This appendix presents a brief summary of the groundwater (Section H.1) and the multipathway (Section H.2) risk assessment for the land storage of newly-identified mineral processing wastes under the modified prior treatment baseline. This effort builds on previous efforts on the identification of recycled waste streams, the estimation of recycled volumes, the identification of waste management strategies, and in the development of waste constituent concentration data, described in the December 1995[1] and August 1996[2] Draft RIAs for the Phase IV LDRs and in the Technical Memorandum reporting the Revised Risk Assessment Results for groundwater submitted to EPA in July, 1996[3].

The analyses presented in this appendix differ from the previous risk assessments for mineral processing wastes, first, in that risks from land storage, rather than ultimate disposal, are evaluated. The assessment is limited to only those waste streams that have been identified as being recycled by the Agency. This effort also differs from previous risk assessments in that it only addresses risks under the modified prior treatment baseline, and does not quantitatively evaluate residual risks under any of the proposed regulatory alternatives. This is because, under three of the regulatory alternatives, requirements would be imposed requiring the storage of recycled materials in either buildings or tanks, rather than on the ground, and release and transport models appropriate to evaluating risks associated with these technologies are not available. Thus, the assessment presented below evaluates only "baseline" risks by identifying specific waste streams and constituents posing risks of regulatory concern under the modified prior treatment assumptions. These risks would be reduced under the proposed regulatory controls, but quantitative estimates of the benefits of these regulatory controls (e.g. the numbers of facilities going from high-risk to low-risk categories) are not developed. Under Option 4, no controls would be imposed on the storage of recycled materials, so there would be no health benefits.

Finally, the risk assessment described in this appendix differs from previous risk assessments for mineral processing wastes in that risks are evaluated for pathways other than groundwater ingestion. As in previous risk assessments, we evaluate leachate releases from land-based units to groundwater and subsequent groundwater ingestion. However, in Section H.2 we evaluate the risks associated with other release events, transport and exposure media, and exposure pathways. This multipathway analysis evaluates risks associated with air particulate and surface runoff releases from waste piles, and risks arising

[1] ICF Incorporated, *Regulatory Impact Analysis of the Supplemental Proposed Rules Applying Phase IV Land Disposal Restrictions to Newly Identified Mineral Processing Wastes*, Submitted to the Office of Solid Waste, US Environmental Protection Agency, December 1995.

[2] ICF Incorporated, *Regulatory Impact Analysis of the Application of Phase IV Land Disposal Restrictions to Newly Identified Mineral Processing Wastes*, Submitted to the Office of Solid Waste, US Environmental Protection Agency, August 1996.

[3] ICF Incorporated, *Revised Results of Mineral Processing Wastes Risk and Benefits Assessments Using Constituent-Specific DAFs Derived for Mineral Processing Waste*, Submitted to the Office of Solid Waste, US Environmental Protection Agency, July 1, 1996.

from surface impoundment runon events and inlet/outlet control failures. The transport and exposure media that are evaluated include air, soil and surface water, as well as home-grown crops and game fish.

H.1 RISK ASSESSMENT METHODS AND RESULTS FOR THE GROUNDWATER PATHWAY

This section of Appendix H presents a brief summary of the groundwater pathway risk assessment for the land storage of newly identified mineral processing wastes under the Modified Prior Treatment baseline. The analyses presented below employ very similar methods for estimating constituent releases, groundwater exposure concentrations, and health risks as were employed in the previous analyses. The only major differences from previous efforts are that groundwater DAFs have been derived using constituent concentration data for only those facilities and waste streams identified as being involved in recycling, and that the DAFs have been derived assuming a release duration of 20 years, corresponding the assumed life of the recycling storage facilities, instead of the much longer release period assumed for disposal facilities. This section addresses only those potential health risks arising from exposures through consumption of contaminated groundwater. Potential risks associated with other release, transport, and exposure pathways are evaluated in Section H.2.

H.1.1 Methods and Assumptions

H.1.1.1 Regulatory Scenarios

As noted previously, risks have been assessed for the modified prior treatment baseline. Under this baseline, it is assumed that recycled spent materials and sludges and byproducts from mineral processing will be stored land-based units prior to recycling. Nonwastewaters would be stored in unlined waste piles, and wastewater and liquid nonwastewater streams would be stored in unlined surface impoundments. Unlike the situation for disposal facilities, it is assumed that, where two or more recycled streams are generated at a facility, the streams would be stored in separate units prior to recycling, and that there would be no comanagement. Also, it has been assumed that the storage units would be sized to just accommodate the required amount of recycled material; three months' generation rates in the case of nonwastewaters, and one month's generation in the case of wastewaters and liquid nonwastewaters. The assumptions used to evaluate the size and configuration of storage facilities are described in detail in the December 1995 RIA and in Appendix D of this RIA.

H.1.1.2 Identification of Waste Streams

Under the modified prior treatment baseline, it is assumed that all recycled spent materials and recycled sludges and byproducts would be managed in land-based units. Thus, all of these waste streams were candidates for the storage risk assessment. Constituent concentration data were available for only some of these streams, however. Risks were therefore evaluated only for the 14 recycled waste streams listed in Exhibit H-1. Two of the waste streams (aluminum and alumina cast house dust and zinc waste ferrosilicon) are nonwastewaters, and the remainder are wastewaters or liquid nonwastewaters.

Although groundwater pathway risks were calculated for only 14 of the 118 total mineral processing waste streams, these streams represent substantial proportions of the total generated wastes and an even higher proportion of the recycled wastes. Depending on which estimate of waste generation is used (minimum, expected, or maximum), the 14 recycled streams included in the risk analysis represent between 32 and 42 percent of the total waste generation, and account for between 57 and 68 percent of the total recycled volume. This is because constituent concentration data are available for a substantial proportion of the high-volume waste streams. The extent of coverage of the storage risk assessment for the various commodity sectors is summarized in Attachment H.A to this appendix.

H.1.1.3 Waste Characterization Data and Release Concentration Estimates

The original source of constituent concentration data for the recycled materials used in the pre-LDR risk estimates is the same as that used in the RIA sample-specific risk assessment. These data are summarized in Appendix K of the December 1995 RIA. Consistent with the previous risk assessment, constituent concentration data from both bulk samples and EP extraction analysis were used in the risk assessment, when they were available, to develop separate risk estimates for the same waste streams. This was done in order to make the best possible use of the available data, and because in many cases we could not be sure that EP and bulk analyses from a given waste stream were from the same samples or batch of waste.

EXHIBIT H-1. RECYCLED STREAMS INCLUDED IN THE STORAGE RISK ANALYSIS

Commodity	Recycled Stream
Aluminum and Alumina	Cast House Dust
Beryllium	Chip Treatment Wastewater
Copper	Acid Plant Blowdown
Elemental Phosphorus	AFM Rinsate
Elemental Phosphorus	Furnace Scrubber Blowdown
Rare Earths	Process Wastewater
Selenium	Plant Process Wastewater
Tantalum, Columbium, and Ferrocolumbium	Process Wastewater
Titanium and Titanium Oxide	Leach Liquor and Sponge Wastewater
Titanium and Titanium Oxide	Scrap Milling Scrubber Water
Zinc	Waste Ferrosilicon
Zinc	Spent Surface Impoundment Liquids
Zinc	Waste Water Treatment Plant Liquid Effluent
Zinc	Process Wastewater

Constituent data from 187 waste samples were used to develop DAF values and to evaluate risks from land storage. Exhibit H-2 presents a breakdown of the samples by facility and types of analysis. It can be seen that the large majority of the data come from bulk samples, and the majority of the samples are from facilities whose identities and locations are unknown. Only three of the 185 samples are from nonwastewater streams managed in waste piles, with the remainder from wastewater and liquid nonwastewater streams managed in surface impoundments.

EXHIBIT H-2. DISTRIBUTION OF SAMPLES BY WASTE, SAMPLE, AND FACILITY TYPE

Waste Type	Bulk Samples	EP Extraction Sample	Known Facilities	Unknown Facilities
Nonwastewater	2	1	0	3
Wastewater	92	25	63	54
Liquid Nonwastewater	51	16	12	55

Arsenic concentration data were available for 75 of the waste samples, allowing the calculation of cancer risks for these samples. Noncarcinogen concentration data for constituents having DAF values and RfDs were available for 135 samples, which include all 75 of those with arsenic data. For WW streams, the bulk concentration sample results were used directly as release concentration estimates. For LNWW and NWW, EP leachate concentrations were also used directly as release concentrations. For LNWW and WW bulk samples, release concentrations (mg/l) were conservatively estimated as being equal to the bulk constituent concentrations (mg/kg) divided by 20. This approach conservatively assumes that all waste constituents are 100 percent leachable.

All of the analytical results from every sample were used to evaluate risks, with one exception. A single bulk analytical result for selenium (100,000 mg/l) in zinc process wastewater was omitted from the risk analysis because this value far exceeds the maximum solubility of most naturally occurring selenium compounds, and is clearly spurious, based on the results for other samples from the same waste stream.

H.1.1.4 Exposure Assessment

Analogous to the procedures used in previous risk assessments, two sets exposure of exposure estimates were developed. Central tendency (CT) exposure concentrations were estimated by dividing the release concentrations of each constituent from each waste stream by the 75th percentile DAF value derived for that constituent. High-end (HE) exposure concentrations were estimating by dividing the release concentrations by the 95th percentile DAF values. The rationale for using the 75th percentile DAFs rather than, for example, the 50th percentile value was that the EPACMTP model used to derive DAFs does not consider fractured or channeled flow or other facilitated transport mechanisms which may occur at some sites, resulting in higher groundwater concentrations than those predicted for homogeneous flow processes modeled by EPACMTP. The 75th percentile of the DAF distribution was therefore judged by EPA to be more nearly representative of dilution conditions for the entire population of facilities than the 50th percentile. The 95th percentile constituent-specific DAF values were used to estimate high-end (HE) groundwater concentrations in keeping with the definition of a high-end receptor as someone exposed at levels between the 90th and 99th percentiles of all exposed individuals. Separate exposure and risk estimates were developed for each waste sample, analogous to the approach used for the analysis of disposal risks. DAF values were derived separately for waste piles and surface impoundments, and used to estimate exposure concentrations for nonwastewaters and liquid nonwastewaters/wastewaters, respectively. The DAF values derived by EPA for use in the mineral processing recycled materials storage risk assessment are shown in Exhibit H-3.

It can be seen from this exhibit that the DAF values (both the 75th and 95th percentile) derived for the management of recycled materials in waste piles are generally very much higher than those derived for surface impoundments. This is due primarily to the lower leachate volume generated by the waste piles than by surface impoundments. In the waste piles, leachate generation is limited by rainfall (and a large

proportion of the facilities are in relatively dry areas), whereas surface impoundments provide their own leachate source to drive releases, in the form of the aqueous liquid wastes which they contain.

For all of the constituents, the CT DAF values for waste piles are greater than 10^{12}, implying, as will be seen below, such high dilutions of leachate that CT risks from all the constituents released from waste piles are well below levels of regulatory concern. The HE DAF value for waste piles are much lower for most constituents (in the range of 10^4 to 10^6), but still generally several orders of magnitude above even the corresponding HE values for surface impoundments. Thus, even the HE exposure concentrations associated with releases from waste piles result in relatively low risks.

The CT surface impoundment DAF values for all of the constituents but cyanide, lead, and mercury are all around 1000. The HE DAF values surface impoundments are mostly less than 100, with the exceptions being vanadium, cyanide and lead. As will be seen below, these lower DAF values imply higher risks for given constituent concentrations than do the DAF values for waste piles.

EXHIBIT H-3. DAF VALUES USED IN THE STORAGE RISK ASSESSMENT

	Waste Pile DAF		**Surface Impoundment DAF**	
Constituent	75th Percentile	95th Percentile	75th Percentile	95th Percentile
Antimony	$>10^{12}$	$2.0X10^6$	$2.7X10^3$	$5.3X10^1$
Arsenic	$>10^{12}$	$1.8X10^5$	$1.1X10^3$	$3.37X10^1$
Barium	$>10^{12}$	$1.2X10^4$	$1.5X10^2$	2.9
Cadmium	$>10^{12}$	$2.4X10^6$	$2.1X10^3$	$1.3X10^1$
Chromium (+6)	$>10^{12}$	$9.9X10^4$	$6.3X10^2$	$2.4X10^1$
Cyanide	NA*	NA	$2.9X10^9$	$1.8X10^3$
Lead	$>10^{12}$	$>10^{12}$	$>10^{12}$	$1.2X10^3$
Mercury	$>10^{12}$	$3.3X10^6$	$1.5X10^3$	$2.6X10^1$
Nickel	$>10^{12}$	$3.4X10^6$	$1.6X10^3$	$1.2X10^1$
Selenium	$>10^{12}$	$2.4X10^4$	$1.9X10^2$	6.2
Silver	$>10^{12}$	$2.5X10^4$	$4.3X10^2$	4.2
Thallium	NA	NA	$3.5X10^3$	$9.0X10^1$
Vanadium	NA	NA	$>10^{12}$	$>10^{12}$
Zinc	$>10^{12}$	$5.8X10^6$	$6.7X10^2$	3.9

* DAFs were not derived for these constituents because no analytical data were reported for these constituents in any of the wastes disposed in waste piles.

H.1.1.5 Risk Characterization

Daily intakes of the waste constituents due to groundwater ingestion are estimated in precisely the same way as described in the July memorandum.[2] For arsenic, (the only constituent considered to be carcinogenic by ingestion), lifetime daily intake is calculated for a 70-kg adult drinking 1.4 liters per day of contaminated groundwater 350 days per year for nine years, assuming a 70-year life-span for averaging purposes. Daily intakes of noncarcinogens are calculated using the same assumptions, except that the dose is averaged over the period of exposure, rather than over a lifetime.

Lifetime cancer risks are calculated by multiplying the lifetime daily arsenic intake (from those waste streams having arsenic as a constituent) by the ingestion Cancer Slope Factor for arsenic of 1.5 $(mg/kg\text{-}day)^{-1}$. Noncancer hazard quotients for exposure to waste constituents are calculated by dividing the daily constituent intake of each constituent by its ingestion pathway Reference Dose (RfD). The toxicity values used in the assessment all come from EPA's IRIS database, and are current as of December 1996.

H.1.2 Estimation of Numbers of Facilities at Specific Risk Levels

In the previous risk and benefits assessments performed for the disposal of mineral processing wastes, the measure of the benefits of the regulatory alternatives was the reduction in the number of facilities at which waste management would results in risks above levels of regulatory concern. As noted previously, no risk assessment has been performed for the storage of recycled materials under any of the regulatory alternatives. Thus, a similar quantitative benefits assessment is not possible for waste storage under the various regulatory alternatives.

This risk assessment, however, does provide an estimate of the numbers of facilities in the various commodity sectors at which risks exceed levels of regulatory concern under the modified prior treatment baseline. This estimate presents an upper bound limit on the possible regulatory benefits; e.g., if regulation reduced releases of all waste constituents to zero, then all of the baseline risks greater than levels of regulatory concern would go to zero as well. Less than perfect control of releases would result in correspondingly smaller reductions in the number of facilities at high risk levels (yielding lower benefits), although the magnitude of risk reduction has not been quantified.

H.1.2.1 Estimation of the Numbers of Facilities Storing Recycled Materials

Risks have been assessed for all of the commodity sectors and waste stream which have been identified by the Agency as being involved in recycling under the modified prior treatment baseline, and for which constituent concentration data are available, as identified in Exhibit H-1. In this analysis, it has been assumed that all of the facilities in each commodity sector not only generate but store recycled sludges, byproducts, and spent materials. Thus, the numbers of facilities included in the assessment are simply equal the estimated numbers of facilities in the commodity sector, which is exactly analogous to the approach taken in the analysis of the risks associated with waste disposal.

In this analysis, it has been assumed that all of the facilities identified as generating the recycled streams also recycle them, under both CT and HE assumptions. Thus, the number of waste stream-facility combinations in each commodity sector is the same for the HE and CT risk estimates. Analogous to the previous analysis, where a single facility stores more than one waste stream, it is counted as more than one "waste stream-facility combination."

H.1.2.2 Attribution of Risks to Facility-Waste Stream Combinations

As was the case for the analysis of disposal risks, the number of risk estimates (one from each sample) does not always equal (in fact, rarely equals) the number of facilities storing the wastes. Thus, in estimating the distribution of risks across a commodity sector it is necessary to apportion the risks from individual samples to the CT and HE numbers of facilities.

The procedures used to do this are described in detail in Section 2.2.2 of the July memorandum. Basically, the approach involves distributing risk levels across the facilities in commodity sector in as close to the same proportions as they are distributed across the individual waste samples from the waste generated by that sector. For example, if there are two samples of a given waste stream in the data base, one with an estimated cancer risk of 10^{-6} and one with an estimated risk of 10^{-2}, half the facilities in the commodity sector would be placed in the "$<10^{-5}$" category, and half would fall into the "10^{-3} to 10^{-2}" category. One of the outcomes of using this approach is that not every risk result above a level of concern translates into a waste stream-facility combination. For example if there is only a very small proportion of samples (for example, one in 20) giving high risks, this may translate into zero waste stream facility combinations if there are, for example, only two or three total facilities in the industry. The July memo describes the approaches used for rounding the estimates of facilities in the various risk categories, where apportionment cannot be done evenly, and for combining risk estimates from multiple samples from a single facility so, so as not to give them undue weight across an entire industry.

H.1.3 Results of the Groundwater Risk Assessment

H.1.3.1 Risk Assessment Results by Sample

Exhibit H-4 summarizes the carcinogenic groundwater risk results for the 75 samples identified as containing arsenic, the sole ingestion pathway carcinogen among the waste constituents. Using the central tendency DAF values, the calculated cancer risks for 49 of these samples were less than 10^{-5}, the level of regulatory concern, and the risks for 26 of the samples exceeded this value. Cancer risks exceeded 10^{-5} for one or more samples from only four waste streams; copper acid plant blowdown, elemental phosphorous furnace scrubber blowdown, tantalum, columbium, and ferrocolumbium process wastewater, and zinc spent surface impoundment liquids. The highest risks cancer risks were associated with three samples of copper acid plant blowdown (10^{-3} to 10^{-2}). This waste stream accounted for 14 of the 16 samples with the highest CT cancer risks. The next highest risks (in the 10^{-4} to 10^{-3} range) were associated with one sample each from tantalum process wastewater and zinc spent surface impoundment liquids.

Using the high-end (CT) DAF values, cancer risks calculated for the groundwater pathway exceeded 10^{-5} for 50 of the 75 samples. Under this set of assumptions, risks for at least one sample exceeded 10^{-5} for 10 of the 14 waste streams evaluated. The highest risks (25 of 30 samples > 10^{-5}, highest risk category >10-1) were again associated with copper acid plant blowdown, with the next highest risk (10^{-2} to 10^{-1}) being associated with the single sample of zinc spent surface impoundment liquids. Of the wastes whose CT cancer risks were below 10^{-5} for all samples, six (elemental phosphorous AFM rinsate, rare earths process wastewater, selenium plant wastewater, titanium/TiO_2 leach liquor and sponge wash water and scrap milling scrubber water, and zinc process wastewaters), had at least one sample with HE cancer risks above this level.

Cancer risks for most of the samples increased about two orders of magnitude from the CT to HE case. This is consistent with the difference between the CT and HE DAF values for arsenic managed in surface

EXHIBIT H-4

Distribution of Samples by Groundwater Risk Category: Cancer Risks

Commodity	Waste Stream	Number of Samples with Cancer Risk	Central Tendency						High End					
			<10-5	10-5 to 10-4	10-4 to 10-3	10-3 to 10-2	10-2 to 10-1	>10-1	<10-5	10-5 to 10-4	10-4 to 10-3	10-3 to 10-2	10-2 to 10-1	>10-1
Aluminum and Alumina	Cast house dust	2	2	0	0	0	0	0	2	0	0	0	0	0
Beryllium	Chip treatment WW	1	1	0	0	0	0	0	1	0	0	0	0	0
Copper	Acid plant blowdown	30	9	7	8	3	3	0	5	3	5	8	5	4
Elemental Phosphorus	AFM rinsate	2	2	0	0	0	0	0	0	1	1	0	0	0
Elemental Phosphorus	Furnace scrubber blowdown	8	7	1	0	0	0	0	3	3	2	0	0	0
Rare Earths	PWW	2	2	0	0	0	0	0	0	2	0	0	0	0
Selenium	Plant PWW	2	2	0	0	0	0	0	0	1	1	0	0	0
Tantalum, Columbium, and Ferrocolumbium.	PWW	13	10	2	1	0	0	0	7	3	0	3	0	0
Titanium and TiO_2	Leach liquor & sponge wash water	2	2	0	0	0	0	0	0	1	1	0	0	0
Titanium and TiO_2	Scrap milling scrubber water	1	1	0	0	0	0	0	0	1	0	0	0	0
Zinc	Waste ferrosilicon	0	0	0	0	0	0	0	0	0	0	0	0	0
Zinc	Spent s.i. liquids	1	0	0	1	0	0	0	0	0	0	0	1	0
Zinc	WWTP liquid effluent	0	0	0	0	0	0	0	0	0	0	0	0	0
Zinc	Process wastewater	11	11	0	0	0	0	0	7	1	3	0	0	0
Totals		75	49	10	10	3	3	0	25	16	13	11	6	4

impoundments. In the case of the NWW waste streams managed in piles, both the CT and HE cancer risks for all samples were below 10^{-5}. For aluminum/alumina cast house dust, this reflected the much higher CT and HE DAF values for arsenic managed in waste piles, compared to surface impoundments, as well on the relatively small mass of arsenic present in the waste pile. Arsenic was not detected in the single sample of waste ferrosilicon from zinc production. Thus, no carcinogenic risks were calculated for this waste. The two other streams for which all HE sample-specific cancer risks were below 10^{-5} were beryllium chip treatment wastewater and zinc wastewater treatment plant liquid effluent.

