This chapter describes the screening level risk assessment and benefits assessment conducted for the mineral processing waste LDRs. The focus of this discussion is on the quantitative aspects of the analysis, although qualitative descriptions of regulatory benefits that could not be estimated quantitatively also are provided.

The end products of the quantitative portion of the benefits assessment are estimates of how the levels of health risks associated with waste disposal would change from the pre-LDR baseline situation to post-LDR regulatory conditions. These benefits are summarized in terms of changes in the numbers of facilities disposing of the various wastes (facility-waste stream combinations) at which the calculated health risks exceed selected levels.

Because of the limited data available supporting the risk and benefits analysis (particularly constituent concentration data), these analyses were performed using two different methods to generalize from the limited data to the universe of wastes and facilities affected by the regulatory options. The first approach–the mean-concentration approach–used the arithmetic average constituent concentrations as the basis for evaluating the risks for each waste stream and for extrapolating risk levels to all facilities disposing of a given waste stream. Based on sensitivity analyses of these results, which showed a wide range of variability in key constituent concentrations in many waste streams (and in the resultant risks), a supplemental analysis was performed. This latter approach–the sample-specific approach–used the individual waste sample concentrations to evaluate the risks from disposal of the various waste streams. Similar to the first approach, sample-specific risk assessment results were then used to extrapolate regulatory benefits across all disposal facilities. These different approaches to evaluating waste disposal risks, and to estimating regulatory benefits, are described in detail in the following sections.

The major steps in the quantitative screening risk assessment and benefits estimation were:[1]

- Initial data preparation, including the identification of waste streams for inclusion in the benefits assessment and development of hazardous waste constituent concentration data for the waste streams;

- Estimation of representative release concentrations for each sample, facility, and waste stream with adequate data;

- Characterization of environmental transport (dilution and attenuation in groundwater) of hazardous constituent releases from waste management units (e.g., surface impoundments and landfills) to potential exposure points;

- Estimation of toxic constituent intakes and individual cancer and noncancer health risks for hypothetical "central tendency" (CT) or "high end" (HE) exposed receptors characterized on the basis of release and transport assumptions;

- Screening calculation of risks for pre-LDR conditions using mean release concentration estimates for each waste stream as an initial step (sample-specific and facility-specific screening risk calculations are described below), and for the post-LDR conditions (Regulatory Options 1 and 2) assuming treated wastes would meet UTS requirements;

- Evaluation of the benefits of Regulatory Options 1 and 2 in terms of the estimated numbers of facilities where significant reductions in estimated health risks occur under the two options;

[1] The risk analysis conducted for this benefits assessment follows the recently provided EPA guidance: EPA, "EPA Risk Characterization Program," Memorandum from Carol M. Browner, March 1995.

- Evaluation of the sensitivity of the first approach to the screening risk assessment and benefits analysis to uncertainties in waste constituent concentrations and fate and transport assumptions;

- Based on the results of the sensitivity analysis, screening calculation of pre- and post-LDR risks and benefits of Regulatory Options 1 and 2 using sample-specific concentration data rather than mean waste stream concentration values;

- Comparison of mean-concentration and sample-specific concentration screening risk and benefits results; and

- Summarization of qualitative benefits of all four regulatory options, and discussion of the limitations of the screening risk and benefits assessments.

The remainder of this chapter is organized into five sections. **Section 5.1** discusses the selection of waste streams for inclusion in the benefits analysis, the approaches used to estimate waste constituent concentrations and to estimate the numbers of facilities affected by LDRs in the mean-concentration screening risk analysis and benefits assessments. The assumptions made regarding waste treatment and disposal technologies underlying the two baseline scenarios are described. The coverage of the benefits assessment relative to the cost and economic impact analysis is also discussed. **Section 5.2** describes the mean-concentration screening risk methodology (the first of the two methods used to estimate risks and evaluate benefits), and describes how it has been applied to the prior treatment and no prior treatment baseline scenarios, and to Regulatory Options 1 and 2. **Section 5.3** summarizes the results of the mean-concentration screening risk assessment, and the changes in individual risks associated with the disposal of the various waste streams and constituents under the two baselines and the regulatory options. **Section 5.4** summarizes the benefits of the regulatory options (numbers of facility-waste stream combinations with significant changes in mean-concentration risks), estimated using mean constituent concentrations, in comparison to the baseline scenarios. The potential sources and magnitude of uncertainty in the mean-concentration screening risk and benefits are also described. The results of this sensitivity analysis led to the further investigations described in **Section 5.5,** which includes screening risk and benefits analyses conducted using constituent concentration data from individual waste samples, rather than using mean-concentration values. EPA also compares the results of the mean-concentration and sample-specific assessments in this section.

5.1 DATA ANALYSIS AND DEFINITION OF BASELINE SCENARIOS FOR THE MEAN-CONCENTRATION SCREENING RISK ASSESSMENT

The benefits assessment depends, first, on the ability to identify and characterize the wastes that would be affected by the proposed regulatory options. Benefits of the regulatory options for mineral processing waste LDRs are then estimated for this universe of wastes in reference to two sets of "baseline" assumptions regarding the current management practices for these wastes. This section describes the methods used to identify wastes for inclusion in the benefits assessment, the methods used to estimate representative constituent concentrations for screening risk assessment purposes, and the basic features of the two pre-regulatory baselines.

5.1.1 Constituent Concentrations, Waste Quantities, and Numbers of Facilities

The initial steps in the screening risk assessment and benefits analysis were aimed at identifying the waste streams and facilities that would be included in the quantitative screening risk assessment and benefits analysis, characterizing the concentrations of toxic constituents in the various wastes streams, and developing estimates of waste quantities for purposes of the screening risk assessment. The initial methodology that was employed used representative constituent concentrations to evaluate the risks associated with the disposal of each waste stream. Where multiple waste samples were available, arithmetic mean constituent concentrations were used as an estimate of pre-LDR release concentrations (see below). Since the concentrations of specific constituents were the major criteria used to determine whether waste streams and facilities were included in the quantitative benefits assessment, the first two activities were conducted in parallel. The basic steps in the initial data analysis, as indicated in the numbered nodes on **Exhibit 5-1**, were as follows.

