. *In* A.A. Orio (ed.) Environmental contamination, 3rd International Conference. CEP Consultants Ltd., Edinburg, UK.

Poole, C.F. 1978. Advances in silylation of organic compounds for GC. p. 152–200. *In* K. Blau and G.S. King (ed.) Handbook of derivatives for chromatography. Heydon and Son, Ltd., London.

Ryan, J.A., R.M. Bell, J.M. Davidson, and G.A. O'Connor. 1988. Plant uptake of non-ionic organic chemicals from soils. Chemosphere 17:2299–2323.

Schäfer, W., and H. Sandermann, Jr. 1988. Metabolism of pentachlorophenol in cell suspension cultures of wheat (*Triticum aestivum* L.). Tetrachlorocatechol as a primary metabolite. J. Agric. Food Chem. 36:370–377.

Scheel, D., W. Schäfer, and H. Sandermann, Jr. 1984. Metabolism of pentachlorophenol in cell suspension cultures of soybean (*Glycine max* L.) and wheat (*Triticum aestivum* L.). General results and isolation of lignin metabolites. J. Agric. Food Chem. 32:1237–1241.

Speitel, G.E., Jr., C. Lu, M. Turakhia, and X. Zhu. 1989. Biodegradation of trace concentrations of substituted phenols in granular activated carbon columns. Environ. Sci. Technol. 23:68–74.

Topp, E., I. Scheunert, A. Attar, and F. Korte. 1986. Factors affecting the uptake of ^{14}C-labelled organic chemicals from soil. Ecotox. Environ. Safety 11:219–228.

U.S. Environmental Protection Agency. 1979. Water related environmental fate of 129 priority pollutants. Vol. 1 and 2. USEPA Rep. 440/4-79-029b. NTIS, Springfield, VA.

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Adsorption, Degradation, and Plant Availability of 2,4-Dinitrophenol in Sludge-Amended Calcareous Soils

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ABSTRACT

2,4-Dinitrophenol (DNP) is a moderately weak acid that is expected to be highly labile (leachable and plant available) in high-pH soils. The adsorption and degradation behavior of DNP in two sludge-amended, calcareous soils was determined and used to explain DNP uptake by plants grown in the soils in the greenhouse. The DNP adsorption was minor in both soils and was only slightly affected by sludge. The DNP degradation was rapid in both soils and was unaffected by sludge. Thus, despite limited soil adsorption, plant uptake of DNP was minor in all crops and plant parts owing to rapid soil DNP degradation. Even if a municipal sludge highly contaminated with DNP was identified (an unlikely occurrence), concerns over possible plant contamination should not limit sludge applications to calcareous soils at agronomic rates. Rapid degradation will minimize opportunities for plant uptake of DNP from contaminated soils or leaching of DNP to groundwater, given careful water management.

2,4-DINITROPHENOL (DNP) is an active compound of considerable phytotoxicity to both animals and plants (Shea et al., 1983). It occurs as a waste contaminant originating from several industrial sources, and may occur as a degradation product of other compounds. Shea et al. (1983) reviewed the various biological activities of DNP in plant-soil systems, and noted that DNP behavior is highly pH-dependent owing to its weak acid character (pKa = 4.09). Adsorption of DNP is favored by low soil pH, but degradation is favored by high soil pH. They cautioned that careful management is necessary if DNP is a predominant component in land-applied waste materials.

The purpose of this study was to determine DNP behavior in calcareous (high-pH) soils amended with municipal sewage sludge and to determine the extent to which DNP is available to food-chain crops grown in such soils.

In high-pH, calcareous soils, DNP adsorption should be minimal and DNP should be readily available to plants (Shea et al., 1983; Ryan et al., 1988). Sludge additions to soils may, however, increase DNP adsorption (Shea et al., 1983) and thereby reduce DNP plant availability. Studies of DNP uptake by plants from sludge-amended soils have not been conducted. This study was intended to supply such information and to explain the uptake data in terms of DNP soil behavior.

MATERIAL AND METHODS

Adsorption, degradation, and plant uptake studies were conducted with two calcareous soils from New Mexico. The Glendale clay loam [pH 8.0, 6.5 g organic C (OC)/kg, and 132 g $CaCO_3$/kg] is classified as a fine-silty, mixed thermic Typic Torrifluvent. The Bluepoint sandy loam (pH 8.0, 1.2 g OC/kg and 30 g $CaCO_3$/kg) is classified as a mixed, thermic Typic Torripsamment. Surface (0–15 cm) samples of both were air-dried and sieved (<2 mm) before use.

Soils were amended with a secondary, anaerobically digested sewage sludge from Albuquerque, NM. Air-dried sludge was gamma-irradiated (^{137}Cs, 10 kGy) to further reduce pathogens. Sludge was thoroughly mixed, sieved (<2 mm), and stored in 30-L plastic containers for use in all studies. A screen for priority pollutants (USEPA method 625), utilizing a gas chromatograph with a flame ionization detector, revealed no detectable (1 mg/kg) DNP in the sludge. Sludge rates included 0 and 45 Mg/ha equivalent (20 g/kg).

Adsorption Study

The batch equilibration technique was used at a soil/solution ratio of 10:11 (w/v). Initial DNP soil concentrations were 1.0, 10.0, 50.0, and 100.0 mg/kg. Each solution consisted of 10 mL of nonradioactive DNP of appropriate concentration and 1 mL of ^{14}C-DNP. The rates were chosen to cover a wide range of concentrations, including rates much higher than expected in normal municipal sludge-amended soils. The median concentration of most organics in municipal sludges is ≤10 mg/kg. Such sludges applied at agronomic rates would result in DNP concentrations in soil-sludge mixtures of ≤0.1 mg/kg (Jacobs et al., 1987). Uniformly ring-labeled ^{14}C-DNP (specific activity 1.89 Bq/kg, >99% purity, Sigma Chemical Company, St. Louis, MO) was present in each flask at 2.442 dpm/kg soil (disintegrations per minute per kilogram). All DNP solutions were prepared in 10^{-3} *M* NaCl (as background electrolyte) adjusted to pH 8.0 with NaOH. This pH was chosen to match the soil initial pH values to reduce pH variations of soil solutions that might otherwise occur with the different DNP (acid) concentrations.

The suspensions were shaken on a wrist-action shaker in the laboratory (22 °C) for 18 h. Such vigorous agitation promotes thorough mixing of the solute-containing solution with soil, and normally promotes rapid attainment of adsorption equilibrium. A preliminary time study suggested that equilibrium was not attained in 18 h. We nevertheless chose 18 h of shaking for convenience and to minimize degradation effects (see below) on the disappearance of DNP from solution. Extrapolation of a preliminary percent adsorbed vs. time curve suggested adsorption equilibrium would require about 36 h and would yield an additional 5% adsorption.

After shaking, the suspension pH of the lowest (1.0) and highest (100.0 mg/kg) treatments was measured. Suspensions were then centrifuged (900 × *g*) to separate supernatants. An aliquot (0.5 mL) of the clear supernatant was added to scintillation cocktail and counted to 2 sigma percent error. An external standard was used to correct for quenching. Counting efficiency was typically ~0.75.

Mass of DNP adsorbed was calculated as the difference in mass of DNP originally added and that remaining after 18 h. Two soil-less blanks were included for each soil to detect DNP losses from solution by mechanisms other than adsorption. The soil-less blanks consisted of 100 mg DNP/L.

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Degradation Study

The DNP degradation was measured in a simple flow-through incubation system. Air under vacuum was first bubbled through water to humidify the air and to minimize soil drying. The humid air was then drawn through the air space of a flask containing 60 g of soil amended with 0 or 45 Mg/ha (0.02 g/g) sludge. The DNP was added at 3.7 mg/kg, and consisted of reagent-grade DNP plus 18.5 kBq ^{14}C-DNP/flask. Carbon dioxide (including $^{14}CO_2$) was trapped in a final test tube containing 1 *M* NaOH. Soils were moistened to water contents representative of moisture conditions maintained in the greenhouse study of DNP plant uptake (see below). The moisture contents (0.16 kg/kg for the Bluepoint sandy loam and 0.38 kg/kg for the Glendale clay loam) were slightly greater than field capacity. A total of 14 flasks for each soil-sludge mixture allowed duplicate sampling at 0, 1, 2, 4, 8, 16, and 32 d. (Preliminary studies suggested most of the DNP would degrade between 2 and 16 d, and that volatilization losses of DNP were negligible.) Flasks were removed from the train at appropriate times, and soils were immediately extracted with a mixture of 30 mL methanol and 30 mL 0.1 *M* NaCl. The suspensions were shaken 3 h, and then centrifuged (900 × *g*) for 30 min to separate equilibrium solutions. The supernatants were transferred to glass bottles. One-milliliter aliquots were removed for ^{14}C assay; the remainder was refrigerated for subsequent high pressure liquid chromatography (HPLC) analysis. Approximately 1 mL of each supernatant was centrifuged (12 680 × *g*) for 2 min to further clear the samples. Aliquots (25 to 250 μL, depending on DNP concentration) were injected for HPLC analysis.

The HPLC conditions were: RP-18 column, 4.6 × 250 mm, mobile phase, methanol: 1% acetic acid at pH 2.8 (50:50 v/v), flow rate 1 mL/min. Under these conditions, the retention time was 8.4 to 8.8 min. Detector: UV at 254 mm, limit of detection was equivalent to 0.0174 mg DNP/kg soil dry weight (~0.5% of initial).

Extracted soil was air-dried and ground (mortar and pestle) for combustion. About 300 mg soil was weighed into ceramic boats, covered with catalyst (activated alumina/cupric oxide powder, 5:1 w/w), and combusted at 1000 °C for 8 min in an oxygen stream of 150 mL/min. Evolved $^{14}CO_2$ was trapped in a mixture of 10 mL ethanolamine-ethoxyethanol (2:3, v/v) and counted by liquid scintillation. Combustion efficiencies were typically ~95%. Samples corresponding to Day 0 and 1 were combusted to determine mass balance and extraction efficiency. Mass balance (^{14}C extracted + ^{14}C remaining in soil) averaged 101% across soils for times Day 0 and 1. Extraction efficiencies averaged 65% (SD = 3.9%) for the Glendale soil and 69% (SD = 7.7%) for the Bluepoint soil, with no sludge effect in either soil. A preliminary incubation study utilized alcohol (95% MeOH) as the extractant (Overcash et al., 1982), but yielded extraction efficiencies that varied from 91 to 52% depending on soil and sludge treatment, even over short (1 d) times. Overcash et al. (1982) also reported that the alcohol extraction efficiency varied (80–120%) with DNP concentration. The mixture of MeOH and NaCl used in this study seems preferable to alcohol alone. The extraction efficiency was less than ideal, but was reproducible, and similar for both soils.

Greenhouse Study

Sludge, soils (2 kg), and 2 g of fertilizer (18–48–0; 18–21–0, N–P–K) were thoroughly mixed in a blender. Fertilizer additions were intended to equalize fertility differences of soils and sludge treatments.

Six DNP rates (0, 0.1, 0.5, 1.0, 5.0, and 10.0 mg/kg) were imposed on the soil and sludge treatments. The rates were chosen to avoid possible phytotoxic (>20 mg DNP/kg) effects on the crops (Overcash et al., 1982) and to encompass both reasonable and excessively high DNP rates expected from additions of sludge containing priority pollutants at concentrations <10 mg/kg (Jacobs et al., 1987).

The 24 treatments (six DNP rates, two sludge rates, two soils) per crop were replicated twice. The decision to invest in several DNP rates rather than more replicates was made to detect the nature of plant response trends with DNP rate more precisely than with fewer DNP rates and more replicates. With the same total number of pots, the estimate of variance is the same, but the precision of the estimate of the mean response is better with more DNP rates than with more replicates. Each DNP treatment, except the control, consisted of nonradioactive, reagent-grade DNP plus uniformly ring-labeled ^{14}C-DNP (specific activity 1.89 Bq/kg, >99% purity; Sigma Chemical Company, St. Louis, MO). The 0.1 mg/kg rate consisted of only ^{14}C-DNP. The amount of ^{14}C added to each pot was the same (11 100 dpm/kg, SD = 10%). Soil-sludge-fertilizer mixes were spread uniformly on a tray covered with aluminum foil. The DNP solutions (labeled and unlabeled) were uniformly applied to the soils with a syringe. The soils were mixed for several minutes by hand and then transferred to plastic pots. The mass of prepared soil in each pot was 1.8 kg. The excess soil (200 g) was retained for DNP analysis (see below) and for covering seeds.

Soils were seeded with fescue (*Festuca arundinacea* Schreb., 'KY 31'), carrot (*Daucus carota* L., 'Nantes'), lettuce (*Lactuca sativa* L., 'Black Seeded Simpson'), and chile pepper (*Capsicum annuum* L., 'Espanola' improved). Seeds were sprinkled uniformly on the soil surface, and covered with a few millimeters of dry soil reserved from each pot. After seeding, pots were watered to pot-holding capacity (0.16 kg/kg for Bluepoint, 0.38 kg/kg for Glendale). Each pot was then covered with newspaper to minimize evaporation during germination. Some leaching occurred from several pots in the initial watering, but leachate was collected in plastic saucers and was reapplied to the respective pots. Plants were watered approximately every 2 d to return them to initial pot-holding weights. No subsequent drainage occurred.

Fescue was grown for 32 d, lettuce for 43 d, carrot for 70 d, and chile pepper for 90 d. Natural light was supplemented as needed by eight 400-W sodium lamps to supply 16 h of light. Temperatures in the greenhouse varied from 16 to 35 °C during the experiment.

Fescue, lettuce, and chile were harvested by cutting plants about 3 cm above the soil surface. Carrot plants were removed with the main tap root intact. Tops and roots were separated with a razor blade and were washed with distilled water until no visible soil particles remained. The roots were then peeled, weighed, and stored in paper bags. Peels were also weighed and stored in bags. Plant fresh weight yields were recorded immediately after harvest. All plant material was then dried (50 °C) for a minimum of 12 h. Dried plant material was weighed, ground, and stored in plastic bags for later analysis.

