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**Human Health Risk
Assessment for the
Use and Disposal of
Sewage Sludge:
Benefits of Regulation**

Prepared for:

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Acknowledgments

Table A

Estimated Baseline Cancer Risks from Sewage Sludge

	Aggregate Cancer Risk (cases/yr)[a]	Risk to Highly Exposed Individual (risk/lifetime)[a]	Risk to Average Exposed Individual (risk/lifetime)[a]
Incineration[b]	0.3 - 4	6×10^{-4} - 7×10^{-3}	2×10^{-7} - 3×10^{-6}
Land Application	0.5	6×10^{-4}	1×10^{-7}
Surface Disposal	0.07	6×10^{-4}	2×10^{-6}
All Practices Combined[b,c]	0.9-5	6×10^{-4} - 7×10^{-3}	3×10^{-7} - 1×10^{-6}

[a] All values independently rounded to one significant figure. Values may not sum to totals because of independent rounding.
[b] Lower value based on "best estimate" of emissions for organic pollutants; higher value based on "worst case" estimate of emissions for organic pollutants.
[c] Reports total in aggregate risk column, maximum in column for risks to HEI, and combined average for average exposed individual column (see text for further discussion).

Figure A

Baseline Cancer Risk By Pollutant[*]

Arsenic 21%

BAP 0.4%

Cadmium 8%

Chromium 0.2%

Aldrin/Dieldrin 17%

Other Organics 10%

PCBs 11%

HCB 33%

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this analysis. Exposure to arsenic is responsible for less than one percent of total risks through inhalation and dietary pathways, but dominates estimated cancer risks from the groundwater and direct ingestion pathways, ultimately contributing more than 20 percent of total estimated cancer risks from all pathways. Arsenic was detected in 80 percent of sludge sampled for the NSSS. The next largest contributors to estimated cancer risks are aldrin and dieldrin, which were detected in 8 percent of samples for the NSSS and contributed 17% of total cancer risk. Exposure to aldrin/dieldrin is estimated to occur primarily through emissions from incinerators. Neither of these compounds has been detected in emissions from incinerators; as with hexachlorobenzene, our estimates are based on limits of detection. PCBs were detected in 19 percent of samples from the NSSS, and account for about 11 percent of estimated cancer risk. More than 90 percent of the cancer risk estimated for PCBs occurs through the dietary pathway, for which direct ingestion by grazing animals can lead to human exposure. Exposure to cadmium (which was found in 69 percent of samples) is responsible for about 8 percent of estimated cancer risk. All estimated cancer risk from cadmium occurs through the inhalation pathway from incinerators. Most of the remainder of the estimated cancer risk is caused by products of incomplete combustion (PICs) emitted by incinerators. This analysis includes the modeling of 85 organic pollutants, most of them PICs, emitted by incinerators.

For non-cancer health effects, we compare estimated exposure to available risk reference doses (RfD). In all cases, average lifetime exposure for each of the human populations modeled falls beneath the risk reference dose for all contaminants considered. For two management practices, however, hypothetical (reasonable worst case) exposure scenarios for the HEI suggest that exposure to arsenic might exceed risk reference doses under baseline conditions. Numerical criteria in the regulation, which are based on estimated exposure for the HEI, are expected to eliminate this condition. For lead and cadmium, we use additional calculations to determine potential health effects. We find that fewer than one individual per year is expected to exceed a threshold concentration of cadmium in the kidney cortex through exposure to cadmium from sludge. For lead, Table B summarizes our estimates of the number of persons crossing blood lead thresholds (7 μg/dl for men and 10 μg/dl for women and children) because of exposure to sludge. We estimate that about 2000 persons per year could cross these thresholds because exposure to sludge; of these, about 70 percent are due to exposure to land application. Included in this total are children who cross blood lead thresholds because they ingest small amounts of garden soil contaminated by sludge used for home gardening. Table B also contains estimates for the number of cases of lead-related disease caused by exposure to sludge. Health endpoints evaluated include cases of IQ reduced below 70 for children; and hypertension, stroke, cardiovascular disease and death in adult men. About 80 percent of the estimated cases are caused by land application of sludge, mostly through residential uses. Because of the relative immobility of lead in soil, estimated risks from exposure to lead from surface disposal are quite low.

Under current, baseline conditions, more than half of the sludge represented by the NSSS is applied to land, with about 30 percent incinerated and about 15 percent managed with surface disposal. Because regulation could possibly lead to the shifting of sludge from one practice to another it is useful to consider the relative risk of alternative management practices for sludge. Table C compares risks from each type of management practice per million tons of sludge managed. Based on the "best" estimate of emissions from incinerators, cancer risks per million

Table B

Estimated Risks from Exposure to Lead from Sewage Sludge

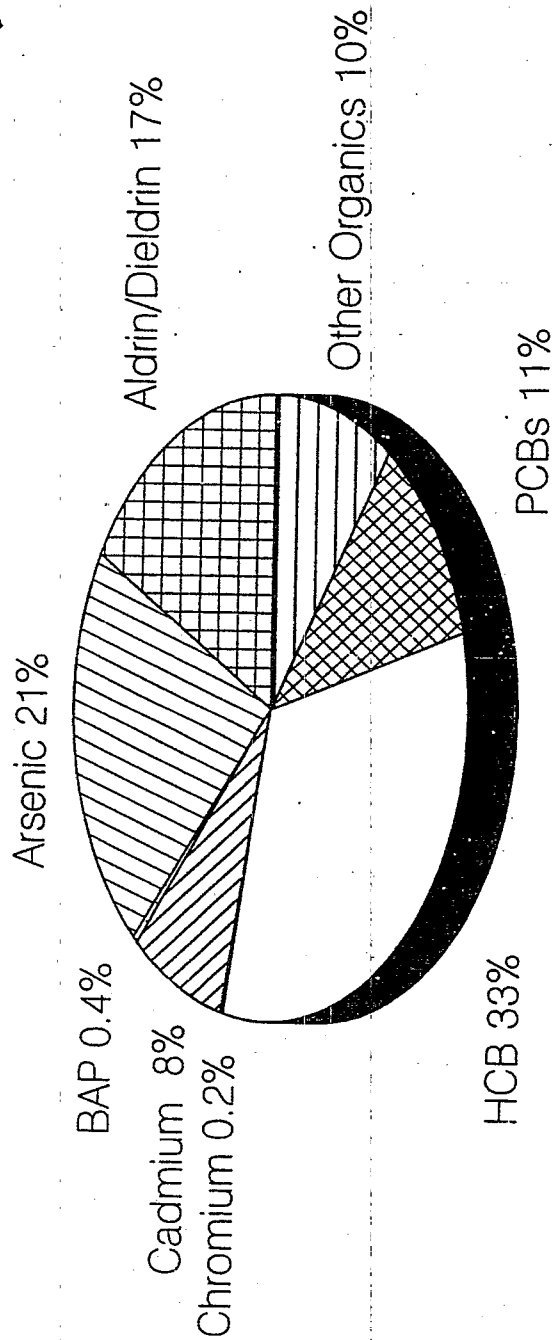|	Persons Crossing Blood Lead Thresholds[a,b]	Expected Cases of Health Effects[a,c] (cases/yr)
Incineration	700	100
Land Application	1000	500
Surface Disposal	<1	<1
All Practices Combined	2000	700

[a] All values independently rounded to one significant figure. Values may not sum to totals because of independent rounding.

[b] Includes men crossing blood lead threshold of 7 μg/dl, and women and children crossing blood lead threshold of 10 μg/dl.

[c] Includes hypertension, stroke, cardiovascular disease, and death for men; cases of IQ reduced below 70 for children.

Table C

Comparison of Baseline Risks per Million Metric Tons of Sewage Sludge
For Alternative Management Practices

	Cancer[a,b] (cases/mmt)	Persons Crossing Lead Threshold[a,c] (cases/mmt)	Cases of Disease from Lead[a,d] (cases/mmt)
Incineration	0.4 - 5	800	100
Land Application			
Agriculture: Dietary Pathways	0.4	300	20
Groundwater/Surface Water/Air	0.07	<1	<1
Residential Uses	0.08	10,000	4000
Total	0.3	1000	400
Surface Disposal			
Monofills	0.1	<1	<1
Surface Impoundments and Other	0.2	<1	<1
Total	0.2	<1	<1
All Practices Combined	0.3-2	800	200

[a] All values reflect expected health effects per million metric tons (mmt) of sludge managed, and are independently rounded to one significant figure. Values may not sum to totals because of independent rounding.

[b] Incremental cases of cancer per year from exposure to pollutants from sludge. Where a range is provided, the lower value is based on "best estimates" of emissions for organic pollutants from incineration. The higher value is based on "worst case" estimates of emissions.

[c] Number of persons crossing thresholds for concentration of lead in blood: 7 μg/dl for men, and 10 μg/dl for women and children.

[d] Includes hypertension, stroke, cardiovascular disease, and death for men; cases of IQ reduced below 70 for children.

tons are of comparable magnitude for all three groups of practices considered. Within these groups, residential uses for land-applied sludge possibly offer the lowest cancer risk per ton, followed by the disposal of sludge in monofills.

Estimated risks from lead follow a different pattern. As shown by the second and third columns of Table C, risks per ton are highest for the residential uses of sludge, where potential direct ingestion of contaminated soil by children presents significant estimated risks. Estimated risks from lead are lowest for surface disposal.

Health Benefits from Regulation

In response to the regulation, treatment works operating incinerators are expected to retrofit their facilities with additional pollution control devices, or to operate them at higher efficiency. These additional controls are expected to achieve significant reductions in emissions and resulting cancer risks, as shown in Table D. The regulation's impact on risks from land application and surface disposal could not be determined with available data and the methods used for this analysis; however, potential reductions in risk should not exceed the baseline estimates listed in the table.

Similarly, Table E shows how the regulation is expected to reduce health risks from exposure to lead. Installation of electrostatic precipitators at some incinerators is expected to eliminate about 80 percent of the cases of lead-related disease estimated for baseline conditions, and about 90 percent of the estimated number of persons crossing thresholds in blood lead. Depending on its effects on land application and surface disposal, the regulation is expected to result in the avoidance of 600-2000 persons crossing blood lead thresholds, and 90-600 cases of lead-related disease per year.

Conclusions

Results from this study should be viewed in light of the limitations to be discussed in Chapter 1. These results should serve as a useful input to the regulatory decision-making process, but further research is needed to focus on a number of areas omitted from this study. One such area is ecological effects, for which the Agency has not yet developed an accepted methodology for assessing risks. Another is risks from pathogenic organisms. In practice, these are a major determinant in choices of methods for sludge treatment or disposal. However, methods are not currently available for providing useful quantitative estimates of risks from pathogenic organisms. Additional pathways of human exposure to chemical pollutants should also be considered. In particular, future research should assess risks from indirect pathways of exposure to pollutants from incineration of sludge. Such research would assess risks from deposition of pollutants onto surface water, soil, or crops. It would also consider potential health risks from the disposal of incinerator ash. Additional technical research is needed to quantify the uncertainty implicit in the estimates reported here, and would ideally provide confidence intervals for all risk estimates.

Table D

Expected Reduction in Cancer Risks
From Regulating Sewage Sludge[a]

	Baseline Risk[a]	Risk After Regulation[a]	Reduction in Risk[b]
INCINERATION[c]			
Aggregate Risk	0.3 - 4	0.2 - 4	0.09
Risk to AEI	$2x10^{-7}$ - $3x10^{-6}$	$1x10^{-7}$ - $3x10^{-6}$	2 - 28%
Risk to HEI	$6x10^{-4}$ - $7x10^{-3}$	$4x10^{-4}$ - $7x10^{-3}$	3 - 39%
LAND APPLICATION			
Aggregate Risk	0.5	0 - 0.5	0 - 0.5
Risk to AEI	$1x10^{-7}$	0 - $1x10^{-7}$	0 - 100%
Risk to HEI	$6x10^{-4}$	0 - $6x10^{-4}$	0 - 100%
SURFACE DISPOSAL			
Aggregate Risk	0.07	0 - 0.07	0 - 0.07
Risk to AEI	$2x10^{-6}$	0 - $2x10^{-6}$	0 - 100%
Risk to HEI	$6x10^{-4}$	0 - $6x10^{-4}$	0 - 100%
ALL PRACTICES COMBINED[d]			
Aggregate Risk	0.9 - 5	0.2 - 4	0.09 - 0.7
Risk to AEI	$3x10^{-7}$ - $1x10^{-6}$	$6x10^{-8}$ - $1x10^{-6}$	2 - 75%
Risk to HEI	$6x10^{-4}$ - $7x10^{-3}$	$4x10^{-4}$ - $7x10^{-3}$	3 - 39%

[a] For aggregate risk, reports total incremental cases of cancer expected per year. For risk to average exposed individual (AEI) and highly exposed individual (HEI), reports incremental individual risk of cancer from lifetime of exposure to pollutants from sludge. All values are independently rounded to one significant figure. Values may not sum to totals because of independent rounding.

[b] For aggregate risk, reports estimated number of cancer avoided per year. For risk to AEI and HEI, reports percent reduction in risk, calculated as: (Baseline - Control)/Baseline.

[c] Lower value based on "best estimate" of emissions for organic pollutants; higher value based on "worst case" estimate of emissions for organic pollutants.

[d] Reports total for aggregate risks, average for risk to AEI, and maximum for risks to HEI.

Table E

Expected Reduction in Lead-Related Health Risks
From Regulating Sewage Sludge[a]

	Baseline Risk	Risk After Regulation	Reduction in Risk
INCINERATION[b]			
Persons Crossing Lead Threshold	700	90	600
Cases of Disease from Lead	100	30	90
LAND APPLICATION			
Persons Crossing Lead Threshold	1000	0 - 1000	0 - 1000
Cases of Disease from Lead	500	0 - 500	0 - 500
SURFACE DISPOSAL			
Persons Crossing Lead Threshold	<1	<1	<1
Cases of Disease from Lead	<1	<1	<1
ALL PRACTICES COMBINED			
Persons Crossing Lead Threshold	2000	20 - 2000	600 - 2000
Cases of Disease from Lead	700	30 - 600	90 - 600

[a] All values are independently rounded to one significant figure. Values may not sum to totals because of independent rounding.
[b] Lower value based on "best estimate" of emissions for organic pollutants; higher value based on "worst case" estimate of emissions for organic pollutants.

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α	=	empirical constant
α_i	=	intermediate variable used to calculate emissions from contaminated soil (m^2/sec)
α_l	=	longitudinal dispersivity (m)
β	=	empirical constant
ΔBP	=	change in diastolic blood pressure (mm Hg)
$\Delta Pr(CHD)$	=	change in probability of CHD event (dimensionless)
$\Delta Pr(HYP)$	=	change in probability of hypertension (dimensionless)
$\Delta Pr(MRT)$	=	change in probability of death from all causes (dimensionless)
$\Delta Pr(STR)$	=	change in probability of stroke (dimensionless)
κ	=	intermediate variable used to calculate volatile emissions from treated soil
θ	=	angle subtended by width of sludge management area or disposal site at distance equal to estimated virtual distance from site (degrees)
θ_a	=	air-filled porosity (dimensionless)
θ_{ca}	=	air-filled porosity in cover layer of soil (dimensionless)
θ_{ct}	=	total porosity in cover layer of soil (dimensionless)
θ_e	=	effective porosity of soil (dimensionless)
θ_t	=	total porosity (dimensionless)
θ_w	=	water-filled porosity (dimensionless)
μ_a	=	viscosity of air (g/cm-sec)
μ_w	=	viscosity of water (g/cm-sec)
ρ_a	=	density of air (g/cm^3)
ρ_{sl}	=	particle density of sludge (kg/m^3)
ρ_{so}	=	particle density of soil (kg/m^3)
ρ_{ss}	=	particle density of sludge-soil mixture (kg/m^3)
ρ_w	=	density of water $(g/cm^3$ or $kg/l)$
σ_z	=	standard deviation of the vertical distribution of contaminant in ambient air (m)
v^*	=	friction velocity of wind (m/sec)
τ	=	empirically derived exponent (dimensionless)
ϕ	=	effective porosity (dimensionless)
ψ	=	pressure head (m)
ψ_a	=	air pressure head (m)
a	=	first coefficient for calculating σ_z
A	=	surface area of sludge management area or surface disposal facility (m^2)
A_{ij}	=	ambient air concentration of pollutant j in cell i due to emissions from all facilities impacting that cell $(\mu g/m^3)$
A_{sma}	=	area affected by sludge management (ha)
A_{ws}	=	area of the watershed (ha)
ADLE	=	average daily lifetime exposure (mg/kg-day)
AF	=	adsorption fraction (dimensionless)

AP_{ij}	=	application rate to soil for crop i of pollutant j (kg/ha, adjusted for background soil concentration and for additional soil mass from added sludge)
AR	=	application rate for sludge (Mg/ha-yr)
AR_i	=	sludge application rate to crop i (Mg/ha-yr)
b	=	second coefficient for calculating σ_z
BAF	=	bioaccumulation factor (l/kg)
BCF	=	contaminant-specific bioconcentration factor (l/kg)
BD	=	bulk density of sludge/soil mix (kg/m^3)
BI_j	=	background intake of pollutant j (mg/kg-day)
BP_1	=	diastolic blood pressure before change in exposure to lead (mm Hg)
BP_2	=	diastolic blood pressure after change in exposure to lead (mm Hg)
BS_j	=	background soil concentration (dry wt) of pollutant j (mg/kg)
BW	=	average body weight (kg)
C	=	concentration of contaminant in sludge (mg/kg)
C_1	=	concentration of contaminant in liquid layer of surface impoundment (kg/m^3)
C_2	=	concentration of contaminant in sediment layer of surface impoundment (kg/m^3)
C_a	=	concentration of contaminant in air-filled pore space of sludge/soil mixture (kg/m^3)
C_{air}	=	average concentration of contaminant in ambient air at the receptor location (μg/m^3)
C_b	=	background concentration of contaminant in groundwater (mg/l)
C_{dw}	=	dry weight concentration of contaminant in soil (mg/kg)
C_i	=	concentration of contaminant in inflow to surface impoundment (kg/m^3)
C_j	=	concentration of contaminant j in sludge (g/DMT)
C_{lec}	=	concentration of contaminant in water leaching from site (kg/m^3 or mg/l)
C_o	=	concentration of contaminant in outflow from surface impoundment (kg/m^3)
C_s	=	concentration of contaminant adsorbed to solids (kg/kg)
C_{sed}	=	dry weight concentration of contaminant in eroded soil (mg/kg)
C_{sep}	=	concentration of contaminant in seepage from surface impoundment (mg/l)
C_{sma}	=	the concentration of contaminant in soil eroding from the sludge management area (mg/kg)
C_{sw}	=	concentration of contaminant in surface water (mg/l)
C_t	=	total concentration of contaminant in sludge/soil mixture (kg/m^3)
C_u	=	concentration of contaminant in unsaturated zone (g/m^3)
C_w	=	concentration of contaminant in water-filled pore space of sludge/soil mixture (kg/m^3)
C_{wel}	=	concentration of contaminant in well-water (mg/l)
CB	=	background concentration of contaminant in soil (mg/kg)

CD_{ij}	$=$	tissue concentration (dry wt) of pollutant j in crop i (mg/kg)
CF_{jk}	$=$	weighted average concentration of pollutant j across all food sources for animal producing meat or dairy product k (μg/g)
CI	$=$	incremental cancer risk for exposed individual (incremental risk of developing cancer per lifetime of exposure)
CI_j	$=$	incremental cancer risk from contaminant j for exposed individual (incremental risk of developing cancer per lifetime of exposure)
CI_{max}	$=$	maximum detected total incremental cancer risk among all modeled individuals (incremental risk of developing cancer per lifetime of exposure)
CP	$=$	total aggregate cancer risk for exposed population (expected incremental cases of cancer per year)
CP_j	$=$	aggregate cancer risk from pollutant j (expected cases per year)
CR	$=$	contact rate (mass per time or volume per time)
CT_{ij}	$=$	concentration of contaminant j in sludge-amended soil used to grow crop i, adjusted for background soil concentration and for additional soil mass for added sludge (mg/kg)
d_1	$=$	depth of liquid layer in surface impoundment (m)
d_2	$=$	depth of sediment layer in surface impoundment (m)
d_a	$=$	depth of aquifer (m)
d_c	$=$	depth of soil cover (m)
d_e	$=$	depth of soil eroding annually from sludge management area (m/yr)
d_i	$=$	depth to which sludge is incorporated into soil (m)
d_{mf}	$=$	depth of monofill (m)
d_{si}	$=$	total depth of surface impoundment (m)
D^*	$=$	effective molecular diffusion coefficient (m^2/sec)
D_{af}	$=$	anti-dilution factor
D_{ca}	$=$	diffusivity of contaminant in air (cm^2/sec)
D_{cw}	$=$	diffusivity of contaminant in water (cm^2/sec)
D_{eth}	$=$	diffusivity of diethyl ether in water (7.5 x 10^{-6} cm^2/sec)
D_{ip}	$=$	dispersion coefficient for cell i from facility p (μg/m^3 per g/sec)
D	$=$	total dose (mg)
DA	$=$	duration adjustment for childhood exposure through direct ingestion of soil (dimensionless)
DC_i	$=$	average daily dietary consumption of crop i (g/kg-day)
DC_k	$=$	average daily dietary consumption of meat or dairy product k (g/kg-day)
de	$=$	effective diameter of site (m)
D_{ci}	$=$	intermediate variable used to calculate emissions from contaminated soil (m^2/sec)
D_f	$=$	dilution factor (dimensionless)
DV	$=$	rate of change of volume, positive for sediment layer in surface impoundment, negative for liquid layer (m^3/sec)
E_{jp}	$=$	emission rate for contaminant j at incineration facility p (grams/sec)

ED	=	exposure duration (yr)
EXP_{ij}	=	exposure to contaminant j for persons in living in cell i (mg/kg-day)
EXP_j	=	incremental exposure to pollutant j for a particular population (mg/kg-day)
f_{ac}	=	fraction of total contaminant loading lost during active operation of monofill facility (dimensionless)
f_{act}	=	fraction of annual loading of contaminant lost from surface impoundment each year to all processes combined (dimensionless)
f_{co}	=	fraction of facility's active lifetime that typical cell contains sludge and temporary soil cover (dimensionless)
f_{d1}	=	fraction of contaminant in the liquid layer of surface impoundment that is dissolved (dimensionless)
f_{d2}	=	fraction of contaminant in the sediment layer of surface impoundment that is dissolved (dimensionless)
f_{da}	=	fraction of contaminant loss during monofill's active lifetime that is lost to degradation (dimensionless)
f_{deg}	=	fraction of contaminant loss caused by degradation (dimensionless)
f_{deg1}	=	fraction of contaminant loss from liquid layer that is lost to degradation (dimensionless)
f_{deg2}	=	fraction of contaminant loss from sediment layer that is lost to degradation (dimensionless)
f_{del1}	=	fraction of total contaminant lost from the liquid layer as a result of the diminishing volume of the liquid layer (dimensionless)
f_{del2}	=	fraction of contaminant entering the sediment layer that is trapped in the accumulating sediment layer (dimensionless)
f_{di}	=	fraction of contaminant loss from inactive monofill that is lost to degradation (dimensionless)
f_{ero}	=	fraction of contaminant loss caused by erosion (dimensionless)
f_{la}	=	fraction of contaminant loss during monofill's active lifetime that is lost to leaching (dimensionless)
f_{lec}	=	fraction of total contaminant loss caused by leaching (dimensionless)
f_{li}	=	fraction of contaminant loss from inactive monofill that is lost to leaching (dimensionless)
f_{oc}	=	organic carbon as fraction of soil mass (dimensionless)
f_{out}	=	fraction of total contaminant loss attributable to effluent (dimensionless)
f_{out1}	=	fraction of contaminant loss from a surface impoundment that is lost to outflow (dimensionless)
f_{p1}	=	fraction of contaminant in liquid layer of surface impoundment adhering to solid particles (dimensionless)
f_{p2}	=	fraction of contaminant in sediment layer of surface impoundment adhering to solid particles (dimensionless)
f_{sep}	=	fraction of total contaminant loss attributable to seepage (dimensionless)
f_{sep1}	=	fraction of mass lost from the liquid layer of surface impoundment that is lost to seepage (dimensionless)

List of Mathematical Symbols (continued)

f_{sep2} = fraction of mass entering the sediment layer of surface impoundment that is lost to seepage (dimensionless)

f_{sl} = fraction of monofill's total volume containing pure sludge (m^3/m^3 or dimensionless)

f_{sol} = fraction of solids in sludge (kg/kg or dimensionless)

f_{un} = fraction of monofill's active lifetime that typical cell contains uncovered sludge (dimensionless)

f_{va} = fraction of contaminant loss during monofill's active lifetime that is lost to volatilization (dimensionless)

f_{vi} = fraction of contaminant loss from inactive monofill that is lost to volatilization (dimensionless)

f_{vls} = fraction of total contaminant loading to monofill that is lost to volatilization during a time interval equivalent to a human lifetime (dimensionless)

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

f_{voll} = fraction of contaminant leaving the liquid layer that leaves through volatilization (dimensionless)

f_{wel} = ratio of contaminant concentration in well-water to concentration in seepage beneath the surface disposal facility (dimensionless)

f_{LS} = fraction of total cumulative loading lost in human lifetime (dimensionless)

F_a = 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)

F_{ik} = fraction of animal's food from crop i for animal product k (dimensionless)

FA_l = annual flux of contaminant leaching from treated land (kg/ha-yr)

FA_v = annual average flux of contaminant volatilizing from treated land (kg/ha-yr)

FC = flux of leached contaminant from mixture of sludge and soil (g/m^2-yr)

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

FC_k = fraction of dietary consumption of animal product k produced with sludge (dimensionless)

FD = ratio of effective diameter to depth of surface impoundment (dimensionless)

FM = food chain multiplier (dimensionless)

FS_k = direct ingestion of sludge (adherence pathway) as fraction of animal's diet for animal producing food product k (dimensionless)

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

$\bar{H}$ = Henry's Law constant (dimensionless at specified temperature)

i = index for grid cells

List of Mathematical Symbols (continued)

i	=	index for crops
I_a	=	inhalation volume (m^3/day)
I_f	=	daily consumption of fish (kg/day)
I_{ij}	=	individual intake for persons living in cell i of combined emitted contaminant j from all incinerators (mg/kg-day)
I_{sa}	=	average rate of soil ingestion for adults (g/kg-day)
I_{sc}	=	average rate of soil ingestion for children (g/kg-day)
I_t	=	daily dose of contaminant from fish (mg/day)
I_w	=	rate of water ingestion (l/day)
IC_j	=	estimate of maximum individual cancer risk for contaminant j (probability of developing cancer from lifetime dose)
j	=	index for contaminants
k_{rw}	=	effective permeability (dimensionless)
K	=	saturated hydraulic conductivity (m/sec)
K_{deg}	=	degradation rate coefficient for monofill or sludge management site (yr^{-1})
K_{deg1}	=	degradation rate coefficient for the liquid layer (sec^{-1})
K_{deg2}	=	degradation rate coefficient for the sediment layer (sec^{-1})
K_{deg3}	=	degradation rate coefficient for unsaturated soil zone (sec^{-1})
K_{del1}	=	rate coefficient for loss of contaminant from liquid layer as result of decreasing volume (m^3/sec)
K_{dis2}	=	rate coefficient for dissolved contaminant gained in sediment layer as result of increasing volume (m^3/sec)
K_{ero}	=	rate coefficient for loss of contaminant to erosion from treated land (yr^{-1})
K_g	=	mass transfer coefficient for liquid layer (m/sec)
K_l	=	mass transfer coefficient for liquid layer (m/sec)
K_{lec}	=	rate coefficient for loss of contaminant to leaching from monofill (yr^{-1})
K_{out1}	=	rate coefficient for loss of contaminant from liquid layer through outflow (m^3/sec)
K_{sep1}	=	rate coefficient for loss of contaminant through seepage from the liquid layer (m^3/sec)
K_{sep2}	=	rate coefficient for loss of contaminant through seepage from the sediment layer (m^3/sec)
K_{ta}	=	rate coefficient for total loss of contaminant during monofill's active lifetime (yr^{-1})
K_{ti}	=	rate coefficient for total loss of contaminant from inactive monofill (yr^{-1})
K_{tot}	=	total rate coefficient for contaminant loss from treated land (yr^{-1})
K_{tot1}	=	total rate coefficient for contaminant loss from the liquid layer of surface impoundment (m^3/sec)
K_{tot2}	=	total rate coefficient for contaminant loss from the sediment layer of surface impoundment (m^3/sec)

K_{va}	=	rate coefficient for loss of contaminant to volatilization from active monofill (yr^{-1})
K_{vi}	=	rate coefficient for loss of contaminant to volatilization from inactive monofill (yr^{-1})
K_{vol}	=	rate coefficient for loss of contaminant to volatilization from treated land (yr^{-1})
K_{voll}	=	rate coefficient for loss of contaminant to volatilization from the liquid layer of surface impoundment (m/sec)
KD	=	equilibrium partition coefficient for contaminant (m^3/kg)
KD_{sw}	=	partition coefficient for contaminant in the stream (l/kg)
KOC	=	organic carbon partition coefficient (m^3/kg)
KOW	=	octanol-water partition coefficient for contaminant
L_{sma}	=	distance between the SMA and the receiving water body (m)
LA_j	=	mass of contaminant added to soil treated with sludge (kg/ha)
LB_j	=	mass of contaminant from non-sludge sources in treated soil (kg/ha)
LF	=	active lifetime of monofill facility: the period in which the facility accepts sludge (yr)
LS	=	lifespan of average individual (yr)
M_{ca}	=	mass of gaseous contaminant (kg)
M_{cs}	=	mass of adsorbed contaminant (kg)
M_{ct}	=	total mass of contaminant in soil (kg)
M_{cw}	=	mass of dissolved contaminant (kg)
M_{LF}	=	mass of contaminant in monofill after end of year LF (kg/ha)
M_{LS}	=	mass of contaminant in treated soil at end of period equal to average human lifespan (kg/ha)
M_N	=	mass of contaminant in soil after N applications (kg/ha)
M_s	=	mass of soil (kg)
M_p	=	mass of sludge incinerated each year at incinerator p (DMT/yr)
M_t	=	mass of contaminant in treated soil at time t (kg/ha)
MCL	=	Maximum Contaminant Level for drinking water (mg/l)
ME_{sma}	=	mass of soil eroding annually (i.e., rate of soil loss) from one hectare of sludge management area (kg/ha-yr)
ME_{ws}	=	estimated rate of soil loss for the watershed (kg/ha-yr)
MS	=	mass of solids in one m^3 of pure sludge (kg/m^3)
MSH	=	mass of soil in zone of incorporation (mixing zone) for sludge (Mg/ha)
M_t	=	mass of contaminant in soil at time t (kg/ha)
MW	=	molecular weight of contaminant (g/mol)
n	=	empirically derived exponent (dimensionless)
N	=	number of years sludge is applied to land (yr)
N_i	=	number of years sludge is applied to land used to grow crop i (yr)
N_j	=	concentration of pollutant j in sludge (μg/g) and
Na	=	average rate of emissions from the soil surface (kg/m^2)
Na_s	=	average rate of emissions from the soil surface in first second after application of sludge (kg/m^2-sec)

List of Mathematical Symbols (continued)

Na_y = average rate of emissions from the soil surface in first year after application of sludge (kg/m²-yr)

NCI_j = individual risk from non-carcinogenic pollutant j, expressed as the ratio of exposure to Risk Reference Dose (percent)

NCP_j = number of persons exceeding RfD of pollutant j due to sludge exposure (persons)

NCP_{j1} = number of persons exceeding RfD of pollutant j without exposure to sludge (i.e., with background intake alone) (persons)

NCP_{j2} = number of persons exceeding RfD of pollutant j with sludge exposure and background intake combined (persons)

NR = net recharge (m/yr)

O_{jp} = emission rate for organic pollutant j for a unit sludge feed rate for facility of same type as facility p (g/sec per DMT/sec)

p = index for incinerator facilities

P_1 = percent solids in liquid layer of surface impoundment (kg/kg)

P_2 = percent solids in sediment layer of surface impoundment (kg/kg)

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

P_{ij} = number of persons in cell i exceeding RfD for pollutant j (persons)

P_l = percent liquid in the water column of surface water body (kg/kg, or dimensionless)

P_s = percent solids in the water column (kg/kg or dimensionless)

PA = total annual loading of contaminant to site (kg/yr)

PbB = concentration of lead in human blood (μg/dl)

PbB_1 = concentration of lead in human blood without exposure to lead from sludge (μg/dl)

PbB_2 = concentration of lead in human blood with exposure to lead from sludge (μg/dl)

POP = total exposed population (persons)

POP_i = population living in cell i (persons)

q^*_j = human cancer potency for pollutant j (mg/kg-day)$^{-1}$

q^* = human cancer potency (mg/kg-day)$^{-1}$

q_{ac} = time-weighted average rate of contaminant volatilization from a monofill (g/m²-sec)

q_{co} = rate of contaminant volatilization from a covered monofill cell (g/m²-sec)

q_{un} = rate of contaminant volatilization (emission rate) from an uncovered monofill cell (g/m²-sec)

Q_i = rate of inflow for sludge into a surface impoundment (m³/sec)

Q_o = rate of outflow from a surface impoundment (m³/sec)

Q_{sep} = seepage rate for both liquid and sediment layers (m/sec)

r' = distance from center of sludge disposal facility to the receptor's location (m)

R = ideal gas constant (m³-atm/mol-K)

R_{jp}	=	combined removal efficiency for pollutant j for furnace and control p expressed as fraction of original contaminant remaining in emissions (dimensionless)
RAC	=	reference air concentration ($\mu g/m^3$)
RCP	=	reference concentration of contaminant in seepage beneath a surface impoundment (kg/m^3)
RCS	=	reference concentration of contaminant in surface-disposed sludge (mg/kg)
RE	=	relative effectiveness of exposure (dimensionless)
RF	=	retardation factor (dimensionless)
RFA	=	reference annual flux of contaminant beneath the site (kg/ha-yr)
RfD_j	=	Risk Reference Dose for pollutant j (mg/kg-day)
RL	=	risk level (incremental risk of cancer per lifetime)
RWC	=	reference water concentration (mg/l)
S	=	intermediate variable used to calculate volatile emissions from soil
S_1	=	solids concentration in the liquid layer of a surface impoundment (kg/m^3)
S_2	=	solids concentration in the sediment layer of a surface impoundment (kg/m^3)
S_e	=	effective water saturation (dimensionless)
S_s	=	specific storage (m^{-1})
S_{sma}	=	sediment delivery ratio for the SMA (dimensionless)
S_w	=	water saturation (dimensionless)
S_{wr}	=	residual water saturation (dimensionless)
S_{ws}	=	sediment delivery ratio for the watershed (dimensionless)
Sc_g	=	Schmidt number on gas side (dimensionless)
Sc_i	=	Schmidt number on liquid side (dimensionless)
SC	=	mass of sludge contained in one hectare of monofill (kg/ha)
SMA	=	sludge management area
SRR	=	source-receptor ratio (sec/m)
t	=	time (sec or yr)
t_c	=	duration of emissions (sec)
t_{un}	=	time that typical monofill cell contains sludge without soil cover (yr)
T	=	temperature (K)
TF	=	active lifetime of surface disposal facility (sec)
TP	=	duration of "square wave" for approximating the loading of contaminant into the top of the unsaturated soil zone (yr)
TSS	=	total suspended solids content of the stream (mg/l)
u	=	wind speed (m/sec)
u_{10}	=	wind velocity at 10 meters altitude (m/sec)
U_{ij}	=	rate of uptake of pollutant j into tissue of crop i (mg/kg dry weight per kg/ha)

List of Mathematical Symbols (continued)

U_{jk} = rate of uptake of pollutant j into meat or dairy product k per unit of concentration in animal's food (mg/kg dry weight in animal tissue per mg/kg dry weight in feed)

v = vertical term for dispersion of contaminant in air (dimensionless)

v_d = darcy velocity (m/sec)

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

v_i = superimposed radial velocity from water seeping from the 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 volume of soil (m^3)

V_v = volume of void space in soil (m^3)

V_w = volume of water in soil (m^3)

x = distance from sludge management area or disposal facility to receptor (km)

x_y = lateral virtual distance to receptor location (m)

z = vertical coordinate in unsaturated zone (m)

1. INTRODUCTION

The Environmental Protection Agency (U.S. EPA) is currently developing comprehensive regulations to control the use and disposal of municipal sewage sludge under Section 405(d) of the Clean Water Act (CWA). This project supports that regulatory effort by providing estimates of both current human health risks and the benefits of controlling these risks for the major sludge management practices. The key sludge management practices examined are consistent with the CWA 405(d) regulatory effort and include incineration, surface disposal, and land application (food chain, non-food chain and distribution and marketing for residential uses).

In general, this report provides quantitative estimates of human health risks. We focus on both carcinogenic and non-carcinogenic effects from exposure to sludge pollutants of concern as identified by EPA's Office of Science and Technology (U.S. EPA, 1987d). We rely on fate, transport and exposure methodologies developed by EPA's Office of Research and Development and recommended by the Agency's scientists. Those compounds exhibiting carcinogenic effects are examined using the best available dose-response estimates obtained from the Agency's Integrated Risk Information System (IRIS). Non-carcinogenic substances are evaluated in terms of the likelihood that exposure will exceed threshold values or Risk Reference Doses (RfDs).

There are several important considerations excluded from the quantitative risk estimates. Most importantly, we did not attempt to a characterize ecological risks. Currently, available methodologies and data do not permit an adequate analysis of such risks. Neither have we attempted to estimate risks associated with exposure to pathogenic organisms. While the sewage sludge regulations do address pathogens, estimating health risks from pathogens was not within the scope of this analysis. This document also excludes the co-disposal of sludge with municipal solid waste, because that practice will be controlled under Subtitle D of the Resource Conservation and Recovery Act. In addition, indirect exposure routes, such as exposure due to deposition from incineration of sewage sludge, were not assessed.

In this chapter we develop the general analytic framework used for the analysis, including first a generic approach to risk assessment and risk-based benefits analysis and the application of this generic framework to municipal sewage sludge. The remaining chapters of this report provide detailed descriptions of the methodology and results for each of the major sewage sludge use and disposal practices to be regulated under CWA 405(d).

1.1 GENERAL FRAMEWORK FOR RISK ASSESSMENT AND RISK-BASED BENEFITS ANALYSIS

The aggregate risk assessment is intended to estimate both the expected national human health risks associated with current use or disposal of sludge ("baseline" conditions) and the benefits of the regulation measured in terms of estimated reductions in human health risks. Methods for determining these risks differ for each of the management practices (incineration, surface disposal and land application). In general, we use a sample of plants from the analytical component of the National Sewage Sludge Survey (NSSS) to represent the larger universe of

actual facilities. Estimated health risks from these plants under current practices were scaled according to weighting factors from the survey to calculate risks at the national level.

In general, risk assessment couples information on hazard identification and chemical specific assessments of dose-response relationships, with an exposure assessment based on fate and transport estimates and health (mortality/morbidity) and ecological effects. Benefits are estimated as the reduction in baseline risk associated with imposing regulatory controls. These are estimated by re-computing risks based on implementation of the regulation and represent the change (usually reduction if benefits are positive) in risks resulting from the relevant controls.

The remainder of this section describes the approach to human health risk assessment used in this analysis. This process was outlined originally by the National Academy of Sciences (NAS, 1983) and was established as final Risk Assessment Guidelines as published in the Federal Register (U.S. EPA, 1986a). Five types of guidelines were issued:

- Guidelines for Carcinogen Assessment,
- Guidelines for Estimating Exposure,
- Guidelines for Mutagenicity Risk Assessment,
- Guidelines for Health Effects of Suspect Developmental Toxicants, and
- Guidelines for Health Risk Assessment of Chemical Mixtures.

The Risk Assessment Methodology consists of four distinct steps: hazard identification, dose-response evaluation, exposure evaluation, and finally, characterization of risks. Each of these steps is discussed below.

Hazard Identification

Hazard identification consists of gathering and evaluating all relevant data that help determine whether a chemical poses a specific hazard, and making a qualitative evaluation based on the type of health effect produced, the conditions of exposure and the metabolic processes within the body that govern chemical behavior. Its goal is to determine whether it is scientifically appropriate to infer that effects observed under one set of conditions (e.g., in experimental animals) are likely to occur in other settings (e.g., in humans), and whether data are adequate to support a quantitative risk assessment.

Information on the toxic properties of chemical substances is obtained principally from animal studies and controlled epidemiological investigations of exposed human populations. The use of animal toxicity studies is based on the longstanding assumption that effects in humans can be inferred from effects in animals. The usual starting point for such investigations is the study of acute toxicity in experimental animals. Acute toxicity studies are necessary to calculate doses that would not be lethal to animals used in studies of longer duration, and typically involve a single dose or exposure of very short duration (e.g., hours of inhalation). Acute toxicity is usually expressed in terms of its LD_{50}, defined as the lethal dose on average for 50% of an exposed population. Substances exhibiting a low LD_{50} (e.g., for sodium cyanide, 6.4 mg/kg) are more acutely toxic than those with higher values (e.g., for sodium chloride, 3000 mg/kg).

In some experiments, animals are exposed continuously for several weeks or months (sub-chronic toxicity studies) or up to their full lifetimes (chronic toxicity studies). One of the goals of these types of studies is to determine the "no observed adverse effect level" (NOAEL), which is the dose at which no effect is seen. In addition to this important objective, chronic toxicity studies seek to identify specific organs or systems of the body that may be damaged by exposure to a chemical and to identify the specific disease or abnormality that a chemical may produce (e.g., cancer, neurotoxic effects.).

Animal studies often vary widely in design and implementation. Although standardized tests have been developed for various types of toxicity (e.g., National Toxicology Program carcinogenic bioassays), some tests are conducted using specialized study designs for situations where established guidelines have not been developed. Factors that need to be considered when evaluating the design and results of animal toxicity studies include selection of animal species (e.g., rats, monkeys), dose and duration (e.g., acute vs. chronic), and route of exposure (e.g., ingestion, inhalation).

Epidemiological studies are useful for determining whether a chemical poses a hazard to human health. These studies compare the health status of a group of persons who have been exposed to a suspected causal agent with that of a comparable non-exposed group. Most are either case-control studies or cohort studies. In case-control studies, a group of individuals with specific disease is identified (cases) and compared to individuals without the disease (controls) in an attempt to ascertain commonalities in exposures they may have experienced in the past. Cohort studies start with a study population, or cohort, considered free of the disease under investigation. The health status of the cohort known to have a common exposure is examined over time to determine whether any specific condition or cause of death is indicated which is greater than what might be expected due to other causes.

Because epidemiological studies assess the effects of a chemical on a human population directly, they are considered to elicit more convincing evidence of risks to human health than the results of animal studies. In general, a robust epidemiological study requires:

- determination of exposure levels, including the degree and duration of exposure,
- designation of control groups that control for other risk related factors that affect exposure and/or health status,
- availability of systematic information on health effects,
- duration of investigation sufficiently long to detect certain health effects, such as cancer, and
- adequate sizes of studied populations to provide statistical power of detection.

The next step in hazard identification is to combine the pertinent data to ascertain the degree of hazard associated with each chemical. In general, EPA uses different approaches for the qualitative assessment of risk or hazard associated with carcinogenic versus non-carcinogenic effects.

EPA's guidelines for carcinogenic risk assessment (U.S. EPA, 1986a) group all human and animal data reviewed into the following categories based on degree of evidence of carcinogenicity:

- Sufficient Evidence,
- Limited Evidence (e.g., in animals, an increased incidence of benign tumors only),
- Inadequate Evidence,
- No Data Available, and
- No Evidence of Carcinogenicity.

Human and animal evidence of carcinogenicity in these categories is combined into a weight-of-evidence classification scheme (described in Table 1-1) that includes the following groups:

- Group A - Human carcinogen,
- Group B - Probable human carcinogen,
  - B1 Higher degree of evidence,
  - B2 Lower degree of evidence,
- Group C - Possible human carcinogen,
- Group D - Not classifiable as to human carcinogenicity, and
- Group E - Evidence of non-carcinogenicity.

Table 1-1

Classification Scheme for Weight of Evidence for Carcinogenicity

Human Evidence	Animal Evidence				
	Sufficient	Limited	Inadequate	No Data	Evidence of No Effect
Sufficient	A	A	A	A	A
Limited	B1	B1	B1	B1	B1
Inadequate	B2	C	D	D	D
No Data	B2	C	D	D	E
Evidence of No Effect	B2	C	D	D	E

Group B (probable human carcinogens) are usually divided into two subgroups in which B1 is used to categorize the chemicals for which there is some limited evidence of

carcinogenicity from epidemiological studies, and B2 for which there is sufficient evidence from animal studies but inadequate evidence from epidemiological studies. EPA treats chemicals classified in categories A and B as suitable for quantitative risk assessment. Chemicals classified as Category C received varying treatment with respect to the dose-response assessment and are determined on a case-by-case basis. Chemicals in Groups D and E do not have sufficient evidence to support a quantitative dose-response assessment.

The following factors are evaluated based on a judgment of the relevance of the data for a particular chemical:

- quality of data,
- resolving power of the studies (significance of the studies as a function of the number of animals or subjects),
- relevance of route and timing of exposure,
- appropriateness of dose selection,
- replication of effects,
- number of species examined, and
- availability of human epidemiologic study data.

As with weight-of-evidence for carcinogenicity, this information is not used to quantitatively estimate risk. Rather, hazard identification characterizes the body of scientific data in such a way as to provide both a determination as to whether a chemical is a hazard and if quantitative assessment is appropriate. For non-carcinogenic health effects, the Agency's weight of evidence categories have not been formalized.

Dose-Response Evaluation

The next step in the risk assessment methodology is to estimate the dose-response relationships for the chemical under review. The evaluation of dose-response data involves quantitatively characterizing the connection between exposure to a chemical (measured as quantity and duration) and the extent of toxic injury or disease. In most cases, dose-response relationships are estimated based on animal studies because even good epidemiological studies rarely have reliable information on exposure.[1] Therefore, this discussion focuses primarily on dose-response evaluations based on animal data.

There are two general approaches to dose-response evaluation, depending on whether the health effects are based on threshold or non-threshold characteristics of the chemical. In this context, thresholds refer to exposure levels below which no adverse health effects are assumed to occur. For effects that involve the alteration of genetic material (including carcinogenicity and mutagenicity), the Agency's position is that effects may take place at very low doses, and therefore are modeled with no thresholds. For most other biological effects, it is usually (but not always) assumed that "threshold" levels exist.

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[1]An important exception is lead, which will be discussed in detail in Chapter 2.

For non-threshold effects, the key assumption is that the dose-response curve achieves zero risk only at zero dose. A mathematical model is used to extrapolate response data from doses in the observed (experimental) range to response estimates in the low dose ranges. Scientists have developed several mathematical models to estimate low dose risks from high dose experimental results. Each model is based on general theories of carcinogenesis rather than on data for specific chemicals. The choice of extrapolation model can have a significant impact on the dose response estimate. The Agency's cancer assessment guidelines recommend the use of the multistage model, which yields estimates of risk that are conservative, representing a plausible upper limit of risk. With this approach, the estimate of risk is not likely to be lower than the true risk (U.S. EPA 1986a).

EPA has acknowledged that a procedure does not yet exist for making the "most likely" or "best" estimates of risk within the range of uncertainty defined by the upper and lower limits. It can be shown that using the maximum likelihood estimates of dose-response curves may be quite different than those produced by the upper confidence limit. For example, the maximum likelihood estimate for formaldehyde yielded dose-response curve estimates five orders of magnitude less stringent than those produced by the upper confidence limit estimate, yet both were derived with the multistage model.

The Agency also provides guidance on extrapolating dosages from animal species to humans. Several methods have been proposed including:

- milligram per kilogram body weight per day,
- milligram per square meter body surface area per day, and
- parts per million in the air, water or diet.

The Agency advocates the use of the surface area approach, the most scientifically conservative, for extrapolating results from animals to humans (U.S. EPA, 1986a).

The resulting cancer potency estimate, which is referred to by the Carcinogenic Assessment Group as q^*, is the quantitative expression derived from the linearized multi-stage model and represents a plausible upper- bound estimate to the slope of the dose-response curve in the low dose range. The q^* is expressed in terms of risk-per-dose, and has units of $(mg/kg\text{-}day)^{-1}$. These values should be used only in dose ranges for which the statistical dose-response extrapolation is appropriate.

Dose-response relationships are assumed to exhibit threshold effects for systemic toxicants or other compounds exhibiting non-carcinogenic, non-mutagenic health effects. Dose-response evaluations for substances exhibiting threshold responses involve calculating what is known as the Risk Reference Dose or Reference Concentration (RfD/RfC). This measure is used as a threshold level for critical non-cancer effects below which a significant risk of adverse effects is not expected.

The RfD/RfC methodology involves taking the experimental dose at which little or no effect is observed and dividing by an appropriate uncertainty factor. These experimental levels can consist of the No Observed Effect Level (NOEL), No Observed Adverse Effect Level

(NOAEL), Lowest Observed Effect Level (LOEL) or lowest Observed Adverse Effect Level (LOAEL) and can be derived from either laboratory animals and/or human epidemiological studies. The safety factors, ranging from 100 to 10,000, are used to extrapolate from acute to chronic effects, inter-species sensitivity, variation in sensitivity in human populations, and extrapolation from a LOAEL to a NOAEL. Their magnitude can vary from 100 to 10,000, depending on the nature and quality of the data from which the experimental levels are derived. Ideally, for all threshold effects, a set of route-specific and effect-specific thresholds should be developed. If information is available for only one route of exposure, this value is used in a route-to-route extrapolation to estimate the appropriate threshold. Once these values are derived, the next step is to estimate actual human (or animal) exposure.

Exposure Evaluation

Exposure evaluation involves the nature and size of the population exposed to a substance and the level, timing and duration of their exposure. The major areas to be evaluated when estimating exposures are:

- source assessment,
- pathways and fate analysis,
- estimation of environmental concentrations,
- population analysis, and
- integrated exposure analysis.

Ideally, the source assessment should account for all mass flows of a chemical from creation to destruction. Environmental releases should account for temporal (e.g., seasonal) and geographic distributions in all environmental media. For this analysis, the source assessment for pollutants in sewage sludge begins with the characterization of pollutant concentrations, the quantities of sludge generated, and sites for the use or disposal of sludge.

A pathway and fate analysis describes how a chemical may be transported from a source to the potentially exposed population. This part of exposure assessment involves both an analysis of chemical transport and transformation, and an identification of principal pathways of exposure. Transport refers to physical properties, such as volatilization, dispersion or advection, that may effect the chemical's ultimate fate. Transformation accounts for chemical processes such as hydrolysis, photolysis or biodegradation that cause parent compounds to break down into progeny compounds. This part of exposure assessment should also account for factors such as inter-media transfers and should identify those pathways that may result in the greatest potential for exposure.

Two distinct approaches are available for estimating environmental concentrations:

- direct measurement of levels of chemicals (monitoring), and
- use of mathematical models to predict concentrations.

Measurements are a direct and preferred source of information for exposure analysis. However, they are usually expensive and are often limited geographically. The best use of such data is

to calibrate mathematical models that can be more widely applied. When estimating concentrations using mathematical models, the analyst needs not only to account for physical and chemical properties but also to document mathematical properties (e.g., analytical integration vs. statistical approach), spatial properties (e.g., one-, two- or three dimensions) and time properties (steady-state vs. nonsteady-state). Also, to the extent possible, selection of the appropriate fate and transport model should follow guidelines specified for particular media. For example, the Guidelines on Air Quality Models (U.S. EPA, 1986b) provide guidance for selecting models for the dispersion of pollutants in ambient air.

Population analysis involves a description of the size and characteristics (e.g., age or sex distribution), location, and habits (e.g., food consumption or work place) of potentially exposed human and non-human populations. Census and other survey data are useful in identifying and describing human populations exposed to a chemical.

Integrated exposure analysis involves the calculation of exposure levels along with a description of the exposed populations. An integrated exposure analysis quantifies the contact of an exposed population to each chemical under investigation via all routes of exposure and all pathways from the sources to the exposed individuals. Finally, uncertainty should be described and quantified to the extent possible.

For risk assessments involving chronic exposure, human exposure is calculated as:

$$ADLE = \frac{D}{365 \times LS \times BW}$$

where:

ADLE	=	average daily lifetime exposure (mg/kg-day),
D	=	total dose (mg),
LS	=	average lifespan (yr), and
BW	=	body weight (kg).

The total dose can be expanded as:

$$D = C \times CR \times ED \times AF$$

where:

C	=	Contaminant Concentration, or the concentration of the chemical in the medium (air, food, drinking water) contacting the body. Typical units are mass/volume (e.g., $\mu g/1$ or $\mu g/m^3$) or mass/mass (e.g., mg/kg).
CR	=	Contact Rate, or the rate at which the medium contacts the body (through inhalation, ingestion or dermal contact). Typical units are mass/time (e.g., mg/day) or volume/time (e.g., m^3 or 1/day).
ED	=	Exposure Duration, or the length of time for contact with the chemical (e.g., lifetime).

AF   =   Absorption Fraction, or the effective portion of total chemical contacting and entering the body. Entering the body means that the chemical crosses one of the three exchange membranes: alveolar membrane, gastrointestinal tract, or skin.

Characterization of Risks

The final step in the risk assessment methodology is risk characterization. This step essentially involves combining the information developed in hazard identification, dose-response assessment and exposure assessment to derive quantitative estimates of risk. Qualitative information should also accompany the numerical risk estimates, including a discussion of uncertainties, limitations, and assumptions. It is useful to distinguish methods used for chemicals exhibiting threshold effects (i.e., most non-carcinogens) from those believed to lack a response threshold (i.e., carcinogens).

EPA has recently developed Guidance on Risk Characterization for Risk Managers and Risk Assessors (U.S. EPA, 1992b) that defines three types of descriptors of risk that should be developed as part of a risk assessment: (1) individual risk, which should include both the central tendency and high end portions of the risk distribution, (2) important subgroups, such as highly exposed or highly susceptible groups or individuals, and (3) population risks. EPA specified recently that the definition of "high end" should be used to describe a plausible estimate associated with a value above 90th percentile of the actual distribution. Thus, the highly exposed individual (HEI) is differentiated from worst case, Maximally Exposed Individual (MEI) or bounding estimates in that it should not overestimate risks to the actual population.

For carcinogens, individual risks generally are represented as the probability that an individual will contract cancer in a lifetime as a result of exposure to a particular chemical or group of chemicals. Population risks are usually estimated based on expected or average exposure scenarios (unless information on distributions of exposure is available.) The number of persons above a certain risk level, such as 10^{-6} (one in a million), or a series of risk levels (10^{-5}, 10^{-4}, etc.) is another useful descriptor of population risks. Thus, individual risks also may be presented using cumulative frequency distributions where the total number of persons exceeding a given risk level is plotted against the individual risk level.

For non-carcinogens, dose-response data above the threshold are usually lacking. Therefore, risks are characterized by a comparison to the threshold level by the ratio of dose or concentration to the threshold level. Aggregate population risks for non-carcinogens can be characterized by the number of persons exposed above the RfD or RfC. The same approach can be used to assess both acute and chronic hazards. For assessing acute effects, the toxicity data and exposure assessment methods must account for the appropriate duration of exposure.

For carcinogens, risk to an individual can be represented as the maximum probability that an individual will develop cancer in a lifetime as a result of exposure to a particular chemical or group of chemicals. This probability is calculated from the estimated dose (or concentration): where:

$$CI_j = q_j^* \ EXP_j$$

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

q_j^* = human cancer potency for pollutant j (mg/kg-day)$^{-1}$, and

EXP_j = exposure to contaminant j (mg/kg-day).

Dose is a function of the concentration in environmental media, the contact rate for the chemical (inhalation rate, dermal contact rate, food consumption rate, etc.) and its rate of biological absorption.

Individual risk from a non-carcinogen is expressed as the ratio of the dose to the Reference Dose (expressed as a percentage):

$$NCI_j = \frac{EXP_j}{RfD_j} \times 100\%$$

where:

NCI_j = individual risk from non-carcinogenic pollutant j, expressed as the ratio of exposure to Risk Reference Dose (percent), and

RfD_j = Risk Reference Dose or reference concentration of pollutant j.

The pattern of environmental contamination from a given chemical varies across geographic areas. As a result, different portions of the population are exposed at different levels. The population risk assessment for both carcinogens and non-carcinogens involves calculating the risk estimates for every combination of population and concentration or dose.[2] These risk estimates are then summed across the entire area of concern. For a particular exposure group, the aggregate risk for a carcinogen is calculated from:

$$CP = \frac{q^*}{LS} \ EXP \ POP$$

where:

CP = expected number of incremental cancer cases for this exposure group (cases/yr),

LS = average human lifespan (yr),

EXP = exposure for persons in group (mg/kg-day), and

POP = population of exposure group (persons).

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[2]The expected number of cancer cases is computed per year so risk values based on a lifetime exposure must be adjusted by the average duration of a human lifetime.

1-10

These estimates are summed across all exposure groups for an estimate of aggregate cancer risks.

The aggregate risk for a non-carcinogen is defined as the total number of persons exposed to a yearly average concentration greater than the reference concentration (or dose). For each exposure group, the dose is compared to the reference dose. Aggregate risk is calculated by summing the number of persons whose dose exceeds the RfD.

Information on the distribution of risk is often also useful to provide a means of combining information on individual risk levels with the expected size of populations exposed at those levels. To accomplish this, random simulation or other methods can be used to generate a distribution of risk based on the individual distributions of the individual input parameters. This approach provides a means of incorporating uncertainty directly into the estimates.

1.2 APPLICATION OF METHODOLOGY TO THE USE AND DISPOSAL OF SEWAGE SLUDGE

The objective of our analysis is to estimate the reduction in human health risks due to the regulatory standards and controls for the use and disposal of sewage sludge. The first step in this analysis was to estimate baseline human health risks. Once baseline risks were defined, the appropriate control options were identified and the expected changes in risks were quantified. This information can also be coupled with cost information to produce cost-risk measures of the regulation.

The key inputs for estimating baseline risks include information on the sources (publicly-owned treatment works, or POTW's), sludge pollutants and ultimate disposal site characteristics. Baseline risks from the major sludge disposal options were characterized based on initial sludge concentrations, the actual quantity of sludge generated for each POTW and a number of different environments, depending upon disposal option. All these components are discussed in more detail later in this chapter.

On the basis of available information, we modeled the fate and transport of the key pollutants for the primary pathways of human health exposure. We then estimated the potential populations exposed, coupled this information with chemical-specific dose-response data to characterize baseline human health risks to both the highly exposed individual and to the aggregate population as a whole. This information is presented by pollutant, exposure pathway and disposal practice.

Once baseline risks were derived, we adjusted our assumptions for management practices to correspond with responses to regulatory controls. We then repeated all calculations to estimate the change in risk as a result of regulatory controls. This change (expressed as the number of cases avoided per year) provides a benefit measure for the regulation.

The remainder of this section briefly discusses key components of the risk assessment methodology, including:

- sludge pollutants,
- disposal options,
- fate and transport modelling,
- exposure pathways,
- exposed populations,
- health effects,
- risk characterization,
- compliance strategies, and
- benefit analysis.

1.2.1 Sludge Pollutants

Municipal sewage sludge typically contains a wide array of pollutants, from heavy metals to organic pollutants and pathogenic organisms. A 1982 EPA study identified over 200 chemical constituents from a number of sludge samples and studies (Booz-Allen & Hamilton, 1982). Based on that list, EPA conducted a preliminary screening analysis in 1983 to select a subset of pollutants of concern based on the frequency of occurrence of these constituents in sludges (from EPA's 40-City study) and on available human health and ecological toxicity information. Expert panels were then convened for each of the major disposal options to obtain consensus on whether the Agency selected the correct list of pollutants or whether additions and/or deletions should be made. The collective judgments of these panels resulted in a list of 50 pollutants that were deemed to be of concern.

During 1984 and 1985, EPA's Office of Water developed a series of profile documents for each of the 50 pollutants. For each of the key disposal practices, these profile documents examined, using screening-level fate, transport and exposure models and generally worst case assumptions, the propensity of any of the pollutants to pose a hazard. Human health hazard was determined for carcinogens by using an equivalent 1×10^{-6} cancer risk determined to be *de minimus*, and for non-carcinogens using available RfD information. Ecological effects were also examined using available threshold effect criteria, such as water quality criteria. The objective of this exercise was to narrow further the list of pollutants of possible concern.

This effort yielded the revised list of pollutants shown in Table 1-2. Note that in this report, as in the proposed sludge regulations, not all practices are considered for each of the sludge pollutants of concern. This is because the sludge profile screening analysis documents were able to eliminate certain disposal practices from further consideration. Again, this screening analysis was based on a worst-case exposure and risk assessment for each of the pollutants listed. The pollutants and practices noted in Table 1-2 serve as the basis for the risk and benefits analysis.[3]

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[3]As will be discussed in Chapter 3, we consider additional organic pollutants are considered when estimating risks from incineration, because of the creation of additional pollutants as products of incomplete combustion (PICs) within the furnace.

Table 1-2

Sludge Pollutants by Management Practice

	Incineration	Land Application	Surface Disposal
Aldrin/Dieldrin	X	X	
Arsenic	X	X	X
Benzene	X		X
Benzidine			
Benzo(a)pyrene	X	X	X
Beryllium	X		
Bis(2-ethylhexyl)phthalate	X		X
Cadmium	X	X	X
Carbon Tetrachloride	X		
Chordane	X	X	X
Chloroform	X		
Chromium	X	X	X
Copper		X	X
DDT/DDD/DDE	X	X	X
Fluorine			
Heptachlor	X	X	
Hexachlorobenzene	X	X	
Hexachlorobutadiene	X	X	
Iron			
Lead	X	X	X
Lindane		X	X
Mercury	X	X	X
Molybdenum		X	
Nickel	X	X	X
Nitrosodimethylamine		X	X
PCBs	X	X	X
Selenium		X	
Toxaphene	X	X	X
Trichloroethylene	X		X
Vinyl Chloride	X		
Zinc		X	

*Organic pollutants considered for incinerators include additional products of incomplete combustion. A list of pollutants considered for this analysis is provided in Table 3-9 of Chapter 3.