Step 1. EPA first obtained concentration data for each of the potentially hazardous mineral processing waste streams subject to this LDR rulemaking. These waste streams include wastewaters (WW; <1 percent solids), liquid nonwastewaters (LNWW; 1 to 10 percent solids), and nonwastewaters (NWW; >10 percent solids). Bulk concentration data were used for WW wastes, and leachate concentration data were used for LNWW and NWW wastes. The concentration data were obtained from RTI surveys and EPA development documents for effluent limitations guidelines (see **Appendix H**).[2] The majority of these wastes are TC metal wastes (D004-D011). In addition, other waste streams that are not TC for metals, e.g., those that are only ignitible, corrosive, and/or reactive (D001, D002, D003) and/or organic TC (D018-D043) may also pose risks. However, quantitative risk assessment generally is not possible for the first category of wastes due to a lack of concentration data. The organic and pesticide wastes may also pose risks, but because of the much lower volume and concentrations of these wastes compared to the TC metals wastes, no quantitative risk assessment was performed. However, the potential risks associated with non-TC metal waste streams are addressed qualitatively for the baselines and regulatory options.

Step 2. EPA then examined the results of the qualitative and quantitative data reviews described in Chapter 3 in order to characterize the degree of certainty that the waste stream is hazardous according to the TC criteria. If a waste stream was categorized as "Y" (indicating high

[2] In calculating constituent concentrations from bulk concentration or leachate data, the arithmetic means of all the reported concentrations were used, and non-detect values were included in the concentration calculations as values equal to one-half the detection limits. Most sample data were available from only one or two waste streams or facilities, which may contribute significant uncertainty to the analysis.

EXHIBIT 5-1

Graphic Not Available.

likelihood for being TC hazardous), the total number of facilities generating the waste stream[3] was used directly in the benefits analysis without adjustment. (This step does not affect the quantitative screening risk characterization for the waste, but only the benefits assessment, where the numbers of facilities where risks will change under the regulatory options is estimated, as discussed in **Section 5.4**.) For waste streams categorized as "Y?" (indicating a lower likelihood of being TC hazardous) the number of facilities used in the benefits estimation was reduced by 50 percent under the CT benefits assessment. The number of facilities used in the HE benefits analysis was still equal to the entire universe of facilities generating the wastes. This approach may have resulted in over- or under-estimation of regulatory benefits in the CT case, depending on what proportion of wastes categorized as "Y?" actually exceeded the TC concentrations. This approach likely overestimates benefits for the HE case because it includes all facilities, when some are likely to generate wastes that are not TC hazardous. This approach parallels the approach taken to selecting facilities for inclusion in the cost analysis, as described in **Chapter 4**.

Step 3. Several waste streams that are suspected of being hazardous (i.e., indicated by "Y?") do not have constituent concentration data. Because concentration data are needed to quantify risks, EPA only addressed these waste streams qualitatively. See **Section 5.4.3** for a discussion of the benefits potentially resulting from the regulation of these waste streams.

Step 4. The quantities of wastes generated at each facility determines the value of the dilution and attenuation factor (DAF) value used in the exposure assessment and risk characterization (see **Section 5.2.1.1** and **Appendix I**). The quantities of each waste stream used in the fate and transport analysis were adjusted to account for the uncertainty associated with the degree of recycling currently taking place. This approach is again analogous to the procedure employed in the cost analysis. If a waste was identified in the data base as not being recycled at all (i.e., indicated by "N"), then all (100 percent) of the waste is assumed to be treated and disposed and thus was included in the benefits analysis. If wastes are classified as "Y" (high likelihood of full recycling) then 20 percent of the waste is included in the benefits analysis.[4] If a waste is categorized as "Y?" (possibly fully recycled), then 50 percent is included in the analysis. If the waste is categorized as "YS" (high likelihood of being partially recycled) or "YS?" (possibly partially recycled), then 80 percent is included in the analysis. As was the case for the adjustments to the number of facilities in the previous two steps, this approach introduces uncertainty into the numerical risk and benefits estimates. Benefits may either be over- or under-estimated depending on the actual amounts which are recycled.

The major inputs to the mean-concentration screening risk estimation are the identities and average concentrations of the waste contaminants and leachate species, and the amounts of wastes being generated (which are used in the fate and transport analysis). Thus, after these four steps were completed, the major waste-related input data to the mean-concentration screening risk estimation had been developed. The following waste constituents were included in the screening risk modeling for the health benefits analysis: aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, and zinc. The Agency did not have concentration data for every constituent for each waste stream. For some wastes, data were not available or were available for only a few constituents.

The activities described above also produced the remaining major input to the benefits analysis, namely, estimates of the numbers of facilities that would be generating/disposing of the various wastes streams under CT and HE assumptions under the baseline and regulatory options.

5.1.2 Definition of the Baseline Scenarios

As noted in Chapter 4, this RIA measures the benefits of the regulatory options for mineral processing wastes against two sets of baseline assumptions. These sets of assumptions have been defined as the "prior treatment baseline" and the "no prior treatment baseline". These two sets of assumptions were developed based on available information regarding the current management practices for the various mineral processing wastes potentially affected by the regulatory options. Two sets of baseline assumptions

[3] See Chapter 3 for a description of how the total numbers of facilities generation specific wastes streams were estimated.

[4] In the sample-specific approach, this assumption is reduced to zero percent.

have been developed because of the high level of uncertainty about the current management practices for specific waste streams.

The **prior treatment baseline** assumes that all mineral processing wastes would be treated to meet the TC leachate concentration requirements[5] (primarily through neutralization and dewatering, as appropriate, and stabilization), even in the absence of new regulatory initiatives. Because the UTS and TCLP concentrations for most inorganic analytes are so similar, this is essentially equivalent–from a risk screening perspective–to the post-regulatory waste management practices we have assumed (compliance with the UTS) under regulatory Options 1 and 2 for most of the treated wastes. Thus, the benefits of either of the options, measured against the prior treatment baseline, will be relatively low.[6]

The **no prior treatment baseline**, on the other hand, employs the assumption that all of the mineral processing wastes currently being generated would be disposed without treatment in land-based units (surface impoundments or waste piles)[7] in the absence of regulatory changes. In contrast to the prior treatment baseline, this set of assumptions may overestimate actual baseline risks since some mineral processing wastes are likely being treated currently. The estimated benefits of regulatory options thus are greater under this baseline than under the prior treatment baseline.

These two baselines span the reasonable range (from a risk and cost standpoint) of management practices likely to be employed to manage the various wastes under current regulatory requirements. Actual practice in the affected industries are likely to be somewhere in between these two extremes. The general structure of the risk screening of the prior treatment and no prior treatment baselines are described in **Sections 5.2.1 and 5.2.2**, respectively.

[5] Since LDR requirements are not in effect for these wastes prior to this proposed rule, they would not have to meet UTS concentration requirements.