Noncancer hazard quotient values for groundwater pathway for the individual samples of recycled materials are summarized in Exhibit H-5. Using the CT DAF values, hazard quotient values exceeding 1.0 were calculated for 43 of 135 total samples from the 14 waste streams. As was the case for cancer risks, copper acid plant blowdown had the highest number of samples with noncancer hazard quotients above 1.0 (18 of 35 samples), and had the highest number of samples (4) in the highest risk category (HQ = 100 to 1000). Samples from zinc production (11 of 22 for spent surface impoundment liquids and 8 of 16 for process wastewater) account for the bulk of the remaining hazard quotients above 1.0. The only other waste streams with CT hazard quotients above 1.0 included beryllium chip treatment wastewater (one sample), elemental phosphorous furnace scrubber blowdown (one sample), tantalum, etc., process waste water (three samples), and zinc wastewater treatment plant liquid effluent (one sample).

When the HE DAF values are used to calculate exposures, hazard quotients exceed 1.0 for 100 of the 135 samples. As was the case for cancer risks, most of the hazard quotient values for individual samples are increased one to two orders of magnitude in the HE case compared to the CT case, reflecting the changes in the DAF values for the risk driving constituents managed in surface impoundments. As for cancer risks, both the CT and HE DAF values for waste piles for all of the constituents are so high, and the masses of constituents are so low, that no samples of the two waste streams managed in waste piles have hazard quotients exceeding 1.0 in either the CT or HE case. Hazard quotient values for five waste streams which were all below 1.0 in the CT case exceeded 1.0 in the HE case for at least one sample (elemental phosphorous AFM rinsate, rare earth process wastewater, selenium process wastewater, and titanium/TiO_2 leach liquor and sponge wash water and scrap milling scrubber sludge).

H.1.3.2 Risk Driving Constituents

For all of the cancer risk calculations, arsenic, being the only ingestion pathway carcinogen among the constituents evaluated, was always the risk driver. In the case of noncancer risks, many constituents drove risks (had the highest hazard quotients) for the samples evaluated. The noncancer risk driving constituents (constituents with the highest HE hazard quotients) for the various waste streams are identified in Exhibit H-6.

Overall, cadmium was the most common driving constituent, having the highest hazard quotient for one-half (50/100) of the samples with hazard quotients above 1.0. Arsenic and zinc (16 samples each) were the next most common drivers, followed by thallium (8 samples), and chromium (6 samples). None of the other constituents were noncancer risk drivers for more than one sample. Among the recycled streams with the highest numbers of samples, arsenic and cadmium were the predominant risk drivers for copper acid plant blowdown (26 out of 30 samples), cadmium was the dominant driver for elemental phosphorous furnace scrubber blowdown (8 of 10 samples), and cadmium and zinc were the predominant risk drivers for the three liquid recycled streams from zinc production.

EXHIBIT H-5

Distribution of Samples by Groundwater Hazard Category: Non-Cancer Hazards

Commodity	Waste Stream	Number of Samples with Non-cancer Hazard	Central Tendency						High End					
			<1	1 to 10	10 to 100	100 to 1k	1k to 10k	>10k	<1	1 to 10	10 to 100	100 to 1k	1k to 10k	>10k
Aluminum and Alumina	Cast house dust	2	2	0	0	0	0	0	2	0	0	0	0	0
Beryllium	Chip treatment WW	1	0	0	1	0	0	0	0	0	0	0	1	0
Copper	Acid plant blowdown	35	17	10	4	4	0	0	3	7	12	7	4	2
Elemental Phosphorus	AFM rinsate	2	2	0	0	0	0	0	0	0	2	0	0	0
Elemental Phosphorus	Furnace scrubber blowdown	14	13	1	0	0	0	0	4	4	5	1	0	0
Rare Earths	PWW	4	4	0	0	0	0	0	2	2	0	0	0	0
Selenium	Plant PWW	2	2	0	0	0	0	0	0	2	0	0	0	0
Tantalum, Columbium, and Ferrocolumbium	PWW	21	18	3	0	0	0	0	13	3	0	5	0	0
Titanium and TiO_2	Leach liquor & sponge wash water	2	2	0	0	0	0	0	0	1	1	0	0	0
Titanium and TiO_2	Scrap milling scrubber water	1	1	0	0	0	0	0	0	1	0	0	0	0
Zinc	Waste ferrosilicon	1	1	0	0	0	0	0	1	0	0	0	0	0
Zinc	Spent s.i. liquids	22	11	5	4	2	0	0	4	3	2	7	2	4
Zinc	WWTP liquid effluent	3	2	0	0	1	0	0	0	1	1	0	0	1
Zinc	Process wastewater	24	16	7	1	0	0	0	5	4	5	8	2	0
Totals		134	91	26	10	7	0	0	34	28	28	28	9	7

EXHIBIT H-6

Constituents Driving Non-Cancer Hazard Quotients in Recycled Streams

Commodity	Waste Stream	Driving Constituent (number of samples)
Aluminum and Alumina	Cast house dust	2 samples total; no hazard quotients greater than 1
Beryllium	Chip treatment WW	Beryllium (1/1)
Copper	Acid plant blowdown	Arsenic (15/35), Cadmium (11), Chromium (1), Lead (1), Selenium (1), Thallium (1), Zinc (2)
Elemental Phosphorus	AFM rinsate	Cadmium (2/2)
Elemental Phosphorus	Furnace scrubber blowdown	Cadmium (8/14), Chromium (1), Thallium (1)
Rare Earths	PWW	Thallium (2/4)
Selenium	Plant PWW	Arsenic (½), Thallium (1)
Tantalum, Columbium, and Ferrocolumbium	PWW	Antimony (1/21), Cadmium (3), Chromium (4)
Titanium and TiO_2	Leach liquor & sponge wash water	Thallium (2/2)
Titanium and TiO_2	Scrap milling scrubber water	Thallium (1/1)
Zinc	Waste ferrosilicon	1 sample total; no hazard quotients greater than 1
Zinc	Spent s.i. liquids	Cadmium (12/22), Zinc (6)
Zinc	WWTP liquid effluent	Cadmium (2/3), Zinc (1)
Zinc	Process wastewater	Cadmium (12/24*), Zinc (7)

* A sample with a selenium concentration of 100,000 ppm was excluded from the analysis

H.1.2.3 Risk Assessment Results by Facility

The cancer risk results for the individual samples, distributed across the numbers of facilities generating and storing the wastes, are summarized in Exhibit H-7. Using the methods described in Section H.1.1.2, it was estimated that CT groundwater pathway cancer risks would exceed 10^{-5} at 11 of the 57 waste stream-facility facilities[4]. All of these waste stream-facility combinations were managing either copper acid plant blowdown (7 facility-waste stream combinations) or zinc spent surface impoundment liquids (3 combinations). These results, of course generally reflect the pattern of sample-specific risk results for the various waste sectors. It will be noted, however, that for two waste streams, findings of one or more sample with greater than 10^{-5} risks did not translate into any facility-waste combinations above 10^{-5} risks. In the case of elemental phosphorous furnace scrubber blowdown, only one of seven samples had a cancer risk of just above 10^{-5}. Distributed across two facilities estimated to be storing this waste, this result (one-seventh of the samples having risks above 10^{-5}) was rounded down to zero. Similarly, in the case of tantalum, etc., process wastewater, three of thirteen samples with risks above 10^{-5} was again rounded downward to zero of two facility-waste stream combinations. This occurrence is the almost inevitable result of having so few facilities in so many industries, and the fact that non-integral numbers of waste-stream facility combinations are meaningless as risk or benefit indicators. It would be reasonable to interpret these results as indicating that either zero or one facility in these industries might have CT cancer risk above 10^{-5}.

When HE DAF values are used, the number of facility-waste stream combinations with cancer risks above 10^{-5} increases to 24 of 57 facilities. Under HE assumptions, most of the waste streams show one or more facilities at risk levels above 10^{-5}, the exceptions being the four low-risk waste streams identified in Exhibit H-4. These include both the two NWW streams that would be stored in waste piles, as well as beryllium chip treatment wastewater and zinc wastewater treatment plant liquid effluent. As noted previously, arsenic is not reported as a constituent of the latter waste.

The distribution of facility-waste stream combinations by noncancer risk category is summarized in Exhibit H-8. Using the CT DAF values, 12 waste stream-facility combinations are identified as having noncancer hazard quotients greater than 1.0. Five of these facilities are managing copper acid plant blowdown, two are managing beryllium chip treatment wastewater, and two of the facility-waste stream combinations are associated with the management of zinc spent surface impoundment liquids.

Using HE DAF values, 28 waste stream-facility combinations are identified as being associated with noncancer hazard quotients above 1.0. Again, four waste streams have no facility- waste stream combinations with hazard quotients above levels of concern: aluminum/alumina cast house dust, rare earth chip treatment wastewater, tantalum, etc., process wastewater, and zinc spent waste ferrosilicon.

[4] Note that the totals in the risk categories do not sum exactly due to rounding. This is true for the following exhibit as well.

EXHIBIT H-7

Distribution of Waste Stream-Facility Combinations by Groundwater Risk Category: Cancer Risks

Commodity	Waste Stream	Number of Waste Stream/ Facility Combinations* #		Central Tendency						High End					
		Central Tendency	High End	<10-5	10-5 to 10-4	10-4 to 10-3	10-3 to 10-2	10-2 to 10-1	>10-1	<10-5	10-5 to 10-4	10-4 to 10-3	10-3 to 10-2	10-2 to 10-1	>10-1
Aluminum and Alumina	Cast house dust	23	23	23	0	0	0	0	0	23	0	0	0	0	0
Beryllium	Chip treatment WW	2	2	2	0	0	0	0	0	2	0	0	0	0	0
Copper	Acid plant blowdown	10	10	3	2	3	1	1	0	2	1	2	2	2	2
Elemental Phosphorus	AFM rinsate	2	2	2	0	0	0	0	0	0	1	1	0	0	0
Elemental Phosphorus	Furnace scrubber blowdown	2	2	2	0	0	0	0	0	1	1	1	0	0	0
Rare Earths	PWW	1	1	1	0	0	0	0	0	0	1	0	0	0	0
Selenium	Plant PWW	2	2	2	0	0	0	0	0	0	1	1	0	0	0
Tantalum, Columbium, and Ferrocolumbium	PWW	2	2	2	0	0	0	0	0	1	1	0	0	0	0
Titanium and TiO_2	Leach liquor & sponge wash water	2	2	2	0	0	0	0	0	0	1	1	0	0	0
Titanium and TiO_2	Scrap milling scrubber water	1	1	1	0	0	0	0	0	0	1	0	0	0	0
Zinc	Waste ferrosilicon	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Zinc	Spent s.i. liquids	3	3	0	0	3	0	0	0	0	0	0	0	3	0
Zinc	WWTP liquid effluent	3	3	0	0	0	0	0	0	0	0	0	0	0	0
Zinc	Process wastewater	3	3	3	0	0	0	0	0	2	0	1	0	0	0
TOTALS*		57	57	42	3	6	1	1	0	30	8	6	3	5	2

* Sums by risk category may not add to the number of central or high-end waste stream/facility combinations due to rounding.

Includes waste stream/facility combinations with no cancer risk (but with an associated non-cancer hazard)

EXHIBIT H-8

Distribution of Waste Stream-Facility Combinations by Groundwater Hazard Category: Non-Cancer Hazards

Commodity	Waste Stream	Number of Waste Stream/ Facility Combinations*		Central Tendency						High End					
		Central Tendency	High End	<1	1 to 10	10 to 100	100 to 1k	1k to 10k	>10k	<1	1 to 10	10 to 100	100 to 1k	1k to 10k	>10k
Aluminum and Alumina	Cast house dust	23	23	23	0	0	0	0	0	23	0	0	0	0	0
Beryllium	Chip treatment WW	2	2	0	0	2	0	0	0	0	0	0	0	2	0
Copper	Acid plant blowdown	10	10	4	3	1	1	0	0	1	2	3	2	1	1
Elemental Phosphorus	AFM rinsate	2	2	2	0	0	0	0	0	0	0	2	0	0	0
Elemental Phosphorus	Furnace scrubber blowdown	2	2	2	0	0	0	0	0	1	1	1	0	0	0
Rare Earths	PWW	1	1	1	0	0	0	0	0	1	0	0	0	0	0
Selenium	Plant PWW	2	2	2	0	0	0	0	0	0	2	0	0	0	0
Tantalum, Columbium, and Ferrocolumbium	PWW	2	2	2	0	0	0	0	0	1	0	0	0	0	0
Titanium and TiO_2	Leach liquor & sponge wash water	2	2	2	0	0	0	0	0	0	1	1	0	0	0
Titanium and TiO_2	Scrap milling scrubber water	1	1	1	0	0	0	0	0	0	1	0	0	0	0
Zinc	Waste ferrosilicon	1	1	1	0	0	0	0	0	1	0	0	0	0	0
Zinc	Spent s.i. liquids	3	3	2	0	1	1	0	0	0	0	0	1	0	1
Zinc	WWTP liquid effluent	3	3	2	0	0	1	0	0	0	1	1	0	0	1
Zinc	Process wastewater	3	3	2	1	0	0	0	0	1	1	1	1	0	0
TOTALS*		57	57	45	5	4	3	0	0	29	9	9	4	4	2

* Sums by hazard category may not add to the number of central or high-end waste stream-facility combinations due to rounding.

H.1.4 Summary of Groundwater Pathway Risk Results

The preceding analysis indicates that the storage of some mineral processing recycled materials in land-based units under modified prior treatment baseline assumptions may be associated with significant health risks due to groundwater consumption. Cancer risks greater than 10^{-5} and hazard quotients greater than 1.0 are predicted for the minority of waste streams and individual samples under CT exposure assumptions and for the majority of waste streams and samples under HE exposure assumptions. Estimated cancer risks range up to 10^{-1} for some samples under CT exposure assumptions and exceed 10^{-1} under HE assumptions. Hazard quotient values similarly approach 1,000 under CT assumptions and exceed 10,000 for a few waste streams using the HE DAF values.

Copper acid plant blowdown has the largest number of samples with high cancer risks, and the highest cancer risks for this recycled stream exceed those for the next highest stream by one to two orders of magnitude. This stream also has the largest number of samples with hazard quotients above 1.0, followed by zinc spent surface impoundment liquids and process wastewater.

Aluminum/alumina cast house dust and zinc waste ferrosilicon are the only two waste streams for which no samples exceed 10^{-5} cancer risk or noncancer hazard quotient value of 1.0 under either CT or HE assumptions. These are the only two nonwastewater streams evaluated, and the low risk results are primarily a function of the very high DAF values for waste piles compared to the values derived for surface impoundments. Two other waste streams (beryllium chip treatment wastewater and zinc wastewater treatment plant liquid effluent) have low cancer risks even under HE assumptions, but one or more samples of each of these wastes is associated with hazard quotients greatly exceeding 1.0, even under CT assumptions.

Aluminum and alumina cast house dust (23 facilities) and copper acid plant blowdown (10 facilities) account for almost half the facilities evaluated in the analysis. As noted above, risks for the former stream are all low, so cast house dust has no waste stream-facility combinations above risk levels of concern. The majority of the waste stream-facility combinations managing copper acid plant blowdown, in contrast, are placed into risk categories above levels of concern under both CT and HE assumptions, and this waste stream contributes the largest number of waste stream-facility combinations at high risk levels of any waste stream.

On a volume basis, two streams (copper acid plant blowdown and zinc process wastewater) account for approximately 80 percent of the total recycled materials volume for which constituent concentration data are available. As noted above, copper acid plant blowdown is one of the highest-risk waste streams. While the cancer risks estimated for zinc process wastewater generally fall into the low-risk categories, the noncancer hazard indices associated with this waste stream are generally quite high, especially under HE assumptions.

H.1.5 Uncertainties/Limitation of Analysis

Most of the major sources of uncertainty for this risk assessment of storage of mineral processing recycled materials are the same as those for the previous analyses of mineral processing waste disposal. These uncertainties are discussed in detail in the cited references. To summarize briefly, the major uncertainties include:

- Limitations in data concerning the identities, amounts, constituent concentrations, and leaching behavior of the recycled materials.

- Limitations in data concerning the amounts of the specific recycled streams generated at specific facilities and the management methods used during storage.

- Limitation in knowledge concerning the locations, climatic, and hydrogeological settings at mineral processing facilities.

- Uncertainties and variability in the methods used to model leaching and groundwater transport (DAFs) of the toxic constituents of recycled materials.

- Uncertainties in the methods used to identify exposed receptors, estimate human exposures, and in characterizing the toxicological impacts of exposure to toxic constituents of recycled materials.

All of these sources of uncertainty (and variability) apply at least as much to the evaluation of storage risks as they did to the evaluation of risks from waste disposal. As noted above, the number of samples used to derive DAFs, and for estimating risks for the recycled materials, is quite limited, even more so than in the case of the waste disposal risk assessment. This is especially true for the nonwastewaters managed in waste piles, for which only three samples from two waste streams (all from unknown facilities) were available.

H.2 RISK ASSESSMENT METHODS AND RESULTS FOR NON-GROUNDWATER PATHWAYS

This section presents a summary of the risk assessment for the land storage of newly-identified mineral wastes under the non-groundwater modified prior treatment baseline.

H.2.1 Methods and Assumptions

H.2.1.1 Overview of Risk Assessment Methods

The multimedia risk assessment for the storage of mineral processing wastes employs many of the methods and assumptions used by EPA to develop the proposed risk-based exit levels for the Hazardous Waste Identification Rule (HWIR-Waste). The HWIR-Waste Technical Support Document[5] provides a detailed description of methods for evaluating releases, characterizing transport, and estimating exposures and risks associated with a number of non-hazardous waste management units. Individual algorithms and equations from HWIR-Waste are used to evaluate human exposures and risks associated with specific types of release events from land-based units (waste piles and landfills) that manage mineral processing recycled materials. In most cases, the HWIR-Waste methods are used without significant modification. However, in some instances, models were adjusted or simplified to reflect the specific characteristics of the facilities and constituents being modeled. For example, since none of the constituents addressed in this effort are appreciably volatile, the volatilization release and depletion equations from the HWIR-Waste models were not used and, since the recycling storage units were assumed to operate for only 20 years, the long-term steady-state assumptions employed in HWIR-Waste to estimate media concentrations were not valid, and time-dependent methods were substituted. Because of the shorter operating life spans of the storage units, compared to the assumptions made in HWIR-Waste, we also eliminated the soil depletion algorithms related to leaching and runoff. Thus, all soil contaminants were assumed to be fully conserved

[5] USEPA, *Technical Support Document for the Hazardous Waste Identification Rule: Risk Assessment for Human and Ecological Receptors*, Office of Solid Waste, August 1995.

for the entire exposure period. Finally, particulate release and transport models were used which differed slightly from those used in HWIR-Waste, and generic climatic assumptions were used in the evaluation of air transport. These methods are described in detail in Attachment H-B.

The same general assumptions regarding receptors and receptor behavior were employed in this analysis as were used in HWIR-Waste. With a few exceptions, the same values for exposure frequency and duration and other exposure factors are used as were employed in HWIR-Waste. Most of the exposure factors corresponded either to the adult resident, child resident, subsistence farmer, or subsistence fisher receptors defined in HWIR-Waste, depending on whichever had the highest exposures and risks. The only major exception was again related to the characteristics of the facilities being evaluated, in that release and exposure durations were adjusted to 20 years for "direct" pathways, corresponding to the assumed life-span of the management units. A full 30 years high-end exposure assumption is employed, however, for exposures to persistent constituents in soils.

Input data for the release models come from the database of waste constituent concentrations developed in support of the RIA (see Section H.1.1.3). In this case, however, only those streams are included which EPA has identified as having non-zero recycled volumes in the expected cost scenario of the modified prior treatment baseline. Facility characteristics and sizes from the least-cost management strategies developed in the RIA are used, as discussed in Section 3.0 of this RIA.

The exposure and risk assessment algorithms are applied in a screening mode to identify those management units, release events, and exposure pathways that may be associated with risks exceeding regulatory levels of concern. In the screening mode, relatively conservative assumptions regarding releases, exposures, and the toxicity characteristics of the waste constituents are used to provide a high degree of assurance that exposures that could be associated with significant risks are not missed. For most of the release events, high-end (HE) assumptions are first used to identify the highest risks pathways and constituents. If HE assumptions indicate that all risks are below levels of concern for a given pathway, no further risk assessment is performed. If HE risks are above levels of concern, central tendency (CT) assumptions are used to determine whether risks are still of concern for particular waste management units, waste streams, and constituents, and to help characterize the variability in risks that is associated with changes in key variables.

The risk assessment presented below summarizes risks for single-release events and exposure pathways. Risks are not summed across exposure pathways, unless it clear that exposure through one pathway would reasonably be associated with exposure through another pathway for the same receptor (risks from the ingestion of home-grown root and above ground vegetables are summed, for example). The risk assessment has not been structured to consider detailed mass balances across release events or exposure media, although each release event is evaluated to determine if it would result in a substantial reduction of the amount of constituent available for release by other events. As will be seen in Section H.2.2, no individual events were found that release substantial portions of the annual recycled volumes from any of the management units.

H.2.1.2 Regulatory Scenarios

As for the groundwater pathway, risks have been assessed for the modified prior treatment baseline. Under this baseline, it is assumed that recycled spent materials, sludges, and byproducts from mineral processing will be stored in land-based units prior to recycling. Nonwastewaters would be stored in unlined waste piles, and wastewater and liquid nonwastewater streams would be stored in unlined surface impoundments. Unlike the situation for disposal facilities, it is assumed that where two or more recycled streams are generated at a facility, the streams would be stored in separate units prior to recycling, and that

there would be no comanagement. Also, it has been assumed that the storage units would be sized to just accommodate the required amount of recycled material — three months' recycled volume in the case of nonwastewaters, and one month's recycled volume in the case of wastewaters and liquid nonwastewaters. The assumptions used to evaluate the size and configuration of storage facilities are described in detail in the August 1996 RIA.

H.2.1.3 Identification of Recycled Waste Streams

The same 14 waste streams were evaluated as in the groundwater pathway assessment. As noted previously, the 14 streams which are evaluated account for between approximately 32 and 42 percent of the total waste generated, and for between about 57 and 68 percent of the annual recycled volume, depending on which estimates are used, from the mineral processing industries that have been evaluated. The extent of coverage of waste streams from the individual industry sectors is summarized in Attachment H.A to this appendix.

H.2.1.4 Waste Characterization Data

The same data sources related to waste constituent concentration were used as described for the groundwater pathway assessment in Section H.1. No data were available related to the particulate characteristics of the two waste streams managed in waste piles. A reasonably conservative set of assumptions were therefore developed regarding waste silt content, particle size distribution, and particle size density, based partially on assumptions used in HWIR-Waste and on assumptions made by EPA as part of previous risk assessment efforts for similar mineral processing waste streams. These assumptions are described in more detail in Attachment H.B.