1.2.2 Disposal Options

The current 405(d) sludge regulation addresses all use or disposal practices for sewage sludge, with the following exceptions:

- landfilled sludge that is co-disposed with municipal solid waste (this will be addressed by the RCRA Subtitle D regulation),
- ocean disposal (this practice is being phased out completely), and
- co-incineration of sludge with municipal solid waste (this will be addressed at a later date).

In the regulation and in this report, use and disposal practices are grouped into three general categories:

- incineration,
- land application (including food chain, non-food chain, and residential uses), and
- surface disposal.

To determine the quality and quantity of sludge managed with each of these options, and to address serious shortcomings in data used for previous analyses of costs and benefits from regulating sewage sludge, the EPA conducted the National Sewage Sludge Survey. This survey, conducted between August, 1988 and September, 1989, characterized pollutant concentrations and sewage sludge practices, and consisted of two major components: a questionnaire which covers management practices and related data, and an analytical survey which focusses on pollutant concentrations found in sewage sludge samples. Candidate POTWs for the survey were based on over 11,000 POTWs in the U.S. and Puerto Rico, which were identified in the 1986 Needs Survey as having secondary or advanced treatment. A random sample was then stratified to be representative based on estimated wastewater flow and sewage sludge disposal practices. Out of the 479 POTWs randomly chosen to receive the questionnaire, 208 facilities were then selected for the analytical survey. Survey weights were utilized to estimate national risks associated with each practice.

The questionnaire component of NSSS was used to obtain general information about the POTW to derive national estimates of the total quantities of sewage sludge generated and estimates of treatment practices, sewage sludge use and disposal practices, quantities associated with each practice, and sewage sludge treatment and disposal costs. The analytical component of NSSS was designed to obtain sewage sludge samples for 419 pollutants, which were selected from several existing EPA regulations, including the Clean Water Act Section 307(a) priority pollutants, Appendix VII pollutants for the Resource Conservation and Recovery Act (40 CFR Part 264), toxic compounds identified in the Domestic Sewage Study, and other contaminants of potential concern for municipal sewage sludge. Table 1-3 depicts the current disposal practices by number of POTW and by percent of total sludge generated.

Table 1-3

Quantity of Sludge by Management Practice

	Quantity of Sludge (DMT/yr)[a]	Percent of Total[b]
INCINERATION	820,000	20%
LAND APPLICATION		
Food Chain Agriculture	970,000	23%
Residential	130,000	3%
Other	350,000	8%
Total	1,500,000	34%
SURFACE DISPOSAL		
Monofills	140,000	3%
Other	88,000	2%
Dedicated	240,000	6%
Total	470,000	11%
OTHER[c]	1,500,000	35%
TOTAL	**4,200,000**	**100%**

[a]Quantities reported in two significant figures.
[b]Due to independent rounding, the percentages may not sum to totals.
[c]"Other" includes not regulated (31% of total), ineligible out-of-business, or unknown.

1.2.3 Fate and Transport Modeling

This study couples fate and transport models to predict risk for each sewage sludge use disposal practice (incineration, land application and surface disposal). The models we use to predict baseline risks and benefits are consistent with those used by the Office of Science and Technology to derive numerical criteria. These models, together with accompanying assumptions, have been peer reviewed both within the Agency and externally to EPA.

Table 1-4 summarizes the models used to predict fate and transport of sludge constituents by disposal practice and medium of concern. These models combine data for pollutant-specific physical/chemical properties (e.g., decay rate, solubility, etc.) with data for the characteristics of use or disposal sites to predict chemical concentrations at selected point(s) in time and space. Therefore, the combination of pollutant concentrations in sludge, amount of sludge generated, the sludge disposal option and the region or site where disposal is practiced, can all greatly affect fate and transport model predictions. Moreover, while models chosen are among the best available, they often do not consider all relevant factors, or not all scientific and engineering factors are available to characterize the number and range of disposal scenarios and environments possible across these United States. Only after many of these models are field-tested and validated with monitoring data from sludge disposal, can we hope to narrow the uncertainty in these predictions. More detailed explanations concerning each model and application are included in later chapters of this report.

1.2.4 Exposure Pathways

Once pollutant concentrations are predicted using the fate and transport models, we determine the primary routes of human exposure for each of the disposal options. Consumption of drinking water; inhalation of airborne pollutants; ingestion of grain, vegetables, meat, poultry, dairy products, and fish; and direct ingestion of treated soil are the main routes of exposure from the use or disposal of sludge.

Exposure through drinking water can occur from the contamination of either surface water or groundwater supplies. Pollutants from both land-applied and surface-disposed sludge can migrate to aquifers beneath the use or disposal site, where they can be transported laterally to nearby drinking water wells. For land application, soil eroding from the application site can carry adsorbed pollutant to nearby surface water bodies used as sources for drinking water.

Exposure through inhalation can occur when pollutants from an incinerator are released through emissions and transported by wind to nearby human populations. Similarly, pollutant volatilizing from land application or surface disposal sites can be transported to humans living near the sites where sludge is used or disposed.

Exposure is also possible through dietary consumption of grains, vegetables, or other products grown on sludge-amended soil. Less directly, pasture or grains from treated land can be used to feed animals, which may produce meat, eggs or dairy products for human consumption. Fish caught for consumption from surface water bodies polluted by eroded soil

Table 1-4

Models Used to Simulate Transport of Pollutants

Management Practice	Transport Process	Model / Developer
INCINERATION	Air Transport	ISC-LT model /EPA-OAQPS
LAND APPLICATION		
Cropland	Crop and Animal Uptake	Linear model / EPA-OW
All Types	Mass balance in mixing zone	EPA-OW
	Transport through Unsaturated Zone	VADOFT model / EPA-ORD-Athens
	Transport through Saturated Zone	AT123D model / Oak Ridge Nat. Lab
	Volatilization to Air	EPA-OW vapor pathway model
SURFACE DISPOSAL		
Monofill Prototype	Mass Balance in Monofill Cells	EPA-OW
	Volatilization in Air	EPA-OW, ES&E
Surface Impoundment Prototype	Mass Balance within Impoundment	EPA-OW, EPA-ORD-Cin.
	Volatilization	Two-film resistance model / EPA-ORD-Cin,
Both Prototypes	Transport through Unsaturated Zone	VADOFT model / EPA-ORD-Athens
	Transport through Saturated Zone	AT123D / Oak Ridge National Laboratory

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from land-application site may provide an additional source of potential dietary exposure. Finally, children can be exposed to pollutants from sludge if they directly ingest treated soil. An additional route of possible dietary exposure, the deposition of pollutants emitted by incinerators and subsequent uptake into crops or fish (or direct ingestion by children), is excluded from this analysis.

1.2.5 Populations Exposed

Table 1-5 presents a summary by disposal option of the critical populations exposed to sludge pollutants, and the data sources used to estimate the sizes of those populations. For the dietary pathways of exposure from land application, the entire U.S. population is assumed to be exposed. Because of the complex distribution networks for grains, meat, and dairy products in the U.S., it is virtually impossible to match individual land application sites with particular populations of consumers. This analysis therefore assumes that all crops and animal products produced with sludge are distributed uniformly in the national food supply. For residential uses of land-applied sludge, we base our analysis on members of households using sewage sludge for vegetable or ornamental gardening.

For incineration, those persons living nearest to sludge incinerators are assumed to be most vulnerable. As will be explained in more detail in Chapter 3, our analysis maps predicted ground-level concentrations of emitted pollutants for about 800 locations within about 40 km of each incinerator. Persons living within this range of at least one incinerator are considered the exposed population. Similarly, for the groundwater and volatilization pathways, those individuals living within about 3 km of a surface disposal or land application site are considered for this analysis. Details are provided in Chapters 5 and 7.

1.2.6 Health Effects

For the sludge pollutants of concern, we attempt to identify the potential adverse health effects that may be associated with chronic exposure. We rely primarily on findings from the EPA's Office of Research and Development to estimate the effects of each pollutant. An important distinction is the difference in methods used for carcinogenic and non-carcinogenic compounds.

As discussed previously, for known and suspected carcinogens we rely on estimates of potency from the EPA's Office of Health and Environmental Assessment (OHEA). OHEA uses a linearized multistage procedure with zero as the threshold to derive the plausible upper bound slope (or q^*) on the low dose portion of the dose-response curve. Table 1-6 and Table 1-7 list estimates of human cancer potency for the organic pollutants and metals (respectively) considered for this analysis. The tables also include Risk Reference Doses for non-cancer health effects. For non-carcinogens, we rely on the Agency's Risk Reference Dose (RfD) as the threshold value obtained from EPA's Integrated Risk Information System (IRIS).

Table 1-5

Exposed Populations by Management Practices

	Population Exposed	Data Source
INCINERATION	Populations Residing Near Sludge Incinerators	U.S. Census (1980) from MARF (PC-GEMS)
LAND APPLICATION		
Food Chain Agriculture	Entire U.S. Population	Statistical Abstracts (1991)
All Non-residential	Population Exposed through Air, Groundwater, Surface Water, or Fish	National Well-Water Association Data Base (Wellfax)
Residential Uses	Home Gardeners	National Home Gardening Survey (1987)
SURFACE DISPOSAL	Population Exposed through Air or Groundwater	National Well-Water Association Data Base (Wellfax)

Table 1-6

Health Effects Data for Organic Contaminants in Sewage Sludge[a]

	Risk Reference Dose (mg/kg-day)	Human Cancer Potency (mg/kg-day)$^{-1}$	Level of Evidence	Reference
Aldrin/dieldrin	3×10^{-5}	17	B-2	IRIS, IRIS
Benzene	NA	0.029	A	IRIS
Benzidine	0.003	230	A	IRIS, IRIS
Benzo(a)pyrene	NA	7.3	B-2	IRIS
Bis(2-ethylhexyl)phthalate	0.02	0.014	B-2	IRIS, IRIS
Carbon tetrachloride	0.0007	0.13	B-2	IRIS, IRIS
Chlordane	6×10^{-5}	1.3	B-2	IRIS, IRIS
Chloroform	0.01	0.0061	B-2	IRIS, IRIS
DDT	0.0005	0.34	B-2	IRIS, IRIS
Dimethylnitrosamine	NA	51	B-2	IRIS
Heptachlor	0.0005	4.5	B-2	IRIS, IRIS
Hexachlorobenzene	0.0008	1.6	B-2	IRIS, IRIS
Hexachlorobutadiene	0.002	0.078	C	IRIS, IRIS
Lindane	0.0003	1.3	B-2	IRIS, HEAST
PCBs	NA	7.7	B-2	IRIS
Toxaphene	NA	1.1	B-2	IRIS
Trichloroethylene	NA	0.011	B-2	IRIS
Vinyl chloride	NA	1.9	A	HEAST

[a] Note that human cancer potency values are for oral exposure. Potencies for inhalation exposure are shown in Table 3-10, Chapter 3.
HEAST = Health Effects Assessment Summary Tables, March 1992
IRIS = Integrated Risk Information System, 1992

Table 1-7

Health Effects Data for Metals in Sewage Sludge[a]

	Risk Reference Dose (mg/kg-day)	Human Cancer Potency (mg/kg-day)$^{-1}$	Weight of Evidence (carcinogens only)	Reference
Arsenic	0.0003	1.75	A	IRIS, IRIS
Beryllium	0.005	4.3	B-2	IRIS, IRIS
Cadmium	0.0005	--	--	IRIS
Chromium(VI)	0.005	--	--	IRIS
Copper(cyanide)	0.005	--	--	IRIS
Mercury	0.0003	--	--	HEAST
Molybdenum	0.005	--	--	HEAST
Nickel	0.02	--	--	IRIS
Selenium	NA	--	--	--
Zinc	0.2	--	--	HEAST

[a] Note that human cancer potency values are for oral exposure. Potencies for inhalation exposure are shown in Table 3-10, Chapter 3.
HEAST = Health Effects Assessment Summary Tables, March 1992
IRIS = Integrated Risk Information System, 1992

Consistent with EPA's final risk assessment guideline for Chemical Mixtures (U.S. EPA 1986a), we will assume additivity of cancer potency and risk assessment estimates across pollutants for each disposal option. We do not consider synergistic or antagonistic interactions between pollutants.

One final point concerns the human absorption or the relative effectiveness (RE) of exposure to each pollutant. RE is a unitless factor that indicates the relative toxicological effectiveness of an exposure by, for example, ingestion of drinking water. The value of RE may reflect an observed or estimated difference in absorption rates between the inhalation and ingestion routes, that is then assumed to translate into a difference in the toxicant's effectiveness. When such information is not available, we assume an RE equal to one.

1.2.7 Risk Characterization

Once the results of our assessments are complete, there are two forms in which risks can be presented: quantitative and qualitative. In depicting risks quantitatively, we present risk to the average exposed individual (AEI), highly exposed individual (HEI) and aggregate risks to the population as a whole. In defining the HEI, we use reasonable worst-case assumptions consistent with those used by the agency to derive numerical criteria. For risks to the AEI and the total population, we generally rely on central tendency estimates for input parameters. Specific choices are discussed in Chapters 3-7 of this report.

For estimating risks to the highly exposed individual, the calculation is straightforward:

$$CI_j = EXP_j \, q_j^*$$

where:

CI_j	=	lifetime incremental risk of cancer from exposure to pollutant j from sludge,
EXP_j	=	incremental exposure to pollutant j for a particular population, in this case the the highly exposed individual (mg/kg-day), and
q_j^*	=	human cancer potency for pollutant j (mg/kg-day)$^{-1}$.

This risk is usually expressed as the incremental probability that an individual will contract a disease (usually cancer) over a lifetime of exposure. For non-carcinogens, we compare the estimated dose to the RfD. Results are reported by listing exposure to each pollutant as a fraction of the risk reference dose.

Individual risk to the AEI is estimated with identical methods, except the calculation is based on average exposure to the population through a particular management practice or exposure pathway. Exposure for the average individual (EXP$_j$) is multiplied by the cancer potency to estimate the average individual lifetime risk of developing cancer.

For aggregate population risk, the calculation is similar except that risks are calculated separately for each individual sub-population and summed to derive an aggregate total. In

addition, the size of the exposed population and the average human life expectancy is inserted into the equation, so that the result is reported in units of incremental cases of cancer expected per year:

$$CP_j = \frac{EXP_j \; q_j^* \; POP}{LS}$$

where:

CP_j = expected incremental cases of cancer per year as result of exposure to pollutant j from sludge (cases/yr),

EXP_j = incremental exposure to pollutant j for particular population, in this case, to a particular sub-population for which exposure has been estimated (mg/kg-day),

POP = size of population with this level of incremental exposure to pollutant j from sludge (persons), and

LS = average human lifespan (yr).

Results are summed across all populations for which exposure has been estimated to derive an estimate of aggregate risk (in cases/yr). For non-carcinogenic pollutants, results are reported by listing average exposure to each pollutant as a fraction of the risk reference dose, and by estimating the number of persons for which exposure exceeds the risk reference dose because of exposure to pollutants from sludge.

For any single management practice or exposure pathway, aggregate cancer risks (in cases/yr) can be related to individual cancer risks for the average exposed individual by multiplying the individual risk by the size of the exposed population and dividing by the average life expectancy. Conversely, multiplying the estimated aggregate risk by the average life expectancy and dividing by the size of the exposed populations gives the risk to the AEI. This same relationship holds for groups of management practices or exposure pathways (e.g., all exposure pathways for land application, or all management practices for sludge). Where such groupings are considered, the relevant population size is defined as the size of the population of individuals exposed through one or more of the pathways under consideration.

1.2.8 Benefits Analysis and Compliance Strategies

Once the baseline risks and regulatory controls are identified, the final step involves the quantification of the incremental change in risk as a result of the regulation. The estimate of benefit we choose is the reduction of human health morbidity and/or mortality as a consequence the regulatory requirement. This approach is best illustrated with a hypothetical example.

Suppose a hypothetical sludge incinerator emits a single pollutant j with a cancer potency of 0.07 (mg/kg-day)$^{-1}$. We use a mathematical model to estimate expected concentrations of pollutant j at locations near the incinerator and to map those concentrations with actual human populations. For an exposed population of 100,000 residing near the incinerator, we estimate that exposure averages 0.04 mg/kg/day, so baseline aggregate risk is calculated as:

$$CP_j = \frac{EXP_j \ q^* \ POP}{LS}$$

$$= \frac{0.04\,(mg/kg-day) \ 0.07\,(mg/kg-day)^{-1} \ 100,000}{70}$$

$$= 4\,(cases/yr)$$

Now assume that in response to regulation the incinerator installs additional emission controls, and that emissions of pollutant j are reduced 75 percent. Using the same mathematical model, we re-calculate exposure to find that it has been reduced to an average of about 0.01 mg/kg-day, so that the new estimate or aggregate risk is:

$$CP_j = \frac{EXP_j \ q^* \ POP}{LS}$$

$$= \frac{0.01\,(mg/kg-day) \ 0.07\,(mg/kg-day)^{-1} \ 100,000}{70}$$

$$= 1\,(case/yr)$$

The net incremental change in risk is equal to the difference between these two estimates: the avoidance of 3 cases of cancer per year.

For the calculations of benefits prepared with this analysis we combine benefits estimated for individual POTWs, and individual sub-populations affected by each facility to derive an estimate of total risk and benefits. Individual POTWs are expected to respond to the regulation by:

- installing additional pollution control equipment (for incinerators only),
- changing to different management options for sludge (e.g., charging from land application to co-disposal),
- refining management practices (e.g., greater distances to streams, operating an incinerator at a higher temperature), or
- requiring pre-treatment of wastewater.

Of these alternative actions, only the first is explicitly modeled for this analysis. As will be discussed in Chapter 3, some POTWs practicing incineration are expected to retrofit their incinerators to achieve reduced emissions of pollutants. This analysis estimates the extent of such reductions and re-calculates human exposure and risk to derive estimates of health benefits to be achieved by the regulation.

As will be explained in Chapters 4-7, relatively few plants are expected to reduce their use of land application or surface disposal as a result of the regulation. We have not attempted to quantity potential benefits from refinements to management practices or possible pre-treatment, except to suggest that these benefits are unlikely to exceed our estimates of baseline risks for these practices.

1.2.9 Limitations/Uncertainty

Because this analysis utilizes data provided by the National Sewage Sludge Survey, assumptions about the quantity and quality of sludge are believed to be more reliable than assumptions used for previous analyses. Nevertheless, if the random, stratified sample of facilities used for the analytic component of the NSSS (the basis for this analysis) is not truly representative (in all respects) of the full universe of POTWs, our estimates of risk may be impacted.

More serious uncertainties are implicit in assumptions required for modeling the transport of pollutants through the environment. As with almost any attempt to estimate aggregate health risks from exposure to pollutants in the environment, this analysis is limited by significant uncertainty in many of the key input parameters and the use of mathematical relationships to perform the calculations. To reduce the analytic problem to tractable proportions, we are forced to rely on simplifying assumptions, many of which are based on incomplete or imperfect data. In particular, our assumptions for human behavior represent gross simplifications of the near-infinite inter-individual variety of human behavior. For example, all exposed persons are assume to consume the same lifetime-average mix and quantity of foods in their diet. Perhaps more importantly, this analysis ignores mobility and time spent indoors or outdoors when estimating potential exposure from inhalation. In other words, it is assumed that each exposed individual resides in the same location and breathes outdoor air 24 hours per day for his or her entire lifetime.

Significant uncertainty surrounds values selected for characterizing soil types, uptake rates, human behavior, and other types of parameters required for the analysis. Where possible, the analysis gives preference to average or expected values for these parameters, but in some cases (i.e., the groundwater, surface water, and vapor pathways of exposure from surface disposal and land application) we rely on certain reasonable worst-case assumptions to provide upper bound estimates of exposure and risk.

Once we have derived estimates of human exposure, we use dose-response relationships to predict impacts on human health. As summarized above, this step is highly uncertain, and generally involves the extrapolation of results from animal experiments (usually conducted at relatively high doses) to humans (at significantly lower doses). For cancer, the values used to estimate health risk are based on upper confidence limits from a conservative model of dose-response, and are likely to over-estimate true risks.

Another limitation is that this analysis ignores certain pathways for possible human exposure. It has focussed on those pathways of potential exposure expected to be most

significant; however, risks from the pathways that were not assessed cannot necessarily be ruled out. For example, the analysis does not consider potential risks from indirect exposure to pollutants emitted by incinerators (via deposition of pollutant on soil or surface water), or potential risks from disposal of incinerator ash. Analyses for the incineration of municipal solid waste suggest that these indirect pathways might actually surpass inhalation as a source of risks to humans. Similarly, research on human exposure to lead suggests that indirect exposure can be a significant contributor to total exposure and risk (U.S. EPA 1986c, as discussed in Chapter 2).

Another important limitation concerns the absence of an analysis of risks from pathogenic organisms. EPA plans to regulate pathogens using a technology-based approach, and the methodology for quantifying pathogenic risks is not yet complete. As a consequence, we cannot at this time quantify the baseline risks and benefits for controlling pathogenic organisms.

Finally, this analysis does not attempt to quantify risks to wildlife or ecological systems. Although risks to aquatic and terrestrial wildlife were involved in deriving numerical criteria for the regulation (for land application only), no attempt has been made in this analysis to quantify any reductions in ecological impacts that may be achieved as a result of the regulation. While some methodologies do exist for quantifying ecological effects, these methodologies are usually in the form of indices rather than actual damage measurements. Many uncertainties and data gaps exist in the use of these methodologies for predictive purposes, and it is quite difficult to generate results directly compatible with a comparison of costs and benefits from a regulation.

Despite these limitations, we feel that this analysis provides a useful framework for relating regulatory action to environmental benefit. The use of sophisticated analytical tools and the best available data ensures that the risk calculations presented in this report represent meaningful (if rough) approximations of true risks and potential benefits.

2. LEAD AND CADMIUM

As explained in Chapter 1, this analysis compares exposure levels to Risk Reference Doses (RfD) for non-carcinogenic sludge contaminants. For lead and cadmium, additional techniques are used that provide a more detailed estimate of potential health risks from human exposure. This section discusses those techniques.

2.1 ESTIMATING HEALTH EFFECTS FROM LEAD EXPOSURE

Scientific understanding of health effects from environmental exposure to lead is more advanced than that for most other contaminants considered in this analysis. Adverse health effects from lead exposure have been recognized for many years, and pathways of human exposure have been examined in detail by numerous studies. In addition, epidemiological studies, both cross-sectional and longitudinal, have provided convincing evidence of association between levels of lead exposure and the incidence of various diseases. They also provide numerical estimates that can be used to quantify health risks from lead exposure. Animal studies provide details of the physiological processes involved. The U.S. EPA Office of Environmental Criteria and Assessment Office has assembled much of the available data for lead exposure and health effects into a single document: Air Quality Criteria for Lead, Vols. I-IV, (U.S. EPA, 1986c). That work provides the primary source of data and assumptions used in this analysis, and will be referenced as the "Criteria Document" or "CD" hereafter.

Assumptions and Data

This report will not attempt to duplicate the extensive discussion of exposure and health effects provided by the CD. Instead, it will briefly outline the major assumptions used for the present analysis, with frequent references to that source. These assumptions can be loosely categorized into four groups: (1) background exposure to lead from other sources and the resulting distribution of lead "body burden" in the U.S. population, (2) absorption or uptake of lead into the body as a result of additional exposure, (3) the manner in which additional exposure to lead from sludge shifts the distribution of blood lead levels in exposed populations, and (4) the relationship between blood lead concentrations and detectable health effects.

2.1.1 Background Exposure

Humans are exposed to lead through multiple environmental pathways. Anthropogenic sources appear to dominate human exposure, so humans living in remote areas evidence significantly lower levels of exposure than those living in urban areas of modern societies (CD, Sections 11,13). While direct measure of an individual's exposure to lead through these numerous environmental pathways is difficult, internal lead exposure levels can be measured through samples of any one of several biological tissues, including blood, teeth, and bone. For this analysis, levels of lead in human blood will be used as the measure of internal lead exposure in human populations. Human blood lead levels (or PbB) are typically expressed in micrograms

of lead per deciliter of human blood (μg/dl). Blood lead in daily equilibrium with other compartments in adult humans appears to have a half life of 25-28 days (CD, Section 10), so blood lead levels are best interpreted as an indication of an individual's recent level of exposure to lead.

Since the physiological response to incremental lead exposure depends to a large extent on an individual's existing body burden of lead from other sources, assumptions about background distributions of blood lead levels in the United States are central to any attempt to estimate expected health risks from additional exposure. The best data currently available for estimating this distribution are provided by the second National Health and Nutrition Examination Study (NHANES II), which was conducted from February, 1976 to February 1980, and included the sampling of blood from 16,563 individuals aged 6 months to 74 years from throughout the United States. These data suggest that blood lead levels are best approximated by log-normal distributions, with geometric means and standard deviations that differ according to sex, race, and degree of urbanization (CD, Section 11.3.4). Based on NHANES II, the CD reports mean blood lead levels of about 15 μg/dl for children aged 6 months to 5 years (of all races), about 11 μg/dl for women of age 18-74 years (all races), and 15.6 and 18.1 μg/dl for males (white and black, respectively) of age 18-74.

Proper use of the NHANES II data in estimating the current distribution of blood lead levels requires an additional consideration: levels of lead exposure in the United States (and consequently blood lead levels) are thought to have decreased significantly since completion of the NHANES II study. In particular, the phasedown of lead in gasoline has reduced lead emissions from automobiles, which accounted for an estimated 90 percent of total U.S. air emissions in 1984 (CD, Section 13). The phaseout of lead-soldered cans, reductions of lead in drinking water, and other changes are further reducing lead exposure to the U.S. population. These changes necessitate an effort to update estimates of average blood lead levels to reflect reduced exposure from those sources.

Within the four years spanned by the NHANES II study, a strong association can be detected between the use of leaded gasoline and average blood lead levels (CD, Section 11.3.6), and researchers have used this association to predict changes in blood lead due to the gasoline lead phasedown. Regression coefficients derived by Joel Schwartz of the U.S. EPA Office of Policy Analysis have been used by the U.S. EPA Office of Air Quality Planning and Standards (OAQPS) to project average blood lead levels to the year 1990. Results were provided by that office (Cohen, 1988), and are believed to provide the best available estimates of current blood lead levels. For this analysis, baseline concentrations of lead in human blood are assumed to be 4.59 μg/dl for adults and 5.41 μg/dl for children.

Also important in estimating health effects is the degree to which blood lead levels vary within each population. If the distribution of blood lead levels is to be approximated by a log-normal distribution, this degree of variation is best expressed as the geometric standard deviation of blood lead within a particular population sample. As discussed in Section 11.3.4 of the CD, analysis of the NHANES II and other data suggests geometric standard deviations of 1.3 to 1.4 for blood lead in U.S. populations. After adjustment for the impact of gasoline lead, the CD derives a geometric standard deviation of about 1.42 for children of ages 0.5-6 years.

Table 11-9 of the CD reports corresponding values of 1.34 to 1.39 for adult women, and 1.37 to 1.40 for adult men. The rounded value of 1.4 has been used in this analysis for both adults and children.

2.1.2 Absorption and Uptake of Lead

The rates at which lead is absorbed from environmental media into human tissue depend on the environmental medium involved, the age and sex of the exposed individual and other factors. Predicting health effects from lead in sludge requires consideration of exposure through air, water, and dietary pathways, and assumptions about intake from each of these sources. As with the distribution of blood lead levels in U.S. populations, assumptions used for this analysis are drawn primarily from the CD. These are summarized in Table 2-1 and are outlined briefly below.

Air. For exposure to lead emitted by sludge incinerators, the primary pathway of concern is the inhalation of lead from ambient air. The relationship between concentrations of lead in ambient air and blood lead concentrations has been evaluated by a variety of methodologies. These include experimental studies of adult volunteers, as well as epidemiological studies of different populations of children and adults. Section 11.4.1 of the CD reviews more than a dozen studies relating blood lead to ambient air exposures, and reaches the following conclusion about the slope of blood lead to ambient air (β):

(1) The experimental studies at lower air lead levels, 3.2 $\mu g/m^3$ or less, and lower blood levels, typically 30 $\mu g/dl$ or less, have linear blood lead inhalation relationships with slopes β of 0-3.6 for most subjects. A typical value of 1.64+0.22 may be assumed for adults.

(2) Population cross-sectional studies at lower air lead and blood lead levels are approximately linear with ß of 0.8-2.0 for inhalation contributions.

(3) Cross-sectional studies in occupational exposures in which air lead levels are higher (much above 10$\mu g/m^3$) and blood levels are higher (above 40$\mu g/dl$) show a much more shallow linear blood lead inhalation relation. The slope ß is in the range of 0.03-0.2.

(4) Cross-sectional and experimental studies at levels of air lead somewhat above the higher ambient exposures (9-36 $\mu g/m3$) and blood lead levels of 30-40 $\mu g/dl$ can be described either by a nonlinear relationship with decreasing slope or by a linear relationship with intermediate slope, approximately ß = 0.5.

Table 2-1

Estimated Intake Slopes:
Increment in Blood Lead Concentration per Unit of Exposure

	Low Estimate	Middle Estimate[a]	High Estimate
Air (μg/dl per μg/m^3)			
General Atmosphere			
Adults[b]	1.8	1.8	1.8
Children	1.1[c]	1.5[c]	5.0[d]
Point Source			
Adults[e]	2.3	3.5	4.6
Children	1.9[f]	1.9[f]	5.0[d]
Dietary and Drinking (μg/dl per μg/day)			
Adults	0.03[g]	0.04[h]	0.06[g]
Children	0.15[h]	0.2[i]	0.27[h]

[a]Middle Estimates are used for this analysis.
[b]From Cohen (1988).
[c]Derived from Point Source estimate.
[d]From Criteria Document (p. 13-21). Includes soil.
[e]Derived from General Atmospheres estimate.
[f]From Criteria Document (p. 13-21).
[g]From Criteria Document (p. 13-22, 13-23).
[h]From this report.
[i]Midpoint.

(5)   The blood lead inhalation slope for children is at least as steep as that for adults, with a median estimate of 1.92 from three major studies.

(6)   Slopes which include both direct (inhalation) and indirect (via soil, dust, etc.) air lead contributions are necessarily higher than those estimates for inhaled air lead alone. Studies using aggregate analyses (direct and indirect air impacts) typically yield slope values in the range 3 to 5, about double the slope due to inhaled air lead alone.

The calculations of change in blood lead level per change in air requires three steps: (1) estimating deposition of inhaled lead, (2) estimating absorption, and (3) estimating the incremental change in blood lead per unit of lead absorbed. As shown in Table 2-1, the present analysis uses air intake slopes that are roughly consistent with the ranges suggested by the CD, if indirect effects from deposition are excluded. For children, however, slopes vary with the distance of an child's residence from the source of lead emissions. The analysis distinguishes "generalized" atmospheres from those in areas close to sludge incinerators because of expected significant differences in particle size distributions.

The U.S. EPA Office of Air Quality Planning and Standards (OAQPS) has estimated differential rates of lead absorption according to distance from a lead smelter (Cohen, 1987a). Larger lead particles predominate at distances within 2-5 km of lead smelters, and are more readily deposited and absorbed in the tracheo-bronchial regions than smaller particles. For the present analysis, it has been assumed that similar distributions in lead particle size are found within 5 km of sludge incinerators. At greater distances, large lead particles are assumed to be less common. As described below, these different absorption estimates were applied to lead uptake slopes to yield lead intake estimates that vary with distance.

Based on particle size distributions and lung deposition data OAQPS has found that on average, 26-42 percent of inhaled airborne lead particles deposit in the respiratory tracts of adults in residential locations not near a point source of lead. Differential absorption rates can be applied to the deposition in each respiratory region to estimate that 15-32 percent of inhaled lead is absorbed by adults in "generalized" U.S. atmospheres. For submicron particles which dominate general atmospheres, OAQPS concluded that deposition in the lungs of a two-year-old child is approximately 1.5 times higher than that in the lungs of adults. Using this factor to adjust the adult absorption estimate, they conclude that 25-56 percent of inhaled lead is absorbed by children in general atmospheres (Cohen, 1987a).

OAQPS used the same methods to estimate total respiratory absorption of inhaled lead particles for individuals living near point sources of lead emissions. The expected particle size distributing was taken from Sledge (1987) and deposition efficiencies were taken from the Criteria Document for particulate matter (PM) and sulfuric oxides (SO$_x$). By combining these data, OAQPS estimated that near point sources of lead emissions, adults absorb approximately 38 percent of inhaled lead, and children absorb approximately 42 percent.

These absorption efficiencies were applied to uptake slopes to obtain lead intake estimates. For children, the CD reports that U.S. EPA analyses of three population studies (Yankel *et al.*, 1977; Roels *et al.*, 1980; Angle and McIntire, 1979) suggest the median blood lead increase is approximately 1.92 $\mu g/dl$ per $\mu g/m^3$ of inhaled air lead. All of these studies involved children living in the vicinity of lead smelters, so they are more applicable for children living near emission sources than for children living in generalized atmospheres. If it is assumed that children living near sources of lead emissions absorb 42 percent of inhaled lead and children living at various distances from these sources are assumed to absorb 25-45 percent, then it can be concluded that the appropriate slope for children in general conditions is (25/42 x 1.92) to (45/52 x 1.92) or 1.1-2.0 $\mu g/dl$ blood lead per $\mu g/m^3$ increment in air lead concentration. A midpoint value of 1.5 has been used for the present analysis.

These slope estimates deliberately exclude indirect exposure to incinerated lead via deposition to soil and subsequent ingestion through food, soil, or drinking water. The CD reports that inclusion of these indirect pathways results in estimated slopes of 3-5 $\mu g/dl$ increment in children's blood lead per $\mu g/m^3$ increment in air lead concentration. For consistency with exposure analyses for other sludge constituents, the present analysis ignores these indirect pathways of exposure for lead.

For adults, analysis by OAQPS found that the most reliable and relevant studies described in the CD yield inhalation slopes values in the range of 1.3-2.0 $\mu g/dl$ per $\mu g/m^3$ with a weighted slope of 1.4 (which is also the unweighted midpoint of the 0.8-2.0 range listed above). Adjusting that value by a factor of 1.3 to account for the resorption of lead from bone tissue (according to Chamberlain, 1983), OAQPS has derived an adjusted slope estimate of 1.8 $\mu g/dl$ blood lead per $\mu g/m^3$ increment in air lead concentration (Cohen, 1988).

In contrast to the available date for children, the data upon which these slopes are based were not confined to individuals living near lead smelters. As a result, they are likely to be representative of general conditions in the U.S. and perhaps less appropriate for modeling lead absorption at locations near to emission sources (where particle sizes are likely to be larger). Using the absorption rates derived by Cohen (1987a), the slope estimate above can be adjusted for higher adult absorption rates near sludge incinerators. This step yields an adjusted slope of about 3.5 $\mu g/dl$ per $\mu g/m^3$ in air lead concentration,[1] a value somewhat higher than the range of estimates for general atmospheres quoted from the CD.

---

[1] For comparison, deposition and absorption rates derived by Cohen (1987a) can be combined with uptake rates relating changes in blood lead levels to increases in the amount of absorbed lead. Based primarily on results from tracer studies by Rabinowitz (1976, 1977) and Marcus (1988) estimates that each $\mu g/day$ of lead absorbed by adults results in an approximate 0.4 $\mu g/dl$ increment in blood lead. If this estimate is combined with the finding of Cohen(1987) that adults absorb approximately 38 percent of inhaled lead when residing near lead emissions, and with the assumption that a typical adult inhales approximately 20 m^3/day (U.S. EPA, 1986c) it can be concluded that each $\mu g/m^3$ of lead in ambient air would result in an increment of 20x0.38x0.4 or about 3 $\mu g/dl$ in blood lead for these individuals, a conclusion in approximate agreement with the above estimate.

2.1.3 Dietary or Drinking Water Pathways

The CD reports that typical absorption rates for ingested lead are 10 percent for adults and 25-50 percent for children. Combining these estimates with uptake rates of 0.4 for adults and 0.5 for children (from Marcus, 1988), yields an intake slope of 0.04 for adults and 0.13-0.27 (midpoint 0.2) for children. By comparison, the CD reports intake slopes of 0.04 for adults and 0.16 for infants. Estimation of intake from drinking water is based on the same results. It is assumed that the average adult consumes 2 liters of drinking water per day and the average child consumes 1 liter daily. Lead consumed through drinking water is assumed to have the same rates of absorption and uptake as lead consumed through the diet.

2.1.4 Shifting the Blood Lead Distributions

The preceding section of this report discusses the rates at which exposure to additional lead can be expected to affect the concentration of lead in an individual's blood. More directly pertinent to the estimation of health risks from sludge is the question of how additional exposure to lead might affect the distribution of blood lead levels in an entire exposed population. Since it has been assumed that the distribution of blood lead values in a given population is approximately log-normal, and since a log-normal distribution is completely characterized by its geometric mean and geometric standard deviation (GSD) this question is equivalent to asking how additional lead exposure affects the geometric mean and GSD of blood lead in the exposed population.

A log-normal distribution is described by $x=e^{(\mu+\sigma\epsilon)}$ where e is the base of the natural logarithm function, μ is the natural log of the distribution's geometric mean, σ is the log of its geometric standard deviation, and ϵ is a normal random variable with mean of 0 and variance of 1. Exposure of a given population to additional lead is likely to shift the mean of that distribution upward, but it is difficult to predict the effect that the additional exposure might have on the GSD of the distribution. If intake slopes decline at higher levels of blood lead, the additional exposure might shift the lower part of the distribution more than the upper tail, resulting in a more compressed curve with a lower GSD. If, instead, those at higher blood lead levels are most affected by increased environmental concentrations, then the distribution might spread out as a result of additional exposure, with an increase in the distribution's GSD. Alternatively, the distribution's GSD could be unaffected by the additional exposure.

To analyze the health impacts from sludge re-use or disposal it has been assumed that the GSD of an exposed population is not affected by small increases in exposure. To estimate the effects of incremental lead exposure from sludge, the methodology assumes that increases to a population's mean blood lead level can be predicted on the basis of increases in average lead exposure. The distribution's GSD is assumed unchanged. Table 2-2 illustrates the methodology with a sample calculation, where the concentration of lead in a population's drinking water is assumed to increase by 0.2 parts per billion.

Table 2-2

Sample Calculation:
Lead in Drinking Water

	Adults	Children
Geometric Mean Blood Lead Without Exposure from Sludge (μg/dl)	4.59	5.41
Increment to Water Concentration (mg/l)	0.0002	0.0002
Incremental Lead Ingested (μg/day)	0.4[a]	0.2[b]
Average Increment to Blood Lead Level	0.016[c]	0.04[d]
Geometric Mean Blood Lead After Exposure from Sludge (μg/dl)	4.606	5.45

[a]Assumes adults ingest 2 liters of water daily.
[b]Assumes children ingest 1 liter of water daily.
[c]Assumes intake slope for adults of 0.4 μg/dl PbB per μg/day of lead ingested.
[d]Assumes intake slope for children of 0.2 μg/dl PbB per μg/day of lead ingested.
[e]The geometric standard deviation for the distribution of blood lead is assumed to remain constant at 1.4 μg/dl.

2.1.5 Health Effects

This section discusses the health effects quantified in this analysis. The model described was originally developed in support of the Office of Water's proposed regulation for sewage sludge. Later, it was refined and used in support of drinking water regulations by U.S. EPA's Office of Ground Water and Drinking Water (OGWDW), as described in U.S. EPA (1990d). Categories of potential health benefits are summarized in Table 2-3; those health effects quantified in this analysis are highlighted.

Background. U.S. EPA has conducted numerous studies on the health effects associated with lead exposure. In a pioneering study (U.S. EPA. 1985c), Schwartz *et al.* quantified a number of health benefits that would result from reductions in the lead content of gasoline. The work was extended by U.S. EPA's analysis of lead in drinking water (U.S. EPA, 1986d) and by a U.S. EPA-funded study of alternative lead National Ambient Air Quality Standards (U.S. EPA, 1987c). Despite this substantial research, much uncertainty remains. Many categories of health effects from lead exposure cannot be quantified -- credible dose-response functions are not yet available. There is also uncertainty regarding the significance of many of these health effects. It is not clear whether the estimates provided in this chapter overestimate or underestimate the actual values.

Threshold Approach. Based on information from the CD, thresholds were selected above which adverse health effects from lead were considered possible: 7 μg/dl for men, and 10 μg/dl for women and children. Using the estimated background distributions of blood lead levels discussed above, the number of persons exceeding these thresholds was estimated. Next, using estimates of potential exposure to lead from sludge, these distributions were shifted. From these revised distributions, the numbers of men, women, and children exceeding the selected thresholds were calculated again. By subtracting the number of persons exceeding the thresholds before and after exposure from sludge, the <u>incremental</u> number of persons at risk of health effects from lead in sludge could be calculated.

Children with IQs Less than 70. Based on encoded expert opinion, Wallsten and Whitfield (1986) provide estimates of expected percentages of children with IQs below 70 for each level of population mean blood lead. For this analysis, the opinions of the experts were averaged.

Health Benefits to Adult Men. With the availability of the Second National Health and Nutrition Survey (NHANES II) several studies, including Pirkle *et al.*, (1985) and Harlan *et al.* (1985), have found a statistically significant relationship between blood lead and hypertension in adult males. In particular, Pirkle *et al.* found that blood lead levels were a significant predictor of blood pressure in adult white males. They found that each increase of one log unit in blood lead in males of age 40-59 could be associated with 3.954 points of diastolic blood pressure and 8.436 points of systolic blood pressure. These relationships held when blood lead was evaluated in a regression with all factors previously known to be correlated with blood pressure, and 87 additional variables representing combinations of every dietary and serologic variable in the NHANES II survey. Results from a large-scale study of British men (Pocock *et al.*, 1985) are consistent with these results.

Table 2-3

Potential Health Benefits from Reducing Exposure to Lead

Men

*   Hypertension in adult men
*   Myocardial infarction, stroke and death in men of ages 40-59

        Myocardial infarction, stroke, and death in men of other ages

        Cancer

Women

        Hypertension, myocardial infarction, stroke, and death

        Fetal effects from maternal exposure, including diminished IQ, decreased gestational age, and reduced birth weight

        Possible increases in infant mortality resulting from maternal exposure

        Cancer

Children

        Interference with growth

*   Reduced intelligence

        Impaired hearing, behavioral changes

        Interference with development of Peripheral Nervous System

        Metabolic effects, impaired heme synthesis, anemia

        Cancer

---

* These health effects have been quantified for this analysis.

Reduced Incidence of Hypertension. One of the authors of the Pirkle *et al.* study, Joel Schwartz, has continued to investigate the relationship between blood lead and hypertension in support of the benefit analysis for the phasedown of the lead in gasoline (U.S. EPA, 1985c). Using multiple logistic regression, Schwartz derived a function that could be used to predict the incidence of hypertension in white males of ages 40-59 from individual blood lead concentrations and other variables. Results of the regression analyses are reported in U.S. EPA (1985c). Schwartz used this equation to simulate changes in the probability of hypertension for each observation in the NHANES data as a result of changes in each observation's blood lead level. By aggregating results, potential impacts of the gasoline lead phasedown were estimated. Similar techniques were later used in U.S. EPA (1986d) for estimating benefits from reducing lead in drinking water.

U.S. EPA (1987c) used results from Schwartz's multiple logistic regression model to derive a univariate logistic function which predicts the probability of hypertension (diastolic pressure above 90 mm Hg) based on blood lead levels alone:

$$\Delta Pr(HYP) = [1 + \exp(-(-2.744 + 0.793 \, (\ln PbB_1)))]^{-1} -$$

$$[1 + \exp(-(-2.744 + 0.793 \, (\ln PbB_2)))]^{-1}$$

where:

$\Delta Pr(HYP)$	=	the change in the probability of hypertension,
PbB_1	=	blood lead level before some change, and,
PbB_2	=	blood lead level after some change.

U.S. EPA's Office of Air Quality Planning and Standards derived this univariate function by replacing each of the independent variables with corresponding mean values from the NHANES II sample. Since the logistic equation is non-linear, this method of reducing the Schwartz equation to a single independent variable is likely to introduce error into the estimation process, but is thought to provide a reasonable approximation of the effect of blood lead on blood pressure. The function derived by OAQPS (which is graphed in Figure 2-1) provides the basis for estimating cases of hypertension associated with lead in sludge.

As reported in U.S. EPA (1985c) and U.S. EPA (1986d) these regression equation results are based on a sample population of white males of ages 40-59. More recently, Schwartz (1988) has tested the relationship between blood lead and blood pressure with a broader population, including all males (black and white) from 20 to 74 years of age. He found that the relationship was consistent over the entire population tested. For the present analysis, it is assumed that the univariate equation derived by U.S. EPA (1987c) can be applied to all adult males (black or white) between 20 and 74 years of age. By combining the dose-response curve with pre- and post-regulatory blood lead distributions, we calculate the number of cases of hypertension that are associated with lead in sewage sludge disposal practices.

Reduced Incidence of Coronary Heart Disease Events. Serious health consequences often result from hypertension. Several large epidemiological studies have shown that elevated

Figure 2-1
Probability of Hypertension vs. Blood Lead
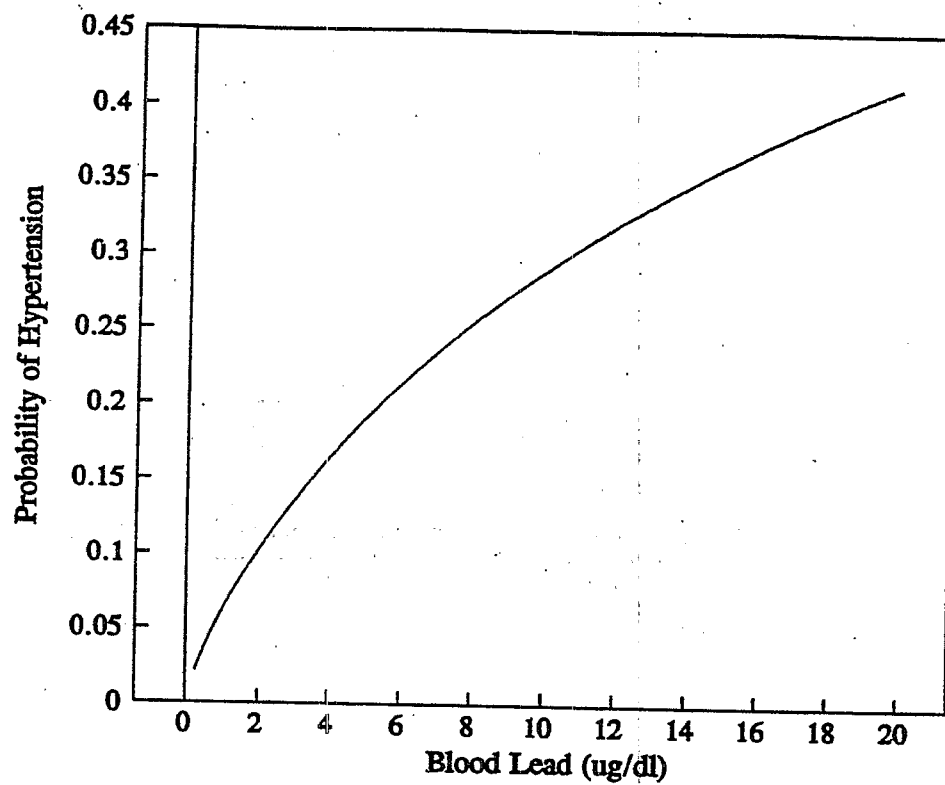
blood pressure increases the risk of coronary heart disease (CHD). The Pooling Project (1978) and the Framingham studies (Shurtleff, 1974 and McGee and Gordon, 1976) are well known studies that estimate the incidence of coronary disease as a function of blood pressure, smoking, and other risk factors. Levy et al. (1984) have shown that the risk coefficients from the studies have accurately predicted the decline in coronary incidence in the 1970s associated with reported declines in blood pressure, smoking, and cholesterol levels over the same period.

Unfortunately, none of these prospective studies recorded measures of blood lead levels for the men investigated. It is therefore impossible to use their data _directly_ to derive estimates of the relationship between blood lead levels and heart attacks, strokes, or premature death. However, indirect methods can be used to estimate dose response relationships. For each health endpoint, two steps are involved. First, estimates of blood lead levels in a population to predict its expected mean diastolic blood pressure are used. Next, this blood pressure estimate is used in conjunction with results from the Framingham and Pooling Project studies, to derive estimates of the incidence of heart attacks, strokes, and death. To calculate changes in these health endpoints, all calculations are repeated for a population whose mean blood lead level has changed. This value is subtracted from the baseline to estimate health consequences of increments in lead exposure.

Estimating Shifts in Population Mean Blood Pressure as a Function of Blood Lead Level. As mentioned above, Pirkle et al. (1985) estimated that each log unit increase in an individual's blood lead level could be associated with an expected increase of 3.954 millimeters of diastolic blood pressure. Schwartz performed additional regressions on the same data to derive a coefficient of 4.609 (U.S. EPA, 1985c). As reported in U.S. EPA (1986d), Schwartz' results were later challenged for his having failed to control for the 64 sites involved in the survey. Subsequent work by Schwartz (1986b) repeated the regressions with the addition of dummy variables for each of the sampling sites in the survey, to yield a revised coefficient of 2.74 for diastolic pressure, an approximate 40 percent decrease from that reported in U.S. EPA (1985c). More recently, Schwartz (1988) reports having used a random effects model to test the effect of site variation on the regression results, and finds that regression coefficients for the log of blood lead are reduced by about 25 percent from original estimates for systolic and diastolic pressure, to yield a somewhat higher coefficient than the 2.74 value reported in U.S. EPA (1986c).

U.S. EPA (1987c) recommends using the 4.609 coefficient from U.S. EPA (1985c) to construct the equation:

$$\Delta BP = 4.609 \ (\ln PbB_1 - PbB_2) \qquad (2\text{-}1)$$

where:

ΔBP = the change in blood pressure expected to result from a change in blood lead as a result of regulatory controls,

PbB_1 = blood lead level before regulatory controls, and

PbB_2 = blood lead level after regulatory controls.

Since the model assumes that blood pressure is linear with respect to all other explanatory variables, no error is introduced if the multi-variable model of U.S. EPA (1985c) is reduced to an equation in two variables.

For the present analysis, the site-adjusted coefficient of 2.74 is used (from U.S. EPA 1986c) to predict changes in blood pressure with respect to changes in blood lead levels. Results from Schwartz' more recent work with the random effects model to adjust for site variation yielded a somewhat higher coefficient, but that coefficient is not listed explicitly in Schwartz (1988) and has not been used in this analysis. The equation for estimating changes in diastolic blood pressure as a result of changes in blood lead levels is described by the equation:

$$\Delta BP = 2.74 \, (\ln PbB_1 - \ln PbB_2) \tag{2-2}$$

Using Shifts in Blood Pressure to Estimate Changes in Rates of CHD Events. Once changes in an exposed population's mean blood pressure have been predicted, results from the Pooling Project and Framingham studies can be used to predict changes in other health endpoints. In Pirkle *et al.* (1985), U.S. EPA (1985c), and U.S. EPA (1986d), regression results described above were used to shift the expected blood pressure for each individual observation of middle aged males in the NHANES II survey. Using these shifted estimates for blood pressure along with each observation's data for smoking, age, and serum cholesterol from NHANES II, the authors applied logistic models from the Pooling Project and Framingham studies to predict each sample individual's odds of a heart attack, stroke, or death. By aggregating results, they predicted changes in expected rates of these heath endpoints for the population as a whole.

U.S. EPA (1987c) offers a similar, but simplified technique for performing these calculations. They begin by applying Equation 2-1 to predict a shift in the mean diastolic blood pressure of an exposed population. Next, they used a simplified version of the multiple logistic regression model to derive a simplified univariate equation describing the probability of a CHD event as a function of diastolic blood pressure. As with the logistic equation for hypertension, OAQPS substitutes population mean values from the Pooling Project data into the multi-variable models based on these studies to derive a substitute equation of a single explanatory variable: diastolic blood pressure. For first coronary heart disease events, the resulting equation is:

$$\Delta Pr(CHD) = [1 + \exp(-(-4.996 + 0.030365 \, BP_1))]^{-1} -$$
$$[1 + \exp(-(-4.996 + 0.030365 \, BP_2))]^{-1} \tag{2-3}$$

where:

$\Delta Pr(CHD)$	=	the change in the probability of occurrence of a CHD event,
BP_1	=	mean diastolic blood pressure before regulatory controls, and
BP_2	=	mean diastolic blood pressure after controls.

By combining Equation 2-3 with Equation 2-1, OAQPS estimates changes in the rates of first cardiac events (over a 10 year follow up period) based on changes in population mean blood lead levels.

To what extent does this simplification introduce error into the estimation process? Since the logistic equation used to predict first CHD events is non-linear with respect to age, serum cholesterol and smoking, the substitution of mean values for these variables will not necessarily result in an equation equivalent to the one developed from the Pooling Project data. To evaluate the possible error introduced by this procedure, we compared dose response curves generated by Pirkle et al. (1985) to results generated with the OAQPS methodology (using the 3.954 coefficient from Pirkle et al. for blood pressure versus the log of blood lead). From this comparison, it seems that results from the two methods are quite similar, suggesting that the practical advantages of the OAQPS methodology outweigh any losses to accuracy introduced by this analytic step.

Therefore, the OAQPS methodology was adopted with one modification: to estimate shifts in diastolic pressure, Equation 2-2 is substituted for Equation 2-1. Annual cases are predicted by dividing the results by 10 years to derive a dose-response curve. Using this function, the number of additional CHD events expected to result from exposure to lead in sludge is estimated.

One limitation to the use of Pooling Project data to predict rates of CHD events is that the regression models from those studies were restricted to white middle-aged men (about a third of the entire adult male population). If logistic regression coefficients were available for a broader population of adult males, then blood pressure - blood lead regression coefficients derived by Schwartz (1988) for males 20-74 could be used to estimate changes in rates of CHD events for this larger population. Recent research suggests that the same relationship holds for black men as well.[2] For this analysis, changes in rates of CHD events are predicted for all males of ages 40-59. The estimates ignore reductions in rates of CHD events that might result from men of other age groups.

Reduced Incidence of Stroke. U.S. EPA (1987c) used similar methods to derive univariate equations relating the incidence of stroke and death from all causes to changes in population mean blood lead levels (through changes in diastolic blood pressure). For strokes, they use results from Shurtleff (1974) as listed in Tables 2-4 and 2-5. These univariate logistic equations are applied to estimates of shifts in population mean blood pressure (derived with Equation 2-2) to yield changes in rates of stroke as a result of changes in lead exposure. For estimates of effects of reduced lead exposure on rates of stroke, the OAQPS methodology is modified only slightly, by using a site-adjusted coefficient (Equation 2-3) to estimate changes in blood pressure. Combining the regression equations of Shurtleff (1974) with Equation 2-3 yields the dose-response curve used in the current analysis. This function is entered into the exposure model to estimate changes in the incidence of stroke due to lead in sewage sludge.

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[2]See Schwartz (1988).

Table 2-4

Logistic Regression Relating Blood Pressure
to the Probability of Initial
Cerebrovascular Accident in White Men Aged 45-74

Variable	Coefficient	t-Statistic
Constant	-8.58889	*
Diastolic Blood Pressure	0.04066	5.72

* Not reported

Source: Shurtleff (1974), as taken from U.S. EPA (1987c). p. 6-13.

Table 2-5

Logistic Regression Relating Blood Pressure
to the Probability of Initial
Atherothrombotic Brain Infarction in White Men Aged 45-74

Variable	Coefficient	t-Statistic
Constant	-9.95160	*
Diastolic Blood Pressure	0.04840	5.16

* Not reported

Source: Shurtleff (1974), as taken from U.S. EPA (1987c). p. 6-13.

Reduced Incidence of Premature Death. Data from the Framingham Study have also been used to estimate the relationship between blood pressure and rates of death from all causes. Pirkle *et al.* (1985), U.S. EPA (1985c), and U.S. EPA (1986b) use logistic equation coefficients from the Framingham Study to predict changes in the rates of death as a function of changes in blood lead levels.

U.S. EPA (1987c) used population mean values for serum cholesterol and smoking to reduce results from the Framingham Study to an equation in one explanatory variable:

$$\Delta Pr(MRT) = [1 + \exp(-(-5.3158 + 0.03516(BP_1)))]^{-1} -$$

$$[1 + \exp(-(-5.3158 + 0.03516(BP_2)))]^{-1}$$

(2-4)

where:

$\Delta Pr(MRT)$	=	the change in probability of death,
BP_1	=	the level of diastolic blood pressure before some change, and
BP_2	=	the level of diastolic blood pressure after some change.

As with estimates for CHD, dose-response curves generated by this simplified equation are compared with those using the multi-variable model and individual data observations as reported in Pirkle *et al.* Once again, the results compare favorably with curves reported from the original study (if a coefficient of 3.954 is substituted into Equation 2-1). For estimates of death caused by lead in sludge, therefore Equation 2-2 is combined with Equation 2-4 to generate the dose-response curve shown in Figure 2-2.

Health Benefits for Women. None of the methods outlined thus far includes consideration of possible health consequences of women's exposure to lead in sludge. Nevertheless, at least some available evidence suggests the possibility of such benefits. Recent expanded analysis of NHANES II by Schwartz (1990) indicates a significant association between blood pressure and blood lead in women. Another study, by Rabinowitz *et al.* (1987), has found a small but demonstrable association between maternal blood lead and pregnancy hypertension and blood pressure at time of delivery. Finally, a recent study of NHANES II data by Silbergeld *et al.* (1988) suggests that accumulated lead stores in the bone tissues of women may be mobilized into blood during conditions of bone demineralization associated with pregnancy, lactation and osteoporosis. The authors note that "lead may interact with other factors in the course of postmenopausal osteoporosis, to aggravate the course of the disease, since lead is known to inhibit activation of vitamin D, uptake of dietary calcium, and several regulatory aspects of bone cell function." No quantitative relationship has yet been established, however, between lead stores in women and postmenopausal health endpoints. For lack of sufficient data to quantify these and other potential impacts of lead exposure on women's health, this analysis does not attempt to quantity health benefits from reductions in women's exposure to lead.

2.2 ESTIMATING HEALTH EFFECTS FROM CADMIUM

Methods for estimating health risks from cadmium exposure are outlined in U.S. EPA (1985a). In that document, human health effects from cadmium exposure are described as follows:

> Damage to the kidney's ability to reabsorb blood protein is the known (non-carcinogenic) effect having the lowest exposure threshold. Increasing degrees of cadmium induced renal tubular dysfunction are manifested in β_2-microglobulin proteinuria, general proteinuria, aminoaciduria, and glycosuria, in order of increasing severity. Effects on bone and mineral metabolism have accompanied kidney damage in sever (sic) cases as found in the Itai-Itai or "Ouch-Ouch" disease in Japan (U.S. EPA 1979, 1980).

> Elevated β_2-microglobulin excretion is not equivalent to clinically significant proteinuria. Without continued high exposure to cadmium, there is little evidence of either a progression of severity of kidney dysfunction, or a significant shortening of life expectancy. Nevertheless, while some elevation of β_2-microglobulin excretion appears to be a relatively benign condition, it is usually taken as the threshold health effect in setting ambient cadmium criteria (U.S. EPA 1979, 1980).

As with lead, data are available describing potential health effects as a function of the burden of cadmium accumulated in an individual's body. Unlike lead, the depuration rate of cadmium is relatively slow, with a half life of 18-38 years, and the commonly used measure of cadmium accumulation is the concentration in an individuals's kidney cortex. U.S. EPA (1985a) reports that 25 percent of inhaled cadmium enters the blood stream, as compared to 5-6 percent of ingested cadmium. The authors continue:

> β_2-Microglobulin proteinuria may occur when the concentration in kidney cortex reaches approximately 200-400 $\mu g/g$ (wet weight), although individual susceptibilities may fall outside this range. At the present time a kidney concentration of 200 $\mu g/g$ is the most widely accepted estimated of the critical threshold (Ryan *et al.*, 1982).

> This concentration is estimated to result from a daily retention (absorption) rate of 10-15 $\mu g/day$ over a 25-50 year period (U.S. EPA 1979, 1980). If 12 $\mu g/day$ is taken as the absorbed dose that will produce a kidney cadmium level of 200 $\mu g/day$ over a 25-50 year period, then at a 6 percent absorption efficiency this corresponds to a gross ingestion of 200 $\mu g/day$, a value sometimes cited as a threshold (Commission of the European Communities, 1978).

As with lead, the present analysis combines information describing background levels of tissue concentration, estimated exposure from sludge, and estimated intake slopes to calculate the extent to which the background distribution of a population's burden of cadmium will shift

as a result of additional exposure to cadmium from sludge. Given this estimated shift, the number of persons crossing a specified kidney cadmium "threshold" ($200\mu g/g$) is used to represent the number of persons at risk from sludge disposal.

Based on the information quoted above, 200 μg/day of cadmium ingested over 25-50 years results in an eventual kidney cortex cadmium level of about 200 $\mu g/g$. If the relationship between kidney cortex concentration and ingested cadmium is approximately linear, then it can be reasoned that each μg/day of lifetime average cadmium ingestion regulates in about 1 $\mu g/g$ in eventual tissue concentration. Since only 5-6 percent of the ingested cadmium is absorbed, this means that each μg/day of absorbed cadmium contributes about 16-20 $\mu g/g$ to the eventual burden. These estimates were used to translate incremental exposure to cadmium from sludge into estimated average increments in kidney cadmium concentrations for the exposed populations.

An estimate of the distribution of the background concentration of cadmium in tissues was obtained from U.S. EPA (1985a). This report describes autopsy studies of 93 men, all more than 30 years in age, that revealed an approximately log-normal distribution of kidney cadmium concentrations, with the following characteristics:

Category	Number	Geometric Mean ($\mu g/g$) [95% Confidence Range]	Geometric Standard Deviation ($\mu g/g$) [95% Confidence Range]
Nonsmokers	21	15.0 [11.7-19.3]	1.74 [1.53-2.19]
Smokers	72	27.9 [25.0-31.2]	1.60 [1.50-1.75]
Combined	93	24.2 [21.6-27.1]	1.73 [1.61-1.89]

The men had no known occupational exposure to cadmium.

If it is assumed that exposure to the additional quantities of cadmium from sludge does not appreciably alter the geometric standard deviation of kidney cadmium concentrations in the exposed population, then the number of persons exceeding the 200 $\mu g/g$ threshold of potential health effects can be calculated before and after exposure to cadmium from sludge, and the results subtracted to yield possible health risks from sludge disposal.