Soils retained from the initial mixing and labeling with ^{14}C-DNP and dried plant samples were assayed for ^{14}C by combustion as in the degradation study. Approximately 50 mg plant tissue or 300 mg ground (mortar and pestle) soil was combusted.

Analysis of plant material for intact (parent compound) DNP was performed by an independent analytical firm using approved USEPA extraction and clean up procedures. Detection was by gas chromatography with flame ionization detection. Given the limited mass of plant tissue available for extraction (~9 g, reps combined), the limit of detection for DNP was 0.146 mg/kg.

Treatments were arranged in a random complete block design. Analyses of variance were conducted for the variables crop, plant parts, DNP rates, sludge treatment, and

bioconcentration factors based on ^{14}C. An LSD test was performed for variable means exhibiting significant differences in the analysis of variance.

Results and Discussion

Adsorption

Adsorption data for the sludge-amended calcareous soils are summarized in Table 1 along with the pH values of the equilibrium suspensions measured. The pH values of intermediate DNP treatments are assumed to be similar. The paste pH values of both soils in distilled water are 8.0. The lower pH values of the adsorption equilibrium suspensions are primarily caused by the background salt (10^{-3} *M* NaCl) rather than acidifying effect of the DNP. Sludge-amended soil pH values were consistently lower than unamended soil pH values, but the effect on adsorption was probably minor. The pKa of DNP (4.09) is at least 2.6 units lower than the soil pH values, so <0.25% of the DNP exists as undissociated acid in the most acidic treatment (pH 6.7) and <0.14% at pH 7.0.

The DNP adsorption was minor in both soils owing to their negative charges and the dominance of the anionic form of DNP. The Glendale soil exhibited positive adsorption (Freundlich K = 0.67 and 0.35, unamended and amended, respectively) consistent with adsorption of other weak acid organics 2, 4-D and 2,4,5-T (O'Connor et al., 1981) and pentachorophenol (Bellin et al., 1990) on this soil. Despite the soil's negative charge, adsorption of weak acid compound occurs, and is primarily associated with the organic fraction (O'Connor and Anderson, 1974). Sludge additions slightly decreased DNP adsorption (Table 1), but had no effect on phenoxy herbicide adsorption by the Glendale soil (O'Connor et al., 1981).

The DNP was repelled (negatively adsorbed) from the Bluepoint soil at all DNP concentrations. This soil is extremely low in organic matter (1.2 g OC/kg) and apparently offered no positive adsorption sites. Sludge addition reduced DNP repulsion (Table 1), but DNP was negatively adsorbed in almost all treatments.

Given the minimal adsorption of DNP in both soils in the presence and absence of sludge, DNP mobility is expected to be great. Careful water management would be necessary in DNP-contaminated soils to avoid groundwater pollution (Shea et al., 1983). The DNP activity toward plants would be maximal in both soils and unaffected by sludge additions. Almost all of the chemical would remain in solution available for plant uptake, or other removal processes, e.g., degradation and leaching.

Degradation

The DNP degradation in the sludge-amended calcareous soils (Fig. 1 and 2) is presented as percent DNP remaining (as determined by HPLC) plotted as a function of time. The data have been corrected for extraction efficiencies. Similar plots (not presented) of percent ^{14}C remaining in methanol/NaCl extracts matched closely the data in both figures. Thus, ^{14}C extracted by methanol/NaCl could have served as a surrogate for intact DNP, contrary to results we have obtained for pentachlorophenol (Bellin et al., 1990).

A semilog plot of the data was used to identify first-order degradation kinetics. Neither soil, however, demonstrated the single linear decrease in DNP remaining with time consistent with simple first-order kinetics. An initial linear decrease, lasting a few days, was followed by another linear decrease, of much greater slope, until only a few percent of DNP remained (Fig. 1 and 2). Half-lives could be estimated for DNP in both soils (~5 d in Glendale and ~9 d in Bluepoint), but are misleading. Much less DNP remains in the soils after 2 half-lives than the 25% expected from first-order kinetics. A more meaningful description would be that DNP has almost completely degraded in the Glendale soil in 8 d, and in the Bluepoint soil in 16 d. There was no significant effect of sludge on DNP degradation in either soil.

The DNP degradation in these soils was more rapid than noted by other investigators. Overcash et al. (1982) reported 62 to 66% DNP loss after 4 wk in the acid Davidson clay loam [clayey, kaolinitic, thermic (oxidic) Rhodic Paleudults]. The USEPA (1979) reports a half-life of 50 d for DNP. Miller (1977) classified phenolic compounds as slowly degradable. The DNP can be rather persistent in both soils and aquatic systems, but decomposition by certain strains of bactericide and by a fungus has been demonstrated (Shea et al., 1983). The DNP is bactericide at high concentrations and low pH, thus, both these factors influenced toxicity and metabolism. The optimum pH for microbial decomposition (by reduction of the nitro groups to amino groups, followed by oxidative deamination, or by the release of a nitro group as nitrite) is near neutrality (Shea et al., 1983). Decomposition by release of NO_2 by *Corynebacterium simplex* was maximal at pH 8.0 (Gundersen and Jensen, 1956).

Table 1. The DNP adsorption on two sludge-amended calcareous soils.†

Treatment		Glendale				Bluepoint		
DNP	Sludge	Ci	Ce	x/m	pH	Ce	x/m	pH
mg/kg	Mg/ha	mg/L		mg/kg		mg/L	mg/kg	
1.0	0	0.93	0.54	0.43	7.0	1.01	−0.09	7.1
	45	0.93	0.70	0.26	6.8	0.93	0.00	6.9
10.0	0	9.1	6.9	2.5		9.7	−0.6	
	45	9.1	7.4	2.0		9.3	−0.2	
50.0	0	45.5	39.1	7.2		50.5	−5.6	
	45	45.5	41.1	4.6		47.3	−2.0	
100.0	0	90.9	82.6	9.5	7.0	99.2	−3.2	7.2
	45	90.9	87.5	3.9	6.9	95.9	−5.6	6.7

† Ci = initial concentration; Ce = equilibrium concentration; x/m = amount adsorbed per unit mass of soil. All values are the average of two replicates.

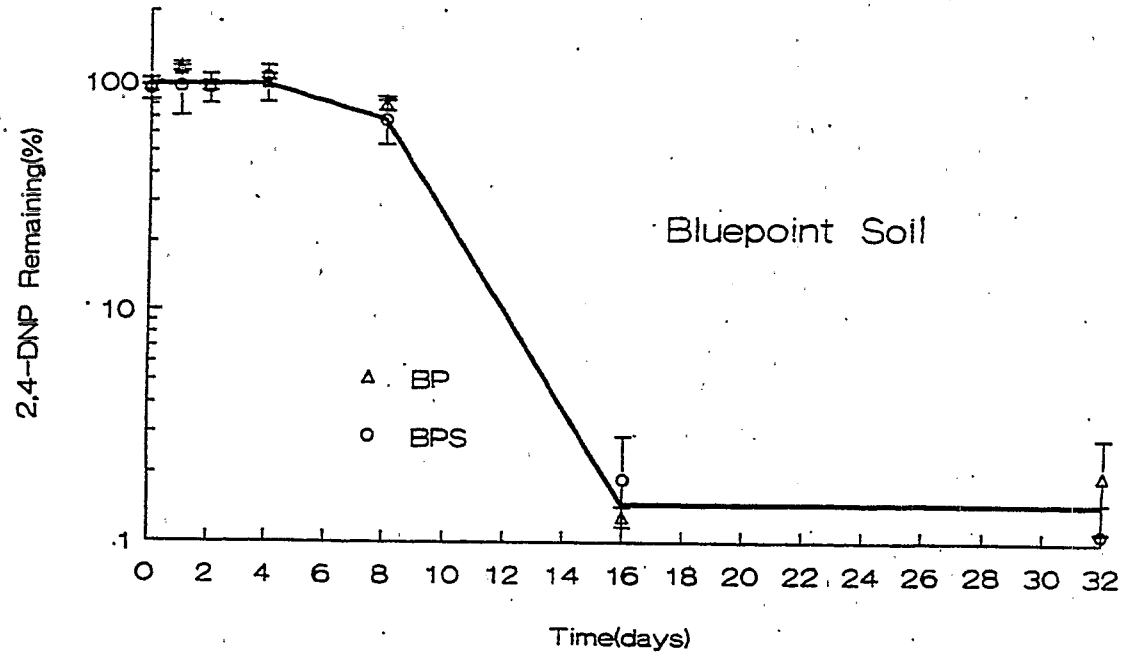

Fig. 1. The DNP degradation in Bluepoint soil (means ± 1 SD).

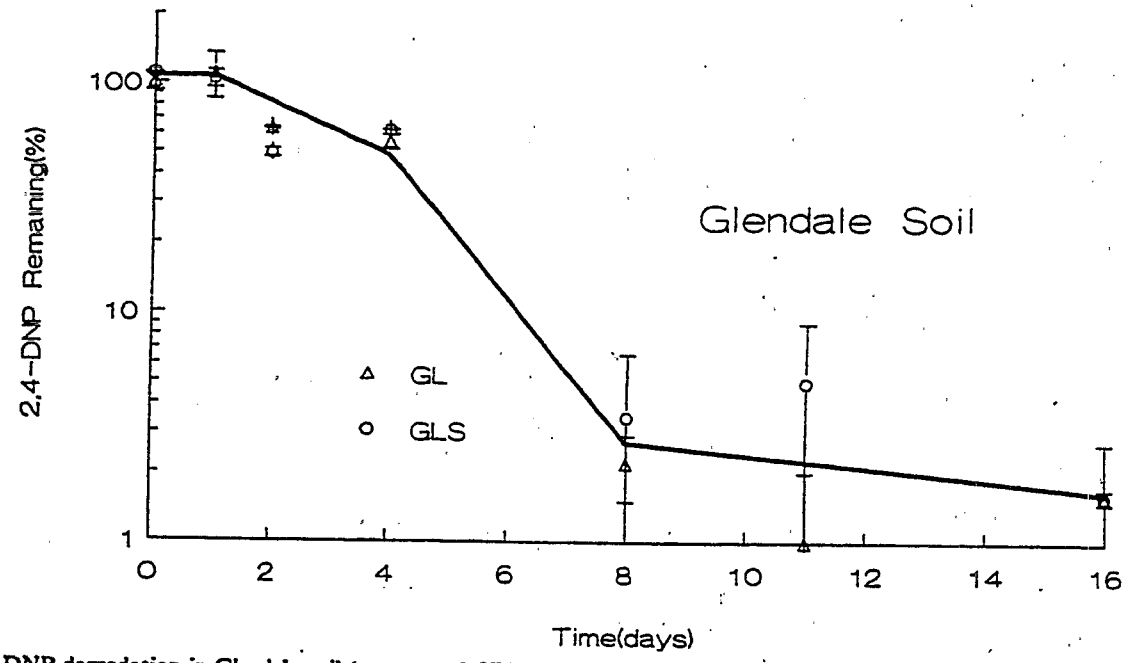

Fig. 2. The DNP degradation in Glendale soil (means ± 1 SD).

Photochemical hydrolysis of DNP has not been demonstrated, but has been suggested as possible (Shea et al., 1983). Flasks in our study, however, were wrapped with aluminum foil to exclude light, and there was no evidence of photolysis in the adsorption study (exposed containers). Volatilization was also considered unlikely based on previous work (Overcash et al., 1982) and DNP's low Henry's constant (2.7×10^{-8}, dimensionless). Lack of volatilization was confirmed in a preliminary study with soil-less blanks. Thus, the rapid dissipation of DNP in our soils was attributed to microbial activity, favored by the initial low (3.7 mg/kg) DNP concentration and high pH.

The rapid degradation of DNP in high pH soils has important environmental consequences. The essentially complete degradation of DNP in 8 or 16 d (Glendale and Bluepoint, respectively) means that little chemical remains in the soils long enough for significant plant uptake. Also, DNP is weakly, or negatively, adsorbed in high-pH soils and could threaten groundwater if the soils are leached excessively. Careful water management that ensures chemical residence times of

Table 2. Effect of DNP and sludge on yields† of crops in high pH soils in the greenhouse.

Treatment		Fescue		Glendale					
					Carrot			Chile	
DNP	Sludge	Glendale	Bluepoint	Lettuce	Tops	Roots	Peels	Plant	Fruit
mg/kg	Mg/ha	g dry wt./pot							
0	0	2.60	1.65	4.55	4.30	2.45	1.30	6.25	1.35
	45	3.80	3.05	4.00	4.15	1.80	0.80	8.15	1.75
0.1	0	3.45	2.80	5.50	3.85	1.15	0.75	6.70	2.25
	45	4.70	2.85	5.90	4.10	0.85	0.60	6.95	1.55
0.5	0	4.50	3.20	5.65	4.85	2.60	1.30	6.35	1.65
	45	4.00	2.80	4.25	4.40	1.20	0.75	6.35	2.85
1	0	4.60	3.80	5.35	4.80	2.25	1.75	8.05	4.90
	45	4.20	3.15	6.00	4.95	1.10	1.10	8.40	4.20
5	0	4.35	3.20	3.55	5.50	1.70	1.55	5.30	2.35
	45	4.30	3.30	5.55	6.15	1.55	0.95	6.30	3.05
10	0	3.15	3.00	5.15	5.60	2.00	1.50	6.05	3.00
	45	3.65	2.00	4.65	4.15	1.15	0.75	6.90	4.55

† Mean of two replicates.

Table 3. The DNP bioconcentration factors† (dry-wt. basis) based on ^{14}C and intact DNP analysis.