H.2.1.5 Facility Characterization Data

As noted above, facility size and configuration were determined for each recycled waste stream as part of the cost and economic impact analysis for the proposed Mineral Processing LDR. These methods are described in detail of Appendix E of the August 1996 RIA. Under the modified prior treatment scenario, it is assumed that all 14 recycled streams will be managed in unlined land-based units, nonwastewaters in waste piles and wastewaters and liquid nonwastewaters in surface impoundments. The management units were assumed to be sized to just meet the needs of the recycling units. Based on the Agency's evaluation of recycling practices, and considering the constraints on the duration of storage under existing regulations, it was assumed that all recycling storage piles would be sized to accommodate one quarter of the annual recycled volumes for typical facilities in the various commodity sectors, and that surface impoundments would be sized to accommodate one-twelfth of the annual recycled volumes of liquid streams. Thus, all units disposing of the same waste streams in any given commodity sector are assumed to be the same size. Further, it is assumed that no comanagement of multiple waste streams would occur in any management units.

For costing purposes, waste piles have been assumed to be conical, with side slopes of 2:1. Piles are assumed to be unlined and uncovered, with no special controls of runoff or particulate suspension. For purposes of emissions estimation, it is assumed that the piles are at full capacity at all times, and that the entire annual recycled volumes of the waste streams pass through the units each year, being added and removed at uniform rates on every day of operation throughout the year. It is assumed that, below a minimum recycled volume (500 mt/yr per facility) it is cheaper to store recycled materials in roll-off containers than in piles, and recycled streams with an annual recycled volume less than this amount were therefore not included in the risk assessment for waste piles. There was also an upper limit on the height

and area of a single pile, but none of the recycled nonwastewater streams were recycled in large enough volumes to reach this limit.

Surface impoundments used for recycling were assumed to be rectangular in shape, with a 2:1 length:width ratio and a rectangular prism-shaped bottom with a maximum depth of seven feet. Again, streams with annual recycled volumes of less than 500 mt/year per facility were assumed to be managed in tanks or containers, rather than impoundments. All of the recycled wastewater and liquid nonwastewater streams for which constituent data were available equaled or exceeded this volume, and thus all of them were included in the risk assessment.

The characteristics of the units used to store the recycled streams prior to recycling are summarized in Exhibit H-9. It can be seen that the two nonwastewater streams are both relatively low-volume, and the management units are correspondingly small. The sizes of the surface impoundments for the storage of liquid waste streams, on the other hand, span the range of the smallest possible facility size (42 cubic meters for titanium/$Ti0_2$ scrap milling scrubber water) to extremely large (99,167 cubic meters for zinc process wastewater).

H.2.1.6 Identification of Release Pathways

The screening-level risk assessment addressed non-groundwater release events from waste piles and surface impoundments managing mineral processing recycled streams. As an initial step in the risk assessment, release events and pathways were identified and screened to determine which would be the most likely to result in significant health risks to human receptors. The initial menu of events that were considered came from the HWIR-Waste Technical Background Document. The results of the screening are summarized in Exhibit H-10. As noted previously, many release events were screened out because of the characteristics of the units or the wastes involved. For example, volatilization release were eliminated for all management units and streams, because none of the toxic constituents, in the chemical forms that they are likely to be present, would be appreciably volatile.

The release events that have been addressed include the generation of air particulates and runoff from waste piles, and the releases of liquid recycled streams from surface impoundments due to inlet/outlet failures and runon events during large storms. Groundwater releases from these units have been addressed previously and are not further evaluated here.

H.2.1.7 Transport and Exposure Pathways

After releases from the land storage units, waste constituents may be transported or appear as contaminants in various environmental media, depending on the characteristics of the release event, the facility characteristics, and the environmental fate and transport properties of the constituents. In HWIR-waste, a large number of transport and exposure pathways were identified for the various units and waste/constituent types, only a minority of which were evaluated in this risk assessment. Reasons for excluding transport and exposure pathways from the assessment included (1) the pathways were not relevant to the units and waste being evaluated, (2) pathway models were not adequately developed or were too complex to apply in the context of this screening level assessment, or (3) it became apparent that the transport and exposure routes were very unlikely to be associated with significant risks. In some cases, simple screening-level models were substituted for the more detailed transport and exposure models from HWIR-Waste. Exhibit H-11 summarizes the fate and transport pathways that were evaluated in this assessment and provides a general description of the exposure and risk modeling procedures used to evaluate them.

EXHIBIT H-9. FACILITY SIZES FOR THE RECYCLED WASTE STREAMS

Commodity	Recycled Stream	Facility Type[1]	Facility Volume (m^3)	Facility Area (m^2)
Aluminum and Alumina	Cast House Dust	WP	107	108
Beryllium	Chip Treatment Wastewater	SI	417	558
Copper	Acid Plant Blowdown	SI	22,083	10,441
Elemental Phosphorous	AFM Rinsate	SI	167	415
Elemental Phosphorous	Furnace Scrubber Blowdown	SI	17,500	8,429
Rare Earths	Process Wastewater	SI	117	385
Selenium	Plant Process Wastewater	SI	550	631
Tantalum, Columbium, and Ferrocolumbium	Process Wastewater	SI	4,375	2,517
Titanium and Titanium Oxide	Leach Liquor and Sponge Wastewater	SI	4,000	2,341
Titanium and Titanium Oxide	Scrap Milling Scrubber Water	SI	42	340
Zinc	Waste Ferrosilicon	WP	1,093	509
Zinc	Spent Surface Impoundment Liquids	SI	10,500	5,319
Zinc	Waste Water Treatment Plant Liquid Effluent	SI	7,250	3,850
Zinc	Process Wastewater	SI	99,167	43,384

Notes: 1. SI = Surface Impoundment, WP = Waste Pile

EXHIBIT H-10. RELEASE EVENTS RETAINED IN THE MINERAL PROCESSING SCREENING RISK ASSESSMENT

Management Unit	Release Events
Waste Pile	Particulate Generation by Wind
	Particulate Generation by Materials Handling
	Surface Runoff due to Rain Events
Surface Impoundments	Releases Due to Inlet/Outlet Failures
	Releases Due to Runon Events

EXHIBIT H-11. EXPOSURE PATHWAY MODELING SUMMARY FOR MINERAL PROCESSING STORAGE RISK ASSESSMENT

Unit Type	Release Event/ Medium	Transport Medium I	Transport Medium II	Transport Medium III	Exposure Pathway	Receptors	Modeling Approaches
Waste Pile	Particulate Generation by Wind, Materials Handling	Air	--	--	Inhalation	Adult Resident	SCREEN3 (Emissions) ISCST3 (Deposition) HWIR (Exposure/Risk)
		Air	Soil (deposition)	--	Ingestion	Child/Adult Resident	HWIR-Waste (Exposure/Risk)
					Dermal	Child Resident	HWIR-Waste
		Air	Soil (deposition)	Crops	Ingestion	Subsistence Farmer	HWIR-Waste, modified for non-steady-state conditions (concentration in crops, vegetable intake, risk)
		Air	Soil/Water	Surface Water/Fish	Ingestion	Subsistence Fisher	Bounding analysis (100 percent deposition in water body)
Waste Pile	Runoff	Soil	--	--	Ingestion	Child Resident	Bounding analysis; 100 percent runoff to adjacent garden/yard, HWIR-Waste (exposure and risk)
					Dermal	Child Resident	Bounding analysis; 100 percent runoff to adjacent garden/yard, HWIR-Waste (exposure and risk)
		Soil	Crops	--	Ingestion	Subsistence Farmer	Bounding Analysis; HWIR-Waste
		Soil	--	Surface Water/Fish	Ingestion	Subsistence Fisher	Bounding analysis; 100 percent deposition to surface water; HWIR-Waste
Surface Impoundment	Control/ Berm Failure	Surface Water	--	--	Ingestion	Adult Resident	HWIR-Waste (Release algorithms, exposure, drinking water ingestion)
		Surface Water	Fish	--	Ingestion	Subsistence Fisher	HWIR-Waste (Releases, dilution, fish ingestion, risk)

Probably the most significant transport pathway that was omitted from the assessment was the discharge of groundwater to surface water. This pathway was not considered because of the absence of applicable DAF values and groundwater discharge volume estimates that would have allowed EPA to estimate surface water exposure concentrations.

H.2.1.8 Release, Transport, and Exposure Modeling

H.2.1.8.1 Air Particulate Generation, Meteorological Modeling, and Deposition on Soils

Airborne particulates generation from waste piles storing the two nonwastewater streams was estimated using EPA's SCREEN3 model. Long-term concentrations of particulates in air and long-term particulate deposition rates were calculated using the more detailed ISCST3 model. Because the locations of the facilities managing these streams are not known (all the analytical data come from facilities without identifiers), it was not possible to use site-specific meteorological or climatic data in the modeling of particulate generation and transport. Therefore, the models were run using a generic "worst-case" set of meteorological input data that is provided on part of ISCST3 for use in screening level analyses. High-End (HE) exposure concentrations (air and deposition values) were estimated for the point of maximal long-term impact (111 meters from the unit boundary in the case of aluminum cast house dust, and 248 meters in the case of zinc waste ferrosilicon), and the central tendency (CT) exposure estimates were deriving using the air concentrations and deposition rates averaged at every 100 meters from the unit boundary out to a distance of 2000 meters in the direction of maximal impact. The procedures and assumptions used in particulate generation and transport modeling are described in more detail in Attachment H-B.

As was the case for the meteorological data, very little information was available related to the physical characteristics (fraction of particulate present in waste, particle size distribution, particle density) of the nonwastewater streams. Data developed by EPA in previous analyses of potential risks from similar mineral processing wastes[6] known to be managed in piles were used in the absence of information specific to aluminum cast house dust and zinc waste ferrosilicon. These data are summarized in Exhibit H-11, along with the other parameter values used to estimate exposure concentrations in soil resulting from particulate deposition.

Accumulation of particulate materials in soils was assumed to occur for the entire 20-year lifespan of the waste piles. Exposure to the contaminated soil was assumed to begin at the end of the deposition period (when soil concentrations of deposited constituents would be the greatest), and it was assumed that the deposited constituents would not be depleted from the soil by leaching, runoff, or volatilization (ks = zero). This latter assumption adds a degree of conservatism to the estimation of soil concentrations, as some proportion of the deposited inorganics might, in the real world, be removed by runoff or leaching.

The soil concentrations from soil deposition were calculated using a variation of Equation 6-1 in the HWIR-Waste Technical Support Document. The equation was first rearranged to allow the calculation of soil concentrations from deposition rates, instead of vice-versa, and the exponential terms relating to the depletion of deposited material from the soil were eliminated from the equation. The result is a simple relationship describing the dilution of the deposited constituents uniformly in the mass of soil represented by the mixed layer. Consistent with HWIR-Waste, shallow mixing depths (1 and 2.5 cm) were used to calculate exposure concentrations for use in the soil ingestion and dermal contact pathways, and greater mixing depths (10 and 20 cm), corresponding to tilled depths, were used to calculate soil concentrations

[6] USEPA, *Risk Screening Analysis of Mining Wastes, Appendix F: Development of Particulate Emission Factors*, (Draft), Office of Solid Waste, October 25, 1987.

for the root vegetable consumption pathway. All other parameter values were the same as those to calculate soil concentration in the HWIR-Waste assessment.

H.2.1.8.2 Deposition of Airborne Particulates into Surface Water

The parameter values used in the estimation of particulate pathway emissions and transport modeling are summarized in Exhibit H-12. In HWIR-Waste, the relationship between airborne particulates and surface water contamination is modeled by a complex set of equations that simulate the both the direct deposition of particulates to surface water, and the deposition to soils onto a watershed, followed by overland transport to surface water bodies. Given the lack of knowledge about the locations of the storage piles relative to watersheds and surface water bodies, the relatively small size of the piles, and the relatively small mass of particulate that is generated, we have employed a much simpler screening approach to estimate the maximum long-term surface water concentrations that could result from the deposition of airborne particulates.

The methods simply assumes that, ultimately, all of the particulate emitted from the storage piles will end up in surface water. This is equivalent to making the conservative assumption that all of the particulates will either be directly deposited onto a surface water body, or that for that fraction of particulates that are initially deposited to soil, the sediment delivery ratio for the watershed will be equal to 1.0.

An additional simplifying assumption has been made regarding the behavior of deposited particulate in the surface water bodies, and regarding the speciation and solubility of particle-bound constituents. To estimate surface water concentrations, it is assumed that all of the constituents will be in the dissolved or suspended phase, and that none will remain bound to, or buried in, bottom sediment. This assumption probably overestimates the concentrations of some constituents in the water column, as some proportion of them would probably remain insoluble and bound to sediment.

Following the HWIR-Waste methodology, the airborne particulate matter is assumed to be deposited in either a “fifth order” or “third order” stream. These are streams or rivers of a given size and annual flow rate that have been selected (HWIR-Waste Technical Background Document Section 7.7.6.2) as the HE and CT surface water bodies, respectively. The long-term average concentration of constituents in surface water resulting from airborne particulate deposition is thus:

$$Csw\ (mg/l) = \frac{PG * Cwaste}{DV * 1000\ L/m^3} \qquad (1)$$

where PG is the annual particulate generation rate (in kg) from the waste pile, and DV is the surface water annual dilution volume as defined in Exhibit H-12. Since deposition is assumed to occur continuously throughout the year into a continuously flowing stream, there is no need to multiply by the 20-year facility life span.

EXHIBIT H-12. PARAMETER VALUES USED IN PARTICULATE PATHWAY EMISSIONS AND TRANSPORT MODELING

Variable	Description	CT Value	HE Value	Units	Source
SC	Silt Content (both streams)	1.6	9.1	percent	Footnote 5
--	Particulate Size Distribution	66 49 32 18	66 49 32 18	percent < 15 um percent < 10 um percent < 5 um percent < 2.5 um	Footnote 5
PD	Particle Density	2.65	2.65	gm/cc	Value for SiO_2 (sand)
Achd	Area of Waste Pile (Cast House Dust)	108	108	m^2	Waste data base, cost/economic impact methodology
Afesi	Area of Waste Pile (Ferrosilicon)	509	509	m^2	Waste data base, cost/economic impact methodology
Z	Soil Mixing Depth (Dermal and Ingestion Exposures)	2.5	1	cm	Typical values for untilled soils
Z	Soil Mixing Depth (Root Vegetable Ingestion)	20	10	cm	Typical tillage depths
BD	Soil Bulk Density	1.5	1.2	gm/cm^3	Typical for U.S. soils
ks	Soil Loss Constant	0	0	$years^{-1}$	Assumes no soil depletion of deposited materials
t	Deposition Period	20	20	years	Assumes unit lifespan of 20 years
DV	Surface Water Dilution Volume	$3.0X10^8$	$1.3X10^7$	m^3/year	Third- and Fifth-Order Stream Flow, respectively, HWIR-Waste Equation 7-69

H.2.1.8.3 Runoff to Surface Soils

The amount of waste released to surrounding soils from waste piles through runoff events was calculated using the Universal Soil Loss Equation (USLE), in a manner very similar to that described for waste piles in Section 7-4 of the HWIR-Waste Technical Support Document. As in the case of particulate releases, some of the assumptions and parameter values that were used were changed to reflect the characteristics of the units and wastes being addressed, and to address the specific geometry of the delivery of sediment to the surrounding soils.

The parameter values used in the estimation of releases to soils from the waste piles and the resulting concentrations of constituents in soils and surface water are summarized in Exhibit H-13. In calculating runoff releases, in the absence of data related to the specific wastes and pile configurations being evaluated, we used the same values for soil erodability (k) and length-slope factors (LS) as were used for Subtitle D waste piles in HWIR-Wastes. The rainfall factor values (R) were changed slightly, however. The CT value used in the analysis was selected from the data in Table 7-42 of the HWIR-Waste Technical Support Document to reflect rainfall frequencies in the western US (where the majority of mineral processing waste, by volume, is managed), while the HE value was selected to be more representative of nationally-averaged conditions. In this analysis, the values for the USLE cover factor © and control practices factor were increased to 1.0 in both the CT and HE cases. These values reflect the likelihood that an active storage pile would not have any vegetative cover, and the conservative assumption (consistent with the cost and economic analysis) that there would be no special precautions taken to prevent runoff losses.

A very simple sediment delivery model was used to estimate the concentrations of waste constituents in soils resulting from runoff. Currently, the sediment delivery model in the HWIR-Waste modeling system is under review, and final decisions about the configurations of waste management units, buffer zones, and receiving areas have not been made. In the absence of a definitive model, soil concentrations were simply calculated by assuming that the conical waste piles would generate circular “plumes” of runoff that would deposit evenly within defined distances from the center of the piles. For HE exposure estimates, the area of soil contaminated by runoff was assumed to be 100 meters in diameter, while for CT exposures, the area of contaminated soil was assumed to be 200 meters in diameter, or four times larger. This approach assumes that a storage pile would be located near the edge of a facility, that exposed receptors would reside directly adjacent to the facility boundary, and that there would be no preferred runoff path or deposition areas. As was the case for air particulate deposition, it was again assumed that deposition would occur for 20 years, and that the deposited constituents would not be further depleted by runoff or leaching after initial deposition.

The soil concentration resulting from surface runoff from waste piles was thus calculated using the following equation:

$$\text{Csoil (mg/kg)} = \frac{\text{Xe (kg/m}^2\text{-year)} * \text{Achd (or Afesi)(m}^2\text{)} * \text{Cwaste (mg/kg)} * \text{t (years)}}{\text{BD (gm cm}^3\text{)} * \text{Z (cm)} * \text{X} * \text{r}^2\text{ (cm}^2\text{)} * 0.001\text{ kg/gm}} \qquad (2)$$

where the variable definitions and values are given in Exhibit H-13.

EXHIBIT H-13
PARAMETER VALUES USED IN RUNOFF RELEASE AND TRANSPORT MODELING FROM WASTE PILES

Variable	Description	CT Value	HE Value	Units	Source
Xe	Runoff loss from waste pile	calculated	calculated	kg/m^2-year	HWIR-Waste equation 7-52
Achd	Area of Waste Pile (Cast House Dust)	108	108	m^2	Waste data base, cost/economic impact methodology
Afesi	Area of Waste Pile (Ferrosilicon)	509	509	m^2	Waste data base, cost/economic impact methodology
R	USLE Rainfall Factor	50	110	$years^{-1}$	CT= Typical of western US HE = US Median value
LS	Length-Slope Factor	1	3		HWIR-Waste value for Subtitle D ash piles
K	Soil Erodability Factor	0.25	0.25	unitless	HWIR-Waste value for Subtitle D ash piles
C	Cover Factor	1	1	unitless	Assumes no vegetative cover on waste piles
P	Control Practices Factor	1	1	unitless	Assumes no measures to control runoff
r	Radius of area contaminated by runoff	5,000	10,000	cm	Contamination is assumed to be distributed uniformly in a circular area around the conical piles
DV	Surface Water Dilution Volume	$3.0X10^8$	$1.3X10^7$	m^3/year	Third- and Fifth-Order Stream Flow, respectively, HWIR-Waste Equation 7-69

This approach to estimating soil concentrations from waste pile runoff greatly simplifies the potentially complex processes that would, in the real world, govern the generation and distribution of runoff contamination. It is intended only as a conservative screening tool to provide indications of the relative risks associated with the various waste and constituents, and to provide a high degree of assurance in ruling out wastes and constituents that pose no significant risks through this pathway.

H.2.1.8.4 Runoff To Surface Water

The deposition of runoff to surface water bodies was evaluated using a screening approach analogous to that used to evaluate the impacts of airborne particulate deposition on surface water quality. Again, it was assumed that 100 percent of the runoff-borne constituents would eventually find their way into the CT or HE streams. Thus, the equation used to estimate the concentration of runoff-borne constituents in surface water during the operation of the storage piles is:

$$\text{Csw (mg/l)} = \frac{\text{Xe (kg/m}^2\text{-year) * Achd (or Afesi)(m}^2\text{)* Cwaste (mg/kg)}}{\text{DV (m}^3\text{/year) * 1000 L/m}^3} \qquad (3)$$

The annual average runoff from the piles is again released to surface water and diluted in the CT or HE stream dilution volume (DV, see Exhibit H-13) to provide a long-term average water concentration. Again, it is assumed that none of the runoff materials would become buried in bottom sediment.

H.2.1.8.5 Surface Impoundment Releases to Surface Water

To evaluate surface water concentrations associated with releases from surface impoundments, we used precisely the same method as used in HWIR-Waste (Equation 7-70). Again, the release model has been simplified by removal of all of the equations related to volatilization.

The equation from HWIR-Waste estimates releases to surface water from runon events (overtopping due to unusually high rainfall) and from inlet-outlet control failures. It does not include releases due to berm failure or leakage. The model is probabilistic, estimating long-term average releases of impoundment contents as a function of annual event probabilities. As was the case for air deposition and runoff from waste piles, the average surface water concentrations during facility operations are calculated assuming that the annual waste releases due to the two types of events (summed) are diluted into the annual flow of the CT and HE streams, without partitioning to sediment. The major variables used to estimate surface water concentrations of constituents from impoundment failure are summarized in Exhibit H-14.

The uncertainty associated with release and exposure estimates from this pathway must be regarded as very high. The model was originally intended to estimate long-term average releases from rare acute events occurring over the course of very many years. Thus, it may not be appropriate for estimating releases from rather short-lived storage impoundments (20 years) being evaluated in this assessment. In addition, the model does not capture the effects of the single acute releases on water quality in the short-term. Finally, the model and the parameter values used to estimate releases were originally derived by EPA based on data from a sample of surface impoundments in the pulp and paper industry.[7] It is likely

[7] DPRA, *Surface Water Control Berms for Pulp and Paper Mill Sludge Landfills and Surface Impoundments*, Memo to Priscilla Halloran, OSW, July 18 1991.