[6] Under Option 1, however, several additional waste streams (characteristic sludges and byproducts being recycled) would be brought under regulatory control, thus providing additional health benefits. These wastes are not brought under regulatory control under Option 2. As discussed in **Section 5.2**, two of these waste streams for which constituent concentration data are available are included in the quantitative risk assessment and benefits estimation, while the benefits of better management of the other wastes are addressed qualitatively.

[7] Underground injection is another possible baseline disposal technology for some wastes that has not been addressed in this assessment due to a lack of data and models that would support a quantitative risk assessment. The omission of this technology probably results in an underestimation of baseline risks and in the aggregate health benefits of the regulatory options.

5.1.3 Comparison of Coverage of the Risk/Benefits and Cost/Economic Impact Analysis

As noted previously, there is relatively little data related to the concentrations of constituents in the waste streams that may be affected by the mineral processing LDRs. In fact, during implementation of the mean-concentration risk and benefits analysis, concentration data were available for only about 26 percent of the waste streams included in the cost and economic impact analysis. However, in terms of the volumes of wastes covered, the risk and benefits analyses cover a much larger fraction of the wastes addressed in the cost analysis. The risk assessment includes approximately 92 percent of the total minimum waste volume estimate used in the cost analysis, 81 percent of the average volume estimate, and 71 percent of the maximum waste volume used to estimate regulatory costs. In addition, the extent of risk assessment coverage of most of the high-volume wastes is quite high; over 97 percent of the wastes from the three highest-volume commodities (lead, copper, and zinc) is covered. Coverage on some of the lower volume sectors is not nearly as high, and there is no chemical concentration data available for many of the small-volume sectors.

The fact that some mineral processing hazardous wastes are not covered in the risk and benefits assessments biases the results of these analyses. Since the compositions (and hence the toxicities) of the excluded wastes are not known, it is not possible to determine the extent of this bias, but it is clear that the net affect is to underestimate pre-LDR risk and regulatory benefits. Therefore, EPA investigated several options for increasing the coverage, as described in **Section 5.5.1.1.** A detailed breakdown of the extent of the coverage of the screening risk and benefits assessment by sector is given in **Appendix J**.

5.2 MEAN-CONCENTRATION SCREENING RISK ASSESSMENT METHODOLOGY

This section provides a discussion of the methods used to develop quantitative estimates of health risks under the two baselines and under the Regulatory Options 1 and 2 using the mean constituent concentrations in each waste stream as our estimate of the release concentration (i.e., the first of the two approaches used to assess risks). In **Section 5.2.1**, the methods used to evaluate risks associated with the prior treatment baseline and with the two regulatory options are described. (As noted above, the basic methods used to evaluate these three cases are essentially the same.) **Section 5.2.2** discusses the slightly different approach that is taken to evaluate risks for the no prior treatment baseline.

5.2.1 Prior Treatment Baseline and Regulatory Options 1 and 2

This section describes the screening level methods used to quantitatively evaluate the health risks associated with waste disposal activities under the prior treatment baseline scenario and under Regulatory Options 1 and 2.[8] The same methods are used to address risks for all three cases because, as noted in **Section 5.1.2**, the waste selection and treatment technology assumptions under the prior treatment baseline are essentially identical to those under regulatory Options 1 and 2. The only major exceptions to this are the recycled waste streams that are not currently regulated, but that would require source control under Option 1. Constituent concentration data are available for two of these waste streams, and these wastes are included in the screening risk assessment and benefits analysis for Option 1. Potential changes in risks for the remainder of the recycled wastes under baseline conditions and regulatory Options 1 and 2 are evaluated qualitatively, as described in **Section 5.4.** In addition, the risk impacts of Regulatory Options 3 and 4 are discussed qualitatively.

The overall scheme for the mean-concentration screening risk assessment for the prior treatment baseline and regulatory Options 1 and 2 is shown in **Exhibit 5-2**. The methods used to select waste streams for inclusion in the screening risk assessment were discussed in **Section 5.1.2.** Once the waste streams were chosen for inclusion, a screening level approach was used to evaluate risks associated with the management of each waste stream under baseline and regulated scenarios. To estimate release concentrations, it was assumed that all wastes would be treated to achieve TCLP (in the baseline) or UTS (under Options 1 and 2)

[8] This quantitative analysis does not address the health risks associated with other aspects of waste management, such as the risks associated with land-based reclamation or other recycling activities under either the baseline cases or Options 1 and 2. The risks associated with these activities are addressed qualitatively in **Section 5.4.3**.

concentrations[9]. For wastewaters and liquid non-wastewaters, the treatment train would involve dewatering and stabilization. For non-wastewaters, only stabilization would be employed.

Exposures to waste constituents were assessed using a generic groundwater exposure pathway model (described below) that characterized exposures faced by hypothetical individuals exposed to contaminated groundwater near waste disposal facilities. The outputs of the screening risk analysis were estimates of individual lifetime incremental cancer risks, as well as hazard quotients indicating the potential for the occurrence of noncancer adverse effects. These individual risk estimates served as inputs to the benefits estimation, as described in **Section 5.4.1.** Health risks were estimated assuming that the ingestion of contaminated groundwater would be the only significant exposure pathway affected by the management of these wastes under the baseline assumptions and the various regulatory options. Other pathways, such as inhalation of particulate or vapors, ingestion of surface water contaminated by runoff or groundwater discharge, use of groundwater for crop irrigation, and consumption of game fish in nearby surface water bodies, were not evaluated. Given the predominant composition of the mineral processing wastes, risks from inhalation exposures to volatile contaminants are not likely to be significant. In addition, individual risks arising from groundwater discharge to surface water are also likely to be much lower than the risks from the direct ingestion of groundwater. Under some circumstances, however, particularly under the no prior treatment baseline, human exposures and risks due to particulates released to the air and due to runoff-contaminated surface water might be significant. Omission of these pathways from the risk assessment may have resulted in an underestimation of baseline risks and in a corresponding underestimate of regulatory benefits relative to the no prior treatment baseline.

Chemical exposures and risks for the hypothetical receptor were calculated using two sets of assumptions reflecting the likely range of variability in groundwater transport parameters and exposure conditions. As discussed below, the first set of estimates (central tendency, or CT estimates) were derived using representative assumptions about groundwater dilution and receptor well locations, while the second set of exposure and risk estimates (high end, or HE estimates) were derived using high-end groundwater dilution and exposure assumptions. For both the CT and HE estimates, the same set of central tendency release, intake, and exposure assumptions were used to estimate risks.