For this report, potential health risks were estimated separately for smokers and nonsmokers. For exposure from distribution and marketing of sludge, for which children were modeled separately from adults, accumulated cadmium from the time-weighted childhood exposure was added to that from adult exposure to approximate the eventual total cadmium burden of an adult with 25-50 years of exposure. Absorption efficiency for children (through ingestion) was assumed to be the same as for adults. Since data were not available to describe the background distribution of kidney cadmium concentrations in adult women, it was assumed that their distribution can be approximated by the study of adult men; the results reported in the U.S. EPA (1985a) were used to estimate potential health risks for both men and women combined. To the extent that these data under- or overestimate kidney cadmium concentrations for women or younger men, these results may under- or over-estimate potential health risks.

It should also be noted that the counts of person crossing kidney cadmium thresholds as presented in this report do not necessarily reflect the number of persons expected to experience adverse health effects from cadmium exposure, since not all individuals exceed the threshold can be expected to suffer health effects. Nevertheless, the results should provide a rough estimate of the number of persons at risk of adverse health effects.

3. INCINERATION

3.0 INTRODUCTION

This chapter describes the methodology and data used to estimate human health risks from the incineration of sludge. It reports our estimates of risks under current ("baseline") conditions, and our estimates of the risk reductions likely to be achieved by the regulation. Only direct exposure through inhalation of pollutants is considered; risks from indirect exposure through the deposition of pollutants onto soil, crops, or surface water bodies have not been examined. Also excluded from the analysis are potential risks from the disposal of incinerator ash. Finally, this analysis considers human health risks only, and does not account for potential adverse effects on plants and animal life.

Section 3.1 will discuss the mathematical models and other methods used to calculate human health risks from the incineration of municipal sewage sludge. Section 3.2 will discuss the sources of data used for this analysis, and will provide values for the input parameters used for the modeling. Section 3.3 will provide estimates of health risks before implementation of regulatory controls, and estimates of the health benefits to be achieved by regulating the incineration of sewage sludge.

3.1 METHODOLOGY

This analysis uses four steps to estimate baseline risks from incineration of sludge:

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

(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.

Based on assumptions about reductions in emissions from regulatory controls, these four steps are repeated to estimate human health risks after the regulation is in place. The difference between the two estimates describes the health benefits to be achieved by the regulation.

As discussed in Chapter 1, the basic strategy of this aggregate risk analysis is to assume that the stratified sample of POTWs in the analytic component of the National Sewage Sludge Survey (NSSS) can be used to represent the full inventory of POTWs in the U.S. In general, aggregate risks estimated for this sample of plants are scaled by sample weights to derive estimates of risk at the national level. For incineration, we modify this approach slightly to

exploit additional data available from other sources, and to account for the fact that some populations will be exposed to emissions from more than one incinerator.

3.1.1 Estimating Emissions of Pollutants

The first step in estimating human exposure and risk is to determine the rate at which pollutants are emitted from the stacks of incinerator facilities. The method for calculating emissions differs by type of pollutant. For metals, it is based on the mass of pollutant entering the incinerator, the removal efficiency of the furnace, and any operating pollution control devices:

$$E_{jp} = 3.17 \times 10^{-8} \; C_j \; M_p \; (1 - R_{jp})$$

where:

E_{jp}	=	emission rate for contaminant j at incinerator facility p (g/sec),
j	=	index for pollutants,
p	=	index for incinerator facilities,
3.17×10^{-8}	=	constant to convert units from (g/yr) to (g/sec),
C_j	=	concentration of metal j in sludge (grams per dry metric ton, or g/DMT),
M_p	=	mass of sludge incinerated at facility p each year (DMT/yr), and
R_{jp}	=	combined removal efficiency for pollutant j of furnace and control devices for incinerator p expressed as fraction of original contaminant retained by the furnace or pollution control devices (dimensionless).

To calculate the average rate that metals are emitted from each facility, we first calculate the rate at which a contaminant enters the facility, based on the feed rate for sludge (M_p) and the concentration of contaminant in the feed (C_j). For a given mass of a metal contaminant 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 contaminant 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 facilities represent stack emissions in units of grams per second for each contaminant (E_{jp}).

For organic pollutants, predicting rates of emission is more complicated, because organic pollutants from sludge can be destroyed within the incinerator, and other organic compounds can be formed in the incinerator as products of incomplete combustion (PICs). Emission rates for organic pollutants are therefore estimated directly from results of monitoring studies for sludge incinerators. Emissions are assumed to be determined by the type of furnace used, the quantity of sludge incinerated, and the use of pollution control devices:

$$E_{jp} = 3.17 \times 10^{-8} \, O_{jp} \, M_p \, (1 - R_{jp})$$

where:

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

3.17×10^{-8} = constant to convert units from (yr^{-1}) to (sec^{-1}),

O_{jp} = emission rate for organic pollutant j for a unit sludge feed rate at a facility of the same type as facility p (g/sec per DMT/sec),

M_p = mass of sludge incinerated per year at facility p (DMT/yr), and

R_{jp} = removal efficiency of control for pollutant j for facility of same type as facility p, expressed as the fraction of pollutant retained by the control device (dimensionless).

The mass feed rate of sludge (M_p) (converted to units of DMT per second) is multiplied by the unit emission rate for the appropriate furnace type (O_{jp}) (estimated with methods to be described in Section 3.2) to obtain estimated emissions in grams per second for each pollutant and each facility. Estimated emissions are reduced proportionately to adjust for the effectiveness of pollution control devices operating at each facility (R_{jp}). We assume organic emissions are reduced only by afterburners: other pollution control devices have negligible impact on organic emissions. Note that, unlike the calculation for metals, concentrations of organic contaminants in the sludge are not considered when estimating expected emissions of organic pollutants.

3.1.2 Modeling the Dispersion of Pollutants in Air

Dispersion of pollutants in air is simulated with the Industrial Source Complex Long Term (ISCLT) model (Bowers *et al.*, 1980; U.S. EPA, 1986j) as implemented in the current version of the Graphical Exposure Modeling System for personal computers, or PC-GEMS (U.S. EPA, 1989d). The model describes the dispersion of pollutants as steady-state, Gaussian plumes, and allows the user several modeling options. Aerodynamic downwash can be considered if nearby building dimensions are known. Plume rise can be predicted as a function of distance. Chemical degradation during transport can also be considered in the calculations. Emissions can be classified as originating from either point, area or volume sources.

The ISCLT model requires several parameters describing the gas plume as it exits the incinerator stack, as well as further information to determine how the plume is affected by surrounding buildings and terrain. The height and inner diameter of the stack are required inputs, and the velocity and temperature of exit gases must also be specified. The height and effective width (square root of the area) of the building nearest the stack are needed to evaluate the effects of potential downwash on the plume.

We model all incinerator stacks as point sources. Depending on the velocity and temperature of exit gases, plume rise is modeled as either momentum- or buoyancy-induced; the appropriate option is selected automatically by the program. We invoke both the downwash and plume-rise-by-distance options, but (for lack of sufficient data) ignore the effects of surrounding terrain. We conservatively assume that pollutants do not degrade significantly during air

transport before reaching human receptors. This last assumption simplifies our calculations, because transport modeling for different pollutants is identical if decay is not considered. For computational efficiency, the dispersion of pollutants near each facility is modeled only once, using a unit rate of emissions (i.e., 1 g/sec of pollutant emitted per kg/sec of sludge incinerated). Resulting dispersion estimates are converted to ground-level concentrations at individual locations, scaled by expected emissions of each pollutant from each individual facility.

3.1.3 Mapping Dispersion and Pollutant Concentrations Onto a Unified Grid

Results from the ISCLT model are reported as dispersion ratios in units of $\mu g/m^3$ of pollutant concentration in ambient air per g/sec of emissions. Separate coefficients are provided for selected locations in the area surrounding an individual incinerator facility. The model allows the user to choose between a rectangular or polar grid for specifying these locations. We select the rectangular grid and specify coordinates in such a way that results from the modeling of individual incinerators can be integrated into a unified mapping of dispersion ratios for the U.S. as a whole.

For non-carcinogenic, threshold-acting pollutants, the potential overlapping of plumes from multiple incinerators can affect the characterization of health risks to the exposed population. In other words, although pollution from a single incinerator might not be sufficient to cause an exposed individual to exceed a risk reference dose, additional pollution from another (nearby) sludge incinerator might raise total exposure to a level exceeding the RfD. Moreover, health effect relationships established for lead and cadmium are non-linear with respect to dose. For these pollutants, summing estimated health effects from individual incinerators in the national inventory would not necessarily provide an accurate estimate of health risks from all incinerators operating simultaneously.

To account for potential overlapping of pollutant plumes, we construct a unified grid system onto which results from the modeling of individual incinerators can be integrated. One desirable characteristic for such a grid is that the cells of overlapping grids of neighboring facilities be perfectly aligned. A simple and intuitive coordinate system for that grid would be to use fractions of degrees latitude and longitude to describe nodes in the grid system. The latitude and longitude of most incinerating POTWs is recorded in data from the 1986 NEEDS survey, and PC-GEMS is designed to access meteorological and population data automatically based on latitude and longitude coordinates. However, a complication with relying exclusively on this coordinate system is that the distance represented by a single degree of longitude decreases with increasing latitude; if the same dimensions in units of latitude and longitude are used to define the area modeled near each incinerator, the widths of areas modeled for facilities in the northern U.S. will be significantly smaller than the widths of corresponding areas for southern facilities. If too small an area is modeled for a particular facility, risks may be under-counted. Conversely, if the grid spacing is too large, the model might fail to resolve rapid changes in pollutant concentrations near incinerators.

To achieve consistent dimensions while aligning grid cells into an integrated system, we construct a transformed, two-dimensional grid system to approximate the locations of

incinerators and human populations. Any transformation from mapping on a sphere to mapping on a flat surface produces distortion. Our goal has been to derive a simple mapping scheme that holds this distortion to levels of little significance for model calculations. First, the latitude and longitude coordinates of each facility are converted to spherical coordinates in a system with its origin at the center of the earth (assumed to be a perfect sphere of radius 6452 km). Second, the spherical coordinates are transformed to a three-dimensional cartesian grid, maintaining exact locations. Finally, positions of each incinerator are approximated with a two-dimensional (2-D) grid. The origin of the new grid is chosen to be the approximate center of the continental U.S. (latitude: 37, longitude: 96). Because the distortion inherent in this step is greatest at large distances from the origin, setting the origin at the grid center helps reduce maximum error.

To determine coordinates in the 2-D system, exact distances are first calculated along arcs on the earth's surface between each facility and the origin of the 2-D system. Positions on the 2-D grid are selected so these distances remain exact. A facility's y-coordinate in the new system, which represents distance in the North-South direction, is defined by the facility's angle of latitude with respect to the 2-D origin. At all locations, one degree of latitude is equivalent to approximately 113 km. This y-coordinate is used together with the true distance between the facility and the origin to determine the x-coordinate, which represents distance in the East-West direction. Distances between facilities as calculated based on the resulting 2-D grid consistently fall within a few percent of true distances along the surface of the earth, with agreement improving as distances decrease.

Within this coordinate system, we use two sizes of grid cell to cover sufficient area while providing sufficient detail at small distances from each plant. As shown in Figure 3-1, we define large grid cells with dimensions of 4 km by 4 km. For simplicity, modeled locations for plants are adjusted by a maximum of 2 km to the center of a grid cell. The grid system extends for ten cells in each direction from the facility, for a total of 84 km in both width and length (42 km from the center to each edge of the grid, and approximately 59 km from the incinerator to each corner). Because greater refinement is desired closer to the incinerator (where concentrations are expected to be greatest), each large grid cell containing a facility is further divided into 400 small grid cells (each of dimensions 200 x 200 m) with the incinerator located at the central node (as shown in Figure 3-2). After slightly adjusting the latitude and longitude coordinates of each actual facility and specifying receptor locations according to the grid system, we use ISCLT to calculate dispersion ratios appropriate for the center of each cell in both the large and small grid systems. The only location for which a dispersion ratio is not calculated is the center of the large grid system, which represents the location of the incinerator. Within the small grid system, receptor locations are specified at distances from 100-1900 m at intervals of 200. Within the larger system, they are specified at distances from 4-40 km at intervals of 4 km. Two separate executions of ISCLT are necessary to calculate dispersion ratios for the receptor locations in the small and large grid systems for each facility.

The next step is to calculate expected concentrations of each pollutant within each cell of the grid system. To do so, we combine estimates of emissions from each facility with results from ISCLT to calculate each incinerator's contribution to pollution within surrounding cells. Where a cell is included in the grid system for more than one incinerator, results for each pollutant are summed to calculate total expected concentrations. The total concentration

Figure 3-1
Large Grid For Modeling Incinerators

84 km

4 km
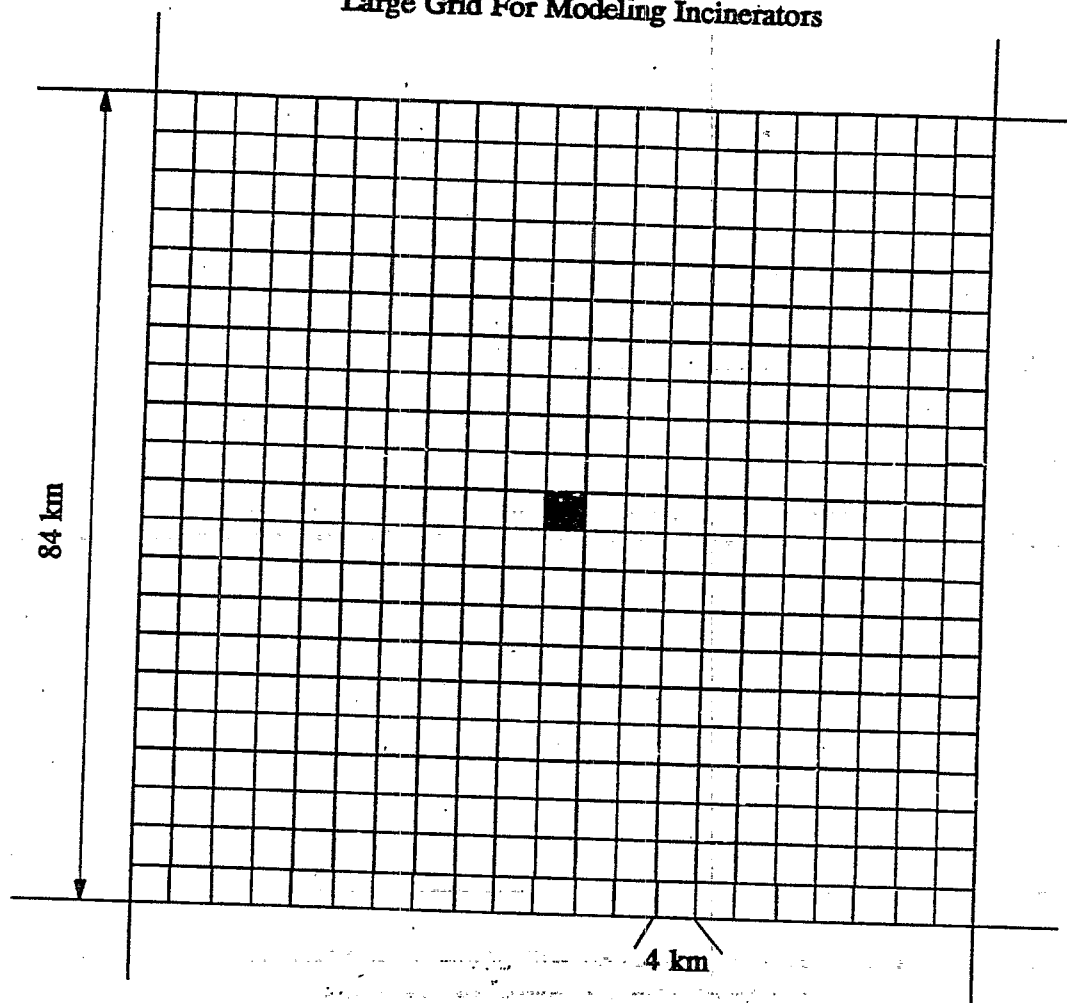
Figure 3-2
Facility Location in Small Grid
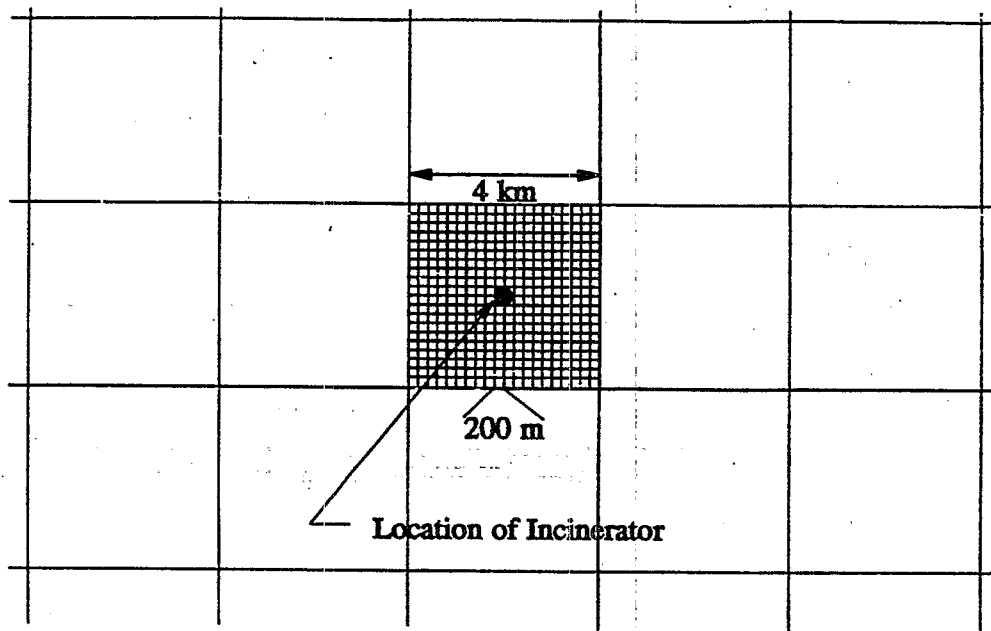
estimated in each cell for each pollutant is described by:

$$A_{ij} = \sum_{p=1}^{N} D_{ip} \, E_{jp}$$

where:

A_{ij}	=	ambient air concentration of pollutant j in cell i due to emissions from all facilities impacting that cell ($\mu g/m^3$),
i	=	index for grid cells,
j	=	index for pollutants,
p	=	index for incinerator facilities,
N	=	number of incinerator facilities modeled,
D_{ip}	=	dispersion ratio for cell i from facility p ($\mu g/m^3$ per g/sec), and
E_{jp}	=	emission rate for pollutant j from facility p (g/sec).

When a particular grid cell i falls outside the range of a particular facility p, D_{ip} is set to zero.

3.1.4 Estimating Human Exposure and Risk

Once the concentrations of each pollutant in ambient air near each incinerator have been estimated, the next step is to determine the extent of human exposure and risk. We begin by estimating the size of the human population residing in each cell of the grid system discussed above. This step is accomplished with the Master Area Resource File (MARF) data base included in PC-GEMS. MARF contains population data from the 1980 Census, mapped at the enumeration district, block group level (ED/BG) by locations in latitude and longitude. We convert the location of each ED/BG centroid from units of latitude and longitude into x- and y- coordinates of the 2-D coordinate system described above, and assign appropriate populations to each grid cell assigned a dispersion ratio.

For the larger, 4 x 4 km grid cells, several ED/BG observations from MARF typically fall within each grid cell. The corresponding populations are summed to calculate the total population of the grid cell. In cases where no observation from MARF falls within a large grid cell, that cell is ignored for calculations of aggregate risk. For smaller 200 x 200 m grid cells, the "lumpiness" of Census data creates complications for assigning populations to grid cells. Within MARF, the full population of each enumeration district or block group is assigned to the centroid of the appropriate area. For the small grid, such assignments can mean that certain populated cells in the grid will be assigned zero populations, while populations in neighboring cells will be assigned over-stated populations. To avoid undue influence from this aggregation of populations in Census data, we "smooth" estimates of populations within the small grid, based on the total density of human populations within the 4 x 4 km boundary of the small system. We combine all populations whose ED/BG centroid fall within the large grid cell containing an incinerator facility to calculate the total population for the large cells containing the small grid. This total is then divided by 400 to derive an estimated population for each cell in the small grid around the facility. In rare cases where no centroid falls within the large cell at the center of the grid system, we conservatively calculate a density from large grid cells within range of the

plant. This average population is then assigned to the large cell containing the facility. As before, this population is divided equally among the small cells. This adjustment results in a slight over-counting of exposed populations, but its effect on aggregate risk estimates is believed to be insignificant.

We generate an estimate of human exposure for each cell by combining estimated concentrations of a pollutant with assumptions about daily inhalation volume, body weight, and background intake of the pollutant from other sources. As mentioned earlier, we assign populations from the 1980 Census to each of the 840 grid cells for which dispersion ratios are estimated near each incinerator. For carcinogens, only incremental cancer risks from sludge incineration are of concern in this study; therefore, background pollutant levels are not considered in assessing carcinogenic risk. However, background intakes for non-carcinogenic pollutants are important for estimating non-carcinogenic risks, since background levels contribute to total exposure, the exposure measure of interest when evaluating the possibility of exceeding a threshold. Individual exposure to each pollutant in each cell is calculated as:

$$EXP_{ij} = \frac{A_{ij} \, I_a \, 10^{-3}}{BW} + BI_j$$

where:

EXP_{ij}	=	exposure to pollutant j for individuals living in grid cell i (mg/kg-day),
A_{ij}	=	estimated average concentration of pollutant j in grid cell i, including contributions from all relevant incinerators and excluding contributions from other sources ($\mu g/m^3$);
I_a	=	inhalation volume (m^3/day),
10^{-3}	=	constant to convert units from μg to mg,
BI_j	=	background intake for pollutant j, set to zero for carcinogens (mg/kg-day), and
BW	=	average body weight (kg).

As can be seen from the equation, we conservatively assume that each person residing in a given grid cell inhales air at the estimated (outdoor) concentration for 24 hours per day for his or her entire lifetime. We also assume that all of the inhaled pollutant is absorbed into the body, and thus exposure is effectively equivalent to dose.

The final step in estimating human health risk is to combine estimates of exposure and population with pollutant-specific dose-response relationships. For carcinogenic compounds, the risk to an exposed individual is expressed as the probability the individual will develop cancer within his or her lifetime as a result of the incineration of sewage sludge. For an exposed individual in cell i, we calculate the incremental risk of cancer as:

$$CI_i = \sum_{j=1}^{M} EXP_{ij} \, q_j^*$$

where:

$$CI_i \quad = \quad \text{individual cancer risk for individual in cell } i \text{ (probability of developing}$$
cancer from a lifetime exposure),

$M \quad = \quad$ number of pollutants, and

$q_j^* \quad = \quad$ human cancer potency for pollutant j (mg/kg-day)$^{-1}$.

The HEI is defined as the individual for whom this sum is highest:

$$CI_{max} = \underset{i}{MAX} \; CI_i$$

where:

$CI_{max} \quad = \quad$ maximum individual risk (incremental risk of developing cancer within lifetime).

Aggregate risk is defined as the number of incremental cancer cases resulting annually from exposure to incinerator emissions. Results from individual grid cells and individual pollutants are summed to calculate total cancer risks from all pollutants combined:

$$CP = \sum_{i=1}^{N} CI_i \; POP_i \; / \; LS$$

where:

$CP \quad = \quad$ total cancer risk from all pollutants across all cells (expected cases per year),

$POP_i \quad = \quad$ human population living in cell i (persons),

$LS \quad = \quad$ lifespan of average individual (yr), and

$N \quad = \quad$ number of cells.

Health risks from non-carcinogenic pollutants are expressed as comparisons between exposure and risk reference doses (RfDs). An individual's risk from exposure to a given pollutant is expressed as the percent of the RfD to which he or she is exposed. A percent greater than 100% indicates that the reference dose is exceeded. For non-carcinogenic pollutants, the HEI for each pollutant is defined as the individual for whom the ratio of exposure to RfD is highest. Aggregate non-carcinogenic risk is expressed as the total number of persons exceeding the reference concentration for a given pollutant:

$$P_{ij} = POP_i \qquad (EXP_{ij} \geq RfD_j)$$
$$P_{ij} = 0 \qquad (EXP_{ij} < RfD_j)$$

and:

$$NCP_j = \sum_{i=1}^{N} P_{ij}$$

where:

P_{ij} =   number of persons in cell i exceeding RfD for pollutant j,

RfD_j =   risk reference dose for pollutant j, and

NCP_j =   number of people exceeding the RfD for pollutant j.

3.1.5 Estimating Benefits from Regulation

To estimate benefits from the regulation, we first estimate risks under baseline (pre-regulatory) conditions. We next determine which pollution control devices will be installed at particular POTWs. We then estimate risks under the new conditions, with estimated emissions adjusted as appropriate for the presence of additional controls. However, the decision to install additional controls is influenced by the concentrations of metals in the sludge incinerated at a particular facility, and unfortunately, data describing those concentrations are not available for this analysis except for those 23 incinerators included in the analytic survey of the NSSS. To model behavior for the larger set of 172 POTWs practicing incineration would require data for the concentrations of metals in the remaining 149 facilities. In the absence of such data, we use the following six steps to estimate risks after installation of additional pollution controls and subsequently to estimate benefits from the regulation:

(1)   Estimate exposure and risk under current conditions for the 23 incinerating POTWs in the analytic component of the NSSS, using known quantities of sludge and concentrations of metals for each facility, together with estimated emissions of organic pollutants.

(2)   Estimate exposure and risk under baseline conditions for the larger set of 172 known incinerators, after assigning average concentrations of metals to the sludges burned in these incinerators based on size category, and after scaling estimates of sludge volumes (from U.S. EPA, 1989g) so that the total sludge volume matches the total estimated from the NSSS after scaling with sample weights.

(3)   Calculate the ratio of estimates derived in Step (2) and Step (1) to yield a scaling factor that includes the effects of the increased volume of sludge incinerated, meteorology, population densities, types of furnaces and controls, and overlapping plumes from incinerators not included in the analytical component of the NSSS.

(4)   Estimate exposure and risk from the 23 incinerating POTWs in the analytic component of the NSSS, after installation of pollution control devices in response to regulatory controls.

(5)   Scale the estimate derived in Step (4) by the ratio derived in Step (3) to estimate expected aggregate risks after the regulation.

(6)   Subtract results of Step (5) from results of Step (2) to estimate the reduction in health risks to be achieved by the regulation.

These six steps provide estimates of aggregate risks both before and after implementation of regulatory controls. Estimates are based on sludge concentrations and volumes from those POTWs in the analytic component of the NSSS, and have been scaled to accommodate both the sample weights of that survey, and all available site-specific data for incinerators in the larger universe of POTWs.

3.2 DATA SOURCES AND MODEL INPUTS

Mathematical modeling of exposure and risk from the incineration of municipal sewage sludge requires data describing characteristics of the incinerators, the quantity and quality of sludge incinerated, local meteorology, the locations of human populations, and dose-response functions for pollutants of concern. This section provides the sources we have used to obtain these data, and provides tables for key input parameters used for the models.

3.2.1 National Sewage Sludge Survey

Our only source of data for current concentrations of contaminants in the sludge of individual POTWs is the National Sewage Sludge Survey (NSSS). The survey's analytic component includes a sampling of sludge from each POTW in a stratified sample of 25 facilities reporting the use of incineration as a sludge disposal method, and includes measurements of the concentration of 10 metals and over 400 organic analytes. The final analytic data set includes only 23 facilities, because sludge from two of the original 25 POTWs is incinerated off-site. Of particular interest for this analysis are the measured concentrations of metals; concentrations of organic contaminants are not known to correlate with emissions from incinerator stacks.

For 63 POTWs responding to the questionnaire (including those in the analytic survey), the NSSS also provides data for the quantity of sludge incinerated and certain characteristics of the individual incinerator facilities, including the types of furnace and pollution control devices in use.

3.2.2 Other Sources of Facility-Specific Data

Additional site-specific data are available from an earlier analysis of aggregate risks from the use and disposal of municipal sewage sludge (U.S. EPA, 1989g). That analysis used data from the 1986 NEEDS Survey and other sources to identify 169 POTWs using incineration to manage their sludge. Names, facility identification numbers, and latitude and longitude for each POTW were collected, along with data describing the physical characteristics of their incinerator facilities (stack height, etc.). For this analysis, we have supplemented results from that earlier study with data collected through the NSSS, to generate a revised data base that includes information for 172 POTWs using incineration. For those POTWs not included in the NSSS, earlier estimates of the quantities of sludge incinerated have been scaled so that the total quantity of sludge incinerated by the larger set of 172 facilities matches the current estimate of the national total (823,005 DMT/yr) based on data and sample weights from the NSSS. The total

estimated mass of sludge incinerated in each state, together with each state's contribution to the set of 172 facilities, is presented in Table 3-1.

3.2.3 Furnace Types

At least three types of furnaces are used to incinerate sewage sludge: multiple hearth, fluidized bed and electric. Fluidized bed furnaces burn relatively cleanly and tend to operate at a constant temperature, whereas multiple hearth combustors are subject to wider fluctuations. The higher furnace temperatures typical of the fluidized bed combustors tend to improve destruction of organic pollutants, but can also increase the fraction of metals entering the flue. Because applicable data are not available to describe emissions from electric furnaces, we assume for this analysis that emissions from electric furnaces are comparable to those from fluidized bed combustors.

3.2.4 Air Pollution Control Devices

Pollution control devices can be roughly categorized by the mechanism they use to capture pollutants, and by their use of wet or dry removal equipment. The four most common methods of pollution control include: wet scrubbers, dry cyclones and fabric filters, wet electrostatic precipitators, and afterburners.

Wet Scrubbers

Most sludge incinerators in the U.S. use some type of wet scrubbing device to control particulate emissions. Incinerators tend to be located near POTWs, which provide effluent as a readily available and inexpensive supply of feed water for the scrubber. Wet scrubbing devices have demonstrated a long history of success in meeting emission control standards for particulate matter.

A typical example is the venturi scrubber. Water introduced at the throat of the venturi is dispersed by the high velocity gas stream. Pollutants are dissolved or adhere to the drops, which are generally larger than particulates and easier to remove. The efficiency of a venturi scrubber depends on the pressure drop across the throat and the temperature after the venturi. If the temperature is too high, further evaporation occurs and condensation is reduced. Cyclones may be used upstream of the scrubber for better removal of particles with diameters larger than 10 micrometers. The gas stream can then be bubbled through impingement trays filled with water (located downstream of the scrubbers) which further entrain solid particles and increase the removal of pollutants from flue gas.

Table 3-1

Mass of Sludge Incinerated and Number of Incinerators by State

	Number of Facilities in Analytic Survey[a]	Quantity Incinerated in Analytic Survey (DMT/year)[a]	Percent of Total Quantity for Analytic Survey[a]	Number of Facilities in Larger Sample[b]	Quantity Incinerated in Larger Sample[b,c] (DMT/year)	Percent of Total Quantity for Larger Sample[b,c]
Alaska	--	--	--	2	446	0.05%
California	1	11,748	3.30%	9	40,848	4.96%
Connecticut	1	5,297	1.49%	11	19,381	2.35%
Florida	1	4,137	1.16%	1	4,137	0.50%
Georgia	2	182	0.05%	6	6,040	0.73%
Hawaii	--	--	--	2	3,914	0.48%
Indiana	--	--	--	1	11,018	1.34%
Iowa	1	2,786	0.78%	3	13,004	1.58%
Kansas	--	--	--	4	3,124	0.38%
Louisiana	1	277	0.08%	4	10,952	1.33%
Maryland	1	11,500	3.23%	3	13,536	1.64%
Massachusetts	2	5,705	1.60%	7	22,020	2.68%
Michigan	4	152,012	42.69%	14	174,858	21.25%
Minnesota	1	59,874	16.81%	3	68,619	8.34%
Missouri	--	--	--	4	44,640	5.42%
Nebraska	--	--	--	1	4,368	0.53%
Nevada	--	--	--	1	114	0.01%
New Hampshire	--	--	--	2	2,511	0.31%
New Jersey	--	--	--	9	12,640	1.54%
New York	1	3,975	1.12%	26	105,023	12.76%
North Carolina	1	2,268	0.64%	3	3,717	0.45%
Ohio	3	66,998	18.81%	12	139,504	16.95%
Oregon	--	--	--	1	2,060	0.25%
Pennsylvania	2	22,731	6.38%	16	46,333	5.63%
Rhode Island	--	--	--	1	3,308	0.40%
South Carolina	--	--	--	2	4,917	0.60%
Tennessee	--	--	--	5	10,002	1.22%
Virginia	1	6,622	1.86%	11	42,930	5.22%
Washington	--	--	--	4	4,233	0.51%
West Virginia	--	--	--	1	529	0.06%
Wisconsin	--	--	--	3	4,280	0.52%
TOTAL	23	356,110	100.00%	172	823,005	100.00%

[a]Describes 23 POTWs in analytic survey of NSSS. Quantities shown have not been scaled by sample weights; scaled total is 823,005 dry metric tons per year.

[b]Based on list of 172 incinerating POTWs compiled from U.S. EPA (1989g) and NSSS.

[c]Quantities of sludge for plants not included in the NSSS have been estimated from inflow reported in the 1986 NEEDS Survey (U.S. EPA, 1989g). For this analysis, estimates for those excluded plants have been proportionately scaled so that the total quantity of sludge for all facilities matches the total estimated with sample weights from analytic survey of NSSS.

Dry Cyclones and Fabric Filters

Dry removal devices rely on inertial, adhesive or electrical forces. A dry cyclone is typical of an inertial force separator. In the vortex chamber of a cyclone, the gas is set in a rotational pattern so that particles and droplets are separated from the gas by centrifugal action. This process is most effective for the more massive particles and droplets.

A bag, or fabric filter, uses adhesive forces to remove pollutants. Fabric filters use a porous fabric that traps particulates as the gases pass through a series of suspended fabric filter tubes. The collected particulates are periodically shaken from the fabric for accumulation and removal from the bottom of the apparatus. If lime is added to the gases the acids are neutralized, and the fabric filter collects the lime and other particles before the gases leave the stack. The first sewage sludge incinerator with a dry scrubber and fabric filter system has been designed for a POTW in California and should soon be operating.

Wet Electrostatic Precipitators

Wet Electrostatic Precipitators (ESPs) operate by electrically charging particles in the gases as they flow through a chamber. Oppositely charged plates or tubes then attract and collect the particles. The particles are periodically washed off the collection plates and gathered by an ash slurry system. Wet ESPs have been specifically tested as a retrofit option for sludge incinerators. Particles and droplets with sizes as low as approximately 0.1 μm can be collected using a wet ESP (Brauer and Varma, 1981). However, small particles and drops tend to migrate more slowly than larger ones. To capture the lower size ranges, the collection areas must be large and gas velocities low. Of special importance is the effect of temperature on collection efficiency. The electrical resistivity of the particles or droplets determines how readily they migrate to the collection surfaces. Graphs of electrical resistivity against temperature from solid waste incinerator samples show a very pronounced peak occurring between 100 and 200°C (Vancil and White, 1988). Away from the peak resistivity temperature, resistivity falls off rapidly. Moreover, above a resistivity of about 10^{10} ohm-cm, a "back corona" develops, which can significantly reduce migration. Both of these factors affect the optimum temperature for operation of a wet ESP. Results of tests on sludge incinerators show that removal efficiency decreases dramatically as temperatures depart from design temperature.

According to Gerstle and Albrinck (1982), approximate ranges of particle sizes targeted by the pollution control devices just discussed are:

Wet Scrubber	0.05 to 300 μm
Filter	0.05 to 16 μm
Cyclone	4 to 800 μm
Electrostatic Precipitator	0.05 to 12 μm

Particles of fly ash range in size from 1-100 μm.

Afterburners

Afterburners oxidize, in a secondary combustion chamber organic compounds not destroyed during the primary incineration process. The secondary combustion is achieved using gaseous or liquid fuels or by exposing the primary combustion gases to a bed of catalytic agents. A review of twelve studies shows that afterburners remove up to 98% of most trace organic pollutants and 99% of dioxins and furans (Environment Canada, 1986).

3.2.5 Current Inventory of Furnaces and Pollution Controls

Table 3-2 summarizes our assumptions for the types of furnaces and pollution control devices operating at each of the 23 incineration facilities included in the analytic survey of the NSSS. As can be seen from the table, only three of the 23 facilities modeled (13%) use fluidized bed furnaces, and all incinerators except St. Paul's rely on conventional wet scrubbers to control emissions of metals. Only one of the 23 facilities (in Martinez, California) is believed to operate an afterburner currently. Similarly, of the 172 plants in our larger sample, 164 (95 percent) use multiple hearth furnaces and only 8 (5 percent) use fluid bed combustors. Martinez is the only facility known to operate an afterburner.

3.2.6 Expected Response to Regulatory Controls

Established under Section III of the Clean Air Act, New Source Performance Standards (NSPS) require all new sludge incinerators to install scrubbers meeting the National Ambient Air Quality Standard (NAAQS) for particulate matter. Data gathered for individual sludge incinerators show that most facilities already have an air pollution control device (i.e., scrubber system) in place. However, additional pollutant controls might be necessary to comply with the regulation of sludge.

Changes in the sludge feed, the combustion process and the type of pollution control devices installed will affect pollutant emissions. For this analysis, we consider only the retrofitting of pollution controls devices for reducing stack emissions. Baseline modeling for facilities in the analytic data set yields estimates of the maximum concentrations of each pollutant to which a highly exposed individual might be exposed. These maximum concentrations are compared to reference levels specified by the regulatory criteria. Any facilities whose emissions result in unacceptably high concentrations will be required to install additional pollution control devices. For simplicity, we consider only one additional control technology. To reduce metal emissions, a facility could add a wet electrostatic precipitator (ESP) operating at either high or medium efficiency; reduction of organic emissions is addressed by adding an afterburner. Based on a standard of 100 parts per million of total hydrocarbons (THC), EPA predicts that existing incinerators will be able to comply with crtieria for emissions of organic pollutants with minimal changes to current practices. Increased use of afterburners is not expected as a result of the regulation. Table 3-2 shows the pollution control devices expected to be installed by the 23 incinerators in the analytic survey.

Table 3-2

Furnaces and Pollution Control Devices for Incinerators in Analytic Survey
Before and After Regulatory Controls

	Furnace Type	Current Control	Additional Control[*]
Martinez, CA	Multiple Hearth	Wet Scrubber, Afterburner	Wet ESP at High Eff.
Waterbury, CT	Multiple Hearth	Wet Scrubber	None
Pensacola, FL	Multiple Hearth	Wet Scrubber	None
Atlanta, GA	Multiple Hearth	Wet Scrubber	None
Decatur, GA	Multiple Hearth	Wet Scrubber	None
Dubuque, IA	Fluidized Bed	Wet Scrubber	None
Natchitoches, LA	Fluidized Bed	Wet Scrubber	None
Baltimore, MD	Multiple Hearth	Wet Scrubber	Wet ESP at High Eff.
Fall River, MA	Multiple Hearth	Wet Scrubber	None
Fitchburg, MA	Multiple Hearth	Wet Scrubber	Wet ESP at Med. Eff.
Ann Arbor TWP, MI	Multiple Hearth	Wet Scrubber	None
Detroit, MI	Multiple Hearth	Wet Scrubber	Wet ESP at Med. Eff.
Port Huron, MI	Fluidized Bed	Wet Scrubber	None
Wyandotte, MI	Multiple Hearth	Wet Scrubber	Wet ESP at Med. Eff.
St. Paul, MI	Multiple Hearth	Wet Scrubber	Wet ESP at Med. Eff.
Hilton, NY	Multiple Hearth	Wet Scrubber	None
Rocky Mount, NC	Multiple Hearth	Wet Scrubber	None
Cincinnati, OH	Multiple Hearth	Wet Scrubber	Wet ESP at Med. Eff.
Cleveland, OH	Multiple Hearth	Wet Scrubber	Wet ESP at Med. Eff.
Euclid, OH	Multiple Hearth	Wet Scrubber	Wet ESP at Med. Eff.
Apollo, PA	Multiple Hearth	Wet Scrubber	Wet ESP at High Eff.
Pittsburgh, PA	Multiple Hearth	Wet Scrubber	None
Norfolk City, VA	Multiple Hearth	Wet Scrubber	None

[*]Pollution control device to be installed to achieve compliance with regulation.

An additional constraint is that ambient concentrations of lead must meet the lead NAAQS of 1.5 $\mu g/m^3$. According to the modeling performed for this analysis, this concentration is not exceeded by any of the 172 incinerators considered. In fact, the highest concentration simulated under baseline conditions is about 0.14 $\mu g/m^3$, or just under 10 percent of the NAAQS standard.

3.2.7 Data for Estimating Emissions of Inorganic Pollutants

Incinerator emissions depend on the type of furnace operating, the quantity of sludge incinerated and the effectiveness of pollution control devices. For inorganic pollutants, the quantity of sludge incinerated and its concentrations of contaminants determine the mass of pollutants potentially available for emissions. Final emissions are determined by the combustion process and the pollution control devices installed. Characterizing emissions from a variety of combustor and pollution control combinations therefore requires estimates of the rate at which each metal is fed into the incinerator, the metal-specific removal efficiencies of the combustor, and the removal efficiencies for the pollution control devices.

Sludge incinerated at the 23 facilities in the analytic survey of the NSSS was tested for ten metals; results are summarized in Table 3-3. To estimate concentrations for the remaining 149 plants in our larger sample, POTWs in the analytic data set have been divided into four categories based on the volume of wastewater treated per year. Average concentrations of each metal have been determined for the plants in each of these four categories and assigned to comparable size plants not included in the analytic survey. Results are listed in Table 3-4.

To predict emissions of metals for this analysis, we use the average concentrations of metals described in Tables 3-3 and 3-4, together with estimated feed rates for each facility, to predict the mass of each metal entering the incinerator per unit time. We then adjust this mass for the "removal efficiency" achieved by each type of incinerator and control device to predict emissions. Removal efficiencies have been calculated by averaging available test results from incinerator facilities with similar furnaces and control devices.

One important consideration in estimating emissions is that different types of pollution controls target different particle and droplet sizes for optimum collection. The combustion conditions occurring in different furnace types will produce different arrays of particle and droplet sizes. For example, increased air velocities through the combustion bed will increase particle entrainment. Changes in the amount of excess oxygen available will affect the vaporization of volatile metals (U.S. EPA, 1987h). As a result, removal efficiencies depend on particular couplings of furnace and control.

Furnace temperature is also important in determining metal emissions at a given furnace. A study by Gerstle and Albrinck (1982) examined the volatility of metals commonly found in sewage sludge, and compiled data to investigate the relation between furnace temperature and percent of input metals that are emitted to the atmosphere. At conventional temperatures for incineration (760-815°C) arsenic, cadmium, mercury and zinc are volatile. Lead is potentially volatile at temperatures above 980°C. These metals tend to be emitted in a vapor state while

Table 3-3

Metal Concentrations in Sludge Incinerated at Plants in Analytic Survey[a]

Plant	Arsenic (mg/kg)	Beryllium (mg/kg)	Cadmium (mg/kg)	Chromium[b] (mg/kg)	Copper (mg/kg)	Lead (mg/kg)	Mercury (mg/kg)	Molybdenum (mg/kg)	Nickel[c] (mg/kg)	Selenium (mg/kg)	Zinc (mg/kg)
Martinez	3.9	0	3.1	21.4	355	73.2	5.3	0	18.9	4.9	392
Waterbury	0	0	15.5	173	1830	263	0	0	488	7.9	1830
Pensacola	6.1	0	10.2	39	259	166	2	0	15	3.5	732
Atlanta	4.2	0	19.05	174	573	757.5	5.05	20.1	34.65	4.1	3275
Decatur	1.2	0	6.9	154	92.3	91.4	0.6	4.4	25.7	1.3	317
Dubuque	3.6	0	16.8	229	465	522	2	5.7	122	3	5190
Natchitoches	3.05	0	2.8	33.2	199.5	63.15	5.95	3.55	14.5	1.15	567.5
Baltimore	4.1	0	10.6	270	810	90	0.9	5.5	58.6	0	605
Fall River	9.3	0	58.6	56.8	657	1670	2	2.7	63.2	3.7	10700
Fitchburg	9.1	0	3.8	1670	1570	138	2.2	0	15	12.5	443
Ann Arbor	13.2	0	8	82.8	236	27.8	1.2	4.3	28.4	2.2	337
Detroit	0	0	128	483	389	211	1.5	8.3	192	4.1	2120
Port Huron	5.1	0	2.75	55.3	279	162.15	0.95	18.15	29.8	1.7	543.5
Wyandotte	14.9	0	499	279	415	144	2	5.2	52.5	3.1	1990
St. Paul	7.25	0	30.35	314.5	1045	154	4	0	95.1	4.2	1040
Hilton	7.3	0	7.6	17.4	490	52.7	2.4	31.3	17.9	3.4	515
Rocky Mount	12.7	0	10.5	23.8	607	166	1.5	5.2	55.7	7.5	909
Cincinnati	4.7	0	89.9	1220	1790	237.5	1.55	25.15	399	4.8	1300
Cleveland	16.5	0	333	1010	789	291	2.5	11.2	236	23.1	3260
Euclid	16.4	0	210	136	605	45.4	2.7	37.3	70.2	3.3	940
Apollo	5.7	0	5.95	276.5	336	112.5	3.5	5.45	76.35	4.8	578.5
Pittsburgh	4.75	0	22.25	231	569.5	142	1.55	2.05	34.15	14.85	1985
Norfolk City	3.5	0	10.05	22.85	541	49.9	1.2	1.5	14.05	3.3	875

[a] Source: National Sewage Sludge Survey
[b] Total chromium.
[c] Total nickel.

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Table 3-4

Metal Concentrations by Flow Group
For Sludge Incinerated at Plants in the Analytic Survey

	Average Concentration for POTWS with 0-1 MGD (mg/kg)	Average Concentration for POTWS with 1-10 MGD (mg/kg)	Average Concentration for POTWS with 10-100 MGD (mg/kg)	Average Concentration for POTWS with 100+ MGD (mg/kg)
Arsenic	6.64	7.4	3.088	5.2
Cadmium	120.7	58.38	6.388	3.7
Chromium[a]	651.7	211.69	150.675	26.5
Copper	916.5	621.3	346.1	954
Lead	207.1	259.8	174.4	55.7
Mercury	2.22	2.0	2.863	3.7
Molybdenum	9.34	9.04	3.675	0
Nickel[b]	191.25	65.84	53.212	15.9
Selenium	10.21	4.16	2.238	4.100
Zinc	1941	1627	1670	460

[a]As total chromium.
[b]As total nickel.

others (including chromium, copper and nickel) are generally emitted as fly ash. Graphs of temperature against metals emitted for both controlled and uncontrolled emissions show an increase in the concentration of volatile metals in stack gas with higher furnace temperature. Conventional wet scrubbers appear relatively ineffective for capturing volatilized metals.

Mercury is the most volatile of the metals considered. Both inorganic and organic mercury compounds can volatilize at temperatures as low as 365°C (Dewling et al, 1980). As a result, mercury is emitted as a vapor or as volatilized mercury compounds. A study conducted in 1980 at the Northwest Bergen Sewer Authority recorded concentrations of metals in the feed sludge, bottom ash, scrubber water and stack emissions. According to these data, which were normalized for the mass balance, 98% of the mercury in the feed sludge was emitted to the atmosphere. Scrubber water contained about 2% and only 0.4% remained in the ash. The incinerator used a fluidized bed combustor with a wet scrubber and average furnace temperature of 788°C. Discussions from other sources agree that typical pollution control devices are not effective in removing mercury from incinerator emissions.

Chromium is most likely to be emitted from incinerators in either a hexavalent or trivalent state. Hexavalent chromium is a suspected human carcinogen, whereas trivalent chromium is a necessary trace element for humans. In general, the hexavalent chromium is expected to be more common in emissions from furnaces operating at higher temperatures with more available oxygen. Based on results from analysis of chromium emitted from Site 8, we assume that 1.3 percent of that emitted from a fluidized bed incinerator is hexavalent. For multiple hearth furnaces, testing at multiple sites has shown that hexavalent chromium ranges from about 1 percent of total to about 13.2 percent of total emitted chromium; the value of 13.2 percent is assumed since it represents a conservative value that possibly over-estimates cancer risks. For nickel, the carcinogenic species is nickel subsulfide (Ni_3S_2). Based on test results, we assume that one percent of the nickel emitted from the average multiple hearth or fluidized bed incinerator is in this form.

Ideally, efficiencies for each type of furnace and control device would be evaluated individually. For example, the removal efficiency of a furnace would determine how much of the metal contaminant in the feed sludge would remain in the bottom ash, and how much would be emitted as uncontrolled emissions. A scrubber efficiency would relate the quantity of contaminant entering the scrubber (uncontrolled emissions from the furnace) to the quantity leaving the scrubber and entering the atmosphere. Emissions from a particular incinerator could then be determined as the product of the efficiencies of its components (e.g., furnace and scrubber). However, available data are not always sufficient for accurate determination of the efficiency of individual components in the incineration process. In some cases, efficiencies can only be estimated for selected couplings of furnaces and controls.[1]

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[1]As mentioned above, removal efficiencies are sensitive to the operating conditions of furnaces and pollution control devices. These variables are not modeled explicitly in this analysis, although they are captured implicitly if conditions at those plants sampled are representative of the larger inventory of incinerators.

Sampling incinerator stack emissions and analyzing the samples is a complex, expensive process. The type of material combusted will to some extent affect the form of the metals entering the flue and therefore the effectiveness of the controls. For this reason data from incinerators burning materials other than sludge (e.g., municipal solid waste incinerators) cannot be used to represent the combustion and emission processes in a typical incinerator for sewage sludge. The availability of useful data is further reduced if one discards poor quality data with likely errors in sampling procedures. Where removal efficiency data are reported or can be calculated, they are usually based on the ratio of the rate at which the pollutant is emitted to the inflow rate for contaminant mass.

For predicting emissions under baseline conditions, three separate estimates of average removal efficiencies have been prepared for three furnace/control couplings: for multiple hearth incinerators with wet scrubbers, for fluidized bed incinerators with wet scrubbers, and for multiple hearth incinerators with dry cyclones and fabric filters. Table 3-5 contains results from tests performed on multiple hearth incinerators with wet scrubbers. Data from eight such incinerators were used; data available from two other sites, the Osborne Wastewater Treatment Plant in Greensboro, NC, and Site 9, were not used. For the Osborne plant, estimated removal efficiencies were greater than 100 percent, and for Site 9 they were significantly lower than for the other eight sites and were therefore discarded as outliers. Available estimates for each metal have been averaged to yield the values in the last column of the table. For this analysis, we assume that these average removal efficiencies are achieved by the 20 plants in the analytic survey (and the 164 plants in the larger sample of 172 facilities) using multiple hearth furnaces with wet scrubbers.

Table 3-6 contains estimates for fluidized bed furnaces with wet ESPs. Data from Glens Falls Waste Water Treatment Plant, Glens Falls, New York were excluded because the inflow rates were very low. As with multiple hearth incinerators, we have averaged these values to derive the results listed in the last column of the table. These averaged removal efficiencies are applied to the 3 facilities in the analytic survey of the NSSS (and 8 facilities in the larger sample) known to use fluidized bed furnaces with wet scrubbers. Table 3-7 summarizes removal efficiencies for wet scrubbers from both multiple hearth and fluidized bed furnaces.

In response to the regulation, some (but not all) POTWs are expected to retrofit their incinerators with additional pollution control devices. As mentioned above, further removal of metals is assumed to be achieved by installation of wet electrostatic precipitators. We assume that wet ESPs remove 95 percent of all remaining metals when operated at high efficiency, or 90 percent when operated at medium efficiency. An important distinction should be noted between efficiency data for wet ESPs and for all other types of controls for metals. The 90 and 95 percent removal efficiencies assumed for wet ESPs apply only to the control device itself; control efficiencies reported in Tables 3-5 through 3-7 pertain to specific couplings of furnace types and control devices. Data for the wet ESP are therefore useful for evaluating this technology as a retrofit option and removal efficiencies can be applied to the concentrations of metals emitted from the control devices already installed. For comparison with values listed in Tables 3-5 through 3-7 and used in this analysis, Table 3-8 summarizes results from other studies that investigated emissions of metals from incineration of either sludge or municipal solid wastes.

Table 3-5

Combined Removal Efficiencies for Multiple Hearth Incinerators with Wet Scrubbers[a]

	Site 1[b,c]	Site 2[c,d]	Site 4[c,e]	Site 6[f]	Derry[g]	Hopewell[h]	Lemay[i]	St. Paul[j]	Average[k]
Arsenic	--	93.83	98.53	--	--	99.99	99.75	95.30	97.48
Beryllium	100.00	99.82	--	89.8	--	--	99.70	--	97.33
Cadmium	79.32	79.82	80.89	80.4	99.73	99.99	92.85	95.32	88.54
Chromium	99.85	99.67	93.53	99.5	99.96	100	99.80	99.91	99.03
Copper	--	--	--	--	99.89	100	99.90	99.91	99.93
Lead	67.96	95.77	90.58	80	99.79	100	99.30	99.31	91.59
Mercury	--	--	--	--	--	--	--	--	--
Nickel	98.95	99.18	99.85	95.3	--	99.96	99.90	99.70	98.98
Selenium	--	--	--	--	--	--	99.80	--	99.80
Zinc	--	--	--	--	99.91	100	--	99.78	99.90

[a]Percent of metal retained by incinerator and pollution controls. Excludes available data for Osborne Wastewater Treatment Plant, Greensboro, NC, Site 4 with operating afterburner, and Site 9. See text for discussion.

[b]Site 1 Draft Emission Test Report, Sewage Sludge Test Program (1987), prepared for U.S. EPA Water Engineering Research Laboratory. (3-tray impingement scrubbers.)

[c]Removal efficiency calculated from quantity of sludge feed, concentrations of metals in feed, and data for emissions.

[d]Site 2 Draft Emission Test Report, Sewage Sludge Test Program (1987), prepared for U.S. EPA Water Engineering Research Laboratory. (Variable-throat Venturi followed by a tray scrubber.)

[e]Site 4 Draft Final Emission Test Report, Sewage Sludge Test Program (1988), prepared for U.S. EPA Water Engineering Research Laboratory. (Water injection Venturi and a tray scrubber. Values are for tests with afterburner inactive.)

[f]Sampling and Analysis of Municipal Wastewater Sludge Incinerator Emissions for Metals, Metal Species, and Organics, (undated), prepared for U.S. EPA Office of Water Regulations and Standards.

[g]Derry Township Municipal Authority, Derry Township, PA. State Compliance Test Report (1991). (Venturi with impingement tray scrubber.)

[h]Results of an Incinerator Performance Test, Hopewell Regional Wastewater Treatment Facility, Hopewell, VA. (1990). (Venturi Scrubber.)

[i]Emission Measurement Test Report of Multiple Hearth Sludge Incinerator for Particulate and Metals Emission(s), Lemay Waste Water Treatment Plant, St. Louis, MO. (1989).

[j]Summary of Stack Testing Data, based on 1989-1990 data, Metropolitan Wastewater Treatment Plant, St. Paul, MN. Metropolitan Waste Control Commission.

[k]Unweighted average of estimated values. These values are used in this analysis to represent removal efficiencies for all multiple hearth incinerators.

Table 3-6

Combined Removal Efficiencies for Fluidized Bed Incinerators with Wet Scrubbers[a]

	Site 3[b]	Site 8[c]	Atherton[d]	Blacksburg[e]	Average[f]
Arsenic	99.91	--	--	--	99.91
Beryllium	100.00	99.990	100	--	99.997
Cadmium	97.12	99.954	100	99.99	99.27
Chromium	99.78	99.976	99.999	99.89	99.91
Copper	--	--	99.999	99.97	99.98
Lead	99.90	99.979	99.991	99.67	99.89
Mercury	--	--	--	--	--
Nickel	99.54	99.969	99.999	--	99.84
Selenium	--	--	--	98.53	98.53
Zinc	--	--	--	99.46	99.46

[a]Percent of metal retained by incinerator and scrubber. Excludes available data for Glens Falls Waste Water Treatment Plant, Glens Falls, New York (Source Emission Testing of the Sewage Sludge Incinerator, 1990) because of low rates of inflow.

[b]Site 3 Draft Emission Test Report, Sewage Sludge Test Program (1987), prepared for U.S. EPA Water Engineering Research Laboratory. (Venturi with tray impinger.)

[c]Emissions of Metals and Organics from Municipal Wastewater Sludge Incinerators, Volume VI: Site 8 Draft Emission Test Report, (undated), prepared for U.S. EPA Office of Water Regulations and Standards.

[d]Little Blue Valley Sewer District's Atherton Plant, Independence, MO. (1989).

[e]Air Emission Testing for Blacksburg-Virginia Polytechnic Institute Sanitation Authority, Blacksburg, VA. (1990). (Venturi / cooler.)

[f]Unweighted average of non-missing values in table.

Table 3-7

Summary of Removal Efficiencies for Metals

	Multiple Hearth With Wet Scrubber[a]	Fluidized Bed With Wet Scrubber[b]
Arsenic	97.48	99.91
Beryllium	97.33	99.997
Cadmium	88.54	99.27
Chromium	99.03	99.91
Copper	99.93	99.98
Lead	91.59	99.89
Nickel	98.98	99.84
Selenium	99.80	98.53
Zinc	99.90	99.46

[a]From Table 3-5.
[b]From Table 3-6.

Table 3-8

Removal Efficiencies of Pollution Control Devices: Summary of Studies

Study		Type of Material Incinerated	Air Pollution Control Device	Contaminant	Removal Efficiencies	
Reference[a]	Description				Range	Mean
I	Applications of incineration technologies in Northern European countries.	Municipal Solid Waste	Spray Dryer Absorption /Baghouse	Dioxin	84-99.5%	-
				Mercury	-	90%
II	Applications of incineration technologies in Munich, West Germany.	Municipal Solid Waste	Dry sorption (w/ lime)/ Electrostatic Precipitators	Arsenic	-	96%
				Cadmium	-	99%
				Chromium	-	98%
				Lead	-	98%
				Mercury	-	99%
				Nickel	-	96%
II	Pilot study comparison of ESP and wet scrubber systems. (part 1 - ESP)	Municipal Sludge	Dry Electrostatic Precipitators	Particulates	92-98%	94.4%
				<1 um	61-73%	69.0%
				>10 um	99.7-99.8%	99.8%
				Arsenic	90-94%	93.5%
				Cadmium	78-91%	83.3%
				Chromium	71-99.8%	85.6%
				Lead	53-91%	77.5%
				Nickel	95-99.7%	97.3%
II	Pilot study comparison of ESP and wet scrubber systems. (Part 2 - Wet Scrubbers)	Municipal Sludge	Two cyclones, water quench, Venturi scrubber, tray scrubber.	Particulates	97.6-98.5%	98%
				<1 um	61-73%	69%
				>10 um	99.7-99.8%	99.8%
				Arsenic	90-94%	93.5%
				Cadmium	78-91%	83.3%
				Chromium	71-99.8%	85.6%
				Lead	53-91%	77.5%
				Nickel	95-99.7%	97.3%

[a] A list of references follows this table.

Table 3-8 (continued)

Removal Efficiencies of Pollution Control Devices: Summary of Studies

Study		Type of Material Incinerated	Air Pollution Control Device	Contaminant	Removal Efficiencies	
Reference[a]	Description				Range	Mean
IV	Pilot study comparison of Fabric Filters and wet scrubber systems. (Part 1 - Fabric Filters)	Municipal Sludge	Fabric Filters after MH furnace.	Particulates	-	99.1%
				<1 um	-	98.6%
				>10 um	-	97.7%
				Arsenic	97.6-98.6%	98.1%
				Cadmium	98.2-98.5%	98.5%
				Chromium	84-91%	87.2%
				Lead	98.9-99.6%	99.4%
				Nickel	90-94%	91.6%
IV	Pilot study comparison of Fabric Filters and wet scrubber systems. (Part 2 - Wet Scrubbers)	Municipal Sludge	Two cyclones, water quench, Venturi scrubber, tray scrubber.	Particulates	-	94.3%
				<1 um	-	59%
				>10 um	-	97.7%
				Arsenic	-	96.5%
				Cadmium	-	82.0%
				Chromium	-	95.7%
				Lead	-	86.5%
				Nickel	-	89.8%
V	Evaluation of fabric filter performance at resource recovery plant. Ames, Iowa.	Municipal Solid Waste	Fabric Filter	Particulates	-	97.8%
VI	Comparison of wet-dry scrubber system (spray dryer) plus fabric filter, to dry scrubber plus fabric filter. Quebec City, Canada. (Part 1 - spray dryer)	Municipal Solid Waste	Wet-Dry Scrubber plus Fabric Filter	Arsenic	-	99.9+%
				Cadmium	-	99.9+%
				Chromium	-	99.9+%
				Lead	-	99.9+%
				Mercury	-	94.7%
				Nickel	-	99.8%

[a] A list of references follows this table.

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Table 3-8 (continued)

Removal Efficiencies of Pollution Control Devices: Summary of Studies

Study		Type of Material Incinerated	Air Pollution Control Device	Contaminant	Removal Efficiencies	
Reference[a]	Description				Range	Mean
VI	Comparison of wet-dry scrubber system (spray dryer) plus fabric filter, to dry scrubber plus fabric filter. Quebec City, Canada. (Part 2 - dry scrubber)	Municipal Solid Waste	Dry Scrubber plus Fabric Filter	Arsenic Cadmium Chromium Lead Mercury Nickel	-- -- -- -- -- --	99.9+% 99.9+% 99.9+% 99.9+% 93.7% 99.9%
VII	Summary statistics on metals' destruction efficiencies from a database of 16 sludge incinerators.	Municipal Sludge	Several Types - See reference.	Arsenic Beryllium Cadmium Chromium Lead Mercury Nickel	93.9-100% 99.98-100% 40.25-99.98% 88.92-99.75% 34.22-99.97% -298.5-79.84% 89.15-100%	98.62% 99.99% 88.54% 99.16% 92.24% -72.06% 98.68%
VIII	Sewage Sludge Test Program Site 2	Municipal Sludge	Venturi Scrubber and 3-tray impingement scrubber on 8-hearth furnace	Arsenic Beryllium Cadmium Chromium Lead Mercury Nickel	77.2-77.7%(b) 87.9%(c) 62.8-68.1%(b) 40.9-81.4%(b) 76.5-78.1%(b) -31.7-54.8%(b)	-- -- -- -- -- -- --

[a] A list of references follows this table.