EXHIBIT H-14. PARAMETERS USED IN THE ESTIMATION OF SURFACE IMPOUNDMENT RELEASES TO SURFACE WATER AND SOILS

Variable	Description	CT Value	HE Value	Units	Source
RV	Release volume (annual average)	Calculated	Calculated	m^3/year	HWIR-Waste, equation 7-70
Prunon	Probability of runon event	$2X10^{-4}$	$2X10^{-4}$	$years^{-1}$	DPRA, 1991 (see text)
Tflood	Duration of Flooding	21,600	21,600	seconds	DPRA, 1991
Vrunon	Runon velocity	0.5	0.5	m/sec	DPRA, 1991
-h	Difference in height between flood and berm	0.0127	0.0127	m	DPRA, 1991
A	Area of surface impoundment	Waste-specific	Waste-specific	m^2	Mineral Processing data base, cost/economic analysis
Pio	Probability of inlet/outlet control event	0.0107	0.0107	$years^{-1}$	DPRA, 1991
h	Berm height	0.457	0.457	m	DPRA, 1991

that the designs, sizes and operating parameters for impoundments in the mineral processing industry are substantially different, and the expected releases could also be different. One feature of the model that tends to result in conservatism in the exposure estimates from this pathway is that no dilution of recycled materials by runon events is assumed. In an actual extreme runon event, dilution of the wastes could be substantial, lowering the concentration of released materials.

H.2.1.9 Exposure and Risk Characterization

H.2.1.9.1 Toxicological Criteria

With a single exception, quantitative risk estimates have been developed using toxicity criteria values obtained from USEPA's IRIS data base or the HEAST tables and updates. To calculate inhalation pathway cancer risks and noncancer hazard quotients, inhalation Unit Risk and chronic inhalation pathway Reference Concentration (RfC) values are used. For the ingestion pathways, Cancer Slope Factors (CSFs) and chronic Reference Doses (RfDs) are used. The IRIS values are current as of December 1996. These values are summarized in Exhibit H-15.

EXHIBIT H-15. TOXICITY CRITERIA VALUES USED IN THE MINERAL PROCESSING STORAGE RISK ASSESSMENT

Constituent	Ingestion Pathway Cancer Slope Factor $(mg\text{-}kg\text{-}day)^{-1}$	Inhalation Pathway Unit Risk $(ug/m^3)^{-1}$	Chronic Ingestion Pathway Reference Dose (mg/kg-day)	Chronic Inhalation Pathway Reference Concentration (mg/m^3)
Antimony			$4X10^{-4}$	
Arsenic	1.5	$4.3x10^{-3}$		
Barium			$7X10^{-2}$	$5X10^{-4}$
Beryllium	4.3 [1]	$2.4X10^{-3}$	$5X10^{-3}$	
Cadmium		$1.8X10^{-3}$	$5X10^{-4}$	
Chromium (VI)		$1.2X10^{-2}$	$5X10^{-3}$	
Lead			0.015 mg/L [2]	
Mercury			$3X10^{-4}$	$3X10^{-4}$
Nickel		$4.8X10^{-4}$	$2X10^{-2}$	
Selenium			$5X10^{-3}$	
Silver			5X10-3	
Thallium			8X10-5	
Vanadium			7X10-3	
Zinc			3X10-1	

Notes:

1. Not used in risk assessment because of low weight of evidence
2. Based on the Safe Drinking Water Act MCL for inorganic lead.

Ingestion pathway RfD values are available for all of the constituents except lead (see below). Arsenic is the only constituent that is considered to be an ingestion pathway carcinogen in this assessment, so it is the only constituent with an ingestion pathway CSF. Inhalation pathway RfCs were available for only two of the constituents (barium and mercury), so inhalation pathway hazard quotients could be calculated only for these elements. Inhalation cancer Unit Risk values are available for five constituents considered to be inhalation pathway carcinogens, however.

Inorganic lead was the only constituent for which a different approach to risk characterization was employed. Since there is no RfD or RfC value for lead, the toxicity criterion that was used to evaluate potential noncancer risks associated with lead exposure was the Clean Water Act MCL of 15 ug/L. This value was used to evaluate concentrations in surface water arising from particulate deposition and runoff, assuming, in effect, that the water body would be used as a drinking water supply. Risks associated with lead exposure through other pathways were not evaluated because of the lack of acceptable toxicity criteria for these pathways.

For purposes of the assessment, it was assumed that all of the chromium present in the stored waste streams would be in the more toxic hexavalent form. This assumption will overstate risks when (as in most cases) the bulk of the chromium is in lower oxidation states.

H.2.1.9.2 Inhalation

Risks associated with inhalation pathway exposure to particulates released from waste piles are calculated directly from the estimated particulate concentrations in air generated by the ISCST3 model. Lifetime cancer risks associated with exposure to airborne particulates are calculated as:

$$\text{Risk} = C_{part}\ (ug/m^3) * C_{waste}\ (mg/kg) * 10^{-6}\ kg/mg * UR\ (ug/m^3)^{-1} \qquad (4)$$

where C_{part} is the particulate concentration from the ISCST model, C_{waste} is the concentration of arsenic in the waste sample, and UR is the unit risk value constituent.

Inhalation noncancer hazard quotients are calculated as:

$$HQ = \frac{C_{part}\ (ug/m^3) * C_{waste}\ (mg/kg) * 10^{-6}\ kg/mg * 10^{-3}\ mg/ug}{RfC\ \ (mg/m^3)} \qquad (5)$$

In both cases, the receptor is an adult resident, residing at either the point of maximum long-term air concentration (HE estimate), or at the point of average concentration within 2000 m of the facility (CT estimate). For screening purposes, exposure is assumed to be continuous for 365 days per year, and for carcinogenic constituents, the exposure duration is assumed to be the 20-year operating lifespan of the facility. As will be seen in Section H.2.1, in both the CT and HE cases, cancer risks were all below 10^{-5} and inhalation hazard quotients were all below 1.0 under these very conservative screening assumptions, so more refined modeling scenarios were not developed for this pathway.

H.2.1.9.3 Soil Ingestion and Dermal Contact

Cancer and chronic noncancer risks were evaluated for dermal and incidental ingestion exposures to soil contaminated by particulate deposition (Section H.2.1.9.3) and by deposition of surface runoff (Section H.2.1.9.4). For each pathway, the soil concentrations after 20 years of deposition were used as inputs to the risk assessment, assuming no depletion of deposited materials from soils by volatilization, leaching, or runoff. For both the dermal and ingestion pathways, the shallower soil mixing depths (1.0 and 2.5 cm) were used to estimate soil concentrations of constituents consistent with the assumption of no tillage or soil disturbance.

Risks associated with soil ingestion were calculated using Equation 5-6 from the HWIR-Waste Technical Support Document, adapted to calculate risk as a function of concentration, instead of vice versa, and with the soil constituent depletion terms removed. Cancer risks were calculated for lifetime exposures to a child/adult resident, consistent with the HWIR-Waste approach, and noncancer hazard quotients were calculated for the child resident receptors, who receive the highest dose per body weight by this pathway. The exposure parameter values used to calculate contaminant intake and risks from soil ingestion are summarized in Exhibit H-16.

For the most part, these are standard values used in Agency rulemaking and risk assessments for contaminated sites. Differences from the HWIR-Waste assumptions include more frequent exposures (350 days/year) for adults and children, and a slightly shorter HE exposure duration (30 years, as opposed to 40 years in HWIR-Waste).

EXHIBIT H-16. EXPOSURE FACTOR VALUES FOR SOIL INGESTION AND DERMAL CONTACT PATHWAYS

Variable	Description	CT Value	HE Value	Units	Source
AT	Averaging time (carcinogens)	70	70	years	Assumed full life span
EF	Exposure frequency	350	350	days/year	Worst-case assumption of year-round residency
IRc	Soil ingestion rate (child)	200	200	mg/day	HWIR-Waste Equation 5-6
IRa	Soil Ingestion rate (adult)	100	100	mg/day	HWIR-Waste Equation 5-6
BWc	Body weight (child)	15	15	kg	HWIR-Waste Equation 5-6
BWa	Body weight (adult)	70	70	kg	HWIR-Waste Equation 5-6
EDc	Exposure duration (child)	6	6	years	HWIR-Waste Equation 5-6
EDa	Exposure duration (adult)	3	24	years	Assumes 30 years' total residential tenure, six as a child, remainder as an adult
Kpw	Skin permeability constant for water	0.001	0.001	cm/hr	HWIR-Waste Equation 5-14
Y	Soil particle density	2.65	2.65	gm/cc	HWIR-Waste Equation 5-14
AF	Adherence factor	0.2	1.0	gm/cm^2	HWIR-Waste Equation 5-23
Tevent	Event Duration	5	12	hours	HWIR-Waste Equation 5-23

Risks from dermal exposures to contaminated soils were likewise calculated using equations based on the HWIR-Waste methodology. Specifically, equations 5-14 and 5-20 through 5-23 (adjusted as for the ingestion pathway) were used to calculate dermal contact rates with soil, dermal permeability constants, dermal absorbed doses, and risks from dermal exposures.

The soil concentration inputs were again the concentrations resulting from 20 years of contamination by runoff or air particulate deposition. As was the case for soil ingestion, exposure factor values were essentially the same as those used in the HWIR-Waste methodology. These values are summarized in the bottom rows of Exhibit H-16. All of the values for body weights, exposure duration, and exposure frequency are the same as those used for the ingestion pathway.

H.2.1.9.3 Ingestion of Home-Grown Vegetable

Crops grown near waste piles may become contaminated either from being grown in contaminated soil or from the deposition of particulates directly on the above-ground portions of the vegetables. In this analysis, risks were calculated for vegetable consumption by a subsistence farmer on soils contaminated either by particulate deposition or runoff. In each case, the methods used to calculate the intake of toxic constituents and risks were the same as those used in Equations 5-58, 5-59, 6-48, and 6-49 in the HWIR-Waste Technical Support Document.

Soil concentrations used to calculate root vegetable constituent concentrations were calculated as described previously. In the case of the root vegetable pathway, the soil mixing depths were either 20 cm (CT) or 10 cm (HE) instead of the shallower values used for the ingestion and dermal contact pathways. The exposure factor values used to estimate intake and risks for this pathway are summarized in Exhibit H-17. These values are essentially the same as those used on HWIR-Waste, the primary exception being the use of an HE exposure duration of 20 years, corresponding to the assumed life of the storage units, rather than the 40-year value used in HWIR-Waste.

H.2.1.9.4 Ingestion of Surface Water

Releases to surface water from surface impoundment failures and runoff from waste piles have been modeled. For both types of releases, the methods used to estimates constituent intakes and health risk are the same, and consistent with that used in the HWIR-Waste methodology.

As described previously, releases to surface water are assumed to be diluted into either a typical third-order (CT) or fifth-order (HE) stream. In this analysis, it is assumed that the surface water body in question would be used as a drinking water source, without further treatment to reduce exposure concentrations. Adult residents would then ingest either 1.4 liters (CT) or 2.0 liters (HE) of surface water for 350 days per year for 20 years. Lifetime doses of carcinogens are calculated based on an assumed lifespan (averaging time) of 70 years as for the other pathways, with residential exposure durations of either 9 (CT) or 30 (HE) years. Cancer risks are calculated as follows:

$$\text{Risk} = \frac{C_{water}\ (\text{mg/l}) * \text{WI (l/day)} * \text{EF (days/year)} * \text{ED (years)} * \text{CSF (mg/kg-day)}^{-1}}{\text{BW (kg)} * \text{AT (years)} * 365\ (\text{days/year})} \qquad (6)$$

where WI is the daily water intake, in liters. Noncancer hazard quotients are calculated as:

$$\text{Hazard Quotient} = \frac{C_{water}\ (\text{mg/l}) * \text{WI (l/day)} * \text{EF (days/year)}}{\text{BW (kg)} * 365\ (\text{days/year}) * \text{RfD (mg/kg-day)}} \qquad (7)$$

In the actual risk assessment, these equations were used, similar to the approach taken in HWIR-Waste, to calculate water concentrations of the constituents that would result in lifetime cancer risks of 10^{-5} or hazard quotients of 1.0 under CT and HE assumptions. These health-based levels (HBLs) were then used as a screening tool to determine which, if any, waste samples or constituents exceeded cancer risks or hazard quotient values of concern under either the CT or HE assumptions, so that more detailed analysis could be confined only to those wastes posing significant risks.

EXHIBIT H-17. EXPOSURE FACTOR VALUES USED FOR CROP INGESTION PATHWAY

Variable	Description	CT Value	HE Value	Units	Source
t	Deposition period	20	20	years	20 years = facility life span
kd	Soil-Water Dissociation Constant	constituent-specific	constituent-specific	l/kg	HWIR-Waste data base
RCF	Root Concentration Factor	constituent-specific	constituent-specific	mg/kg (veg.) mg/kg (soil)	HWIR-Waste data base
Vg	Surface correction factor for volatiles	1	1	unitless	All constituents are inorganic
Br	Plant-Soil BCF	constituent-specific	constituent-specific	ug/kg (veg.) ug/kg (soil)	HWIR-Waste data base
Rp	Interception fraction	0.05	0.05	unitless	HWIR-Waste Equation 6-48
kp	Plant surface loss coefficient	18	18	$years^{-1}$	HWIR-Waste Equation 6-48
tp	Plant exposure to deposition	0.16	0.16	years	HWIR-Waste Equation 6-48
Yp	Crop yield	1.7	1.7	kg/m^2 (D^W)	HWIR-Waste Equation 6-48
BWa	Adult Body Weight	70	70	kg	Standard Assumption
F	Fraction from contaminated soil	0.4	0.9	unitless	HWIR-Waste Subsistence Farmer
Cra	Consumption of above-ground vegetables	19.7	19.7	gm/day	HWIR-Waste Subsistence Farmer
Crr	Consumption of root vegetables	28	28	gm/day	HWIR-Waste Subsistence Farmer
EF	Exposure Frequency	350	350	days/year	HWIR-Waste Subsistence Farmer
ED	Exposure duration	9	20	years	HWIR-Waste (CT), = deposition period (HE)
AT	Averaging time	70	70	years	full life span

H.2.1.9.5 Ingestion of Fish from Contaminated Surface Water Bodies

In addition to being screened for potential risks associated with ingestion, the estimated surface water concentrations resulting from air particulate deposition, runoff, and surface impoundment failure were also screened to determine the potential risks associated with ingestion of fish from the contaminated surface water bodies. The primary inputs to this analysis were the surface water concentrations resulting from the

various release pathways. The concentrations of toxic constituents in fish tissue were calculated as follows:

$$C_{fish}\ (mg/kg) = C_{water}\ (mg/l) * \max(BCF, BAF)\ (l/kg)$$

To calculate fish tissue concentrations, the estimated surface water concentrations were multiplied by the higher of either the fish bioconcentration factor (BCF) or fish bioaccumulation factor (BAF) values for the constituents. The primary source of these values was the chemical-specific data base from HWIR-Waste, but the values from that source were supplemented by values from other literature sources, as summarized in Attachment H-C. Where both a BCF and a BAF value were available for a constituent, the higher of the two values was chosen. Where multiple BCF values were found, we generally took what we considered to be the highest reliable value from either HWIR or the literature. Since the values in HWIR were intended to be representative, rather than conservative, this procedure resulted in our using higher BCF values for a number of constituents than were used in HWIR-Waste risk calculations, and the resultant hazard-based levels (HBLs) for this pathway were thus lower than those derived in HWIR-Waste for some constituents.

Constituent intakes and risks from fish ingestion were calculated using equations 5-67 and 5-68 from HWIR-Waste. Consistent with the HWIR-Waste approach, health-based levels (HBLs) were calculated for surface water exposures through fish ingestion for the adult subsistence fisher who is assumed to consume 60 gms (CT) or 130 gms (HE) of fish per day for 350 days per year, using a target cancer risk level of 10^{-5} and a target hazard quotient value of 1.0. These HBLs were then used to screen the surface water concentrations resulting from air deposition, runoff, and surface impoundment failures.

H.2.2 Results of Multipathway Risk Assessment

This section presents the results of the multimedia risk assessment for the storage of mineral processing recycled streams. It begins with a review of the release modeling from the point of view of mass balance considerations, and then presents discussions of the risk results for each of the release events, exposure media, and pathways.

H.2.2.1 Mass Balance for Release Pathways

As noted above, the risks associated with releases from for mineral processing facilities presented in this analysis have been evaluated separately. In other words, it has been assumed that releases occur independently of one another, and that all of the materials in the storage units are available for release by all release pathways. In this section, we review whether this assumption is valid by comparing the amounts of materials released from the storage units by the different release events.

The masses of recycled materials released from the various storage units are summarized in Exhibit H-18. It can be seen that only a very small proportion of the total annual recycled volume of all of the waste streams are released from waste piles. In the case of the two nonwastewater streams managed in piles, the annual release volumes from the two types of release events (particulate generation and runoff) are both far below one percent of the total annual recycled volume. Thus, depletion of material by these pathways will not seriously affect the total mass of material remaining in the piles, and thus the release estimates for runoff and particulate generation do not bias each other significantly. Similarly, releases from these pathways do not deplete the amount of materials available for leaching to groundwater.

EXHIBIT H-18
MASSES AND PROPORTIONS OF RECYCLED STREAMS RELEASED BY SPECIFIC RELEASE EVENTS

Commodity	Recycled Stream	Management Unit	Annual Recycled Volume (kg/year)	Release Event	HE Amount Released (kg/year)	Proportion of Annual Volume
Aluminum and Alumina	Cast House Dust	Waste Pile	581,000	Air Particulate Generation	324	0.06%
				Runoff	5,416	0.93%
Zinc	Waste Ferrosilicon	Waste Pile	5,950,000	Air Particulate Generation	1,520	0.03%
				Runoff	25,527	0.43%
Beryllium	Chip Treatment Wastewater	Surface Impoundment	2,500,000	Runon, Inlet/Outlet Control Failure	2,012	0.08%
Copper	Acid Plant Blowdown	Surface Impoundment	265,000,000	Runon, Inlet/Outlet Control Failure	28,330	0.01%
Elemental Phosphorous	AFM Rinsate	Surface Impoundment	2,000,000	Runon, Inlet/Outlet Control Failure	1,573	0.08%
Elemental Phosphorous	Furnace Scrubber Blowdown	Surface Impoundment	210,000,000	Runon, Inlet/Outlet Control Failure	23,127	0.01%
Rare Earths	Process Wastewater	Surface Impoundment	700,000	Runon, Inlet/Outlet Control Failure	1,480	0.21%
Selenium	Plant Process Wastewater	Surface Impoundment	3,300,000	Runon, Inlet/Outlet Control Failure	2,232	0.07%
Tantalum, Columbium, Ferrocolumbium	Process Wastewater	Surface Impoundment	37,500,000	Runon, Inlet/Outlet Control Failure	7,530	0.02%
Titanium, TiO_2	Leach Liquor, Sponge Wash Water	Surface Impoundment	24,000,000	Runon, Inlet/Outlet Control Failure	7,050	0.03%
Titanium, TiO_2	Scrap Milling Scrubber Water	Surface Impoundment	5,00,000	Runon, Inlet/Outlet Control Failure	1,337	0.27%
Zinc	Spent Surface Impoundment Liquids	Surface Impoundment	63,000,000	Runon, Inlet/Outlet Control Failure	15,005	0.02%
Zinc	Wastewater Treatment Plant Liquid Effluents	Surface Impoundment	43,500,000	Runon, Inlet/Outlet Control Failure	11,115	0.03%
Zinc	Process Wastewater	Surface Impoundment	850,000,000	Runon, Inlet/Outlet Control Failure	111,784	0.01%

Likewise, the estimated annual releases from surface impoundments also represent very small proportions of the total impoundment capacities and the annual recycled volumes. Thus, releases due to these events will not in the long run seriously deplete the amount of materials available for release by other pathways. For surface impoundments, the only other significant release pathway is infiltration to groundwater, since particulate generation and runoff are not important. The issue of whether leaching to groundwater might reduce the concentration of some constituents in the storage units has not been specifically addressed. EPA does not believe that, given the short operation life of these units and their continual replenishment with recycled materials, leaching to groundwater would seriously deplete any of the constituents.

H.2.2.2 Risk Results for Inhalation of Particulate

The estimated health risks associated with inhalation of particulates released from storage waste piles are quite low, as summarized in Exhibit H-19. Because of the lack of inhalation toxicity criteria, cancer risks could only be calculated for four constituents, and noncancer hazard quotients could be calculated for only two constituents. Since no inhalation toxicity criteria were available for the only two constituents analyzed for in zinc waste ferrosilicon (lead and zinc), no inhalation pathway risks could be calculated for that waste.

Exhibit H-19

Estimated Inhalation Pathway Risks for Aluminum Cast House Dust

Constituent	CT Constituent Concentration in Air (ug/m3)	HE Constituent Concentration in Air (ug/m3)	CANCER RISK		HAZARD QUOTIENT	
			CT	HE	CT	HE
Antimony	1.73E-05	2.42E-04				
Arsenic	7.36E-05	1.03E-03	3.90E-13	1.22E-11		
Barium	2.30E-05	3.23E-04			1.92E-01	1.92E-01
Cadmium	1.66E-05	2.33E-04	3.67E-14	1.15E-12		
Chromium(VI)	2.53E-04	3.55E-03	3.74E-12	1.17E-10		
Lead	3.91E-05	5.49E-04				
Mercury	2.30E-10	3.23E-09			3.20E-06	3.20E-06
Nickel	5.98E-04	8.40E-03	3.54E-13	1.10E-11		
Selenium	2.12E-06	2.97E-05				
Silver	4.37E-06	6.14E-05				
Zinc	2.76E-04	3.88E-03				

In the case of aluminum cast house dust, the highest cancer risks were associated with exposures to chromium (VI), followed by arsenic, nickel, and cadmium exposures. The HE cancer risk estimates for these constituents ranged from 10^{-12} to 10^{-10}, far below the 10^{-5} cancer risk level of regulatory concern, and the CT risks were even lower. As noted previously, the assumption that all of the chromium present would by hexavalent is very conservative, and risks for chromium exposures are likely to substantially overestimated for this reason.

The estimated inhalation hazard quotient values for aluminum cast house dust are also below levels that indicate the potential for significant adverse effects. The highest HE hazard quotient value (for barium) is 0.2, while for mercury the HE hazard quotient is less than 10^{-5}. Both of these values are below the 1.0 value, which indicates the potential for adverse effects, although the HE hazard quotient for barium approaches the level of concern.

H.2.2.3 Risk Results for Soil Particulate Deposition

Particulate matter generated from waste piles may also be deposited onto soils and crops, resulting in direct exposure to contaminated soils and through the consumption of home-grown vegetables. In addition, impacts of particulate deposition to surface water have also been modeled. The risk results for these pathways are discussed in the following sections.