[9] As noted above, these concentrations are generally so similar that there is very little risk reduction under Regulatory Options 1 or 2 in comparison to the prior treatment baseline.

EXHIBIT 5-2

Graphic Not Available.

5.2.1.1 Release and Transport Models

The general approach used to evaluate constituent release concentrations under the prior treatment baseline and regulatory Options 1 and 2 is summarized in **Exhibit 5-2**. Nonwastewater (NWW) streams were assumed to be stabilized prior to disposal. Wastewaters (WW) and liquid nonwastewaters (LNWW) were assumed to be neutralized, dewatered, and then stabilized. For all of the WW, NWW, and LNWWs, it was assumed that stabilization would reduce all leachate constituent concentrations above the UTS levels to their respective UTS concentrations. Thus, if the mean leachate concentrations in the untreated NWW and LNWW streams (or the mean bulk concentrations in WWs) were less than UTS concentrations, these mean values were used directly as estimates of release concentrations from the treated wastes. If the mean bulk or leachate concentrations from the untreated wastes were above UTs levels, then the UTS values were used as estimates of release concentrations for those constituents. See **Appendix I** for a list of the UTS levels for key inorganic analytes.

The structure of the compliance requirements of the LDR treatment standards suggest that average post-treatment leachate concentrations will be lower than the UTS concentrations, and performance data from waste stabilization studies, as well as data from other sources, support this conclusion. None the less, in the mean-concentration screening risk and benefits analysis, EPA used the UTS concentrations as the estimate of post-LDR release concentrations. When sensitivity analyses indicated that the benefits estimates were highly dependent on post-LDR concentration estimates, (**Section 5.4.4**) we developed alternative assumptions about post-LDR release concentrations, and employed them as part of the sample-specific screening risk and benefits analyses which are described in **Section 5.5**.

Exposure via ingestion of groundwater was modeled for each waste constituent. The release and exposure characterization are both time-invariant, using steady-state assumptions about release concentrations and transport conditions. Two sets of dilution-attenuation factor (DAF) values were used to estimate groundwater exposure concentrations. The CT exposure concentrations were estimated using a landfill DAF value of 500. The value of 500, selected from the Toxicity Characteristic Regulatory Impact Analysis,[10] represents the 75th percentile of the probability distribution of DAF values derived for hazardous waste landfills. The 75th percentile DAF was used to take into account possible complex subsurface conditions which could facilitate groundwater transport. The HE landfill DAF values varied, based on the quantities of waste generated. The annual mass of waste generated (in metric tons per year) was converted to volume of waste generated annually (cubic yards per year) assuming an average waste density 2,000 pounds per cubic yard. As noted previously, the dewatered volume of WW and LNWW, rather than the pretreatment volume, was used to estimated HE DAF values. As shown in **Appendix I**, the landfill DAFs range from 12 to 100, with the lowest DAFs assigned to the largest waste volumes.[11] EPA is currently revising the DAF values as part of regulatory development effort for the Hazardous Waste Identification Rule (HWIR), and this screening risk assessment and benefits analysis will be revisited when the revised values become available.

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[10] USEPA, March 1990.

[11] This approach is described in "DAFs for Landfills and Surface Impoundments" in 56 Federal Register 33000 (July 18, 1991); Table 1 of this document provides DAFs as a function of waste volume.

5.2.1.2 Groundwater Intake Estimation

Lifetime individual cancer risks and noncancer hazard quotients were estimated based on the average daily intakes of contaminants at mean concentrations through groundwater ingestion. Four different groundwater pathway contaminant intake estimates were developed in this analysis:

(1) cancer - CT;
(2) cancer - HE;
(3) noncancer - CT; and,
(4) noncancer - HE.

As noted above, the CT intake estimates were calculated using the CT DAF values, while the HE intake estimates were developed using HE DAF values. For both CT and HE exposure estimates, the daily groundwater consumption rate used for intake estimation was 1.4 liters day for a 70-kg adult. The intake frequency was assumed to be 350 days/year over a nine-year period[12] for both cancer and noncancer risk calculations. The exposure averaging time used to estimate daily intakes for cancer risk assessment was 70 years, while the averaging time used to estimate intakes for calculating noncancer hazard quotients was nine years (equal to the exposure period).[13] The DAF values also reflect a range of exposure locations (distance of drinking water wells from the treatment unit boundary). The exposure parameters used to estimate intakes are shown in **Appendix I**.

5.2.1.3 Toxicity Evaluation

EPA used standard toxicity parameter values (ingestion pathway Cancer Slope Factors and chronic noncancer Reference Doses)[14] to derive risk estimates for all of the toxic waste constituents except lead. In the case of lead, the potential for adverse noncancer effects was evaluated by comparing daily intake values to the dose level corresponding to EPA's drinking water Maximum Contaminant Level (MCL) of 15 ug/l in drinking water. The toxicity values used in the screening risk assessment are listed in **Appendix I**.

5.2.1.4 Risk Characterization

Lifetime cancer risk estimates for the hypothetical receptors were derived by multiplying daily lifetime intake of carcinogenic waste constituents by the ingestion pathway Cancer Slope Factors for these constituents.[15] To calculate the total lifetime cancer risks associated with exposure to multiple carcinogenic constituents in the same waste stream, the calculated cancer risks for each constituent were summed. Among all of the waste streams evaluated, arsenic and beryllium were the only constituents for which ingestion-pathway slope factors were available. Thus, all cancer risks in this assessment are due to arsenic and beryllium exposures.

Noncancer hazard quotients were calculated for all of the noncarcinogenic constituents of the waste streams by dividing the average daily dose to the receptor during the exposure period by the chronic ingestion pathway Reference Dose for each constituent. In displaying noncancer risk results for the waste streams, only the highest hazard quotient value among those for all the constituents is given. Hazard quotients were not summed across constituents within waste streams because, unless the target organ and

[12] The nine-year exposure period corresponds to EPA's estimate of the average residential tenure at a single location.

[13] The difference in exposure averaging times reflects the differences in the assumed causative mechanism for cancer and noncancer effects. Cancer risks are believed to be related to exposures averaged over a full lifetime, while noncancer adverse effects may occur in response to exposures over a shorter time period. All of the parameter values relating to pollutant intake estimation come from EPA's Exposure Factors Handbook (1989).

[14] These data were obtained from the IRIS data base (1995) and HEAST (1993).

[15] Cancer risk estimates developed using this procedure were bounded at 1.0.

mode of action for various chemicals are the same, adding hazard quotients is not justified as an indicator of overall risk.