Table 3-8 (continued)

Removal Efficiencies of Pollution Control Devices: Summary of Studies

Study		Type of Material Incinerated	Air Pollution Control Device	Contaminant	Removal Efficiencies	
Reference[a]	Description				Range	Mean
IX	Sewage Sludge Test Program Site 4	Municipal Sludge	Variable-Throat Venturi S., 3-tray impingement scrubber, and afterburner on a 6-hearth furnace.	Arsenic	98.4%(d)	-
				Beryllium	-	-
				Cadmium	47.7(d)-52(e)%	-
				Chromium	98%(d)	-
				Lead	37.1(d)-60.57(e)%	-
				Mercury	-	-
				Nickel	92.3(e)-95.6(d)%	-
X	Notes about Scrubber and Fabric Filter Removal Efficiency. Norfolk, Virginia.	Municipal Sludge	1) Venturi Scrubbers	Arsenic	93.48-96.51%	-
				Cadmium	81.93-83.30%	-
				Chromium	85.57-95.72%	-
				Lead	77.46-86.63%	-
				Nickel	89.80-97.32%	-
			2) Fabric Filters	Arsenic	-	98.08%
				Cadmium	-	98.53%
				Chromium	-	87.16%
				Lead	-	99.38%
				Nickel	-	91.61%
			3) ESPs	Arsenic	-	92.44%
				Cadmium	-	99.54%
				Chromium	-	91.72%
				Lead	-	-
				Nickel	-	62.69%
XI	Proceedings: U.S.-Japan Conference on Sludge Incineration	Municipal Sludge	1) Wet ESPs on Multiple Hearth Furnaces	Particulates	84-89%	-
			2) Wet-dry Scrubber on Fluidized Bed Furnace using Lime	Particulates	97-98%	-
			3) Dry ESPs on Fluidized Bed Furnace using Lime	Particulates	84-85%	-
			4) Venturi Scrubber on Fluidized Bed Furnace using Polymers	Particulates	98-99%	-

[a] A references follows this table.

Table 3-8 (continued)

References for Studies

Study Number	Reference
I	Donnelly, J.R. Quach, M.T., and Moller, J.T., 1985. Joy/Niro Spray Dryer Absorption Flue Gas Cleaning System. Joy Manufacturing Company (Los Angeles) and A/S Niro Company (Copenhagen, Denmark). Presented at: Acid Gas and Dioxin Control for Waste to Energy Facilities Conference, Washington, D.C.
II	Deutsche Babcock, 1984/85. Cleaning the Flue Gas from the Munich North Refuse Fired Power Plant. Munich, West Germany.
III	U.S. EPA, 1985. Particulate Removal Evaluation of an Electrostatic Precipitator Dust Removal System Installed on a Multiple Hearth Incinerator Burning Sewage Sludge. Water Engineering Research Laboratory. Cincinatti, Ohio.
IV	U.S. EPA, 1985. Particulate Removal Capability of a Baghouse Filter on the Exhaust of a Multiple Hearth Furnace Burning Sewage Sludge. Water Engineering Research Laboratory. Cincinatti, Ohio.
V	Hall, F.D., Bruck, J.M., and Albrinck, D.N., 1985. Evaluation of Fabric Filter Performance at Ames Solid Waste Recovery System. Hazardous Waste Engineering Research Laboratory. Cincinatti, Ohio.
VI	Flakt Canada Ltd. and Environment Canada, 1986. The National Incinerator Testing and Evaluation Program: Air Pollution Control Technology. Industrial Programs Branch. Ottawa, Canada.
VII	U.S. EPA, 1988. Data Base Documentation and Summary Statistics on Metals' Destruction Efficiency Estimates From Incineration of Municipal Sludge. Analysis and Evaluation Division. Final Draft. September.
VIII	U.S. EPA, 1987. Site 2 Draft Emission Test Report, Sewage Sludge Test Program. Volume I: Emission Test Results. Water Engineering Research Laboratory. Cincinatti, Ohio.
IX	U.S. EPA, 1988. Site 4 Draft Final Emission Test Report, Sewage Sludge Test Program. Volume I: Emission Test Results. Water Engineering Research Laboratory. Cincinatti, Ohio.
X	Crumpler, G. Notes From Scrubber and Baghouse Removal Efficiency Tests. Norfolk, Virginia multiple hearth furnace.
XI	Okuno, N., 1978. Japan Sewage Works Agency. Facts About Emission Control Equipment for Sewage Sludge Incinerators. In: Proceedings: Sixth United States/Japan Conference on Sewage Treatment Technology. Cincinatti, Ohio (Oct. 30-31, 1978) and Washington, D.C. (Nov. 2-3, 1978).

3.2.8 Emissions of Organic Pollutants

To date, no useful correlation has been identified between the mix of organic compounds entering a sludge incinerator and the mix of organic compounds released in stack emissions. For this analysis, emissions of organic pollutants are considered a function of sludge feed rate and the type of furnace, and are not related to the concentrations of organic compounds in the sludge feed. Assumed profiles of emissions of organic compounds are based on measurements obtained from seven actual incinerators for wastewater sludge. Of these seven facilities, five were multiple hearth incinerators and two were fluidized bed combustors. This distinction is important because the improved burning conditions in fluidized bed combustors allow more complete oxidation of organic compounds and minimize the creation of products of incomplete combustion (PICs).

For both our "best estimate" and "worst case" estimate of organic emissions, we considered all compounds tested for in at least one facility sampled in the monitoring studies, regardless of whether the compounds were detected. For the "best" estimate, expected emissions are based on the arithmetic mean of values for the multiple samples taken for each furnace type. Samples with concentrations below limits of detection are assigned values corresponding to the appropriate detection limit. Where possible, each non-detect observation has been assigned the detection limit concentration for the particular site and chemical tested; otherwise, it has been assigned an average concentration based on detection limits at other sites or for similar compounds. Results are listed in the first and third columns of Table 3-9. Based on the available sample of measured (or assigned) values, a 99th percentile confidence or "worst case" estimate has been derived for the average emission rate for each pollutant for multiple hearth or fluidized bed combustors, as listed in the second and fourth columns of Table 3-9.

These "worst case" estimates for emissions are further adjusted to account for additional organic pollutants not tested. The Municipal Waste Combustion Study Report to Congress (U.S. EPA, 1987a) asserts that "a significant portion (80% or more) of the organic emissions from the stacks of municipal waste combustors have not been identified and quantified. Although some portion of the mixture may be carcinogenic, the carcinogenic fraction, its composition, and its potency remain unknown." If a similar fraction of emissions from sludge incinerators has not been identified or quantified, calculations based on known emissions might understate true exposure and risk. To adjust for this possible error, we assume that the average cancer potency of these unknown compounds is comparable to that for the chemicals evaluated. Assuming that in fact 80% of organic emissions have not been characterized, estimated health risks from incineration of wastewater sludge are increased by a factor of five to derive a conservative estimate of total risk.

3.2.9 Population Data

As discussed in Section 3.1.4, we obtain population data from the 1980 Census at the enumeration district or block group (ED/BG) level of aggregation, as provided within PC-GEMS. The MARF data base of PC-GEMS provides a convenient listing of population by the latitude and longitude of each ED/BG centroid.

Table 3-9

Unit Emissions of Organic Pollutants
(g/s emitted per kg/s sludge feed)

	Multiple Hearth Mean[a]	Multiple Hearth 99%ile[b]	Fluidized Bed Mean[c]	Fluidized Bed 99%ile[d]
1,2-dichloroethane	8.89×10^{-6}	2.11×10^{-5}	2.83×10^{-6}	8.79×10^{-6}
1,2-dichlorobenzene	7.36×10^{-4}	1.67×10^{-3}	9.78×10^{-6}	3.07×10^{-5}
1,2,3,4,6,7,8-heptachlorodibenzofuran	2.08×10^{-8}	2.08×10^{-8}	1.51×10^{-11}	1.51×10^{-11}
1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin	3.30×10^{-8}	9.41×10^{-8}	1.03×10^{-11}	1.03×10^{-11}
1,2,3,4,7,8-hexachlorodibenzo-p-dioxin	1.15×10^{-9}	1.15×10^{-9}	1.50×10^{-12}	1.50×10^{-12}
1,2,3,4,7,8-hexachlorodibenzofuran	3.66×10^{-8}	3.66×10^{-8}	9.70×10^{-12}	9.70×10^{-12}
1,2,3,4,7,8,9-heptachlorodibenzofuran	1.83×10^{-9}	1.83×10^{-9}	3.26×10^{-12}	3.26×10^{-12}
1,2,3,6,7,8-hexachlorodibenzo-p-dioxin	1.45×10^{-9}	2.43×10^{-9}	1.70×10^{-12}	1.70×10^{-12}
1,2,3,6,7,8-hexachlorodibenzofuran	8.77×10^{-9}	8.77×10^{-9}	4.94×10^{-12}	4.94×10^{-12}
1,2,3,7,8-pentachlorodibenzo-p-dioxin	1.38×10^{-9}	3.19×10^{-9}	1.91×10^{-12}	1.91×10^{-12}
1,2,3,7,8-pentachlorodibenzofuran	3.21×10^{-8}	3.21×10^{-8}	8.71×10^{-12}	8.71×10^{-12}
1,2,3,7,8,9-hexachlorodibenzofuran	1.06×10^{-9}	1.06×10^{-9}	1.41×10^{-12}	1.41×10^{-12}
1,2,3,7,8,9-hexachlorodibenzo-p-dioxin	2.41×10^{-9}	2.41×10^{-9}	2.23×10^{-12}	2.23×10^{-12}
1,2,4-trichlorobenzene	7.13×10^{-3}	2.21×10^{-2}	1.41×10^{-5}	4.69×10^{-5}
2-chlorophenol	6.33×10^{-4}	2.06×10^{-3}	9.39×10^{-6}	3.12×10^{-5}
2-methylphenol	5.22×10^{-5}	5.22×10^{-5}	4.08×10^{-8}	4.08×10^{-8}
2,3,4,6,7,8-hexachlorodibenzofuran	2.27×10^{-8}	2.27×10^{-8}	4.05×10^{-12}	4.05×10^{-12}
2,3,4,7,8-pentachlorodibenzofuran	1.32×10^{-7}	1.32×10^{-7}	5.29×10^{-12}	5.29×10^{-12}
2,3,7,8-tetrachlorodibenzofuran	1.64×10^{-7}	4.02×10^{-7}	3.98×10^{-12}	3.98×10^{-12}
2,3,7,8-tetrachlorodibenzo-p-dioxin	1.15×10^{-9}	2.82×10^{-9}	2.06×10^{-12}	2.06×10^{-12}
2,4-dinitrotoluene	9.46×10^{-5}	9.46×10^{-5}	2.14×10^{-8}	2.14×10^{-8}
2,4-dimethylphenol	8.17×10^{-4}	2.67×10^{-3}	1.41×10^{-5}	4.69×10^{-5}
2,4-dichlorophenol	7.39×10^{-4}	2.41×10^{-3}	1.88×10^{-5}	6.25×10^{-5}
2,4-dinitrophenol	5.23×10^{-3}	1.58×10^{-2}	3.43×10^{-4}	1.14×10^{-3}
2,4,5-trichlorophenol	5.08×10^{-5}	5.08×10^{-5}	2.08×10^{-8}	2.08×10^{-8}
2,4,6-trichlorophenol	1.00×10^{-3}	3.24×10^{-3}	2.82×10^{-5}	9.40×10^{-5}

(continued next page)

Table 3-9 (continued)

Unit Emissions of Organic Pollutants
(g/s emitted per kg/s sludge feed)

	Multiple Hearth Mean[a]	Multiple Hearth 99%ile[b]	Fluidized Bed Mean[c]	Fluidized Bed 99%ile[d]
3,3'-dichlorobenzidine	9.59×10^{-5}	9.59×10^{-5}	4.31×10^{-8}	4.31×10^{-8}
4-methylphenol	5.92×10^{-5}	5.92×10^{-5}	1.38×10^{-8}	1.38×10^{-8}
Acenaphthene	1.64×10^{-5}	4.13×10^{-5}	7.58×10^{-9}	7.58×10^{-9}
Acenaphthylene	2.04×10^{-5}	2.04×10^{-5}	2.96×10^{-8}	2.96×10^{-8}
Acetonitrile	9.58×10^{-3}	9.58×10^{-3}	--	--
Acrylonitrile	2.19×10^{-2}	3.68×10^{-2}	1.61×10^{-5}	5.31×10^{-5}
Aldrin	4.28×10^{-3}	1.39×10^{-2}	3.75×10^{-5}	3.75×10^{-5}
Anthracene	1.07×10^{-5}	1.07×10^{-5}	7.44×10^{-9}	7.44×10^{-9}
Benzene	6.01×10^{-2}	1.74×10^{-1}	2.00×10^{-4}	6.64×10^{-4}
Benz(a)anthracene	7.56×10^{-5}	7.56×10^{-5}	1.20×10^{-8}	1.20×10^{-8}
Benzo(a)pyrene	3.29×10^{-4}	1.05×10^{-3}	1.41×10^{-5}	4.68×10^{-5}
Benzo(b) fluoranthene	2.21×10^{-5}	2.21×10^{-5}	2.13×10^{-8}	2.13×10^{-8}
Benzo(k) fluoranthene	1.92×10^{-5}	1.92×10^{-5}	1.73×10^{-8}	1.73×10^{-8}
Benzoic acid	1.10×10^{-1}	1.10×10^{-1}	5.00×10^{-6}	5.00×10^{-6}
Bis(2-chloroisopropyl) ether	2.55×10^{-5}	2.55×10^{-5}	1.44×10^{-8}	1.44×10^{-8}
Bis(2-ethylhexyl)phthalate	1.02×10^{-3}	2.05×10^{-3}	1.34×10^{-4}	4.42×10^{-4}
Bis(2-chlorethyl) ether	4.95×10^{-5}	4.95×10^{-5}	1.41×10^{-8}	1.41×10^{-8}
Butylbenzyl phthalate	2.08×10^{-5}	2.08×10^{-5}	4.61×10^{-8}	4.61×10^{-8}
Carbon tetrachloride	3.35×10^{-5}	9.82×10^{-5}	3.84×10^{-6}	1.26×10^{-5}
Chlordane	3.33×10^{-4}	3.98×10^{-4}	5.63×10^{-4}	5.63×10^{-4}
Chlorobenzene	9.72×10^{-4}	1.86×10^{-3}	2.12×10^{-6}	7.03×10^{-6}
Chloroform	1.65×10^{-3}	4.38×10^{-3}	1.94×10^{-3}	6.45×10^{-3}
Chrysene	7.91×10^{-6}	7.91×10^{-6}	1.46×10^{-8}	1.46×10^{-8}
Di-n-butylphthalate	1.80×10^{-5}	1.80×10^{-5}	1.60×10^{-7}	1.60×10^{-7}
Dibenz(a,h)anthracene	4.86×10^{-5}	4.86×10^{-5}	4.81×10^{-8}	4.81×10^{-8}

Table 3-9 (continued)

Unit Emissions of Organic Pollutants
(g/s emitted per kg/s sludge feed)

	Multiple Hearth Mean[a]	Multiple Hearth 99%ile[b]	Fluidized Bed Mean[c]	Fluidized Bed 99%ile[d]
Dieldrin	3.88×10^{-3}	1.25×10^{-2}	5.63×10^{-5}	5.63×10^{-5}
Diethylphthalate	2.95×10^{-5}	2.95×10^{-5}	1.02×10^{-7}	1.02×10^{-7}
Ethylbenzene	1.76×10^{-3}	3.22×10^{-3}	9.23×10^{-6}	2.94×10^{-5}
Fluoranthene	4.55×10^{-4}	4.55×10^{-4}	5.68×10^{-8}	5.68×10^{-8}
Fluorene	2.14×10^{-5}	2.14×10^{-5}	7.05×10^{-9}	7.05×10^{-9}
Hexachlorobenzene	8.53×10^{-5}	8.53×10^{-5}	2.40×10^{-8}	2.40×10^{-8}
Hexachlorobutadiene	5.26×10^{-5}	5.26×10^{-5}	3.16×10^{-8}	3.16×10^{-8}
Hexachlorocyclopentadiene	2.10×10^{-4}	2.10×10^{-4}	2.55×10^{-8}	2.55×10^{-8}
Hexachloroethane	1.88×10^{-4}	1.88×10^{-4}	3.34×10^{-8}	3.34×10^{-8}
Indeno (1,2,3-cd) pyrene	5.97×10^{-5}	5.97×10^{-5}	2.89×10^{-8}	2.89×10^{-8}
Isophorone	1.42×10^{-5}	1.42×10^{-5}	7.82×10^{-9}	7.82×10^{-9}
Methyl ethyl ketone	4.94×10^{-3}	4.94×10^{-3}	--	--
Methylene chloride	1.43×10^{-3}	2.62×10^{-3}	4.70×10^{-5}	1.03×10^{-4}
N-Nitroso-di-n-propylamine	5.61×10^{-5}	5.61×10^{-5}	2.14×10^{-8}	2.14×10^{-8}
N-Nitrosodiphenylamine	3.12×10^{-5}	3.12×10^{-5}	1.21×10^{-8}	1.21×10^{-8}
Naphthalene	8.36×10^{-3}	2.49×10^{-2}	1.79×10^{-6}	1.79×10^{-6}
Nitrobenzene	2.23×10^{-5}	2.23×10^{-5}	1.40×10^{-8}	1.40×10^{-8}
Other-pentachlorodibenzo-p-dioxin	2.25×10^{-8}	5.75×10^{-8}	1.15×10^{-11}	1.15×10^{-11}
Other-tetrachlorodibenzofuran	7.00×10^{-7}	7.00×10^{-7}	1.08×10^{-10}	1.08×10^{-10}
Other-tetrachlorodibenzo-p-dioxin	1.73×10^{-7}	3.31×10^{-7}	2.58×10^{-11}	2.58×10^{-11}
Other-pentachlorodibenzofuran	5.23×10^{-7}	5.23×10^{-7}	7.66×10^{-11}	7.66×10^{-11}
Other-hexachlorodibenzo-p-dioxin	2.41×10^{-8}	3.00×10^{-8}	1.17×10^{-11}	1.17×10^{-11}
Other-heptachlorodibenzofuran	1.39×10^{-8}	1.39×10^{-8}	1.15×10^{-11}	1.15×10^{-11}
Other-heptachlorodibenzo-p-dioxin	2.95×10^{-8}	8.12×10^{-8}	1.48×10^{-11}	1.48×10^{-11}
Other-hexachlorodibenzofuran	7.07×10^{-8}	7.07×10^{-8}	2.87×10^{-11}	2.87×10^{-11}

(continued next page)

Table 3-9 (continued)

Unit Emissions of Organic Pollutants
(g/s emitted per kg/s sludge feed)

	Multiple Hearth Mean[a]	Multiple Hearth 99%ile[b]	Fluidized Bed Mean[c]	Fluidized Bed 99%ile[d]
PCBs	3.48×10^{-4}	4.49×10^{-4}	6.11×10^{-4}	6.11×10^{-4}
Pentachlorophenol	1.52×10^{-3}	4.67×10^{-3}	7.52×10^{-5}	2.50×10^{-4}
Phenol	7.85×10^{-4}	1.76×10^{-3}	9.42×10^{-6}	3.12×10^{-5}
Pyrene	2.41×10^{-4}	2.41×10^{-4}	3.78×10^{-8}	3.78×10^{-8}
Tetrachloroethylene	4.27×10^{-3}	1.22×10^{-2}	2.75×10^{-5}	8.69×10^{-5}
Toluene	3.79×10^{-2}	1.11×10^{-1}	3.38×10^{-5}	1.09×10^{-4}
Trans 1,2-dichloroethene	7.31×10^{-5}	2.14×10^{-4}	4.86×10^{-6}	4.86×10^{-6}
Trichloroethylene	1.02×10^{-3}	2.03×10^{-3}	4.94×10^{-6}	1.37×10^{-5}
Vinyl chloride	3.73×10^{-3}	6.45×10^{-3}	7.85×10^{-6}	2.60×10^{-5}

[a]Mean of available emissions data for five multiple hearth furnaces, with non-detect observations set to detection limit.
[b]99th percentile confidence limit for estimated mean of available emissions data for five multiple hearth furnaces, with non-detect values set to detection limit.
[c]Mean of available emissions data for two fluidized bed furnaces, with non-detect observations set to detection limit.
[d]99th percentile confidence limit for estimated mean of available emissions data for two fluidized bed furnaces, with non-detect values set to detection limit.

3.2.10 Meteorological Data

PC-GEMS also includes a Stability Array (STAR) database. Average wind velocity, wind directions, mixing heights and stability categories from STAR stations (usually airports) across the country are compiled to provide a national meteorological database. As implemented in PC-GEMS, these data are accessed automatically; we use the STAR station nearest to each incinerator for describing average weather conditions at the site.

3.2.11 Health Effects Data

As discussed in Chapter 1, health effects are grouped into two categories: cancer and non-cancer effects. For carcinogenic pollutants, a human cancer potency slope is characterized, and no threshold in exposure is assumed to exist. For most non-carcinogens, a threshold dose has been identified by the U.S. EPA below which no adverse health effects are expected. These threshold doses are termed "Risk Reference Doses" or RfDs. For this analysis, the primary source for our health effects data is the Integrated Risk Information System (IRIS) database maintained by the U.S. EPA. Health effects data are listed in Table 3-10.

3.3   RESULTS AND DISCUSSION

3.3.1 Baseline Risks

From our modeling of the emission and dispersion of pollutants, we estimate that a total of about 96,000,000 persons live within a 84 km square grid system centered on at least one sludge incinerator. In other words, a population of approximately this size lives within 40-50 km of at least one facility. Of this population, a surprisingly large fraction (about 70 percent) is exposed to pollutant plumes from more than one facility. Figure 3-3 shows the sizes of populations falling within the grids of varying numbers of incinerators. For example, it shows that about 1% of the total U.S. population (about 2,000,000 persons) is exposed simultaneously to pollutant plumes from 11 or more incinerators, while about 40% (96,000,000) is exposed to at least one plume (i.e., more than zero).

We estimate that under current conditions, the annual incineration of about 0.8 million dry metric tons of sewage sludge is responsible for an estimated incremental risk of about 0.3 to 4 cases of cancer per year. The lower of these two estimates is based on the "best estimate" of organic emissions discussed above; the higher value is based on the "worst case" estimate of organic emissions. Risks from exposure to metals account for approximately 0.07 cases each year while between 0.2 and 4 cases are expected to result from emissions of organic pollutants.

Based on "best estimate" assumptions for emissions, the highest aggregate risks from individual chemicals are 0.09 cases/yr from aldrin, 0.08 cases/yr from dieldrin, and 0.01 cases/yr from 2,3,4,7,8-pentachlorodibenzofuran (2,3,4,7,8-PCDF), as shown in Tables 3-11 and 3-12. Aldrin and dieldrin were never actually detected in stack emissions, although they were tested for. The detection limits at each test site were used as an upper bound on the

Table 3-10

Health Effects Data for Pollutants from Incineration of Sewage Sludge[a]

	Human Cancer Potency (mg/kg-day)$^{-1}$	Risk Reference Dose (mg/kg-day)	Reference
Arsenic	15	0.0003[b]	IRIS, IRIS
Beryllium	8.4	0.005[b]	IRIS, IRIS
Cadmium	6.3	0.0005[b]	IRIS, IRIS
Chromium(VI)	42	0.005[b]	IRIS, IRIS
Copper (cyanide)	NA	0.005[b]	IRIS
Mercury	NA	0.000086	HEAST
Nickel	1.7	0.02[b]	IRIS, IRIS
Selenium	NA	0.005[b]	IRIS
Zinc	NA	0.2[b]	HEAST
1,2-dichlorobenzene	NA	0.04	HEAST
1,2-dichloroethane	0.091	NA	IRIS
trans 1,2-dichloroethene	NA	0.02[b]	IRIS
1,2,3,4,6,7,8-heptachlorodibenzofuran	1600	NA	EPA
1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin	1600	NA	EPA
1,2,3,4,7,8-hexachlorodibenzofuran	16000	NA	EPA
1,2,3,4,7,8-hexachlorodibenzo-p-dioxin	16000	NA	EPA
1,2,3,4,7,8,9-heptachlorodibenzofuran	1600	NA	EPA
1,2,3,6,7,8-hexachlorodibenzo-p-dioxin	16000	NA	EPA
1,2,3,6,7,8-hexachlorodibenzofuran	16000	NA	EPA
1,2,3,7,8-pentachlorodibenzofuran	7800	NA	EPA
1,2,3,7,8-pentachlorodibenzo-p-dioxin	78000	NA	EPA
1,2,3,7,8,9-hexachlorodibenzofuran	16000	NA	EPA
1,2,3,7,8,9-hexachlorodibenzo-p-dioxin	16000	NA	EPA

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Table 3-10 (continued)

Health Effects Data for Pollutants from Incineration of Sewage Sludge

	Human Cancer Potency (mg/kg-day)$^{-1}$	Risk Reference Dose (mg/kg-day)	Reference
1,2,4-trichlorobenzene	NA	0.003	HEAST
2-chlorophenol	NA	0.005[b]	IRIS
2,3,4,6,7,8-hexachlorodibenzofuran	16000	NA	EPA
2,3,4,7,8-pentachlorodibenzofuran	78000	NA	EPA
2,3,7,8-tetrachlorodibenzofuran	16000	NA	EPA
2,3,7,8-tetrachlorodibenzo-p-dioxin	160000	NA	EPA
2,4-dimethylphenol	NA	0.02[b]	IRIS
2,4-dinitrotoluene	0.68[b]	NA	IRIS
2,4-dichlorophenol	NA	0.003[b]	IRIS
2,4-dinitrophenol	NA	0.002[b]	IRIS
2,4,5-trichlorophenol	NA	0.1[b]	IRIS
2,4,6-trichlorophenol	0.01	NA	IRIS
3,3'-dichlorobenzidine	0.45[b]	NA	IRIS
4-methylphenol	NA	0.05[b]	HEAST
Acenaphthene	NA	0.06[b]	IRIS
Acetonitrile	NA	0.01	HEAST
Acrylonitrile	0.24	0.00057	IRIS, IRIS
Aldrin	17	0.00003[b]	IRIS, IRIS
Anthracene	NA	0.3[b]	IRIS
Benzene	0.029	NA	IRIS
Benz(a)anthracene	6.1	NA	B(a)P
Benzo(a)pyrene	6.1	NA	HEAST
Benzo(b)fluoranthene	6.1	NA	B(a)P

(continued next page)

Table 3-10 (continued)

Health Effects Data for Pollutants from Incineration of Sewage Sludge

	Human Cancer Potency (mg/kg-day)$^{-1}$	Risk Reference Dose (mg/kg-day)	Reference
Benzo(k)fluoranthene	6.1	NA	B(a)P
Benzoic acid	NA	4[b]	IRIS
Bis(2-chlorethyl)ether	1.1	NA	IRIS
Bis(2-ethylhexyl)phthalate	0.014[b]	0.02[b]	IRIS, IRIS
Bis(2-chloroisopropyl)ether	0.035	0.04[b]	HEAST, IRIS
Butylbenzyl phthalate	NA	0.2[b]	IRIS
Carbon tetrachloride	0.052	0.0007[b]	IRIS, IRIS
Chlordane	1.3	0.00006[b]	IRIS, IRIS
Chlorobenzene	NA	0.005	HEAST
Chloroform	0.081	0.01[b]	IRIS, IRIS
Chrysene	6.1	NA	B(a)P
Di-n-butylphthalate	NA	0.1[b]	IRIS
Dibenz(a,h)anthracene	6.1	NA	B(a)P
Dieldrin	16	0.00005[b]	IRIS, IRIS
Diethylphthalate	NA	0.8[b]	IRIS
Ethylbenzene	NA	0.3	IRIS
Fluoranthene	NA	0.04[b]	IRIS
Fluorene	NA	0.04[b]	IRIS
Hexachlorobenzene	1.6	0.0008[b]	IRIS, IRIS
Hexachlorobutadiene	0.078	0.002[b]	IRIS, IRIS
Hexachlorocyclopentadiene	NA	0.00002	HEAST
Hexachloroethane	0.014	0.001[b]	IRIS, IRIS
Indeno(1,2,3-cd)pyrene	6.1	NA	B(a)P

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Table 3-10 (continued)

Health Effects Data for Pollutants from Incineration of Sewage Sludge

	Human Cancer Potency $(mg/kg-day)^{-1}$	Risk Reference Dose $(mg/kg-day)$	Reference
Isophorone	0.0041	0.2[b]	IRIS, IRIS
Methyl ethyl ketone	NA	0.3	IRIS
Methylene chloride	0.0016	0.9	IRIS, HEAST
N-nitrosodiphenylamine	0.0049[b]	NA	IRIS
N-Nitroso-di-n-propylamine	7[b]	NA	IRIS
Naphthalene	NA	0.04[b]	HEAST
Nitrobenzene	NA	0.0006	HEAST
Other-pentachlorodibenzo-p-dioxin	160000	NA	EPA
PCBs	7.7[b]	NA	IRIS
Pentachlorophenol	0.12[b]	0.03[b]	IRIS, IRIS
Phenol	NA	0.6[b]	IRIS
Pyrene	NA	0.03[b]	IRIS
Tetrachloroethylene	0.0018	0.01[b]	HEAST, IRIS
Toluene	NA	0.1	IRIS
Trichloroethylene	0.017	NA	HEAST
Vinyl chloride	0.3	NA	HEAST

[a] Human cancer potency values and risk reference doses are for inhalation exposure unless otherwise noted.
[b] Oral exposure value.
B(a)P = Human cancer potency value for benzo(a)pyrene used for all polycyclic aromatic hydrocarbons considered to be B2 carcinogens.
EPA = Estimating Exposure to Dioxin-Like Compounds, U.S. EPA 1989i.
HEAST = Health Effects Assessment Summary Tables, March 1992.
IRIS = Integrated Risk Information System, 1992.

Figure 3-3
Population by Number of Overlapped Plumes
Incineration of Sewage Sludge
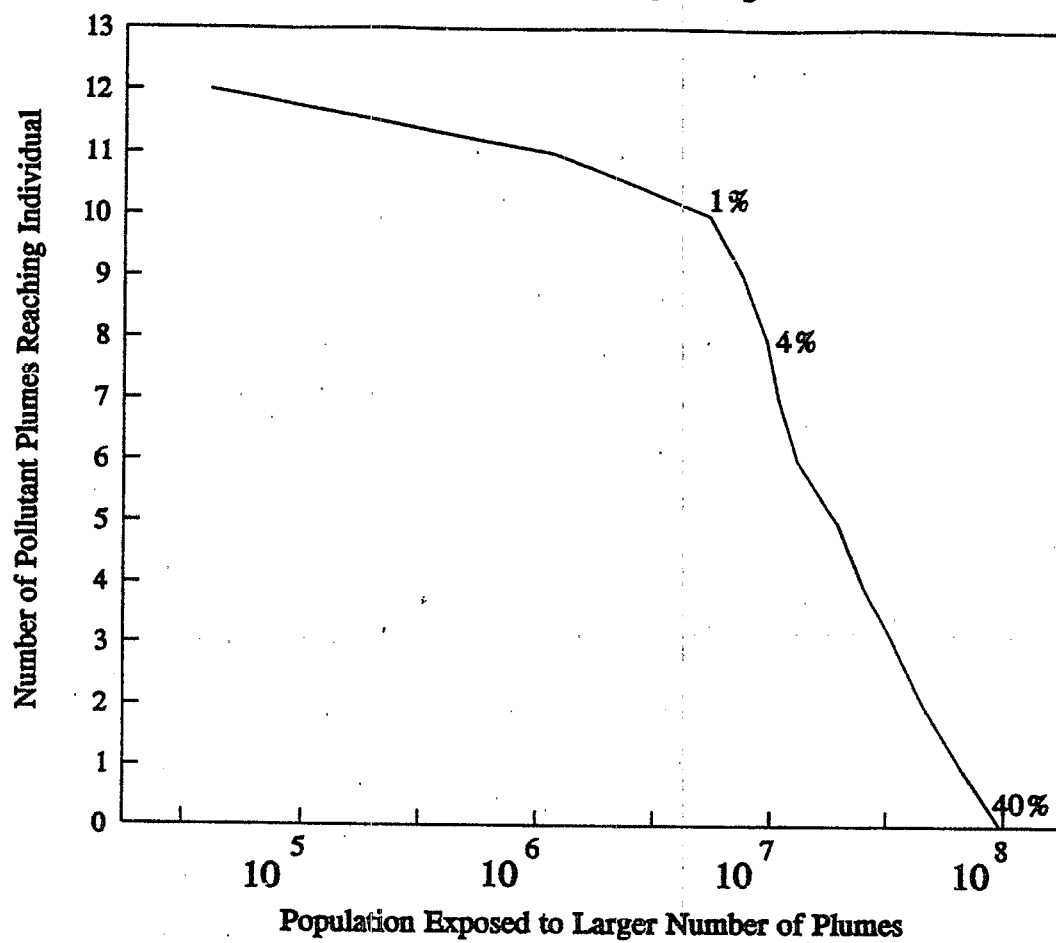
Table 3-11

Baseline Cancer Risk by Pollutant
For Incineration of Sewage Sludge

	HEI Risk	Mean Individual Risk	Total Aggregate Risk*
Arsenic	7×10^{-6}	2×10^{-9}	2×10^{-3}
Cadmium	3×10^{-4}	5×10^{-8}	7×10^{-2}
Chromium	1×10^{-5}	1×10^{-9}	2×10^{-3}
Nickel	2×10^{-6}	2×10^{-10}	3×10^{-4}
1,2-dichloroethane	2×10^{-9}	8×10^{-13}	1×10^{-6}
1,2,3,4,6,7,8-heptachlorodibenzofuran	7×10^{-8}	3×10^{-11}	4×10^{-5}
1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin	1×10^{-7}	5×10^{-11}	7×10^{-5}
1,2,3,4,7,8-hexachlorodibenzo-p-dioxin	4×10^{-8}	2×10^{-11}	2×10^{-5}
1,2,3,4,7,8-hexachlorodibenzofuran	1×10^{-6}	5×10^{-10}	8×10^{-4}
1,2,3,4,7,8,9-heptachlorodibenzofuran	6×10^{-9}	3×10^{-12}	4×10^{-6}
1,2,3,6,7,8-hexachlorodibenzo-p-dioxin	5×10^{-8}	2×10^{-11}	3×10^{-5}
1,2,3,6,7,8-hexachlorodibenzofuran	3×10^{-7}	1×10^{-10}	2×10^{-4}
1,2,3,7,8-pentachlorodibenzo-p-dioxin	2×10^{-7}	1×10^{-10}	1×10^{-4}
1,2,3,7,8-pentachlorodibenzofuran	5×10^{-7}	2×10^{-10}	3×10^{-4}
1,2,3,7,8,9-hexachlorodibenzofuran	3×10^{-8}	2×10^{-11}	2×10^{-5}
1,2,3,7,8,9-hexachlorodibenzo-p-dioxin	8×10^{-8}	4×10^{-11}	5×10^{-5}
2,3,4,6,7,8-hexachlorodibenzofuran	7×10^{-7}	3×10^{-10}	5×10^{-4}
2,3,4,7,8-pentachlorodibenzofuran	2×10^{-5}	1×10^{-8}	1×10^{-2}
2,3,7,8-tetrachlorodibenzo-p-dioxin	4×10^{-7}	2×10^{-10}	2×10^{-4}
2,3,7,8-tetrachlorodibenzofuran	5×10^{-6}	2×10^{-9}	3×10^{-3}
2,4-dinitrotoluene	1×10^{-7}	6×10^{-11}	8×10^{-5}
2,4,6-trichlorophenol	2×10^{-8}	9×10^{-12}	1×10^{-5}
3,3'-dichlorobenzidine	9×10^{-8}	4×10^{-11}	6×10^{-5}
Acrylonitrile	1×10^{-5}	5×10^{-9}	7×10^{-3}
Aldrin	1×10^{-4}	7×10^{-8}	9×10^{-2}
Benzene	4×10^{-6}	2×10^{-9}	2×10^{-3}
Benz(a)anthracene	9×10^{-7}	4×10^{-10}	6×10^{-4}
Benzo(a)pyrene	4×10^{-6}	2×10^{-9}	3×10^{-3}
Benzo(b)fluoranthene	3×10^{-7}	1×10^{-10}	2×10^{-4}
Benzo(k)fluoranthene	2×10^{-7}	1×10^{-10}	2×10^{-4}

(continued on next page)

Table 3-11 (continued)

Baseline Cancer Risk by Pollutant
For Incineration of Sewage Sludge

	HEI Risk	Mean Individual Risk	Total Aggregate Risk*
Bis(2-ethylhexyl)phthalate	3×10^{-8}	1×10^{-11}	2×10^{-5}
Bis(2-chlorethyl)ether	1×10^{-7}	5×10^{-11}	7×10^{-5}
Bis(2-chloroisopropyl)ether	2×10^{-9}	8×10^{-13}	1×10^{-6}
Carbon tetrachloride	4×10^{-9}	2×10^{-12}	2×10^{-6}
Chlordane	1×10^{-6}	4×10^{-10}	6×10^{-4}
Chloroform	3×10^{-7}	1×10^{-10}	2×10^{-4}
Chrysene	1×10^{-7}	5×10^{-11}	6×10^{-5}
Dibenz(a,h)anthracene	6×10^{-7}	3×10^{-10}	4×10^{-4}
Dieldrin	1×10^{-4}	6×10^{-8}	8×10^{-2}
Hexachlorobenzene	3×10^{-7}	1×10^{-10}	2×10^{-4}
Hexachlorobutadiene	8×10^{-9}	4×10^{-12}	5×10^{-6}
Hexachloroethane	5×10^{-9}	2×10^{-12}	3×10^{-6}
Indeno(1,2,3-cd)pyrene	7×10^{-7}	3×10^{-10}	5×10^{-4}
Isophorone	1×10^{-10}	5×10^{-14}	8×10^{-8}
Methylene chloride	5×10^{-9}	2×10^{-12}	3×10^{-6}
N-nitrosodiphenylamine	3×10^{-10}	1×10^{-13}	2×10^{-7}
N-nitroso-di-n-propylamine	8×10^{-7}	4×10^{-10}	5×10^{-4}
other-pentachlorodibenzo-p-dioxin	7×10^{-6}	3×10^{-9}	5×10^{-3}
Pentachlorophenol	4×10^{-7}	2×10^{-10}	2×10^{-4}
PCBs	6×10^{-6}	2×10^{-9}	4×10^{-3}
Tetrachloroethylene	2×10^{-8}	7×10^{-12}	1×10^{-5}
Trichloroethylene	4×10^{-8}	2×10^{-11}	2×10^{-5}
Vinyl chloride	2×10^{-6}	1×10^{-9}	1×10^{-3}
TOTAL	6×10^{-4}	2×10^{-7}	3×10^{-1}

*Based on "best estimate" for emissions of organic pollutants.

Table 3-12

Contribution of Individual Pollutants to Total Estimated Cancer Risk from Incineration[a]

	Number Plants Detected[b]	Number Plants Tested[c]	Percent Plants Detected[d]	Total Aggregate Risk[a]	Percent Total	Cum. Percent Total
Aldrin	0	4	0%	0.08	31%	31%
Cadmium	NA	NA	NA	0.07	27%	58%
Dieldrin	0	4	0%	0.07	26%	84%
2,3,4,7,8-pentachlorodibenzofuran	3	3	100%	0.01	4%	88%
Acrylonitrile	5	6	83%	0.006	2%	90%
other pentachlorodibenzo-p-dioxins	3	3	100%	0.004	2%	92%
PCBs	1	3	33%	0.003	1%	93%
2,3,7,8-tetrachlorodibenzofuran	3	3	100%	0.003	1%	94%
Arsenic	NA	NA	NA	0.002	0.8%	95%
Benzo(a)pyrene	0	6	0%	0.002	0.8%	96%
Benzene	6	6	100%	0.002	0.7%	96.7%
Chromium	NA	NA	NA	0.002	0.7%	97.4%
Vinyl chloride	5	6	83%	0.001	0.5%	97.9%
1,2,3,4,7,8-hexachlorodibenzofuran	3	3	100%	0.0007	0.2%	98.1%
Benz(a)anthracene	1	1	100%	0.0005	0.2%	98.3%
Chlordane	0	3	0%	0.0005	0.2%	98.5%
N-Nitroso-di-n-propylamine	0	2	0%	0.0004	0.2%	98.7%
Indeno(1,2,3-cd)pyrene	0	2	0%	0.0004	0.2%	98.8%
2,3,4,6,7,8-hexachlorodibenzofuran	3	3	100%	0.0004	0.2%	99.0%
SUBTOTAL FOR 19 POLLUTANTS:	--	--	--	0.3	99%	--
SUBTOTAL FOR 19 POLLUTANTS EXCL. NON-DETECTS[e]	--	--	--	0.1	38%	--
TOTAL FOR 95 POLLUTANTS[e]	--	--	--	0.3	100%	--

[a]Table contains 19 pollutants contributing 99% of total risk, listed in descending order of contribution. Based on "best estimate" of organic emissions.
[b]Number of tested plants at which pollutant was detected in emissions. Each plant counted only once, regardless of multiple detected samples. Not applicable for metals, for which predicted emissions are based on concentration in sludge.
[c]Number of plants at which emissions were tested for pollutant. Not applicable for metals.
[d]Number of plants with detected pollutant divided by number of plants tested.
[e]Excludes pollutants not detected at one or more plants.

emissions of these pollutants. The pollutant 2,3,4,7,8-PCDF, however, was detected at 3 sites. The lifetime incremental cancer risk for the HEI is estimated to be $6x10^{-4}$ based on the "best estimate" of emissions, or $7x10^{-3}$ based on "worst case" estimates. Figure 3-4 shows the estimated distribution of individual cancer risk within the exposed population.

Aggregate non-cancer effects are expressed as the number of persons exceeding the risk reference dose (RfD) for each pollutant, or the ratio of exposure to RfD when this level is not exceeded. As shown for organic pollutants in Table 3-13, we find that the Risk Reference Dose for any pollutant emitted from sludge incinerators is not exceeded for any of the pollutants. Where available, background intake of pollutant from non-sludge sources is included in the calculations. Typical background intakes for metals are generally two or more orders of magnitude higher than those resulting from the incineration of sludge. The most significant contribution from sludge is for mercury, for which the intake resulting from incineration is less than 4% of the background level. In general, the contribution to ambient levels of these metals is negligible, as shown in Table 3-14.

Based on methods and assumptions outlined in Chapter 2, we estimate that about 700 persons cross blood lead thresholds each year as a result of the incineration of sludge. Exposure to lead from the incineration of sludge also causes an estimated 100 cases of non-cancer disease per year, of which more than 95% consist of additional cases of hypertension expected in adult males. The remainder are due to other cardiovascular effects in adults and neurological developmental effects in children. For cadmium, fewer than one person per year is expected to cross the kidney cadmium threshold because of exposure from sludge.

3.3.2 Benefits from Regulatory Controls

Analysis of each facility's individual response to the regulation has been completed for only those 23 POTWs included in the analytic subset of the NSSS. In order to estimate risk under the new regulations, all exposure and risk calculations are repeated after adjusting emissions from each of these modeled facilities to reflect the expected installation of additional pollution controls. For simplicity, only two additional control technologies are considered. To reduce emissions for metals, a wet electrostatic precipitator (ESP) operating at either high (95% of maximum) or medium (90% of maximum) is assumed to be installed; further reduction of organic emissions is assumed to be unnecessary. Estimates of exposure and risk after installation of controls are scaled to the national level based on dispersion results from the full inventory of known incinerators, and on sample weights from the NSSS. Expected reductions in risk are derived by subtracting these results from comparable estimates of exposure and risk under baseline conditions. Table 3-15 shows the estimated reduction in the number of cancer cases expected each year. Total risks from emissions of metal pollutants are expected to decline by about 90%; risks from organic pollutants are not affected by the regulation.

Additional pollution controls reduce the number of people crossing the lead threshold from about 700 to about 90 and the associated number of disease cases from about 100 to about 30. Table 3-16 shows how the incremental body burden of lead from incinerated sludge (in $\mu g/dl$) is distributed among exposed individuals both before and after implementation of controls

Figure 3-4
Baseline Individual Cancer Risk by Population Size For Incineration
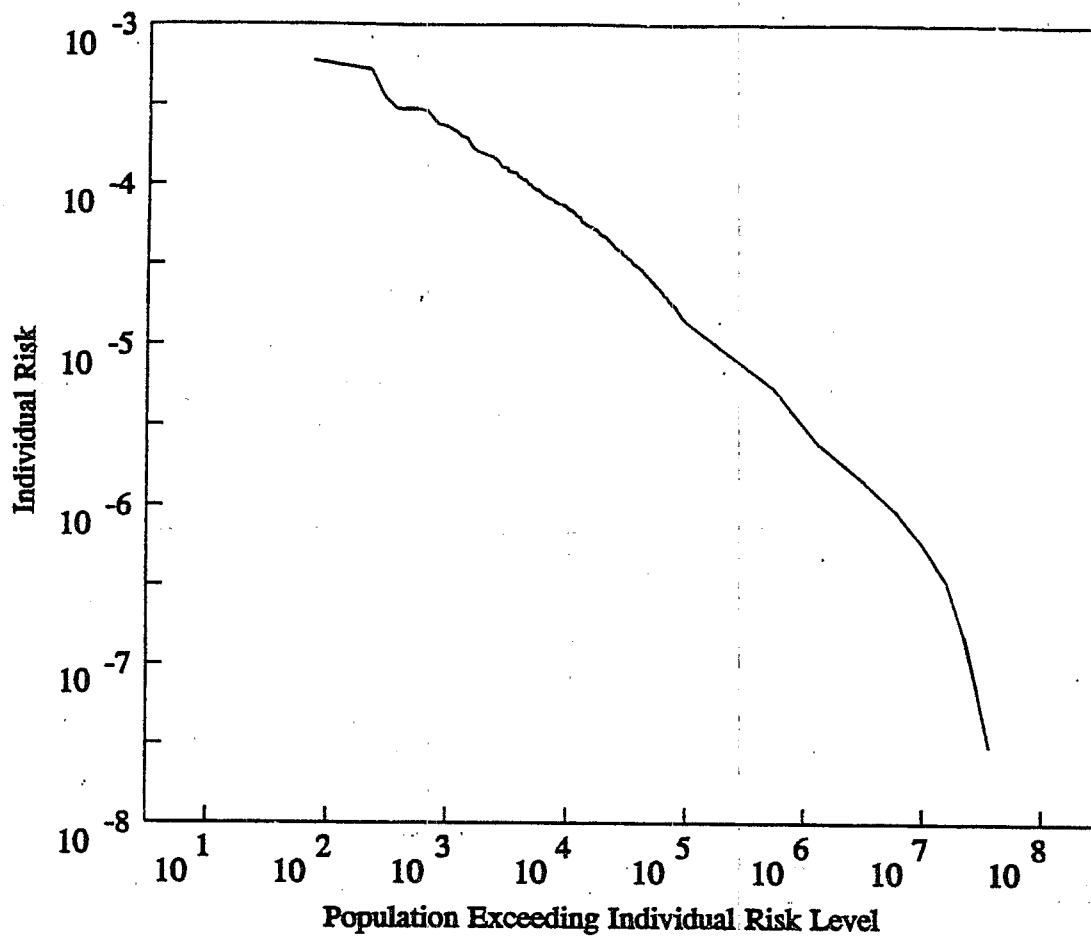
Table 3-13

Comparison of Exposure to Risk Reference Doses
For Organic Pollutants from Incineration of Sewage Sludge[a]

	Risk Reference Dose (mg/kg-day)	Maximum Exposure (mg/kg-day)	Maximum Exposure as Percent of RfD	Typical Exposure (mg/kg-day)	Typical Exposure as Percent of RfD
1,2-dichlorobenzene	0.04	1×10^{-6}	0.004%	7×10^{-10}	<0.001%
1,2,4-trichlorobenzene	0.003	1×10^{-5}	0.47%	7×10^{-9}	<0.001%
2-chlorophenol	0.005	1×10^{-6}	0.025%	6×10^{-10}	<0.001%
2,4-dimethylphenol	0.02	2×10^{-6}	0.008%	8×10^{-10}	<0.001%
2,4-dichlorophenol	0.003	2×10^{-6}	0.067%	7×10^{-10}	<0.001%
2,4-dinitrophenol	0.002	1×10^{-5}	0.52%	5×10^{-9}	<0.001%
Acetonitrile	0.01	2×10^{-5}	0.19%	9×10^{-9}	<0.001%
Acrylonitrile	0.00057	4×10^{-5}	7.5%	2×10^{-8}	0.003%
Aldrin	0.00003	9×10^{-6}	30%	4×10^{-9}	0.013%
Benzoic acid	4	2×10^{-4}	0.005%	1×10^{-7}	<0.001%
Bis(2-ethylhexyl)phthalate	0.02	2×10^{-6}	0.01%	9×10^{-10}	<0.001%
Carbon tetrachloride	0.0007	7×10^{-8}	0.01%	3×10^{-11}	<0.001%
Chlordane	0.00006	8×10^{-7}	1.3%	3×10^{-10}	<0.001%
Chlorobenzene	0.005	2×10^{-6}	0.038%	9×10^{-10}	<0.001%
Chloroform	0.01	4×10^{-6}	0.035%	2×10^{-9}	<0.001%
Dieldrin	0.00005	8×10^{-6}	16%	4×10^{-9}	0.008%
Ethylbenzene	0.3	4×10^{-6}	0.001%	2×10^{-9}	<0.001%
Fluoranthene	0.04	9×10^{-7}	0.002%	4×10^{-10}	<0.001%
Hexachlorobenzene	0.0008	2×10^{-7}	0.021%	8×10^{-11}	<0.001%
Hexachlorobutadiene	0.002	1×10^{-7}	0.005%	5×10^{-11}	<0.001%
Hexachlorocyclopentadiene	0.00002	4×10^{-7}	2.07%	2×10^{-10}	0.001%
Hexachloroethane	0.001	4×10^{-7}	0.037%	2×10^{-10}	<0.001%
Methyl ethyl ketone	0.3	1×10^{-5}	0.003%	5×10^{-9}	<0.001%
Naphthalene	0.04	2×10^{-5}	0.041%	8×10^{-9}	<0.001%
Nitrobenzene	0.0006	5×10^{-8}	0.008%	2×10^{-11}	<0.001%
Pentachlorophenol	0.03	3×10^{-6}	0.010%	1×10^{-9}	<0.001%
Pyrene	0.03	5×10^{-7}	0.002%	2×10^{-10}	<0.001%
Tetrachloroethylene	0.01	9×10^{-6}	0.09%	4×10^{-9}	<0.001%
Toluene	0.1	8×10^{-5}	0.08%	4×10^{-8}	<0.001%
Trans 1,2-dichloroethene	0.02	1×10^{-7}	0.001%	7×10^{-11}	<0.001%

[a] Only those compounds with at least one exposure greater than (0.001 x RfD) are shown.

Table 3-14

Comparison of Exposure to Risk Reference Doses
For Metals from Incineration of Sewage Sludge

	Background Exposure (mg/kg-day)	Risk Reference Dose (mg/kg-day)	Maximum Incremental Exposure (mg/kg-day)	Mean Incremental Exposure (mg/kg-day)	Background as Percent of Risk Reference Dose	Maximum Incremental Exposure as Percent of RfD	Total Exposure as Percent of Risk Reference Dose
Arsenic	1.2×10^{-3}	3.0×10^{-4}	5×10^{-7}	1×10^{-10}	390%	0.15%	390%
Chromium	1.5×10^{-3}	5.0×10^{-3}	2×10^{-7}	3×10^{-11}	30%	0.005%	30%
Copper	2.2×10^{-3}	5.0×10^{-3}	2×10^{-6}	4×10^{-10}	45%	0.05%	45%
Mercury	9.4×10^{-5}	8.6×10^{-5}	4×10^{-6}	2×10^{-9}	110%	4.4%	114%
Nickel	2.5×10^{-3}	2.0×10^{-2}	1×10^{-6}	1×10^{-10}	12%	0.005%	12%
Selenium	1.5×10^{-4}	5.0×10^{-3}	5×10^{-8}	1×10^{-11}	3.1%	0.001%	3.1%
Zinc	1.9×10^{-1}	2.0×10^{-1}	4×10^{-6}	2×10^{-9}	95%	1×10^{-6}%	95%

Table 3-15

Cancer Risks from Incineration
Before and After Regulatory Controls
For Incineration of Sewage Sludge

	Baseline	With Controls	Reduction
METALS[a]			
Multiple Hearth Furnaces	0.1	0.01	0.09
Fluidized Bed Furnaces	0.00001	1×10^{-6}	9×10^{-6}
Total	0.1	0.01	0.09
ORGANIC POLLUTANTS[a,b]			
Multiple Hearth Furnaces	0.2-4	0.2-4	0
Fluidized Bed Furnaces	0.00009	0.00009	0
Total	0.2-4	0.2-4	0
ALL POLLUTANTS[a,b]			
Multiple Hearth Furnaces	0.3-4	0.2-4	0.09
Fluidized Bed Furnaces	0.0001	0.00009	9×10^{-6}
Total	0.3-4	0.2-4	0.09
AVERAGE INDIVIDUAL RISK[d]	$2 \times 10^{-7} - 3 \times 10^{-6}$	$2 \times 10^{-7} - 3 \times 10^{-6}$	2-28%[e]
INDIVIDUAL RISK FOR HEI[f]	$6 \times 10^{-4} - 7 \times 10^{-3}$	$4 \times 10^{-4} - 7 \times 10^{-3}$	3-39%[e]

[a]All estimates are total expected incremental risk of cancer as result of exposure to pollutants from sludge. Estimates are in units of expected cancer cases per year. Values do not sum to totals because of independent rounding to one significant figure.

[b]Lower estimates derived with "best estimate" of emissions for organic pollutants. Higher estimates for "worst case" estimate of organic emissions.

[c]Estimates expressed as incremental risk that individual will develop cancer from lifetime of exposure to pollutants from sludge.

[d]Risk for average exposed individual of developing cancer from lifetime of exposure to pollutants from sludge.

[e]Equals (baseline-control)/(baseline).

[f]Risk for the Highly Exposed Individual (HEI) of developing cancer from lifetime of exposure to pollutants from sludge.

Table 3-16

Risks from Lead and Cadmium
For Incineration of Sewage Sludge

	Baseline (cases/yr)	Control (cases/yr)	Benefit (cases/yr)
Persons Crossing Cadmium Threshold	<0.01	<0.01	<0.01
Persons Crossing Lead Threshold			
Men	500	80	500
Women	90	9	80
Children	40	2	30
TOTAL	700	90	600
Expected Disease Cases from Lead	100	30	90

Note: all values have been independently rounded to one significant figure. Values do not sum to totals because of independent rounding.

and installation of wet ESPs. As can be seen from Figure 3-5, the average increment in blood lead for the worst grid cell modeled would be about 0.5 μg/dl under baseline conditions and about 0.1 after controls.

Figure 3-5
Increment to Blood Lead by Population Size for Incineration
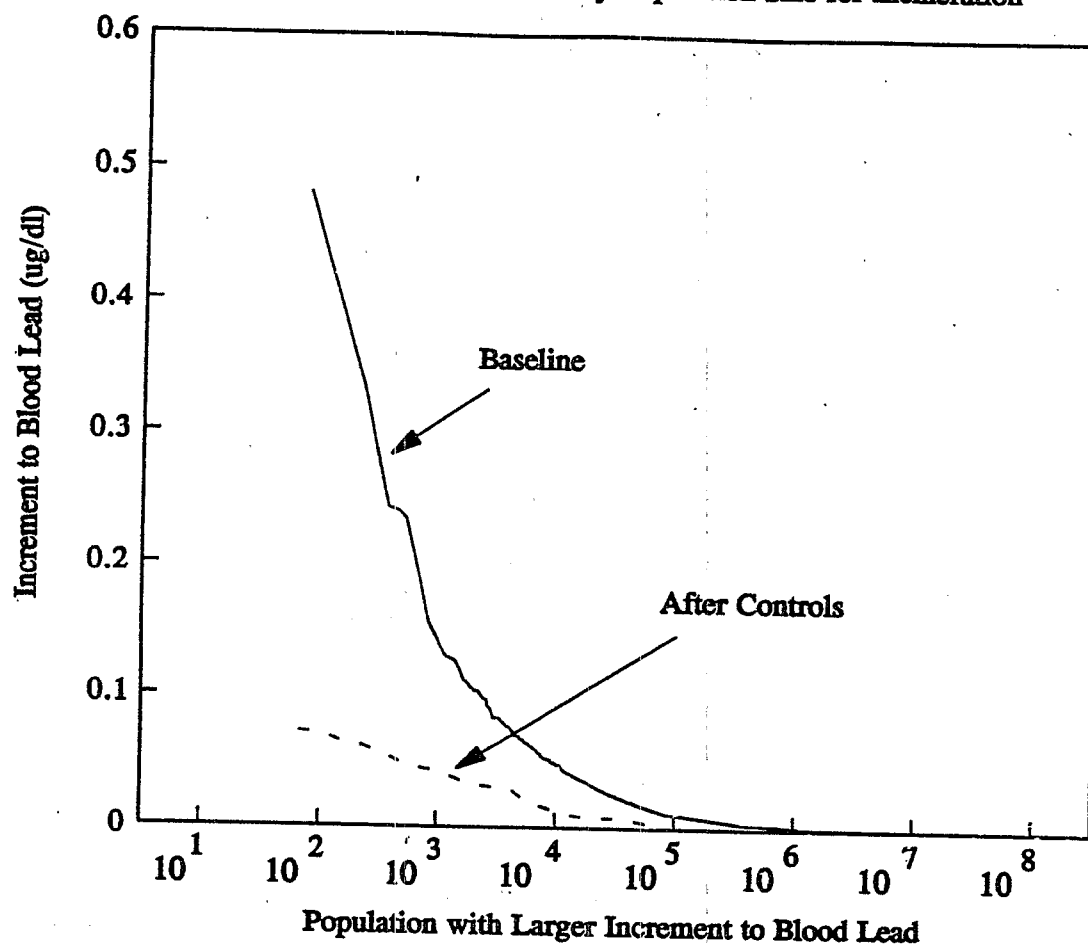
4. LAND APPLICATION: DIETARY PATHWAYS

4.0 INTRODUCTION

Each year in the United States, approximately 1.4 million dry metric tons of municipal sludge are applied to a variety of land types, including forests, roadsides, nurseries, pasture, and agricultural land. Approximately one million dry metric tons of municipal sludge are applied each year to land used for production of animal feed or human food crops. Land application, while serving as a method for managing sludge, also serves the beneficial purpose of fertilizing and conditioning soil. There are several pathways, however, through which small quantities of heavy metal and organic contaminants in the sludge might affect human health. This chapter evaluates the application of sludge to pasture and cropland, and examines the following potential pathways of exposure:

(1) Sludge is incorporated into the soil of farmland used for producing food crops. Contaminants in the sludge are drawn from the soil into the tissues of those crops, and are then ingested by humans who consume the crops directly.

(2) Sludge is incorporated into the soil of farmland used for producing animal feeds or for pasture. Contaminants in the sludge are absorbed into the tissues of these feeds or pasture grasses, which are then consumed by livestock. Meat and dairy products from these livestock are consumed by humans.

(3) Sludge is applied to the surface of pasture land, and adheres to pasture grasses. Grazing cattle or lambs ingest the sludge directly as a fraction of their pasture consumption. Humans then consume beef, dairy products, or lamb.

Section 4.1 describes the methods we use to estimate health risks for both the total exposed population and a highly exposed individual (HEI); Section 4.2 contains the data sources and model inputs; and Section 4.3 presents both the results of the analysis for the baseline (current) practices and a discussion of potential benefits from regulations. Application of sludge to food and animal feed crops may also result in surface water, groundwater, and air contamination. Our methods for assessing human health risks through these pathways are developed in Chapter 5.

4.1 METHODOLOGY

4.1.1 Overview

To estimate health risks from land application of municipal sludge, we have created a computer model in Borland International Inc.'s Turbo Pascal programming language, and executed the model on an IBM-compatible personal computer. The model first calculates the uptake of contaminants by crops and by animals feeding on crops and pasture. It then calculates human exposure and health risks using data for human dietary consumption of these animal

products and crops. Model inputs include data on sludge application rates, concentrations of individual contaminants in the sludge, uptake rates of soil contaminants into various crop tissues, uptake rates of contaminants in animal feed into meat or dairy products, the fraction of each type of feed in animal diets, human dietary data, the fraction of crops produced in sludge-amended soil, health risk reference doses and human cancer potency slopes for sludge contaminants, and an estimate of the population exposed. Estimates of exposure and health risks for an average individual, for the HEI, and for the entire population exposed are estimated. Separate tabulations are provided for estimated cancer cases and for estimated risk of cadmium-related or lead-related diseases.

4.1.2 National Versus Local Aggregation

As discussed in Chapters 3 and 7, human health risks from the incineration or surface disposal of municipal sludge tend to be concentrated in the proximity of disposal facilities. Produce grown with land-applied sludge, however, may be distributed nationally, so the health effects from dietary exposure to sludge contaminants are not necessarily dependent on the location of land application practices. Although a plant-by-plant or state-by-state analysis of land application might consider local differences in land application practices, this analysis of health risks has been conducted at the national level of aggregation. The discussion below outlines some of the issues raised by the aggregation of the analysis of both food production and food consumption to the national level.

Food Production

Municipal sludge may be applied more frequently to certain crops than to others. In addition, land application practices may differ among states due to state regulations. However, data detailing the particular crops to which each state's sludge is applied are not available at this time. For this analysis we therefore rely on data from the U.S. Census of Agriculture to estimate the fraction of national food production originating in sludge-amended soils. In doing so, we assume that the relative frequencies of crops grown with sludge do not differ appreciably from those grown without sludge. We also assume that land application practices are uniform across all states.