H.2.2.3.1 Incidental Ingestion and Dermal Contact Pathways

Risk results for the incidental ingestion and dermal contact pathways for soils contaminated by particulate deposition are summarized in Exhibit H-20. As was the case for the inhalation pathway, estimated cancer risks and hazard quotients for all of the constituents in both nonwastewater streams are below levels of concern for exposure by both pathways. The cancer risks and hazard quotients for the two pathways are generally within about one order of magnitude of each other, with higher risks for the ingestion pathways in some cases and higher risks for dermal contact in others.

The HE lifetime cancer risk associated with soil ingestion exposures to arsenic in aluminum cast house dust is $7X10^{-7}$, while the CT value is $4X10^{-8}$. In comparison, the HE and CT cancer risk estimates for dermal exposures are $1X10^{-6}$ and $1X10^{-8}$, respectively. The highest HE hazard quotient for ingestion exposures (again associated with exposures to arsenic) is $1X10^{-2}$, while the highest HE hazard quotient for dermal exposures is $4X10^{-2}$ (for arsenic). Hazard quotients for the remaining constituents range downward by many orders of magnitude from these values.

Zinc is the only constituent in zinc waste ferrosilicon for which a toxicity value is available for the ingestion and dermal pathway. Hazard quotient values for ingestion and dermal exposures to zinc from this stream are on the order of 10^{-4} to 10^{-2}, which is similar to the values for aluminum cast house dust.

While there is no ingestion pathway toxicity parameter for lead, it should be noted that the predicted HE soil concentration (48 mg/kg) is about ten times lower than EPA's recommended risk-based cleanup standard for lead in residential soils of 500 mg/kg.

Exhibit H-20

Soil Ingestion and Dermal Contact Pathway Risk Assessment Results for Particulate Deposition										
			Ingestion				Dermal Contact			
			CANCER RISK		HAZARD QUOTIENT		CANCER RISK		HAZARD QUOTIENT	
Constituent	**CT Soil Concentration at 20 Years (mg/kg)**	**HE Soil Concentration at 20 Years (mg/kg)**	**CT**	**HE**	**CT**	**HE**	**CT**	**HE**	**CT**	**HE**
Aluminum Cast House Dust										
Antimony	5.26E-03	7.12E-02			1.75E-04	2.37E-03			2.98E-04	2.01E-02
Arsenic	2.24E-02	3.04E-01	3.88E-08	7.13E-07	9.97E-04	1.35E-02	1.34E-08	1.44E-06	9.79E-04	3.89E-02
Barium	7.01E-03	9.49E-02			1.33E-06	1.81E-05			1.05E-07	3.44E-06
Cadmium	5.05E-03	6.83E-02			1.35E-04	1.82E-03			3.31E-05	1.12E-03
Chromium(VI)	7.71E-02	1.04E+00			2.06E-04	2.78E-03			2.62E-04	1.15E-02
Lead	1.19E-02	1.61E-01								
Mercury	7.01E-08	9.49E-07			3.11E-09	4.22E-08			1.39E-12	4.53E-11
Nickel	1.82E-01	2.47E+00			1.21E-04	1.64E-03			5.43E-05	1.90E-03
Selenium	6.45E-04	8.73E-03			1.72E-06	2.33E-05			2.91E-06	1.86E-04
Silver	1.33E-03	1.80E-02			3.55E-06	4.81E-05			6.04E-06	4.09E-04
Zinc	8.41E-02	1.14E+00			3.74E-06	5.06E-05			2.95E-06	1.12E-04
Zinc Waste Ferrosilicon										
Lead	3.50E+00	4.75E+01								
Zinc	2.80E+01	3.80E+02			1.25E-03	1.69E-02			9.84E-04	3.72E-02

These values in and of themselves are, as noted above, below the levels of concern. In fact, the values are low enough so that simultaneous exposures to all of the contaminants through both pathways results in a summed cancer risk less than 10^{-5} and a combined hazard index of less than 1. Given the conservative methods used to derive these values, and the small size of the units being evaluated, these results provide a high degree of assurance that risks for actual receptors would be below levels of concern.

H.2.2.3.2 Ingestion of Home-Grown Crops

The risk results for exposures to particulates deposited on soils and crops are summarized in Exhibit H-21. The HE estimated cancer risks associated with exposures to arsenic in aluminum cast house dust ($7X10^{-7}$) is very close to that for the ingestion pathway. The CT cancer risk for this pathway is $3X10^{-8}$. The highest HE noncancer hazard quotient for this pathway is $6X10^{-3}$, again associated with arsenic exposures, and the CT value for arsenic is one order of magnitude lower ($5X10^{-4}$). Hazard quotient values for the other constituents through the ingestion of home-grown crops are all much lower than the corresponding values for arsenic.

H.2.2.3.5 Particulate Deposition to Surface Water

Because the releases to air are so small and the surface water dilution volumes are so high, risks associated with surface water deposition are evaluated using a screening approach not unlike that used in the HWIR-Waste Technical Background Document to establish media concentrations corresponding to risk levels of concern. In this analysis, the methods and assumptions described in Section H.2.1 were used to calculate concentrations in surface water that corresponded to calculated cancer risk levels of $1X10^{-5}$ and hazard quotients of 1.0. HE exposure assumptions were used to evaluate exposures through the drinking water and fish ingestion pathways. These HE health-based levels were then used as a basis for comparison with the results of the concentration modeling for particulate deposition to surface water, as shown in Exhibit H-22.

As can be seen from the exhibit, the predicted surface water concentrations of the toxic waste constituents associated with air particulate deposition are all many orders of magnitude below the HBLs for drinking water or fish ingestion (corresponding to 10^{-5} cancer risk and hazard quotient equal to 1.0). Cadmium, with an HE predicted concentration of about two orders of magnitude below the HBL for fish ingestion, and chromium (VI), with an HE concentration of about four orders of magnitude below the HBL for drinking water ingestion, come the closest to any of the HBLs among the constituents of aluminum cast house dust. In the case of zinc waste ferrosilicon, the HE surface water concentration of zinc is about two orders of magnitude below the HBL for fish ingestion, and the HE concentration of lead is about thirty-fold below the drinking water HBL, which is based on the Clean Waster Act MCL. All of these results indicate little cause for concern for adverse health effects through this pathway, especially considering the conservativeness of the exposure assumptions (e.g., 100 percent of the particulate is deposited in surface water).

Exhibit H-21

Home-Grown Crop Ingestion Pathway Risk Assessment Results for Particulate Deposition										
							CANCER RISK		HAZARD QUOTIENT	
Constituent	**CT Soil Concentration at 20 Years (mg/kg)**	**HE Soil Concentration at 20 Years (mg/kg)**	**CT Concentration in Above-Ground Vegetables (mg/kg)**	**HE Concentration in Above-Ground Vegetables (mg/kg)**	**CT Concentration in Root Vegetables (mg/kg)**	**HE Concentration in Root Vegetables (mg/kg)**	**CT**	**HE**	**CT**	**HE**
Aluminum Cast House Dust										
Antimony	6.57E-04	7.12E-03	4.35E-04	2.74E-03	9.86E-06	1.07E-04			1.26E-04	1.83E-03
Arsenic	2.80E-03	3.04E-02	1.40E-03	6.71E-03	7.73E-07	8.38E-06	2.91E-08	7.00E-07	5.25E-04	5.67E-03
Barium	8.76E-04	9.49E-03	5.36E-04	3.18E-03	2.48E-08	2.69E-07			8.63E-07	1.15E-05
Cadmium	6.31E-04	6.83E-03	5.19E-04	3.72E-03	2.52E-07	2.73E-06			1.17E-04	1.89E-03
Chromium(VI)	9.64E-03	1.04E-01	4.53E-03	2.01E-02	2.41E-06	2.61E-05			1.02E-04	1.02E-03
Lead	1.49E-03	1.61E-02	6.89E-04	2.98E-03	4.79E-11	5.19E-10				
Mercury	8.76E-09	9.49E-08	4.12E-09	1.83E-08	1.29E-15	1.40E-14			1.55E-09	1.55E-08
Nickel	2.28E-02	2.47E-01	1.13E-02	5.35E-02	2.22E-06	2.41E-05			6.34E-05	6.78E-04
Selenium	8.06E-05	8.73E-04	3.86E-05	1.75E-04	4.12E-07	4.47E-06			8.81E-07	9.21E-06
Silver	1.66E-04	1.80E-03	1.44E-04	1.05E-03	4.16E-05	4.51E-04			4.56E-06	8.59E-05
Zinc	1.05E-02	1.14E-01	7.49E-03	4.95E-02	1.16E-05	1.25E-04			2.82E-06	4.20E-05
Zinc Waste Ferrosilicon										
Lead	4.38E-01	4.75E+00	2.03E-01	8.78E-01	1.41E-08	1.53E-07				
Zinc	3.50E+00	3.80E+01	2.50E+00	1.65E+01	3.85E-03	4.18E-02			9.39E-04	1.40E-02

EXHIBIT H-22

Screening Results for Particulate Deposition to Surface Water										
Constituent	Concentrations Resulting from Releases of Aluminum Cast House Dust			Concentrations Resulting from Releases of Zinc Waste ferrosilicon			Surface Water HBL Concentrations (mg/L)[1]			
	Maximum Concentration in Waste (mg/kg)	CT Water Concentration (mg/L)	HE Water Concentration (mg/L)	Maximum Concentration in Waste (mg/kg)	CT Water Concentration (mg/L)	HE Water Concentration (mg/L)	Fish - Noncancer	Fish - Cancer	Drinking Water - Noncancer	Drinking Water - Cancer
Antimony	7.5	8.10E-09	8.77E-07						1.40E-02	
Arsenic	32	3.46E-08	3.74E-06					7.40E-04		8.40E-04
Barium	10	1.08E-08	1.17E-06				3.77E-01		2.45E+00	
Beryllium							2.84E-02		1.75E-01	
Cadmium	7.2	7.78E-09	8.42E-07				7.35E-05		3.50E-02	
Chromium(VI)	110	1.19E-07	1.29E-05				9.00E-01		1.75E-01	
Lead	17	1.84E-08	1.99E-06	5000	5.40E-06	5.85E-04			1.50E-02	
Mercury	0.0001	1.08E-13	1.17E-11						1.05E-02	
Nickel	260	2.81E-07	3.04E-05				1.02E-01		7.00E-01	
Selenium	0.92	9.94E-10	1.08E-07				8.40E-03		1.75E-01	
Silver	1.9	2.05E-09	2.22E-07				1.80E-02		1.75E-01	
Thallium							3.02E-05		2.80E-03	
Vanadium									2.45E-01	
Zinc	120	1.30E-07	1.40E-05	40000	4.32E-05	4.68E-03	3.12E-01		1.05E+01	

[1] HBLs correspond to an estimated lower risk of 10^{-5}, a noncancer hazard quotient of 1.0, or for lead, the MCL.

H.2.2.4 Risk Results for Runoff Releases to Surface Soils

The screening methods used to estimate constituent concentrations in surface soils due to runoff release from waste piles are summarized in Section H.2.1.8.3. Risks from this pathway were again evaluated by comparison of the resultant concentrations to HBLs. In this case, however, the HBLs were soil concentrations derived for the incidental ingestion and dermal contact pathways, and for the ingestion of contaminated root vegetables. The results of this analysis are summarized in Exhibit H-23.

As was the case for the air deposition pathway, the concentrations of toxic constituents in soils resulting from runoff releases are all below levels that would be associated with concern for adverse health effects. Both in the case of the ingestion and dermal exposure pathways, where shallow mixing depths were used, and in the case of the root vegetable ingestion pathway, where greater mixing depths were used, the estimated HE and CT concentrations of toxic constituents in soils are generally several orders of magnitude below the levels that might be associated with significant adverse health effects. (In the case of arsenic, the HBLs correspond to soil concentrations that would be associated with an HE cancer risk of 10^{-5}. For the other constituents, the HBLs correspond to soil concentrations resulting in HE noncancer hazard quotient values of 1.0.)

These results hold true both for aluminum cast house dust and zinc waste ferrosilicon, even though, in the latter case, the predicted HE concentration of zinc is quite high (4,000 mg/kg). This finding is a result of zinc's relatively low human toxicity. The predicted HE concentration of lead (497 mg/kg) is just below EPA's recommended risk-based cleanup standard for lead in residential soils.

H.2.2.5 Risk Results for Runoff Deposition to Surface Water

Runoff from the waste piles may also be deposited into surface water. Long-term concentrations of waste constituents in surface water resulting from runoff loading were calculated for both waste streams, as described previously, and the resulting concentrations were compared to HBLs for surface water in the same fashion as was done for deposition of airborne particulates.

The results of that analysis are summarized in Exhibit H-24. As might be expected, since the amounts of materials released through surface runoff are roughly comparable to the amounts of air particulate generated, the results of the screening surface water risk analysis for this pathway are similar to those for air particulate deposition, in that all of the calculated concentrations of constituents in the surface water bodies are far below the HBLs for either surface water ingestion or the ingestion of fish.

For aluminum cast house dust, the highest HE surface water concentrations (of antimony, arsenic, chromium, lead, and nickel) associated with runoff releases were all in the range of 10^{-6} to 10^{-5} mg/l, all of which were lower than the corresponding HBLs. In the case of zinc waste ferrosilicon, the estimated HE concentrations of lead and zinc, the two constituents for which concentration data were available, are both about ten-fold lower than the lowest HBLs. These results indicate that runoff releases to surface water are unlikely to be associated with significant risks to human health.

EXHIBIT H-23

COMPARISON OF SOIL CONCENTRATIONS FROM RUNOFF RELEASES TO HEALTH-BASED LEVELS											
				Aluminum Cast House Dust				Zinc Waste Ferrosilicon			
Constituent	Soil Ingestion Health-Based Level (mg/kg)	Soil Dermal Contact Health-Based Level (mg/kg)	Home-Grown Vegetable Consumption Health-Based Level (mg/kg)	CT Soil Concentration (Ingestion and Dermal) (mg/kg)	HE Soil Concentration (Ingestion and Dermal Contact) (mg/kg)	CT Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)	HE Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)	CT Soil Concentration (Ingestion and Dermal) (mg/kg)	HE Soil Concentration (Ingestion and Dermal Contact) (mg/kg)	CT Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)	HE Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)
Antimony	30	3.54	74.1	1.92E-03	1.58E-01	2.39E-04	1.58E-02				
Arsenic	4.26	2.11	24.5	8.17E-03	6.74E-01	1.02E-03	6.74E-02				
Barium	525	27,600	>1,000,000	2.55E-03	2.11E-01	3.19E-04	2.11E-02				
Beryllium	NA	NA	NA								
Cadmium	37.5	61.1	3470	1.84E-03	1.52E-01	2.30E-04	1.52E-02				
Chromium	375	90.7	55600	2.81E-02	2.32E+00	3.51E-03	2.32E-01				
Cyanide	NA	NA	NA								
Lead	NA	NA	NA	4.34E-03	3.58E-01	5.43E-04	3.58E-02	6.02E+00	4.97E+02	7.52E-01	4.97E+01
Mercury	22.5	21,000	>1,000,000	2.55E-08	2.11E-06	3.19E-09	2.11E-07				
Nickel	1,500	1,300	569,000	6.64E-02	5.48E+00	8.30E-03	5.48E-01				
Selenium	375	47	2,710	2.35E-04	1.94E-02	2.94E-05	1.94E-03				
Silver	375	44.1	55.6	4.85E-04	4.00E-02	6.07E-05	4.00E-03				
Thallium	NA	NA	NA								
Vanadium	NA	NA	NA								
Zinc	22,500	10,200	758,000	3.07E-02	2.53E+00	3.83E-03	2.53E-01	4.82E+01	3.97E+03	6.02E+00	3.97E+02

EXHIBIT H-24

COMPARISON OF SURFACE WATER CONCENTRATIONS DUE TO SOIL RUNOFF RELEASES TO HEALTH-BASED LEVELS						
			Aluminum Cast House Dust		Zinc Waste Ferrosilicon	
Constituent	**Drinking Water Health-Based Level (mg/l)[1]**	**Fish Ingestion Health-Based Level (mg/l)**	**Maximum High-End Surface Water Concentration, Bulk Samples (mg/l)**	**Maximum Central Tendency Surface Water Concentration, Bulk Samples (mg/l)**	**Maximum High-End Surface Water Concentration, Bulk Samples (mg/l)**	**Maximum Central Tendency Surface Water Concentration, Bulk Samples (mg/l)**
Antimony	0.014	NA	7.52E-09	1.15E-06		
Arsenic	0.00084	0.00074	3.21E-08	4.89E-06		
Barium	2.45	0.377	1.00E-08	1.53E-06		
Beryllium	0.175	0.0284				
Cadmium	0.035	0.0000735	7.22E-09	1.10E-06		
Chromium	0.175	0.9	1.10E-07	1.68E-05		
Cyanide	0.7	36.5				
Lead	0.015	NA	1.71E-08	2.60E-06	2.36E-05	3.60E-03
Mercury	0.0105	0.00000125	1.00E-13	1.53E-11		
Nickel	0.7	0.102	2.61E-07	3.97E-05		
Selenium	0.175	0.0084	9.23E-10	1.41E-07		
Silver	0.175	0.018	1.91E-09	2.90E-07		
Thallium	0.0028	0.0000302				
Vanadium	0.245	NA				
Zinc	10.5	0.312	1.20E-07	1.83E-05	1.89E-04	2.88E-02

[1] HBLs correspond to a lower risk of 10^{-5}, a noncancer hazard quotient of 1.0, or, for lead, the MCL value.

H.2.2.6 Risk Results for Surface Impoundment Releases to Surface Water

The surface water concentrations of toxic constituents resulting from surface impoundment releases were also compared to surface water HBLs. Unlike the other pathways evaluated, the screening comparison indicates the potential for adverse effects on human health above levels of concern for a few constituents from some samples from several waste streams. These results are summarized below.

H.2.2.6.1 Ingestion of Surface Water

Exhibit H-25 summarizes the results of the comparison of surface water concentrations from impoundment releases to HBLs. Because there are multiple samples available for most of the waste streams managed in surface impoundments, the results of the comparison to HBLs are reported in terms of

EXHIBIT H-25

COMPARISON OF SURFACE WATER CONCENTRATIONS FROM SURFACE IMPOUNDMENT RELEASES TO HEALTH-BASED LEVELS DRINKING WATER												
				Maximum High-End Surface Water Concentration, Bulk Samples		**Maximum High-End Surface Water Concentration, EP Samples[2]**			**Central Tendency Surface Water Concentration, Bulk Samples**		**Central Tendency Surface Water Concentration, EP Samples[2]**	
				Compared to HBL[1]		**Compared to HBL**			**Compared to HBL**		**Compared to HBL**	
Constituent	**Commodity**	**Wastestream**	**Total No. Samples**	**1-10x**	**10-100x**	**1-10x**	**10-100x**	**100-1000x**	**1-10x**	**10-100x**	**1-10x**	**10-100x**
Arsenic	Copper	Acid plant blowdown	40	3	1			1			1	
Cadmium	Zinc	Spent surface impoundment liquids	24	1								
Lead	Copper	Acid plant blowdown	40	1								
	Zinc	Spent surface impoundment liquids	24	1	1							

NOTES:

1. HBLs correspond to a lower risk of 10^{-5}, a noncancer hazard quotient of 1.0, or, for lead, the MCL value.
2. EP samples are adjusted (i.e., have been multiplied by 1.95) to extrapolate to bulk concentrations.

the numbers of samples and recycled streams for which the HE and CT surface water concentrations from impoundment releases exceed the HBLs, presented in order-of-magnitude categories.

Releases from surface impoundment failures were modeled as resulting in potential exceedences of HBLs for water ingestion for three constituents: arsenic, cadmium, and lead. Under high-end dilution assumptions, the arsenic concentrations in five samples (four bulk samples, one EP extraction) would exceed the drinking water HBL by up to one thousand-fold. (This is equivalent, in this case, to saying that the estimated cancer risks under HE assumptions would exceed the 10^{-5} level of concern by up to a factor of 1000.) All of these samples came from the copper acid plant blowdown stream, and under CT dilution assumptions the surface water concentration for arsenic exceeds the HBL for only one of the 40 total samples from this stream.

The concentration of cadmium in one of 24 samples from the zinc spent surface impoundment liquid stream results in surface water concentrations exceeding the drinking water HBL under HE assumptions. The HBL is exceeded by a factor of ten or less. Under CT assumptions, there are no HBL exceedences for cadmium. For cadmium, an HBL exceedence corresponds to a hazard quotient value exceeding 1.0 for its critical toxic effect on kidney function.

The lead concentrations in bulk samples from two waste streams result in calculated surface water concentrations exceeding the drinking water HBL. One sample of copper acid plant blowdown shows a concentration of lead such that the HE concentrations exceeds the HBL by a factor of less than ten. Under CT assumptions, this sample no longer exceeds the HBL. Two bulk samples of zinc spent surface impoundment liquids result in HE lead concentrations in surface water that exceed the HBL by a factor of up to 100. Again, under the CT dilution assumptions, the predicted lead concentrations in surface water are reduced to below the drinking water HBL. As noted previously, the HBL for lead is simply the Drinking Water MCL of 15 ug/l.

H.2.2.6.2 Ingestion of Contaminated Fish

The predicted surface water concentrations of six contaminants released from surface impoundments also were such that HBLs derived for the ingestion of fish by subsistence fishers were exceeded. The results are presented in Exhibit H-26. Six arsenic samples (again all from copper acid plant blowdown) were associated with HE surface water concentrations exceeding the fish consumption HBLs by up to a factor of 1000. Four of these were bulk samples, and the remainder were EP extraction samples. Under CT assumptions, only one sample exceeded the arsenic fish ingestion HBL.

A total of 20 samples (one EP extraction, the rest bulk) contained cadmium concentrations which resulted in surface water concentrations exceeding the fish ingestion HBL by a up to 1000-fold. These samples came from zinc spent surface impoundment liquids (10), zinc process wastewater (6), copper acid plant blowdown (2 samples), and one sample each from rare earths process wastewater and zinc wastewater treatment plant liquid effluent. Under CT dilution assumptions, the number of samples exceeding the cadmium HBL is reduced to 3 samples, and the maximum level of exceedence is reduce to less than 100-fold.