Neither cancer risks nor hazard quotients are summed across waste streams within a commodity, consistent with the assumption that exposure only occurs to one waste stream at a time at a given facility. Furthermore, note that because estimates of both cancer risk and hazard quotient estimates are very uncertain, they are not appropriate for use in making fine distinctions among wastes or regulatory options. Risk results are displayed in terms of order-of-magnitude ranges (10^{-5} to 10^{-4}, for example), and only changes is risk moving from one order of magnitude to another are considered to be "significant", when calculating regulatory benefits. This issue is discussed in more detail in **Section 5.4**.

The results of the mean-concentration screening risk characterization for the prior treatment baseline and regulatory Options 1 and 2 are summarized in **Section 5.3.**

5.2.2 No Prior Treatment Baseline Risk Screening

This section provides a brief discussion of the methods used to calculate health risks for the no prior treatment baseline. For the most part, the methods used are the same as for the prior treatment baseline.

5.2.2.1 Release and Transport Model

Slightly different methods were used to estimate waste constituent release concentrations under the no prior treatment baseline than were used for the prior treatment baseline, as shown in **Exhibit 5-3**. In the case of wastewaters (WW), release concentration estimates were simply the mean bulk constituent concentrations from the reported data for each waste stream. For nonwastewaters (NWW) and liquid nonwastewaters (LNWW), the mean leachate concentrations, rather than bulk concentrations, were used. Unlike the prior treatment baseline and regulatory Options 1 and 2, release concentrations were not limited to the UTS levels.

Groundwater dilution of releases from the no prior treatment case were estimated using DAFs, similar to the approach taken for the prior treatment baseline. As in that analysis, the CT exposure concentrations were estimated by dividing the release concentrations by the CT DAF values, while the HE exposure concentrations were estimated by dividing the release concentrations by the HE DAF values. In the case of the wastewaters, however, the DAF values for surface impoundments were used, rather than the DAF values for landfills, as were used in the prior treatment baseline. The CT DAF value for surface impoundments is 500, the same as that for landfills, while the HE DAF values for surface impoundments range from 100 to 6.0, depending on annual waste flows, as shown in **Appendix I**. The exposure concentrations in groundwater for constituents of LNWWs and NWWs were calculated using the landfill DAF values, the same as for the prior treatment case.

EXHIBIT 5-3

Graphic Not Available.

5.2.2.2 Groundwater Intake, Toxicity Evaluation, and Risk Characterization

The models and parameters used to estimate groundwater intakes of waste constituents, to characterize the toxicity of waste constituents, and to characterize health risks under the no prior treatment baseline are the same as those used to characterize risks under the prior treatment baseline. Thus, the discussion of these methods presented in **Section 5.2.1.2** through **5.2.1.4** are not repeated here.

5.3 RESULTS OF MEAN-CONCENTRATION SCREENING RISK ASSESSMENT

This section discusses changes in the actual screening risk assessment results for the various waste streams under the baselines and under Options 1 and 2 calculated using mean constituent concentrations to estimate pre-LDR no prior treatment releases and the UTS concentrations to estimate post-LDR and prior treatment baseline release concentrations. The risk outputs of these analyses are used in the estimation of quantitative regulatory benefits (numbers of facility-waste stream combinations where risks are reduced below regulatory criteria) described in **Section 5.4.** Non-quantified benefits of Options 1 and 2, as well as the benefits of Options 3 and 4, also are described in **Section 5.4.** Results of the other benefits assessment approach – using sample-specific constituent concentrations to estimate risks and benefits – are presented and discussed in **Section 5.5**.

5.3.1 Risk Screening Results for the Mean-Concentration No Prior Treatment Baseline

Exhibit 5-4 provides a summary of the screening level risk assessment results for mineral processing waste disposal activities under the no prior treatment baseline. (As noted in **Section 5.2**, the prior treatment baseline risk screening results are essentially identical to those for Options 1 and 2, and thus the results for the prior treatment baseline are not discussed independently in this section.) Central tendency (CT) and high-end (HE) baseline risks are shown in the columns labeled "Pre-LDR", with cancer risks on the left half of the table and non-cancer hazard quotients on the right half. Quantitative risk estimates are provided only for those waste streams for which constituent concentration and toxicity data are available. As seen in **Exhibit 5-4**, sufficient data were available for 36 of the non- or partly-recycled materials (for both Options 1 and 2) and two of the fully recycled materials (Option 1 only).

Several observations concerning cancer risks can be made from the results in **Exhibit 5-4**:

- The majority of both the CT (20 of 28) and HE (27 of 38) cancer risk estimates under the mean-concentration no prior treatment case exceeded the 10^{-5} level of concern.

- The commodities/waste streams with the highest calculated mean-concentration CT cancer risks were chip treatment wastewaters from beryllium production, autoclave filtrate from antimony production, and acid plant blowdown from zinc production. The estimated individual cancer risks for these wastes were on the order of 10^{-2} or greater.

- The disposal of six waste streams from various industries were associated with calculated cancer risks in the range of 10^{-3}, with the remainder of the waste stream cancer risks below this level.

EXHIBIT 5-4

Graphic Not Available.

- The highest estimated HE mean-concentration cancer risks under the no prior treatment baseline were in the range of 10^{-1} to 1.0. In addition to the three wastes mentioned above from antimony, beryllium, and zinc production sectors, wastes with the highest estimated cancer risks were acid plant blowdown and spent bleed electrolyte from copper production, acid plant blowdown and process wastewater from lead production, and spent surface impoundment liquids from zinc production.

- The mean-concentration HE cancer risks calculated for nine additional waste streams were in the range of 10^{-3} to 10^{-4}, with the remainder of the waste stream cancer risks below these levels.

Similar to the situation seen for cancer risks, the majority of the no prior treatment baseline non-cancer hazard quotients calculated under both CT and HE assumptions exceeded the level of regulatory concern (in this case, a value of 1.0).

- Under CT mean-concentration assumptions, 23 of the 38 calculated hazard quotient values exceeded 1.0, while 35 of the 38 HE hazard quotients exceeded this level.

- Under CT assumptions, the three waste streams with the highest mean-concentration hazard quotients (approaching or exceeding 1,000) were all from the zinc industry (acid plant blowdown, spent surface impoundment liquids, and wastewater treatment plant liquid effluent). These wastes also had the highest hazard quotients under HE assumptions (values greater than 70,000).