Extending the analysis of crop production to a state or regional level could provide a more accurate estimate of the health risks associated with land application. With the availability of data from the National Sewage Sludge Survey, such an analysis could consider plant-specific data for quantities of sludge, application rates, and concentrations of contaminants. State-level resolution of crop information might also point out local differences among crops grown in those regions where land application is most common. Furthermore, since many individual states have their own regulations or guidelines governing the land application of sludge to agricultural land, describing statewide practices according to these restrictions would give a more accurate and realistic picture of application patterns and crop production.

One indirect means of assessing these local differences might be the use of the Agricultural Census, from which crop production and acreage is available at the county level.

If the land application activity in any particular county reflects the relative distribution of crops in that county, then those distributions, along with information about local sludge production quantities and state restrictions or regulations, could be used to estimate actual farm production by crop from sludge-amended soil.

Preliminary analysis of the AGDATC data base from the Oak Ridge National Laboratory's Radiation Shielding Information Center (Baes *et al.*, 1985) suggests that patterns of crop production in counties where land application is practiced differ appreciably from national averages. Additional investigation of these differences could offer further refinement to estimates of dietary health risks from land application.

Food Consumption

Data are not currently available to describe precisely how harvested food crops are mixed and distributed throughout the United States. In the absence of information on the transport of these crops, population risks have been computed assuming complete national mixing for all crops grown on sludge-treated soil. This assumption neglects the possibility that certain geographic areas might consume higher fractions of sludge-grown foods. Insofar as the distribution network mix departs from a complete national mix, non-cancer risks may be underestimated. In particular, if food distribution is localized around an agricultural area with a high level of land application activity, the population in that area will have a higher exposure than estimated from this analysis, and will be more likely to exceed threshold doses for non-cancer health effects. (Since no threshold dose is assumed for cancer, the food distribution pattern would not affect the number of predicted cancer cases, but it would affect the geographical distribution of those cancer cases.)

4.1.3 Description of Calculations

The risk calculations consist of four general steps. First, tissue concentrations of contaminants in each crop as a result of the land application of sludge are modeled. Second, concentrations of these contaminants are estimated for meat and dairy products. As discussed above, contaminants are assumed to enter meat and dairy products either due to animal ingestion of sludge-treated crops or direct ingestion of sludge adhering to pasture grasses. Third, the amount of each contaminant in all crops and animal products ingested by humans, as well as the background intakes of each contaminant are summed, to estimate total exposure. Finally, we use the exposure estimate together with dose-response and threshold dose information to derive an estimate of the human health risk posed by each contaminant. The application rate is then set to zero and all calculations repeated in order to determine non-cancer risks due to background intakes alone. Details of these calculations are provided below.

Determining Tissue Concentrations of Contaminants in Crops Grown on Sludge-Amended Soil

The background mass of each contaminant in the mixing zone is first calculated from the background concentration of each contaminant and the mass of soil in the mixing zone:

$$LB_j = MSH \ BS_j \ 10^{-3}$$

where:

LB_j	=	mass of contaminant j in mixing zone for one hectare of farmland as result of background concentrations in soil (kg/ha),
MSH	=	mass of soil in mixing zone of one hectare of farmland (Mg/ha),
BS_j	=	background concentration (dry wt) of pollutant j in soil (mg/kg or g/Mg), and
10^{-3}	=	constant to convert units from (g/ha) to (kg/ha).

The mass of contaminant added through the application of sludge is calculated as:

$$LA_{ij} = N_i \ C_j \ AR_i \ 10^{-3}$$

where:

LA_{ij}	=	mass of contaminant j added to one hectare of farmland by land application of sludge to crop i (kg/ha),
N_i	=	total number of years sludge is applied to farmland used to grow crop i,
C_j	=	concentration of pollutant j in sludge (mg/kg or g/Mg),
AR_i	=	application rate of sludge for crop i (dry Mg/ha), and
10^{-3}	=	constant to convert units from (g/ha) to (kg/ha).

We sum these two estimates and divide by the mass of soil in the mixing zone (adjusted for the addition of additional mass from sludge) to approximate the concentration of contaminant in treated soil:

$$CT_{ij} \approx \frac{(LB_j + LA_{ij}) \ 1000}{(N_i \ AR_i) + MSH}$$

where:

CT_{ij}	=	concentration of contaminant j in soil used to grow crop i, adjusted for background soil concentration and for additional soil mass from added sludge (mg/kg), and
1000	=	constant to convert units from (kg/Mg) to (mg/kg).

Note that as N_i approaches infinity, CT_{ij} approaches C_j, so repeated applications of sludge cannot increase the estimated concentration of contaminant in soil beyond the concentration in the sludge.

Once we have estimated the concentration of contaminant in soil, we use estimated uptake rates to calculate the expected concentration of contaminant in crops grown on that soil:

$$CD_{ij} = CT_{ij} \ U_{ij}$$

where:

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

U_{ij} = rate of uptake of pollutant j into tissue of crop i (mg/kg dry weight per mg/kg in soil).

Determining Tissue Concentrations of Contaminants in Meat and Dairy Products Produced from Animals Eating Feeds Grown with Sludge-Amended Soil

If farm animals are given feeds grown in sludge-amended soil, or allowed to graze on treated pastureland, contaminants from sludge may be absorbed into animal tissues, leading to potential exposure and risks for humans. To estimate concentrations of contaminants in animal tissues, we first derive an average concentration of each contaminant in each animal's feed mix:

$$CF_{jk} = F_{ik} \ CD_{ij} + FS_k \ C_j$$

where:

CF_{jk} = weighted average concentration of pollutant j across all food sources for animal producing meat or dairy product k (mg/kg),

F_{ik} = fraction of animal's food from crop i for animal product k (kg/kg or dimensionless), and

FS_k = direct ingestion of sludge (adherence pathway) as fraction of animal's diet for animal product k (kg/kg or dimensionless).

For cattle and lamb, this estimate includes the contribution from direct ingestion of sludge from treated pastureland. We then use animal uptake rates to convert these results into estimated dry weight concentrations of contaminants in each meat or dairy product:

$$CD_{jk} = CF_{jk} \ U_{jk}$$

where:

CD_{jk} = concentration of contaminant j in animal product k (mg/kg), and

U_{jk} = rate of uptake of contaminant j into meat or dairy product k per unit of concentration in animal's feed (mg/kg dry weight in animal tissue per mg/kg dry weight feed).

Determining Individual and Population Risks from Contaminant Ingestion Through Foods Grown in Sludge-Amended Soil

To determine human exposure to contaminants from sludge, we combine these estimated concentrations of contaminant in food products with assumptions about dietary consumption and the fraction of national produce grown in sludge-amended land:

$$EXP_j = \sum_i CD_{ij} \, FC_i \, DC_i \, 10^{-3} + \sum_k CD_{jk} \, FC_k \, DC_k \, 10^{-3}$$

where:

EXP_j	=	exposure of contaminant j from crops, meat and dairy products produced with sludge-amended soil (mg/kg-day),
FC_i	=	fraction of dietary consumption of crop i grown in sludge-amended soil (dimensionless),
DC_i	=	daily dietary consumption of crop i (g/kg-day),
FC_k	=	fraction of dietary consumption of animal product k produced with sludge (kg/kg or dimensionless),
DC_k	=	daily dietary consumption of meat or dairy product k (g/kg-day), and
10^{-3}	=	constant to convert units from (μg/kg-day) to (mg/kg-day).

Cancer Risks

For contaminants classified as carcinogens, we convert this estimate of exposure into an incremental risk of cancer for the exposed individual:

$$CI_j = EXP_j \, q_j^*$$

where:

CI_j	=	incremental cancer risk from contaminant j for exposed individual (incremental risk of developing cancer per lifetime of exposure), and
q_j^*	=	human cancer potency of contaminant j (mg/kg-day)$^{-1}$.

We assume that small incremental risks from individual contaminants are additive, so we sum to calculate the total incremental risk for exposed individuals:

$$CI = \sum_j CI_j$$

where:

CI	=	total individual cancer risk from dietary pathway (incremental risk of developing cancer within lifetime as result of land application of sludge).

We multiply this result by the size of the U.S. population and divide by the human life expectancy to estimate the incremental number of cancer cases caused annually by the application of sludge to agricultural land:

$$CP = \frac{CI \, POP}{LS}$$

where:

CP   =   total aggregate risk of cancer from dietary pathway (number of additional cancer cases expected per year as result of dietary pathways for land application),

POP   =   exposed population, and

LS   =   lifespan of average individual (yr).

Health Risks from Non-Carcinogens

For non-carcinogenic contaminants, we express risk as the ratio of expected exposure to the risk reference dose (RfD) for each contaminant of concern:

$$NCP_j = POP \qquad\qquad (IF\ EXP_j + BI_j > RfD_j)$$
$$NCP_j = 0 \qquad\qquad (IF\ EXP_j + BI_j \le RfD_j)$$

where:

NCP_j =   number of people exceeding RfD of pollutant j due to sludge exposure,

BI_j =   background intake of pollutant j (mg/kg/day), and

RfD_j =   Risk Reference Dose for pollutant j (mg/kg/day).

This calculation is repeated with and without the application of sludge; the difference in NCP_j for these two scenarios provides a measure of aggregate risk.

The methods used for estimating non-cancer health effects from lead and cadmium differ from those used for other non-carcinogenic contaminants, and are explained in Chapter 2. Lead's effects are estimated separately for men, women, and children. The model bases calculations on estimated blood lead levels, and uses nonlinear dose-response functions to estimate non-cancer effects. For cadmium, expected cases of kidney disease are estimated separately for smokers and nonsmokers. Calculations are based on kidney cadmium uptake rates, and an individual exceeding a threshold of 200 μg/g cadmium in the kidney is defined as a "case". These methods do not imply that everyone exceeding this threshold will experience kidney disease; rather, those exceeding this threshold are considered to be at risk from this health hazard.

4.2   DATA SOURCES AND MODEL INPUTS

Assessing dietary exposures and risks from the land application of wastewater sludge requires a number of assumptions and types of input data. These assumptions and their ramifications are listed in Table 4-1. This section describes the sources, assumptions, and methods used to prepare these inputs for risk calculations.

Table 4-1

Major Assumptions for Land Application: Dietary Pathway

Functional Area	Assumption	Ramification
National Level of Aggregation	Crops grown in sludge same as crops grown in unamended soil.	May under- or overestimate risk depending on whether certain crops are more or less frequently grown with sludge.
	Full mixing in national market.	Crops may be locally distributed, leading to a higher fraction of local diets consisting of sludge-amended crops, and higher total risks for non-cancer effects.
	Land application practices are same for all states.	May overestimate risks in particular areas where land application of sludge is tightly controlled.
Fraction of Food Affected by Land Application Equals the Fraction of Farmland to Which Sludge Is Applied	Land treated with sludge is as productive as land treated with other fertilizers and soil conditioners.	May over- or underestimate risk if crops in sludge-amended soil are less or more productive than crops grown with other products.
	National mixing of food supply.	See ramifications listed above for this assumption.
	Land application practices are same for all states.	May under- or overestimate risks if certain states that grow particular crops restrict land application of sludge.

Table 4-1 (continued)

Major Assumptions for Land Application: Dietary Pathway

Functional Area	Assumption	Ramification
Application Rate and Frequency	Application rate same for all crops.	Fraction of production affected by sludge might be higher for crops with lower application rates.
	As amount of sludge per unit area decreases, the amount of farmland affected by land application of sludge increases if the volume of sludge is constant.	Estimates of health risks are relatively insensitive to assumed application rate.
	Assume 20 applications of sludge and 100 percent retention of metals in soil.	Could underestimate risk if sludge is applied more than 20 times; overestimates risk if fewer applications or if metals do not remain available for plant uptake.
Background Levels of Pollutants	Model assumes zero background levels for organic pollutants.	No effect on estimated cancer risks since analysis estimates only incremental risks from organic pollutants in sludge.
Uptake Rates	Plant and animal uptake is linear.	Will overestimate uptake at high contaminant levels if plant and/or animal uptakes approach plateau.
	Metals are assumed to be taken up into fat at the same rates they are taken up into other tissues.	Almost certainly overestimates uptake of metals into fat tissues.

Table 4-1 (continued)

Major Assumptions for Land Application: Dietary Pathway

Functional Area	Assumption	Ramification
Livestock Diets	Animal feed mixes estimated from USDA data.	May under- or overestimate animal uptake of contaminant from feed depending on accuracy of these feed mix data.
	Assumes 8 percent of a grazing animal's diet consists of sludge.	May overestimate animal's ingestion of contaminants if sludge is soil-incorporated or if a waiting period longer than 30 days is observed; may underestimate ingestion for lesser waiting period.
Human Dietary Consumption	Analysis uses lifetime average consumption rates rather than estimating risks for different age groups.	May fail to detect exceedence of RfD at one or more points in an exposed individual's lifetime. Should not affect estimates of total cancer risk.

4.2.1 Application Rates

Wastewater sludge is used both as a source of plant nutrients and as a soil conditioner for agricultural applications. Application rates vary according to the crop involved, state regulations, and the purpose for which the sludge is applied. For simplicity, however, the same application rate (11 metric tons per hectare) is assumed for all crops, and sludge is assumed to be soil-incorporated for all crops except pasture.

Although application rate would at first glance seem to be an extremely sensitive input parameter for national population risk estimations, other assumptions in the model render the results relatively insensitive to choice of application rate. Lowering application rates will, of course, decrease the quantity of sludge contaminants per acre available for plant uptake. However, if baseline application rates are lowered while holding constant the volume of land-applied sludge, the amount of land to which the sludge is applied must increase proportionately, so that a larger fraction of the nation's produce is affected by land application. An increased fraction of produce from sludge-amended soil means that a larger fraction of the average American's diet consists of foods affected by land application.

Because of these two opposing factors, baseline estimates of average individual dose are not significantly decreased by spreading a constant volume of sludge over an increased area of farmland. This result relies on the assumptions that plant and animal uptakes are approximately linear at the concentrations and exposures of interest, and that all crops and animal products produced on sludge-amended land are mixed in the national markets.

On the other hand, if mixing is not complete, decreasing the application rate and increasing the area of application could reduce dietary intakes of contaminants for certain individuals who consume an above-average proportion of locally grown food. Therefore, although the choice of baseline application rates does not affect the estimates of population risk, it does affect the estimated HEI risk, since the fraction of the HEI's diet produced from sludge-amended soil is assumed to be independent of application rate.

4.2.2 Concentration of Contaminants in Sludge

Listed in Table 4-2 are average and 99th percentile concentrations of contaminants in land-applied sludge, background contaminant levels in agricultural soil, and average background contaminant intake levels from other sources. Before application of sludge, soil is assumed to be free of the organic chemicals on the list of possible sludge contaminants, but is assumed to contain background concentrations of many of the metals. This analysis assumes organic contaminants in the soil decay to negligible levels between yearly harvests; therefore they do not accumulate in soil with repeated applications of sludge. However, metals do not decay, and might remain available for plant uptake (see, for example, Heckman *et al.*, 1987). Although metals may bond to the soil and become unavailable for plant uptake, or be carried away from the topsoil by wind erosion or percolating rainwater, we conservatively assume for this analysis that metals accumulate and remain bioavailable, and that sludge is applied for 20 consecutive years to all crops except pasture.

Table 4-2

Pollutant Concentrations in Sludge and Soil

	Mean Concentration in Sludge (mg/kg)[a]	99th Percentile Concentration in Sludge (mg/kg)[a]	Background Concentration in Soil (mg/kg)[b]	Background Intake for Adult (mg/day)
Aldrin/Dieldrin	0.021	0.10	0	0
Arsenic	10	62	3	0.082
Benzo(a)pyrene	11	67	0	0
Cadmium	10	120	0.2	0.027
Chlordane	0.25	1.3	0	0
Copper	520	2,500	19	0.16
DDT/DDE/DDD	0.021	0.12	0	0
Fluoride	0	0	0	0
Heptachlor	0.020	0.10	0	0
Hexachlorobenzene	11	667	0	0
Hexachlorobutadiene	0	0	0	0
Iron	0	0	0	0
Lead	140	490	11	0.11
Lindane	0.025	0.13	0	0
Mercury	3.6	18	0.1	0.0066
Molybdenum	11	51	2	0
Nickel	66	980	18	0.17
PCBs	1.3	6.1	0	0
Selenium	7.4	49	0.21	0.12
Toxaphene	0.99	5.1	0	0
Zinc	1,300	33,000	54	13

[a] Mean and 99th percentile sludge constituent concentrations obtained from the analytic survey of the National Sewage Sludge Survey. Samples where contaminant was not detected have been assigned values equal to the their respective limits of detection.
[b] Background soil concentrations from U.S. EPA (1988d).

For cancer, this analysis is concerned primarily with incremental risks from the land application of wastewater sludge, so background soil concentrations and intakes of carcinogens are not of particular concern. Background soil concentrations and intakes are important in the estimation of non-cancer risks, however, since these risks are assumed to have thresholds (risk reference doses or RfDs), beneath which no adverse health effect is expected. Background concentrations and background intakes must be known to determine whether an individual's total exposure to a contaminant will exceed the RfD when sludge is added to the soil.

4.2.3 Uptake Rates

Uptake Rates into Plant Tissue

Table 4-3 lists uptake rates for pasture and the eight food crop groups selected for this study. Uptake rates for this analysis were obtained from plant uptake response slopes (U.S. EPA, 1992c). The values in Table 4-3 represent geometric means of the plant response curves from field sludge experiments. The plant response curve for peanuts was set equal to that of legumes. Pasture uptake values were only available for lead, copper, and molybdenum; all other values were set equal to those for grains and cereals. All rates are expressed in units of mg/kg of dry weight plant tissue per mg/kg of dry soil. It is assumed that uptake rates for legumes or grains grown for animal feed do not differ from those grown for direct human consumption.

Uptake Rates into Meat or Dairy Products

From crop uptake rates and from information on the typical diets of livestock, the average dry weight concentrations of contaminants in animal feeds can be calculated. The rates at which animals incorporate these feed contaminants into their tissues are then used to calculate concentrations of the contaminants in meat or dairy products. Table 4-4 lists uptake rates for all meat and dairy products included in this study. The rates at which metals are taken up in animal fat were not available, so we conservatively extend metal uptake rates from nonfat tissue to fat tissue.

4.2.4 Animal Feed Mixes

Contaminant uptake rates differ among crops used for animal feed. The particular mix of these crops in the diet of livestock will therefore affect the total quantity of contaminants that animals ingest and the concentrations in the meat or dairy products produced by these animals. Commodity Maps from the U.S. Department of Agriculture (USDA, 1982) provide information on these feed mixes. Table 4-5 summarizes national feed totals (in millions of metric tons per year, mmt/year) for beef and milk cattle, boiler and layer chickens, hogs, and lambs. From these data, and from pasture data described below, the percentage that each type of crop contributes to an animal's total diet can be derived.

Table 4-3

Uptake Rates into Plant Tissue from Land Application of Sludge

	Pasture	Dry Legumes	Garden Fruits	Grains & Cereals	Leafy Vegetables	Non-Dry Legumes	Peanuts	Potatoes	Root Crops
Aldrin/Dieldrin	0	0	0	0	0	0	0	0	0
Arsenic	0.001	0.001	0.0014	0.001	0.018	0.001	0.001	0.0012	0.0025
Benzo(a)pyrene	0	0	0	0	0	0	0	0	0
Cadmium	0.0176	0.0016	0.045	0.0176	0.1116	0.0016	0.0016	0.004	0.0248
Chlordane	0	0	0	0	0	0	0	0	0
Copper	0.031	0.0041	0.019	0.0074	0.012	0.0041	0.0041	0.001	0.0032
DDT/DDE/DDD	0	0	0	0	0	0	0	0	0
Fluoride	0	0	0	0	0	0	0	0	0
Heptachlor	0	0	0	0	0	0	0	0	0
Hexachlorobenzene	0	0	0	0	0	0	0	0	0
Hexachlorobuta-diene	0	0	0	0	0	0	0	0	0
Iron	0.0016	0	0	0	0	0	0	0	0
Lead	0.0022	0.0012	0.002	0.0015	0.0016	0.0012	0.0012	0.001	0.0023
Lindane	0	0	0	0	0	0	0	0	0
Mercury	0.038	0.001	0.0114	0.038	0.0048	0.001	0.001	0.001	0.0095
Molybdenum	0.15	0.53	0	0.012	0	0.53	0.53	0	0
Nickel	0.005	0.0308	0.0066	0.005	0.0194	0.0308	0.0308	0.0085	0.0055
PCBs	0	0	0	0	0	0	0	0	0
Selenium	0.0047	0.028	0.0176	0.0047	0.0082	0.028	0.028	0.0209	0.0156
Toxaphene	0	0	0	0	0	0	0	0	0
Zinc	0.0416	0.0177	0.0258	0.0416	0.1129	0.0177	0.0177	0.014	0.0218

Source: U.S. EPA (1992c). All values in units of mg/kg dry weight of plant tissue per mg/kg dry weight of soil.

Table 4-4

Uptake Rates into Meat and Dairy Products from Animal Feed[a]

	Beef (Lean)	Beef (Fat)	Beef (Liver)	Dairy (Non-fat)	Dairy (Fat)	Egg (Non-fat)	Egg (Fat)	Lamb (Lean)	Lamb (Fat)	Poultry (Lean)	Poultry (Fat)	Pork (Lean)	Pork (Fat)
Aldrin/Dieldrin	0	2[i]	2[b]	0	2[b]	0	11.95	0	2	0	44.07	0	1.17
Arsenic	0	0	0	0	0	0	0	0	0	0	0	0	0
Benzo(a)pyrene	0	0	0	0	0	0	0	0	0	0	0	0	0
Cadmium	0.00002	0.00002	0.005	0.00002	0.00002	0.0002	0.0002	0.0001	0.0001	0.0006	0.0006	0.0001	0.0001
Chlordane	0	0.04	0.04	0	0.22	0	0.04	0	0.04	0	0.04	0	0.04
Copper	0	0	0	0	0	0	0	0	0	0	0	0	0
DDT/DDE/DDD	0	4.57	4.57	0	3.77	0	5	0	3.11	0	5	0	3.77
Fluoride	0	0	0	0	0	0	0	0	0	0	0	0	0
Heptachlor	0	4.94	4.94	0	4.27	0	3.55	0	4.19	0	3.55	0	4.19
Hexachlorobenzene	0	3	3	0	3.2	3.3	3.3	0	7	0	5.9	0	4.6
Hexachlorobutadiene	0	2.48	2.48	0	2.48	0	2.48	0	2.48	0	2.48	0	2.48
Iron	0	0	0	0	0	0	0	0	0	0	0	0	0
Lead	0	0	0	0	0	0	0	0	0	0	0	0	0
Lindane	0	0.48	0.48	0	0.48	0	0	0	0.6	0	0	0	0.6
Mercury	0.002	0.002	0.02	0.007	0.007	2.88	2.88	0.0014	0.0014	2.33	2.33	0.019	0.019
Molybdenum	0	0	0	0	0	0	0	0	0	0	0	0	0
Nickel	0	0	0	0	0	0	0	0	0	0	0	0	0
PCBs	0	2[c]	2[c]	0	2[c]	0	4	0	2[c]	0	4	0	4
Selenium	0.55	0.55	0.42	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.28	0.28
Toxaphene	0	0.85	0.85	0	0.02	0.79	0.79	0	0.74	0	0.79	0	0.79
Zinc	0	0	0	0	0	0.02	0.02	1.1	1.1	0.02	0.02	0	0

[a] Source: U.S. EPA (1988d). All values in mg/kg dry weight of animal tissue per mg/kg dry weight in feed.
[b] Modified to reflect assumption that all beef products should be the same (Fries 1982).
[c] Modified based on Baxter et al. (1983), Rozman et al. (1982), and Fairbanks and O'Connor (1984).

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Table 4-5

Animal Feed Mixes[a]

	Beef	Chicken	Eggs	Hogs	Lamb	Milk	Total
Corn	6.30	9.50	11.10	43.9	—	19.70	90.50
Grain	—	—	—	—	0.50	—	0.50
Hay	73.10	—	—	—	1.06	34.00	108.16
Other Feed	2.70	0.80	1.90	2.20	—	3.60	11.20
Other Feed Grain	1.40	0.40	3.40	2.90	—	3.00	11.10
Other High Protein	0.10	0.80	0.80	1.20	—	1.00	3.90
Other Harvested Roughages	64.20	—	—	—	—	70.00	134.20
Pasture (estimated)[b]	102.51	—	—	—	1.15	99.34	203
Protein Supplement	—	—	—	—	0.05	—	0.05
Sheep Pellets	—	—	—	—	0.14	—	0.14
Soybeans	2.75	4.45	3.94	6.99	—	19.00	37.12
Wheat	0.10	0.20	0.70	0.04	—	0.30	1.34
TOTAL	253.16	16.15	21.84	57.23	2.89	249.94	601.20

[a]All values in millions of metric tons of consumption per year.
[b]From the Oak Ridge AGDATC database, national production of pasture is estimated to be approximately 203 million metric tons per year. This value is divided among beef cattle, milk cattle, and lambs in proportion to total tonnage of other feeds consumed by these animals (50.5 percent to beef cattle, 48.9 percent to milk cattle, and 0.57 percent to lambs).

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Estimates for pasture consumption (also in mmt/year) were derived from the AGDATC data base maintained by the Oak Ridge National Laboratory. The estimated national total of 202.7 million metric tons of pasture consumption per year for beef cattle, milk cattle, and lambs was divided in proportion to total tonnage of other feeds consumed by these animals to obtain estimates for pasture as a fraction of these animals' diets.

Grazing animals (beef cattle, milk cattle, and lambs) also directly ingest some sludge through its adherence to pasture grasses. We assume that the sludge has not been soil-incorporated, and that one third of the time, grazing animals are feeding on forage recently sprayed with sludge (after the normal 1-3 week waiting period). The fraction of sludge ingested by grazing animals is approximately 2.5 percent (Chaney et al., 1987) during this time. However, during the remaining two thirds of the year, the fraction of sludge ingested is assumed to be only 1 percent. Therefore, on average, the grazing animals' diet is 1.5 percent sludge. We use this value for estimating both national population risk and risks to the HEI. In each case, the mass of hay/pasture consumed by the livestock is reduced to accommodate the estimated dietary percentage from sludge ingestion. Sludge ingested by animals is assumed to have the same pollutant concentrations as the sludge had when originally land-applied: that is, none of the pollutants decay before ingestion by animals. Final livestock feed estimates are listed in Table 4-6. For this analysis, we assume that animals feed on only three crops: grains, soybeans, and pasture/hay and the entries in the table been reorganized accordingly.

4.2.5 Dietary Consumption

This analysis uses estimates of dietary consumption rates obtained by averaging the rates across sexes in the 14-16, 25-30, and 60-65 age categories, and then calculates an estimated lifetime rate based on all age categories. Rates are expressed in grams dry weight of food per kilogram body weight per day.

Disaggregation of the national population into separate age groups might identify periods in an individual's life where an RfD threshold was temporarily exceeded that would provide information potentially useful for setting national criteria. However, because cancer risks are assumed to be a linear function of exposure, disaggregating the U.S. population into smaller demographic groups would not affect estimates of total cancer risks. We therefore base our estimates of total cancer risks on estimates of national average rates of food consumption. Table 4-7 lists the dietary assumptions used in the baseline analysis.

4.2.6 Fraction of Crops Grown in Sludge-Amended Soil

To estimate human exposure to sludge contaminants in crops and animal products, we first estimate the fraction of each vegetable or meat food group in the average American's diet produced in sludge-amended soil. Since national mixing of these products is assumed, we use the fraction of national farmland on which land application is practiced to approximate the fraction of national food production and consumption affected by land-applied sludge (or "FC").

Table 4-6

Estimated Feed Mix for Each Animal Product

	Soil	Pasture	Soybeans	Grains	Total
Beef	1.5%	65.0%	10.6%	23.0%	100.0%
Dairy	1.5%	45.2%	29.7%	23.7%	100.0%
Eggs	0.0%	0.0%	20.6%	79.4%	100.0%
Lamb	1.5%	74.8%	5.1%	18.6%	100.0%
Poultry	0.0%	0.0%	30.6%	69.4%	100.0%
Pork	0.0%	0.0%	13.0%	87.0%	100.0%

Note: Assumes 1.5 percent of animal's diet is sludge, based on 7-21 day waiting period between application and grazing. Values may not sum to totals because of independent rounding.

Table 4-7

Dietary Assumptions

	Fraction of Consumption from Sludge-Amended Soil (Aggregate Risk)[a]	Fraction of Consumption from Sludge-Amended Soil (Risk to HEI)[b]	Dietary Consumption[c] (g/kg-day)
Dried Legumes	$2.1x10^{-4}$	$2.5x10^{-2}$	0.04
Garden Fruits	$2.1x10^{-4}$	$2.5x10^{-2}$	0.06
Grains and Cereals	$2.1x10^{-4}$	$2.5x10^{-2}$	1.3
Leafy Vegetables	$2.1x10^{-4}$	$2.5x10^{-2}$	0.03
Non-Dried Legumes	$2.1x10^{-4}$	$2.5x10^{-2}$	0.09
Peanuts	$2.1x10^{-4}$	$2.5x10^{-2}$	0.03
Potatoes	$2.1x10^{-4}$	$2.5x10^{-2}$	0.22
Root Crops	$2.1x10^{-4}$	$2.5x10^{-2}$	0.02
Beef (Lean)	$2.1x10^{-4}$	$9.7x10^{-2}$	0.28
Beef (Fat)	$2.1x10^{-4}$	$9.7x10^{-2}$	0.22
Beef Liver	$2.1x10^{-4}$	$9.7x10^{-2}$	0.02
Dairy (Non-Fat)	$2.1x10^{-4}$	$3.1x10^{-2}$	0.41
Dairy (Fat)	$2.1x10^{-4}$	$3.1x10^{-2}$	0.26
Eggs (Whole)	$2.1x10^{-4}$	$7.9x10^{-2}$	0.12
Lamb (Lean)	$2.1x10^{-4}$	$9.7x10^{-2}$	0.003
Lamb (Fat)	$2.1x10^{-4}$	$9.7x10^{-2}$	0.003
Poultry (Lean)	$2.1x10^{-4}$	$1.08x10^{-1}$	0.10
Poultry (Fat)	$2.1x10^{-4}$	$1.08x10^{-1}$	0.02
Pork (Lean)	$2.1x10^{-4}$	$9.7x10^{-2}$	0.13
Pork (Fat)	$2.1x10^{-4}$	$9.7x10^{-2}$	0.18

[a] Calculated from amount of sludge applied to land at an assumed rate of 11 metric tons DW/ha.
[b] From U.S. EPA (1992c).
[c] From U.S. EPA (1989h).

To obtain the fraction of farmland affected by land application, we divide the estimated annual dry tonnage of sludge for land application to food chain agricultural land by the estimated average sludge application rate. This calculation yields an estimate of the number of hectares of cropland treated with sludge. This number is then divided by the total number of hectares of non-idle farmland used to produce food and animal feed crops to estimate the percent of farmland treated with sludge. Under the assumption that crops grown on sludge-amended soil are no more or less productive than crops grown with other fertilizers or soil conditioners, and that these crops yield the same proportion of waste, export, or other non-food uses as would be expected from non-sludge-amended crops, the proportion of food and animal feed farmland to which sludge has been applied should approximately match the proportion of food production and consumption affected by land application.

According to the National Sewage Sludge Survey, approximately 970,000 dry metric tons of sewage sludge are applied annually to land for food-chain agriculture. If this sludge is applied at about 11 metric tons per hectare, an estimated 88,000 hectares of land are treated with sludge each year. Dividing this number by total non-idle hectares of land used for crops and pasture (418 million hectares) yields the estimate that approximately 0.02 percent of food-chain agricultural land is treated with sludge each year. We use this estimate to represent the percentage of the average American's diet that originates from sludge-amended land.

Since our estimates of FC are inversely related to our assumed application rates, increased (or reduced) sludge application rates reduce (or increase) our estimated FC values proportionately. Population risk estimates are sensitive to the product of application rate and FC, but relatively insensitive to the particular application rate chosen.

For the highly exposed individual (HEI), we use a different approach. This hypothetical individual is assumed to consume an above-average fraction of his or her food from crops or animals raised locally in sludge-amended soil. For these calculations, values of FC are taken from Ryan (1991). Table 4-7 shows the values for both the HEI and for the national population. Because these estimates of FC for the HEI are independent of our assumed application rates, changing the application rate would change estimated cancer risks to the HEI by equal proportions, and would also be expected to increase risks of non-cancer health effects.

4.3 RESULTS AND DISCUSSION

4.3.1 Baseline Risks: Dietary Pathway

Table 4-8 provides estimates of cancer risks from the land application of sludge to food-chain land under current conditions. As shown by the table, we estimate that about 4 cases of cancer might be caused by every 10 years that sludge is land applied. Of this total risk, more than 97 percent is caused by hexachlorobenzene and PCBs, and more than 99 percent can be attributed with the addition of aldrin/dieldrin and toxaphene. As can be seen from Table 4-3, organic contaminants in sludge are not assumed to be taken up into food crops in significant quantities. It follows that concentrations of these contaminants should also be insignificant in animal feed, so that human exposure from animal products produced with feed from sludge-amended land should also be negligible. The remaining pathway through which this analysis estimates human dietary exposure from land-applied sludge is for the adherence of sludge to pasture grasses, with subsequent direct ingestion by animals. For this pathway, cancer risks for each contaminant are determined primarily by each contaminant's concentration in sludge, its rates of uptake into animal tissue, and its human cancer potency. As can be seen from Table 4-4, estimated animal uptake rates for hexachlorobenzene and PCBs are relatively high. This fact, coupled with their relatively high human cancer potencies (from Table 1-5) accounts for their dominance of total estimated risks through dietary pathways.

We estimate that the HEI faces a risk of about 6×10^{-4} of contracting cancer as a result of dietary exposure to pollutants from land-applied sludge. As with total risks, this estimate is dominated by hexachlorobenzene. Individual risks for the average individual are much lower: about 1×10^{-7}.

Uptake rates for animal and crop tissues are available from most metals. With the exception of arsenic, none of the metals are considered potential human carcinogens for this analysis. For these contaminants, non-cancer risks are of concern. As shown in Table 4-10 for the average individual, exposure to metals from sludge is in all cases more than four orders of magnitude lower than estimated background exposure from other sources (except for molybdenum, for which background exposure is unknown.) For arsenic and zinc, these background levels of exposure exceed or are approximately equal to the risk reference dose. Nevertheless, the contribution to exposure from sewage sludge appears to be trivial. As shown in Table 4-10 for the HEI, the contribution to exposure from sewage sludge is in all cases less than background exposure, and for all metals, incremental exposure is less than five percent of the RfD.

For lead and cadmium, we use additional methods to calculate potential health effects from exposure to sludge. As discussed in Chapter 2, potential health effects from lead include increased risk for hypertension, heart attack, stroke or death for men, and neurological effects for children. As shown in Table 4-11, we estimate that about 20 persons per year might suffer adverse health effects from dietary pathways of exposure to lead from land-applied sludge. A higher number, about 300, are expected to cross blood lead thresholds of 7 μg/dl for men, 10 μg/dl for women, and 10 μg/dl for children. For cadmium, about 2 persons are expected to

Table 4-8

Baseline Cancer Risks for Land Application: Dietary Pathway

	Cancer Risk
AGGREGATE RISKS[a]	
Aldrin/Dieldrin	0.004
Arsenic	0.002
Benzo(a)pyrene	0
Chlordane	0.0002
DDT/DDD/DDE	0.0002
Heptachlor	0.002
Hexachlorobenzene	0.3
Hexachlorobutadiene	0
Lindane	9×10^{-5}
PCBs	0.1
Toxaphene	0.003
Total	0.4
AVERAGE INDIVIDUAL RISK[b]	1×10^{-7}
INDIVIDUAL RISKS FOR HEI[c]	6×10^{-4}

[a]All values in incremental number of cancer cases expected per year as result of exposure through dietary pathways from land application of sludge.
[b]Risk for average exposed individual of developing cancer from lifetime of exposure to pollutants from sludge.
[c]Risk for the Highly Exposed Individual (HEI) of developing cancer from lifetime of exposure to pollutants from sludge.

Table 4-9

Comparison of Baseline Exposure to Risk Reference Doses
for Land Application: Dietary Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose[a] (mg/kg-day)	Average Exposure Including Background[b] (mg/kg-day)	Percent RfD[c]	Average Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[c]
Arsenic	1.2×10^{-3}	3×10^{-4}	1.2×10^{-3}	390	1.9×10^{-9}	6.3×10^{-4}
Cadmium	3.8×10^{-4}	5×10^{-4}	3.8×10^{-4}	76	7.6×10^{-9}	1.5×10^{-3}
Copper	2.2×10^{-3}	5×10^{-3}	2.2×10^{-3}	45	1.7×10^{-7}	3.5×10^{-3}
Fluoride	0	6×10^{-2}	0	0	0	0
Mercury	9.4×10^{-5}	3×10^{-4}	9.4×10^{-5}	31	6.6×10^{-9}	2.2×10^{-3}
Molybdenum	0	5×10^{-3}	6.1×10^{-8}	0	6.1×10^{-8}	1.2×10^{-3}
Nickel	2.5×10^{-3}	2×10^{-2}	2.5×10^{-3}	12	6.9×10^{-8}	3.5×10^{-4}
Selenium	1.6×10^{-3}	5×10^{-3}	1.6×10^{-3}	32	2.0×10^{-8}	4.0×10^{-4}
Zinc	1.9×10^{-1}	2×10^{-1}	1.9×10^{-1}	96	2.5×10^{-6}	1.2×10^{-3}

[a]From the Integrated Risk Information System (IRIS) and Health Effects Assessment Summary Tables (HEAST).
[b]Total exposure including background intakes from other sources.
[c](Exposure/RfD) x 100.
[d]Total exposure less background intake from other sources.

Table 4-10

Comparison of Baseline Exposure of the Highly Exposed Individual to
Risk Reference Doses for Land Application: Dietary Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose[a] (mg/kg-day)	Exposure for HEI Including Background[b] (mg/kg-day)	Percent RfD[c]	Exposure for HEI Excluding Background[d] (mg/kg-day)	Percent RfD[c]
Arsenic	1.2×10^{-3}	3×10^{-4}	1.2×10^{-3}	390	2.0×10^{-6}	0.68
Cadmium	3.8×10^{-4}	5×10^{-4}	3.9×10^{-4}	78	9.5×10^{-6}	1.9
Copper	2.2×10^{-3}	5×10^{-3}	2.3×10^{-3}	46	9.6×10^{-5}	1.9
Fluoride	0	6×10^{-2}	0	0	0	0
Mercury	9.4×10^{-5}	3×10^{-4}	1.1×10^{-4}	36	1.3×10^{-5}	4.3
Molybdenum	0	5×10^{-3}	1.7×10^{-5}	0.35	1.7×10^{-5}	0.35
Nickel	2.5×10^{-3}	2×10^{-2}	2.5×10^{-3}	13	4.1×10^{-5}	0.2
Selenium	1.6×10^{-3}	5×10^{-3}	1.7×10^{-3}	34	4.0×10^{-5}	0.81
Zinc	1.9×10^{-1}	2×10^{-1}	2.0×10^{-1}	99	5.5×10^{-3}	2.8

[a]From the Integrated Risk Information System (IRIS) and Health Effects Assessment Summary Tables (HEAST).
[b]Total exposure including background intakes from other sources.
[c](Exposure/RfD) x 100.
[d]Total exposure less background intake from other sources.

Table 4-11

Baseline Risks from Lead and Cadmium
Land Application: Dietary Pathways

	Health Risk
CADMIUM (Persons Crossing Kidney Cadmium Threshold)	
Smokers	0.2
Non-Smokers	0.06
Total	0.02
LEAD (Persons Crossing Blood Lead Thresholds)	
Men	300
Women	30
Children	4
Total	300
LEAD (Estimated Cases/Yr)	
Men	10
Children	9
Total	20

Note: Individual values do not sum to totals because of independent rounding to one significant figure.

cross a kidney cadmium threshold every 10 years as a result of dietary exposure from land-applied sludge.

4.3.2 Benefits from Regulatory Controls

By limiting annual or cumulative loadings of pollutant from land-applied sludge to agricultural land, or by otherwise controlling management practices for land application, the regulation is likely to reduce potential dietary risks. The extent of that reduction could not be determined from existing data and the methods used in this analysis. However, such reductions in risk are unlikely to exceed our estimate of current (or baseline) risk. For this reason, we estimate that the likely health benefit from regulating land application of sludge to agricultural land is the avoidance of up to 0.4 cancer cases per year and about 20 cases of non-cancer disease (mostly hypertension).

5. LAND APPLICATION:
GROUNDWATER, SURFACE WATER AND AIR PATHWAYS

5.0   INTRODUCTION

Chapters 4, 5 and 6 of this report concern the land application of sludge. Chapter 4 discussed potential health risks through dietary pathways, and Chapter 6 will discuss potential risks through dietary and direct ingestion pathways for members of households who apply sludge to home gardens. This chapter estimates risks for the groundwater, surface water and air pathways of exposure from land application.

5.1   METHODOLOGY

Our general strategy for analyzing these pathways is first to determine the expected behavior of organic and inorganic contaminants loaded into soil. We begin by estimating the fraction of contaminant likely to be lost through volatilization, leaching, surface runoff, and chemical degradation. These calculations, which we refer to as "mass balance," are based on the principle that total contaminant mass is conserved; the total mass of sludge contaminant lost to these processes or retained in the soil cannot exceed the total loading.[1]

After completing the mass balance calculations, we use mathematical models to predict the movement of sludge contaminants through various environmental media. We then combine our results with data describing the densities of human populations to estimate likely human exposure and risk. Details of each of these steps are provided below.

5.1.1 Mass Balance

Our calculations of mass balance consider contaminant losses to surface erosion, volatilization, leaching, and degradation. As discussed in Chapter 4, the uptake of sludge contaminants into crop tissue is of concern from the perspective of potential exposure. However, this process is of little importance to the overall loss of contaminant from soil. As shown in Table 4-3, concentrations of contaminant in crop tissue are, at worst, of comparable magnitude to those in sludge-amended soil, and the dry mass of crop tissue harvested from treated land in a given year is small in comparison to the mass of treated soil (assumed to equal approximately 2000 metric tons per hectare within the top 15 cm). For these reasons, we do not believe plant uptake is a significant component of the mass balance for contaminant in soil.

We approximate all loss processes as first-order. In other words, we assume that rates of contaminant loss to each process are always proportional to the concentration of contaminant

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[1]For this analysis, we ignore the possibility that one contaminant may degrade into another contaminant.

remaining in the soil. We begin our calculations of mass balance by estimating first-order loss coefficients for each competing loss process.

Contaminant Losses Through Leaching

As discussed in Appendix B, a coefficient for the rate of contaminant loss to leaching is calculated as:

$$K_{lec} = \frac{NR}{(BD\,KD + \theta_w + \dot{H}\theta_a)\,d_i}$$

where:

$$\dot{H} = \frac{H}{R\,T}$$

and:

BD	=	bulk density of soil in mixing zone (kg/m³),
d_i	=	depth to which sludge is incorporated into soil (m),
H	=	Henry's Law constant for the contaminant (m³-atm/mol),
$\dot{H}$	=	Henry's Law constant for the contaminant (dimensionless at specified temperature),
KD	=	equilibrium partition coefficient for the contaminant (m³/kg),
K_{lec}	=	loss rate coefficient for leaching (yr⁻¹),
θ_a	=	air-filled porosity of soil, (dimensionless),
θ_w	=	water-filled porosity of soil, (dimensionless),
NR	=	annual recharge to groundwater beneath the land application site (m/yr),
R	=	ideal gas constant (8.21×10^{-5} m³-atm/mol-K), and
T	=	temperature (K).

Contaminant Losses to Volatilization

For organic contaminants, estimates of volatile emissions are based on equations provided by Hwang and Falco (1986). After minor adjustments to units used in the original version:

$$Na = \frac{2\,t_e\,\theta_e\,D_{ei}\,C_a}{\sqrt{\pi\,\alpha_i\,t_e}} \tag{5-1}$$

where:

Na	=	total emissions from the soil surface over time interval t_e (kg/m²),
θ_e	=	effective porosity of soil (dimensionless),
D_{ei}	=	intermediate variable to be defined below (m²/sec),
C_a	=	concentration of contaminant in air-filled pore space of treated soil (kg/m³),

α_i = intermediate variable to be defined below (m²/sec), and
t_e = duration of emissions (sec).

Hwang and Falco (1986) estimate C_a with the relation:

$$C_a = \frac{41 H}{KD\, C_s}$$

where:

C_s = concentration of adsorbed contaminant in treated soil (kg/kg), and
41 = constant to convert units from (atm-m³/mol) to (m³/m³) at approximately 298 K.

Of interest for these calculations is the relationship between the total concentration of contaminant in treated soil (in dissolved, adsorbed or vapor phase) and the concentration in vapor phase within the soil's pore space. As discussed in Appendix A, we use the more appropriate equation:

$$C_a = C_t\, /\, [BD\,KD/\dot{H} + \theta_w/\dot{H} + \theta_a]$$

where:
C_t = total concentration of contaminant in treated soil (kg/m³).

Other variables used within Equation 5-1 are:

$$D_{ei} = D_{ca}\, 10^{-4}\, \theta_e^{1/3}$$

and:

$$\alpha_i = \frac{D_{ei}}{1 + \kappa S}$$

where:

$$\kappa = \rho_{ss}\, KD/\dot{H}$$

and:

$$S = \frac{1-\theta_e}{\theta_e}$$

and where:
10^{-4} = constant to convert units from (cm²) to (m²),
D_{ca} = the molecular diffusivity of contaminant in air (cm²/sec), and
ρ_{ss} = particle density of sludge-soil mixture (kg/m³).

These equations provide an estimate of total emissions from an uncovered layer of contaminated soil as a function of time and the initial concentration of contaminant. As is evident from Equation 5-1, however, the estimated loss rate is not proportional to contaminant concentration. For consistency with methods used to estimate losses for other pathways, Equation 5-1 is evaluated for t_e equal to 1 year ($t_e = 3.2 \times 10^7$ sec), and results are used to estimate an approximate loss coefficient. Losses predicted for the first year (Na_y) are divided by the total mass of contaminant in soil to estimate the approximate fraction of available contaminant lost per unit of time. For a unit concentration (1 kg/m^3) of the contaminant in soil, the mass of contaminant 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), which is equal to the depth of incorporation (m). The estimated loss rate (in kg/m^2-yr) is converted to a comparable first-order loss coefficient (in yr^{-1}) as:

$$ K_{vol} \approx -\ln(1 - \frac{Na_y}{d_i}) \qquad (5\text{-}2) $$

where:

K_{vol} = loss rate coefficient for volatilization, used to approximate loss function described by Equation 5-1 (yr^{-1}),

Na_y = emissions from the soil surface in first year (kg/m^2-yr), and

d_i = depth of incorporation for sludge (m, equivalent to kg/m^2 for a unit concentration of contaminant in treated soil).

Because Equation 5-1 was derived by assuming the column of contaminated 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 contaminant. For such cases, Equation 5-2 cannot be evaluated and the rate coefficient is estimated from predicted emissions in the first second ($t_e = 1$):

$$ K_{vol} \approx -3.2 \times 10^7 \ln(1 - \frac{Na_s}{d_i}) $$

where:

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

3.2×10^7 = constant to convert (sec^{-1}) to (yr^{-1}).

Contaminant Losses to Erosion

Annual losses to erosion are calculated based on an average rate of soil loss for agricultural land. If contaminant is evenly incorporated into the zone of incorporation, a coefficient for losses to erosion can be calculated as:

$$ K_{ero} \approx d_e / d_i $$

where:

K_{ero} = rate coefficient for loss of contaminant to erosion from treated land (yr^{-1}), and

d_e = depth of soil eroded from site each year (m/yr).

Individual Loss Processes as Fraction of Total Loss

These three loss rate coefficients are combined with an estimated loss coefficient for degradation of the contaminant in treated soil (obtained for each contaminant from scientific literature) to yield a coefficient for the total rate at which the contaminant is lost from soil:

$$K_{tot} = K_{lec} + K_{vol} + K_{ero} + K_{deg}$$

where:

K_{deg} = degradation rate coefficient for land application site (yr^{-1}), and

K_{tot} = total loss rate for the contaminant in treated soil (yr^{-1}).

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

$$f_{lec} = K_{lec} / K_{tot} \qquad f_{vol} = K_{vol} / K_{tot}$$

$$f_{ero} = K_{ero} / K_{tot} \qquad f_{deg} = K_{deg} / K_{tot}$$

where:

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

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

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

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

5.1.2 Estimating the Concentration of Contaminants in Groundwater

After completing the mass balance calculations describe above, we use two additional steps to calculate the concentration of each contaminant in groundwater near the site:

1) Determine the concentration of contaminant in water leaching through the treated soil, and

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

Calculations for the first of these two steps differ according to whether the contaminant of concern is organic or inorganic. For organic contaminants, we conservatively assume that sludge is applied to the land indefinitely. Concentrations of organic contaminants gradually

increase in the soil until the rates of annual loss equal rates of annual loading, and steady state is achieved. Our calculations of aggregate risks are based on this steady state condition.

At steady state, the rate at which the contaminant 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:

$$FA_l = 0.001 \ AR \ C \ f_{lec}$$

where:

FA_l	=	annual flux of contaminant leaching from the treated land (kg/ha-yr),
0.001	=	constant to convert units from (g/ha-yr) to (kg/ha-yr),
AR	=	application rate for sludge (dry Mg/ha-yr),
C	=	concentration of contaminant in sludge (mg/kg), and
f_{lec}	=	fraction of total contaminant losses attributable to leaching (dimensionless).

For metals, sludge is assumed to be applied for 20 consecutive years, followed by an inactive period in which contaminant is depleted from the treated soil by leaching and erosion. To simulate potential contamination of groundwater for metals, the loading of contaminant into the unsaturated zone is "linearized" into a pulse of constant magnitude to represent the maximum annual loss of contaminant (in kg/ha-yr) occurring over the 300-year simulation period modeled. The duration of that pulse is calculated so that contaminant mass is conserved. For land application sites, the maximum rate of loss is expected in the year immediately following the last application of sludge, since the concentration of contaminant 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:

$$TP = N \ / \ [1 - e^{(-K_{tot}N)}]$$

where:

TP	=	duration of "square" wave for approximating the loading of contaminant into the top of the unsaturated soil zone (yr), and
N	=	number of consecutive years sludge is applied to site (yr).

This result is combined with an estimate of the fraction of total contaminant loss to leaching for a conservative estimate of the average flux of inorganic contaminant leaching from the land application site:

$$FA_l = \frac{N \ AR \ C \ f_{lec} \ 0.001}{TP}$$

where:

FA_l	=	annual flux of contaminant leaching from the site (kg/ha-yr),
AR	=	application rate for sludge (Mg/ha-yr), and
0.001	=	constant for converting units from (g/ha-yr) to (kg/ha-yr).

For both organic and inorganic contaminants, this flux 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 contaminant in this leachate:

$$C_{lec} = \frac{0.1 \; FA_l}{NR}$$

where:

C_{lec} = concentration of contaminant in water leaching from the treated land (mg/l),

0.1 = constant to convert units from (kg/ha-m) to (mg/l), and

NR = net recharge in treated area (m/yr).

The next step is to relate this concentration to the expected concentration of contaminant 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, 1989b,c) is used to estimate flow and transport through the unsaturated zone, and the AT123D analytical model (Yeh, 1981) is used to estimate contaminant 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 contaminants is predicted based on longitudinal dispersion, an estimated retardation coefficient derived from an equilibrium partition coefficient, and a first-order rate of contaminant 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. We assume that the flux of contaminant mass into the top of 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 contaminant 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, which 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 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 (V_d) 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 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 (1989b).

One-dimensional advective-dispersive transport is estimated with VADOFT based on the estimated mass flux of contaminant 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, which simulates contaminant 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 contaminants 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 (1986h) 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 contaminant within 300 years in a receptor well at the downgradient edge of the site.

5.1.3 Estimating the Concentration of Contaminants in Ambient Air

Two steps provide an estimate for the concentration of volatilized contaminants in air near the site:

1) Use the mass balance calculations summarized above to determine the mass of contaminant expected to volatilize from the land application site 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 contaminant in ambient air near the site.

With methods analogous to those for organic contaminants in groundwater, we first estimate the rate at which contaminant mass will volatilize from the site, based on the assumption that equilibrium has been achieved between annual contaminant loadings and total losses:

$$FA_v = 0.001 \ AR \ C \ f_{vol}$$

where:

FA_v = annual average flux of contaminant volatilizing from the treated land (kg/ha-yr), and

0.001 = constant to convert units from (g/ha-yr) to (kg/ha-yr).

The next step is to relate releases of volatilized contaminant to the expected concentrations in ambient air. The model we use to simulate transport of contaminant from treated land is described by U.S. EPA (1986h) 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 edge of the land application site. A source-receptor ratio is calculated to relate the concentration of contaminant in ambient air at that individual's location (g/m³) to the rate at which that contaminant is emitted from the treated soil (g/m²-sec):

$$9SRR = 2.032 \ \frac{A \ v}{(r' + x_y) \ u \ \sigma_z}$$

where:

SRR = source-receptor ratio (sec/m),
2.032 = empirical constant,
A = area of land application site (m²),
v = vertical term for dispersion of contaminant in air (dimensionless),
r' = distance from center of the land application site to the receptor (m),
x_y = lateral virtual distance to receptor location (m),
u = wind speed (m/sec), and
σ_z = standard deviation of the vertical distribution of 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 contaminant 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:

$$x_y = \sqrt{\frac{A}{\pi}} \cot \frac{\theta}{2}$$

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 receptor. Table 5-1 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:

$$\sigma_z = a \ x^b$$

where:

$$x = \frac{10^{-3}\sqrt{A}}{2}$$

and:

x = distance from the land application site to the receptor (km), and
10^{-3} = constant for converting units from (m) to (km).

This result is combined with the estimated average flux of contaminant to predict the average concentration of contaminant in ambient air:

$$C_{air} = \frac{FA_v \ SRR}{316}$$

where:

C_{air} = average concentration of contaminant in ambient air at the receptor location ($\mu g/m^3$), and
316 = constant for converting units from ($\mu g/m^2$-sec) to (kg/ha-yr).

5.1.4 Estimating the Concentration of Contaminants in Surface Water

The first step of the calculations for this pathway is to calculate the expected concentration of contaminant on eroded soil. For organic contaminants, this step is based on the assumption that steady state has been achieved between annual loading and losses. We use the estimated fraction of annual loss attributable to erosion and the estimated mass of soil lost from the site each year to calculate the expected concentration of each organic contaminant on eroded soil:

Table 5-1

Parameters Used to Calculate σ_z Under Stable Conditions[a]

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

[a] Source: Environmental Science and Engineering (1985).
σ_z calculated as $\sigma_z = ax^b$ where x is distance in km.

$$C_{sma} = \frac{1000 \; AR \; C f_{ero}}{ME_{sma}}$$

where:

C_{sma} = average concentration of contaminant on eroded soil from the sludge management area (mg/kg),

1000 = constant to convert units from (g/kg) to (mg/kg),

AR = application rate for sludge to treated land (dry Mg/ha-yr), and

ME_{sma} = mass of soil eroding annually from one hectare of sludge management area (kg/ha-yr).

Annual soil erosion from the site can be calculated from the bulk density of treated soil and the estimated average rate of soil loss for agricultural soil in the U.S:

$$ME_{sma} = 10,000 \; d_e \; BD$$

where 10,000 is a constant to convert units from (kg/m^2) to (kg/ha).

The calculations differ slightly for metals. As discussed in Appendix C, concentrations of metals in the soil are assumed to increase from year to year as contaminant accumulates in the soil and then to decrease after the last application of sludge. Since human exposure continues for the duration of an individual's lifetime, expected concentrations of metal contaminants in surface water are calculated based on maximum estimated average losses of contaminant through surface erosion for a period equal to the human life expectancy.

For the years in which sludge is applied, contaminant is assumed to be loaded once per year at an arbitrary rate of 1 kg/ha-yr, and lost at a continuous first-order rate (K_{tot}). The outcome of combined loading and losses is calculated numerically as:

where:

M_t = mass of contaminant in soil at end of year t (kg/ha),

K_{tot} = lumped rate coefficient for contaminant loss from treated land (yr^{-1}), and

N = number of years in which sludge is applied (yr).

After the last application, no further loading of contaminant to soil takes place, but contaminant continues to be depleted:

$$M_{LS} = M_N e^{-K_{TOT}(LS-N)}$$

where:

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

M_N = mass of contaminant in soil after N applications of sludge (kg/ha), and

LS = average human lifespan (yr).

The fraction of total, cumulative loading lost in the human lifespan is independent of the assumed application rate, and can be calculated as:

$$f_{LS} = 1 - \frac{M_{LS}}{N\,1}$$

where:

f_{LS} =   fraction of total cumulative loading lost in human lifetime (dimensionless),
1   =   arbitrary unit loading rate (kg/ha-yr).

Our estimation of average contaminant concentrations on eroded soil for a metal is identical to the corresponding step for organic contaminants, except that the mass of soil eroding per year is multiplied by the life expectancy to calculate the total mass of soil lost in that period. We multiply our estimated total loading of contaminant by the fraction expected to be lost to erosion, and divide by the total mass of eroded soil to calculate the expected average concentration of each inorganic contaminant on eroded soil:

$$C_{sma} = \frac{1000 \ AR \ C \ N f_{ero} \ f_{LS}}{ME_{sma} \ LS}$$

where:

C_{sma} =   the concentration of contaminant in soil eroding from the sludge management area (mg/kg), and
1000   =   constant to convert units from (g/kg) to (mg/kg).

The next step is to determine the extent to which eroded soil from the sludge management area is diluted by soil from the (untreated) remainder of the watershed. A "dilution factor" describes the fraction of the stream's sediment originating in the land application site:

where:

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

The sediment delivery ratio for the sludge management area (SMA) is calculated with the following empirical relationship for delivery of eroded soil from the site to the stream (U.S. EPA, 1986h):

$$S_{sma} = 0.77 \ [L_{sma}]^{-0.22}$$

where:

$$L_{sma} \quad = \quad \text{buffer zone or distance between the SMA and the receiving water body (m).}$$

The sediment delivery ratio for the watershed is calculated as (Vanoni, 1975):

$$S_{ws} = 0.872 \ [A_{ws}]^{-0.125}$$

If we assume the rates of soil erosion from the SMA and remainder of the watershed are the same, ME_{ws} and ME_{sma} cancel from the equation, and the dilution factor can be calculated as:

$$D_f = \frac{A_{sma} \cdot S_{sma}}{A_{sma} \ S_{sma} + (A_{ws} - A_{sma}) \ S_{ws}}$$

If all contaminant in stream sediment is assumed to originate in the land application site, this same fraction also describes the ratio between the average concentration of contaminant in sediment entering the stream and the average concentration in soil eroding from the site:

$$C_{sed} = DF \ C_{sma}$$

where:

C_{sed} = dry weight concentration of contaminant in eroded soil (mg/kg).

We use this estimated concentration of contaminant in the stream's sediment to calculate the expected concentration of contaminant in the water column. Once the eroded soil enters the stream, contaminant is assumed to partition at equilibrium between the solid and liquid compartments of the stream. The dissolved concentration is related to the concentration of contaminant on the eroded soil entering the stream as:

$$C_{sw} = \frac{C_{sed}}{KD_{sw} + (P_l / P_s)(1/\rho_w)}$$

where:

C_{sw} = the concentration of contaminant in surface water (mg/l),
KD_{sw} = partition coefficient for contaminant in the stream (l/kg),
P_l = fraction liquid in the water column, (kg/kg),
P_s = fraction solids in the water column (kg/kg), and
ρ_w = density of pure water (assumed to equal 1 kg/l).

For metals, a partition coefficient for the contaminant is derived with an equation from U.S. EPA (1982b):

$$KD_{sw} = \alpha \ TSS^{\beta}$$

where:

TSS = total suspended solids content of the stream (mg/l), and

α, β = contaminant-specific empirical constants.

The ratio P_l/P_s is calculated as:

$$\frac{P_l}{P_s} \approx \frac{\rho_w}{TSS \ 10^{-6}}$$

where:

10^{-6} = constant to convert units from (mg) to (kg).

5.1.5 Estimating Human Exposure and Risks

To estimate human exposure we use methods discussed in Sections 5.1.1 through 5.1.4 to estimate concentrations of each contaminant in groundwater, surface water and air near each land application site. Our calculations are based on average measured concentrations of each contaminant in sewage sludge of land-applying POTWs in the analytic survey of the National Sewage Sludge Survey (NSSS), combined with our assumptions about local geological, hydrological, and meteorological conditions.

The NSSS provides data for the quality and quantity of sludge applied to land, but lacks information describing characteristics of the individual sites at which sludge is applied. The survey also does not provide information about the number of sites treated each year by individual POTWs. For simplicity and for lack of site-specific data, we have chosen a single "reasonable worst case" prototypical site to represent sludge management areas for each of the thousands of POTWs believed to practice land application. Based on characteristics of this prototypical site (to be described below), we use mathematical models to predict concentrations of pollutants in groundwater, surface water and air near the site.

These estimated concentrations in environmental media are converted to estimates of human exposure based on assumptions about the rate at which the average individual consumes drinking water and inhales air. For air, we calculate human exposure as:

$$EXP_j = \frac{10^{-3} C_{air} I_a}{BW}$$

where:

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

10^{-3} = constant to convert units from (μg) to (mg),

C_{air} = concentration of contaminant j in air (μg/m^3),

I_a = inhalation volume (m^3/day), and

BW = average body weight (kg).

We calculate potential exposure through ingestion of contaminated groundwater and surface water as:

$$EXP_j = \frac{C_{wel} \, I_w}{BW}$$

and:

$$EXP_j = \frac{C_{sw} \, I_w}{BW}$$

where:

I_w = quantity of water ingested daily (l/day),

C_{wel} = concentration of contaminant in well water (mg/l), and

C_{sw} = concentration of contaminant in surface water (mg/l).