EXHIBIT H-26

COMPARISON OF SURFACE WATER CONCENTRATIONS FROM SURFACE IMPOUNDMENT RELEASES TO HEALTH-BASED LEVELS
FISH INGESTION

				Maximum High-End Surface Water Concentration, Bulk Samples			Central Tendency Surface Water Concentration, Bulk Samples		Maximum High-End Surface Water Concentration, EP Samples[2]			Central Tendency Surface Water Concentration, EP Samples[2]	
				Compared to HBL[1]			Compared to HBL		Compared to HBL			Compared to HBL	
Constituent	**Commodity**	**Wastestream**	**Total No. Samples**	**1-10x**	**10-100x**	**100-1000x**	**1-10x**	**10-100x**	**1-10x**	**10-100x**	**100-1000x**	**1-10x**	**10-100x**
Arsenic	Copper	Acid plant blowdown	40	2	2				1		1	1	
Cadmium	Copper	Acid plant blowdown	40	2									
	Rare Earths	Process wastewater	8						1				
	Zinc	Process wastewater	40	6									
	Zinc	Spent surface impoundment liquids	24	6	3	1	1	1					
	Zinc	WWTP liquid effluent	5			1		1					
Mercury	Copper	Acid plant blowdown	40	2					1				
	Zinc	Spent surface impoundment liquids	24	1	1								
Selenium	Copper	Acid plant blowdown	40	1									
Thallium	Titanium and Titanium Dioxide	Leach liquid & sponge wash water	8	1									
	Copper	Acid plant blowdown	40						1				
Zinc	Zinc	Spent surface impoundment liquids	24	5									
	Zinc	WWTP liquid effluent	5	1									

NOTES:

1. HBL = health-based level derived for fish ingestion based on worst-cast subsistence fisher.
2. EP samples are adjusted (i.e., have been multiplied by 1.95) to extrapolate to bulk concentrations.

Under HE assumptions, five samples give mercury concentrations in surface water exceeding the fish ingestion HBL. These samples come from copper acid plant blowdown (3) and zinc spent surface impoundment liquids (2), and under CT assumptions, none of these samples exceeds the fish HBL. In the case of mercury, an HBL exceedence is equivalent to a hazard quotient greater than 1.0 for reproductive effects.

A single sample result for selenium in copper acid plant blowdown results in surface water concentrations above the HBL, as do two thallium results (one each from titanium/TiO_2 leach liquor and sponge wash water and from copper acid plant blowdown). For all of these samples, no exceedences occur under CT dilution assumptions. The same is true for the six analytical results for zinc (all from zinc commodity streams); all six samples exceed the fish ingestion HBL under HE but not under CT dilution assumptions.

H.2.2.7 Summary of Non-Groundwater Pathway Risk Assessment Results

The findings of this analysis parallel the results of the groundwater risk assessment for the storage of mineral processing wastes, which found generally very low risks for the nonwastewater streams disposed in waste piles, and higher risks (exceeding 10^{-5} cancer risk and hazard quotients of 1.0 in some instances) for the wastewaters and liquid nonwastewater streams disposed in surface impoundments.

In the groundwater analysis, the major reasons for the relatively low estimated risks were the generally low DAF values for waste piles, and the relatively low masses of toxic constituents in the relatively small piles. In this analysis, the small size of the waste piles (corresponding to the low recycled volumes of these streams) is again decisive in determining the generally low risks for the nonwastewater streams. None of the release events and exposure pathways that were evaluated for waste piles resulted in risks greater than the previously-noted levels of concern under either CT or HE assumptions. Estimated releases from both runoff and air particulate generation were low (in the range of a few hundred to a few thousand kilograms per year total mass), and even moderate dilution in exposure media was enough to reduce exposure concentrations below levels of concern with regard to adverse health effects.

The comparatively higher risks associated with waste managed in surface impoundments was primarily a function of the larger volumes of waste being managed and correspondingly larger release volumes. Even though the proportions of the recycled materials released from impoundments were relatively low, there was still enough mass present in the impoundments to result in surface water concentrations exceeding HBLs. It should be noted, however, that even for these high-volume wastes, exceedences of HBLs were limited to only a small minority of the constituents, samples and waste streams, and the greatest numbers of exceedences were for the fish ingestion pathway, where the HBLs for several constituents have been derived quite conservatively. Under HE assumptions, only nine samples (out of 135 having analytical data) resulted in exceedences of the drinking water HBL, and this number dropped to one under CT assumptions. Under HE assumptions, a total of 40 samples exceeded the far more stringent fish ingestion HBLs, and this number dropped to 4 under CT assumptions.

Two of the twelve wastewater and liquid nonwastewater streams evaluated in the analysis accounted for the bulk of the HBL exceedences. Under HE assumptions, samples from copper acid plant blowdown accounted for six of the nine exceedences of the drinking water HBL, and zinc spent surface impoundment liquids accounted for the remaining three. Between them , these two streams also accounted for 34 of the 40 HE exceedences of the fish ingestion HBLs (copper acid plant blowdown 13, zinc spent surface impoundment liquids 21). Two other streams from the zinc commodity sector (six samples from process

wastewater and two samples from waste water treatment plant liquid effluent) also accounted for one or more exceedences of the fish consumption HBL. Beyond that, only two other commodity sectors (rare earths and titanium/TiO_2) had any exceedences (one each, only under HE assumptions).

Thus, this analysis clearly identifies two commodity sectors and four waste streams as dominant in driving potential risks from the storage of mineral processing wastes, at least among the streams for which analytical data are available. Whether there are other streams and commodities for which non-groundwater risks might also exceed levels of concern cannot be determined without additional data concerning waste characteristics and composition.

H.2.2.8 Uncertainties/Limitations of the Analysis

As discussed in Section H.2.1, the multipathway risk assessment for the storage of mineral processing recycled materials relies on relatively simple, generic models of contaminant releases, transport, exposures, and risks. As such, this screening level analysis shares the general limitations of all generic analyses in that high levels of uncertainty and variability may not be adequately treated, since only a limited number of generally applicable models and generally representative data are used to model risks from a wide range of units, wastes, and constituents. Many of these generic sources of uncertainty have been addressed in our previous work on mineral processing wastes, and the following discussion is limited to limitations specific to the multipathway analysis

Constituent concentration data are available for only 14 recycled waste streams, and for some wastes only small numbers of samples are available. It is interesting to note that two of the wastes for which estimated risks are the highest (copper acid plant blowdown and zinc spent surface impoundment liquids) also are those for which the largest number of samples are available. It is not possible to estimate which of the other wastes might also show risks above levels of concern if more data were available.

Limited data are also available concerning waste characteristics, including constituent speciation, solubility, and bioavailability. Throughout this analysis, we have assumed that all constituents would behave in such a manner as to maximize exposure potential. For example, we have assumed that none of the constituents would leach from soils after their initial deposition, and that all of the constituents would be bioavailable in the water column. Generally these assumptions increase the level of conservatism in the risk assessment.

Release events and amounts were simulated mostly using the general methods adopted in HWIR-Waste. The one exception is air particulate generation, which was estimated using the SCREEN3 model, rather than the model recommended in HWIR-Waste. SCREEN3 is a widely-accepted screening level EPA model, however, and EPA believes that it is appropriate for the types of release events that were modeled. The use of SCREEN3 is unlikely to have biased the results of the risk assessment significantly compared to other methods. However, as noted previously, no data were available concerning the particle size characteristics of the two wastes streams that were modeled, so we relied on data from an earlier study of mineral processing wastes stored in waste piles. Based on limited information, we believe that the particle size distribution that was used may overstate the potential for particulate release of the more coarse-grained, high-density zinc waste ferrosilicon, while more accurately describing the potential for particulate releases of aluminum cast house dust.

Runoff releases were evaluated using the same model (USLE) applied in HWIR-Waste, with input parameters varied slightly to reflect the operating characteristics of the waste piles being simulated and the likely geographic distribution of the recycling facilities. The risk results are not particularly sensitive to these changes, as exposure concentrations for runoff events are below the levels of concern for all of the runoff exposure pathways.

The ISCST3 model used to predict particulate air concentrations and deposition rates is a state-of-the-art model that has been used in many regulatory proceedings by EPA. The input data that were used, the "worst-case" meteorological conditions that are supplied with ISCST3 specifically for use in screening level assessments, were somewhat more conservative than the meteorological data used in HWIR-Waste with a similar model. Thus, our estimates of air impacts are likely to be higher than those that would have been achieved had we replicated the HWIR-Waste approach. Again, however, all risks associated with this pathway were far below levels of concern.

The modeling of releases from surface impoundments reproduced exactly the approach used in HWIR-Waste. This release model and its input parameters were derived based on data from management units in the pulp and paper industry, and just how reliably they predict releases from surface impoundments in the mineral processing industries is not known. This is clearly a major source of uncertainty in the risk assessment, as these release events are the only non-groundwater releases for which health risks are predicted to be above levels of concern.

Because of resource limitations and the specific characteristics of the facilities that were evaluated, simplified approaches were developed to estimate the concentrations of waste constituents in surface soils and surface water to substitute for the much more elaborate methods used in HWIR-Waste. In the case of surface runoff, in the absence of site-specific data, we conservatively assume that soil contamination would be limited to relatively small distances (50 or 100 meters) from the piles in arbitrarily defined circular plumes. This is only intended as a bounding analysis, and the finding that this pathway is not a major concern can be supported by the fact that, even with these relatively small exposure areas (and the resultant high soil concentrations), constituent concentrations due to runoff events were below levels of health concern.

Similarly, to be conservative, we assumed that all of the runoff and all of the particulate generated by the waste piles would be deposited on the watershed in such a way that all of these materials would rapidly find their way into surface water. This approach, while it resulted in surface water concentrations far below levels of health concern, may be less conservative than the approach taken for surface soils, in that the CT and HE streams are both rather large, and the model does not take into account possible runoff or deposition into smaller streams, lakes, or ponds where constituents may accumulate in surface water or sediment.

The approach we took in evaluating fish tissue concentrations was also somewhat more conservative than that taken in HWIR-Waste, in that we used the highest reliable BCF or BAF values, rather than representative values, in our calculations. For some constituents (arsenic, cadmium, mercury, thallium), this approach resulted in considerably higher tissue concentrations than would have been calculated had we used the HWIR-Waste values, and considerably lower HBLs. This may be a major source of uncertainty in this analysis, since the fish ingestion pathway resulted in the highest risks for several of the constituents.

ATTACHMENT H.A

Proportion of Mineral Processing Wastes Covered by the Storage Risk Assessment

ATTACHMENT H.A-1

PROPORTION OF RECYCLED MINERAL PROCESSING WASTE STREAMS ADDRESSED IN THE RISK ASSESSMENT FOR RECYCLED MATERIALS STORAGE

		Total Recycled Volume			Recycled Volume Analyzed in Risk Assessment	Percent Analyzed in Risk Assessment		
Commodity	**Waste Stream**	**Min.**	**Ave.**	**Max.**	**Expect.**	**Exp./Min.**	**Expect./Ave.**	**Exp./Max.**
Aluminum, Alumina	Cast House Dust	16,227	16,227	16,227	16,227	100.00%	100.00%	100.00%
	Electrolysis Waste		24,438	48,875				
	Sector	16,227	40,665	65,102	16,227	100.00%	39.90%	24.93%
Beryllium	Chip Treatment Wastewater		10,000	400,000	10,000		100.00%	2.50%
	Sector		10,000	400,000	10,000		100.00%	2.50%
Copper	Acid Plant Blowdown	3,975,000	3,975,000	3,975,000	3,975,000	100.00%	100.00%	100.00%
	WWTP Sludge		2,250	4,500				
	Sector	3,975,000	3,977,250	3,979,500	3,975,000	100.00%	99.94%	99.89%
Elemental Phosphorus	AFM Rinsate	4,000	4,000	4,000	4,000	100.00%	100.00%	100.00%
	Furnace Scrubber Blowdown	420,000	420,000	420,000	420,000	100.00%	100.00%	100.00%
	Furnace Building Washdown	700,000	700,000	700,000				
	Sector	1,124,000	1,124,000	1,124,000	424,000	37.72%	37.72%	37.72%
Rare Earths	Electrol. Cell Caustic Wet APC Slud.		350	7,000				
	Process Wastewater	1,400	1,400	1,400	1,400	100.00%	100.00%	100.00%
	Spent Scrubber Liquor	20	100,000	200,000				
	Wastewater from APC		50,000	200,000				
	Sector	1,420	151,750	408,400	1,400	98.59%	0.92%	0.34%
Selenium	Spent Filter Cake		217	4,335				
	Plant Process Wastewater	13,200	13,200	13,200	13,200	100.00%	100.00%	100.00%
	Slag		51	1,020				
	Tellurium Slime Wastes		217	4,335				
	Sector	13,200	13,685	22,890	13,200	100.00%	96.46%	57.67%
Tantalum, Ferrocolum-bium, etc.	Process Wastewater	127,500	127,500	127,500	127,500	100.00%	100.00%	100.00%
	Sector	127,500	127,500	127,500	127,500	100.00%	100.00%	100.00%
Titanium, Titanium Oxide	Pickel Liquor and Wash Water		270	660				
	Scrap Milling Scrubber Water		500	1,200	500		100.00%	41.67%
	Smut from Mg Recovery	85	18,700	39,100				
	Leach Liquor,Sponge Wash Water	76,000	96,000	116,000	96,000	126.32%	100.00%	82.76%
	Spent Surface Impoundment Liquids		1,458	5,712				
	Sector	76,085	116,928	162,672	96,500	126.83%	82.53%	59.32%
Zinc	Acid Plant Blowdown	130,000	130,000	130,000				
	Waste Ferrisilicon		7,225	14,450	7,225		100.00%	50.00%
	Process Wastewater	4,335,000	4,335,000	4,335,000	4,335,000	100.00%	100.00%	100.00%
	Spent Clothes, Bags, and Filters		75	150				
	Spent Goethite, Leach Cake Residues	15,000	15,000	15,000				
	Spent Surface Impoundment Liquids	378,000	378,000	378,000	378,000	100.00%	100.00%	100.00%
	WWTP Solids		281	563				
	TAC Tower Blowdown		94	188				
	WWTP Liquid Effluent		261,000	522,000	261,000		100.00%	50.00%
	Sector	4,858,000	5,126,675	5,395,351	4,981,225	102.54%	97.16%	92.32%
All Sectors		**14,099,602**	**14,843,809**	**16,805,223**	**9,645,052**	**68.41%**	**64.98%**	**57.39%**

Notes:

Proportion of streams covered = 14/73 = 19.2 percent

Commodities not covered = Antimony, Bismuth, Cadmium, Calcium, Coal Gas, Fluorspar and Hydrofluoric Acid, Germanium, Lead, Magnesium and Magnesia, Mercury, Platinum Group Metals, Pyrobitumens, Rhenium, Scandium, Synthetic Rutile, Tellurium, Tungsten, Uranium, Zirconium and Hafnium

ATTACHMENT H.A-2

PROPORTION OF MINERAL PROCESSING WASTE STREAMS ADDRESSED IN THE RISK ASSESSMENT FOR RECYCLED MATERIALS STORAGE

		Generation Rate			Recycled Volume			Percent Recycled		
Commodity	**Waste Stream**	**Min.**	**Ave.**	**Max.**	**Min.**	**Expect.**	**Max.**	**Min./Min.**	**Expect./Ave.**	**Max./Max.**
Aluminum, Alumina	Cast House Dust	19,000	19,000	19,000	16,227	16,227	16,227	85.41%	85.41%	85.41%
	Electrolysis Waste	58	58	58						
	Sector	19,058	19,058	19,058	16,227	16,227	16,227	85.15%	85.15%	85.15%
Beryllium	Chip Treatment Wastewater	200	100,000	2,000,000	0	10,000	400,000	0.00%	10.00%	20.00%
	Filtration Discard	200	450	90,000						
	Sector	400	100,450	2,090,000	0	10,000	400,000	0.00%	9.96%	19.14%
Copper	Acid Plant Blowdown	5,300,000	5,300,000	5,300,000	3,975,000	3,975,000	3,975,000	75.00%	75.00%	75.00%
	WWTP Sludge	6,000	6,000	6,000						
	Sector	5,306,000	5,306,000	5,306,000	3,975,000	3,975,000	3,975,000	74.92%	74.92%	74.92%
Elemental Phosphorus	Anderson Filter Media	460	460	460						
	AFM Rinsate	4,000	4,000	4,000	4,000	4,000	4,000	100.00%	100.00%	100.00%
	Furnace Scrubber Blowdown	410,000	410,000	410,000	420,000	420,000	420,000	102.44%	102.44%	102.44%
	Furnace Building Washdown	700,000	700,000	700,000						
	Sector	1,114,460	1,114,460	1,114,460	424,000	424,000	424,000	38.05%	38.05%	38.05%
Rare Earths	Spent NH_4NO_3 Solution	14,000	14,000	14,000						
	Electrol. Cell Caustic Wet APC Slud.	70	70	700						
	Process Wastewater	7,000	7,000	7,000	1,400	1,400	1,400	20.00%	20.00%	20.00%
	Spent Scrubber Liquor	100	500,000	1,000,000						
	Solvent Extraction Crud	100	2,300	4,500						
	Wastewater from APC	100	500,000	1,000,000						
	Sector	21,370	1,023,370	2,026,200	1,400	1,400	1,400	6.55%	0.14%	0.07%
Selenium	Spent Filter Cake	50	500	5,000						
	Plant Process Wastewater	66,000	66,000	66,000	13,200	13,200	13,200	20.00%	20.00%	20.00%
	Slag	50	500	5,000						
	Tellurium Slime Wastes	50	500	5,000						
	Waste Solids	50	500	5,000						
	Sector	66,200	68,000	86,000	13,200	13,200	13,200	19.94%	19.41%	15.35%
Tantalum, Ferrocolum-bium, etc.	Digester Sludge	1,000	1,000	1,000						
	Process Wastewater	150,000	150,000	150,000	127,500	127,500	127,500	85.00%	85.00%	85.00%
	Spent Raffinate Solids	2,000	2,000	2,000						
	Sector	153,000	153,000	153,000	127,500	127,500	127,500	83.33%	83.33%	83.33%
Titanium, Titanium Oxide	Pickel Liquor and Wash Water	2,200	2,700	3,200						
	Scrap Milling Scrubber Water	4,000	5,000	6,000		500	1,200	0.00%	10.00%	20.00%
	Smut from Mg Recovery	100	22,000	45,000						
	Leach Liquor,Sponge Wash Water	380,000	480,000	580,000	76,000	96,000	116,000	20.00%	20.00%	20.00%
	Spent Surface Impoundment Liquids	630	3,400	6,700						
	Waste Acids (Sulfate Process)	200	39,000	77,000						
	WWTP Sludges/Solids	420,000	420,000	420,000						
	Sector	807,130	972,100	1,137,900	76,000	96,500	117,200	9.42%	9.93%	10.30%
Zinc	Acid Plant Blowdown	130,000	130,000	130,000						
	Waste Ferrisilicon	17,000	17,000	17,000	0	7,225	14,450	0.00%	42.50%	85.00%
	Process Wastewater	5,000,000	5,000,000	5,000,000	4,335,000	4,335,000	4,335,000	86.70%	86.70%	86.70%
	Discarded Refractory Brick	1,000	1,000	1,000						
	Spent Clothes, Bags, and Filters	150	150	150						
	Spent Goethite, Leach Cake Residues	15,000	15,000	15,000						
	Spent Surface Impoundment Liquids	1,900,000	1,900,000	1,900,000	378,000	378,000	378,000	19.89%	19.89%	19.89%
	WWTP Solids	750	750	750						
	Spent Synthetic Gypsum	16,000	16,000	16,000						
	TAC Tower Blowdown	250	250	250						
	WWTP Liquid Effluent	2,600,000	2,600,000	2,600,000	0	261,000	522,000	0.00%	10.04%	20.08%
	Sector	9,680,150	9,680,150	9,680,150	4,713,000	4,981,225	5,249,450	48.69%	51.46%	54.23%
	All Sectors	**21,523,000**	**24,411,000**	**32,246,000**	**9,346,327**	**9,645,052**	**10,323,977**	**43.42%**	**39.51%**	**32.02%**

Notes:

Proportion of streams covered = 14/118 = 11.9 percent

Commodities not covered = Antimony, Bismuth, Cadmium, Calcium, Coal Gas, Fluorspar and Hydrofluoric Acid, Germanium, Lead, Magnesium and Magnesia, Mercury, Molybdenum, Ferromolybdenum, and Ammonium Molybdate, Pyrobitumens, Rhenium, Scandium, Synthetic Rutile, Tellurium, Tungsten, Uranium

ATTACHMENT H.B

Summary of Particulate Generation, Air Transport, and Deposition Modeling

ATTACHMENT H.B. Air Quality Modeling in Support of Mineral Processing Storage Analysis

Model Selection and Options

The Industrial Source Complex Short Term (ISC3ST- version 96113) was used to model the impacts of fugitive emissions from materials handling and wind erosion at the mineral processing facilities. The ISC3ST model is the model recommended by EPA in the Guideline On Air Quality Models (Revised), EPA-450/2-78-027R, Appendix W of 40 CFR Part 51 and Part 52. As stated in the guidance document:

> "Fugitive emissions are usually defined as emissions that come from an industrial source complex. They include the emissions resulting from the industrial process that are not captured and vented through a stack but may be released from various locations within the complex. Where such fugitive emissions can be properly specified, the ISC model, with consideration of gravitational settling and dry deposition, is the recommended model."

The ISC3ST model was set-up to run using the following regulatory default options:

- Final plume rise
- Stack-tip downwash
- Buoyancy-induced dispersion
- Calms processing
- Default wind profile exponents
- Default vertical potential temperature gradients
- No exponential decay.

However, since the only sources included were fugitive area sources, the options applicable to stack point sources (e.g. stack-tip downwash) were not applied.