		Fescue				Glendale									
								Carrot				Chile			
		Glendale		Bluepoint		Lettuce		Tops		Roots		Peels		Plant		Fruit	
DNP	Sludge	^{14}C	DNP	^{14}C	DNP	^{14}C	DNP	^{14}C	DNP	^{14}C	DNP	^{14}C	DNP	^{14}C	DNP	^{14}C	DNP
mg/kg	Mg/ha																
0	0	–		–		–		–		–		–		–		–	
	45	–		–		–		–		–		–		–		–	
0.1	0	0.432	<13.0‡	2.05	<28.0	0.068	<5.00	0.130	<25.0	0.032	<26.0	0.083	<26.0	0.045	<5.00	0.010	<26.0
	45	0.295	<19.0	3.23	<24.0	0.131	<5.00	0.103	<21.0	0.000	<26.0	0.066	<26.0	0.071	<5.00	0.029	<26.0
0.5	0	0.781		3.87		0.095		0.097		0.031		0.067		0.040		0.031	
	45	0.472		1.90		0.141		0.062		0.000		0.000		0.056		0.030	
1.0	0	0.840	<2.00	3.93	<2.10	0.154	<1.50	0.115	<0.800	0.024	<1.60	0.043	<2.60	0.058	<0.500	0.016	<0.500
	45	0.838	<2.00	3.22	<2.10	0.158	<1.10	0.142	<0.800	0.000	<2.60	0.006	<2.60	0.058	<0.500	0.018	<0.500
5.0	0	1.46		4.19		0.196		0.138		0.069		0.056		0.113		0.072	
	45	0.542		2.82		0.258		0.028		0.000		0.004		0.065		0.026	
10.	0	1.07	<0.200	3.46	<0.240	0.199	<0.170	0.042	<0.050	0.028	<0.260	0.027	<0.260	0.057	<0.050	0.019	<0.050
	45	1.05	<0.200	2.59	<0.250	0.383	<0.170	0.055	<0.050	0.000	<0.260	0.048	<0.260	0.073	<0.050	0.012	<0.190

† Bioconcentration factors calculated as net ^{14}C (dpm/g) or μg DNP/g in plant tissue ÷ ^{14}C (dpm/g) or μg DNP/g in soil initially.
‡ Less than symbol (<) signifies less than detection limit (0.146 mg/kg) in plant material. Limit varies with mass of plant material available for analysis.

about 2 wk in the active root zone, however, should minimize groundwater pollution with DNP. The relatively toxic DNP will degrade almost completely to much less toxic constituents (Shea et al., 1983) in high pH soils. The DNP degradation is slower in acid soils ($t_{1/2}$ = 30–50 d; Overcash et al., 1982), but adsorption is greater so residence times are longer (Shea et al., 1983).

Greenhouse Study

Acceptable yields of fescue were obtained from both soils (Table 2), but growth of lettuce, carrot, and chile pepper was very poor in the Bluepoint soil regardless of treatment (data not presented). Often seeds failed to germinate or plants died before harvest even in the controls. Even fescue yields were low in the Bluepoint soil, but sufficient plant material resulted in all treatments for analysis. The reasons for the differences in crop yields in the two soils are unknown. Thus, yields in Table 2 are primarily for crops grown in Glendale soil.

There was no significant effect of DNP rate or sludge treatment on any crop or plant part yield. The only significant treatment effect on yield was a soil effect, on fescue yield. This lack of treatment effect (except soils) was fortunate as plant accumulations of DNP or ^{14}C could be interpreted without regard to complications associated with yield differences. The lack of DNP rate effects on yield indicates that there was no DNP phytotoxicity at the rates used.

Bioconcentration Factors

Plant and plant part contamination is expressed as bioconcentration factors in Table 3. Bioconcentration factors (BCFs) are calculated as the ratio of ^{14}C (dpm) or intact DNP on a plant tissue dry-weight basis to the concentrations of ^{14}C or intact DNP originally added to the soil. The BCFs calculated on a plant tissue fresh-weight basis (Table 4) are approximately 10-fold less than those calculated on a dry-weight basis, depending on the moisture content of the tissue.

Plant tissue samples from both replicates of all treatments were assayed for ^{14}C by combustion. Confirmational analysis for intact DNP was performed on tissue pooled across replicates for the low (0.1), medium (1.0), and high (10.0 mg/kg) DNP rates. In gen-

Table 4. Various estimates of DNP bioconcentration factors (fresh-wt. basis) at the highest initial DNP soil concentration (10 mg/kg).

Crop	Bioconcentration factor		
	^{14}C	DNP	Likely maximum
Fescue (Glendale)	0.21	<0.040	<0.040
Fescue (Bluepoint)	0.64	<0.045	<0.045
Lettuce	0.034	<0.010	<0.010
Carrot tops	0.01	<0.009	<0.009
Peels	0.003	<0.050	<0.003
Roots	0.000	<0.030	0.000
Chile foliage	0.012	<0.008	<0.008
Fruit	0.001	<0.019	<0.001

eral, BCFs based on ^{14}C were greatest in fescue (both soils) and much less in the other crops. The fescue BCFs imply significant (active, BCFs > 1) bioaccumulation, whereas the BCFs for the other crops imply much less (passive, BCFs < 1) bioaccumulation.

The BCFs calculated on the basis of ^{14}C are common in the literature, but are misleading as the ^{14}C is assumed to represent intact parent compound. If the ^{14}C-labeled compound degrades in soil or is metabolized within the plant, ^{14}C contents of plant tissue falsely describe parent compound contents. Some ^{14}C was detected in the plant tissues from control treatments. Because there was no ^{14}C in the control soils (and no DNP volatilized in the degradation study), ^{14}C in plant tissue in the controls was regarded as $^{14}CO_2$ released from ^{14}C-tagged soils. This contamination was accounted for by subtracting the average ^{14}C content (~0.300 dpm/kg, effective BCF = 0.027) of all controls from the ^{14}C content of each treatment. Nevertheless, net ^{14}C contents of plants may still represent ^{14}C species accumulated other than ^{14}C-DNP. Degradation of DNP was rapid in both soils, being almost complete in 8 d in the Glendale soil and 16 d in the Bluepoint soil. Thus, even before the plants germinated (5–20 d after planting), significant reductions in actual soil DNP concentrations occurred, especially in the Glendale soil. Fescue germinated 5 d after seeding and was harvested 32 d after seeding. The three food-chain crops grew for longer times, but germinated more slowly. Lettuce germinated 20 d after seeding and was harvested on Day 54; carrot germinated on Day 12 and was harvested on Day 70; chile germinated on Day 20 and was harvested on Day 90. Given these growth characteristics, one would expect greatest contamination in fescue, less in lettuce and carrot, and much less in chile. The ^{14}C-based BCFs in Table 3 generally reflect the expected trend.

Evidence exists that DNP is metabolized within plants (Berlin et al., 1971; Klepper, 1979). Thus, even if intact ^{14}C-DNP were accumulated by plants, ^{14}C contents of harvested plant material would not be clear evidence of DNP in tissue. That plant metabolism can completely obviate meaningful interpretations of BCFs based on ^{14}C was clearly demonstrated in studies similar to this utilizing diethylhexyl phthalate (DEHP) as the target chemical (Aranda et al., 1989). Based on all of the above discussion, it is clear that BCFs based on ^{14}C represent very conservative, and probably erroneously high, values.

Attempts to improve on the ^{14}C data by analyzing for intact DNP were disappointing. Because of limited plant tissue, the limit of detection (LOD) for DNP in extracted plant tissue was only 0.146 mg/kg dry weight. Replicates were combined to yield plant tissue for analysis, but only 8 to 9 g dry tissue was obtained (Table 2) resulting in the stated LOD. When less plant tissue was available, the LOD, (per gram tissue) was higher. Thus, BCFs calculated on the basis of detected, intact DNP were limited to LOD values and usually exceeded the BCFs based on ^{14}C (Table 3). Exceptions were for the highest initial DNP treatments (10 mg/kg). Particularly for fescue grown in Bluepoint soil, actual DNP analysis showed the ^{14}C-based BCFs to be grossly in error.

Various estimates of DNP bioconcentration factors for each plant and plant part are given in Table 4. The BCFs are expressed on a fresh-weight basis, because all of these plants would be consumed fresh. Data for unamended and sludge-amended soils have been averaged because there was no significant effect of sludge on BCFs across all treatments. The BCFs are presented for the highest DNP concentration (10 mg/kg) because DNP accumulation is likely greatest at this concentration, and because DNP determinations of intact parent compound are useful only for this DNP rate. The last column represents our estimate of the likely maximum BCFs. The values are the smaller (but more reasonably accurate) than the BCFs based on ^{14}C or actual DNP determinations. Even this estimate is conservative as *no* intact DNP was indicated in any GC chromatogram. Further, our degradation data suggest minimal DNP existing in the soils long enough for plant uptake. Plant metabolism of even the small amounts of DNP accumulated could also occur.

All of the BCFs (Table 4) are low, suggesting minimal plant contamination with DNP. Contamination is minor regardless of sludge treatment, at DNP concentrations an order of magnitude (or more) higher than expected under normal conditions. Calcareous soils more highly polluted (>20 mg/kg) with DNP would likely result in plant phytotoxicities (Shea et al., 1983). If the soils are acid, phytotoxicities occur at lower DNP concentrations (Simon, 1953).

Thus, even if a municipal sewage sludge highly contaminated with DNP was identified (an unlikely occurrence), concerns over possible plant contamination should not limit sludge application to calcareous soils at agronomic rates. Degradation of DNP in these soils would minimize the amount of chemical available for plant uptake. Careful water management for the first 30 d or so following DNP additions to high pH soils would also minimize the amount of chemical available for leaching to groundwater.

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REFERENCES

Aranda, J.M., G.A. O'Connor, and G.A. Eiceman. 1989. Effects of sewage sludge on diethylhexyl phthalate uptake by plants. J. Environ. Qual. 18:45–50.

Bellin, C.A., G.A. O'Connor, and Jin Yan. 1990. Sorption and degradation of pentachlorophenol in sludge-amended soils. J. Envi-

ron. Qual. 19:603–608 (this issue).
Berlin, J., W. Barz, H. Harms, and K. Harder. 1971. Degradation of phenolic compounds in cell cultures. FEBS Lett. 16:141–146.
Gundersen, K., and H.L. Jensen. 1956. A soil bacterium decomposing organic nitrocompounds. Acta Agric. Scand. 6:100–114.
Jacobs, L.W., G.A. O'Connor, M.R. Overcash, M.J. Zabek, and P. Rygwiecz. 1987. Effect of trace organics in sewage sludges on soil-plant systems and assessing their risk to humans. p. 101–143. *In* A.L. Page et al. (ed.) Land application of sludge. Lewis Publ., Chelsea, MI.
Klepper, L.A., 1979. Effects of certain herbicides and their combination on nitrate and nitrite reduction. Plant Physiol. 64:273–275.
Miller, R.H. 1977. The soil as a biological filter. p. 70. *In* S. Sopper and L.T. Kardos (ed.) Recycling treated municipal wastewater and sludge through forest and cropland. The Penn. State Univ. Press, University Park, PA.
O'Connor, G.A., and J.U. Anderson. 1974. Soil factors affecting the adsorption of 2,4,5-T. Soil Sci. Soc. Am. Proc. 38:433–436.
O'Connor, G.A., B.C. Fairbanks, and E.A. Doyle. 1981. Effects of sewage sludge on phenoxy herbicide adsorption and degradation in soils. J. Environ. Qual. 10:510–515.
Overcash, M.R., J.B. Weber, and M.L. Miles. 1982. Behavior of organic priority pollutants in the terrestrial system: di-N-butyl phthalate ester, toluene, and 2,4-dinitrophenol. Rep. 171. Water Resources Res. Inst., Raleigh, NC.
Ryan, J.A., R.M. Bell, J.M. Davidson, and G.A. O'Connor. 1988. Plant uptake of non-ionic organic chemicals from soils. Chemosphere 17:2299–2323.
Shea, P.J., J. Weber, and M.R. Overcash. 1983. Biological activities of 2,4-dinitrophenol in plant-soil systems. Residue Rev. 87:2–41.
Simon, E.W. 1953. Mechanisms of dinitrophenol toxicity. Biol. Rev. 28:453–479.
U.S. Environmental Protection Agency. 1979. Water related environmental fate of 129 priority pollutants. Vol. 1 and 2. EPA-440/4-79-0296. NTIS, Springfield, VA.

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PLANT UPTAKE OF NON-IONIC ORGANIC CHEMICALS FROM SOILS

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ABSTRACT

Methodologies utilizing simple properties of chemicals - half-life ($T_{1/2}$), log octanol-water partition coefficient (log K_{ow}) and Henery's Law constant (Hc) - are developed to screen organic chemicals for potential plant uptake.

INTRODUCTION

Early in 1983, the American Chemical Society's Chemical Abstract Service registered its 6,000,000th chemical. The Toxic Substance Control Act Inventory list 63,000 chemical substances whose manufacture, processing and ultimate use for commercial purposes has occurred in the United States since January, 1975 (TSCA Inventory, USEPA, 1985). Additionally, the number of synthetic organic chemicals used and disposed of by society is increasing at a rate of about 1000 new chemicals per year, (Loehr and Malina, 1986). This endless supply of compounds together with the variety of reactions they can undergo in the environment makes describing their environmental impact exceptionally challenging.

Of the possible locations for the disposal of wastes - surface waters, atmosphere or land the latter represents a common location for waste disposal as well as an opportunity to manage wastes with minimal environmental impact. The object of the land disposal practice is to degrade, immobilize, and/or transform the wastes into beneficial, or at least non detrimental constituents. There are over 200 industrial waste land treatment sites in the United States, and a larger number of land treatment sites for municipal wastewater and sludge (Loehr and Malina, 1986). Land disposal of wastes has increased during the past decade and is projected to continue to increase in the future (Loehr and Malina, 1986).

The study of organic chemicals in the soil environment has been dominated by agricultural chemicals (e.g., insecticides, nematicides and herbicides) and specific compounds that persist in the soil (e.g., PCB's, PBB's etc.). This narrow perspective probably occurred because of the prevalence of agricultural chemicals in soil, complexity of reactions, large number of compounds, and cost associated with organic analysis. Specific compound attention has been propagated by the formation of lists of specific compounds, such as the organic priority pollutant list of 1976. Even with this narrowing of focus, the cost associated with a chemical by chemical investigation is prohibitive. The approach therefore

has been to utilize physicochemical parameters, or to group compounds on the basis of their chemical or physical properties and study selected compounds from each group. Clearly, we must insure that the grouping of compounds is correct and that the factors used in the groupings predict the behavior and impact of compounds not studied.