We calculate exposure through ingestion of fish based on our estimates for the bioaccumulation of contaminant in fish and the assumed rate of fish ingestion. Bioaccumulation is the process by which aquatic organisms accumulate contaminants, from both water and food, at concentrations higher than the ambient concentration. The process by which a compound is absorbed from water through gill membranes or other external body surfaces is called bioconcentration, and the measure of a chemical's tendency to bioconcentrate is described by the bioconcentration factor. Biomagnification, in contrast, denotes the process by which the concentration of a compound increases in different organisms occupying successive trophic levels. The combined accumulation from these two sources is represented by the bioaccumulation factor, which is calculated as the product of the bioconcentration factor and a food chain multiplier. This product is multiplied by the concentration of contaminant in surface water and the ratio of concentration in fillet to that of the whole fish to calculate exposure:

$$EXP_j = \frac{C_{sw} \, BCF \, FM \, P_f \, I_f}{BW} \qquad (5\text{-}3)$$

where:

BCF = contaminant-specific bioconcentration factor (l/kg),

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

I_f = daily consumption of fish (kg/day), and

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

To complete our calculations of risk we combine estimates of individual exposure through groundwater, surface water, ambient air and fish with estimates for the sizes of exposed populations. Based on an estimated density (persons per hectare) for human populations living near land application sites and an estimate for the area of land affected by each exposure pathway, we derive an estimate for the number of persons potentially exposed at each site. This value is combined with estimates of individual exposure and risk at each site to derive an estimate of aggregate risk. Finally, we scale by the estimated number of land application sites in the U.S. to calculate our estimates of total risk.

5.2 DATA SOURCES AND MODEL INPUTS

Within the analytic survey of the NSSS, about 100 POTWs report applying sludge to agricultural land, forests, public land, reclaimed land, dedicated land, and undefined land uses. Based on sample weights from the survey, these sites represent an estimated 4803 POTWs. As mentioned above, we use a single idealized prototype (based on reasonable worst-case soil, hydrogeologic, meteorologic, and other site conditions) to represent all land application sites. We first estimate risks for this representative site, and then scale results to the national level. Because our analysis is based on conservative assumptions for many key parameters, actual risks to the exposed population are unlikely to exceed those estimated here.

This section discusses each of the key input parameters we use to model land application sites. These include: concentrations of contaminants in sludge, site parameters, soil parameters, hydrologic parameters, chemical-specific parameters, and parameters describing human populations.

5.2.1 Sludge Parameters and Site Parameters

To estimate exposure and risk we begin by characterizing the concentrations of contaminants in land-applied sludge, and the sites to which the sludge is applied.

Concentrations of Pollutants in Sludge

Table 5-2 lists mean and 99th percentile concentrations of contaminants for POTWs reporting land application of sludge in the analytic survey of the NSSS. We use the mean values for our calculations of total (aggregate) risk, and for our calculations of individual risk for the average exposed individual. We use the 99th percentile concentrations for calculating risks for the Highly Exposed Individual (HEI).

Area of Land Application Site

The areas of individual land application sites used by plants in the NSSS are not known. We make the simplifying (probably conservative) assumption that each POTW applies all of its land-applied sludge to a single contiguous area in a given year, and calculate an expected size for that area based on plants in the NSSS. We calculate this value from the estimated total quantity of sludge used for land application (1,499,861 dry metric tons per year), the estimated number of treatment works using land application (4803), and an average application rate for agriculturally applied sludge (11 mt/ha). From these values we calculate that the average POTW treats about 28 hectares of land per year, and use this estimate as the size of our prototypical site. This result, together with other assumed characteristics of the land application site, is listed in Table 5-3.

Area of Watershed

We assume the area of the watershed is 440,300 ha, the mean size hydrologic cataloguing unit as defined by the U.S. Geological Survey, (U.S. EPA, 1990a). We also assume the entire

Table 5-2

Pollutant Concentrations in Land-Applied Sludge

	Mean Concentration in Sludge (mg/kg)	99th Percentile Concentration in Sludge (mg/kg)
Aldrin/Dieldrin	0.021	0.10
Arsenic	10	62
Benzo(a)pyrene	11	67
Cadmium	10	120
Chlordane	0.25	1.3
Copper	520	2,500
DDT/DDE/DDD	0.021	0.12
Fluoride	0	0
Heptachlor	0.020	0.10
Hexachlorobenzene	11	667
Hexachlorobutadiene	0	0
Iron	0	0
Lead	140	490
Lindane	0.025	0.13
Mercury	3.6	18
Molybdenum	11	51
Nickel	66	980
PCBs	1.3	6.1
Selenium	7.4	49
Zinc	1,300	33,000

Source: Mean and 99th percentile sludge constituent concentrations obtained from the analytic survey of the National Sewage Sludge Survey.

Table 5-3

Site and Sludge Parameters for Land Application

Sludge Management Area (ha)	28[a]
Watershed Area (ha)	440,300[b]
Depth of Incorporation for Sludge (m)	0.15[c]
Lateral Distance to Well (m)	0,1000
Width of Buffer Zone (m)	10
Wind Velocity (m/sec)	4.5[d]
Average Air Temperature (K)	288[e]
Number of Applications of Sludge	20[f]
Yearly Loss to Soil Erosion (m/yr)	0.00060[g]

[a] Estimated from total quantity of sludge used for land application, total number of land application facilities, and average application rate.
[b] Mean size hydrologic cataloging unit (U.S. EPA, 1990a).
[c] U.S. EPA (1986h).
[d] Typical value for U.S. (U.S. EPA, 1990c).
[e] U.S. EPA (1986f).
[f] Typical site life (U.S. EPA, 1983b).
[g] U.S.D.A. (1987).

watershed is used for agricultural purposes, so the topography, cover, and other characteristics of the remainder of the watershed are identical to those of the sludge management area.

Depth of Incorporation

We assume sludge is incorporated into treated soil to a depth of 6 inches or 15 cm (U.S EPA, 1986h).

Distance to Well

As a worst-case exposure scenario, we estimate the concentration of contaminants in well-water at the downgradient edge of the site. We then conservatively extend exposure to concentrations at this location to all individuals residing within a 90 degree angle centered on the downgradient direction of groundwater flow and within a distance of 1 km. Similarly, we extend estimated concentrations at 1 km downgradient from the edge of the site to all individuals residing within the same 90 degree angle and at distances ranging from 1-3 km.

Distance to Human Receptor (Inhalation Pathway)

As a worst-case exposure scenario, we assume that all persons residing within 1 km of the site are exposed to ambient air concentrations estimated at the down-wind edge of the land-application site, and that persons residing between 1 km and 3 km from the site are exposed to ambient air concentrations estimated at 1 km downwind.

Width of Buffer Zone

We assume that the width of the "buffer zone" between the land application site and the nearest stream is 10 meters (U.S. EPA, 1986h). A buffer zone of identical width is assumed to separate surface water from the remainder of the watershed.

Velocity of Wind at Ground Surface

Wind velocity affects the transport of volatilized contaminant from the site. We have selected a value of 4.5 m/sec (10 mph) for wind velocity at the ground surface. This value is based on mean annual wind speeds in the United States (U.S. EPA, 1990c).

Air Temperature

The model air temperature of 15°C represents the annual average for the U.S. (U.S. EPA, 1986f).

Application Rate

A typical application rate for agricultural utilization is 11 mt/ha (U.S. EPA, 1983b). We use this value for our calculations.

Number of Applications of Sludge

For organic contaminants, calculations are based on annual loadings of contaminant to treated land, and sludge is assumed to be applied indefinitely (year after year). For metals, we assume sludge is applied to each site for 20 years. This value is consistent with the "useful life of application sites" as described by U.S. EPA (1983b).

Yearly Loss of Soil to Erosion

The model erosion rate of 0.00060 m/yr represents a weighted average of annual soil loss rates presented in the U.S.D.A. Soil Conservation Survey National Resources Inventory Summary Report (USDA, 1987). This average value has been converted from 3.8 tons/acre-yr based on an assumed bulk density of 1400 kg/m^3 for treated soil.

5.2.2 Soil Parameters

The unsaturated zone is characterized by pore space containing both air and water, whereas pore space in the saturated zone contains water only. Because of differences in flow regimes, these two zones require different equations and input parameters for tracking contaminant transport. A simplifying assumption used for these calculations is that the basic soil characteristics (including soil type, porosity, and bulk density) of the two zones are identical. Values for parameters describing the mixing, unsaturated, and saturated soil zones are provided in Table 5-4.

Soil Type

The types of soil in the mixing zone, unsaturated, and saturated zones affect the ability of a contaminant to move vertically to the aquifer and laterally to a nearby well. In general, the ease of contaminant transport through a soil (ignoring the adsorption properties of the soil) is largely affected by the type of clay present, the shrink/swell potential of that clay, and the grain size of the soil. The less the clay shrinks and swells and the smaller the grain size of the soil, the more difficult it is for contaminants to move through that soil. Soil types in the unsaturated zone in order of increasing ease of transport are (1) non-shrinking 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, 1985d).

Sand has been selected as a reasonable worst case soil to use in model scenarios for these calculations. Wherever possible all values for parameters describing soil characteristics for model simulations are based on values estimated for sand.

Porosity of Sludge/Soil

Porosity is the ratio of the void volume of a given soil or rock mass to the total volume of that mass. If the total volume is represented by V_t and the volume of the voids by V_v, the porosity can be defined as $\theta_t = V_v/V_t$. Porosity is usually reported as a decimal fraction or

Table 5-4

Soil and Hydrologic Parameters
For Land Application

Soil Type	Sand
Porosity of Sludge/Soil	0.4[a]
Bulk Density of Sludge/Soil (kg/m^3)	1400[b]
Bulk Density of Soil in Unsaturated and Saturated Zones (kg/m^3)	1600[c]
Saturated Hydraulic Conductivity of Soil (m/hr)	0.61[d]
Water Retention Parameters	
θ_r	0.045[e]
α (m^{-1})	14.5[e]
β	2.68[e]
Fraction of Organic Carbon in Soil or Sludge	
Mixing Layer	0.01
Unsaturated Zone	0.001[f]
Saturated Zone	0.0001
Depth to Groundwater (m)	1[g]
Net Recharge or Seepage (m/yr)	0.5[h]
Thickness of Aquifer (m)	1[g]
Hydraulic Gradient	0.005[i]
Total Suspended Solids in Surface Water (mg/l)	16[j]

[a] Todd (1980), Carsel and Parrish (1988).

[b] Chaney (1992).

[c] Based on porosity of 0.4. Freeze and Cherry (1979).

[d] 95th percentile value from Carsel and Parrish (1988).

[e] Mean values reported for sand in Carsel and Parrish (1988).

[f] Lower bound of range reported in U.S. EPA (1986f).

[g] Conservative value.

[h] Average of range reported in U.S. EPA (1986f).

[i] Average value for groundwater surveyed in U.S. EPA (1986f).

[j] Geometric mean of values reported in U.S. EPA's STORET data base, U.S. EPA (1992a).

percentage, and ranges from 0 (no pore space) to 1 (no solids).

For this analysis, we assume a total porosity of 0.4, taken from Todd (1980). This value is consistent with the average value for sand (0.43) reported in Carsel and Parrish (1988). It is used to represent total porosity within the soil treated with sludge, and within the unsaturated and saturated soil zones beneath the land application site.

Effective porosity is calculated as the difference between the average saturated water content and the approximate average residual water content, and refers to the amount of interconnected pore space available for fluid flow. For these calculations, the average residual water content in the unsaturated zone is assumed to be less than 0.05, and effective porosity has been approximated with the same value used for total porosity (0.4) in mass balance and groundwater transport calculations.

Bulk Density of Sludge/Soil

The bulk density of soil is defined as the mass of dry soil divided by its total (or bulk) volume. Bulk density directly influences the retardation of solutes and is related to soil structure. In general, as soils become more compact, their bulk density increases. Bulk density can be related to the particle density and porosity of a given soil as:

$$BD = \rho_{so}(1 - \theta_t)$$

where:

BD = bulk density of soil (kg/m^3),
ρ_{so} = particle density of soil (kg/m^3), and
θ_t = porosity of soil (dimensionless).

Typical mineral soils have particle densities of about 2650 kg/m^3 (Freeze and Cherry, 1979). This value and a soil porosity of 0.4 suggest a bulk density of about 1600 kg/m^3 for pure soil, somewhat higher than the 1300-1500 kg/m^3 range typically encountered for soil mixed with sludge (Chaney, 1992). We assume the bulk density of the soil in the mixing zone is 1400 kg/m^3, and the bulk density of soil in the unsaturated and saturated zones is 1600 kg/m^3.

Saturated Hydraulic Conductivity of Soil

Saturated hydraulic conductivity refers to the ability of soil to transmit water, which is governed by the amount and interconnection of void spaces in the saturated zone. These voids may occur as a consequence of inter-granular porosity, fracturing, bedding planes, or macropores. In general, high hydraulic conductivities are associated with high rates of pollutant transport. We use a value for saturated hydraulic conductivity (0.61 m/hr) based on the 95th percentile of a probability distribution for hydraulic conductivity in sand derived by Carsel and Parrish (1988). This value thus represents a conservative or "reasonable worst case" estimate.

Unsaturated Hydraulic Conductivity of Soil

In the unsaturated zone, the hydraulic conductivity, which is based on the effective permeability of soil and fluid properties, is a function of the moisture content, which is in turn a function of the pressure head. These relationships are central to the simulation of water flow through the unsaturated zone. As inputs, the VADOFT model accepts sets of data points describing effective permeability-saturation curves and the saturation-pressure head curves. Alternatively, it accepts van Genuchten water retention parameters defining the curves (U.S. EPA, 1989b; Carsel and Parrish, 1988); we select this latter option for this analysis.

Based on soils data from the Soil Conservation Survey (SCS), Carsel and Parrish derived distributions for the three parameters required (Θ_r, α, and β) according to twelve SCS textural classifications (Carsel and Parrish, 1988). Values used for our calculations (0.045, 14.5 m^{-1}, and 2.68 for Θ_r, α, and β, respectively) correspond to the values reported for sand.

Fraction of Organic Carbon in Soil or Sludge

The model combines the fraction of organic carbon in the soil with each contaminant's organic carbon partition coefficient to determine the partitioning of contaminant between soil and water. In general, a lower fraction of organic carbon implies greater mobility for organic contaminants. The organic carbon content for sludge varies among sludge types, with mean values for various types showing a relatively narrow range of 27.6-32.6 percent (U.S. EPA, 1983b). We conservatively assume that soil within the upper 15 cm of the soil column contains 1 percent organic carbon. We select a value of 10^{-3} for the fraction of organic carbon in the unsaturated zone because it is a typical value for sand, and falls at the lower end of the range (0.001-0.01) reported for soil beneath hazardous waste disposal facilities (U.S. EPA, 1986f). The fraction of organic carbon in the saturated zone is expected to be lower than that of the unsaturated zone, and has been assigned a value of 10^{-4}, or one tenth the fraction we assume for the unsaturated zone.

Depth to Groundwater

The depth to groundwater is defined as the distance from the bottom of the mixing zone to the water table. The water table is itself defined as the subsurface boundary between the unsaturated zone (where the pore spaces contain both water and air) and the saturated zone (where the pore spaces contain water only). It may be present in any type of medium and may be either permanent or seasonal. The depth to groundwater determines the distance a contaminant must travel before reaching the aquifer, and affects the attenuation of contaminant concentration during vertical transport. As this depth increases, attenuation also tends to increase, thus reducing potential pollution of the groundwater. We use a conservative value of 1 m for the distance between treated soil and groundwater.

5.2.3 Hydrologic Parameters

Key hydrologic parameters include net recharge or seepage, the thickness of the aquifer, and the hydraulic gradient. Values used for these calculations were included in Table 5-4 and are discussed below.

Net Recharge

The primary source of most groundwater is precipitation, which passes through the ground surface and percolates to the water table. Net recharge is the volume of water reaching the water table per unit area of land, and determines the quantity of water available for transporting contaminants vertically to the water table and laterally within the aquifer. The greater the recharge rate, the greater the rate of contaminant transport, up to the point at which the amount of recharge is large enough to dilute the contaminant. Beyond that point, the two effects may cancel each other out U.S. EPA (1985b).

For land application sites, the selected recharge rate (0.5 m/yr) represents the average of a range of values presented in (U.S. EPA, 1986f).

Thickness of Aquifer

Saturated zones are considered to be aquifers if they can transmit significant volumes of water. Only aquifers are considered when selecting input parameters for these calculations. For estimating aggregate risks, we conservatively assume the thickness of the aquifer is 1 m.

Hydraulic Gradient

The hydraulic gradient is a function of local geology, groundwater recharge volumes and locations, and the influence of withdrawals (e.g., well fields). It is also very likely to be indirectly related to properties of porous media. Rarely are steep gradients associated with very high conductivities. No functional relationship exists, however, to express this relationship.

For these calculations we select a value of 0.005 m/m or 0.5 percent for the hydraulic gradient, based on an average value for groundwaters surveyed for the Hazardous Waste Management System Land Disposal Restrictions Regulation (U.S. EPA, 1986f).

Total Suspended Solids

Calculating the amount of contaminant partitioning to solid phases in the stream requires a value for the concentration of suspended solids within the stream. Raw data for total suspended solids in streams and rivers in the U.S. were obtained from the EPA's STORET database, under the field "Total Residue". The data consist of annual mean total residues for the U.S. for the years 1903 through 1991. The geometric mean of these annual values is calculated as 16.2 mg/l and the median as 16.41 mg/l. We use a rounded value of 16 mg/l for the suspended solids content of the surface water.

5.2.4 Chemical-Specific Parameters

Distribution Coefficients

Contaminant transport in soil systems is influenced by interactions between the contaminant and soil. The affinity of contaminants for soil particles may result from ion exchange in clay particles, electrostatic forces between contaminants and charged particles, and interactions with organic carbon. When all exchange and interaction sites in a soil are filled, soluble contaminants will move through the soil at the same velocity as the bulk leachate. The affinity between a soil and a contaminant is characterized by the distribution coefficient (KD). Representative KD values (in l/kg or m^3/kg) are defined as the equilibrium ratio of the contaminant concentration in soil (mg/kg) to that in associated water (mg/l or mg/m^3). Values used for this analysis are listed in Table 5-5 and discussed below. Note that the organic contaminants nitrosodimethylamine and toxaphene are not being considered in this analysis because they were never detected in the analytic survey of the NSSS.

For hydrophobic organic contaminants, KD is calculated from a contaminant's partition coefficient between organic carbon and water:

$$KD = KOC\, f_{oc}$$

where:

KD = equilibrium partition coefficient for contaminant (m^3/kg),
KOC = organic carbon partition coefficient (m^3/kg); and
f_{oc} = fraction of soil consisting of organic matter.

The organic carbon partition coefficient for a contaminant can be estimated from its octanol-water partition coefficient, which can be measured in laboratory experiments. Values of KOC used in this analysis are shown in Table 5-6 and are calculated from the following regression equation by Hassett *et al.* (1983):

$$\log(KOC) = 0.0884 + 0.909\log(KOW)$$

where:

KOW = octanol-water partition coefficient for contaminant.

With the exception of PCBs, the KOW values used for this analysis have been obtained from the CHEMEST procedures in the Graphical Exposure Modelling Systems (GEMS and PCGEMS), U.S. EPA (1988a, 1989d).

Polychlorinated biphenyls (PCBs) are a class of chemicals containing 209 possible congeners. The most common PCB mixture is Aroclor 1254, which is dominated by penta-congeners, with about equal amounts of tetra- and hexa-congeners. In a well-aged soil contaminated with PCBs, however, Aroclor 1260, which contains more penta- and hexa-congeners than tetra-congeners, is more representative of the PCBs found (O'Connor, 1992) because the less chlorinated congeners degrade more rapidly. In order to determine a

Table 5-5

Octanol-Water and Organic Carbon Partition Coefficients
for Organic Contaminants

	Log of Octanol-Water Partition Coefficient[a]	Organic Carbon Partition Coefficient[b]
Benzene	2.13	106
Benzo(a)pyrene	6.12	448,000
Bis(2-ethylhexyl)phthalate	5.11	54,100
Chlordane	5.54	133,000
DDT	6.38	772,000
Lindane	3.61	2,340
Polychlorinated biphenyls[c]	6.70	1,510,000
Trichloroethylene	2.42	194

[a] All values except for PCBs taken from the CHEMEST procedure of the Graphical Exposure Modeling System (GEMS), U.S. EPA (1989d).

[b] KOC for organic contaminants derived from KOW with Equation 6 from Chapter 15 of Hasset et al. (1983): $\log(KOC) = 0.0884 + 0.909\log(KOW)$.

[c] Based on Aroclor 1254, the most common PCB mixture in sewage sludge. Derived from O'Connor (1992) and representative values from Anderson and Parker (1990).

Table 5-6

Octanol-Water Partition Coefficients
for PCBs[a]

Congener	Number of Chlorines	Log KOW
2,4'	2	5.1
2,2',5,5'	4	6.1
All Penta	5	6.5[b]
2,2',4,4',5,5'	6	6.9
Average[c]	5.5	6.7

[a] Source: Anderson and Parker (1990).

[b] Estimated based on apparent linear relationship between number of chlorines on congener and log KOW.

[c] log KOW values for penta- and hexa-congeners averaged for representative log KOW.

representative organic carbon partition coefficient for PCBs, we calculate an average from log KOW coefficients (from Anderson and Parker, 1990) listed in Table 5-7. The log KOW for the penta-congener has been estimated to be approximately 6.5 by noting that the log KOW values are approximately linearly related to the number of chlorines in the congener. Averaging that value with the hexa-congener value gives 6.7 for the log KOW. As with other organic contaminants, the regression equation from Hassett et al., (1983) is used to convert this KOW value to an estimate of KOC.

For organic contaminants, KD is also a function of the organic carbon content of the sludge and soil. For the mixing zone, we use a conservative f_{oc} value of 0.01. The value of 10^{-4} is selected for the f_{oc} value of the saturated zone because it is a typical value for sand. The f_{oc} value of 10^{-3} is used for the unsaturated zone to reflect the higher degree of organic carbon associated with the soil directly under the sludge application or disposal site.

For metals, separate KD values are used to describe partitioning between the water and soil within the mixing, unsaturated, and saturated zones, and the partitioning between the water and the sediment in the surface water. KD values used for the mixing, unsaturated and saturated soil zones are taken from a study by Gerritse et al. (1982) and represent the results of laboratory column tests with a sludge-amended sandy loam topsoil. The KD values we use were listed in Table 5-5, and represent the geometric mean of the ranges provided by Gerritse et al. (1982). For surface water, KD values are calculated according to the following regression equation provided in U.S. EPA (1982b):

$$KD_{sw} = \alpha\ TSS^{\beta}$$

where:

KD	=	partition coefficient (l/kg),
TSS	=	total suspended solids (mg/l), and
α, β	=	coefficient and exponential constants determined from regression.

Values of α and β for each contaminant are based on data for streams (U.S. EPA, 1982b), and are presented in Table 5-8. As discussed in Section 5.2.3, we assume the concentration of suspended solids in the stream is 16 mg/l.

Degradation

Pollutant concentrations in the subsurface regime may be decreased by various degradation processes, including abiotic hydrolysis and aerobic or anaerobic microbial degradation. Although rates of hydrolysis are dependent only on Ph and temperature (and can be estimated with reasonable accuracy), estimates of rates for microbial degradation are fraught with uncertainty. This uncertainty is due to many confounding influences in the field, such as substrate availability (fraction of organic carbon present), temperature, the microbial consortium, and microbial acclimation to a given pollutant. Nevertheless, the range of microbial degradation rates obtained in the laboratory by measuring the rate of disappearance of a pollutant in various soil and water grab samples, soil column studies, etc., provides a rough estimate of the rate that microbial activity is likely to degrade a particular pollutant in the field.

Table 5-7

Distribution Coefficients for Organic and Inorganic Contaminants

	Mixing Zone (l/kg)	Unsaturated Zone (l/kg)	Saturated Zone (l/kg)
Arsenic	20	20	20
Cadmium	431	431	431
Chromium	59	59	59
Copper	98	98	98
Lead	621	621	621
Mercury	330	330	330
Nickel	63	63	63
Benzene	1.06	0.106	0.0106
Benzo(a)pyrene	4,480	448	44.8
Bis(2-ethylhexyl)phthalate	541	54.1	5.41
Chlordane	1,330	133	13.3
DDT	7,720	772	77.2
Lindane	23.4	2.34	0.234
Polychlorinated biphenyls	15,100	1,510	151
Trichloroethylene	1.94	0.194	0.0194

Note: The distribution coefficient for organic contaminants (KD) is the product of the organic carbon partition coefficient (KOC) and the fraction of organic carbon in the medium (f_{oc}). Assumes foc of 1% for mixing layer, and 0.1% and 0.01% in the unsaturated and saturated soil zones, respectively. Distribution coefficients for metals are geometric means of values reported for "sandy loam" soil in Gerritse et al. (1982).

Table 5-8

Statistical Parameters for Predicting the
Equilibrium Partitioning of Metals in Surface Water[a]

	α	β	KD_{sw} (l/kg)[b]
Arsenic	0.48×10^6	-0.7286	63,700
Cadmium	4.00×10^6	-1.1307	174,000
Chromium	3.36×10^6	-0.9304	255,000
Copper	1.04×10^6	-0.7436	132,000
Lead	0.31×10^6	-0.1856	185,000
Mercury	2.91×10^6	-1.1356	125,000
Nickel	0.49×10^6	-0.5719	100,000

[a] Source: U.S. EPA (1982b).
[b] $KD_{sw} = \alpha TSS^\beta$. Assumes $TSS = 16$ mg/l.

As shown in Table 5-9, this work utilizes several sources for representative microbial degradation rates. Where a range of values is reported by these sources, we have selected values from the lower end of the range to derive conservative estimates of health risks. Studies of biodegradation in soil have been favored over studies of biodegradation in aquatic environments. If estimates of only aerobic biodegradation rates are available for a given contaminant, a half-life for anaerobic biodegradation has been conservatively estimated to be four times longer (Howard et al., 1991). However, if available data fail to show any indication that a pollutant degraded in a particular regime, a value of zero has been assumed for the microbial degradation rate.

For the zone of incorporation for land treated with sludge, estimated rates of degradation are based on studies of microbial degradation under aerobic conditions. For the unsaturated soil zone, aerobic microbial degradation and hydrolysis are assumed to be the two dominant degradation processes. Lindane and trichloroethylene are the only two compounds that are assumed to undergo significant hydrolysis: since hydrolysis rates are far more accurately quantifiable than microbial degradation rates, hydrolysis rates are used for these two chemicals. For the other six organic contaminants, 10 percent of the aerobic biodegradation decay rate is assumed to be appropriate for the unsaturated zone. This decision is based on the observation that f_{oc} tends to decrease with depth in the soil, thereby reducing the amount of suitable substrate for microbial populations which might degrade these chemicals (O'Connor, 1992).

In the saturated zone, all three degradation processes can occur because some groundwater is anaerobic and some aerobic. To capture this mix of processes, an arithmetic mean has been calculated from the aerobic and anaerobic biodegradation decay rates discussed above. For lindane and trichloroethylene, the only two chemicals where hydrolysis is a significant degradation process, estimated anaerobic decay rates are significantly higher than hydrolysis rates.

For PCBs, it is difficult to assign an anaerobic degradation rate. Highly chlorinated congeners may be partially degraded very slowly in reducing conditions, but then oxidative conditions must be established for further degradation to occur. Adequate information on anaerobic degradation rates is not currently available from the scientific literature. We have conservatively assumed that anaerobic degradation of PCBs occurs at 1 percent of the aerobic biodegradation rate.

Henry's Law Constants

Henry's Law constants are used to calculate the rate at which organic contaminants volatilize from sludge. Determining appropriate values for these constants is complicated by the wide variation in estimates provided by various sources. Table 5-10 shows values taken from four different sources, along with the value selected for this analysis. Whenever possible, values are taken from Lyman et al. (1990); otherwise values are taken from: the GEMS data base (U.S. EPA, 1988a), the PCGEMS data base (U.S. EPA, 1989d), or the Aquatic Fate Process Data for Organic Priority Pollutants (U.S. EPA, 1982a). The decision process is as follows: if a value is published in Lyman et al. (1990) it is used. If not, but if two values from other sources are similar, the mean of those two values is used. If there is no value in Lyman et al.

Table 5-9

Degradation Rates

	Aerobic Degradation Rate (yr^{-1})[a]	Anaerobic Degradation Rate (yr^{-1})[b]	Unsaturated Zone Degradation Rate (yr^{-1})[c]	Saturated Zone Degradation Rate (yr^{-1})[d]
Benzene	16[e]	0[f]	1.6	0.8
Benzo(a)-pyrene	0.48[g]	0.12[h]	0.048	0.084
Bis(2-ethylhexyl) phthalate	11[i]	0[f,j]	1.1	0.55
Chlordane	0[k]	36[i]	0	18
DDT	0.04[l]	2.5[k]	0.004	1.3
Lindane	1.2[m]	8.3[n]	1.2	4.8
PCBs	0.063[o]	0.00063[p]	0.0063	0.0035
Trichloro-ethylene	0.78[q]	3.3[r]	0.78	2.0

[a] Based on microbial degradation rates, except for lindane and trichloroethylene, where hydrolysis rates are used.
[b] Based on microbial degradation rates.
[c] Estimated as 10% of aerobic biodegradation rates. Hydrolysis rates used for lindane and trichloroethylene.
[d] Estimated as the arithmetic average of the unsaturated zone degradation rates and the anaerobic degradation rate.
[e] Vaishnav and Babeu (1987).
[f] Horowitz et al. (1982).
[g] Coover and Sims (1987).
[h] Anaerobic rate assumed to equal 25% of aerobic rate.
[i] Howard et al. (1991).
[j] Shelton et al. (1984).
[k] Castro and Yoshida (1971).
[l] Stewart and Chisholm (1971).
[m] Ellington et al. (1988).
[n] Zhang et al. (1982).
[o] Fries (1982).
[p] Anaerobic rate assumed to equal 1% of aerobic rate.
[q] Dilling et al. (1975).
[r] Bouwer and McCarthy (1983).

Table 5-10

Henry's Law Constants

	GEMS[a,b] (atm-m³/mol)	PCGEMS[a,b] (atm-m³/mol)	Lyman et al.[d] (atm-m³/mol)	Aquatic Fate[a,b] (atm-m³/mol)	Selected Values[f] (dimensionless)	Reference
Benzene	3.5×10^{-3}	5.6×10^{-3} (M)	5.5×10^{-3}	5.5×10^{-3}	2.3×10^{-1}	(d)
Benzo(a)pyrene	1.9×10^{-10}	--	--	--	8.0×10^{-9}	(a)
Bis(2-ethylhexyl)phthalate	4.4×10^{-8}	1.5×10^{-5}	--	3.0×10^{-7}	1.9×10^{-6}	(a)
Chlordane	--	4.9×10^{-5} (M)	--	9.4×10^{-5}	3.0×10^{-3}	(c,e)
DDT	2.8×10^{-8}	--	3.8×10^{-5}	1.6×10^{-5}	1.6×10^{-3}	(d)
Nitrosodimethylamine	6.5×10^{-4}	2.6×10^{-7}	--	3.3×10^{-5}	1.1×10^{-5}	(c)
Lindane	4.3×10^{-6}	1.4×10^{-5}	4.8×10^{-7}	--	2.0×10^{-5}	(d)
Polychlorinated biphenyls	--	--	--	--	1.4×10^{-2}	(g)
Toxaphene	--	6.0×10^{-6} (M)	--	2.1×10^{-1}	2.5×10^{-4}	(c)
Trichloroethylene	2.9×10^{-3}	1.0×10^{-2} (M)	1.0×10^{-2}	9.1×10^{-3}	4.2×10^{-1}	(d)

[a]CHEMEST procedure in U.S. EPA (1988a).
[b]Values estimated using $\dot{H}$ = VP/WSOL where VP is the vapor pressure and WSOL is the water solubility, unless designated as a measurement (M).
[c]CHEMEST procedure in U.S. EPA (1989b).
[d]Lyman, Reehl and Rosenblatt, Handbook of Chemical Property Estimation Methods, McGraw-Hill, 1990.
[e]U.S. EPA (1982a).
[f]Converted to non-dimensional value using $\dot{H}$ = H/(RT) where T = 288 K, R is the Universal Gas Constant and H is the dimensional Henry's Law constant.
[g]Average values from Anderson and Parker (1990) adjusted to 15°C.

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and no two values agree, a measured value is chosen in preference to an estimated one. If only estimated, dissimilar values are available, the value most conservative for groundwater (i.e., the lowest Henry's Law constant) is chosen. This last circumstance occurs only for bis(2-ethylhexyl)phthalate.

The only exception to the decision process described above is for polychlorinated biphenyls (PCBs), which include a variety of possible congeners with different chemical characteristics. Anderson and Parker (1990) provide a compilation of non-dimensional Henry's Law constants for one penta-congener and three hexa-congeners. To derive a representative Henry's Law Constant for PCBs, the three values for hexa-congeners were averaged to a single value which was then averaged with the penta-congener value to obtain the single constant reported in Table 5-10.

For all organic contaminants except PCBs, the dimensional estimate of Henry's Law Constant reported in Table 5-10 has been converted to an equivalent non-dimensional constant based on an assumed temperature of 15°C (288 K) and the following equation:

$$\dot{H} = \frac{H}{R\,T}$$

where:

T	=	temperature (assumed to be 288 K),
R	=	Universal Gas Constant (m^3-atm/mol-K),
H	=	Henry's law constant (m^3-atm/mol), and
$\dot{H}$	=	Henry's Law constant (dimensionless at specified temperature).

Because Anderson and Parker report non-dimensional values for PCBs at 25°C, the average value derived from this source has been adjusted to an equivalent non-dimensional value at 15°C.

Diffusion Coefficients

As discussed in Section 5.1, volatilization of contaminant from a land application site is modeled with equations provided by Hwang and Falco (1986). These equations require estimates for the diffusivity of each contaminant in air. To calculate diffusivity coefficients, we use Wilke and Lee's method as described in Lyman et al., (1990). Our resulting estimates, which are based on a temperature of 15°C, are listed in Table 5-11.

Bioconcentration Factor and Food Chain Multiplier

Our estimates of exposure through fish consumption (surface water pathway) require an estimate for the bioaccumulation of contaminants in fish swimming within contaminated surface water. As discussed in Section 5.1.5, this bioaccumulation can be described as the product of two parameters, the bioconcentration factor (BCF) and the food chain multiplier (FM). For metals, we take values of these two parameters from U.S. EPA (1989e). For organic contaminants, we use each contaminant's octanol water partition coefficient (from Table 5-6) to derive the BCF and FM values listed in Table 5-12. The food chain multiplier is also a function

Table 5-11

Diffusion Coefficients for Contaminant in Air

	Diffusivity in Air (cm²/sec)[a]
Benzene	9.06×10^{-2}
Benzo(a)pyrene	4.58×10^{-2}
Bis(2-ethylhexyl)phthalate	3.27×10^{-2}
Chlordane	4.51×10^{-2}
DDT	4.13×10^{-2}
Lindane	4.98×10^{-2}
Polychlorinated biphenyls	5.69×10^{-2}
Trichloroethylene	8.18×10^{-2}

[a] Calculated according to Wilke and Lee's method, as described in Lyman *et al.* (1990).

Table 5-12

Bioconcentration Factors and Food Chain Multipliers

	Bioconcentration [a] Factor (l/kg)	Food Chain [b] Multiplier (dimensionless)
Arsenic	350	1
Cadmium	330	1
Chromium	130	1
Copper	120	1
Lead	180	1
Mercury	100	1
Nickel	50	1
Benzene	7.6	1
Benzo(a)pyrene	11000	10
Bis(2-ethylhexyl)phthalate	1700	10
Chlordane	3700	10
DDT	17000	10
Lindane	110	1
Polychlorinated biphenyls	31000	10
Trichloroethylene	13	1

[a] BCF values for inorganic contaminants are taken from U.S. EPA (1989e). BCF values for organic contaminants are derived from the following regression equation, taken from U.S. EPA (1990b): $\log(BCF) = 0.79 \log(K_{ow}) - 0.8$. Log(KOW) values for each contaminant are listed in Table 5-6.

[b] Food chain multipliers are determined from the procedure in U.S. EPA (1990b), assuming a Trophic Level of 3 for fish.

of the trophic level of the species under consideration. We base this analysis on a trophic level of 3, and take food chain multipliers from the following table (U.S. EPA, 1990b):

Range of Log(K_{ow})	Food Chain Multiplier
≤ 5	1
5 - 5.5	10
≥ 5.5	10

According to that source, BCF values can be calculated from the log of KOW values (based on a 3 percent lipid content) according to the following regression equation:

$$\log(BCF) = 0.79 \log(KOW) - 0.80$$

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

Inhalation Volume

Estimates of exposure through the air pathway are based on an inhalation volume of 20 m^3 of contaminated air per day.

5.2.5 Size of Exposed Population

The size of the population exposed through groundwater, surface water, and air pathways depends on:

1) the size of the affected area near each site,
2) the density of wells and human populations in the area surrounding each site, and
3) the number of land application sites.

Each of these components is described below.

Size of Affected Area

Figure 5-1 provides a schematic representation of the prototype land application site we use for this analysis. As shown in the figure, the exposed area for all three pathways is based on the area of a circle extended 3 km from the edge of the site (itself idealized as a circle), less the area of the site. For the surface water and air pathway, this entire area is used as the area of exposure. For the groundwater pathway, the area of exposure is assumed to be one quarter

Figure 5-1

Zones of Human Exposure for Groundwater, Surface Water, and Air Pathways
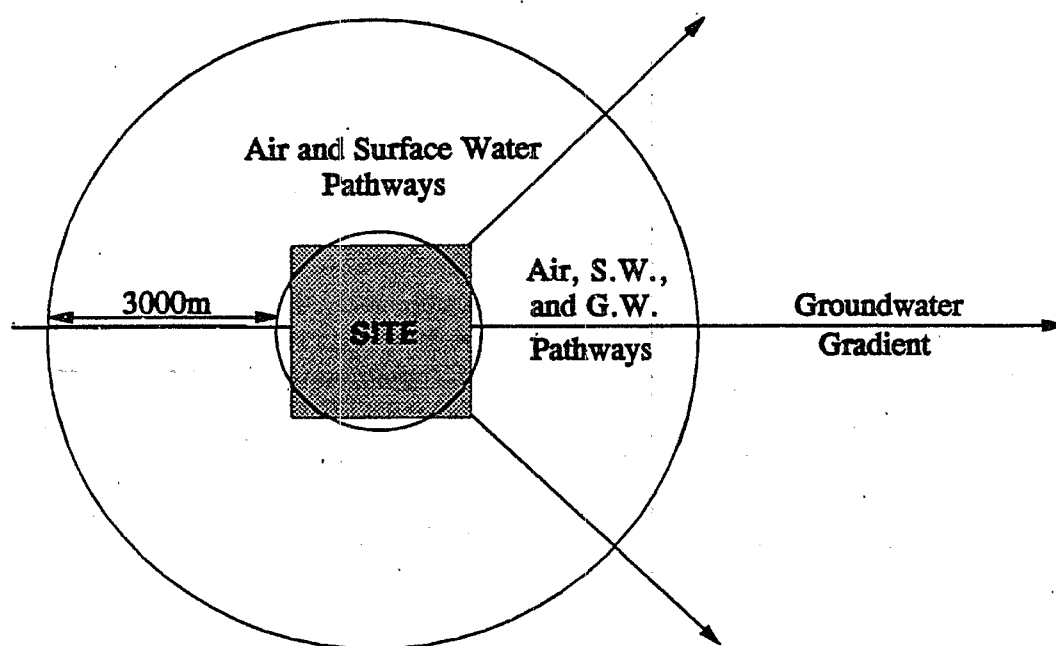
of exposure. For the groundwater pathway, the area of exposure is assumed to be one quarter of the 3 km circle, less the area of the site, and is centered on the downgradient direction of groundwater flow.

The potential for exposure to contaminated groundwater is conservatively assigned such that persons with drinking water wells within 1000 m of the site are exposed to the concentrations predicted for the downgradient edge of the site, and persons with wells located from 1000-3000 m from the site are exposed to the concentrations predicted at 1000 m downgradient of the site. Exposure from the air pathway is calculated in the same way, so that persons living within 1000 m of the site are exposed to the ambient air concentrations estimated at the property boundary, and persons living 1000-3000 m from the site are exposed to the air concentrations estimated at 1000 m from the property boundary. For the surface water pathway, all individuals living within 3000 meters of the site and using surface water as their drinking water source are assumed to be exposed to water concentrations predicted in the stream near the prototype land application site. The population potentially exposed through ingestion of fish includes all individuals living within the 3000 meter distance.

Density of Exposed Population

We have selected a representative population density by calculating the average population density of the eight states which account for more than 50 percent of land application reported in the analytic survey. Using state census data (summarized in Table 5-13), we have determined the average population density for those eight states (California, Florida, Illinois, Indiana, Michigan, Ohio, Pennsylvania, and Texas), which we apply to our prototype site. Within the exposed population, we assume that 49 percent use well water as their source of drinking water, and the remaining 51 percent use surface water.

Number of Sites

Within the analytic survey of the NSSS, about 100 plants reported the use of land application. Based on sample weights from the survey, these plants represent 4803 actual treatment facilities. To estimate risks at the national level of aggregation, we scale our estimated risks for cancer, and exposures to lead and cadmium, for the single prototypical site by a factor of 4803.

Exposed Population

Using the prototype area of 28 ha and the methods described above, the areas of exposure between 0 - 1000 m and 1000 - 3000 m are roughly 2.7 mi^2 and 15 mi^2, respectively. Combining these values with the estimated population density (194 persons/mi^2) and the total number of sites (4,803), yields the total exposed population estimates for the air pathway. The exposed populations for the groundwater pathway are adjusted to one quarter of the air pathway, multiplied by the percentage of people using well water as their drinking water source (49 percent (U.S.G.S., 1985)). The population exposed to surface water as a drinking source is determined from the total exposed to air, multiplied by the percentage of people who do not use well water as their drinking source (51 percent). Finally, the population exposed to surface

Table 5-13

State Population Densities

State	Number of Treatment Works Facilities	Population density (per mi²)[a]
California	4	190.8
Florida	4	239.6
Illinois	6	205.6
Indiana	7	154.6
Michigan	5	163.6
Ohio	9	264.9
Pennsylvania	5	265.1
Texas	5	64.9
AVERAGE:		194

[a] U.S. Bureau of the Census, 1991.

water through the ingestion of fish is set equal to the total population within a 3000 m radius of the site, or the total population exposed through the air pathway. As a result, we estimate 14 million people reside within 3000 mi radius of a site and may be exposed through the air pathway, 1.7 million people are exposed through groundwater, 7.1 million through direct consumption of surface water as a source of drinking water, and 14 million through the ingestion of fish.

5.3 RESULTS AND DISCUSSION

5.3.1 Baseline Risks: Groundwater, Surface Water, and Air Pathways

Table 5-14 provides estimates of cancer risks from the land application of sludge through groundwater, surface water, and air pathways. As shown by the table, we estimate that land application of sludge under baseline conditions causes about 1 case of cancer per ten years. This total is dominated primarily by exposure to PCBs, which accounts for about 54 percent of the total: 36 percent is from the air pathway and 18 percent is from the surface water pathway. BAP also contributes a significant amount to the total risk (about 37 percent), through the surface water pathway. The cancer risks from both PCBs and BAP through the surface water pathway are the result of ingestion of contaminated fish (due to the fact that they both have high partition coefficients and low degradation rates). Interestingly, arsenic, which has the highest individual risk ($4x10^{-5}$ for the HEI) through the groundwater pathway accounts for less than 6 percent of the total cancer risk. This is due to the smaller population exposed to groundwater, compared to that through the air and surface water pathways. However, actual risks are likely to be lower than those estimated, since these estimates are often based on generally conservative assumptions.

Exposure to non-carcinogens through the groundwater and surface water pathways is generally quite low. Average exposure from sludge (without background) is never greater than one tenth of one percent of the RfD (Table 5-15), and estimated exposure from sludge (without background) for the HEI is less than 10 percent of the RfD (Table 5-16).

5.3.2 Benefits from Regulatory Controls

By limiting annual or cumulative loadings of pollutant from land-applied sludge to agricultural land, or by otherwise controlling management practices for land application, the regulation is likely to reduce potential exposure through these pathways. However, such reductions in risk are unlikely to exceed our estimate of current (or baseline) risk. For this reason, we estimate that the likely health benefit for groundwater, surface water and air pathways for this management practice is less than one case of cancer or disease from lead per hundred years.

Table 5-14

Baseline Cancer Cases[a]
Land Application: Groundwater, Surface Water, and Air Pathways

	Groundwater	Vapor	Surface Water	Total[b]
AGGREGATE RISK				
Arsenic	<0.01	--	<0.01	<0.01
Benzene	<0.01	<0.01	<0.01	<0.01
Benzo(a)pyrene	<0.01	<0.01	0.04	0.04
Bis(2)ethylhexylphlate	<0.01	<0.01	<0.01	<0.01
Chlordane	<0.01	<0.01	<0.01	<0.01
DDT	<0.01	<0.01	<0.01	<0.01
Lindane	<0.01	<0.01	<0.01	<0.01
PCBs	<0.01	0.04	0.02	0.06
Trichloroethylene	<0.01	<0.01	<0.01	<0.01
Total	<0.01	0.04	0.06	0.1
AVERAGE INDIVIDUAL RISK[c]	1×10^{-6}	6×10^{-7}	3×10^{-7}	5×10^{-7}
INDIVIDUAL RISK FOR HEI[d]	4×10^{-5}	5×10^{-6}	5×10^{-6}	4×10^{-5}

[a] These results are based on reasonable worst-case input parameters and assumptions.
[b] Individual totals may not sum to totals because of independent rounding
[c] Risk for average exposed individual of developing cancer from lifetime of exposure to pollutants from sludge.
[d] Risk for Highly Exposed Individual (HEI).

Table 5-15

Comparison of Baseline Exposure to Risk Reference Doses
for Land Application: Groundwater, Surface Water and Air Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	Average Exposure Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg-day)	Average Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[c] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.2×10^{-3}	410%	8×10^{-7}	< 0.1%
Cadmium	3.8×10^{-4}	5×10^{-4}	3.8×10^{-4}	76%	7×10^{-12}	< 0.1%
Chromium III	1.5×10^{-3}	1×10^{0}	1.5×10^{-3}	<0.1%	3×10^{-6}	< 0.1%
Copper	2.2×10^{-3}	5×10^{-3}	2.2×10^{-3}	49%	2×10^{-6}	< 0.1%
Mercury	9.4×10^{-5}	3×10^{-4}	9.4×10^{-5}	31%	4×10^{-11}	< 0.1%
Nickel	2.5×10^{-3}	2×10^{-2}	2.5×10^{-3}	12%	1×10^{-6}	< 0.1%

[a] From the Integrated Risk Information System (IRIS) and Health Effects Assessment Summary Tables (HEAST).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Total exposure less background intake from other sources.

Table 5-16

Comparison of Baseline Exposure for the HEI to Risk Reference Doses
for Land Application: Groundwater and Air Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	HEI Exposure Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg-day)	HEI Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[c] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.3×10^{-3}	420%	2.6×10^{-5}	9%
Cadmium	3.8×10^{-4}	5×10^{-4}	3.8×10^{-4}	76%	8.5×10^{-11}	<0.1%
Chromium III	1.5×10^{-3}	1×10^{0}	1.5×10^{-3}	<0.1%	4.2×10^{-5}	<0.1%
Copper	2.2×10^{-3}	5×10^{-3}	2.2×10^{-3}	44%	1.4×10^{-5}	0.3%
Mercury	9.4×10^{-5}	3×10^{-4}	9.4×10^{-5}	31%	5.2×10^{-10}	<0.1%
Nickel	2.5×10^{-3}	2×10^{-2}	2.5×10^{-3}	12%	1.5×10^{-5}	0.1%

[a] From the Integrated Risk Information System (IRIS) and Health Effects Assessment Summary Tables (HEAST).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Total exposure less background intake from other sources.

6. LAND APPLICATION: RESIDENTIAL USES

6.0 INTRODUCTION

As discussed in Chapters 4 and 5, municipal sewage sludge is applied to cropland, forest land, reclaimed land, and other types of land application sites. In addition, some sludge is sold or given away to the public for residential uses, and can be used as a soil conditioner for either vegetable or ornamental gardening. As with other types of land application, the use of sludge for home gardening involves the conversion of a waste product to a useful resource. However, it may also present human health risks through two pathways:

(1) Sludge is acquired by home gardeners, who apply the product to their home vegetable gardens. Contaminants in the sludge enter the tissues of home-grown produce, which is then consumed by the gardening household.

(2) Sludge is acquired by home gardeners, who apply the sludge to home vegetable gardens, or to other home gardening uses such as lawns and ornamental shrubs. Young children in the gardening household ingest small quantities of the sludge-amended soil through normal mouthing behavior or through "pica" behavior. Adults also ingest small quantities of sludge-amended soil.

In this chapter, we estimate human health risks for these two pathways. Section 6.1 outlines methods we use for estimating risks from residential uses of sludge, Section 6.2 provides data sources and model inputs, and Section 6.3 discusses the results and potential benefits of regulatory controls.

6.1 METHODOLOGY

6.1.1 Overview

Health risks from residential uses of sludge have been estimated with a computer model written in Borland International Inc.'s Turbo Pascal programming language. The model was executed on an IBM-compatible personal computer to estimate human exposure and risks through dietary and soil ingestion pathways.

Inputs to the model are similar to those required for the analysis of sludge applied to food chain crops, with a few notable exceptions. This analysis assumes that produce grown in home gardens treated with sludge is consumed by the households maintaining the gardens. Thus, the population exposed to risks from residential uses of sludge is assumed to be smaller than that exposed to risks from land application to agricultural land, where produce from sludge-amended soil was assumed to be distributed through national markets. The analysis of risks from residential uses also departs from the analysis of agricultural uses in that it excludes exposure through meat and dairy products. We assume that home gardeners do not produce their own meat or dairy products from vegetables grown in their gardens.

Another difference between residential and non-residential land application is that the analysis for home gardeners pays special attention to a particular demographic group: children of ages one to six years. These children are assumed to be exposed to additional risks from the ingestion of sludge-treated soil. To estimate population risks from this pathway, we assume the average child in a gardening household ingests soil at a rate of about 0.2 grams per day. This value is based on a 1985 pilot study in which tracer elements were used to estimate the amount of soil ingested daily by 70 children living near a lead smelter (Binder et al., 1986).

Using findings based on aluminum levels measured in soil, dust, and the children's stool samples, we assume that 0.2 grams per day represents mean soil ingestion for all children. Since children are likely to ingest soil from a number of locations other than their yard or garden (e.g., schoolyard, playground), the models assume that only 10 percent of all soil ingested has been treated with sludge. The contaminant doses ingested through this pathway are then added to those the children receive by eating vegetables from the home garden. Our estimates for the amount of vegetables consumed by these children are based on the amounts reported for the one to six year age range of the Pennington (1983) study. Because exposure to this amount of soil and these quantities of vegetables is limited to approximately 5 of the 70 years in an individual's expected lifetime, we adjust estimated cancer risks for children by a factor of 5/70 when estimating total lifetime risk or incremental cases per year.

Adults are also assumed to ingest some soil, but in quantities much smaller than those ingested by children. U.S. EPA (1988d) cites a value for adults' dust ingestion of 0.02 g/day, or about 3×10^{-4} g/kg-day. As with children, we assume that only 10 percent of the total amount of soil ingested is from sludge-amended areas. Exposure through soil ingestion is added to exposure through consumption of home-grown vegetables. We take our assumed rates of consumption for vegetables from the Pennington (1983) study. Cancer effects are adjusted by a factor of 65/70 to reflect the fraction of an average lifetime when soil is consumed at adult rates.

The individual exposed to the most health risks from the use of sludge for residential purposes (the HEI) is assumed to be a young child who lives in a household using sludge on its vegetable garden and who has a tendency to ingest soil materials at above-average rates. Such a child is assumed to ingest 0.8 g of soil per day, or about 0.08 g/kg-day (U.S. EPA, 1989h).

Finally, this analysis models risks to those who use sludge only for ornamental gardening, such as landscaping or lawn care. These risks are incurred through ingestion of soil by children and adults. All assumptions regarding soil ingestion are the same as those described above. The model assumes that households that use sludge for ornamental gardening do not also use it for home vegetable gardens, and vice versa.

6.1.2 Description of Calculations

Our calculations for exposure and risk to individuals using sludge on their home gardens differ from those for land application in the following respects:

(1)   We assume that no meat is produced from crops grown in home gardens.

(2)   We do not assume that produce from home gardens is distributed to national markets. The exposed population is limited to those applying sludge to their home gardens, lawns and shrubs.

(3)   Children of home gardeners are assumed to ingest a certain average amount of garden soil daily. In addition, adults are assumed to ingest some garden soil daily, but in lesser quantities than children.

(4)   To estimate health risks for children through direct ingestion of soil, we assume children ingest 0.02 g/kg-day (average) or 0.08 g/kg-day (for the highly exposed individual) of soil. Our estimates of cancer risk through this pathway are adjusted by a factor of 5/70 to reflect the estimated five-year duration of the soil ingestion behavior. We assume that adults ingest soil at a rate of 0.0003 g/kg-day, and adjust their estimated cancer risk from soil ingestion by a factor of 65/70. For both adults and children, exposure from direct ingestion is added to exposure from ingestion of garden produce.

(5)   To estimate risks from cadmium exposure, estimated cadmium burden from childhood exposure is added to estimated burden from adult exposure when calculating adult health risks.

For households with vegetable gardens, the calculations proceed in three steps. First, we estimate concentrations of sludge constituents in the tissues of produce grown in treated gardens. Second, we calculate individual and population risks based on dietary ingestion of that produce. Finally, we sum estimated risks from all crops and soil ingestion to estimate total risk from each contaminant. For ornamental gardeners, we calculate risks from soil ingestion only. All calculations are repeated with and without application of sludge, so that health risks caused by background soil concentrations or intake from other sources are not attributed to sludge. The calculations used for each component of the analysis are described below.

Determining Tissue Concentrations of Contaminants in Produce from Sludge-Amended Gardens

We first calculate the mass of each contaminant in the mixing zone at background levels from the background concentration of each contaminant and the mass of soil in the mixing zone:

$$LB_j = MSH \ BS_j \ 10^{-3}$$

where:

LB_j =   mass of contaminant j in mixing zone as a result of background concentrations in soil (kg/ha),

MSH =   mass of soil in mixing zone (Mg/ha),

BS_j = background concentration (dry wt) of pollutant j in soil (mg/kg or g/Mg), and

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

We calculate the mass of contaminant added through the application of sludge as:

$$LA_j = N \, C_j \, AR \, 10^{-3}$$

where:

LA_j = mass of contaminant j added by application of sludge to home garden i (kg/ha),

N = number of years sludge is applied to home garden (yr),

C_j = concentration of pollutant j in sludge (mg/kg or g/Mg),

AR = application rate of sludge (dry Mg/ha), and

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

We sum these two estimates and divide by the mass of soil in the mixing zone (adjusted for additional soil mass from sludge) to approximate the concentration of contaminant in treated soil:

$$CT_j \approx \frac{(LB_j + LA_j) \; 1000}{(N \, AR) + MSH}$$

where:

CT_j = concentration of contaminant j in garden soil, adjusted for background soil concentration and for additional soil mass from added sludge (mg/kg or g/mg), and

1000 = constant to convert units from (kg/Mg) to (mg/kg).

Note that as N approaches infinity, CT_j approaches C_j, so the application of sludge cannot increase estimated concentrations of contaminant in the soil beyond the concentration in the sludge.

Once we have estimated the concentration of contaminant in soil, we use estimated uptake rates to calculate the expected concentration of contaminant in crops grown on that soil:

$$CD_{ij} = CT_j \, U_{ij}$$

where:

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

U_{ij} = rate of uptake of pollutant j into tissue of crop i (mg/kg dry weight per mg/kg in soil).

To determine human exposure to contaminants from sludge, we combine these estimated concentrations of contaminant in garden produce with assumptions about dietary consumption and the fraction of national produce grown in sludge-amended land. We also consider additional exposure that may occur through the inadvertent ingestion of small quantities of sludge by children or adults:

$$EXP_j = \sum_i CD_{ij} \, FC_i \, DC_i \, 10^{-3} + \frac{DA \, I_{sc} \, AR \, C_j \, 10^{-3}}{MSH} + \frac{(1-DA) \, I_{sa} \, AR \, C_j \, 10^{-3}}{MSH}$$

where:

EXP_j = total exposure of pollutant j from fruits or vegetables grown in the home garden (mg/kg-day),

FC_i = fraction of dietary consumption of vegetable or fruit i grown in home garden (dimensionless),

DC_i = daily dietary consumption of vegetable or fruit (g/kg-day),

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

DA = duration adjustment for childhood exposure through direct ingestion of soil (dimensionless),

I_{sc} = average rate of soil ingestion for children (g/kg-day),

LS = average human lifespan (yr), and

I_{sa} = average rate of soil ingestion for adults (g/kg-day).

Determining Individual and Population Risks from Contaminant Ingestion Through Foods or Soil from Sludge-Amended Gardens

Cancer Risks

For contaminants classified as carcinogens, we convert this estimate of exposure into an incremental risk of cancer for the exposed individual:

$$CI_j = EXP_j \, q_j^*$$

where:

CI_j = upper-bound estimate of individual cancer risk for contaminant j (probability of developing cancer from lifetime dose of EXP_j), and

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

We assume that small incremental risks from individual contaminants are additive, and sum to calculate the total incremental risk for exposed individuals:

$$CI = \sum_j CI_j$$

where:

CI   =   total individual cancer risk (incremental risk of developing cancer within lifetime as result of land application of sludge).

We combine this result with an estimate of the number of home gardens using sludge, to calculate the incremental number of cancer cases caused annually by the use of sludge in home gardens:

$$CP = \frac{CI\ POP}{LS}$$

where:

CP   =   total aggregate risk of cancer from the use of sludge on home gardens (number of additional cancer cases expected per year), and

POP   =   exposed population.

Health Risks from Non-carcinogens

For non-carcinogenic contaminants, we express risk as the ratio of expected exposure to the risk reference dose (RfD) for each contaminant of concern:

$$NCP_j = POP \qquad (IF\ EXP_j + BI_j > RfD_j)$$
$$NCP_j = 0 \qquad (IF\ EXP_j + BI_j \leq RfD_j)$$

where:

NCP_j =   number of people exceeding RfD of pollutant j due to exposure from sludge,

BI_j =   background intake of pollutant j (mg/kg-day), and

RfD_j =   Risk Reference Dose for pollutant j (mg/kg-day).

This calculation is repeated with and without the application of sludge; the difference in NCP_j for these two scenarios provides a measure of aggregate risk.

The methods we use for estimating non-cancer health effects from lead and cadmium differ from those used for other non-carcinogenic contaminants, as explained in Chapter 2. Lead's effects are estimated separately for men, women, and children. The model bases calculations on estimated blood lead levels, and uses nonlinear dose-response functions to estimate non-cancer effects. For cadmium, expected cases of kidney disease are estimated separately for smokers and nonsmokers. Calculations are based on kidney cadmium uptake rates, and exceedence of a threshold of 200 μg/g cadmium in the kidney is defined as a "case". These methods do not imply that everyone exceeding this threshold will experience kidney disease; rather, those exceeding this threshold are considered to be at risk from this health hazard.

6.2 DATA SOURCES AND MODEL INPUTS

Our assessment of health risks from residential uses of sludge relies on a number of inputs and assumptions. The discussion below describes the inputs to the risk assessment model, as well as the assumptions and data from which the inputs were derived. Table 6-1 summarizes these input parameters and assumptions. Although we use the best available estimate for each input in the baseline analysis, the true value of many inputs is uncertain.

6.2.1 Volume of Sludge to Home Gardens

According to the National Sewage Sludge Survey, about 125,000 dry metric tons of sludge are given away or sold for residential uses annually. The National Gardening Survey (1987) found that 34 million of the 69 million households involved in gardening activity in 1986 grew vegetables. We therefore assume that about half of the households using sludge apply it to vegetable gardens, while the other half apply sludge to lawns or ornamental shrubs and flowers. We also assume that approximately the same quantities of sludge are used for both ornamental and vegetable gardening.

6.2.2 Application Rates

Based on application rates estimated for agricultural uses of sludge, we assume the home gardener applies 11 dry metric tons of sludge per hectare per year to his or her home vegetable garden. If we assumed higher application rates but held constant the quantity of sludge for residential uses, we would derive a smaller estimate for the number of households affected. Population cancer risks are therefore relatively insensitive to application rate assumptions as long as cancer risks are assumed to have no threshold, but non-cancer risks would be affected by higher application rates, since heavier applications to any particular garden increase the household's probability of exceeding RfD thresholds for one or more contaminants in sludge.

6.2.3 Concentrations of Contaminants in Sludge

Table 6-2 lists mean and 99th percentile concentrations of sludge constituents, background soil concentrations for these contaminants, and adult and child background intake rates. We assume that organic contaminants do not accumulate in soil with repeated applications of sludge, but rather decay to negligible levels between yearly harvests. Metals do not decay; they may bond to the soil and become unavailable for plant uptake, or be carried away from the topsoil by wind erosion or percolating rainwater. However, field studies suggest that metals can remain available for plant uptake for several years (see, for example, Heckman et al., 1987). For this analysis we assume that sludge is applied for 20 consecutive years and that metals accumulate and remain bioavailable. Since this analysis is concerned primarily with incremental cancer risks from the land application of wastewater sludge, background soil concentrations and intakes of carcinogens are not of particular concern for the estimation of cancer risks. They are important in the estimation of non-cancer risks, however, since non-cancer risks are assumed

Table 6-1

Major Assumptions and Their Ramifications for
Land Application: Residential Uses

Functional Area	Assumption	Ramification
Number of Households Affected	Homegrown produce consumed only by gardening household.	May overestimate individual exposure and underestimate number of persons exposed if produce is shared with other households.
	Size of average garden is 0.016 ha.	Smaller/larger gardens would mean more/fewer households affected.
Residential Uses of Sludge	Half the sludge used in residential settings is for vegetable gardens, half for ornamental gardens.	May over- or underestimate number of persons using sludge for vegetable gardening, depending on true distribution of uses.
	Households use sludge for either vegetable gardening or ornamental gardening, not both.	May overestimate size of exposed population if there is overlap between these populations.
Application Rate	Sludge is applied to gardens at 11 dry metric tons/ha.	May under- or overestimate non-cancer risk if application rate is higher or lower.
	Sludge is applied for 20 consecutive years, without loss of inorganic contaminants from soil.	Likely to overestimate risks if fewer applications are applied or metals are lost from soil.
Exposure	No loss of contaminants from sludge prior to home use.	Will overestimate exposure to sludge contaminants if decay of organic contaminants or loss of metals occurs during composting or other processing.