Emission Estimates

Emissions associated with the storage of mineral processing waste (aluminum/alumina cast house dust and ferrosilicon waste from zinc production) were estimated to occur from the aggregate handling of the waste materials and from the wind erosion of the waste piles. Emissions from the aggregate handling of the waste piles vary in proportion to the mean wind speed and the moisture content of the waste. Emissions generated by wind erosion of the waste piles were related to threshold friction velocity and the wind gusts of the highest magnitude routinely measured as the fastest mile. Because the lack of data, we made a few assumptions in estimating these emissions:

a) The material in the storage piles has a moisture content of 4.8 percent

b) The threshold friction velocity for the waste piles is the same as the threshold friction velocity for fine coal dust stored on a concrete pad. This assumption would overestimate emissions for the waste piles since fine coal dust on concrete pad has a greater erosion potential than the waste piles.

c) The fastest mile, (i.e., the wind gusts of the highest magnitude) occurs during period between disturbances to the piles.

d) The surface area of the storage pile which is disturbed during each work day is equal to 25 percent of the total pile surface area.

e) Data for the annual mean wind speed and for the fastest mile were taken for Kansas City which has an average values of the cities surveyed in "Extreme Wind Speed at 129 Stations in the Contiguous United States".

Emissions from handling of the waste materials and from the wind erosion of the waste piles were estimated using equations from EPA's AP-42, Compilation of Air Pollution Emission Factors, Volume I: Stationary Point and Area Sources. As previously stated, these equations relate parameters such as exposed surface area, moisture content, mean wind speed, threshold friction velocity, fastest mile to total TSP and PM_{10} emissions.

Meteorological Assumptions

In addition to the meteorological assumptions needed to estimate emissions from mineral processing waste piles, meteorological data was required to complete the air quality dispersion modeling analysis using ISC3ST. To conservatively predict the impacts of the emission sources, worst-case meteorological data was used in ISC3ST.

The worst-case meteorological data is similar to that incorporated in the EPA model, SCREEN3. The worst-case meteorological data set contains an array of all possible combinations of wind speed, wind direction and stability class that could exist in an actual location. The data set of meteorological conditions consisted of:

- Mixing heights of 1000 meters
- Ambient temperatures of 298 DegK
- Wind directions varying from 10 to 360 degrees
- Wind speeds (varying from 1.0 m/sec to 20.0 m/sec) assigned to stability classes A through F

A few additional parameters are required to estimate deposition using the ISC3ST model. Those parameters include: The variables are: friction velocity at the application site (m/s), Monin-Obukhov length at the application site (m) and roughness length at the application site (m). The EPA model RAMMET, version 95227 was used to estimate these parameters. RAMMET requires data on surface roughness length at application site, noon time albedo and Bowen ratio, which vary by season and land-use type. Values by season and land-use type (10% urban, deciduous forest, coniferous forest, grassland and desert shrubland share the 90%, i.e., 22.5% each) were estimated. The appropriate fraction velocity, Monin-Obukhov length and roughness length values were extracted from the RAMMET output and added to worst case meteorological data for the deposition calculations.

Location of Maximum and Area-Average Concentrations and Concentrations

As with many Gaussian dispersion models, ISC3ST results are accurate no closer than 100 meters from each source. Thus to calculate impacts of the two sets of storage piles, both piles were placed in a prototypical facility with property boundaries located approximately 100 meters from the edge of each storage pile. Two sets of receptor grids were used to determine maximum peak 24 hour and annual average concentration and deposition values at points located around the property boundary. To pinpoint the maximum values, a grid of receptor points, with receptors located from 100 meters to 250 meters in each direction, with a resolution of 50 meters was input to ISC3ST. An array of polar receptors, at 45 degree intervals, from 200 to 3,000 meters was used to estimate area average concentrations.

Results

The maximum predicted 24 hour and annual average concentration (ug/m^3) and deposition (g/m^2) values are listed in Table 1. These maximum concentration were predicted to occur 180 meters from the Al cast house dust storage pile and 104 meters from the Ferrosilicon storage pile. Area average values were estimated over the entire polar receptor grid.

Table 1- Modeling Results

Pollutant	TSP		PM_{10}	
	24 Hour	Annual Average	24 hour	Annual Average
Max. Concentration	258.4	64.6	192.2	32.3
Area Average	18.5	4.6	9.2	2.3
Max Deposition	2.6e-3	6.5e-4	5.4e-4	1.3e-4
Area Ave. Dep.	5.9e-4	1.5e-4	1.1e-4	2.9e-5

ATTACHMENT H.C

Fish Bioconcentration and Bioaccumulation Factor Values and Data Sources

Attachment H.C
Fish BCF, and Toxicity Values

Chemical	Cas Number	BAF fish (L/kg body weight) (total)	Source	BCF fish (L/kg) (dissolved)	Source	RfD (mg/kg/day)	Source	Oral CSF (mg/kg/day)-1	Source	RfC (mg/m3)	Source	Inhal URF (ug/m3)-1	Source
Antimony	7440-36-0	NA		not significant	Barrows et. al 1980 (in EPA 1988)*	4.00E-04	IRIS	NA		NA		NA	
Arsenic	7440-38-2	NA		4	Barrows et al. 1978*	3.00E-04	IRIS	1.50E+00	IRIS	NA		4.30E-03	IRIS
Barium	7440-39-3	NA		100	Schroeder 1970*	7.00E-02	IRIS	NA		5.00E-04	HEAST	NA	
Beryllium	7440-41-7	NA		19	Barrows et al. 1978*	5.00E-03	IRIS	4.30E+00	IRIS	NA		2.40E-03	IRIS
Cadmium	7440-43-9	NA		3-7,440	Benoit et al. 1976 (in EPA 1985a)*; Giesy et al. 1977 (in Eisler 1985)*	5.00E-04	IRIS	NA		NA		1.80E-03	IRIS
Chromium (VI)	18540-29-9	NA		3	EPA 1985b	5.00E-03	IRIS	NA		NA		1.20E-02	IRIS
Cyanide	57-12-5			0.3	Kenaga 1980 (KCN)*	2.00E-02	IRIS	NA		NA		NA	
Lead	7439-92-1	8		1-726	Maddock and Taylor 1980 (in Eisler 1988)*; Wong et al. 1981 (in Eisler 1988)*		NA	NA		NA		NA	
Mercury	7439-97-6	6.00E+04	EPA 1993b	129-10,000 (mercury(II)); 10,000-85,700 (methylmercury)	Various refs. in EPA 1985c*	3.00E-04	IRIS (HgCl2)	NA		3.00E-04	IRIS	NA	
Nickel	7440-02-0	NA		47-106	Lind et al. manuscript (in EPA 1986)*	2.00E-02	IRIS (soluble salts)	NA		NA		NA	
Selenium	7482-49-2	0.5-1.0	Cleveland et. al 1993	5-322	Cleveland et. al 1993*; Ingersoll et. al 1990*	5.00E-03	IRIS	NA		NA		NA	
Silver	7440-22-4	NA		11-150	EPA 1987	5.00E-03	IRIS	NA		NA		NA	
Thallium	7440-28-0	NA		27-1430	Zitko et al. 1975; Barrows et al. 1978*	8.00E-05	IRIS (Tl2Ch2O3,TlCl, or Tl2H2SO4)	NA		NA		NA	
Vanadium	7440-62-2	NA		NA		7.00E-03	HEAST	NA		NA		NA	
Zinc	7440-66-6	4.4		275-519	Xu and Pascoe 1993*	3.00E-01	IRIS	NA		NA		NA	

ATTACHMENT H.D

Risk Characterization and Screening Spreadsheets

H.D-1 Inhalation Pathway

H.D-2 Particulate Depostion Soil Ingestion and Dermal Contact

H.D-3 Particulate Deposition to Surface Water Risk Screening Results

H.D-4 Runoff Deposition to Soils Screening Results

H.D-5 Runoff Deposition to Surface Water Screening Results

H.D-6 Surface Impoundment Releases to Surface Soils Screening Results

ATTACHMENT H.D-1
Inhalation Pathway

Exposure and Risk Calculations for Particulate Deposition

COMMODITY:	Alumina and Aluminum	
WASTE STREAM:	Cast house dust	
CT PM 10 Concentration	2.3	ug/m3
HE PM10 Concentration	32.3	ug/m3

Constituent	RfC (mg/m3)	Unit Risk (ug/m3)-1	Maximum Concentration in Waste (mg/kg)	CT Constituent Concentration in Particulate (ug/m3)	HE Constituent Concentration in Particulate (ug/m3)	CT Cancer Risk	HE Cancer Risk	CT Noncancer Hazard Quotient	HE Noncancer Hazard Quotient
Antimony			7.5	1.73E-05	2.42E-04	0.00E+00	0.00E+00		
Arsenic		4.30E-03	32	7.36E-05	1.03E-03	3.90E-13	1.22E-11		
Barium	5.00E-04		10	2.30E-05	3.23E-04	0.00E+00	0.00E+00	1.92E-01	1.92E-01
Beryllium		2.40E-03	0	0.00E+00	0.00E+00	0.00E+00	0.00E+00		
Cadmium		1.80E-03	7.2	1.66E-05	2.33E-04	3.67E-14	1.15E-12		
Chromium(VI)		1.20E-02	110	2.53E-04	3.55E-03	3.74E-12	1.17E-10		
Lead			17	3.91E-05	5.49E-04	0.00E+00	0.00E+00		
Mercury	3.00E-04		0.0001	2.30E-10	3.23E-09	0.00E+00	0.00E+00	3.20E-06	3.20E-06
Nickel		4.80E-04	260	5.98E-04	8.40E-03	3.54E-13	1.10E-11		
Selenium			0.92	2.12E-06	2.97E-05	0.00E+00	0.00E+00		
Silver			1.9	4.37E-06	6.14E-05	0.00E+00	0.00E+00		
Thallium			0	0.00E+00	0.00E+00	0.00E+00	0.00E+00		
Vanadium			0	0.00E+00	0.00E+00	0.00E+00	0.00E+00		
Zinc			120	2.76E-04	3.88E-03	0.00E+00	0.00E+00		

Exposure Variables		**CT**	**HE**	**Units**
EF	Exposure Frequency	350	350	days/year
EDa	Exposure Duration (Adult)	9	20	years

Cancer Risk = U.R. * PM10 * Max Conc.* 10^-6* (EF/365) * (ED/70)

Hazard Quotient = (EF/365) * (Max. Conc.* 10^-6) / RfC

ATTACHMENT H.D-2
Particulate Deposition - Soil Ingestion and Dermal Contact

Exposure and Risk Calculations for Particulate Deposition

COMMODITY:	Alumina and Aluminum			
WASTE STREAM:	Cast house dust			
CT Long-Term Deposition	1.50E-04	g/m2-hour	4.17E-08	g/m2-sec
HE Long-Term Deposition	6.50E-04	g/m2-hour	1.81E-07	g/m2-sec

1. Soil Ingestion

Constituent	Ingestion CSF	Ingestion RfD	Maximum Concentration in Waste (mg/kg)	CT Constituent Concentration in Deposition (g/m2-sec)	HE Constituent Concentration in Deposition (g/m2-sec)	CT Soil Concentration at 20 Years (mg/kg)	HE Soil Concentration at 20 Years (mg/kg)	CT Lifetime Constituent Combined Intake Due Soil to Ingestion (mg/kg-day)	HE Lifetime Constituent Combined Intake Due to Soil Ingestion (mg/kg-day)	CT Constituent Child Intake Due Soil to Ingestion-Hazard (mg/kg-day)	HE Constituent Child Intake Due Soil to Ingestion-Hazard (mg/kg-day)	CT Lifetime Cancer Risk	HE Lifetime Cancer Risk	CT Noncancer Hazard Quotient (Child)	HE Noncancer Hazard Quotient (Child)
Antimony	0.00E+00	4.00E-04	7.5	3.13E-13	1.35E-12	5.26E-03	7.12E-02	6.07E-09	1.11E-07	7.01E-08	9.49E-07	0.00E+00	0.00E+00	1.75E-04	2.37E-03
Arsenic	1.50E+00	3.00E-04	32	1.33E-12	5.78E-12	2.24E-02	3.04E-01	2.59E-08	4.75E-07	2.99E-07	4.05E-06	3.88E-08	7.13E-07	9.97E-04	1.35E-02
Barium	0.00E+00	7.00E-02	10	4.17E-13	1.81E-12	7.01E-03	9.49E-02	8.09E-09	1.49E-07	9.34E-08	1.27E-06	0.00E+00	0.00E+00	1.33E-06	1.81E-05
Beryllium	0.00E+00	5.00E-03	0	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cadmium	0.00E+00	5.00E-04	7.2	3.00E-13	1.30E-12	5.05E-03	6.83E-02	5.83E-09	1.07E-07	6.73E-08	9.11E-07	0.00E+00	0.00E+00	1.35E-04	1.82E-03
Chromium(VI)	0.00E+00	5.00E-03	110	4.58E-12	1.99E-11	7.71E-02	1.04E+00	8.90E-08	1.63E-06	1.03E-06	1.39E-05	0.00E+00	0.00E+00	2.06E-04	2.78E-03
Lead	0.00E+00	0.00E+00	17	7.08E-13	3.07E-12	1.19E-02	1.61E-01	1.38E-08	2.53E-07	1.59E-07	2.15E-06	0.00E+00	0.00E+00		
Mercury	0.00E+00	3.00E-04	0.0001	4.17E-18	1.81E-17	7.01E-08	9.49E-07	8.09E-14	1.49E-12	9.34E-13	1.27E-11	0.00E+00	0.00E+00	3.11E-09	4.22E-08
Nickel	0.00E+00	2.00E-02	260	1.08E-11	4.69E-11	1.82E-01	2.47E+00	2.10E-07	3.86E-06	2.43E-06	3.29E-05	0.00E+00	0.00E+00	1.21E-04	1.64E-03
Selenium	0.00E+00	5.00E-03	0.92	3.83E-14	1.66E-13	6.45E-04	8.73E-03	7.44E-10	1.37E-08	8.60E-09	1.16E-07	0.00E+00	0.00E+00	1.72E-06	2.33E-05
Silver	0.00E+00	5.00E-03	1.9	7.92E-14	3.43E-13	1.33E-03	1.80E-02	1.54E-09	2.82E-08	1.78E-08	2.40E-07	0.00E+00	0.00E+00	3.55E-06	4.81E-05
Thallium	0.00E+00	8.00E-05	0	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Vanadium	0.00E+00	7.00E-03	0	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zinc	0.00E+00	3.00E-01	120	5.00E-12	2.17E-11	8.41E-02	1.14E+00	9.71E-08	1.78E-06	1.12E-06	1.52E-05	0.00E+00	0.00E+00	3.74E-06	5.06E-05

2. Soil Dermal Contact

Constituent	Ingestion CSF	Ingestion RfD	Soil-Water Partition Coefficient (cm3/g)	CT Soil Concentration at 20 Years (mg/kg)	HE Soil Concentration at 20 Years (mg/kg)	Skin Permeability Constant (cm/hr)	CT Soil Disappearance Rate Constant (1/hr)	HE Soil Disappearance Rate Constant (1/hr)	CT Absorption Fraction (unitless)	HE Absorption Fraction (unitless)	CT Absorbed Dose, Carcinogens (mg/kg-day)	HE Absorbed Dose, Carcinogens (mg/kg-day)	CT Absorbed Dose, Noncarcinogens (mg/kg-day)	HE Absorbed Dose, Noncarcinogens (mg/kg-day)	CT Cancer Risk	HE Cancer Risk	CT Noncancer Hazard Quotient	HE Noncancer Hazard Quotient
Antimony	0.00E+00	4.00E-04	2.00E+00	5.26E-03	7.12E-02	1.89E-04	2.50E+00	5.00E-01	1.00E+00	9.98E-01	3.64E-09	6.61E-07	1.19E-07	8.05E-06	0.00E+00	0.00E+00	2.98E-04	2.01E-02
Arsenic	1.50E+00	3.00E-04	2.90E+01	2.24E-02	3.04E-01	1.30E-05	1.72E-01	3.45E-02	5.78E-01	3.39E-01	8.96E-09	9.59E-07	2.94E-07	1.17E-05	1.34E-08	1.44E-06	9.79E-04	3.89E-02
Barium	0.00E+00	7.00E-02	5.30E+02	7.01E-03	9.49E-02	7.12E-07	9.43E-03	1.89E-03	4.61E-02	2.24E-02	2.23E-10	1.98E-08	7.32E-09	2.41E-07	0.00E+00	0.00E+00	1.05E-07	3.44E-06
Beryllium	0.00E+00	5.00E-03	7.00E+01	0.00E+00	0.00E+00	5.39E-06	7.14E-02	1.43E-02	3.00E-01	1.58E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cadmium	0.00E+00	5.00E-04	1.60E+02	5.05E-03	6.83E-02	2.36E-06	3.13E-02	6.25E-03	1.45E-01	7.23E-02	5.05E-10	4.60E-08	1.65E-08	5.60E-07	0.00E+00	0.00E+00	3.31E-05	1.12E-03
Chromium(VI)	0.00E+00	5.00E-03	1.80E+01	7.71E-02	1.04E+00	2.10E-05	2.78E-01	5.56E-02	7.51E-01	4.87E-01	4.00E-08	4.73E-06	1.31E-06	5.76E-05	0.00E+00	0.00E+00	2.62E-04	1.15E-02
Lead	0.00E+00	0.00E+00	2.80E+05	1.19E-02	1.61E-01	1.35E-09	1.79E-05	3.57E-06	8.93E-05	4.29E-05	7.36E-13	6.44E-11	2.41E-11	7.84E-10	0.00E+00	0.00E+00		
Mercury	0.00E+00	3.00E-04	9.50E+04	7.01E-08	9.49E-07	3.97E-09	5.26E-05	1.05E-05	2.63E-04	1.26E-04	1.28E-17	1.12E-15	4.18E-16	1.36E-14	0.00E+00	0.00E+00	1.39E-12	4.53E-11
Nickel	0.00E+00	2.00E-02	8.20E+01	1.82E-01	2.47E+00	4.60E-06	6.10E-02	1.22E-02	2.63E-01	1.36E-01	3.31E-08	3.13E-06	1.09E-06	3.81E-05	0.00E+00	0.00E+00	5.43E-05	1.90E-03
Selenium	0.00E+00	5.00E-03	4.30E+00	6.45E-04	8.73E-03	8.78E-05	1.16E+00	2.33E-01	9.97E-01	9.39E-01	4.45E-10	7.63E-08	1.46E-08	9.29E-07	0.00E+00	0.00E+00	2.91E-06	1.86E-04
Silver	0.00E+00	5.00E-03	4.00E-01	1.33E-03	1.80E-02	9.43E-04	1.25E+01	2.50E+00	1.00E+00	1.00E+00	9.21E-10	1.68E-07	3.02E-08	2.04E-06	0.00E+00	0.00E+00	6.04E-06	4.09E-04
Thallium	0.00E+00	8.00E-05	7.40E+01	0.00E+00	0.00E+00	5.10E-06	6.76E-02	1.35E-02	2.87E-01	1.50E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Vanadium	0.00E+00	7.00E-03	5.00E+01	0.00E+00	0.00E+00	7.55E-06	1.00E-01	2.00E-02	3.93E-01	2.13E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zinc	0.00E+00	3.00E-01	4.00E+01	8.41E-02	1.14E+00	9.43E-06	1.25E-01	2.50E-02	4.65E-01	2.59E-01	2.70E-08	2.75E-06	8.86E-07	3.35E-05	0.00E+00	0.00E+00	2.95E-06	1.12E-04

Pathway Variables		**CT**	**HE**	**Units**
Skin Permeability Constants(5-14)				
Kow	Skin Perm. Constant for H20	0.001	0.001	cm/hr
rho	Soil Particle Density	2.65	2.65	gm/cc
Dermal Absorption Fraction (Child) (5-23)				
AF	Adherence Factor	0.2	1	mg/cm2
rho	Soil Particle Density	2.65	2.65	gm/cc
ks	Soil Loss Constant	0	0	1/years
kvol	Volatilization Loss Constant	0	0	1/years
tevent	Event Duration	5	12	hours

		CT	**HE**	**Units**
Dermal Absorbed Dose (5-20, 5-21,5-22)				
BWc	Body Weight (Child)	15	15	kg
EV	Event Frequency	1	1	events/day
A	Exposed Surface Area	1700	1700	cm2
EF	Exposure Frequency	130	350	days/year
EDc	Exposure Duration (Child)	6	6	years
AT	Averaging Time	70	70	years
Soil Concentration due to Deposition: Dermal and Ingestion (6-1)				
Z	Mixing Depth	2.5	1	cm
BD	Soil Bulk Density	1.5	1.2	gm/cc
ks	Soil Loss Constant	0	0	1/years
t	Deposition Period	9	20	years

		CT	**HE**	**Units**
Intake/Cancer Risk due to Soil Ingestion (5-6)				
AT	Averaging Time	70	70	years
EF	Exposure Frequency	350	350	days/year
IRc	Ingestion Rate (Child)	200	200	mg/day
IRa	Ingestion Rate (Adult)	100	100	mg/day
BWc	Body Weight (Child)	15	15	kg
BWa	Body Weight (Adult)	70	70	kg
EDc	Exposure Duration (Child)	6	6	years
EDa	Exposure Duration (Adult)	3	24	years
CSF	Oral Cancer Slope Factor	chemical-specific		(mg/kg-day)-1

ATTACHMENT H.D-2 (Continued)
Particulate Deposition - Vegetable Ingestion

Exposure and Risk Calculations for Particulate Deposition

COMMODITY:	Alumina and Aluminum					
WASTE STREAM:	Cast house dust					
CT Long-Term Deposition	1.50E-04	g/m2-hour	4.17E-08	g/m2-sec	1.31E+00	g/m2-year
HE Long-Term Deposition	6.50E-04	g/m2-hour	1.81E-07	g/m2-sec	5.69E+00	g/m2-year