The following attempts to provide a framework which uses physicochemical parameters to evaluate potential plant uptake of neutral or weakly ionized organic chemicals from soil. The procedure does not predict plant concentration of organics in a field situation, but provides a procedure for grouping chemicals by their relative potential for plant uptake. As such, it should allow compound screening for their likelihood for plant uptake and, therefore, justify experimental evaluation as well as identify chemicals of low concern where testing may be counterproductive. It should also reveal where information is needed to confirm the screening model.

BEHAVIOR OF ORGANIC CHEMICALS

Many processes impact organic chemicals in the soil environment. The sum of these actions determine the compounds environmental impact (Figure 1). Factors such as pH, CEC, OM content, clay content and soil water content all impact the rate and extent of these processes (Goring and Hamaker, 1972). In a given situation (soil and environmental conditions) however, the processes are dependant upon the physical and chemical properties of the chemical. The characteristics of a chemical that determine its distribution between vapor, solid, liquid and adsorbed phases in the soil, and its degradation rate become the characteristics that determine its environmental fate and impact upon plants. These processes determine not only the form of the compound that is present, but also the speed at which the compound moves or spreads through the soil and atmosphere to achieve its impact. The importance of each of these processes will be discussed separately.

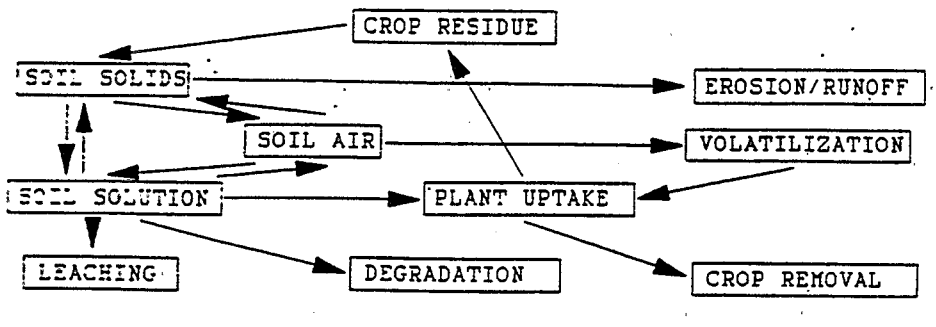

FIGURE 1 SOIL TRANSFORMATIONS

Degradation

Plant uptake of most chemicals is concentration dependent, therefore a compound's persistence can alter its ultimate fate and environmental impact. An assessment of the half-life of a particular compound is a relatively simple way of limiting the number of soil borne organic compounds that need to be considered as likely to impact a plant grown in contaminated soil. The concentration of synthetic organic compounds in the soil decrease with time, providing no further additions occur. Processes contributing to the decrease with time are biological and/or chemical degradation. These processes have been shown to be dependent on soil and environmental factors (ie., temperature, water content, soil pH, and organic C), (Hamaker, 1972). Without the quantitative information necessary to describe the functional dependence of degradation on these factors, it has been shown that degradation of a specific organic chemical can be described by a first order rate constant, μ, (Nash, 1980; Rao and Davidson, 1980; Jury et al., 1983; Gillett,1983). This parameter is usually measured by determining the fraction of an applied chemical remaining after a time t according to Equation 1 :

$$M(t) = M(0) \exp^{(-\mu t)} \qquad [1]$$

where M(t) is the quantity of the compound remaining in the soil at time t. The half-life, $T_{1/2}$, of a compound is defined as the time required for one half of the concentration of the chemical at any point in time to be lost from the soil. This is related to the rate constant (μ) by :

$$T_{1/2} = \frac{0.693}{\mu} \qquad [2]$$

Half-lives of many chemicals have been published (USEPA, 1979; Jury et al., 1983; Smith and Dragun, 1984). Unfortunately, reported values of μ may vary enormously because measured half-lives of compounds in the soil do not always reflect degradation. Often losses include other pathways (i.e., volatilization, leaching, etc.). Additionally, water content, microbial population, and temperature can significantly influence the rate of loss thus, a chemicals life may vary from soil to soil. Half-lives are reported in Table 1 from data in USEPA, 1979. Compounds are distinguish from one another on the basis of half-life in the soil: less than 10 days, (Class A); between 10 and 50 days, (Class B); and greater than 50 days, (Class C). Gillett considered compounds of $T_{1/2}$ greater than 14 days of sufficient stability to be of concern (Gillett, 1983). The impact of chemical half-lives on concentration of a pollutant in the soil over time is shown in Figure 2. Pollutants with half-lives of less than 10 days, for example, are reduced to less than 0.10% of their original concentration after 100 days in the soil. In contrast, pollutants with half-lives of greater than 50 days are still present at >25% of their original concentrations after 100 days. Their impact, and relative potential for plant uptake, are much more pronounced than that for

compounds with half lives of less than 10 days.

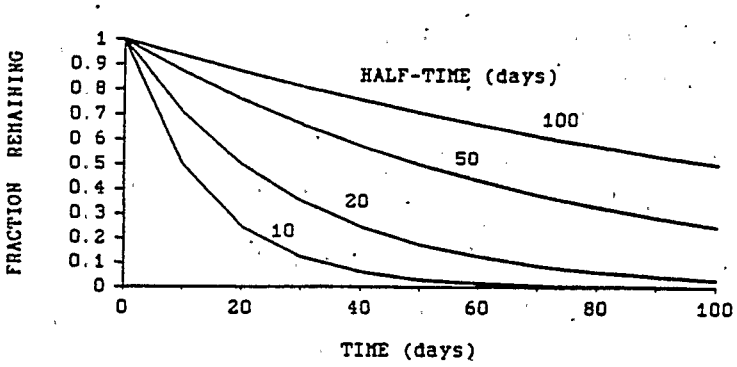

FIGURE 2 EFFECT OF CHEMICAL HALF LIFE AND TIME ON FRACTION REMAINING

The average concentration present during the plant growing period can be calculated by integration of Equation 1 between the limits 0 and t (growth period) and dividing by t. Assuming a growth period (i.e. 50 or 100 days) the effect of half-life on the average soil concentration as a fraction of the amount originally applied illustrates that the limits for classification of compounds based on half-lives are arbitrary (Figure 3). The length of exposure (i.e. plant growth period) and relative average exposure must be specified before compounds can be classified by their half-lives. For example, our use of 10 and 50 day half lives as classification end points was based on a 100 day growth period and relative average exposures of 0.15 and 0.5. Using the same half-life end points but a 50 day growth period means relative average exposures of 0.3 and 0.7.

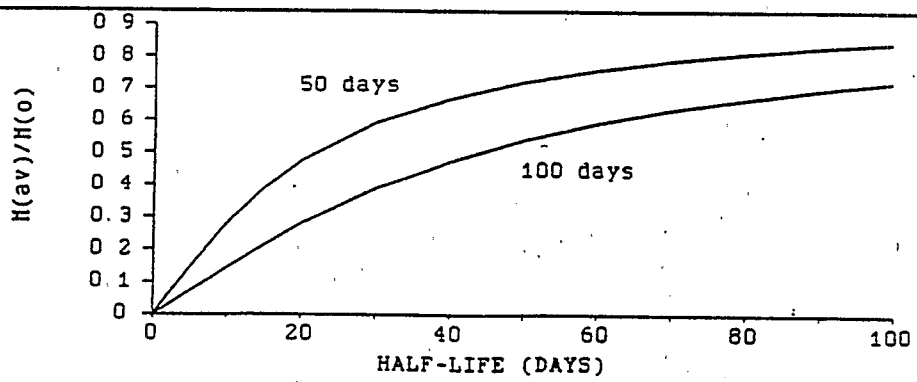

FIGURE 3 AVERAGE SOIL CONCENTRATION VS HALF LIFE FOR 50 AND 100 DAYS OF GROWTH

TABLE 1. Log K_{ow}, Half-life and Hc for the Priority Pollutants

Compound	log K_{ow}	$T_{1/2}$	Hc	Compound	log K_{ow}	$T_{1/2}$	Hc
PESTICIDES							
20.Acrolein	-0.09	B	2.8E-03	21.Aldrin	4.62	C	4.9E-04
22.Chlordane	4.3	C	3.9E-03	23.DDD	5.99	C	0.5E+00
24.DDE	5.69	B	9.0E-04	25.DDT	5.98	C	2.0E-03
26.Dieldrin	2.9	C	3.0E-04	27.Endosulfan	3.55	C	nd
28.Endrin	4.5	C	1.7E-05	29.Heptachlor	3.9	A	6.2E-02
30.Heptachlor epoxide	3.9	C	3.2E-05	31.Hexachlorocyclohexane	3.8	B	3.0E-04
32.Lindane	3.72	C	6.0E-04	33.Isophorone	1.70	nd	nd
34.TCDD	6.14	C	nd	35.Toxaphene	3.85	C	2.1E-01
POLYCHLORINATED BIPHENYLS							
36a.Arochlor 1016	4.38	C	8.6E-01	36b.Arochlor 1221	4.09	C	1.3E-02
36c.Arochlor 1232	4.54	C	2.1E+00	36d.Arochlor 1242	4.11	C	5.5E-02
36e.Arochlor 1248	5.6	C	1.1E-01	36f.Arochlor 1254	6.06	C	1.1E-01
36g.Arochlor 1260	6.11	C	2.9E-01	37.2-chloronaphthalene	4.12	C	1.3E-02
HALOGENATED ALIPHATIC HYDROCARBONS							
38.Chloromethane	0.91	C	1.6E+01	39.Dichloromethane	1.25	B	8.5E-02
40.Trichloromethane	1.9	B	1.2E-01	41.Tetrachloromethane	2.64	nd	9.6E-01
42.Chloroethane	1.54	B	6.1E-01	43.1,1-dichloroethane	1.79	B	1.7E-01
44.1,2-dichloroethane	1.48	B	3.8E-02	45.1,1,1-trichloroethane	2.17	nd	1.3E+00
46.1,1,2-trichloroethane	2.17	nd	3.1E-01	47.1,1,2,2-tetrachloroethane	2.56	A	1.6E-02
48.Hexachloroethane	4.62	nd	4.1E-01	49.Chloroethene	0.60	A	6.2E-01
50.1,1-dichloroethene	1.48	A	1.8E-01	51.1,2-trans-dichloroethene	1.48	A	2.7E-01
52.Trichloroethene	2.29	A	3.8E-01	53.Tetrachloroethene	2.88	A	6.4E-01
54.1,2-dichloropropane	2.28	nd	1.2E-01	55.1,3-dichloropropene	1.98	A	5.6E-02
56.Hexachlorobutadiene	3.74	C	4.3E-01	57.Hexachlorocyclopentadiene	3.99	A	1.5E+00
58.Bromomethane	1.10	B	4.4E+00	59.Bromodichloromethane	1.88	nd	nd
60.Dibromochloromethane	2.09	nd	nd	61.Tribromomethane	2.30	nd	2.4E-02
62.Dichlorodifluoromethane	2.16	C	6.3E+01	63.Trichlorofluoromethane	2.53	nd	2.4E+00
HALOGENATED ETHERS							
64.Bis(chloromethyl)ether	-0.38	A	8.6E-02	65.Bis(2-chloroethyl)ether	1.58	nd	4.7E-05
66.Bis(2-chloroisopropyl)ether	2.58	nd	4.7E-02	67.2-chloroethyl vinyl ether	1.28	A	1.0E-02
68.4-chlorophenyl phenylether	4.08	nd	1.0E-02	69.4-bromophenyl phenyl ether	4.28	nd	nd
70.Bis(2-chloroethoxy)methane	1.26	C	1.1E-05				
MONOCYCLIC AROMATICS							
71.Benzene	2.13	A	2.3E-01	72.Chlorobenzene	2.84	nd	1.5E-01
73.1,2-dichlorobenzene	3.38	nd	1.5E-01	74.1,3-dichlorobenzene	3.55	nd	1.1E-01
75.1,4-dichlorobenzene	3.55	nd	9.9E-02	76.1,2,4-trichlorobenzene	4.26	A	9.6E-01
77.Hexachlorobenzene	6.18	C	7.0E-02	78.Ethylbenzene	3.15	A	2.6E-01
79.Nitrobenzene	1.85	C	5.4E-06	80.Toluene	2.69	A	2.8E-01
81.2,4-dinitrotoluene	2.01	nd	1.3E-02	82.2,6-dinitrotoluene	2.05	nd	1.3E-02
83.Phenol	1.46	A	1.9E-05	84.2-chlorophenol	2.17	nd	1.9E-04
85.2,4-dichlorophenol	3.3	A	2.3E-04	86.2,4,6-trichlorophenol	3.61	nd	1.7E-04
87.Pentachlorophenol	5.01	A	1.2E-04	88.2-nitrophenol	1.76	C	3.2E-04
89.4-nitrophenol	1.91	B	2.6E-04	90.2,4-dinitrophenol	1.53	C	2.7E-08
91.2,4-dimethylphenol	2.50	nd	7.5E-04	92.p-chloro-m-cresol	3.11	nd	1.0E-04
93.4,6-dinitro-o-cresol	2.85	nd	nd				
PHTHALATE ESTERS							
94a.Dimethyl phthalate esters	2.12	B	8.8E-06	94b.Diethyl	3.22	B	1.9E-03
94c.Di-n-butyl	5.20	B	1.2E-05	94d.Di-n-octyl	9.20	B	1.2E-05
94e.Bis(2-ethylhexyl)	8.73	B	1.2E-05	94f.Butyl benzyl	5.80	B	4.2E-05
POLYCYCLIC AROMATIC HYDROCARBONS							
95a.Acenaphthene	4.13	C	1.0E-02	95b.Acenaphthylene	4.07	C	4.8E-03
95c.Fluorene	4.18	C	4.8E-03	95d.Naphthalene	3.37	C	2.0E-02
96a.Anthracene	4.45	C	1.1E-02	96b.Fluoranthene	5.33	C	4.0E-06
96c.Phenanthrene	4.46	C	1.6E-03	97a.Benzo(a)anthracene	5.61	C	4.1E-05
97b.Benzo(b)fluoranthene	6.57	nd	nd	97c.Benzo(k)fluoranthene	6.84	C	nd
97d.Chrysene	5.61	C	8.8E-02	97e.Pyrene	5.32	C	2.8E-01
98a.Benzo(ghi)perylene	7.23	C	nd	98b.Benzo(a)pyrene	6.04	C	4.9E-01
98c.Dibenzo(a)anthracene	5.97	C	nd	98d.Indeno(123-cd)pyrene	7.66	C	nd
MISCELLANEOUS COMPOUNDS							
99.Dimethyl nitrosamine	0.06	nd	nd	100.Diphenyl nitrosamine	2.57	nd	nd
101.Di-n-propyl nitrosamine	1.31	nd	nd	102.Benzidine	1.81	A	nd
103.3,3-dichlorobenzidine	3.02	A	nd	104.1,2-diphenylhydrazine	3.03	nd	nd
105.Acrylonitrile	0.25	A	3.8E-03				

where log K_{ow} and half-lives (assessed as A = < 10 days, B = 10-50 days, C = > 50 days) are based on the principal fate process in the environment (USEPA,79); where Henry's Constant (dimensionless) has been obtained from the WERL Treatability Database