Table 6-1 (continued)

Major Assumptions and Their Ramifications for Land Application: Residential Uses

Functional Area	Assumption	Ramification
Exposure (continued)	Plant uptake rates are linear.	May overestimate uptake at high concentrations of contaminants.
	Children ingest 0.02 g/kg-day of soil; adults ingest 0.0003 g/kg-day.	May over- or underestimate risk if more or less soil ingested.
	Only 10 percent of total daily soil ingestion consists of sludge-amended garden soil.	May under- or overestimate risk.
	Soil is free of background concentrations for organic contaminants.	Does not significantly affect cancer estimates because the analysis calculates only incremental cancer risks.
	Fraction of vegetable consumption derived from home-grown food represented by consumption pattern of non-farm, rural resident.	FC higher for rural farm residents, lower for urban residents.
Consumption	Average U.S. dietary consumption values represent consumption for exposed population.	May underestimate non-cancer risk at one or more points in a person's lifetime.

to have thresholds (risk reference doses or RfDs) below which no adverse health effect is expected. Background concentrations and background intakes must be known to determine whether an individual's total exposure to a contaminant exceeds the RfD when sludge is added to the soil.

6.2.4 Uptake Rates for Crops

Table 6-3 lists the uptake rates for sludge constituents into the tissues of vegetables grown in a typical home garden. The values represent geometric means of the plant response curves from field experiments for sludge. All rates are reported in units of mg/kg dry weight of plant tissue per mg/kg of dry soil.

6.2.5 Dietary Assumptions

The quantity and types of food consumed by those with home gardens are assumed to be the same as for those without gardens. Table 6-4 lists our assumed dietary consumption rates (in g/kg-day) for both the average adult individual and for a one to six year old child (U.S. EPA, 1989h). The values for adult consumption have been obtained by averaging consumption rates across the sexes in the 14-16, 25-30, and 60-65 age categories, and then calculating an estimated lifetime rate based on all age categories. Table 6-4 also lists assumed values for the fraction of each food consumed by a gardening household that has been grown in that household's garden (i.e., that has been produced in sludge-amended soil). These fractions are abbreviated as *FC*, and represent consumption patterns for a rural, non-farm resident (U.S. EPA, 1992c).

6.2.6 Exposed Population

Table 6.5 describes the calculations used to estimate the number of individuals exposed through dietary consumption of foods grown with residential use sludge. From the 1986-1987 National Gardening Survey, the average garden size for combined urban and rural home vegetable gardens is 1,670 square feet, or 0.016 hectares. Three additional assumptions allow estimation of the number of individuals exposed to risks from residential use of sludge: we assume that 125,058 dry metric tons of sludge are applied to residential land annually, that the sludge is applied at an average rate of 11 metric tons per hectare, and that the average gardening household includes an average of 2.7 persons. With these assumptions, we estimate that about 1,900,000 persons are potentially exposed to health risks from residential uses of sludge.

To estimate the population of children exposed through the soil ingestion pathway, we assume the proportion of children in gardening households is the same as the national average (approximately 7.5 percent). As discussed earlier, we also assume these children ingest soil at the median rate estimated for all children, and 10 percent of the soil ingested is composed of sludge-amended soil. With these assumptions, we estimate the children's exposure and health risks from contaminants in the sludge.

Table 6-2

Pollutant Concentrations in Sludge and Soil

	Mean Concentration in Sludge (mg/kg)[a]	99th Percentile Concentration in Sludge (mg/kg)[a]	Background Concentration in Soil (mg/kg)[b]	Background Intake for Adult (mg/day)	Background Intake for Child (mg/day)
Aldrin/dieldrin	0.021	0.10	0	0	0
Arsenic	10	62	3	0.082	0.041
Benzo(a)pyrene	11	67	0	0	0
Cadmium	10	120	0.2	0.027	0.016
Chlordane	0.25	1.3	0	0	0
Copper	520	2,500	19	0.16	0.074
DDT/DDE/DDD	0.021	0.12	0	0	0
Fluoride	0	0	0	0	0
Heptachlor	0.020	0.10	0	0	0
Hexachlorobenzene	11	667	0	0	0
Hexachlorobutadiene	0	0	0	0	0
Iron	0	0	0	0	0
Lead	140	490	11	0.11	0.067
Lindane	0.025	0.13	0	0	0
Mercury	3.6	18	0.1	0.0066	0.0066
Molybdenum	11	51	2	0	0
Nickel	66	980	18	0.17	0.0096
PCBs	1.3	6.1	0	0	0
Selenium	7.4	49	0.21	0.12	0.059
Toxaphene	0.99	5.1	0	0	0
Zinc	1,300	33,000	54	13	6.7

[a] Mean and 99th percentile sludge constituent concentrations obtained from the analytic survey of the National Sewage Sludge Survey.
[b] Background soil concentrations from U.S. EPA (1988d).

Table 6-3

Uptake Rates into Plant Tissue from Sludge Applied to Home Gardens[a]

	Dry Legumes	Garden Fruits	Sweet Corn	Leafy Vegetables	Non-Dry Legumes	Potatoes	Root Crops
Aldrin/Dieldrin	0	0	0	0	0	0	0
Arsenic	0.001	0.0014	0.001	0.018	0.001	0.0012	0.0025
Benzo(a)pyrene	0	0	0	0	0	0	0
Cadmium	0.0016	0.045	0.0586	0.1116	0.0016	0.004	0.0248
Chlordane	0	0	0	0	0	0	0
Copper	0.0041	0.018	0.0015	0.012	0.0041	0.001	0.0032
DDT/DDE/DDD	0	0	0	0	0	0	0
Fluoride	0	0	0	0	0	0	0
Heptachlor	0	0	0	0	0	0	0
Hexachlorobenzene	0	0	0	0	0	0	0
Hexachlorobutadiene	0	0	0	0	0	0	0
Iron	0	0	0	0	0	0	0
Lead	0.0012	0.002	0.001	0.0016	0.0012	0.001	0.0023
Lindane	0	0	0	0	0	0	0
Mercury	0.001	0.0114	0.038	0.0048	0.001	0.001	0.0095
Molybdenum	0.53	0	0	0	0.53	0	0
Nickel	0.0308	0.0066	0.005	0.0194	0.0308	0.0085	0.0055
PCBs	0	0	0	0	0	0	0
Selenium	0.028	0.0176	0.0047	0.0082	0.028	0.0209	0.0156
Toxaphene	0	0	0	0	0	0	0
Zinc	0.0177	0.0258	0.0416	0.1129	0.0177	0.014	0.0218

[a] Source: U.S. EPA (1992c). All values are mg/kg dry weight of plant tissue per mg/kg of dry soil.

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Table 6-4

Dietary Assumptions for Land Application:
Residential Uses

	Fraction of Consumption from Sludge-Amended Soil[a]	Average Consumption for Adults (g/kg/day)[b]	Average Consumption for Children (g/kg/day)[b]
Dried Legumes	0.59	0.036	0.13
Garden Fruits	0.59	0.059	0.17
Sweet Corn	0.59	0.10	0.40
Leafy Vegetables	0.59	0.028	0.049
Non-Dried Legumes	0.59	0.089	0.33
Potatoes	0.37	0.22	1.0
Root Crops	0.59	0.023	0.067

[a] From U.S. EPA (1992c).

[b] From U.S. EPA (1989h), except for corn. Values for corn have been derived from Reanalysis of FDA Study on the Revised Total Diet Food List.

Table 6-5

Estimated Population Affected By Residential Uses of Sludge

Size of Average Garden (ha)	0.016[a]
Application Rate (DMT/ha)	11
Sludge to Home Uses (DMT)	125,058[b]
Persons per Plot	2.7[c]
Persons Exposed	1,900,000

[a] From 1986-87 National Garden Survey (1987).
[b] From the Analytical Survey portion of the National Sewage Sludge Survey (U.S. EPA, 1989a).
[c] Average persons per household, U.S. Bureau of the Census, Statistical Abstracts of the United States: 1992, 112th Ed., Washington, DC., 1992.

6.3 RESULTS AND DISCUSSION

6.3.1 Baseline Risks

Our estimates of total cancer risks for residential uses of sludge are presented in Table 6-6. We estimate that about 1 case of cancer is caused by each 100 years that sludge is used for residential purposes in the U.S. Of this total, more than 85 percent is attributable to arsenic, and more than 97 percent to arsenic and benzo(a)pyrene combined. Because uptake of organic contaminants into plant tissues is assumed to be negligible (as shown in Table 6-3), exposure to organic contaminants through ingestion of home-grown vegetables is also assumed to be negligible. For this reason, all cancer risks estimated for this analysis can be attributed to either arsenic or the direct ingestion of treated soil containing organic contaminants from sludge. We estimate that the HEI faces a risk of about 1×10^{-4} of contracting cancer as a result of exposure to residential uses of sewage sludge. Individual risks for the average individual are much lower: 7×10^{-8}.

Table 6-7 compares estimated exposure to risk reference doses (RfDs) as a measure of risks from non-cancer health effects for the average individual. As can be seen from the table, our assumed background intake exceeds the RfD for arsenic and is comparable to the RfD for cadmium, copper, mercury, nickel, selenium, and zinc. However, exposure for the average individual in a household using sludge for home gardening is never greater than the RfD for any metal except arsenic. For arsenic, our assumed background level of exposure is four times greater than the RfD, but incremental exposure from sludge is less than one percent of the RfD.

Table 6.8 compares estimated exposure to the RfD for the highly exposed individual. To estimate the HEI's exposure, we combine "reasonable worst case" modeling assumptions with maximum concentrations of contaminant in sludge. The HEI is assumed to ingest 0.8 g/day of contaminated soil as a child and then to consume average quantities of home-grown vegetables throughout his or her lifetime. Each year of this hypothetical individual's lifetime, he or she ingests soil or consumes vegetables from a garden that has received 20 applications of sludge and retains 100 percent of its cumulative loading of metals. Although potential exposure to such a hypothetical individual exceeds the risk reference dose for arsenic, cadmium, copper, and zinc under baseline conditions, exposure should be reduced for these contaminants once the regulation is in place.

Table 6-9 shows estimated health risks from exposure to lead and cadmium. We estimate that about 2 persons per year exceed threshold concentrations of cadmium in their kidneys as a result of residential uses of sludge. About 1000 individuals (of which about 70 percent are men) exceed threshold concentrations of lead in blood. A smaller number (about 500) experience adverse health effects.

Table 6-6

Baseline Cancer Risks for
Land Application: Residential Uses[a]

	Cancer Risk
AGGREGATE RISKS[b]	
Aldrin/Dieldrin	4×10^{-6}
Arsenic	0.01
Benzo(a)pyrene	0.001
Chlordane	3×10^{-6}
DDT/DDE/DDD	8×10^{-8}
Heptachlor	1×10^{-6}
Hexachlorobenzene	2×10^{-4}
Hexachlorobutadiene	0
Lindane	4×10^{-7}
PCBs	1×10^{-4}
Toxaphene	1×10^{-5}
Total	0.01
AVERAGE INDIVIDUAL RISK[c]	7×10^{-8}
INDIVIDUAL RISK FOR HEI[d]	1×10^{-4}

[a] Approximate size of exposed population: 1,900,000.
[b] All values in incremental number of cancer cases expected per year as a result of exposure through residential uses of sewage sludge.
[c] Risk for average exposed individual of developing cancer from lifetime exposure to pollutants in sludge.
[d] Risk for the Highly Exposed Indivdual (HEI) of developing cancer from lifetime of exposure to pollutants in sludge.

Table 6-7

Comparison Of Baseline Exposure to Risk Reference Doses for
Land Application: Residential Uses[a]

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	Average Exposure Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg-day)	Average Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[d] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.2×10^{-3}	400	2.6×10^{-6}	0.87
Cadmium	3.8×10^{-4}	5×10^{-4}	3.9×10^{-4}	78	5.7×10^{-6}	1.1
Copper	2.2×10^{-3}	5×10^{-3}	2.3×10^{-3}	46	7.8×10^{-5}	1.6
Fluoride	0	6×10^{-2}	0	0	0	0
Mercury	9.4×10^{-5}	3×10^{-4}	9.5×10^{-5}	32	3.7×10^{-7}	0.12
Molybdenum	0	5×10^{-3}	6.8×10^{-5}	1.4	6.8×10^{-5}	1.4
Nickel	2.5×10^{-3}	2×10^{-2}	2.5×10^{-3}	13	5.8×10^{-5}	0.29
Selenium	1.6×10^{-3}	5×10^{-3}	1.6×10^{-3}	32	3.0×10^{-6}	0.06
Zinc	1.9×10^{-1}	2×10^{-1}	1.9×10^{-1}	96	6.9×10^{-4}	0.34

[a] From the Integrated Risk Information System (IRIS) and Health Effects Assessment Summary Tables (HEAST).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Exposure less background intake from other sources.

Table 6-8

Comparison Of Baseline Exposure of the Highly Exposed Individual to
Risk Reference Doses for Land Application: Residential Uses[a]

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	Exposure for HEI Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg-day)	Exposure for HEI Excluding Background[d] (mg/kg-day)	Percent RfD[d] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.6×10^{-3}	530	4.7×10^{-4}	160
Cadmium	3.8×10^{-4}	5×10^{-4}	6.6×10^{-4}	130	2.8×10^{-4}	56
Copper	2.2×10^{-3}	5×10^{-3}	6.6×10^{-3}	130	4.4×10^{-3}	88
Fluoride	0	6×10^{-2}	0	0	0	0
Mercury	9.4×10^{-5}	3×10^{-4}	1.6×10^{-4}	53	6.3×10^{-5}	21
Molybdenum	0	5×10^{-3}	4.0×10^{-4}	8	4.0×10^{-4}	8.0
Nickel	2.5×10^{-3}	2×10^{-2}	4.4×10^{-3}	22	1.9×10^{-3}	9.5
Selenium	1.6×10^{-3}	5×10^{-3}	1.8×10^{-3}	36	1.7×10^{-4}	3.4
Zinc	1.9×10^{-1}	2×10^{-1}	2.6×10^{-1}	130	7.1×10^{-2}	36

[a] From the Integrated Risk Information System (IRIS) and Health Effects Assessment Summary Tables (HEAST).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Exposure less background intake from other sources.

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Table 6-9

Baseline Noncancer Health Risks for
Land Application: Residential Uses[a]

	Health Risk
CADMIUM (Persons Crossing Kidney Cadmium Threshold	
Smokers	1
Non-Smokers	0.5
Total	2
LEAD (Persons Crossing Blood Lead Thresholds)	
Men	900
Women	100
Children	200
Total	1000
LEAD (Estimated Cases/Yr)	
Men	40
Children	500
Total	500

Note: Individual values do not sum to totals because of independent rounding to one significant figure.

6.3.2 Benefits from Regulatory Controls

Regulation of residential uses for land applied sludge might reduce these baseline risks by eliminating residential uses for sludge with relatively high concentrations of pollutants, or by otherwise altering current practices. The extent of this benefit could not be determined from existing information, but the benefit is unlikely to exceed our estimates of risks under current (baseline) conditions. If the regulation eliminated all of the existing health risk, it would provide a benefit equal to the avoidance of about 2 cancer cases per 100 years and about 500 cases of non-cancer disease per year.

7. SURFACE DISPOSAL

7.0 INTRODUCTION

The surface disposal of municipal sewage sludge can cause pollution of groundwater and thus risk to humans who drink the groundwater. Emission of volatile organic pollutants from surface disposal units can also result in exposure and risk for humans who inhale contaminated air. Significant pollution of nearby surface water bodies is assumed to be avoided by responsible management practices at surface disposal facilities; potential exposure through this pathway is not considered for this analysis.

The U.S. EPA has defined the term "surface disposal" broadly: the definition includes the disposal of sludge in waste piles, lagoons, sludge-only monofills, dedicated land-application sites, and other practices. Two idealized prototypical facilities have been defined to represent this diverse mix of related management practices: a monofill and a surface impoundment.

For the monofill prototype, the facility is represented as a sludge-only trench fill. Disposal involves the excavation of trenches in which the sludge is entirely buried below the original ground surface; de-watered sludge may be directly deposited in the trenches from a haul vehicle. Only de-watered sludges with solids content greater than or equal to 20% are assumed to be suitable for disposal and the sludge is often mixed with a bulking agent (e.g., soil) to increase solids content. Normal operating procedures require daily coverage which reduces odors and provides vector control, with a final cover placed on the monofill after closure.

In the surface impoundment prototype, the facility is assumed to receive a continuous inflow of low solids wastewater sludge. 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 and in seepage through the floor of the impoundment. Sludge entering the impoundment is assumed to have a low solids content (between 2-5 percent). Over time, particulate settling occurs and a denser layer of sediment accumulates on the floor of the lagoon. Eventually, this layer of sediment reaches the top of the impoundment and no further inflow is possible. Upon closure, the 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 facility is therefore expected to be greater for a surface impoundment, and may be sufficient to sustain a local "mounding" of the underlying water table. The surface layer of the impoundment is also assumed to be in a liquid state over the active lifetime of the facility. The volatilization of organic contaminants 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 soil cover.

As discussed in Chapters 1-6, our analysis uses the sample of facilities in the analytic survey of the NSSS to represent the complete national inventory of treatment works. For surface disposal, we have assigned each POTW reporting the use of surface disposal in the analytic component of the NSSS to either the monofill or surface impoundment categories. Those reporting the use of "dedicated" or "monofill" surface disposal are represented by the monofill prototype; those reporting the use of "other" types of surface disposal are represented by the surface impoundment prototype. In addition, some facilities originally reporting the use of land application have been reassigned to surface disposal if they appear to be applying the sludge at high rates to dedicated sites; these reassigned sites are modeled using the monofill prototype of land application.

Based on the modeling of human exposure and risk for those treatment works believed to practice surface disposal, estimates of individual and population risks are prepared for each site. Results are scaled to the national level with sample weights from the NSSS. Exposure and risk are estimated both before and after the regulation. Based on several "worst case" assumptions, we provide conservative estimates of likely risks under current or "baseline" conditions. Because most facilities are expected to comply with the numerical criteria even under baseline conditions, the actual reduction in health risks to be achieved by the regulation is expected to be significantly smaller than the predicted baseline estimates.

7.1 METHODOLOGY

As with groundwater and air pathways of exposure from land application (Chapter 5), our general strategy for evaluating these pathways is first to determine the expected behavior of organic and inorganic contaminants loaded into the surface disposal facility. We begin by estimating the fraction of contaminant likely to be lost through volatilization, leaching, and chemical degradation. These calculations, which we refer to as "mass balance," are based on the principle that contaminant mass is conserved; the total mass of sludge contaminant lost to these processes or retained in the soil cannot exceed the total loading.[1] Because the physical characteristics of the two modeled prototypes are different (one contains uncovered liquid, the other a covered mix of sludge and soil), we use different mathematical models to represent losses of contaminant for each prototype.

After completing the mass balance calculations, we use additional mathematical models to predict the movement of sludge contaminants through environmental media. We then combine our results with data describing the densities of human populations to estimate likely human exposure and risk. Details of each of these steps are provided below.

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[1] For this analysis we ignore the possibility that one contaminant may degrade into another contaminant.

7.1.1 Algorithms for the Monofill Prototype

Methodology for Mass Balance

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

Contaminant Losses Through Leaching. A coefficient for the rate of contaminant loss to leaching is calculated by assuming that contaminant mass in a filled monofill cell is partitioned at equilibrium between dissolved and adsorbed phases. Based on mathematical relationships discussed in Appendix A, the concentration of contaminant dissolved in water within the monofill can be estimated from the total concentration of contaminant within the facility:

$$C_w = \frac{C_t}{BD\,KD + \theta_w + \dot{H}\theta_a}$$

where:

$$\dot{H} = \frac{H}{RT}$$

and:

C_w	=	concentration of contaminant in water-filled pore space of sludge/soil (kg/m^3),
C_t	=	total concentration of contaminant in sludge/soil (kg/m^3),
BD	=	bulk density of sludge/soil (kg/m^3),
H	=	Henry's Law constant for the contaminant $(atm\text{-}m^3/mol)$,
$\dot{H}$	=	Henry's Law constant for the contaminant (dimensionless),
KD	=	equilibrium partition coefficient for the contaminant (m^3/kg),
θ_a	=	air-filled porosity of sludge/soil, (dimensionless),
θ_w	=	water-filled porosity of sludge/soil, (dimensionless),
R	=	ideal gas constant $(8.21 \times 10^{-5}\ m^3\text{-}atm/mol\text{-}K)$, and
T	=	temperature (K).

For an arbitrary unit concentration of contaminant in the sludge/soil (1 kg/m^3), a flux of contaminant mass leaching from the facility can be calculated as the product of net recharge (denoted NR) and the expected concentration of contaminant in leachate. Moreover, with a unit concentration of contaminant, the mass of contaminant beneath one square meter of surface is equal to the volume of sludge/soil beneath that area. This volume can be expressed as kg

contaminant/m² area/m depth of the monofill. As discussed in Appendix B, the estimated flux of leached contaminant is divided by this mass to derive a first-order loss coefficient for leaching:

$$K_{lec} = \frac{NR}{(BD\ KD + \theta_w + H\ \theta_a) \cdot d_{mf}}$$

where:

K_{lec} = loss rate coefficient for leaching from monofill (yr^{-1}),

NR = annual recharge to groundwater beneath the monofill (m/yr), and

d_{mf} = depth of a monofill cell (m).

Contaminant Losses to Volatilization. Rates of volatilization from a filled cell in a sludge monofill will vary according to whether or not a cover layer of soil has been applied. We assume that each cell in the monofill contains uncovered wastewater sludge for a few hours on each of the days it receives sludge. Following each deposit, 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 facility (U.S. EPA, 1986h). A time-weighted average of emission rates with and without cover is therefore used to describe the average rate of volatile emissions for an individual cell in the monofill. The fraction of the facility's active lifetime that a typical cell will be uncovered is calculated as:

$$f_{un} = \frac{t_{un}}{LF}$$

where:

f_{un} = fraction of facility's active lifetime that a typical cell contains sludge without soil cover,

t_{un} = time that a typical monofill cell contains uncovered sludge (yr), and

LF = active lifetime of monofill (yr).

Some monofill cells will be filled early in the facility's operation, others closer to the facility's closure. We assume that the average monofill cell will contain sludge for half the active lifetime of the facility. The fraction of the facility's active lifetime that such a cell will contain sludge protected by temporary cover is:

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

f_{co} = fraction of facility's active lifetime that typical cell contains sludge with temporary soil cover (dimensionless).

A time-weighted average rate of emissions from a typical monofill cell is calculated from equations describing emissions from a cell with and without soil cover. According to

Environmental Science and Engineering (1985) as discussed in U.S. EPA (1986h), emissions from an uncovered landfill cell can be described by:

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

(7-1)

where:

q_{un} = emission rate from treated soil for uncovered period (kg/m²-sec),
u = wind speed (m/sec),
T = temperature (K),
C_a = concentration of contaminant in air-filled pore space of treated soil (kg/m³), and
MW = molecular weight of contaminant (g/mole).

For a cell with soil cover:

$$q_{co} = \frac{9.2 \times 10^{-5} \; \theta_{ca}^{(10/3)} \; 1.005^{(T-293)} \; C_a}{\sqrt{MW} \; d_c \; \theta_{ct}^2}$$

(7-2)

where:

q_{co} = emission rate from treated soil for covered period (kg/m²-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 7-1 and 7-2 require an estimate of the concentration of contaminant in air-filled pore-space within the monofill. As explained in Appendix A, this concentration can be related to the total concentration of contaminant in sludge/soil as:

$$C_a = C_t \; / \; [BD \; KD / \dot{H} + \theta_w / \dot{H} + \theta_a]$$

Estimated emissions from an uncovered and temporarily covered monofill cell are combined to derive a time-weighted average rate of emissions from a monofill cell during the facility's active lifetime:

$$q_{ac} = q_{un} \; f_{un} + q_{co} \; f_{co}$$

(7-3)

where:

q_{ac} = time-weighted average rate of emissions from typical landfill cell over the active lifetime of the monofill (kg/m²-sec).

For a unit concentration ($C_t = 1$ kg/m³) of contaminant in sludge/soil, the mass of contaminant beneath one square meter of monofill surface (kg/m²) 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:

$$K_{va} = \frac{q_{ac}\ 3.16\times10^7}{d_{mf}} \qquad (7\text{-}4)$$

where:

K_{va} = first-order loss rate coefficient for volatilization during facility's active operation (yr^{-1}), and

3.16×10^7 = constant to convert units from (sec^{-1}) to (yr^{-1}).

Estimated coefficients for losses to volatilization and leaching are combined with assumed rates of degradation (K_{deg}, obtained for each contaminant from the scientific literature) to yield a "lumped" coefficient describing contaminant loss through all three pathways during the facility's active lifetime:

$$K_{ta} = K_{lec} + K_{va} + K_{deg}$$

where:

K_{ta} = coefficient for total rate of contaminant loss through leaching, volatilization, and degradation during facility's active operation (yr^{-1}).

We calculate the fraction of contaminant loss attributable to each individual process during the facility's active lifetime as:

$$f_{la} = \frac{K_{lec}}{K_{ta}} \qquad f_{va} = \frac{K_{va}}{K_{ta}} \qquad f_{da} = \frac{K_{deg}}{K_{ta}}$$

where:

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

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

f_{da} = fraction of total contaminant loss during facility's active operation attributable to degradation (dimensionless).

The fraction of total loading lost within the facility's active lifetime is calculated numerically from the lumped rate of contaminant loss, assuming a time step of 1 year:

$$M_t = 0 \qquad\qquad (t=0)$$
$$M_t = [M_{t-1} + 1]\ e^{-K_{ta}} \qquad (1 \leq t \leq LF)$$

and:

$$f_{ac} = 1 - \frac{M_t}{1\ LF}$$

where:

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

M_{LF} = mass of contaminant in sludge/soil at end of year LF (kg/ha), and

1 = unit loading of contaminant (kg/ha-yr).

Once the facility'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 contaminant 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 Equations 7-2 through 7-4 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:

$$K_{ti} = K_{lec} + K_{vi} + K_{deg}$$

where:

K_{ti} = coefficient for total rate of contaminant loss from inactive monofill (yr^{-1}), and

K_{vi} = coefficient for rate of contaminant loss through volatilization from inactive monofill (yr^{-1}).

The fraction of loss attributable to each individual process is calculated as:

$$f_{li} = \frac{K_{lec}}{K_{ti}} \qquad f_{vi} = \frac{K_{vi}}{K_{ti}} \qquad f_{di} = \frac{K_{deg}}{K_{ti}}$$

where:

f_{li} = fraction of total contaminant loss from inactive monofill attributable to leaching (dimensionless),

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

f_{di} = fraction of total contaminant loss from inactive monofill attributable to degradation (dimensionless).

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

Methodology for Groundwater Pathway

Upon completion of the mass balance calculations described above, we use two additional steps to calculate the concentration of each contaminant in groundwater:

1)   Determine the concentration of contaminant in leachate from the bottom of the facility, and

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

With the mass balance calculations, we have an estimated total rate at which contaminant is lost from the facility, and the fraction of that loss attributable to leaching. Using methods discussed in Appendix C, we conservatively estimate the amount of time that would be required to deplete the entire mass of contaminant deposited in the monofill at the maximum predicted rate of loss for each contaminant. This approach is conservative because using higher estimates for flux will yield a higher estimate of concentrations at the well.

For monofills, the rate of maximum total contaminant loss (in kg/yr) will occur in the year immediately following the last deposit of sludge, since the total mass of contaminant 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:

$$TP = LF \ / \ [1 - e^{(-K_u LF)}]$$

where:

TP   =   length of "square wave" in which maximum total loss rate of contaminant depletes total mass of contaminant applied to site (yr).

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

$$FA_l = \frac{10^{-6} \ C \ f_{la} \ SC}{TP}$$

where:

FA$_l$ =   annual flux of contaminant leaching from the monofill (kg/ha-yr),
C   =   dry-weight concentration of contaminant in sludge (mg/kg),
SC   =   estimated mass of 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).

Since sludge is combined with soil when disposed in a monofill, the volume (and mass) of sludge in the monofill is only a fraction of the total volume of the monofill. Therefore, the dry mass of sludge contained in one hectare of completed monofill is calculated by multiplying the facility's depth by the fraction of its volume containing pure sludge and by the mass of solids in one cubic meter of sludge:

$$SC = d_{mf}\,f_{sl}\,MS\;10^4$$

where:

$$MS = \frac{f_{sol}\,\rho_{sl}\,\rho_w}{f_{sol}\,\rho_w + (1-f_{sol})\,\rho_{sl}}$$

and:

SC	=	estimated mass of sludge contained in 1 ha of completed monofill (kg/ha),
MS	=	mass of solids in 1 m^3 of pure sludge (kg/m^3),
f_{sl}	=	fraction of monofill's volume containing pure sludge (dimensionless),
ρ_{sl}	=	particle density of sludge (kg/m^3),
ρ_w	=	density of water (kg/m^3),
f_{sol}	=	fraction of solids in sludge (kg/kg), 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 contaminant in leachate:

$$C_{lec} = \frac{0.1\;FA_l}{NR}$$

0.1	=	constant to convert from (kg/ha-m) to (mg/l), and
C_{lec}	=	average concentration of contaminant in water leaching from the monofill site (mg/l).

The next step is to relate the leachate concentration to the expected concentration of contaminant 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, 1989b,c) is used to estimate flow and transport through the unsaturated zone, and the AT123D analytical model (Yeh, 1981) is used to estimate contaminant 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 contaminants is predicted based on longitudinal dispersion, an estimated retardation coefficient derived from an equilibrium partition coefficient, and a first-order rate of contaminant 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 contaminant mass into the top of the unsaturated zone beneath a facility 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 contaminant 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, 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 sludge in the monofill. Transport in the unsaturated zone is determined using the Darcy velocity (V_d) 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 solid 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 (1989b).

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

As in calculations for the unsaturated zone, degradation of organic contaminants 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 (1986h) 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 contaminant (C_{wel}) within 300 years in a receptor well at the downgradient edge of the site's property boundary.

Methodology for Vapor Pathway

Two steps provide an estimate for the concentration of volatilized contaminants in air near the site:

1) Use the mass balance calculations summarized above to determine the mass of contaminant 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 contaminant in ambient air near the site.

We first use results from mass balance calculations to estimate the fraction of total contaminant 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:

$$f_{vls} = f_{vo}f_{ac} + f_{vi}(1 - f_{ac})[1 - e^{-K_a(LS-LF)}]$$

where:

f_{vls} = fraction of contaminant mass which volatilizes over a human lifetime, and
LS = human life expectancy (yr).

Next, we multiply this fraction by the total mass of contaminant deposited in the monofill, and divide by the time of release to calculate an average flux:

$$FA_v = \frac{10^{-6}\ C\ f_{vls}\ SC}{LS}$$

where:

10^{-6} = constant to convert units from (mg/ha-yr) to (kg/ha-yr), and
FA_v = annual average flux of volatilized contaminant from the site (kg/ha-yr).

The next step is to relate releases of volatilized contaminant to the expected concentrations in ambient air. The model used to simulate transport of contaminant from a monofill site is described by U.S. EPA (1986h) 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 contaminant in ambient air at that individual's location (g/m³) to the rate at which that contaminant is emitted from the facility (g/m²-sec):

$$SRR = 2.032 \; \frac{A \; v}{(r' + x_y) \; u \; \sigma_z}$$

where:

SRR	=	source-receptor ratio (sec/m),
2.032	=	empirical constant,
A	=	area of SMA (m²),
SMA	=	sludge management area,
v	=	vertical term (dimensionless),
r'	=	distance from the SMA center to the receptor (m),
x_y	=	lateral virtual distance to the receptor (m),
u	=	wind speed (m/sec), and
σ_z	=	standard deviation of the vertical distribution of contaminant concentration in ambient 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 contaminant release height of zero, $v=1$. The lateral virtual distance is the distance from a virtual point source to the SMA, such that the angle θ subtended by the SMA width is 22.5°. This distance is calculated as:

$$x_y = \sqrt{\frac{A}{\pi}} \; \cot \frac{\theta}{2}$$

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 SMA to the receptor. Table 7-1 provides values for two parameters, a and b, for a range of distances under stable conditions. Based on values from this table, an appropriate value of σ_z is calculated as:

$$\sigma_z = a \; x^b$$

where:

$$x = \frac{10^{-3} \sqrt{A}}{2}$$

and:

10^{-3}	=	constant for converting units from (m) to (km), and
x	=	distance from center of SMA to receptor (km).

Table 7-1

Parameters Used to Calculate σ_z Under Stable Conditions[a]

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

[a] Source: Environmental Science and Engineering (1985).
σ_z calculated as $\sigma_z = ax^b$ where x is distance in km.

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

$$C_{air} = \frac{FA_v \ SRR}{316}$$

where:

C_{air} = average concentration of contaminant in ambient air at the receptor location ($\mu g/m^3$), and

316 = constant for converting units from ($\mu g/m^2$-sec) to (kg/ha-yr).

7.1.2 Methodology for Surface Impoundment Prototype

Algorithms for Surface Impoundment Prototype

Our methods of estimating exposure and risk for surface impoundments are similar to those described in Section 7.1.1 for monofills. As with monofills, we begin with a mass balance of contaminant losses from the facility.

Methodology for Mass Balance

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

1) contaminant is lost to degradation within the facility (e.g., to photolysis, hydrolysis, or microbial decay),

2) contaminant is transported out of the facility by seepage through the floor of the impoundment,

3) contaminant is lost through outflow (possibly for return to the treatment works), and

+) contaminant volatilizes from the liquid surface of the impoundment.

We have adapted our model for describing these four processes 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 contaminant, diffusive exchange between the sediment layer and the water column, degradation, volatilization, the settling of particulate toxicant from the water column to the sediment, and the re-suspension of particulate from the sediment layer to the water column. For the sediment layer, they consider diffusive exchange with the water column, decay processes, particulate settling from the overlying water column, re-suspension flux from the sediment to the water column, and loss of toxicant from the sediment due to net sedimentation or burial.

We use a similar, two-layer model. The "liquid" layer begins at the surface and has the same average solids content as inflow to the facility; the "sediment" 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 contaminant 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 sediment 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 contaminant concentration; i.e., loss rates at any time are proportional to the current concentration of contaminant in the impoundment.

We rely on two additional simplifying assumptions:

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

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

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

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

$$Q_i C_i = Q_o f_{dl} C_1 + K_{degl} A d_1 C_1 + K_{voll} f_{dl} A C_1 + Q_{sep} f_{dl} A C_1 + DVC_1 \qquad (7\text{-}5)$$

where:

Q_i = rate at which sludge enters the impoundment (m³/sec),

C_i = concentration of contaminant in inflow to the impoundment (kg/m³),

Q_o = rate at which outflow leaves the impoundment (m³/sec),

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

C_1 = total concentration of contaminant (adsorbed and dissolved) in liquid layer of surface impoundment (kg/m³),

K_{degl} = rate of contaminant degradation in liquid layer (sec⁻¹),

A = surface area of impoundment (m²),

d_1 = depth of liquid layer (m),
K_{voll} = rate of contaminant 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 layer (m³/sec).

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

$$DV = \frac{d_{si}A}{TF}$$

where:

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

The active lifetime of the facility is calculated as:

$$TF = \frac{d_{si}AS_2}{Q_iS_1}$$

where:

S_1 = concentration of solids in liquid layer (kg/m³), and
S_2 = concentration of solids in sediment layer (kg/m³).

For the first term on the right of Equation 7-5, the volume of outflow from the facility (Q_o) is calculated to be consistent with assumptions about rates of inflow, seepage, and accumulation of the sediment layer:

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

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

$$S_1 = \frac{\rho_{sl}\rho_w P_1}{\rho_w P_1 + (1-P_1)\rho_{sl}} \qquad S_2 = \frac{\rho_{sl}\rho_w P_2}{\rho_w P_1 + (1-P_2)\rho_{sl}}$$

where:

P_1 = percent solids (by mass) in liquid layer (kg/kg),
P_2 = percent solids (by mass) in sediment layer (kg/kg),
ρ_w = density of water (kg/m³), and
ρ_{sl} = particle density of sludge (kg/m³).

In each layer contaminant is partitioned between adsorbed and dissolved phases. As discussed earlier, the partitioning depends on both the chemical-specific partition coefficient and the concentration of solids in the layer:

$$f_{dl} = \frac{1}{1 + KD \, S_l}$$

where:

f_{dl} = fraction of contaminant dissolved in the liquid layer, and

KD = chemical-specific partition coefficient (m^3/kg).

The second term on the right side of Equation 7-5 describes degradation of the contaminant through photolysis, hydrolysis, microbial decay, and other processes. Values for K_{degl} are taken from studies of anaerobic microbial degradation, and are applied to contaminant in both dissolved and adsorbed phases.

The third term on the right side of Equation 7-5 describes contaminant 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:

$$\frac{1}{K_{voll}} = \frac{1}{K_l} + \frac{RT}{HK_g} \qquad (7\text{-}6)$$

where:

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

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

R = ideal gas constant (8.21×10^{-5} atm-m^3/K-mol),

T = temperature (K), and

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

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, 1987f; U.S. EPA, 1989f). This methodology follows an approach described in U.S. EPA (1987f, 1989f) 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:

$$de = 2\sqrt{A/\pi}$$

$$FD = \frac{de}{d_1}$$

where:

de = effective diameter (or fetch) of site (m), and
FD = ratio of fetch to depth (dimensionless).

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

$$K_l = 2.611 \times 10^{-7} u_{10}^2 \ [D_{cw}/D_{eth}]^{2/3} \qquad (7\text{-}7)$$

where:

u_{10} = average wind speed 10 m above surface (m/sec),
D_{cw} = diffusivity of contaminant in water (cm²/sec), and
D_{eth} = diffusivity of diethyl ether in water, 8.5×10^{-6} cm²/sec.

Calculation of the mass transfer coefficient for the gas phase is based on Hwang (1982). For all values of FD and U_{10}, K_g (m/sec) is calculated from

$$K_g = 1.8 \times 10^{-3} U_{10}^{0.78} \ Sc_G^{-0.67} \ de^{-0.11}$$

where Sc_G equals the Schmidt number on the gas side, defined as

$$Sc_G = \frac{\mu_a}{\rho_a D_{ca}}$$

and where:

μ_a = viscosity of air (g/cm-s),
ρ_a = density of air (g/cm³) and
D_{ca} = diffusivity of constituent in air (cm²/sec).

Equations 7-6 through 7-8 are sufficient to estimate K_{voll}, the overall mass transfer coefficient for the dissolved fraction of the contaminant.

The fourth term on the right side of Equation 7-5 describes losses of dissolved contaminant from the liquid layer as a result of the seepage through the sediment layer and the floor of the impoundment. The rate of seepage (Q_{sep}) is based on measured values from sludge lagoons. Only dissolved contaminant is included in this term; adsorbed contaminant is included in the fifth term of the equation, which describes loss of contaminant from the liquid layer as a result of the diminishing volume of that layer.

All terms on the right side of Equation 7-5 are proportional to the concentration of contaminant in the liquid layer. A coefficient for the total rate at which contaminant mass is lost from the liquid layer (K_{totl}, in m^3/sec) can be defined as:

$$K_{totl} = Q_o f_{dl} + K_{deg1} d_1 A + K_{voll} f_{dl} A + Q_{sep} f_{dl} A + DV$$

so that:

$$Q_i C_i = K_{totl} C_1$$

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

$$f_{outl} = \frac{Q_o f_{dl}}{K_{totl}} \qquad f_{degl} = \frac{K_{deg1} d_i A}{K_{totl}} \qquad f_{voll} = \frac{K_{voll} f_{dl} A}{K_{totl}}$$

$$f_{sepl} = \frac{Q_{sep} f_{dl} A}{K_{totl}} \qquad f_{dell} = \frac{DV}{K_{totl}}$$

where:

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

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

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

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

f_{dell} = fraction of total contaminant lost from the liquid layer as a result of the diminishing volume of the liquid layer.

Sediment Layer. Contaminant mass accumulates in the sediment layer as the depth of this layer increases and eventually reaches the surface of the impoundment. If the only source of contaminant mass for the sediment layer is the loss estimated for the liquid layer:

$$Q_{sep} f_{dl} A C_1 + DV C_1 = K_{deg2} d_2 A C_2 + Q_{sep} f_{d2} A C_2 + DV C_2$$

where:

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

d_2 = depth of sediment layer (m),

K_{deg2} = rate of contaminant degradation in sediment layer (sec^{-1}), and

C_2 = total concentration of contaminant in sediment layer (kg/m^3).

A coefficient for the total loss or storage of contaminant in the sediment layer (K_{tot2}, in m³/sec) can be defined as:

$$K_{tot2} = K_{deg2}d_2A + Q_{sep}f_{d2}A + DV$$

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

$$f_{deg2} = \frac{K_{deg2}d_2A}{K_{tot2}} \qquad f_{sep2} = \frac{Q_{sep}f_{d2}A}{K_{tot2}} \qquad f_{del2} = \frac{DV}{K_{tot2}}$$

where:

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

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

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

If concentrations of contaminant in the liquid and sediment layers can be approximated as steady-state for the duration of the impoundment's active lifetime, and if the partitioning of contaminant 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 contaminant lost during each year of the facility's active phase (f_{act}) can be calculated as:

$$f_{act} = f_{vol1} + f_{deg1} + f_{out1} + (f_{sep1} + f_{del1})(f_{deg2} + f_{sep2})$$

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

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

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

$$f_{deg} = \frac{f_{deg1} + (f_{sep1} + f_{del1})f_{deg2}}{f_{act}}$$

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

where:

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

f_{vol} = fraction of total contaminant lost from the impoundment through volatilization (dimensionless),

f_{deg} = fraction of total contaminant lost from the impoundment through degradation, and

f_{out} = fraction of total contaminant lost from the impoundment through outflow.

These results are used to calculate concentrations of contaminant in groundwater and air near the site.

Methodology for the Groundwater Pathway

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

$$TP = \frac{3.2 \times 10^{-8} \ TF}{f_{act}}$$

where:

3.2×10^{-8} = constant to convert units from (sec) to (yr).

TP = duration of "square" wave for approximating the loading of contaminant into the top of the unsaturated soil zone (yr), and

TF = duration of facility's active lifetime (sec).

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

$$FA_l = \frac{0.01 \; f_{sep} \; S_2 \; d_{si} \; C}{TP}$$

where:

0.01 = constant to convert units from (mg/m²-yr) to (kg/ha-yr),

FA_l = average flux of contaminant seeping through the floor of the surface impoundment (kg/ha-yr), and

C = concentration of contaminant in sludge (mg/kg).

Next we use the average flux to estimate the average concentration of contaminant in seepage:

$$C_{sep} = \frac{0.1 \; FA_l}{3.2 \times 10^{-8} \; Q_{sep}}$$

where:

0.1 = constant for converting units from (kg/ha-m) to (mg/l), and

C_{sep} = average concentration of contaminant in water seeping through the bottom of the impoundment (mg/l).

As discussed in Section 7.1.1 for the landfill prototype, two mathematical models are combined for this purpose. The VADOFT component of the RUSTIC model (U.S. EPA, 1989b,c) estimates flow and transport through the unsaturated zone, and the AT123D model (Yeh, 1981) estimates contaminant 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 lagoon 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 sludge contaminants 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 disposal site and the downgradient receptor well. This change in the gradient will increase the expected rate of horizontal transport of the contaminant through the saturated zone.

To accommodate these two effects in the model calculations, we modify an approach used in the RUSTIC model. The first component (VADOFT) of our 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

contaminants 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 contaminant mass entering the top of the saturated zone, and does not consider the extent of the contaminant'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 contaminant 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 contaminant's original dilution could result in non-trivial overestimation of the source concentration, leading to an overestimation of contaminant concentrations at the receptor well. Furthermore, neglect of mounding effects could lead to incorrect assumptions about the velocity of groundwater flow near the site.

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

$$D_f = F_a \ / \ (A Q_{sep} + F_a) \tag{7-9}$$

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 can also 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 slowed up-gradient of the lagoon, and accelerated downgradient of the site. This change in the velocity field might result in reduced time of travel for contaminants moving to receptor wells downgradient of the impoundment site, which could in turn lead to reductions in contaminant 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 contaminant 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 lagoon as a circular source, so that the rate at which seepage passes outward through a cylinder beneath the perimeter of the lagoon's floor is:

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

where:

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

d_a =   depth of aquifer (m).

In addition to increasing the expected velocity of contaminant transport through the aquifer, this superimposed velocity would also have the effect of increasing AT123D's estimate of contaminant dilution within the aquifer. This additional dilution effect must be subtracted back out of the model calculations, since the true dilution is explicitly included in the factor introduced by Equation 7-9. The model performs this calculation automatically, based on the following equation for the anti-dilution factor:

$$D_{af} = (v_v + v_h) / v_h$$

where:

D_{af} =   anti-dilution factor,

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

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

It should be noted that the above methodology is conservative, since 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 contaminant in groundwater at a receptor well can be predicted as a function of the liquid concentration of contaminant near the floor of the impoundment, the rate of seepage from the facility, and hydrogeological characteristics of the area. It should be noted that all of the calculations described above are linear with respect to contaminant concentrations in liquid seeping from the lagoon.

This result is combined with chemical-specific properties and assumptions about the physical characteristics of the unsaturated and saturated zones to provide inputs for the linked VADOFT and AT123D models. Model results include an estimate of the maximum concentration of contaminant (C_{wel}) expected in a well at the downgradient edge of the site's property boundary within the 300-year period simulated.

Methodology for Vapor Pathway

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

$$f_{vls} = \frac{3.2 \times 10^7 \; LS \; f_{act} \; f_{voll}}{TF}$$

where:

f_{vls} =   fraction of total contaminant volatilizing during human lifetime,
LS =   lifetime expectancy (yr), and
3.2×10^7 =   constant to convert units from (yr) to (sec).

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

$$FA_v = \frac{C \; f_{vls} \; 100 \; S_2 \; d_{sl}}{LS}$$

where:

FA_v =   annual average flux of contaminant volatilizing from the surface impoundment (kg/ha-yr), and
100 =   constant for converting (kg/ha) to (mg/m^2).

The next step is to relate releases of volatilized contaminant from the site to the expected concentration in ambient air. As before, we use the simplified version of ISCLT described above in Section 7.1.1 to calculate a "source receptor ratio" or SRR. Multiplying the SRR by the average volatilized flux and adjusting units yields a conservative estimate of the expected average concentration of contaminant in ambient air near the site:

$$C_{air} = FA_v \; SRR \; 3.2 \times 10^{-3}$$

where:

C_{air} =   average concentration of contaminant in air at the receptor site (μg-m^3), and
3.2×10^{-3} =   constant for converting units from (kg/ha-yr) to (μg/m^2-sec).

7.1.3 Estimating Human Exposure and Risks

To estimate human exposure, we use methods discussed in Sections 7.1.1 and 7.1.2 to estimate the concentrations of each contaminant in groundwater and air near each surface disposal facility. As with land application, the NSSS provides data for the quality and quantity

of sludge surface disposed, but lacks information describing characteristics of individual disposal sites. For simplicity and for lack of site-specific data, we have chosen "reasonable worst case" scenarios to represent the monofill and surface impoundment prototypes. Only two types of parameter values are assumed to vary from site to site: concentrations of contaminants in the sludge and the density of human populations surrounding the site.

Estimated concentrations in environmental media are converted to estimates of human exposure based on assumptions about the rate at which the average individual consumes drinking water and inhales air. We calculate potential exposure through ingestion of contaminated groundwater as:

$$EXP_j = \frac{C_{wel} I_w}{BW}$$

where:

EXP_j	=	human exposure to pollutant j (mg/kg-day),
BW	=	average body weight (kg),
I_w	=	quantity of water ingested daily (l/d), and
C_{wel}	=	concentration of contaminant in well water (mg/l).

For air, we calculate human exposure as:

$$EXP_j = \frac{10^{-3} C_{air} I_a}{BW}$$

where:

10^{-3}	=	constant to convert units from (μg) to (mg),
C_{air}	=	average concentration of contaminant j in air at the receptor site (μg/m^3), and
I_a	=	inhalation volume (m^3/day).

To complete our calculations of risk, we combine estimates of individual exposure through groundwater and ambient air with estimates for the sizes of exposed populations. Finally, we scale by the estimated number of surface disposal sites in the U.S. to calculate our estimates of total risk.

7.2   DATA SOURCES AND MODEL INPUTS

As described in Section 7.0, we base our estimates of aggregate risks on separate prototypes for surface disposal: a monofill and a surface impoundment. With data from the analytic survey of the NSSS, we model 26 sites to represent an estimated 1,446 surface disposal facilities. Of these, 12 are modeled as monofills (including 3 dedicated land-application sites which were re-classified as surface disposal) and 14 are modeled as surface impoundments. Because reliable data could not be obtained for key site-specific parameters at many of the sites,

we have based our modeling on several conservative assumptions and choices for parameter values such that we do not expect any actual sites to cause greater exposure and risk through the groundwater or air pathways.

To obtain site-specific estimates of exposure and risk, we combine generic parameter values for soil, hydrogeological, meteorological, and other site conditions, and modeling assumptions for each prototype with two site-specific characteristics: site-specific concentrations of pollutants in sludge (from the analytic survey), and site-specific estimates for the density of human populations near each facility.

7.2.1 Site and Sludge Parameters

Area of Surface Disposal Site

Values for input parameters used to characterize the monofill and surface impoundment prototypes are listed in Table 7-2 and Table 7-3. The area for the monofill prototype is set to 1 hectare (10,000 m²), based on a design scenario described in U.S. EPA (1986h). The model area of a surface impoundment is set to 20,236 m². This value represents the 98th percentile of areas for sludge lagoons, as reported in the RCRA Subtitle D Survey (U.S. EPA, 1986f).

Depth of Disposal Facility

The depth of a monofill determines the total quantity of sludge contained in the site. The assumed depth of 3.46 m is based on a design scenario described in U.S. EPA (1986h). The depth of the surface impoundment is assumed to be 4 m. This value represents the average of data collected from an informal survey of municipal sewage sludge practices by Abt Associates Inc. (1989). In this survey, Abt Associates contacted sludge coordinators in the nine states with the largest number of surface impoundments in the 1988 NEEDS survey.

Distance to Well

For both the monofill and surface impoundment prototypes, we conservatively assume that one receptor well is located at the facility's property boundary, directly down-gradient of the site (150 m from edge of the site). Concentrations estimated for this location are conservatively applied to all individuals residing within 1000 m of the site, and within a 90 degree angle centered on the downgradient direction of groundwater flow. A second receptor well is assumed to be located at 1000 m downgradient of the property boundary, and estimated concentrations at this location are applied to all individuals residing within the same angle from 1000-3000 m.

Thickness of Cover for Monofill

The thickness of the active cover for a monofill is assumed to be 0.3 m, and the thickness of the final cover 1 m. These values represent typical thicknesses for cover applied to an area

Table 7-2

Site and Sludge Parameters
Monofill Prototype for Surface Disposal

Parameter	Value
Area of monofill (m²)	10,000[a]
Depth of monofill (m)	3.46[a]
Distance to Well (m)	150[b]
Thickness of Daily Cover (m)	0.3[b]
Thickness of Permanent Cover (m)	1[b]
Time Each Cell Uncovered (hr)	12[a]
Time Average Cell Contains Sludge (hr)	87,660[c]
Sludge as Fraction of Total Volume (m³/m³)	0.63[b]
Active Site Life (yr)	20[b]
Wind Velocity (m/sec)	4.5[d]
Average Air Temperature (K)	288[a]
Solids Content of Sludge (kg/kg)	0.20[b]

[a] U.S. EPA (1986h).
[b] U.S. EPA (1978).
[c] Half of assumed site life.
[d] U.S. EPA (1990c).

Table 7-3

Site and Sludge Parameters
Surface Impoundment Prototype for Surface Disposal

Parameter	Value
Area of Surface Impoundment (m^2)	20,236[a]
Depth of Surface Impoundment (m)	4[b]
Distance to Well (m)	150[a]
Rate of Inflow (m^3/sec)	0.0022[a]
Wind Velocity (m/sec)	4.5[c]
Average Air Temperature (K)	288
Solids Content of Inflow (kg/kg)	0.03[d]
Solids Content of "Liquid" Layer (kg/kg)	0.03[d]
Solids Content of "Sediment" Layer (kg/kg)	0.175[c,e]
Particle Density of Sludge (kg/m^3)	1200[e]

[a] U.S. EPA (1986f).
[b] Abt Associates Inc. (1989).
[c] U.S. EPA (1990c).
[d] U.S. EPA (1978).
[e] Chaney (1992).

fill trench (U.S. EPA, 1978).

Number of Days Average Cell of Monofill is Uncovered

Total emissions from a monofill cell depend on the length of time the cell is uncovered or covered with a soil layer. For these calculations, we assume a cell is open for 4 hours a day for the 3 consecutive days it receives sludge (U.S. EPA, 1986h). For the remaining 20 hours of each of those 3 days, and for the remainder of the active lifetime of the facility, the cell is covered with a temporary layer of soil. Assuming a constant rate of disposal, during the 20 year active lifetime of the monofill, half the cells will contain sludge for more than 10 years and half for less than 10 years. A typical cell is therefore uncovered for (3x4)/24 or 0.5 days and covered for (20x365.25)/2 - 0.5 or about 3652 days. After the facility is filled to capacity, it is covered with a thicker, permanent layer of soil for the remainder of the period simulated.

Inflow Rate for Surface Impoundment

We assume the prototype facility for surface impoundments receives continuous inflow of sludge throughout its active lifetime. The duration of that active lifetime depends on the rate at which sludge enters the facility, the solids content of that sludge, and the volume of the facility. According to the RCRA Subtitle D survey (U.S. EPA, 1986f), almost 96 percent of the sludge lagoons surveyed received less than 50,000 gallons per day of wastewater flow. This number converts to 0.0022 m^3/sec, and is used as the model inflow rate. It is also consistent with the mean inflow rate for surface impoundments, based on data from the analytic survey of the NSSS.

Ratio of Sludge to Total Volume

A typical trench monofill contains parallel trenches filled with sludge and separated by soil, so that the entire monofill site contains both sludge and soil. We do not model these layers of sludge and soil separately, but instead model an idealized, homogeneous mixture of sludge and soil. We calculate the total quantity of sludge likely to be contained within a monofill of specified dimensions from the fraction of the monofill's contents consisting of sludge. U.S. EPA (1978) describes several design scenarios for different types of trench monofills, and reports the approximate quantity of sludge that can be received per acre for each scenario. The wide trench monofill scenario (which receives the most sludge per unit area) is reported to receive about 7,744 m^3 of sludge per hectare to a depth of about 1.22 m. Dividing this volume of sludge by the volume of facility per hectare (10,000 m^2 x 1.22 m or 12,200 m^3) yields the fraction of the monofill's volume which contains pure sludge (0.63).

For the surface impoundment prototype, we assume the entire facility is filled with sludge, so that the volume of sludge contained by the facility is equal to the volume of the impoundment.

Site Life

The site life for monofills is the length of time the monofill is receiving sludge, up until the time that permanent cover is applied. The assumed site life of 20 years is based on the design scenario described in U.S. EPA (1978).

For surface impoundments, the site life is the time required to fill the impoundment with sludge solids. It is calculated from the inflow rate, the percent solids in the sludge, and the dimensions of the facility. Using these values we calculate an expected lifetime of about seven years for the modeled surface impoundment.

Wind Velocity

Wind velocity affects the transport of volatilized contaminant. As explained in Section 7.1, the equations used to model emissions from monofills require an estimate of average wind velocity at the ground surface. For surface impoundments, wind velocity is specified at 10 m above the surface. Although wind velocities at 10 m are likely to be somewhat higher than those at ground level, a single value of 4.5 m/sec (10 mph) has been assumed in this analysis for both ground level and 10 m heights, and for both prototypes. This value represents a typical yearly average wind speed in the United States (U.S. EPA, 1990c).

Air Temperature

The model air temperature of 15°C represents a typical annual average value for the U.S. This value is consistent with the range of temperatures presented in U.S. EPA (1986f).

Solids Content of Sludge

The solids content of sludge depends on the level of treatment (i.e., primary, secondary, [ad]vanced) and on the types of treatment processes used (stabilization, de-watering, etc.). Only [de]-watered sludges with solids contents greater than or equal to 15% are suitable for disposal [in] sludge-only monofills. Monofilled sludge is often mixed with a bulking agent (e.g., soil) to [inc]rease solids content. The solids content of sludge (20%) assumed for the monofill prototype [is b]ased on the value reported for a typical wide trench monofill (U.S. EPA, 1978).

The solids content of sludge in surface impoundments is assumed to differ between two [ideali]zed layers: a "liquid" layer and a "sediment" layer. Sludge entering the impoundment is [assum]ed to contain 3% solids by mass (0.03 kg/kg). This value falls within the range [of concen]trations reported for primary and secondary treated sludge (U.S. EPA, 19[ ]). [The solids conten]t of the sediment layer is assumed to be 17.5%. [ ] [ ] of municipal sewage sludge dis[ ] [ ]ts the midpoint of [ ]

Particle Density of Sludge

A particle density can be derived from the mass and solids content of a typical wet sludge. According to Chaney (1992), a typical wet sludge in a surface impoundment has a specific gravity of about 1.03, equivalent to a density of about 1030 kg/m³. If 17.5 percent of the mass of such sludge is solids and the remainder is water (with a density of 1000 kg/m³), the particle density of pure sludge (ρ_{sl}) can be calculated as:

$$\rho_{sl} = \frac{(1030)\,(kg/m^3) - (1000)\,(kg/m^3)\,(0.825)}{(0.175)} \approx 1200\,(kg/m^3)$$

Concentrations of Pollutants in Sludge

Table 7-4 lists the concentrations of each pollutant measured for each of the 26 surface disposal facilities included in the analytic survey of the NSSS and modeled for this analysis.

7.2.2 Soil and Hydrologic Parameters

The unsaturated zone is characterized by pore space containing both air and water, whereas the pore space in the saturated zone contains water only. Because of differences in fluid flow regimes, these two zones require different equations and input parameters for tracking contaminant transport. A simplifying assumption used for this analysis is that the basic soil characteristics (including soil type, porosity, and bulk density) of the two zones are identical.

For both the monofill and surface impoundment prototypes, sludge is assumed to be placed entirely beneath the ground surface, requiring subsurface excavation. For monofills, the excavated soil is typically used for cover (U.S. EPA, 1978). Parameter values describing soil and hydrogeological characteristics for both monofills and surface impoundments are listed in Table 7-5.

Soil Type

The types of soil in the unsaturated and saturated zones affect the ability of a contaminant to move vertically to the aquifer and laterally to a nearby well. In general, the ease of contaminant transport through a soil (ignoring the adsorption properties of the soil) is largely affected by the type of clay present, the shrink/swell potential of that clay, and the grain size of the soil. The less the clay shrinks and swells and the smaller the grain size of the soil, the

Table 7-4

Concentrations of Pollutants in Surface Disposal Facilities (mg/kg)

Survey ID #	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Benzene	BEHP	TCE
MONOFILLS										
Sacramento Reg. Wastewater	0.0	12.5	188.5	786.5	314	5.6	48.7	0.0	268.11	0.00
Garden City WPCP	5.5	7.1	113	524	142	2.4	58.3	0.00	24.29	0.00
Spillville WWTP	5.7	7.85	204	647	146	2.5	62.8	0.00	26.07	0.00
Pittsfield Wastewater TP	0	2.4	15.5	237	34.8	0	0	0.00	0.51	0.00
Hibbing WWTP	17.3	5.2	37.8	2410	483	5.8	46.5	0.00	62.49	0.00
Southerly WWT Center	0	0	0	17.4	0	0	0	0.00	0.00	0.00
N. Olmstead WWT Plant	4.4	5.8	94.1	459	124	2.2	22.35	0.00	14.23	0.00
Bucklin Pt. STP	1.9	3	21.3	336	46.8	0.4	9.5	0.00	1.61	0.00
Memphis T.E. Maxson WWTP	0.65	9.6	104.65	337.5	311	2.75	15.25	0.01	216.63	0.00
Haskell ST STP	0	2.1	10.7	229	24.5	0	0	0.00	0.00	0.00
Navasota STP	6.9	28.1	420	1035	357	4.7	76.4	0.00	142.32	0.00
San Felipe STP	4.9	6.9	108	485	138	2.35	54.5	0.00	16.75	0.0

Table 7-4, continued

Concentrations of Pollutants in Surface Disposal Facilities (mg/kg)

Survey ID #	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Benzene	BEHP	TCE
SURFACE IMPOUNDMENTS										
Stickney WRP	0	0	2.2	161	0	0	0	0.00	0.00	0.00
Corinth WWTP	4.05	4.8	72.9	404	112.5	1.7	19.1	0.00	11.53	0.00
Hamilton WWTP-S.W.	5.7	8	276.5	702	156	2.8	63.3	0.00	71.70	0.00
Missouri River WWTP	0	1.3	10	223	14.3	0	0	0.00	0.00	0.00
Coldwater Creek WWTP	2.3	3.35	31.6	343	54.7	0.8	10	0.00	2.13	0.00
Holdenville South STP	6	9	318	789	229	3.5	72.15	0.00	103.92	0.00
Kiski Valley WPCA	2.9	4.7	64.5	361.5	111	1.65	16.5	0.00	8.49	0.00
Signal Mountain STP	6.7	9.85	416	860	291	4.6	76.35	0.00	105.07	0.00
Abiline STP	2.85	4.1	38.2	360	87.1	1.6	14.6	0.00	4.24	0.00
E. Cedar Creek Fresh Water Sup. Div	12.7	136.5	752.5	1110	370.5	5.5	236	0.00	168.42	0.00
Quechea WPC	14.95	249	1010	1180	425.5	5.6	236	0.00	174.05	0.00
Wabeno SD #1 Lagoons	16.5	333	1895	1190	475	5.7	262.5	0.22	202.74	0.024

Table 7-5

Soil and Hydrologic Parameters
For Monofill and Surface Impoundment Prototypes

Parameter	Value
Soil Type	Sand
Porosity of Sludge/Soil	0.4[a,b]
Porosity of Soil Cover (monofill only)	0.4[a,b]
Bulk Density for Pure Soil (kg/m^3)	1600[c]
Bulk Density of Sludge/Soil (kg/m^3)	1400[d]
Saturated Hydraulic Conductivity of Soil (m/hr)	0.61[b]
Water Retention Parameters	
θ_r	0.045[b]
α (m^{-1})	14.5[b]
β	2.68[b]
Fraction of Organic Carbon in Soil or Sludge	
Sludge	0.31[e]
Unsaturated Soil Zone	0.001[f]
Saturated Soil Zone	10^{-4}
Depth to Groundwater (m)	1
Net Recharge or Seepage	
Monofill Prototype (m/yr)	0.5[f]
Surface Impoundment Prototype (m/yr)	2.5[g]
Thickness of Aquifer (m)	5
Hydraulic Gradient	0.005[f]

[a] Todd (1980).
[b] Carsel and Parrish (1988).
[c] Calculated from porosity and particle density of 2650 (kg/m^3) from Freeze and Cherry (1979).
[d] Chaney (1992). Assumed for this analysis to be comparable to soil treated with land-applied sludge.
[e] U.S. EPA (1983b).
[f] U.S. EPA (1986f).

model simulations are based on values estimated for sand.