- Other commodities where one or more wastes had mean-concentration HE hazard quotients exceeding 1,000 were antimony production (autoclave filtrate), beryllium production (chip treatment wastewater), copper production (acid plant blowdown and spent bleed electrolyte), elemental phosphorus production (furnace scrubber blowdown), and lead production (acid plant blowdown and process wastewater). In addition, the HE hazard quotient for an additional zinc production waste (process wastewater) also exceeded 1,000.

There does not appear to be a simple pattern of a single industry or type of waste where cancer risks or hazard quotients are highest, although wastes from antimony, beryllium, copper, and zinc production, and to a lesser extent lead production, show up as high-risk industries somewhat more frequently than others. In the case of beryllium production wastes, the finding of high cancer risks is due to beryllium itself being one of only two carcinogenic constituents evaluated.

As noted in **Section 5.2**, the screening risk calculations for the no prior treatment baseline (and for Option 1) include two characteristic materials currently not subject to LDRs. As shown in **Exhibit 5-4**, neither of these two wastes are associated with any of the highest mean-concentration cancer risk or hazard quotient values, although the HE and CT values do exceed the levels of regulatory concern for both of these wastes.

5.3.2 Mean-Concentrations Risk Screening Results for Option 1

The results of the risk screening for Regulatory Option 1 are also summarized in **Exhibit 5-4**. Several observations concerning cancer risks can be made based on these results.

- The estimated cancer risks are substantially reduced for most wastes under Option 1 when compared to the no prior treatment baseline. (This point is made clearer in **Section 5.4.**)

- Under Option 1, the highest CT cancer risks calculated for any of the waste streams are in the range of 10^{-5} to 10^{-4}, compared to the maximum risks of approximately 10^{-2} seen in the no prior treatment baseline case.

- The number of waste streams for which CT risks exceed the 10^{-5} level of concern drops from the 21 of 38 seen in the baseline case to 16 of 38 under Option 1. It should be noted, however, that an additional five waste streams have associated post-LDR CT cancer risk estimates in the range between $1.02x10^{-5}$ and $1.33x10^{-5}$. It is not clear, given the level of uncertainty in the risk estimates, that these risks are actually greater than 10^{-5}. (The potential implications of these findings for the mean-concentration benefits assessment are discussed in **Section 5.4.4.**)

- Under HE assumptions, the highest estimated cancer risks under Option 1 are all less than 10^{-3}, compared to the many risks exceeding 10^{-1} under the no prior treatment baseline.

- The number of waste streams for which HE risks exceed 10^{-5} drops from 27 of 38 under the baseline to 22 of 38 under Option 1. In both the CT and HE cases, the risks dropping from above 10^{-5} to below 10^{-5} were associated with the same five waste streams: antimony production autoclave filtrate, beryllium production bertrandite thickener slurry and chip treatment wastewater, and the two fully-recycled materials for which constituent data were available (lead production wastewater treatment plant effluent and spent goethite and leach cake from zinc production).

- Most of the highest risks identified in the no prior treatment baseline in **Section 5.3.1** stay above 10^{-5} under Option 1. The sole exception is the risk associated with chip treatment wastewaters from beryllium production, which drops from around 10^{-1} in the baseline case to below 10^{-5} under Option 1. The estimated risks for many of the other highest risk wastes are substantially reduced, but not to levels below 10^{-5}. In the case of the HE cancer risks, there are no potential "swing wastes" (wastes with risks very close to 10^{-5}; see **Section 5.4.4.2**) under Option 1 as there are with the CT estimates.

Several observations concerning non-cancer risks can be made based on the results presented in **Exhibit 5-4**.

- In contrast to the situation for cancer risks, the CT mean-concentration hazard quotients for all of the wastes are reduced below 1.0 under Option 1. Thus, none of these wastes pose risks of regulatory concern under Option 1.

- The HE mean-concentration hazard quotients are also substantially reduced, but many (23 of 38) still remain above the level of concern of 1.0.

- Analogous to the case for the CT cancer risks, an additional six of the HE mean-concentration hazard quotients for Option 1 come very close to dropping below the level of concern, ranging in value between 1.0 and 1.5. As was the case for the CT cancer risks, there is a substantial degree of ambiguity as to how to count these wastes for purposes of benefits assessment. This issue is discussed in more detail in **Section 5.4.4.**

- The highest post-LDR mean-concentration hazard quotients are just above ten, compared to the maximum no prior treatment baseline value of 400,000. Wastes in this category include copper production acid plant blowdown and process wastewater, process wastewater and surface impoundment waste liquids from lead production, and zinc production spent surface impoundment liquids.

- As was the case for cancer risk reductions, the mean-concentration non-cancer hazard quotient reductions under Option 1 are highly variable, and not all of the wastes posing the highest risks under the baseline case still pose the highest risks under Option 1.

5.3.3 Mean-Concentration Risk Screening Results for Option 2

As discussed in **Section 5.2**, the only difference between regulatory Options 1 and 2 is that Option 1 would require source control measures for approximately 13 fully-recycled characteristic materials that would not be addressed under Option 2. Thus, the mean-concentration risk screening results for Option 2 are exactly the same as for Option 1, except that the risk (and benefit) contributions of these Option 1 materials would not be included. The bottom panel of **Exhibit 5-4** ("Options 1 and 2") displays the risks excluding the fully-recycled materials. As discussed above, the cancer risks and hazard quotients calculated for the wastes excluded under the Option 2 analysis are not among the highest under either the CT or HE assumptions. However, both the baseline CT and HE mean-concentration cancer risks for these waste streams are above 10^{-5}, and the CT and HE hazard quotients for one of the wastes (WWTP treatment effluent from lead production) are above 1.0. Under the assumptions used in the risk screening, all of these values would drop below levels of concern under Option 1. Thus, the exclusion of these wastes does result in a difference in regulatory benefits between Options 1 and 2, as discussed in **Section 5.4.2.**

5.3.4 Key Uncertainties and Limitation of the Mean-Concentration Screening Risk Assessment

The data and risk assessment methodology used to evaluate the changes in risks associated with the management of mineral wastes under the various LDR options is highly generic and imprecise. Some of the sources of uncertainty in this risk assessment are those that are common to many screening level prospective risk analyses. Others are unique to this study. Among those that are common to screening level analyses in general are:

- The use of highly generic and chemical release and environmental fate and transport models;

- The use of a generic exposure assumptions, a simplified exposure pathway model, and generic exposure factor values;

- The use of toxicological data derived primarily from animal studies to predict human response to pollutant exposures, and the use of simplified toxicological models to evaluate risks; and

- The use of deterministic methods that do not address major sources of uncertainty and variability in the exposure and risk assessment models and parameters.