Adsorbed-Liquid Partition

Considerable research data exists on the equilibrium between an organic sorbed to the soil and that in the soil-water phase. For simplicity, this is often expressed as a linear sorption isotherm (Karickoff, 1981):

$$C_s = Kd\ C_l \qquad [3]$$

where C_s is the sorbed concentration (g/kg soil), C_l is the solution concentration (g/m^3 soil solution) and Kd (m^3/kg) is the slope of the sorption isotherm or distribution coefficient (Kay and Elrick, 1967). Equation 3 assumes complete reversibility and equilibrium between the two phases, which may not strictly occur for some chemicals. Di Toro and Horzempa (1982), reported that the sorptive process of 2,4,5,2',4',5'- hexachlorobiphenyl consisted of both reversible and strongly bound components. Such bound residues could not be extracted by normal analytical techniques, but could be detected by radiolabelling. Similar findings have been reported by others working with herbicides and chlorobenzenes (Khan, 1982; and Scheunert, et al., 1985) and may require the above mathematical approach for sorption be modified to account for bound residuals.

In soils and sediments, where the clay content is relatively low, pollutant sorption occurs primarily on the organic fraction of the soil, (Hamaker and Thompson, 1972; Rao and Davidson, 1980). The degree of sorption of the non ionic organic pollutant is then dependant upon the organic carbon content in the soil, or sediment. Variation between materials, which otherwise exhibit a wide range of physicochemical properties, can then be reduced by defining an organic carbon distribution coefficient (K_{oc}):

$$K_{oc} = \frac{Kd}{f_{oc}} \qquad [4]$$

where Kd is the slope of the sorption isotherm in m^3/kg, and f_{oc} is the organic carbon fraction in the soil or sediment, (Means, et al., 1982). This assumes that all organic matter has the same chemical structure.

K_{ow} is defined as the ratio of the organic chemical concentration in octanol to that in water, when an aqueous solution of the organic chemical is mixed with n-octanol and then the organic chemical allowed to partition between the two phases (Dawson, et al., 1980). There have been many investigations into the relationship between K_{oc} and K_{ow}. Briggs (1973) for example reported:

$$\log K_{oc} = 0.524 \log K_{ow} + 0.62 \qquad [5]$$

from his work with 4 agricultural soils and 30 chemicals chosen for their wide range of properties. Similar relationships, see Equation's 6, 7, 8, 9, and 10, have been reported (Means, et al., 1982; Schwarzenbach and Westall, 1981; Rao, et al., 1982; Karickhoff, 1981; and Brown and Flagg, 1981 respectively).

$$\log K_{oc} = \log K_{ow} - 0.317 \qquad [6]$$

$$\log K_{oc} = 0.721 \log K_{ow} + 0.49 \qquad [7]$$

$$\log K_{oc} = 1.029 \log K_{ow} - 0.18 \quad [8]$$

$$\log K_{oc} = 0.989 \log K_{ow} - 0.346 \quad [9]$$

$$\log K_{oc} = 0.937 \log K_{ow} - 0.006 \quad [10]$$

The relationships are surprisingly similar to one another considering they cover over 100 chemicals, as well as a large number of soils and sediments (Figure 4). Thus when the sorption value of a particular pollutant in a particular soil is not available, advantage can be taken of the relationship between the organic carbon distribution coefficient (K_{oc}) and the octanol water partition coefficient (K_{ow}) of the chemical. Recently, a nonempirical measurement (first-order molecular connective indexes) calculated from the non-hydrogen part of the molecule has been shown to predict the K_{ow} of organic compounds with great success (Sabljic, 1987). As these calculated values for various organic compounds become available it will allow for their use in place of K_{ow} or K_{oc}.

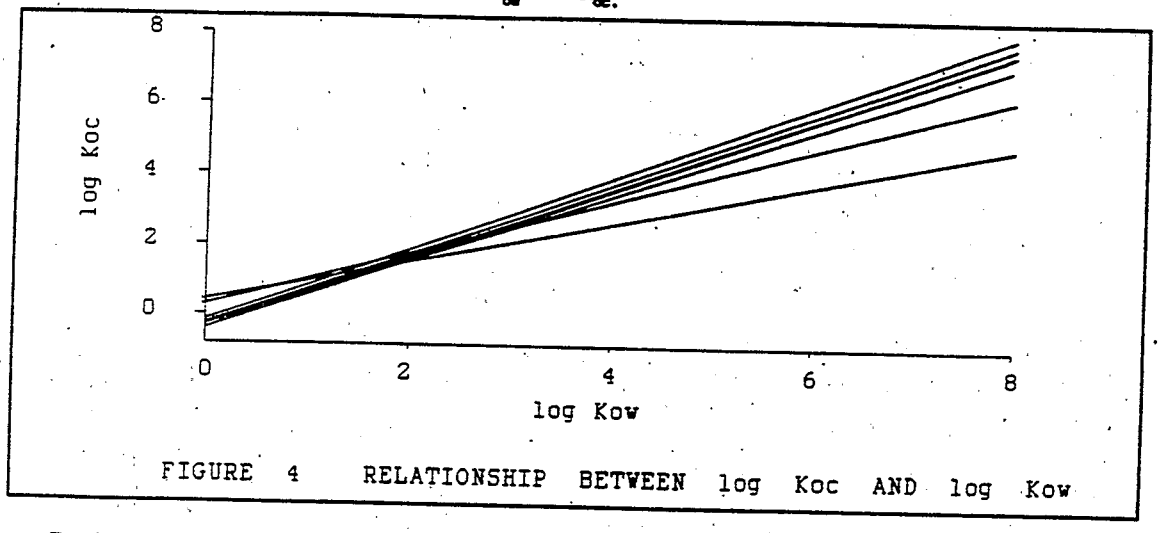

FIGURE 4 RELATIONSHIP BETWEEN log Koc AND log Kow

To have greatest impact upon plant uptake, the organic compound must stay within the vicinity of the plant root, and not be quickly leached away by mass flow. For example most residual soil-acting herbicides have Kd values in the range of 1-20 with values up to 40 being satisfactory for most soil applications (Graham-Bryce, 1984). Compounds with Kd's of greater than 1000 become inactivated by soil sorption (Graham-Bryce, 1984). Based on Equation 4 and Equation 9 for a soil with f_{oc} = 0.0125 (OM ≈ 2%) Kd's of 1, 20, 40, and 1000 would represent log K_{ow}'s of 2.3, 3.6, 3.9, and 5.3, respectively.

Liquid-Vapor Partition

Vapor phase partitioning of a compound in the soil influences the spread of the compound through the soil. Even for chemicals with relatively low vapor pressure, this transport route has been shown to be significant (Mayer,et al., 1974). Those chemicals that have a high vapor pressure may easily move from the soil solution into the soil air phase, where they can move throughout the soil

and across the soil surface. The vapor-phase may be taken up by the plant either through roots or by above ground portions of the plant.

The compartmentalization of the compound between the soil solution and the air spaces in the soil is frequently described by Henry's Law (Jury et al., 1983) with the extent of partitioning described by Henry's Constant (Hc). This can be calculated as:

$$\text{Henry's Constant (Hc)} = \frac{16.04 P M}{T S} \qquad [11]$$

where P = vapor pressure of pure solute in mm/Hg,
M = molecular weight of solute,
T = absolute temperature, and
S = solubility in water mg/L

(Thibodeaux, 1979). Henry's Constant may be expressed in different units and vary by several orders of magnitude depending upon the source of the original data. For example, estimated values for vinyl chloride of 2.3×10^{-2} to 6.39 atm m^3/mol are reported by Mackay and Shiu (1981) and Goldstein (1982), respectively. Experimentally determined Hc values are considered more reliable than calculated values. Henry's Constant, dimensionless, for the priority pollutants is provided in Table 1.

Comprehensive studies have not been conducted to determine the Hc above which volatilization plays an important role in the transport of a chemical in the atmosphere. Thus, it is not possible to select a Hc above which transport in the soil will occur primarily in the vapor phase. However, a partition between the vapor and aqueous phases of greater than 10^{-4} is normally sufficient for a chemical to be a good preemergence herbicide (Graham-Bryce, 1984). Jury et al., (1984) utilized three volatility categories with Hc values of 2.5×10^{-3}, 2.5×10^{-5} and 2.5×10^{-7}. Gillett (1983) utilized values of 10^{-3} and 6×10^{-5} in his classification. Thus, the value of 10^{-4} may be a reasonable transition point for determining when vapor diffusion becomes important. This would mean that vapor diffusion would be important for all PCB's and halogenated aliphatics and unimportant for some of the monocyclic and polycyclic aromatics and many pesticides. Soil sorption can significantly reduce chemical volatilization (Fairbanks et al., 1987) thus, the arbitrary value of 10^{-4} may overestimate the importance of volatilization in high organic carbon soils. Jury et al., (1983) used Hc and K_{ow} to calculate volatilization flux from soil.

PLANT UPTAKE OF ORGANIC CHEMICALS

Chemical uptake by plants is a complex process that may involve a compound specific active processes, and/or a passive process in which the chemical accompanies the transpiration water through the plant. If the former case dominates, a rigorous relationship between plant uptake and the chemicals

physicochemical parameters may not exist, although some general guidelines may be expected. If uptake into the plant is a passive process, rigorous relationships should exist.

It is generally accepted that there are four main pathways by which a chemical in the soil can enter a plant (Topp et al., 1986). These are:

1. root uptake and subsequent translocation by the transpiration stream,
2. vegetative uptake of vapor from the surrounding air,
3. uptake by external contamination of shoots by soil and dust, followed by retention in the cuticle or penetration through it, and
4. uptake and transport in oil cells which are found in oil containing plants like carrots and cress.

The amount of an organic chemical found in a plant will be the sum total of each of these transport routes minus metabolic losses. Their respective importance will depend upon the nature of the organic chemical, the nature of the soil, and the environmental conditions under which plant exposure occurs. Pathways 3 & 4 are significant only in specific situations. Thus, for the purpose of describing the general case of plant uptake, they can be discounted as major routes of plant contamination. Most reported instances of plant uptake of soil-borne organic compounds make no attempt to discriminate between pathways 1 & 2. Therefore, the relative importance of each pathway, under different environmental conditions, has not been assessed at present.

Root Uptake And Translocation

Shone and Wood (1972) investigated the absorption and translocation of the herbicide simazine by 6-day-old barley plants in solution cultures. The experiments were either 24- or 48-hour experiments conducted under different conditions of humidity, light intensity, temperature, and levels of metabolic inhibitors. The relationship between simazine transport and water uptake was described by a transpiration stream concentration factor (TSCF), defined as:

$$\text{TSCF} = \frac{\mu g \text{ simazine in shoots per mL water transpired}}{\mu g \text{ simazine per mL of external solution}}$$

They found that water was taken up preferentially to simazine, because the TSCF was always less than unity, i.e., the concentration of simazine in the plant shoots per mL of water transpired never reached that in the external solution. There was no evidence of loss of or breakdown of the parent compound during the experiment. The concentration of simazine in the plant roots, on a fresh weight basis, however, reached a value greater than unity as a result of physical sorption of the herbicide to the root tissue.

Evaluation of other triazines led to the conclusion that plant uptake was, in general, a passive process because TSCF was less than unity, (Shone et al.,1973).

Plant uptake of 6 herbicides and a fungicide showed that TSCF was independent of concentration and less than unity for all except 2,4-D at pH 4.0 (Shone and Wood, 1974). In the case of 2,4-D at pH 4.0, plant uptake was metabolically influenced. Briggs et al.,(1982) evaluated plant uptake of 18 chemicals and found that the TSCF was less than unity for all chemicals studied. They related the TSCF to the octanol/water partition coefficient (K_{ow}) for the chemicals and found a bell shaped relationship between TSCF and K_{ow}, with a broad maximum around a K_{ow} of 1.8. A Gaussin curve (Figure 5) was fitted to the data such that:

$$TSCF = 0.784e^{-[(\log K_{ow} - 1.78)^2/2.44]} \qquad [12]$$

The authors suggested that at K_{ow} values below 1.8, translocation is limited by the lipid membranes in the root. At K_{ow} values above 1.8, translocation is limited by the rate of transport of the lipophilic chemical from the plant root to the top of the plant. All the TSCF values were below unity, suggesting passive chemical movement into the shoot with the transpiration stream. There was no evidence that chemicals were taken up against a concentration gradient.