Constituent	Ingestion CSF	Ingestion RfD	Root Concentration Factor (RCF) (mg/kg)/(mg/kg) DW	Plant-Soil Bioconcentration Factor (Br) (ug/kg)/(ug/kg) DW	CT Soil Concentration at 20 Years (mg/kg)	HE Soil Concentration at 20 Years (mg/kg)	CT Concentration in Above-Ground Vegetables (mg/kg)	HE Concentration in Above-Ground Vegetables (mg/kg)	CT Concentration in Root Vegetables (mg/kg)	HE Concentration in Root Vegetables (mg/kg)	CT Absorbed Dose, Carcinogens (mg/kg-day)	HE Absorbed Dose, Carcinogens (mg/kg-day)	CT Absorbed Dose, Noncarcinogens (mg/kg-day)	HE Absorbed Dose, Noncarcinogens (mg/kg-day)	CT Cancer Risk for Vegetable Consumption	HE Cancer Risk From Vegetable Consumption	CT Noncancer Hazard Quotient from Vegetable Consumption	HE Noncancer Hazard Quotient from Vegetable Consumption
Antimony	0.00E+00	4.00E-04	3.00E-02	2.00E-01	6.57E-04	7.12E-03	4.35E-04	2.74E-03	9.86E-06	1.07E-04	6.23E-09	2.01E-07	5.06E-08	7.32E-07	0.00E+00	0.00E+00	1.26E-04	1.83E-03
Arsenic	1.50E+00	3.00E-04	8.00E-03	3.60E-02	2.80E-03	3.04E-02	1.40E-03	6.71E-03	7.73E-07	8.38E-06	1.94E-08	4.66E-07	1.57E-07	1.70E-06	2.91E-08	7.00E-07	5.25E-04	5.67E-03
Barium	0.00E+00	7.00E-02	1.50E-02	1.50E-01	8.76E-04	9.49E-03	5.36E-04	3.18E-03	2.48E-08	2.69E-07	7.45E-09	2.21E-07	6.04E-08	8.05E-07	0.00E+00	0.00E+00	8.63E-07	1.15E-05
Beryllium	0.00E+00	5.00E-03	1.50E-03	1.00E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cadmium	0.00E+00	5.00E-04	6.40E-02	3.60E-01	6.31E-04	6.83E-03	5.19E-04	3.72E-03	2.52E-07	2.73E-06	7.20E-09	2.59E-07	5.84E-08	9.44E-07	0.00E+00	0.00E+00	1.17E-04	1.89E-03
Chromium(VI)	0.00E+00	5.00E-03	4.50E-03	7.50E-03	9.64E-03	1.04E-01	4.53E-03	2.01E-02	2.41E-06	2.61E-05	6.29E-08	1.40E-06	5.10E-07	5.10E-06	0.00E+00	0.00E+00	1.02E-04	1.02E-03
Lead	0.00E+00	0.00E+00	9.00E-03	1.30E-05	1.49E-03	1.61E-02	6.89E-04	2.98E-03	4.79E-11	5.19E-10	9.56E-09	2.07E-07	7.75E-08	7.56E-07	0.00E+00	0.00E+00		
Mercury	0.00E+00	3.00E-04	1.40E-02	8.00E-03	8.76E-09	9.49E-08	4.12E-09	1.83E-08	1.29E-15	1.40E-14	5.72E-14	1.27E-12	4.64E-13	4.64E-12	0.00E+00	0.00E+00	1.55E-09	1.55E-08
Nickel	0.00E+00	2.00E-02	8.00E-03	3.20E-02	2.28E-02	2.47E-01	1.13E-02	5.35E-02	2.22E-06	2.41E-05	1.56E-07	3.72E-06	1.27E-06	1.36E-05	0.00E+00	0.00E+00	6.34E-05	6.78E-04
Selenium	0.00E+00	5.00E-03	2.20E-02	1.60E-02	8.06E-05	8.73E-04	3.86E-05	1.75E-04	4.12E-07	4.47E-06	5.43E-10	1.26E-08	4.41E-09	4.60E-08	0.00E+00	0.00E+00	8.81E-07	9.21E-06
Silver	0.00E+00	5.00E-03	1.00E-01	4.00E-01	1.66E-04	1.80E-03	1.44E-04	1.05E-03	4.16E-05	4.51E-04	2.81E-09	1.18E-07	2.28E-08	4.29E-07	0.00E+00	0.00E+00	4.56E-06	8.59E-05
Thallium	0.00E+00	8.00E-05	4.00E-04	4.00E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Vanadium	0.00E+00	7.00E-03	3.00E-03	5.50E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zinc	0.00E+00	3.00E-01	4.40E-02	2.50E-01	1.05E-02	1.14E-01	7.49E-03	4.95E-02	1.16E-05	1.25E-04	1.04E-07	3.45E-06	8.45E-07	1.26E-05	0.00E+00	0.00E+00	2.82E-06	4.20E-05

Pathway Variables

Soil Concentration due to Deposition: Root Vegetables (6-58)

		CT	HE	Units
Z	Mixing Depth	20	10	cm
BD	Soil Bulk Density	1.5	1.2	gm/cc
ks	Soil Loss Constant	0	0	1/years
t	Deposition Period	9	20	years

Root Vegetable Concentration (6-57)

		CT	HE	Units
kd	soil-water dis. constant	chemical-specific		l/kg
RCF	Root Concentration Factor			(mg/kg)/(mg/kg)
Vg	Correction Factor	1	1	unitless

Above-Ground Vegetable Concentrations from Combined Deposition (6-48)

		CT	HE	Units
Br	Plant-Soil BCF	chemical-specific		(ug/kg)/(ug/kg) (DW)
ks	Soil Loss Constant	0	0	1/years
t	Deposition Period	9	20	years
Z	Mixing Depth	20	10	cm
BD	Soil Bulk Density	1.5	1.2	gm/cc
Rp	Interception Fraction	0.05	0.05	unitless
kp	Plant Sur. Loss Coeff.	18	18	year-1
tp	Plant Exposure to Deposition	0.16	0.16	years
Yp	Yield	1.7	1.7	kg/m2 (DW)

Crop Intake (Subsistence Farmer-Noncarcinogen) (5-58, 5-59)

		CT	HE	Units
BWa	Body Weight (Adult)	70	70	kg
F	Fraction From Contaminated Soil	0.4	0.9	unitless
CRa	Consumption Rate (Above-Ground)	19.7	19.7	gm/day
CRr	Consumption Rate (Root)	28	28	gm/day
EF	Exposure Frequency	350	350	days/year

Crop Intake (Subsistence Farmer-Carcinogen) (5-56, 5-57)

		CT	HE	Units
BWa	Body Weight (Adult)	70	70	kg
F	Fraction From Contaminated Soil	0.4	0.9	unitless
CRa	Consumption Rate (Above-Ground)	19.7	19.7	gm/day
CRr	Consumption Rate (Root)	28	28	gm/day
EF	Exposure Frequency	350	350	days/year
ED	Exposure Duration	9	20	years
AT	Averaging Time	70	70	years

ATTACHMENT H.D-3
Air Emissions to Surface Water - Risk Screening Results

Exposure and Risk Calculations for Air Emissions

COMMODITY:	Alumina and Aluminum			
WASTE STREAM:	Cast house dust		**Flow Rate**	
CT Long-Term Emissions	3.24E+08	mg/year	3.00E+11	L/year
HE Long-Term Emissions	1.52E+09	mg/year	1.30E+10	L/year

Constituent	Maximum Concentration in Waste (mg/kg)	CT Water Concentration (mg/L)	HE Water Concentration (mg/L)	Surface Water HBL Concentrations (mg/L)			
				Fish - Noncancer	Fish - Cancer	Drinking Water - Noncancer	Drinking Water - Cancer
Antimony	7.5	8.10E-09	8.77E-07	NA	NA	1.40E-02	NA
Arsenic	32	3.46E-08	3.74E-06	NA	7.40E-04	NA	8.40E-04
Barium	10	1.08E-08	1.17E-06	3.77E-01	NA	2.45E+00	NA
Beryllium	0	0.00E+00	0.00E+00	2.84E-02	NA	1.75E-01	NA
Cadmium	7.2	7.78E-09	8.42E-07	7.35E-05	NA	3.50E-02	NA
Chromium(VI)	110	1.19E-07	1.29E-05	9.00E-01	NA	1.75E-01	NA
Lead	17	1.84E-08	1.99E-06	NA	NA	1.50E-02	NA
Mercury	0.0001	1.08E-13	1.17E-11	NA	NA	1.05E-02	NA
Nickel	260	2.81E-07	3.04E-05	1.02E-01	NA	7.00E-01	NA
Selenium	0.92	9.94E-10	1.08E-07	8.40E-03	NA	1.75E-01	NA
Silver	1.9	2.05E-09	2.22E-07	1.80E-02	NA	1.75E-01	NA
Thallium	0	0.00E+00	0.00E+00	3.02E-05	NA	2.80E-03	NA
Vanadium	0	0.00E+00	0.00E+00	NA	NA	2.45E-01	NA
Zinc	120	1.30E-07	1.40E-05	3.12E-01	NA	1.05E+01	NA

ATTACHMENT H.D-4
Runoff Deposition to Soils Screening Results

Release, Exposure Risk Calculations for Waste Piles

1. Aluminum Cast House Dust

Constituent	Ingestion Pathway Cancer Slope Factor (mg-kg-day)-1	Ingestion Pathway RfD (mg/kg-day)	Maximum Bulk Concentration in Waste (mg/kg)	CT Soil Concentration (Ingestion and Dermal) (mg/kg)	HE Soil Concentration (Ingestion and Dermal Contact) (mg/kg)	CT Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)	HE Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)
Antimony		0.0004	7.5	1.92E-03	1.58E-01	2.39E-04	1.58E-02
Arsenic	1.5	0.0003	32	8.17E-03	6.74E-01	1.02E-03	6.74E-02
Barium		0.07	10	2.55E-03	2.11E-01	3.19E-04	2.11E-02
Beryllium		0.07					
Cadmium		0.0005	7.2	1.84E-03	1.52E-01	2.30E-04	1.52E-02
Chromium(VI)		0.005	110	2.81E-02	2.32E+00	3.51E-03	2.32E-01
Lead			17	4.34E-03	3.58E-01	5.43E-04	3.58E-02
Mercury		0.0003	0.0001	2.55E-08	2.11E-06	3.19E-09	2.11E-07
Nickel		0.02	260	6.64E-02	5.48E+00	8.30E-03	5.48E-01
Selenium		0.005	0.92	2.35E-04	1.94E-02	2.94E-05	1.94E-03
Silver		0.005	1.9	4.85E-04	4.00E-02	6.07E-05	4.00E-03
Thallium		0.00008					
Vanadium		0.009					
Zinc		0.3	120	3.07E-02	2.53E+00	3.83E-03	2.53E-01

Pathway Variables		**CT**	**HE**	**Units**
USLE Release Modeling (7-52)				
AWPd	Area of Waste Pile (Dust)	108	108	m2
AWPf	Area of Waste Pile (Ferrosilicon)	509	509	m2
R	Rainfall factor	50	110	1/year
K	Soil Erodability Factor	0.25	0.25	t/year
LS	Length-Slope Factor	1	3	unitless
C	Cover Factor	1	1	unitless
P	Control Practices Factor	1	1	unitless
SL	Total Soil Loss (Dust)	301	1986	kg/year
SL	Total Soil Loss (Ferrosilicon)	1418	9360	kg/year
Soil Delivery				
r	Radius of contaminated area	10000	5000	cm
Soil Concentration due to Deposition: Dermal and Ingestion (6-1)				
Z	Mixing Depth	2.5	1	cm
BD	Soil Bulk Density	1.5	1.2	gm/cc
ks	Soil Loss Constant	0	0	1/years
t	Deposition Period	20	20	years
SM	Mixed Soil Mass (Dermal and Ingestion)	1.18E+06	9.42E+04	kg
Soil Concentration due to Deposition: Root Vegetables (6-58)				
Z	Mixing Depth	20	10	cm
BD	Soil Bulk Density	1.5	1.2	gm/cc
ks	Soil Loss Constant	0	0	1/years
t	Deposition Period	20	20	years
SM	Mixed Soil Mass (Root Vegetables)	9.42E+06	9.42E+05	kg

ATTACHMENT H.D-4 (Continued)
Runoff Deposition to Soils Screening Results

Release, Exposure Risk Calculations for Waste Piles

2. Zinc Waste Ferrosilicon

Constituent	Ingestion Pathway Cancer Slope Factor (mg-kg-day)-1	Ingestion Pathway RfD (mg/kg-day)	Maximum Bulk Concentration in Waste (mg/kg)	CT Soil Concentration (Ingestion and Dermal) (mg/kg)	HE Soil Concentration (Ingestion and Dermal Contact) (mg/kg)	CT Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)	HE Soil Concentration (Ingestion of Home-Grown Vegetables) (mg/kg)
Antimony		0.0004					
Arsenic	1.5	0.0003					
Barium		0.07					
Beryllium		0.07					
Cadmium		0.0005					
Chromium(VI)		0.005					
Lead			5000	6.018874429	496.5571404	0.752359304	49.65571404
Mercury		0.0003					
Nickel		0.02					
Selenium		0.005					
Silver		0.005					
Thallium		0.00008					
Vanadium		0.009					
Zinc		0.3	40000	48.15099543	3972.457123	6.018874429	397.2457123

Pathway Variables		**CT**	**HE**	**Units**
USLE Release Modeling (7-52)				
AWPd	Area of Waste Pile (Dust)	108	108	m2
AWPf	Area of Waste Pile (Ferrosilicon)	509	509	m2
R	Rainfall factor	50	110	1/year
K	Soil Erodability Factor	0.25	0.25	t/year
LS	Length-Slope Factor	1	3	unitless
C	Cover Factor	1	1	unitless
P	Control Practices Factor	1	1	unitless
SL	Total Soil Loss (Dust)	301	1986	kg/year
SL	Total Soil Loss (Ferrosilicon)	1418	9360	kg/year
Soil Delivery				
r	Radius of contaminated area	10000	5000	cm
Soil Concentration due to Deposition: Dermal and Ingestion (6-1)				
Z	Mixing Depth	2.5	1	cm
BD	Soil Bulk Density	1.5	1.2	gm/cc
ks	Soil Loss Constant	0	0	1/years
t	Deposition Period	20	20	years
SM	Mixed Soil Mass (Dermal and Ingestion)	1.18E+06	9.42E+04	kg
Soil Concentration due to Deposition: Root Vegetables (6-58)				
Z	Mixing Depth	20	10	cm
BD	Soil Bulk Density	1.5	1.2	gm/cc
ks	Soil Loss Constant	0	0	1/years
t	Deposition Period	20	20	years
SM	Mixed Soil Mass (Root Vegetables)	9.42E+06	9.42E+05	kg

ATTACHMENT H.D-5
Runoff Deposition to Surface Water Screening Results

Release, Exposure Risk Calculations for Waste Piles

1. Aluminum Cast House Dust

Constituent	Ingestion Pathway Cancer Slope Factor (mg-kg-day)-1	Ingestion Pathway RfD (mg/kg-day)	Maximum Bulk Concentration in Waste (mg/kg)	CT Waterbody Concentration (mg/l)	HE Waterbody Concentration (mg/l)
Antimony		0.0004	7.5	7.52E-09	1.15E-06
Arsenic	1.5	0.0003	32	3.21E-08	4.89E-06
Barium		0.07	10	1.00E-08	1.53E-06
Beryllium		0.07			
Cadmium		0.0005	7.2	7.22E-09	1.10E-06
Chromium(VI)		0.005	110	1.10E-07	1.68E-05
Lead			17	1.71E-08	2.60E-06
Mercury		0.0003	0.0001	1.00E-13	1.53E-11
Nickel		0.02	260	2.61E-07	3.97E-05
Selenium		0.005	0.92	9.23E-10	1.41E-07
Silver		0.005	1.9	1.91E-09	2.90E-07
Thallium		0.00008			
Vanadium		0.009			
Zinc		0.3	120	1.20E-07	1.83E-05

Pathway Variables		**CT**	**HE**	**Units**
USLE Release Modeling (7-52)				
AWPd	Area of Waste Pile (Dust)	108	108	m2
AWPf	Area of Waste Pile (Ferrosilicon)	509	509	m2
R	Rainfall factor	50	110	1/year
K	Soil Erodability Factor	0.25	0.25	t/year
LS	Length-Slope Factor	1	3	unitless
C	Cover Factor	1	1	unitless
P	Control Practices Factor	1	1	unitless
SL	Total Soil Loss (Dust)	301	1986	kg/year
SL	Total Soil Loss (Ferrosilicon)	1418	9360	kg/year
Surface Water Characteristics				
	Flow Rate	3.00E+11	1.30E+10	liter/year

ATTACHMENT H.D-5 (Continued)
Runoff Deposition to Surface Water Screening Results

Release, Exposure Risk Calculations for Waste Piles

2. Zinc Waste Ferrosilicon

Constituent	Ingestion Pathway Cancer Slope Factor (mg-kg-day)-1	Ingestion Pathway RfD (mg/kg-day)	Maximum Bulk Concentration in Waste (mg/kg)	CT Waterbody Concentration (mg/l)	HE Waterbody Concentration (mg/l)
Antimony		0.0004			
Arsenic	1.5	0.0003			
Barium		0.07			
Beryllium		0.07			
Cadmium		0.0005			
Chromium(VI)		0.005			
Lead			5000	2.36361E-05	0.003599954
Mercury		0.0003			
Nickel		0.02			
Selenium		0.005			
Silver		0.005			
Thallium		0.00008			
Vanadium		0.009			
Zinc		0.3	40000	0.000189089	0.028799636

Pathway Variables		**CT**	**HE**	**Units**
USLE Release Modeling (7-52)				
AWPd	Area of Waste Pile (Dust)	108	108	m2
AWPf	Area of Waste Pile (Ferrosilicon)	509	509	m2
R	Rainfall factor	50	110	1/year
K	Soil Erodability Factor	0.25	0.25	t/year
LS	Length-Slope Factor	1	3	unitless
C	Cover Factor	1	1	unitless
P	Control Practices Factor	1	1	unitless
SL	Total Soil Loss (Dust)	301	1986	kg/year
SL	Total Soil Loss (Ferrosilicon)	1418	9360	kg/year
Surface Water Characteristics				
	Flow Rate	3.00E+11	1.30E+10	liter/year

ATTACHMENT H.D-6
Surface Impoundment Releases to Surface Water Screening Results

COMPARISON OF SURFACE WATER CONCENTRATIONS FROM SURFACE IMPOUNDMENT RELEASES TO HEALTH-BASED LEVELS														
FISH INGESTION														
						Maximum High-End Surface Water Concentration, Bulk Samples		**Central Tendency Surface Water Concentration, Bulk Samples**		**Maximum High-End Surface Water Concentration, EP Samples**			**Central Tendency Surface Water Concentration, EP Samples**	
						Compared to HBL		**Compared to HBL**		**Compared to HBL**			**Compared to HBL**	
Constituent	**Hazard-Based Level (mg/l)**	**Commodity**	**Wastestream**	**Facility**	**State**	**1-10x**	**10-100x**	**1-10x**	**10-100x**	**1-10x**	**10-100x**	**100-1000x**	**1-10x**	**10-100x**
Arsenic	0.00084	Copper	Acid plant blowdown	Unknown	Unknown		x							
		Copper	Acid plant blowdown	Unknown	Unknown	x								
		Copper	Acid plant blowdown	Unknown	Unknown	x								
		Copper	Acid plant blowdown	Unknown	Unknown	x								
		Copper	Acid plant blowdown	Magma, San Manuel	AZ							x	x	
Cadmium	0.035	Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA	x								
Lead	0.015	Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA		x							
		Copper	Acid plant blowdown	Unknown	Unknown	x								
		Zinc	Spent surface impoundment liquids	Big River Zinc	IL	x								

ATTACHMENT H.D-6 (Continued)
Surface Impoundment Releases to Surface Water Screening Results

COMPARISON OF SURFACE WATER CONCENTRATIONS FROM SURFACE IMPOUNDMENT RELEASES TO HEALTH-BASED LEVELS																
FISH INGESTION																
							Maximum High-End Surface Water Concentration, Bulk Samples			Central Tendency Surface Water Concentration, Bulk Samples		Maximum High-End Surface Water Concentration, EP Samples			Central Tendency Surface Water Concentration, EP Samples	
	Hazard-Based Level						Compared to HBL			Compared to HBL		Compared to HBL			Compared to HBL	
Constituent	(mg/l)	Source	Commodity	Wastestream	Facility	State	1-10x	10-100x	100-1000x	1-10x	10-100x	1-10x	10-100x	100-1000x	1-10x	10-100x
Arsenic	0.00074	HBL	Copper	Acid plant blowdown	Unknown	Unknown		x								
			Copper	Acid plant blowdown	Unknown	Unknown		x								
			Copper	Acid plant blowdown	Unknown	Unknown	x									
			Copper	Acid plant blowdown	Unknown	Unknown	x									
			Copper	Acid plant blowdown	Magma, San Manuel	AZ								x	x	
			Copper	Acid plant blowdown	CBI	CBI						x				
Cadmium	0.0000735	HBL	Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA			x		x					
			Zinc	WWTP liquid effluent	Zinc Corp of America	OK			x		x					
			Zinc	Spent surface impoundment liquids	Big River Zinc	IL		x		x						
			Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA		x								
			Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA		x								
			Copper	Acid plant blowdown	Unknown	Unknown	x									
			Zinc	Spent surface impoundment liquids	Big River Zinc	IL	x									
			Zinc	Process wastewater	Zinc Corp, Bartlesville	OK	x									
			Zinc	Process wastewater	Unknown	Unknown	x									
			Copper	Acid plant blowdown	Unknown	Unknown	x									
			Zinc	Process wastewater	Unknown	Unknown	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America	OK	x									
			Zinc	Process wastewater	Zinc Corp, Monaca	PA	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America	OK	x									
			Zinc	Process wastewater	Unknown	Unknown	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America	OK	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America	OK	x									
			Zinc	Process wastewater	Unknown	Unknown	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America	OK	x									
			Rare Earths	Process wastewater	Unknown	Unknown						x				
Mercury	0.00000125	HBL	Zinc	Spent surface impoundment liquids	Big River Zinc	IL		x								
			Zinc	Spent surface impoundment liquids	Big River Zinc	IL	x									
			Copper	Acid plant blowdown	Unknown	Unknown	x									
			Copper	Acid plant blowdown	Unknown	Unknown	x									
			Copper	Acid plant blowdown	CBI	CBI						x				
Selenium	0.0084	HBL	Copper	Acid plant blowdown	Unknown	Unknown	x									
Thallium	0.0000302	HBL	Titanium and Titanium Dioxide	Leach liquid & sponge wash water	Timet, Henderson	NV	x									
			Copper	Acid plant blowdown	Cyprus, Clay Pool	AZ						x				
Zinc	0.312	HBL	Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA	x									
			Zinc	Spent surface impoundment liquids	Big River Zinc	IL	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA	x									
			Zinc	WWTP liquid effluent	Zinc Corp of America	OK	x									
			Zinc	Spent surface impoundment liquids	Zinc Corp of America, Monaca	PA	x									