The use of all of these assumptions and approaches is fully consistent with the screening level nature of this risk assessment. However, the consequence of using these simplified methods is that the quantitative risk estimates for the management of the wastes are highly uncertain, and the level of uncertainty, although large, cannot be estimated reliably. The results of this screening level risk assessment should be interpreted as being no better than order-of-magnitude indicators of risks.

In addition to uncertainty, it is likely that some of the assumptions made will tend to either overestimate or underestimate exposures and risks. Key conservative exposure assumptions include the assumption that groundwater wells would be present near every facility, and the assumption that the contaminated well would supply all of the drinking water for the receptor 350 days per year. The remainder of the exposure factor values are representative values derived from EPA sources, and are not likely to overestimate exposure or risks.

In the case of the toxicological analysis, linear low-dose extrapolation for carcinogens is also generally considered to provide a conservative estimate of risks. Also, the ingestion slope factor values for all of the carcinogenic waste constituents addressed in this assessment are quite uncertain, and their use may also overestimate cancer risks.

The major assumptions that might underestimate risks in this assessment include the inability to estimate non-groundwater pathway risks, and the inability to include wastes for which constituent concentration data are unavailable. Both of these factors would most likely contribute to an underestimation of regulatory benefits as well as risks, since releases of all wastes would be controlled under the regulatory options (even those wastes for which no concentration data are currently available), and because exposures through all pathways are likely to be better controlled under regulation than under the no prior treatment baseline.

In addition to these common sources of uncertainty, there are additional sources of uncertainty that are unique to this screening risk analysis. These include:

- Uncertainty about the chemical composition of wastes and the variability in concentrations of key toxic constituents;
- Uncertainties about the pre- and post-LDR management practices and technologies for specific wastes;
- Uncertainties about the amounts of wastes and waste constituents that would be released from land management units; and
- Uncertainties about site-specific fate and transport pathways and parameter values, and about receptors.

The use of the mean constituent concentration values as release concentration estimates helps to assure that these values will be representative of the reported data taken as a whole, but does not account for the variability in actual waste or leachate concentrations. The variability in constituent concentrations within some of the waste stream samples is actually very large, and the composition of some waste streams is quite heterogeneous. Analytical results from some samples and/or facilities show high levels of constituents that are not present at all, or are present at very low concentrations, in the other samples. The potential uncertainty in the mean-concentration screening risk estimates associated with the variability in waste composition is analyzed in more detail in the sensitivity analyses described in **Section 5.4.4**.

As noted in **Section 5.2**, the uncertainties about groundwater transport are addressed through the use of the "CT" and "HE" DAF values developed by EPA specifically for this purpose. The uncertainties associated with a lack of knowledge of pre-LDR management practices are addressed by defining the two baseline cases, which span the range of likely management practices, at least at an aggregate level. The uncertainty regarding the nature and location of receptors is resolved, as indicated above, by using EPA's previous analysis of the average distance to the nearest well, and by defining a generic adult resident exposure scenario. Thus, as appropriate for a screening analysis, uncertainty is addressed either through the use of defensible generic assumptions, or through the use of multiple exposure scenarios spanning the likely range of exposure conditions.

5.4 MEAN-CONCENTRATION AND QUALITATIVE BENEFITS ASSESSMENT

This section presents a brief description of how the benefits of the proposed regulatory options have been estimated, based on the mean-concentration screening risk estimates (i.e., using the first of the two general approaches), followed by a summary of the estimated regulatory benefits of the four options.

5.4.1 Mean-Concentration Benefits Assessment Methodology

The quantitative benefits assessment develops estimates of the number of facilities at which risks to human health will be reduced under regulatory Options 1 and 2, compared to the baseline conditions. The inputs to the benefits estimation are the numbers of facilities generating and managing the various waste streams (estimated as discussed in **Section 5.1.1**), and the estimated individual groundwater pathway risks associated with management of the waste streams, which were discussed in **Section 5.3.** Benefits are estimated by comparing the numbers of facility-waste stream combinations that fall into order-of-magnitude risk categories in the pre- and post-LDR settings.[16] Tables and histograms of the changes in the risk distributions across facility/waste stream combinations are provided in the following sections.

Of particular concern to EPA are the numbers of facility-waste stream combinations at which risks exceed levels of regulatory concern under the baseline assumptions and regulatory options. In this analysis, an individual cancer risk level of 10^{-5} and a noncancer hazard quotient value of 1.0 have been selected as levels indicative of potential concern. The numbers of facilities at which the mean-concentration risks associated with the disposal of individual waste streams exceed these levels under baseline conditions (pre-LDR) are calculated, as well as the numbers of facilities at which waste disposal risks are reduced below these levels by the regulatory options (post-LDR).

As discussed previously, the prior treatment baseline is essentially equivalent to the post-LDR treatment requirements under Option 1, except that this option brings under regulation a small number of fully-recycled characteristic sludges and byproducts that are not currently subject to LDRs. Thus, there are no estimated benefits of Option 1 relative to the prior treatment baseline, except those arising from the control of the newly-regulated waste streams. Option 2 and the prior treatment baseline are essentially identical, and thus no quantitative benefits are estimated for Option 2 relative to this baseline. For these reasons, the focus of the quantitative benefits assessment is on benefits relative to the no prior treatment baseline.

In addition to the quantitatively estimated benefits, there are also benefits of the various regulatory options that must be characterized qualitatively. Since no quantitative risk assessment has been performed for Options 3 and 4, all of the health risk benefits for these options fall into that category.

5.4.2 Results of Mean-Concentration Quantitative Benefits Assessment for Regulatory Options

This section describes the results of the quantitative benefits estimation, first reviewing the estimated numbers of facility-waste stream combinations counted in the benefits methodology, and then summarizing how the distributions of facility-waste stream combinations by risk level change from the baseline case under the imposition of Options 1 and 2.

5.4.2.1 Estimated Numbers of Facility/Waste Stream Combinations

The numbers of facilities that generate and manage specific hazardous waste were estimated based on constituent concentration data, as described in **Section 5.1.2.** In some cases (for all HE estimates), the numbers of facilities used in the benefits estimation is simply the total number of facilities generating the wastes. For some wastes streams, the number of facilities has been adjusted (reduced by 50 percent) in the CT case to take into account uncertainty about what proportions of some of the waste stream actually is TC hazardous.