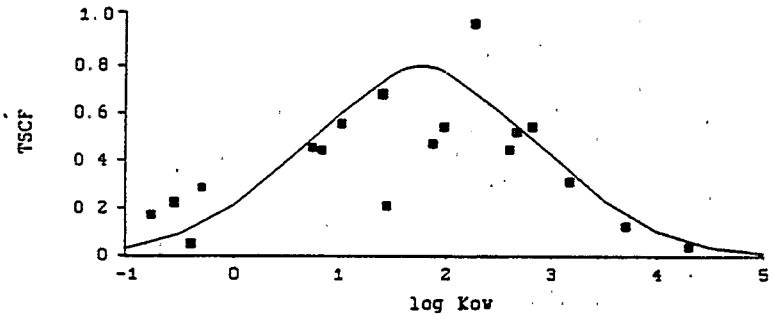

FIGURE 5 RELATIONSHIP BETWEEN log Kow AND TRANSPIRATION STREAM CONCENTRATION FACTOR
Adapted from Briggs et al 1982

Shone and Wood (1974) proposed that the uptake of a chemical into a plant root could be described by a root concentration factor (RCF), defined as:

$$RCF = \frac{\text{concentration in root, (}\mu g/g \text{ fresh wt.)}}{\text{concentration in external solution, (}\mu g/mL)}$$

Using radiolabelled herbicides in solution culture with barley seedlings, they showed that the quantity of the herbicide transported to the plant stems (TSCF) could not be inferred from the concentration in the plant roots (RCF). In addition, although the RCF of some of the tested herbicides exceeded unity, uptake was not affected by temperature. This, suggests the compounds were retained by physical sorption rather than biochemically.

When barley seedlings were transferred from the herbicide amended solution culture to a herbicide free solution, RCF decreased before TSCF was affected by

the change (Shone et al., 1974). Thus lipophilic herbicides appear to penetrate the cortical cells of the root whereas the lipophobic herbicides are largely confined to the free cell space in the root.

Briggs et al.,(1982) found that RCF was related to K_{ow}. Starting with a value of less than unity for polar compounds, RCF increased with increasing K_{ow}. Sorption of chemicals by macerated roots was very closely related to the RCF of living roots, for the more lipophilic chemicals. In contrast, the RCF of macerated roots continued to decrease as the lipophilicity decreased (Figure 6). There was a linear relationship between the log concentration factor of the macerated roots and log K_{ow}:

$$\log RCF_{(\text{macerated root})} = 0.77 \log K_{ow} - 1.52 \qquad [13]$$

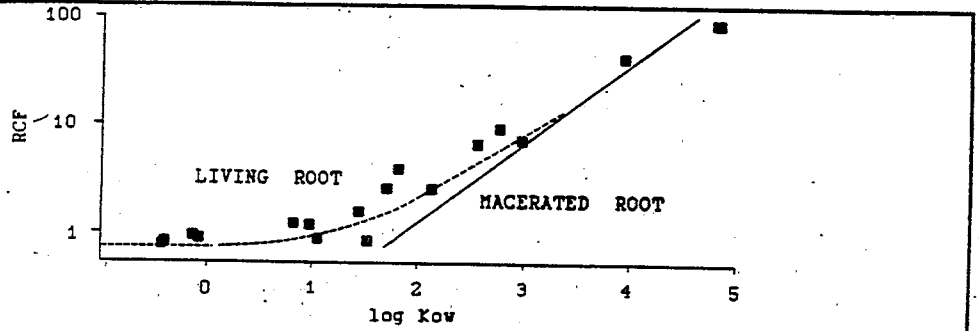

FIGURE 6 EFFECT OF TISSUE STATUS ON THE RELATIONSHIP BETWEEN log Kow AND ROOT CONCENTRATION FACTOR
Adapted from Briggs et al. 1982

Assuming that RCF of living roots could be explained by two processes: (1) a partitioning of the organic chemical between the lipophilic root tissue and external solution culture and (2) a fraction of root that is aqueous and equal in concentration to external solution phase (constant for all compounds, 0.82). Briggs et al., (1983) suggested that sorption of chemicals by the root is a partitioning described by:

$$\log(RCF - 0.82) = 0.77 \log K_{ow} - 1.52 \qquad [14]$$

They proposed an analogous stem concentration factor (SCF):

$$SCF = \frac{\text{concentration in stem } (\mu g/g \text{ fresh wt.})}{\text{concentration in external solution } (\mu g/mL)}$$

Macerated stems sorption of organic compounds was also related to the K_{ow} of the compound:

$$\log SCF_{(\text{macerated stem})} = 0.95 \log K_{ow} - 2.05 \qquad [15]$$

Assuming that the contribution of the aqueous phase in the stem was similar to that in roots (0.82), the partition between the stem and xylem stream is:

$$\log(K_{(\text{stem/xylem sap})} - 0.82) = 0.95 \log K_{ow} - 2.05 \qquad [16]$$

The SCF is then given by the $K_{(stem/xylem\ sap)}$ partition coefficient multiplied by the partition of the external solution present in the xylem sap (TSCF):

$$SCF = [10^{(0.95 \log K_{ow} - 2.05)} + 0.82]^* \quad [17]$$
$$[(0.784)10^{-0.434[(\log K_{ow} - 1.78)^2/2.44]}]$$

For 15 chemicals ($\log K_{ow}$ from -0.57 to 3.7), the experimental points fit the predicted line quite well (Figure 7). The shift in log K_{ow} where TSCF reaches a maximum (1.8) to where SCF reaches a maximum (4.5) arises because sorption of the more lipophilic compounds by the stem tissue increases faster than the TSCF decreases. The predicted decline in SCF for compounds of log K_{ow} > 4.5 was not tested.

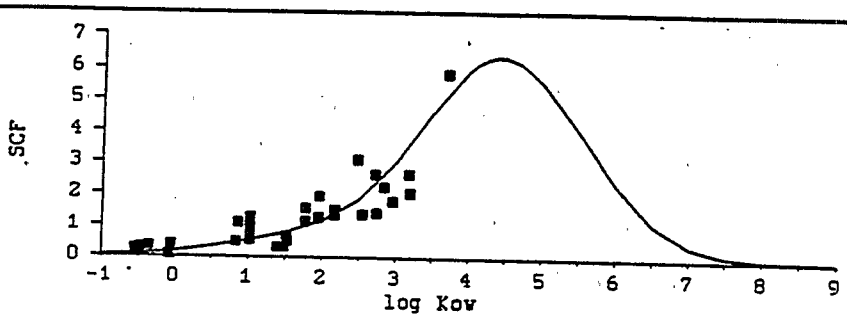

FIGURE 7 RELATIONSHIP BETWEEN log Kow AND PLANT CONCENTRATION FACTOR
Adapted from Briggs et al 1983

There have been other attempts to relate plant uptake and translocation of an organic chemical to either the physical or chemical properties of the chemical. Topp et al., (1986) reported the relationship:

$$\log RCF = 0.63 \log K_{ow} - 0.959 \quad [18]$$

following their exposure of barley seedlings for 7-days to various chemicals in water culture.

The concentration factor (CF) concept is a useful way of describing the relative concentration of an organic chemical in a particular plant part. It has many limitations, however. These arise because the concentration of organic chemicals, both within the soil or nutrient solution and within the plant part do not remain constant with time. Chemicals in the soil, or in nutrient solution, may be depleted by plant uptake or degradation; chemicals in a plant may also be reduced with time by degradation within the plant, or by increases in plant mass effectively diluting the chemical. Changes in uptake as measured by the CF, have been reported, Figure 8 (Topp et al., 1986). Different CF's arise depending upon the timing of the actual sampling. Further it seems logical that the CF would depend upon soil concentration, initial vs soil concentration at time of plant

sampling. Further research on this topic is needed to define the effect of time of sampling (both plant and soil) on CF's so different experiments can be compared.

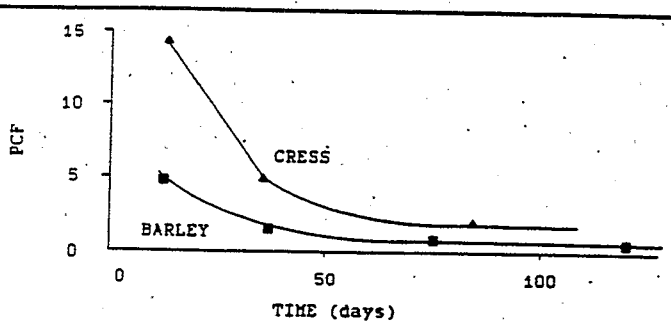

FIGURE 8 EFFECT OF PLANT TYPE AND LENGTH OF GROWTH PEROID ON THE PLANT CONCENTRATION FACTOR
Adapted from Topp et al 1986

The work of Shone, Briggs, and their co-workers reported above was carried out in nutrient solution cultures where sorption and desorption effects of soil organic matter were absent. The application of their results to plant uptake from field soils requires that soil sorption be considered. The effect of soil sorption on soil solution concentration can be mathematically described using the following relationship:

$$\delta C_T = \delta C_s + \theta C_L \qquad [19]$$

where C_T is the total organic chemical concentration in the soil ($\mu g/g$), δ is the soil bulk density (g/cm^3), C_s is the adsorbed chemical concentration ($\mu g/g$), θ is the soil-water content by volume (mL/cm^3), and C_L is the chemical concentration in the soil-water phase ($\mu g/mL$). Using the linear equilibrium relationship in Equation 3 and 4 allows Equation 19 to be rewritten in terms of C_L such that:

$$\frac{C_L}{C_T} = \frac{\delta}{\delta K_{oc} f_{oc} + \theta} \qquad [20]$$

It is now possible to combine equations relating soil sorption and soil solution concentration and calculate RCF, TSCF, and SCF for different chemicals on a total soil concentration basis. Substituting Equation 20 into Equation 17 where C_L is the external solution and:

$$SCF_{(SOIL)} = \frac{\text{concentration in stem}}{\text{concentration in soil}}$$

gives:

$$SCF_{(SOIL)} = \frac{\delta}{\delta K_{ow} f_{oc} + \theta} \left([10^{(0.95 \log K_{ow} - 2.05)} + 0.82] * [(0.784) 10^{-0.434[(\log K_{ow} - 1.78)^2 / 2.44]}] \right) \qquad [21]$$

For nutrient solutions this equation reduces to Eq [17] when $f_{oc} = 0$, $\theta = 1$, and $\delta = 1$. Inclusion of soil sorption into the SCF from Briggs et al.,(1983)

alters the relationship between SCF and log K_{ow} such that the log K_{ow} where plant adsorption is a maximum decreases from 4.5 for nutrient solution to 1 for soils. (Figure 9). The decrease in SCF for chemicals with log K_{ow} greater than 1 is supported by the published literature on plant uptake in soil systems (Travis and Arms, 1988).

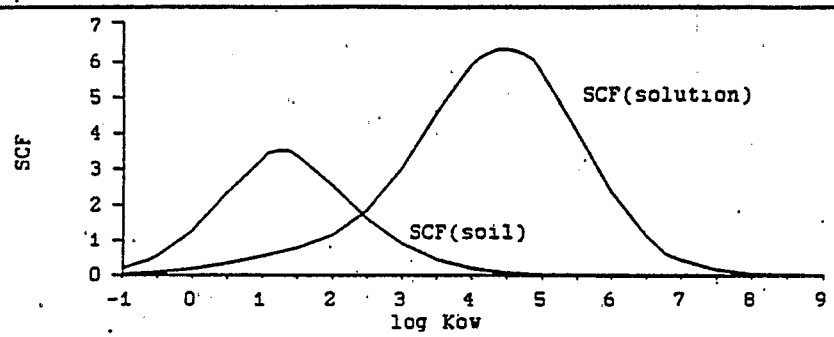

FIGURE 9 EFFECT OF SOIL ON THE RELATIONSHIP BETWEEN log Kow AND STEM CONCENTRATION FACTOR

Equation 21 implies that plant uptake is related to soil organic matter content (Figure 10).Differences in the plant uptake of an organic chemical in soils with different organic carbon contents has been shown experimentally. Lichtenstein et al., (1967) for example, showed higher concentrations of the pesticide aldrin in roots of peas when grown in aldrin- polluted quartz sand compared to a loam soil containing approximately the same total concentration of the pollutant.

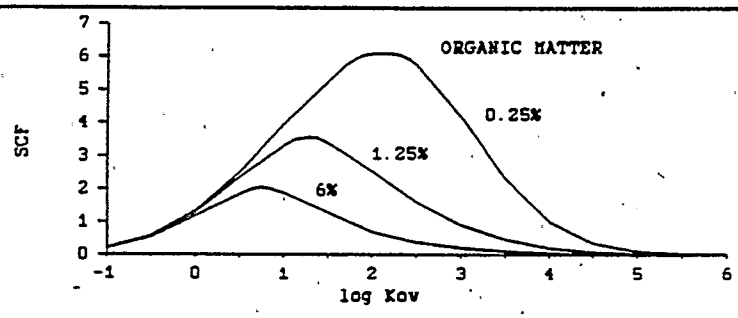

FIGURE 10 EFFECT OF SOIL ORGANIC MATTER ON THE RELATIONSHIP BETWEEN log Kov AND STEM CONCENTRATION FACTOR

It is also apparent from Equation 21 that increases in soil water content reduce SCF (Figure 11). However, for a soil with a f_{oc} of 0.0075 (1.25% organic matter), changes in soil water content over the range 0.1 to 0.5 mL/cm^3 altered SCF less than 10% for chemicals with a K_{ow} greater than 2.5. The fraction in solution, $(\theta C_l/C_T)$, increases as soil water content increases even though the organic chemical concentration (C_l) in the soil solution-phase decreases.

Therefore, if plant transpiration were increased by increasing soil water content, plant concentration could be increased. Walker, (1971) found that the phytotoxicities of the pesticides atrazine, simazine, linuron, lenacil, and aziprotryne were increased as the moisture content of the soil increased. He related the effect to differences in the quantities of the pesticides that were accumulated by the plants, with the degree of accumulation being directly proportional to water uptake.