Porosity of Sludge/Soil

Porosity is the ratio of the void volume of a given soil or rock mass to the total volume of that mass. If the total volume is represented by V_t and the volume of the voids by V_v, the porosity can be defined as $\theta_t = V_v/V_t$. Porosity is usually reported as a decimal fraction or percentage, and ranges from 0 (no pore space) to 1 (no solids).

For this analysis, we assume a total porosity of 0.4, based on Todd (1980). This value is consistent with the average value for sand (0.43) reported in Carsel and Parrish (1988). It is used to represent total porosity within a monofill, in the cover soil applied to a monofill, and within the unsaturated and saturated soil zones beneath both monofills and surface impoundment prototypes. In the unsaturated zone, about half of total porosity on average is assumed to be water-filled, so that both water-filled and air-filled porosities are assigned values of 0.2.

Effective porosity is calculated as the difference between the average saturated water content and the approximate average residual water content, and refers to the amount of interconnected pore space available for fluid flow. For these calculations, the average residual water content in the unsaturated zone is assumed to be less than 0.05 (Carsel and Parrish, 1988), and effective porosity has been approximated with the same value used for total porosity (0.4) in mass balance and groundwater transport calculations.

Bulk Density of Soil

The bulk density of soil is defined as the mass of dry soil divided by its total (or bulk) volume. Bulk density directly influences the retardation of solutes and is related to soil structure. In general, as soils become more compact, their bulk density increases. Bulk density can be related to the particle density and porosity of a given soil as:

$$BD = \rho_{so}(1 - \theta_t)$$

where:

BD	=	bulk density of soil (kg/m³),
ρ_{so}	=	particle density of soil (kg/m³) and
θ_t	=	porosity of soil (dimensionless).

Typical mineral soils have particle densities of about 2650 kg/m³ (Freeze and Cherry, 1979). This value and a soil porosity of 0.4 suggest a bulk density of about 1600 kg/m³ for pure soil, somewhat higher than the 1300-1500 kg/m³ range typically encountered for soil mixed with sludge (Chaney, 1992).

Saturated Hydraulic Conductivity of Soil

Saturated hydraulic conductivity refers to the ability of soil to transmit water, which is governed by the amount and interconnection of void spaces in the saturated zone. These voids may occur as a consequence' of inter-granular porosity, fracturing, bedding planes, or macropores. In general, high hydraulic conductivities are associated with high rates of contaminant transport. We use a value for saturated hydraulic conductivity (0.61 m/hr) based on the 95th percentile of a probability distribution for hydraulic conductivity in sand derived by Carsel and Parrish (1988). This value thus represents a conservative or "reasonable worst case" value.

Unsaturated Hydraulic Conductivity of Soil

In the unsaturated zone, the hydraulic conductivity, which is based on the effective permeability of soil and fluid properties, is a function of the moisture content, which is in turn a function of the pressure head. These relationships are central to the simulation of water flow through the unsaturated zone. As input, the VADOFT model accepts sets of data points describing effective permeability-saturation curves and the saturation-pressure head curves. Alternatively, it accepts van Genuchten water retention parameters defining the curves (U.S. EPA, 1989b; Carsel and Parrish, 1988); this latter option is used for this analysis.

Based on soils data from the Soil Conservation Survey (SCS), Carsel and Parrish derived distributions for the three parameters required (Θ_r, α, and β) according to twelve SCS textural classifications (Carsel and Parrish, 1988). Values used for our calculations (0.045, 14.5 m^{-1}, and 2.68 for Θ_r, α, and β, respectively) correspond to values reported for sand.

Fraction of Organic Carbon in Soil or Sludge

The model combines the fraction of organic carbon in the soil with each contaminant's organic carbon partition coefficient to determine the partitioning of contaminant between soil and water. In general, a lower fraction of organic carbon implies greater mobility for organic contaminants. The organic carbon content for sludge varies among sludge types, with mean values for various types showing a relatively narrow range of 27.6-32.6 percent (U.S. EPA, 1978). For sludge within the surface disposal facility, we use the mean value for all sludges combined of 31 percent (U.S. EPA, 1978). We have selected a value of 10^{-3} for the fraction of organic carbon in the unsaturated zone because it is a typical value for sand, and it falls at the lower end of the range (0.001 - 0.01) reported for soil beneath hazardous waste disposal facilities (U.S. EPA, 1986f). The fraction of organic carbon in the saturated zone is expected to be lower than that of the unsaturated zone, and has been assigned a value of 10^{-4}, or one-tenth the fraction assumed for the unsaturated zone.

Depth to Groundwater

The depth to groundwater is defined as the distance from the lowest point of the surface disposal facility to the water table. The water table is itself defined as the subsurface boundary between the unsaturated zone (where the pore spaces contain both water and air) and the

saturated zone (where the pore spaces contain water only). It may be present in any type of medium and may be either permanent or seasonal. The depth to groundwater determines the distance a contaminant must travel before reaching the aquifer, and affects the attenuation of contaminant concentration during vertical transport. As this depth increases, attenuation also tends to increase, thus reducing potential pollution of the groundwater.

For both monofills and surface impoundments, we conservatively assume that the depth to groundwater is 1 m. According to data in the GRNDWAT data base (U.S. EPA, 1988a), this value is less than any of the depths most typical of counties containing surface disposal facilities in the analytic survey of the NSSS. For this reason, 1 m is believed to be a reasonable worst-case value likely to over-estimate actual exposure and risk.

Net Recharge or Seepage

The primary source of most groundwater is precipitation, which passes through the ground surface and percolates to the water table. Net recharge is the volume of water reaching the water table per unit of land, and determines the quantity of water available for transporting contaminants vertically to the water table and laterally within the aquifer. The greater the recharge rate, the greater the potential for contaminant transport, up to the point at which the amount of recharge is large enough to dilute the contaminant. Beyond that point, the effect of the increased rate of transport is offset by dilution (U.S. EPA, 1985b).

For monofills, the selected recharge rate (0.5 m/yr) represents the average of a range of values presented in (U.S. EPA, 1986f). For surface impoundments, the relatively high water content of sludge can provide an additional source of recharge if water from the sludge seeps through the floor of the impoundment. In impoundments receiving continuous or periodic deposits of sludge, this source may not be depleted during the active lifetime of the facility. Table 7-6 lists seepage rates from municipal lagoons (in inches per day and $1/m^2$-hr) (U.S. EPA, 1987e). The value selected for this analysis (2.5 m/yr) represents the average seepage rate for lagoons over sandy soil.

Thickness of Aquifer

Saturated zones are considered to be aquifers if they can transmit significant volumes of water. Only aquifers are considered when selecting input parameters for these calculations. For estimating aggregate risks, we assume the thickness of the aquifer is 5 m.

Hydraulic Gradient

The hydraulic gradient is a function of the local geology, groundwater recharge volumes and locations, and the influence of withdrawals (e.g., well fields). It is also very likely to be indirectly related to properties of porous media. Rarely are steep gradients associated with very high conductivities. No functional relationship exists, however, to express this relationship.

The hydraulic gradient value selected for our calculations is 0.005 m/m or 0.5 percent, and is based on an average value for groundwaters surveyed for the Hazardous Waste

Table 7-6

Summary of Measured Seepage Rate from Municipal Lagoon Systems[a]

Water Depth (ft)	Lagoon Type	Underlying Soil	Seepage Rate (in/day)	Seepage Rate (l/m^2-hour)
5	Facultative	Heavy silty clay	0.3	0.32
6	Facultative	Light silty clay	0.29	0.31
5	Facultative	Alkaline silt	0.65	0.69
6	Facultative	Fine sand	1.2	1.3
6	Facultative	Gravel and silt	1.3	1.4
-	- -	Sandy soil	0.35	0.37
-	- -	Sand and gravel	0.61[b]	0.65
-	- -	Sandy soil	0.34	0.36
-	- -	Clay loam and shale	0.3	0.32
-	Maturation	Mica and schist	0.06 - 0.23	0.06 - 0.24
5	Facultative	Silt, sand, marl	0.18	0.19
5	Facultative[c]	Sand, silt, marl	1.07	1.13
5	Evaporation[d]	Sand, silt, marl	0.04 - 0.11	0.04 - 0.12
-	Facultative	Sandy soil	0.12	0.13

[a] Source: U.S. EPA (1987e).
[b] Includes net precipitation/evaporation.
[c] Used intermittently.
[d] Sealed with bentonite and soda ash.

Management System Land Disposal Restrictions Regulation (U.S. EPA, 1986f).

7.2.3 Chemical-Specific Parameters

Distribution Coefficients

Contaminant transport in soil systems is influenced by interactions between the contaminant and soil. The affinity of contaminants for soil particles may result from ion exchange in clay particles, electrostatic forces between contaminants and charged particles, and interactions with organic carbon. When all interaction and exchange sites in a soil are filled, soluble contaminants will move through the soil at the same velocity as the bulk leachate. The affinity between a soil and a contaminant is characterized by the distribution coefficient (*KD*). Representative *KD* values (in l/kg or m^3/kg) are defined as the equilibrium ratio of the contaminant concentration in soil (mg/kg) to that in associated water (mg/l or mg/m^3). Values used for this analysis are listed in Table 7-7 and discussed below. Note that the organic contaminants nitrosodimethylamine and toxaphene are not being considered in this analysis because they were never detected in the analytic survey of the NSSS.

For hydrophobic organic contaminants, *KD* is calculated from a contaminant's partition coefficient between organic carbon and water:

$$KD = KOC \, f_{oc}$$

where:

KD	=	equilibrium partition coefficient for contaminant (m^3/kg),
KOC	=	organic carbon partition coefficient (m^3/kg), and
f_{oc}	=	organic carbon as a fraction of soil mass (dimensionless).

As discussed previously, f_{oc} values of 0.31, 0.001 and 0.0001 are assumed for the sludge layer, unsaturated zone, and saturated zone, respectively.

The organic carbon partition coefficient for a contaminant can be estimated from its octanol-water partition coefficient, which can be measured in laboratory experiments. Values of *KOC* used are shown in Table 7-8, and are calculated from the following regression equation by Hassett *et al.* (1983):

$$\log(KOC) = 0.0884 + 0.909 \log(KOW)$$

where:

KOW	=	octanol-water partition coefficient for contaminant.

With the exception of PCBs, the *KOW* values used for this analysis have been obtained from the CHEMEST procedures in the Graphical Exposure Modelling Systems (GEMS and PCGEMS), U.S. EPA (1988a, 1989d).

Table 7-7

Distribution Coefficients for Organic and Inorganic Contaminants

	Within Surface Disposal Facility (l/kg)	Unsaturated Zone (l/kg)	Saturated Zone (l/kg)
Arsenic	20	20	20
Cadmium	431	431	431
Chromium	59	59	59
Copper	98	98	98
Lead	621	621	621
Mercury	330	330	330
Nickel	63	63	63
Benzene	32.8	0.106	0.0106
Benzo(a)pyrene	139,000	448	44.8
Bis(2-ethylhexyl)phthalate	16,800	54.1	5.41
Chlordane	41,200	133	13.3
DDT	239,000	772	77.2
Lindane	726	2.34	0.234
Polychlorinated biphenyls	467,000	1,510	151
Trichloroethylene	60.1	0.194	0.0194

Notes: The distribution coefficient for organic pollutants (KD) is the product of the organic carbon partition coefficient (KOC) and the fraction of organic carbon in the medium (f_{oc}). Assumes f_{oc} of 31 percent within the surface disposal facility and 0.1 percent and 0.01 percent in the unsaturated and saturated soil zones, respectively. Distribution coefficients for metals are geometric means of values reported for a "sandy loam" soil in Gerritse et al. (1982).

Table 7-8

Octanol-Water and Organic Carbon Partition Coefficients for Organic Contaminants

	Log of Octanol-Water Partition Coefficient[a]	Organic Carbon Partition Coefficient[b]
Benzene	2.13	106
Benzo(a)pyrene	6.12	448,000
Bis(2-ethylhexyl)phthalate	5.11	54,100
Chlordane	5.54	133,000
DDT	6.38	772,000
Lindane	3.61	2,340
Polychlorinated biphenyls[c]	6.70	1,510,000
Trichloroethylene	2.42	194

[a] All values except for PCBs taken from the CHEMEST procedure of the Graphical Exposure Modeling System (GEMS), U.S. EPA (1989d).

[b] KOC for organic contaminants derived from KOW with Equation 6 from Chapter 15 of Hasset et al. (1983): $\log(KOC) = 0.0884 + 0.909\log(KOW)$.

[c] Based on Aroclor 1254, the most common PCB mixture in sewage sludge. Derived from O'Connor (1992) and representative values from Anderson and Parker (1990).

Polychlorinated biphenyls (PCBs) are a class of chemicals containing 209 possible congeners. The most common PCB mixture is Aroclor 1254, which is dominated by penta-congeners, with about equal amounts of tetra- and hexa-congeners. In a well-aged soil contaminated with PCBs, however, Aroclor 1260, which contains more penta- and hexa-congeners than tetra-congeners, is more representative of the PCBs found (O'Connor, 1992) because the less chlorinated congeners degrade more rapidly. To determine a representative organic carbon partition coefficient for PCBs, we have calculated an average from log KOW coefficients listed in Table 7-9 (from Anderson and Parker, 1990). The log KOW for the penta-congener has been estimated to be approximately 6.5 by noting that the log KOW values are approximately linearly related to the number of chlorines in the congener. Averaging that value with the hexa-congener value gives 6.7 for the log KOW. As with other organic contaminants, the regression equation from Hassett et al., (1983) is used to convert this KOW value to an estimate of KOC.

For metals, values for KD are taken from Gerritse et al. (1982), and represent results of laboratory column tests with a sludge-amended sandy loam topsoil. The values used for each contaminant were included in Table 7-6, and are based on the geometric mean of the ranges provided by those authors.

Degradation

Pollutant concentrations in the subsurface regime may be decreased by various degradation processes, including abiotic hydrolysis and aerobic or anaerobic microbial degradation. Although rates of hydrolysis are dependent only on Ph and temperature (and can be estimated with reasonable accuracy), estimates of rates for microbial degradation are fraught with uncertainty. This uncertainty is due to many confounding influences in the field, such as substrate availability (fraction of organic carbon present), temperature, the microbial consortium, and microbial acclimation to a given pollutant. Nevertheless, the range of microbial degradation rates obtained in the laboratory by measuring the rate of disappearance of a pollutant in various soil and water grab samples, soil column studies, etc., provides a rough estimate of the rate that microbial activity is likely to degrade a particular pollutant in the field.

As shown in Table 7-10, this work utilizes several sources for representative microbial degradation rates. Where a range of values is reported by these sources, values from the lower end of the range have been selected to derive estimates most protective of public health. Studies of biodegradation in soil have been favored over studies of biodegradation in aquatic environments. If estimates of only aerobic biodegradation rates are available for a given contaminant, a half-life for anaerobic biodegradation has been conservatively estimated to be four times longer (Howard et al., 1991). However, if available data fail to show any indication that a pollutant degraded in a particular regime, a value of 0 has been assumed for the microbial degradation rate.

For the sludge layer of both surface disposal prototypes, estimated rates of degradation are based on studies of microbial degradation in anaerobic conditions. For the unsaturated soil zone, aerobic microbial degradation and hydrolysis are assumed to be the two dominant degradation processes. Lindane and trichloroethylene are the only two compounds that are

Table 7-9

Octanol-Water Partition Coefficients
for PCBs[a]

Congener	Number of Chlorines	Log KOW
2,4'	2	5.1
2,2',5,5'	4	6.1
All Penta	5	6.5[b]
2,2',4,4',5,5'	6	6.9
Average[c]	5.5	6.7

[a] Source: Anderson and Parker (1990).

[b] Estimated based on apparent linear relationship between number of chlorines on congener and log KOW.

[c] log KOW values for penta- and hexa-congeners averaged for representative log KOW.

Table 7-10

Degradation Rates

	Aerobic Degradation Rate (yr⁻¹)[a]	Anaerobic Degradation Rate (yr⁻¹)[b]	Unsaturated Zone Degradation Rate (yr⁻¹)[c]	Saturated Zone Degradation Rate (yr⁻¹)[d]
Benzene	16[e]	0[f]	1.6	0.8
Benzo(a)pyrene	0.48[g]	0.12[h]	0.048	0.084
Bis(2-ethylhexyl) phthalate	.11[i]	0[f,j]	1.1	0.55
Chlordane	0[k]	36[i]	0	18
DDT	0.04[l]	2.5[k]	0.004	1.3
Lindane	1.2[m]	8.3[n]	1.2	4.8
PCBs	0.063[o]	0.00063[p]	0.0063	0.0035
Trichloroethylene	0.78[q]	3.3[r]	0.78	2.0

[a] Based on microbial degradation rates, except for lindane and trichloroethylene, where hydrolysis rates are used.
[b] Based on microbial degradation rates.
[c] Estimated as 10% of aerobic biodegradation rates. Hydrolysis rates used for lindane and trichloroethylene.
[d] Estimated as the arithmetic average of the unsaturated zone degradation rates and the anaerobic degradation rate.
[e] Vaishnav and Babeu (1987).
[f] Horowitz et al. (1982).
[g] Coover and Sims (1987).
[h] Anaerobic rate assumed to equal 25% of aerobic rate.
[i] Howard et al. (1991).
[j] Shelton et al. (1984).
[k] Castro and Yoshida (1971).
[l] Stewart and Chisholm (1971).
[m] Ellington et al. (1988).
[n] Zhang et al. (1982).
[o] Fries (1982).
[p] Anaerobic rate assumed to equal 1% of aerobic rate.
[q] Dilling et al. (1975).
[r] Bouwer and McCarthy (1983).

assumed to undergo significant hydrolysis: since hydrolysis rates are far more accurately quantifiable than microbial degradation rates, hydrolysis rates are used for these two chemicals. For the other six organic contaminants, 10 percent of the aerobic biodegradation decay rate is assumed to be appropriate for the unsaturated zone. This decision is based on the observation that f_{oc} tends to decrease with depth in the soil, thereby reducing the amount of suitable substrate for microbial populations which might degrade these chemicals (O'Connor, 1992).

In the saturated zone, all three degradation processes can occur because some groundwater is anaerobic and some aerobic. To capture this mix of processes, we have calculated an arithmetic mean from the aerobic and anaerobic biodegradation decay rates discussed above. For lindane and trichloroethylene, the only two chemicals where hydrolysis is a significant degradation process, estimated anaerobic decay rates are significantly higher than hydrolysis rates.

For PCBs, it is difficult to assign an anaerobic degradation rate. Highly chlorinated congeners may be partially degraded very slowly in reducing conditions, but then oxidative conditions must be established for further degradation to occur. Adequate information on anaerobic degradation rates cannot be obtained from the scientific literature. We have conservatively assumed that anaerobic degradation of PCBs occurs at 1 percent of the aerobic biodegradation rate.

Henry's Law Constant

Henry's Law constants are used to calculate the rate at which organic contaminants volatilize from sludge. Determining appropriate values for these constants is complicated by the wide variation in estimates provided by various sources. Table 7-11 shows values taken from four different sources, along with the value selected for this analysis. Whenever possible, values are taken from Lyman et al. (1990); otherwise values are taken from: the GEMS data base (U.S. EPA, 1988a), the PCGEMS data base (U.S. EPA, 1989d), or the Aquatic Fate Process Data for Organic Priority Pollutants (U.S. EPA, 1982a). The decision process is as follows: if a value is published in Lyman et al. (1990), it is used. If not, but if two values from other sources are similar, the mean of those two values is used. If there is no value in Lyman et al. and no two values agree, a measured value is chosen in preference to an estimated one. If only estimated, dissimilar values are available, the value most conservative for groundwater (i.e., the lowest Henry's Law constant) is chosen. This last circumstance occurs only for bis(2-ethylhexyl)phthalate.

The only exception to the decision process described above is for polychlorinated biphenyls (PCBs), which include a variety of possible congeners with different chemical characteristics. Anderson and Parker (1990) provide a compilation of non-dimensional Henry's Law constants for one penta-congener and three hexa-congeners. To derive a representative Henry's Law Constant for PCBs, the three values for hexa-congeners were averaged to a single value which was then averaged with the penta-congener value to obtain the single constant reported in Table 7-11.

Table 7-11

Henry's Law Constants

	GEMS[a,b] (atm-m³/mol)	PCGEMS[a,b] (atm-m³/mol)	Lyman et al. [d] (atm-m³/mol)	Aquatic Fate[c,b] (atm-m³/mol)	Selected Values[f] (dimensionless)	Reference
Benzene	3.5×10^{-3}	5.6×10^{-3} (M)	5.5×10^{-3}	5.5×10^{-3}	2.3×10^{-1}	(d)
Benzo(a)pyrene	1.9×10^{-10}	--	--	--	8.0×10^{-9}	(g)
Bis(2-ethylhexyl)phthalate	4.4×10^{-8}	1.5×10^{-5}	--	3.0×10^{-7}	1.9×10^{-6}	(a)
Chlordane	--	4.9×10^{-5} (M)	--	9.4×10^{-5}	3.0×10^{-3}	(c,e)
DDT	2.8×10^{-8}	--	3.8×10^{-5}	1.6×10^{-5}	1.6×10^{-3}	(d)
Lindane	4.3×10^{-6}	1.4×10^{-5}	4.8×10^{-7}	--	2.0×10^{-5}	(d)
Polychlorinated biphenyls	--	--	--	--	1.4×10^{-2}	(g)
Trichloroethylene	2.9×10^{-3}	1.0×10^{-2} (M)	1.0×10^{-2}	9.1×10^{-3}	4.2×10^{-1}	(d)

[a] CHEMEST procedure in U.S. EPA (1988a).
[b] Values estimated using $\dot{H}$ = VP/WSOL where VP is the vapor pressure and WSOL is the water solubility, unless designated as a measurement (M).
[c] CHEMEST procedure in U.S. EPA (1989b).
[d] Lyman, Reehl and Rosenblatt, Handbook of Chemical Property Estimation Methods, McGraw-Hill, 1990.
[e] U.S. EPA (1982a).
[f] Converted to non-dimensional value using $\dot{H}$ = H/(RT) where T = 288 K, R is the Universal Gas Constant and H is the dimensional Henry's Law constant.
[g] Average values from Anderson and Parker (1990) adjusted to 15°C.

For all organic contaminants except PCBs, the dimensional estimate of Henry's Law Constant reported in Table 7-11 has been converted to an equivalent non-dimensional constant based on an assumed temperature of 15°C (288K) and the following equation:

$$\dot{H} = \frac{H}{RT}$$

where:

T	=	temperature (assumed to be 288 K),
R	=	Ideal Gas Constant (m³-atm/mol-K),
H	=	dimensional Henry's law constant (m³-atm/mol), and
$\dot{H}$	=	non-dimensional Henry's Law constant.

Because Anderson and Parker (1990) report non-dimensional values for PCBs at 25°C, the average value derived from this source has been adjusted to an equivalent non-dimensional value at 15°C.

Diffusion Coefficients

Volatilization of contaminant from a surface impoundment is modeled with a mass transfer coefficient derived with a two-layer resistance model. Because contaminant must pass through both the liquid and air to be released into the atmosphere, the overall resistance equals the sum of the liquid and gas phase resistances, which are described as the inverse of mass transport coefficients for each phase. Methods for calculating these mass transfer coefficients are selected according to two types of site characteristics: (1) the ratio of the surface's effective diameter (or "fetch") to its depth and (2) the local average wind speed. Effective diameter is defined as the diameter of a circle of area equal to the facility's. The fetch:depth ratio for the model site is about 80 and is calculated using an effective diameter of 161 m (area = 20,236 m²), and a depth of 2 m (the depth of the liquid layer).

Mass transfer coefficients for the liquid and gas phases are calculated from effective diameter, the fetch:depth ratio, wind-velocity, the viscosity and density of air, and the estimated diffusivity of each contaminant in water and air. Default values for the viscosity of air (1.8x10⁻⁴ g/cm-sec) and the density of air (1.2x10⁻³ g/cm³ at STP) have been taken from Incropera and DeWitt (1985). Wilke and Lee's method provides estimates for the diffusivity of each contaminant in air, and Hayduk and Laudie's method provides estimates for each contaminant's diffusivity in water (Lyman *et al.*, 1990). The resulting estimates, which are based on a temperature of 15°C, are listed in Table 7-12.

Molecular Weights

The values presented in Table 7-13 are standard molecular weights for the contaminants of concern. These weights are used in the vapor loss component of the mass balance program.

Table 7-12

Diffusion Coefficients for Organic Contaminants

	Diffusivity in Air (cm²/sec)[a]	Diffusivity in Water (cm²/sec)[b]
Benzene	9.1×10^{-2}	7.8×10^{-6}
Benzo(a)pyrene	4.6×10^{-2}	4.3×10^{-6}
Bis(2-ethylhexyl)phthalate	3.3×10^{-2}	3.2×10^{-6}
Chlordane	4.5×10^{-2}	3.7×10^{-6}
DDT	4.1×10^{-2}	3.7×10^{-6}
Lindane	5.0×10^{-2}	4.5×10^{-6}
Polychlorinated biphenyls	5.7×10^{-2}	4.2×10^{-6}
Trichloroethylene	8.2×10^{-2}	7.3×10^{-6}

[a] Calculated using the Wilke and Lee method from Chapter 17-4 of Lyman *et al.* (1990), based on temperature of 15°C.

[b] Calculated using the Hayduk and Laudie method from Chapter 17-7 of Lyman *et al.* (1990), based on temperature of 15°C.

Table 7-13

Molecular Weights for Organic Contaminants

	Molecular Weight
Benzene	78.1
Benzo(a)pyrene	252.3
Bis(2-ethylhexyl)phthalate	390.6
Chlordane	409.8
DDT	354.5
Lindane	290.8
Polychlorinated biphenyls (Aroclor 1254)	325.1
Trichloroethylene	131.4

7.2.4 Size of Exposed Population

Estimates of human exposure and risk are derived by using the methods discussed in Sections 7.1.1 and 7.1.2 to estimate concentrations of each contaminant in groundwater and air near each sludge management area.

Estimates of concentrations in groundwater and ambient air are next combined with estimates for the density of human populations surrounding each individual SMA. Based on data from the National Well Water Association, we determine the number of households served by public or private wells for the county containing each surface disposal facility in the analytic survey of the NSSS. This value, when multiplied by the average household size and then divided by the total area of each relevant county, provides an estimate of the average density of populations using groundwater in each county. For the population exposed through the air pathway, a total population density was calculated based on the county population and county area.

These estimated densities are then multiplied by the area of land thought to be affected by each site in order to derive an estimate for the size of the exposed populations. For the air pathway, the exposed area is calculated as the area of a circle extended 3000 m from the edge of the site, less the area of the site. The exposed area for the groundwater pathway is taken as one quarter of the exposed area for air, centered on the downgradient direction of groundwater flow. Figure 7-1 depicts the exposed areas for both the groundwater and the air pathways. The potential for exposure to contaminated groundwater is conservatively assigned such that persons living within 1000 m of the site are exposed to the concentrations estimated at the downgradient edge of the site, and persons living between 1000-3000 m from the site are exposed to the concentrations estimated at 1000 m down gradient of the site. For the air pathway, predicted concentrations of contaminant at the property boundary (150 m from the edge in the down-wind direction) are conservatively applied to the entire population residing in the exposed area. The estimated sizes of exposed populations are listed in Table 7-14 and Table 7-15 for monofills, and Table 7-16 and Table 7-17 for surface impoundments.

7.3 BASELINE RISKS

Tables 7-18 through 7-21 provide our estimates for the health risks caused by monofilling of sewage sludge under baseline conditions. Risks of both cancer and non-cancer health effects have been estimated for an exposed population of about 1,260,000 persons for the air pathway and about 140,000 for the groundwater pathway. As can be seen from the tables, estimated risks of cancer from facilities represented by the monofill prototype are expected to be low, with about two incremental cases of cancer expected every hundred years (Table 7-18). More than 99 percent of this total risk can be attributed to arsenic from the groundwater pathway. Cancer risks to the highly exposed individual are estimated to be about 3×10^{-4}.

For non-carcinogenic risks, Tables 7-19 and 7-20, show that without background sources, none of the exposure of pollutants in monofills alone exceed the RfD, although the estimates for arsenic are 17% of the RfD for the average individual and 55% for the HEI. These values may

Figure 7-1

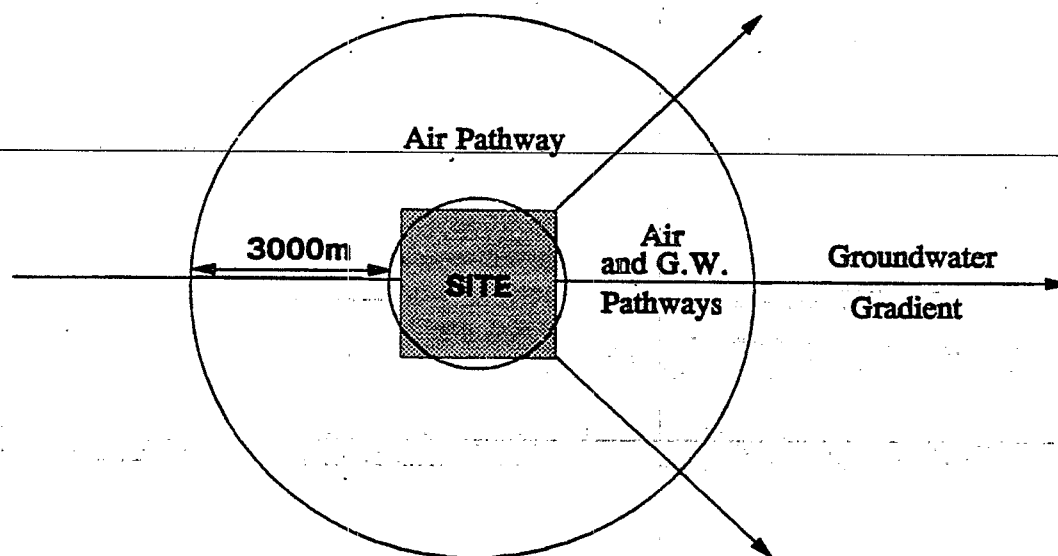**Zones of Human Exposure for Groundwater and Air Pathways**

Air Pathway

3000m

SITE

Air and G.W. Pathways

Groundwater Gradient

Table 7-14

Data for Calculating Sizes of Exposed Populations for Monofills

Facility Name	County	State	Sample Weight[a]	County Area[b] (mi²)	1980 Population[c]	Population for Groundwater[c]	Total Pop. Density (pers./mi²)	Pop. Density for GW (pers./mi²)
Sacramento Regional Wastewater	Sacramento	CA	1	971	783381	342543	807	353
Garden City WPCP	Chatham	GA	135	443	202226	88731	456	200
Spillville WWTP	Winneshiek	IA	135	690	21876	14293	32	251
Pittsfield Wastewater TP	Berkshire	MA	6	929	145110	66861	156	72
Hibbing WWTP-North	St. Louis	MN	30	6125	222229	116995	36	19
Southerly WWT Center	Cuyahoga	OH	1	459	1498400	605287	3264	1,319
N. Olmstead WWT Plant	Lorain	OH	30	495	274909	104847	555	212
Bucklin Pt. STP	Providence	RI	6	416	571349	250865	1373	603
Memphis T.E. Maxson WWTP	Shelby	TN	6	772	777113	288451	1007	374
Haskell ST STP	El Paso	TX	6	1014	479899	154885	473	153
Navasota STP	Grimes	TX	135	799	13580	9183	17	11
San Felipe STP	Val Verde	TX	30	3150	35910	14589	11	5

[a] Sample weights based on design flow reported in NSSS.
[b] U.S. Bureau of the Census, 1988.
[c] National Water Well Association, 1985.

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Table 7-15

Estimated Sizes of Exposed Populations for Monofills

Facility Name	County	State	Population Using GW 150-1000m[a]	Population for Air 150-1000m[a]	Population Using GW 1000m-3000m[b]	Population for Air 1000m-3000m[b]
Sacramento Regional Wastewater	Sacramento	CA	129	1181	899	8221
Garden City WPCP	Chatham	GA	9895	90211	68883	627960
Spillville WWTP	Winneshiek	IA	1023	6265	7124	43613
Pittsfield Wastewater TP	Berkshire	MA	158	1372	1100	9550
Hibbing WWTP-North	St. Louis	MN	210	1593	1460	11091
Southerly WWT Center	Cuyahoga	OH	483	4779	3359	33264
N. Olmstead WWT Plant	Lorain	OH	2325	24389	16187	169774
Bucklin Pt. STP	Providence	RI	1324	12063	9217	83970
Memphis T.E. Maxson WWTP	Shelby	TN	820	8841	5711	61544
Haskell ST STP	El Paso	TX	335	4157	2335	28935
Navasota STP	Grimes	TX	568	3359	3953	23380
San Felipe STP	Val Verde	TX	51	501	354	3485
TOTAL	--	--	17,000	160,000	120,000	1,100,000

[a] Population based on groundwater and air population densities presented in Table 7-14, and an exposed area of 0.37 mi[2] for gw, and 1.46 mi[2] for air.
[b] Population based on groundwater and air population densities presented in Table 7-14, and an exposed area of 2.55 mi[2] for gw, and 10.19 mi[2] for air.

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Table 7-16

Data for Calculating Sizes of Exposed Populations for Surface Impoundments

Facility Name	County	State	Sample Weight[a]	County Area[b] (mi²)	1980 Population[c]	Population for GW[c]	Total Pop. Density (pers./mi²)	Pop. Density for GW (pers./mi²)
Stickney WRP	Cook	IL	1	958	5253655	2037972	5484	2127
Corinth WWTP	Alcorn	MS	30	401	33036	15635	82	39
Hamilton WWTP-S.W.	Caldwell	MO	135	430	8660	5594	20	13
Missouri River WWTP	St. Louis	MO	6	506	973896	362228	1925	716
Coldwater Creek WWTP	St. Louis	MO	6	506	973896	362228	1925	716
Holdenville South STP	Hughes	OK	135	806	14338	8570	18	11
Kiski Valley WPCA	Armstrong	PA	30	646	77768	45542	120	70
Signal Mountain STP	Hamilton	TN	135	539	287740	115545	534	214
Abilene STP	Taylor	TX	6	917	110932	43132	121	47
E. Cedar Ck. Fresh Water Sup. Dis.	Kaufman	TX	135	788	39015	15193	50	19
Quechea WPC	Windsor	VT	135	972	51030	33747	53	35
Wabeno SD # Lagoons	Forest	WI	135	1011	9044	8313	9	8

[a] Sample weight based on design flow reported in NSSS.
[b] U.S. Bureau of the Census, 1988.
[c] National Well Water Association, 1985.

Table 7-17

Estimated Sizes of Exposed Populations for Surface Impoundments

Facility Name	County	State	Population Using GW 150-1000m[a]	Population for Air 150-1000m[a]	Population Using GW 1000-3000m[b]	Population for Air 1000-3000m[b]
Stickney WRP	Cook	IL	838	8636	5528	57000
Corinth WWTP	Alcorn	MS	461	3892	3040	25689
Hamilton WWTP-S.W.	Caldwell	MO	691	4282	4564	28259
Missouri River WWTP	St. Louis	MO	1691	18186	11161	120031
Coldwater Creek WWTP	St. Louis	MO	1691	18186	11161	120031
Holdenville South STP	Hughes	OK	565	3782	3730	24961
Kiski Valley WPCA	Armstrong	PA	833	5687	5496	37538
Signal Mountain STP	Hamilton	TN	11394	113493	75200	749076
Abiline STP	Taylor	TX	111	1143	733	7544
E. Cedar Ck. Fresh Water Sup.	Kaufman	TX	1025	10526	6763	69474
Quechea WPC	Windsor	VT	1845	11161	12179	73667
Wabeno SD # Lagoons	Forest	WI	437	1902	2885	12552
TOTAL	--	--	22,000	200,000	140,000	1,300,000

[a] Population based on groundwater and air population densities presented in Table 7-16, and an exposed area of 0.39 mi^2 for gw, and 1.6mi^2 for air.

[b] Population based on groundwater and air population densities presented in Table 7-16, and an exposed area of 2.60 mi^2 for gw, and 10.4mi^2 for air.

Table 7-18

Baseline Cancer Risks for Surface Disposal: Monofills[a]

Contaminant	Groundwater	Vapor	Total[b]
Arsenic	0.02	--	0.02
Benzene	<0.01	<0.01	<0.01
Benzo(a)pyrene	<0.01	<0.01	<0.01
Bis(2-ethylhexyl)phthalate	<0.01	<0.01	<0.01
Chlordane	<0.01	<0.01	<0.01
DDT	<0.01	<0.01	<0.01
Lindane	<0.01	<0.01	<0.01
PCBs	<0.01	<0.01	<0.01
Trichloroethylene	<0.01	<0.01	<0.01
Total	0.02	<0.01	0.02
AVERAGE INDIVIDUAL RISK[c]	1×10^{-5}	1×10^{-10}	1×10^{-6}
INDIVIDUAL RISK FOR HEI[d]	3×10^{-4}	4×10^{-9}	3×10^{-4}

[a] These results are based on reasonable worst-case input parameters and assumptions.
[b] Individual totals may not sum to totals because of independent rounding.
[c] Risk for average exposed individual of developing cancer from lifetime of exposure to pollutants from sludge.
[d] Risk for Highly Exposed Individual (HEI).

Table 7-19

Comparison of Baseline Exposure to Risk Reference Doses
for Monofills: Groundwater and Air Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	Average Exposure Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg/day)	Average Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[c] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.3×10^{-3}	430%	5.1×10^{-5}	17%
Cadmium	3.8×10^{-4}	5×10^{-4}	3.8×10^{-4}	76%	1.5×10^{-26}	<0.1%
Chromium III	1.5×10^{-3}	1×10^{0}	1.5×10^{-3}	<0.1%	1.2×10^{-6}	<0.1%
Copper	2.2×10^{-3}	5×10^{-3}	2.2×10^{-3}	44%	8.7×10^{-9}	<0.1%
Mercury	9.4×10^{-5}	3×10^{-4}	9.4×10^{-5}	31%	1.1×10^{-21}	<0.1%
Nickel	2.5×10^{-3}	2×10^{-2}	2.5×10^{-3}	12%	2.9×10^{-7}	<0.1%

[a] From the Integrated Risk Information System (IRIS).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Total exposure less background intake from other sources.

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Table 7-20

Comparison of Baseline Exposure for HEI to Risk Reference Doses
for Monofills: Groundwater and Air Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	HEI Exposure Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg/day)	HEI Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[c] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.4×10^{-3}	470%	1.6×10^{-4}	55%
Cadmium	3.8×10^{-4}	5×10^{-4}	3.8×10^{-4}	76%	3.2×10^{-25}	<0.1%
Chromium III	1.5×10^{-3}	1×10^{0}	1.5×10^{-3}	<0.1%	1.4×10^{-5}	<0.1%
Copper	2.2×10^{-3}	5×10^{-3}	2.2×10^{-3}	44%	5.7×10^{-8}	<0.1%
Mercury	9.4×10^{-5}	3×10^{-4}	9.4×10^{-5}	31%	3.4×10^{-21}	<0.1%
Nickel	2.5×10^{-3}	2×10^{-2}	2.5×10^{-3}	12%	1.7×10^{-6}	<0.1%

[a] From the Integrated Risk Information System (IRIS).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Total exposure less background intake from other sources.

Table 7-21

Baseline Non-Cancer Health Risks[a]
Monofilling

	Groundwater	Vapor	Total[b]
Non-Cancer Health Effects			
Persons Exceeding RFD[c]	0	0	0
Persons Crossing Cadmium Threshold[d]	<0.01	<0.01	<0.01
Persons Crossing PbB Threshold[e]			
Men	<0.01	<0.01	<0.01
Women	<0.01	<0.01	<0.01
Children	<0.01	<0.01	<0.01
Total Crossing PbB Threshold	<0.01	<0.01	<0.01
TOTAL NON-CANCER	<0.01	<0.01	<0.01
Expected Disease Cases from Lead[g]			
Men	<0.01	<0.01	<0.01
Women	<0.01	<0.01	<0.01
Children	<0.01	<0.01	<0.01
TOTAL	<0.01	<0.01	<0.01

[a] Based on "worst case" input parameters and assumptions.

[b] Individual values may not sum to totals because of independent rounding.

[c] Number of persons crossing RfD exposure threshold for one or more contaminants because of sludge reuse or disposal.

[d] Estimated number of persons with kidney cadmium levels rising above 200 μg/g. For discussion, refer to Chapter 2.

[e] Estimated number of persons with blood lead (PbB) levels crossing health effect thresholds. For discussion of assumptions and methodology, refer to Chapter 2.

[f] Number of persons crossing thresholds of lead or cadmium body burden because of exposure to sludge. The number of persons experiencing adverse non-cancer health effects from theses two contaminants is likely to be lower.

[g] Not all of the persons exceeding blood lead levels can be expected to suffer adverse health effects. These values represent estimates of actual disease cases resulting from sludge re-use or disposal. For discussion of disease and estimation methodology, refer to Chapter 2.

be significant when accounting for background exposure sources. For instance, combined background and monofill exposure sources yields estimates that exceed the RfD for arsenic by 430% for the average individual and 470% for the HEI; however, note that the background levels alone accounts for most of this exposure in that the background for arsenic exceeds the RfD by 400%.

As shown in Table 7-21, special calculations of risks for lead and cadmium suggest that fewer than one person per hundred years will cross body-burden thresholds or experience adverse health effects because of exposure through the groundwater pathway from sludge monofills. Because these estimates are based on several worst-case assumptions, actual risks for both cancer and non-cancer effects are likely to be lower.

Tables 7-22 through 7-25 provide the corresponding tables for the surface impoundment prototype. Within a total exposed population of about 1,500,000 persons for the vapor pathway and 170,000 for the groundwater pathway, we expect about one incremental case of cancer nationally per each twenty years this practice continues. As with the monofill prototype, more than 99 percent of the total risk is caused by arsenic through the groundwater pathway. Cancer risk to the highly exposed individual is estimated to be about $6x10^{-4}$.

For non-carcinogenic risks, predicted baseline and HEI exposure to all metals, excluding background intake, is lower than RfD levels. For arsenic, however, without background exposure, the baseline HEI exposure in a sludge impoundment represents 96 percent of the RfD (Table 24) and the baseline average exposure is estimated to be about 27 percent (Table 23). This implies that sludges with slightly higher levels of arsenic might lead to exposure levels in excess of the risk reference dose. However, we note that these estimates are based on several worst-case assumptions (e.g. that the well is assumed to be located at the property boundary in the down-gradient direction; the soil is assumed to be sandy; and the depth to groundwater is assumed to be 1 m), so that actual exposure and risks may be likely to be much lower than those predicted.

As shown in Table 7-25, we do not predict significant health risks from lead and cadmium, for which fewer than one person per hundred years is expected to cross body burden thresholds or experience adverse health effects as a result of exposure to lead or cadmium from surface impoundments.

Table 7-26 presents combined estimates of the monofilling and surface impoundment prototypes to give total baseline cancer risks from surface disposal. In total, we estimate fewer than seven incremental cases of cancer might be caused every hundred years as a result of surface disposal. As mentioned earlier, this estimate is based on several worst case assumptions, so actual risks are likely to be even lower.

7.3.1 Benefits from Regulatory Controls

Based on an analysis of those 26 facilities in the analytic survey of the NSSS believed to use surface disposal for their sludge, it appears unlikely that a significant number of facilities

Table 7-22

Baseline Cancer Cases[a]:
Surface Impoundments

	Groundwater	Vapor	Total[b]
Contaminant			
Arsenic	0.05	--	0.05
Benzene	<0.01	<0.01	<0.01
Benzo(a)pyrene	<0.01	<0.01	<0.01
Bis(2)ethylhexylphthalate	<0.01	<0.01	<0.01
Chlordane	<0.01	<0.01	<0.01
DDT	<0.01	<0.01	<0.01
Lindane	<0.01	<0.01	<0.01
PCBs	<0.01	<0.01	<0.01
Trichloroethylene	<0.01	<0.01	<0.01
Total	0.05	<0.01	0.05
AVERAGE INDIVIDUAL RISK[c]	2×10^{-5}	5×10^{-8}	2×10^{-6}
INDIVIDUAL RISK FOR HEI[d]	6×10^{-4}	1×10^{-5}	6×10^{-4}

[a] These results are based on reasonable worst-case input parameters and assumptions.
[b] Individual totals may not sum to totals because of independent rounding.
[c] Risk for average exposed individual of developing cancer from lifetime exposure to pollutants from sludge.
[d] Risk for Highly Exposed Individual (HEI).

Table 7-23

Comparison of Baseline Exposure to Risk Reference Doses
for Surface Impoundments: Groundwater and Air Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	Average Exposure Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg-day)	Average Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[c] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.3×10^{-3}	420%	8.0×10^{-5}	27%
Cadmium	3.8×10^{-4}	5×10^{-4}	3.8×10^{-4}	76%	1.2×10^{-13}	<0.1%
Chromium III	1.5×10^{-3}	1×10^{0}	2.0×10^{-3}	0.2%	5.9×10^{-4}	<0.1%
Copper	2.2×10^{-3}	5×10^{-3}	2.6×10^{-3}	51%	3.6×10^{-4}	7%
Mercury	9.4×10^{-5}	3×10^{-4}	9.4×10^{-5}	31%	7.7×10^{-11}	<0.1%
Nickel	2.5×10^{-3}	2×10^{-2}	2.6×10^{-3}	13%	8.6×10^{-5}	0.4%

[a] From the Integrated Risk Information System (IRIS).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Total exposure less background intake from other sources.

Table 7-24

Comparison of Baseline HEI Exposure to Risk Reference Doses
for Surface Impoundments: Groundwater and Air Pathways

	Background Intake (mg/kg-day)	Risk Reference Dose (mg/kg-day)	HEI Exposure Including Background[b] (mg/kg-day)	Percent RfD[c] (mg/kg-day)	HEI Exposure Excluding Background[d] (mg/kg-day)	Percent RfD[c] (mg/kg-day)
Arsenic	1.2×10^{-3}	3×10^{-4}	1.5×10^{-3}	490%	2.9×10^{-4}	96%
Cadmium	3.8×10^{-4}	5×10^{-4}	3.8×10^{-4}	76%	2.8×10^{-12}	<0.1%
Chromium III	1.5×10^{-3}	1×10^{0}	1.1×10^{-2}	1%	9.3×10^{-3}	.1%
Copper	2.2×10^{-3}	5×10^{-3}	3.1×10^{-3}	61%	8.6×10^{-4}	17%
Mercury	9.4×10^{-5}	3×10^{-4}	9.4×10^{-5}	31%	1.3×10^{-10}	<0.1%
Nickel	2.5×10^{-3}	2×10^{-2}	3.6×10^{-3}	17%	1.1×10^{-3}	5%

[a] From the Integrated Risk Information System (IRIS).
[b] Total exposure including background intakes from other sources.
[c] (Exposure/RfD) x 100.
[d] Total exposure less background intake from other sources.

Table 7-25

Baseline Non-Cancer Health Risks[a]
Surface Impoundment

	Groundwater	Vapor	Total[b]
Non-Cancer Health Effects			
Persons Exceeding RFD[c]	0	0	0
Persons Crossing Cadmium Threshold[d]	<0.01	<0.01	<0.01
Persons Crossing PbB Threshold[e]			
Men	<0.01	<0.01	<0.01
Women	<0.01	<0.01	<0.01
Children	<0.01	<0.01	<0.01
Total Crossing PbB Threshold	<0.01	<0.01	<0.01
TOTAL NON-CANCER	<0.01	<0.01	<0.01
Expected Disease Cases from Lead[g]			
Men	<0.01	<0.01	<0.01
Women	<0.01	<0.01	<0.01
Children	<0.01	<0.01	<0.01
TOTAL	<0.01	<0.01	<0.01

[a] Based on "worst case" input parameters and assumptions.

[b] Individual values may not sum to totals because of independent rounding.

[c] Number of persons crossing RfD exposure threshold for one or more contaminants because of sludge reuse or disposal.

[d] Estimated number of persons with kidney cadmium levels rising above 200 $\mu g/g$. For discussion, refer to Chapter 2.

[e] Estimated number of persons with blood lead (PbB) levels crossing health effect thresholds. For discussion of assumptions and methodology, refer to Chapter 2.

[f] Number of persons crossing thresholds of lead or cadmium body burden because of exposure to sludge. The number of persons experiencing adverse non-cancer health effects from theses two contaminants is likely to be lower.

[g] Not all of the persons exceeding blood lead levels can be expected to suffer adverse health effects. These values represent estimates of actual disease cases resulting from sludge re-use or disposal. For discussion of disease and estimation methodology, refer to Chapter 2.

Table 7-26

Baseline Cancer Cases[a]:
Total Surface Disposal

	Groundwater	Vapor	Total[b]
<u>Contaminant</u>			
Arsenic	0.07	--	0.07
Benzene	<0.01	<0.01	<0.01
Benzo(a)pyrene	<0.01	<0.01	<0.01
Bis(2)ethylhexylphthalate	<0.01	<0.01	<0.01
chlordane	<0.01	<0.01	<0.01
DDT	<0.01	<0.01	<0.01
Lindane	<0.01	<0.01	<0.01
PCBs	<0.01	<0.01	<0.01
Trichloroethylene	<0.01	<0.01	<0.01
Total	0.07	<0.01	0.07
AVERAGE INDIVIDUAL RISK[c]	2×10^{-5}	3×10^{-8}	2×10^{-6}
INDIVIDUAL RISK FOR HEI[d]	6×10^{-4}	1×10^{-5}	6×10^{-4}

[a] These results are based on reasonable worst-case input parameters and assumptions.
[b] Individual totals may not sum to totals because of independent rounding.
[c] Risk for average exposed individual of developing cancer from lifetime exposure to pollutants from sludge.
[d] Risk for Highly Exposed Individual (HEI).

using this method will be forced to change to other practices as a result of the regulation. Although the additional management practices required by the regulation are likely to reduce expected human exposure and risk, their effects are not directly quantifiable by methods used for this analysis. Nevertheless, the potential reduction in risk likely to be achieved by the regulation should not exceed our estimates of risk for baseline conditions. Therefore we estimate that the potential benefit of regulation falls within the range of zero to our estimate of baseline risk.

7.3.2 Uncertainties and Limitations

This analysis is based on numerous simplifying assumptions (almost all of them conservative). To the extent that actual conditions at individual facilities differ from those assumed, true exposure and risk will differ from the estimates provided here. For example, if the local depth to groundwater exceeds 1 m, or if the hydraulic conductivity of the local soil medium is less than that of sand, actual contamination of groundwater beneath a surface disposal facility is likely to be lower than that calculated for this analysis. Similarly, concentrations in groundwater at distances greater than 150 m in directions other than downgradient are likely to be lower than those calculated for this analysis. On the other hand, a non-homogeneous or fractured medium beneath a surface disposal facility might lead to the contamination of groundwater at higher levels than those predicted by VADOFT and AT123D. Moreover, if the number of persons drinking groundwater near a surface disposal facility exceeds the average density calculated for the county as a whole, risks at these sites may be underestimated. Finally, if the density of human populations or concentrations of contaminants in sludge for other surface disposal facilities differs systematically from those predicted based on the analytic component of the NSSS, risks might be underestimated. Nevertheless, estimates of exposure and risk derived in this analysis are believed to be conservative, and unlikely to underestimate true risks.

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APPENDIX A

Partitioning of Contaminant Among Air, Water, and Solids in Soil

Calculations used to derive criteria for groundwater, surface water, and air pathways are based on the assumption that equilibrium is maintained between concentrations of contaminant in the air-filled pore space, the water-filled pore-space, and the solid particles of soil. Equilibrium partitioning between dissolved and gaseous phases is described by Henry's Law constants; partitioning between adsorbed and dissolved phases is described by soil-water partition coefficients. From these assumptions and the definitions of concentration are derived the equations used to describe partitioning.

Define:

C_s = concentration of adsorbed contaminant on solid soil particles (kg/kg),

C_w = concentration of dissolved contaminant in water-filled pore space of soil (kg/m^3),

C_a = concentration of contaminant in air-filled pore space of soil (kg/m^3), and

C_t = total concentration of contaminant in soil (kg/m^3).

Mathematically:

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

and:

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

where:

M_{cs} = mass of adsorbed contaminant (kg),

M_s = mass of soil (kg),

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

V_w = volume of water in soil (m^3),

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

V_a = volume of air in soil (m^3),

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

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

V_s = volume of solids in soil (m^3).

The equilibrium distribution coefficient (KD, in m³/kg) between adsorbed and dissolved phases can be defined as:

$$KD = \frac{[M_{cs}/M_s]}{[M_{cw}/V_w]} = \frac{M_{cs} \; V_w}{M_s \; M_{cw}}$$

The dimensionless Henry's Law constant ($\dot{H}$) describing the partitioning between gaseous and dissolved phases is defined as:

$$\dot{H} = \frac{[M_{ca}/V_a]}{[M_{cw}/V_w]} = \frac{M_{ca} \; V_w}{V_a \; M_{cw}}$$

The bulk density of soil (BD, in kg/m³) is defined as:

$$BD = M_s \; / \; V_t$$

The air-filled porosity of soil (θ_a) is defined as:

$$\theta_a = V_a \; / \; V_t$$

water-filled porosity (θ_w) is defined as:

$$\theta_w = V_w \; / \; V_t$$

and the total porosity of soil (θ_t) is defined as:

$$\theta_t = (V_t - V_s) \; / \; V_t = \theta_a + \theta_w$$

The above definitions can be combined to yield:

$$\frac{C_t}{C_a} = \frac{KD \; BD}{\dot{H}} + \frac{\theta_w}{\dot{H}} + \theta_a$$

and:

$$\frac{C_t}{C_w} = BD \; KD + \theta_w + \dot{H} \; \theta_a$$

and:

$$\frac{C_t}{C_s} = BD + \frac{\theta_w}{KD} + \frac{\theta_a \; \dot{H}}{KD}$$

These relationships are used throughout the calculations used to derive criteria. Where dry-weight concentrations of contaminant in sludge or soil are involved, the equations are modified slightly, based on the definition:
where:

$$C_{dw} = \frac{M_{ct}}{M_s} = \frac{M_{ct}}{V_t \, BD} = \frac{C_t}{BD}$$

C_{dw} =   dry-weight concentration of contaminant in soil.

APPENDIX B

Derivation of First-Order Coefficient for Losses to Leaching

U.S. EPA (1989f) provides an equation for computing a first-order loss rate to leaching of contaminant from treated soil:

$$K_{lec} = \frac{NR}{BD\ KD\ d_i}$$

where:

K_{lec}	=	first-order loss rate coefficient for leaching (yr^{-1}),
NR	=	annual recharge rate (m/yr),
BD	=	bulk density of soil (kg/m^3),
KD	=	soil-water distribution coefficient for contaminant (m^3/kg), and
d_i	=	depth of incorporation for sludge (m).

This appendix describes a modified version of that equation.

The basic strategy for deriving a coefficient for first-order loss to leaching is to estimate the mass of contaminant expected to be lost each year and divide by the available mass of contaminant. The mass of contaminant that will be lost to leaching in any interval of time can be described by the volume of water percolating through the treated soil multiplied by the average concentration of contaminant in that water:

$$FA_{gw} = NR\ C_{lec}\ 10,000$$

where:

FA_{gw}	=	flux of leached contaminant from treated soil (kg/ha-yr),
NR	=	recharge to groundwater beneath the treated soil (m^3/m^2-yr, or m/yr),
C_{lec}	=	concentration of contaminant in water infiltrating through the treated soil (i.e., leaching from the site) (kg/m^3), and
10,000	=	constant to convert units from (kg/m^2-yr) to (kg/ha-yr).

From Appendix A, the concentration of contaminant in leachate is related to the total concentration (by volume) of contaminant in soil as:

$$C_{lec} = C_t\ /\ [BD \cdot KD\ +\ \theta_w\ +\ H\theta_a]$$

where:

C_t	=	total concentration of contaminant in treated soil (kg/m^3),
θ_w	=	water-filled porosity of soil (dimensionless),
H	=	Henry's Law Constant for contaminant (dimensionless), and
θ_a	=	air-filled porosity of soil (dimensionless).

This flux of contaminant must be translated into a first-order loss coefficient so that:

$$\frac{dC_t}{dt} = -K_{lec} C_t$$

where:

K_{lec} = first-order loss rate coefficient for leaching (yr^{-1}),
C_t = total concentration of contaminant in soil (kg/m^3), and
t = time (yr).

K_{lec} is estimated with the approximation:

$$K_{lec} = \frac{[dC_t/dt]}{C_t} \approx \frac{[\Delta C_t/\Delta t]}{C_t} = \frac{[\Delta M_{ct}/\Delta t]}{M_{ct}}$$

which is equivalent to:

$$K_{lec} = \frac{FA_{gw} A}{M_{ct}}$$

where:

M_{ct} = mass of contaminant in soil (kg),
Δt = one year,
FA_l = flux of contaminant leaching to groundwater (kg/ha-yr),
A = area of land application site (m^2), and
10^{-4} = constant to convert units from (m^2) to (ha).

If $A = (V_t/d_i)$, this result can be combined with results from Appendix A to yield:

$$K_{lec} \approx \frac{NR\ C_{lec}}{C_t\ d_i} = \frac{NR}{[BD\ KD + \theta_w + H\theta_a]\ d_i}$$

We use this equation to predict contaminant loss to leaching for both land application and the monofill prototype for surface disposal.

APPENDIX C

Calculation of a "Square Wave" for the Groundwater Pathway

We estimate potential human exposure and risk through the groundwater pathway for both land application and surface disposal of sewage sludge. To prepare input for the VADOFT model of contaminant transport through the unsaturated zone, we conservatively assume for both land application and surface disposal that pollutant is consistently loaded into the top of the unsaturated zone at the maximum rate estimated by mass balance calculations. We constrain the duration of this constant pulse, or "square wave" so that the total mass of contaminant leaching or seeping from the site is conserved. Although our 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 our methods for estimating the magnitude and duration of the "square wave" of pollutant loading for land application and both prototype facilities for surface disposal.

C.1 Land Application

Both metals and organic contaminants accumulate in soil with repeated applications of sludge. We assume that all competing loss processes for contaminant in 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 contaminant in treated soil at time t (kg) and
K_{tot} = lumped, first-order loss rate for contaminant (yr^{-1}).

If contaminant loading to treated soil is approximated as a continuous process, the mass of contaminant 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 contaminant to site (kg/yr).

As t approaches infinity, M_t therefore approaches $(PA)/K_{tot}$ and yearly loss approaches yearly loading.

For organic contaminants, we assume that sludge has been applied repeatedly until steady-state is achieved. In other words, contaminant has accumulated in the soil until total yearly losses through runoff, degradation, leaching, and volatilization (which are assumed to be proportional to the concentration in soil) catch up with yearly loadings of contaminant to soil. Our estimates of risks from organic contaminants 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 contaminant 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 metals, this condition of steady-state is not necessarily reached. According to the loss coefficients calculated in this analysis, arsenic is lost most rapidly from treated soil, and lead least rapidly. With a lumped loss rate of 0.12 per year, arsenic approaches a steady-state concentration equal to about 8 times its annual loading. After about 10-20 years, yearly losses closely approximate yearly loadings of 1 kg/ha. Lead, the least mobile of the metals evaluated, is depleted from treated soil at an estimated annual rate of only 0.0073 per year. If lead were applied repeatedly to the soil, its expected concentration would increase significantly each year for the first 500-600 years of repeat applications; yearly losses would not begin to approximate yearly loadings for several centuries.

Pollution of groundwater by metals from sludge depends not only on the cumulative loading of metals, but also on the period of time in which this cumulative loading takes place. We assume that metals are loaded into treated soil through N consecutive, yearly applications of sludge; after N years, applications are discontinued. To capture the risks associated with the peak rate at which contaminant leaves the soil layer, we use this peak rate (calculated for the Nth year of application) for the calculations. We calculate the length of the square wave by dividing the total (cumulative) loading of contaminant by this maximum rate of loss:

$$TP = \frac{N\,PA}{PA\,(1-e^{-K_{tot}N})} = \frac{N}{(1-e^{-K_{tot}N})}$$

where:

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

C.2   Surface Disposal: Monofill Prototype

Our modeling of the groundwater pathway for the monofill prototype of surface disposal is similar to that for land application. For both cases, we assume the site receives repeated loadings of contaminant 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 contaminant in monofill at end of year LF (kg), and

PA = annual rate that contaminant is loaded into the monofill (kg/yr).

The length of time this maximum rate of loss could be maintained is then:

$$TP = \frac{LF \, PA}{PA \, (1-e^{-K_{tot} \, LF})} = \frac{LF}{1-e^{-K_{tot} \, LF}}$$

C.3 Surface Disposal: Surface Impoundment Prototype

For the surface impoundment prototype of surface disposal, our 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. We also assume that the flux of contaminant leaching from the impoundment is constant with respect to time, at least until the total mass of contaminant 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 contaminant entering the impoundment each year, multiplied by the expected lifetime of the facility and divided by the amount lost each year:

$$TP = \frac{3.2 \times 10^{-8} \, PA \, TF}{PA \, f_{act}} = \frac{3.2 \times 10^{-8} \, TF}{f_{act}}$$

where:

3.2×10^{-8} = conversion factor for (sec) to (yr),

PA = rate that contaminant is loaded into the surface impoundment (kg/yr),

TF = active lifetime of site (sec), and

f_{act} = fraction of each year's contaminant loading lost from the site each year to all processes combined (dimensionless).