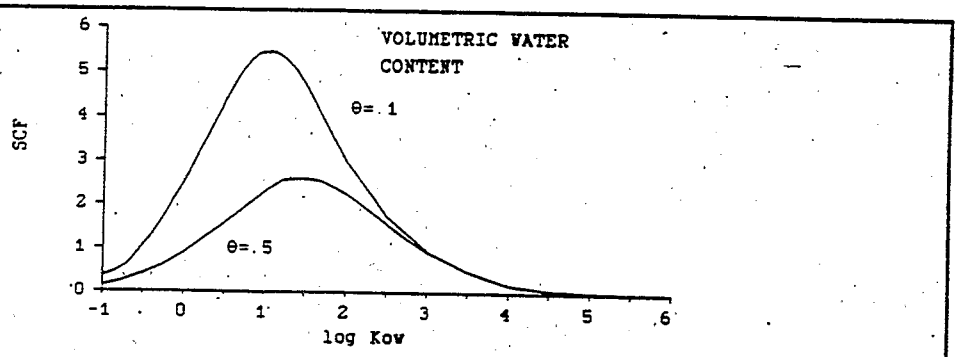

FIGURE 11 EFFECT OF SOIL WATER CONTENT ON THE RELATIONSHIP BETWEEN log Kow AND STEM CONCENTRATION FACTOR

In conclusion, assuming degradation of the organic chemical does not occur within the plant, and plant root uptake and translocation of organic chemicals from the soil is passive, plant uptake can be described as a series of consecutive partitions reactions. Partitioning occurs between soil solids and soil water, soil water and plant roots, plant roots and transpiration stream, and transpiration stream and plant stem. This partitioning can be related to the K_{ow} of organic compounds such that pollutants with high log K_{ow} values, (eg. TCDD (6.14), PCB's (4.12-6.11), some of the phthalate esters (above 5.2) and the polycyclic aromatic hydrocarbons (4.07-7.66)) are most likely to be sorbed by the soil and/or plant root. Chemicals with lower K_{ow} values are likely to be translocated within the plant and may reach significant concentrations within the above ground portions of the plant.

Vapor Phase Uptake

For volatile compounds, diffusion in the vapor phase and subsequent uptake by the root and/or shoot may be an important route of chemical entry into plants (Parker, 1966, and Prendeville, 1968). Two processes precede the penetration of chemicals in the soil into plant tissue via the air: 1) volatilization of the chemical from the soil and 2) deposition from the air onto the plant surface. Soil volatilization depends upon the vapor pressure of the compound which varies according to ambient temperatures, water solubility of the compound, and sorption capacity and physical properties of the soil.

Increasing the soil-water content of a soil will increase the potential for volatilization loss of a chemical (Guenzi and Beard, 1970). Harris and Lichtenstein (1961) showed that the rate of volatilization of aldrin from soil increased with aldrin concentration, soil moisture, relative humidity, temperature and the rate of air movement. Chemical concentration effects cease when the concentration reaches that required to give a maximum saturation vapor density equivalent to that of the pure compound. For dieldrin in a Gila silt loam soil this concentration was 25 ppm (Farmer et al., 1972). These authors also report that under similar environmental conditions the rate of volatilization was lindane > dieldrin > DDT, which is the same order for increasing vapor pressures. Jury et al., (1983 and 1984) developed a behavior assessment model that separates compounds into volatilization categories based on Henry's constants.

There have been few investigations aimed at separating root uptake and translocation of a chemical from vapor phase uptake into plant shoots. In an experiment designed to discriminate these effects, Beall and Nash (1971) found soybean shoots were contaminated by soil applied dieldrin, endrin and heptachlor largely via root uptake and subsequent translocation. Vapor phase foliar sorption however dominated for DDT and was nearly 7 times greater than root sorption and translocation. Foliar contamination from vapor sorption of residues from all four insecticides was similar (about 6.5 ppm plant dry weight), whereas contamination from root sorption and translocation varied from 38 ppm to 1 ppm depending upon the compound.

Using similar experimental techniques, Fries and Marrow (1981) found that PCBs reached the shoots of plants via the vapor phase rather than from root uptake, although the importance of this route for PCB contamination of plants remains inconclusive.

Topp et al., (1986) investigated the uptake of 16 organic chemicals by barley seedlings. Foliar uptake was related to the amount of chemical volatilized from the soil surface. The relationship (Figure 12) after 7 days exposure was:

$$FU = 46.11 + 28.95 \log VOL \qquad [22]$$

where FU was foliar uptake as percent of total ^{14}C uptake, and VOL was the organic ^{14}C trapped from the air plus that sublimated on the walls of the exposure chamber as percent of the total ^{14}C applied (Note that in the original publication the sign in front of log VOL is negative, this is assumed to be a typographical error). Four compounds (benzene, pentachlorophenol, diethylhexylphthalate, and the phenylenediamine pigment) did not fit the calculated line because they were nonpersistent and taken up after mineralization to $^{14}CO_2$.

There are many difficulties in extrapolating vapor phase uptake in the

laboratory to that in the field. Overall, volatilization rates are likely to be higher in the laboratory than in the field. This is because laboratory soils are normally kept moist to encourage plant growth, and this encourages volatilization. In addition, the actual deposition of volatilized chemicals onto a plant in the field is likely to be lower as atmospheric turbulence may be higher.

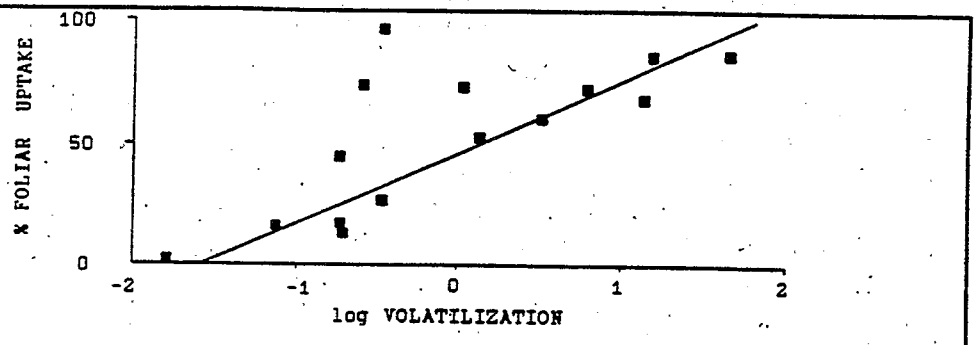

FIGURE 12 RELATIONSHIP BETWEEN VOLATILIZATION AND FOLIAR UPTAKE Adapted from Topp et al. 1986

The importance of plant uptake of organic chemicals via volatilization under field conditions, remains to be determined. However, it appears potentially significant for compounds with Hc greater than 10^{-4}. The impact could be estimated by calculating cumulative volatilization losses over the growing period and assuming that all of it ends up in the plant. The model of Jury et al., 1983 could be utilized for this purpose.

Plant Type

The final variable affecting plant uptake of soil-borne organic pollutants is the plant species itself. There has been no systematic examination of plant responses to organic chemicals in soil, although it does appear that, as with plant uptake of soil-borne heavy metals, there is variation in uptake both between species and within the same species on an individual level (Chaney, 1985; and McNeilly, 1978). For example; Harris and Sans (1967) found that sugar beet roots accumulated more dieldrin from a clay soil that contained dieldrin, than did carrots, potatoes, sugar beet tops, corn, oats, and alfalfa. Lichtenstein and Shulz (1965) on the other hand, report that carrots usually take up more organochlorine insecticides than do other root crops such as potatoes, radish, turnip, and beet. This apparent contradiction can be resolved by consideration of varietal differences which can be as much as 400% when different carrot varieties are grown in soil containing endrin (Hermanson et al., 1970).

CONCLUSIONS

In solution culture, the movement of nonionic organic compounds into roots is a passive process, equivalent to a partitioning between the liquid and solid

phase, and can be related to the octanol water partition coefficient of the compound. Subsequent translocation of the chemical from roots to shoots depends on the K_{ow} of the compound and the transpiration rate of the plant. Based on available data, compounds with a log K_{ow} of approximately 4.5 are most likely to accumulate in the stem and leaf tissue of plants.

In soil systems, there is competition between the plant and soil solids (organic fraction) for the partitioning of organics from solution. As the sorption of the compound by the soil organic phase increases, the quantity available for plant uptake decreases. Based upon these considerations compounds with log K_{ow} of 1 -2 are most likely to have significant transport of the chemical to above ground plant tissue produced in soil systems. If metabolism of the compound in the roots is significant, even compounds with low log K_{ow}'s may not be translocated (McFarlen et al., 1987). Compounds with high log K_{ow} > 5.0 would not be expected to be present in above ground plant tissue if plant uptake is limited by soil solution.

The potential for root or plant sorption of organic compounds from vapor is dependent upon the vapor pressure of the compound. Very few experiments on this route of plant contamination have been conducted. Based upon the movement of herbicides in the soil, a Henry's constant of 10^{-4} may be used as a transition point between primary movement in solution and vapor phases. If it can be assumed that vapor movement in the soil will result in vapor uptake by the plant, then those compounds with Hc $>10^{-4}$ are potential candidates for vapor phase uptake.

Superimposed upon both of these processes is the half-life of the compound. If it is short, i.e., less than 10 days, the chemical is likely lost from the system before it can be taken up by the plant. Those compounds with long half lives, i.e., greater than 6 months or greater than the growing season of the plant, presist long enough to impact plants.

Applying these screening processes to the priority pollutants, listed in Table 1, reduces the number of chemicals likely taken up by plants. For example, if plant uptake and translocation without vaporization is the pathway of concern, the list of 107 chemicals is reduced to 50 on the basis of half-life and K_{ow}, (Table 2). If vaporization is of concern the list is reduced from 107 to 64 on the basis of half-life and Hc, (Table 3).

TABLE 2 Log K_{ow}, Half-life and Hc for Priority Pollutants which are subject to plant uptake from soil

Compound	log K_{ow}	$T_{1/2}$	Hc	Compound	log K_{ow}	$T_{1/2}$	Hc
PESTICIDES							
20.Acrolein	-0.09	B	2.8E-03	26.Dieldrin	2.9	C	3.0E-04
27.Endosulfan	3.55	C	nd	30.Heptachlor epoxide	3.9	C	3.2E-05
31.Hexachlorocyclohexane	3.8	B	3.0E-04	32.Lindane	3.72	C	6.0E-04
33.Isophorone	1.70	nd	nd	35.Toxaphene	3.85	C	2.1E-01
POLYCHLORINATED BIPHENYLS							
HALOGENATED ALIPHATIC HYDROCARBONS							
38.Chloromethane	0.91	C	1.6E+01	39.Dichloromethane	1.25	B	8.5E-02
40.Trichloromethane	1.9	B	1.2E-01	41.Tetrachloromethane	2.64	nd	9.6E-01
42.Chloroethane	1.54	B	6.1E-01	43.1,1-dichloroethane	1.79	B	1.7E-01
44.1,2-dichloroethane	1.48	B	3.8E-02	45.1,1,1-trichloroethane	2.17	nd	1.3E+00
46.1,1,2-trichloroethane	2.17	nd	3.1E-01	54.1,2-dichloropropane	2.28	nd	1.2E-01
56.Hexachlorobutadiene	3.74	C	4.3E-01	58.Bromomethane	1.10	B	4.4E+00
59.Bromodichloromethane	1.88	nd	nd	60.Dibromochloromethane	2.09	nd	nd
61.Tribromomethane	2.30	nd	2.4E-02	62.Dichlorodifluoromethane	2.16	C	6.3E+01
63.Trichlorofluoromethane	2.53	nd	2.4E+00				
HALOGENATED ETHERS							
65.Bis(2-chloroethyl)ether	1.58	nd	4.7E-05	66.Bis(2-chloroisopropyl)ether	2.58	nd	4.7E-02
70.Bis(2-chloroethoxy)methane	1.26	C	1.1E-05				
MONOCYCLIC AROMATICS							
72.Chlorobenzene	2.84	nd	1.5E-01	73.1,2-dichlorobenzene	3.38	nd	1.5E-01
74.1,3-dichlorobenzene	3.55	nd	1.1E-01	75.1,4-dichlorobenzene	3.55	nd	9.9E-02
79.Nitrobenzene	1.85	C	5.4E-04	81.2,4-dinitrotoluene	2.01	nd	1.3E-02
82.2,6-dinitrotoluene	2.05	nd	1.3E-02	84.2-chlorophenol	2.17	nd	1.9E-04
86.2,4,6-trichlorophenol	3.61	nd	1.7E-06	88.2-nitrophenol	1.76	C	3.2E-04
89.4-nitrophenol	1.91	B	2.6E-04	90.2,4-dinitrophenol	1.53	C	2.7E-08
91.2,4-dimethylphenol	2.50	nd	7.5E-04	92.p-chloro-m-cresol	3.11	nd	1.0E-04
93.4,6-dinitro-o-cresol	2.85	nd	nd				
PHTHALATE ESTERS							
94a.Dimethyl phthalate esters	2.12	B	8.8E-06	94b.Diethyl	3.22	B	1.9E-03
POLYCYCLIC AROMATIC HYDROCARBONS							
95d.Naphthalene	3.37	C	2.0E-02				
MISCELLANEOUS COMPOUNDS							
99.Dimethyl nitrosamine	0.06	nd	nd	100.Diphenyl nitrosamine	2.57	nd	nd
101.Di-n-propyl nitrosamine	1.31	nd	nd	106.1,2-diphenylhydrazine	3.03	nd	nd

TABLE 3. Log K_{ow}, Half-life and Hc for the Priority Pollutants which are subject to plant uptake via volatilization

Compound	log K_{ow}	$T_{1/2}$	Hc	Compound	log K_{ow}	$T_{1/2}$	Hc
PESTICIDES							
20.Acrolein	-0.09	B	2.8E-03	22.Chlordane	4.3	C	3.9E-03
23.DDD	5.99	C	0.5E+00	25.DDT	5.98	C	2.0E-03
27.Endosulfan	3.55	C	nd	31.Hexachlorocyclohexane	3.8	B	3.0E-04
33.Isophorone	1.70	nd	nd	34.TCDD	6.14	C	nd
35.Toxaphene	3.85	C	2.1E-01				
POLYCHLORINATED BIPHENYLS							
36a.Arochlor 1016	4.38	C	8.6E-01	36b.Arochlor 1221	4.09	C	1.3E-02
36c.Arochlor 1232	4.54	C	2.1E+00	36d.Arochlor 1242	4.11	C	5.5E-02
36e.Arochlor 1248	5.6	C	1.1E-01	36f.Arochlor 1254	6.04	C	1.1E-01
36g.Arochlor 1260	6.11	C	2.9E-01	37.2-chloronaphthalene	4.12	C	1.3E-02
HALOGENATED ALIPHATIC HYDROCARBONS							
38.Chloromethane	0.91	C	1.6E+01	39.Dichloromethane	1.25	B	8.5E-02
40.Trichloromethane	1.9	B	1.2E-01	41.Tetrachloromethane	2.64	nd	9.6E-01
42.Chloroethane	1.54	B	6.1E-01	43.1,1-dichloroethane	1.79	B	1.7E-01
44.1,2-dichloroethane	1.48	B	3.8E-02	45.1,1,1-trichloroethane	2.17	nd	1.3E+00
46.1,1,2-trichloroethane	2.17	nd	3.1E-01	48.Hexachloroethane	4.62	nd	4.1E-01
54.1,2-dichloropropane	2.28	nd	1.2E-01	56.Hexachlorobutadiene	3.74	C	4.3E-01
58.Bromomethane	1.10	B	4.4E+00	59.Bromodichloromethane	1.88	nd	nd
60.Dibromochloromethane	2.09	nd	nd	61.Tribromomethane	2.30	nd	2.4E-02
62.Dichlorodifluoromethane	2.16	C	6.3E+01	63.Trichlorofluoromethane	2.53	nd	2.4E+00
HALOGENATED ETHERS							
66.Bis(2-chloroisopropyl)ether	2.58	nd	4.7E-02	68.4-chlorophenyl phenylether	4.08	nd	1.0E-02
69.4-bromophenyl phenyl ether	4.28	nd	nd				
MONOCYCLIC AROMATICS							
72.Chlorobenzene	2.84	nd	1.5E-01	73.1,2-dichlorobenzene	3.38	nd	1.5E-01
74.1,3-dichlorobenzene	3.55	nd	1.1E-01	75.1,4-dichlorobenzene	3.55	nd	9.9E-02
77.Hexachlorobenzene	6.18	C	7.0E-02	81.2,4-dinitrotoluene	2.01	nd	1.3E-02
82.2,6-dinitrotoluene	2.05	nd	1.3E-02	93.4,6-dinitro-o-cresol	2.85	nd	nd
PHTHALATE ESTERS							
94b.Diethyl	3.22	B	1.9E-03				
POLYCYCLIC AROMATIC HYDROCARBONS							
95a.Acenaphthene	4.13	C	1.0E-02	95b.Acenaphthylene	4.07	C	4.8E-03
95c.Fluorene	4.18	C	4.8E-03	95d.Naphthalene	3.37	C	2.0E-02
96a.Anthracene	4.45	C	1.1E-02	96c.Phenanthrene	4.46	C	1.6E-03
97b.Benzo[b]fluoranthene	6.57	nd	nd	97c.Benzo[k]fluoranthene	6.84	C	nd.
97d.Chrysene	5.61	C	8.8E-02	97e.Pyrene	5.32	C	2.8E-01
98a.Benzo[ghi]perylene	7.23	C	nd	98b.Benzo[a]pyrene	6.04	C	4.9E-01
98c.Dibenzo[a]anthracene	5.97	C	nd	98d.Indeno(123-cd)pyrene	7.66	C	nd
MISCELLANEOUS COMPOUNDS							
99.Dimethyl nitrosamine	0.06	nd	nd	100.Diphenyl nitrosamine	2.57	nd	nd
101.Di-n-propyl nitrosamine	1.31	nd	nd	104.1,2-diphenylhydrazine	3.03	nd	nd

Clearly, plant uptake of soil borne organic pollutants is a complex phenomena. More work is needed before the potential environmental impact of organic pollutants can be adequately assessed and actions designed to limit such impacts.

REFERENCES

Beall, M.L.,and R.G. Nash. 1971. Organochlorine insecticide residues in soybean plant tops: Root uptake vs. vapor sorption. Agron. J. 63: 460-464.

Bowen, H.J.M. 1977. Residence times of heavy metals in the environment. In Proc. Int. Conf. Heavy Metals in the Environment. Eds. T.C.Hutchinson, et al. Institute for Environmental Studies. University of Toronto. Ontario. Canada. October, 1975.

Briggs, G.G. 1973. A simple relationship between soil adsorption of organic chemicals and their octanol water partition coefficients. Proc. 7 Brit. Insecticide and Fungicide Conf. Nottingham U.K. pp83-86.

Briggs, G.G., R.H. Bromilow, and A.A. Evans. 1982. Relationship between lipophilicity and root uptake and translocation of non-ionized chemicals by barley. Pestic. Sci. 13: 495-504.

Briggs, G.G., R.H. Bromilow, A.A. Evans, and M. Williams. 1983. Relationships between lipophilicity and the distribution of non- ionized chemicals in barley shoots following uptake by the roots. Pestic. Sci. 14: 492-500.

Briggs, G.G., R.H. Bromilow, R. Edmondson, and M. Johnston. 1976. Distribution coefficients and systemic activity. Chem. Soc. Spec. Publ. No. 29: 129-134.

Brown, D.S. and E.W. Flagg. 1981. Empirical prediction of organic pollutant sorption in natural sediments. J. Environ. Qual. 10: 382-386.

Chaney, R.L., 1985. Potential effects of sludge borne heavy metals and toxic organics on soils, plants, and animals, and related regulatory guidelines. Annex 3, Workshop 9, pp 1-56. In Final Report of the Workshop on the International Transportation, Utilization or Disposal of Sewage Sludge Including Recommendations. PNSP/85-01.

Dawson, G.W., C.J. English, and S.E. Petty. 1980. Physical and chemical properties of hazardous waste constituents. Attachment 1. Appendix B. Identification and listing of hazardous wastes. EPA Office of Solid Waste.

Dejonckheere, W., W. Steurbaut, G. Melkebeke and R.H. Kips. 1982. Leaching of aldicarb and thiofanox and their uptake in soils by sugarbeet plants. Pestic. Sci. 14: 99-107.

Di Toro, D.M. and L.M. Horzempa. 1982. Reversible and resistant components of PCb adsorption-desorption isotherms. Environ. Sci. Technol. 16: 594-602.

Fairbanks, B.C., G.A. O'Connor, and S.E. Smith. 1987. Mineralization and volatilization of polychlorinated biphenyls in sludge-amended soils. J. Environ. Qual. 16: 18-25.

Farmer, W.J., K. Igue, W.F. Spencer, and J.P. Martin. 1972. Volatility of organochlorine residues from soil; Effect of concentration, temperature, air flow rate and vapor pressure. Soil Sci. Soc. Amer. Proc. 36: 443-447.

Fries, G.F., and G.S. Marrow. 1981. Chlorobiphenyl movement from soil to soybean plants. J.Agric. Food Che. 29: 757-759.

Gillett, J.W. 1983. A comprehensive prebiological screen for ecotoxicologic effects. Environ. Toxic. Chem. 2: 463-476.

Goldstein, D.J., 1982. Air and steam stripping of toxic pollutants: Vol. II, USEPA, Report No. 68-03-002.

Graham-Bryce, I.J. 1984. Optimization of physicochemical and biophysical properties of pesticides. In Pesticide synthesis through rational approaches. Eds. Magee, P.S., G.K. Kohn, J.J. Menn. American Chemical Society.

Goring, C.A.I.and J.W. Hamaker. 1972. Organic chemicals in the soil environment. Marcel Dekker Inc., New York.

Guenzi, W.D.,and W.E. Beard. 1970. Volatilization of lindane and DDT from soils. Soil Sci. Soc. Amer. Proc. 34 : 443-447.

Harris, C.R. and E.P. Lichtenstein. 1961. Factors affecting the volatilization of insecticidal residues from soil. J.Econ. Entomol. 54 : 1038-1045.

Harris, C.R., and W.W. Sans. 1967. Absorption of organochlorine insecticide residues from agricultural soils by root crops. J. Agr. Food Chem. 15: 861-863.

Hermanson, H.P., L.D. Anderson, and F.A. Gunther. 1970. Effect of variety and maturity of carrots upon uptake of endrin residues from soil. J. Econ. Entomol. 63: 1651-1654.

Jury, W.A., W.F. Spencer, and W.J. Farmer. 1983. Behavior assessment model for trace organics in soil. I. Model description. J. Environ. Qual. 12: 558-564.

Jury, W.A., W.J. Farmer, and W.F. Spencer. 1984. Behavior assessment model for trace organics in soil. II. Chemical classification and parameter sensitivity. J. Environ. Qual. 13: 567-572.

Jury, W.A., W.F. Spencer, and W.J. Farmer. 1984. Behavior assessment model for trace organics in soil. III. Application of screening model. J. Environ. Qual. 13: 573-579.

Kay, B.D., and D.E. Ekrick. 1967. Adsorption and movement of lindane in soils. Soil Sci.104: 314-322.

Karichkoff, S.W. 1981. Semi epirical estimation of sorption of hydrophobic pollutants on natural sediments and soils. Chemosphere 10: 833-846.

Kenaga, E.E. 1972. Guidelines for environmental study of pesticides; Determination of bioconcentration potential. Resid. Revs. 44: 73-113.

Khan, S.U. 1982. Studies on bound 14C-prometryn residues in soil and plants. Chemosphere. 11: 171-195.

Lichtenstein, E.P. and K.R. Schulz. 1965. Residues of aldrin and heptachlor in soils and their translocation into various crops. J. Agr. Food Chem. 13: 57-63.

Lichtenstein, E.P., T.W. Fuhremann, N.E.A. Scopes, and R.F. Skrent. 1967. Translocation of insecticides from soils into pea plants; Effects of the detergent LAS on translocation and plant growth. J. Agric. Food Chem. 15: 864-869.

Loehr, R.C., and J.F. Malina. 1986. Land treatment: A Hazardous Waste Management Alternative. Water Resources Symp. 13.

Mackay, D. and W.Y .Shiu. 1981. Critical review of Henery's Law constants for chemicals of environmental interest. J. Phys. Chem. Ref. Data,10(4):1175.

Mayer, R., J. Letey, and W.J. Farmer. 1974. Models for predicting volatilization of soil incorporated pesticides. Soil Sci. Soc. Amer. Proc. 38: 563-568.

McFarlane, C., C. Nolt, C. Wickliff, T. Pfleeger, R. Shimabuku, and M. McDowell. 1987. The uptake, distribution and metabolism of four organic chemicals by soybean plants and barley roots. Environ. Toxic. Chem. 6: 847-856.

McNeilly, T. 1982. A rapid method for screening barley plants for aluminum tolerance. Euphytica 31: 237-239.

Means, J.C., S.G. Wood, J.J. Haslett, and W.L. Banwart. 1982. Sorption of amino and carboxy-substituted polynuclear aromatic hydrocarbons by sediments and soils. Environ. Sci. Technol. 16: 93-98.

Parker, C. 1966. The importance of shoot entry in the action of herbicides applied to the soil. Weeds. 14 : 117-121.

Prendeville, G.N. 1968. Shoot zone uptake of soil applied herbicides. Weed Res. 8: 104-114.

Rao, P.S.C., J.M. Davidson, V.E. Berkheiser, L.T. Ou, et al.1982. Retention and transformation of selected pesticides and phosphorus in soil water systems. A critical review. EPA 660/3- 82-060.

Ryan, J.A. 1976. Factors affecting plant uptake of heavy metals from land application of residuals. Proc. Conf. Disposal of residues on land. St. Louis.

Sabljic, A. 1987. On the prediction of soil sorption coefficients of organic pollutants from molecular structure: Application of molecular topology model. Environ. Sci. Technol. 21:358-366.

Scheunert, I., E. Topp, J. Schmitzer, W. Klein, F. Korte. 1985. Formation and fate of bound residues of 14C benzene and 14C chlorobenzenes in soils and plants. Ecotoxicol. Environ. Safety 9, 159-170.

Schwarzenbach, R.P., and J. Westall. 1981. Transport of non polar organic compounds from surface water to groundwater. Environ. Sci. Technol. 15: 1360-1367.

Shone, M.G.T., and A.V. Wood. 1976. Uptake and translocation of some pesticides by hypocotyls of radish seedlings. Weed Res. 16: 229-238.

Shone, M.G.T., and A.V. Wood. 1974. A comparison of the uptake and translocation of some organic herbicides and a systemic fungicide by barley : I Absorption in relation to physico-chemical properties. J. Exp. Bot. 25: 390-400.

Shone, M.G.T., B.O. Barlett, and A.V. Wood. 1974. A comparison of the uptake and translocation of some organic herbicides and a systemic fungicide by barley; ii Relationship between uptake by roots and translocation to shoots. J.Exp. Bot. 2: 401-409.

Shone, M.G.T., D.T. Clarkson, J. Sanderson and A.V. Wood. 1973. A comparison of the uptake and translocation of some organic molecules and ions in higher plants. Ion Transport in Plants. Academic Press, chapter 8, 571-582.

Smith, L.R,.J. Dragun. 1984. Degradation of volatile chlorinated aliphatic priority pollutants in groundwater. Environ. Int. 10,291-298.

Thibodeaux, L.J. 1979. Chemodynamics - Environmental movement of chemicals in air, water and soil. Wiley Interscience. p501.

Topp, E., I. Scheunert, A. Attar, and F. Korte. 1986. Factors affecting the uptake of 14C labelled organic chemicals by plants from soil. Ecotoxicol. Environ. Safety 11: 219-228.

Travis, C.C. and A.D. Arms. 1988. Bioconcentration of organics in beef, milk, and vegetation. Environ. Sci. Technol. 22: 271-274.

USEPA. 1979. Water related environmental fate of 129 priority pollutants. Volumes 1 and 2. EPA-440/4-79-029b.

USEPA, 1985. Toxic Substance Control Act, Chemical substance inventory. EPA-560/7-85-002a.

Walker, A. 1971. Effects of soil moisture content on the availability of soil applied herbicides to plants. Pestic. Sci. 2: 56-59.

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