The numbers of facility-waste stream combinations included in this analysis are shown, by risk category, in **Exhibit 5-5a**. As seen in this exhibit, the total for Option 1 CT cancer risk is 98; in the HE cancer risk estimation, the total number is 113. The numbers of facility-waste stream combinations in the CT and HE noncancer hazard index estimation, respectively, are 136 and 157. The differences in these numbers reflect differences in the numbers of facilities generating carcinogenic and noncarcinogenic wastes, and in the CT and HE estimates of the facilities generating wastes that are likely to be TC hazardous. The numbers of facility/waste stream combinations included in the benefits estimates under Option 2 (lower panel of **Exhibit 5-5a**) are slightly lower, since the facilities generating the two fully-recycled characteristic sludges and byproducts are not included in the latter estimates.

[16] Changes in risks are characterized in order-of-magnitude increments because the level of uncertainty in the screening risk estimates does not merit more precise treatment.

EXHIBIT 5-5

Graphic Not Available.

Among the waste streams evaluated, the number of generating facilities varies greatly. The waste stream associated by far with the greatest number of generating facilities (23) is cast house dust from aluminum and alumina production (see **Exhibit 5-4**). The next highest number of facilities (11) is associated with two waste streams from titanium/titanium dioxide production. Three additional wastes from titanium production and three waste streams from copper production have the next highest estimated HE number of facilities at 10 each. In the case of the titanium wastes, the CT estimate of the number of facilities has been adjusted downward to five. One other copper production waste is generated at nine facilities, and the remainder of the wastes are generated at seven or fewer facilities. Nineteen of the 38 waste streams for which concentration data are available are generated at three or fewer facilities.

5.4.2.2 Quantified Mean-Concentration Health Benefits Under Option 1

The numbers of facility-waste stream combinations for which estimated mean-concentration cancer risks and hazard indices fall into specified ranges under the no prior treatment baseline and Options 1 and 2 are summarized in **Exhibit 5-5a**. These results are reproduced in the form of histograms in **Exhibit 5-5b**.

Under Option 1, the estimated numbers of facility-waste stream combinations with CT cancer risks greater than 10^{-5} is reduced only slightly, from 68 out of 98 to 62 out of 98. The number of facility-waste stream combinations with CT cancer risks <u>less</u> than 10^{-5} correspondingly increases from 30 to 36. It can be seen, however, that the numbers of facility/waste stream combinations with high CT cancer risks (greater than 10^{-4}) drops dramatically, from 45 facility-waste stream combinations to zero, under Option 1.

The specific waste streams for which CT disposal cancer risks decrease from above 10^{-5} to below this level are spent goethite and leach cake from zinc production (three facilities) and three waste streams from beryllium production, (one facility each)[17]. CT cancer risks did not decrease below 10^{-5} for any of the waste streams generated at the large numbers of facilities that were identified in **Section 5.4.2.1.** However, the estimated CT cancer risks from five additional waste streams, from an additional 19 facilities, were reduced to levels just above 10^{-5} (between $1.09\text{x}10^{-5}$ and $1.33\text{x}10^{-5}$) under Option 1. Relatively small changes in the CT risk estimates for these wastes would greatly increase the number of facility/waste stream combinations where risks were reduced to below levels of concern, and correspondingly increase the estimated CT regulatory benefits. There were far fewer instances where risks were reduced to just below levels of concern under Option 1, and less potential for overestimation of benefits. The issue of these potential "swing wastes" is discussed in more detail in **Section 5.4.4.2**.

The number of facility/waste stream combinations with HE mean-concentration cancer estimates greater than 10^{-5} drops from 106 of 113 to 98 of 113 under Option 1, with a similar decrease in the numbers of facility/waste stream combinations with the highest risks as was seen in the CT case. The HE benefits of Option 1 (reduced numbers of facility/waste stream combinations with HE cancer risks greater than 10^{-5}) were accounted for by titanium production scrap milling waste (10 facilities), three beryllium production wastes (four facilities total), and by the non-Bevill zinc production waste noted above (three facilities). In the case of the HE cancer risks, there were no potential "swing wastes" (facility-waste stream combinations with Option 1 risks just above or below 10^{-5}).

The numbers of facility-waste stream combinations with CT noncancer hazard quotients greater than 1.0 decreases dramatically from the no prior treatment baseline to Option 1. Under the baseline case 71 of 136 facility/waste stream combinations had CT hazard quotients greater then 1.0. Under Option 1, this number was reduced to zero. Obviously, many waste streams contributed to this reduction. The greatest numbers of facility/waste stream combinations where CT hazard quotients decreased from above 1.0 to below 1.0 were three titanium/titanium dioxide production wastes (10, 10, and 11 facilities, respectively), and copper production scrubber and acid plant blowdown (ten and nine facilities, respectively).

The number of facility-waste stream combinations with mean-concentration HE hazard quotients greater than 1.0 decreased less dramatically than the corresponding numbers of CT estimates, from 151 of 157 facility-waste stream combinations to 100 of 157. The wastes contributing the most to this reduction were cast house dust from aluminum/alumina production (23 facilities), and two waste streams from

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[17] Note that these waste streams may well have been generated at the same facility. This illustrates why the total number of facilities generating wastes and the total numbers of facility-waste stream combinations used to aggregate benefits are not the same.

titanium production (ten facilities each). In addition, similar to the situation for the CT cancer risks, there were three more waste streams (one from copper production and two from titanium production), accounting for 32 facility/waste stream combinations, where the post-LDR HE hazard quotients were decreased to levels just above 1.0 (between 1.006 and 1.36). If these risk results had been lower by even a few percent, the estimated HE benefits of Option 1 (in terms of the number of facility/waste stream combinations with noncancer risks reduced to acceptable levels) would have been changed (increased) substantially. There are far fewer potential "swings" in the other direction (facility/waste stream combinations with HE hazard quotients just below 1.0). The issue of potential swings in waste streams from one risk category to another is discussed in more detail in **Section 5.4.4.2**.

5.4.2.3 Quantified Mean-Concentration Health Benefits Under Option 2

Analogous to the situation for the risk screening, the only difference between the quantitative regulatory benefits between Options 1 and 2 is the inclusion of the fully-recycled materials in the former Option, but not in the latter. Thus, the benefits for Option 2, shown in the bottom panel of **Exhibit 5-5a**, differ only with regard to the risks and numbers of facilities associated with these waste streams.

The first waste stream, wastewater treatment plant liquid effluents, is estimated to affect benefits at four facilities under both CT and HE assumptions. While this waste does not pose any carcinogenic risk, both the CT and HE noncancer hazard quotients under baseline assumptions exceed 1.0, and these values are reduced below 1.0 under Option 1. In the case of the other waste (spent goethite and leach cake residues from zinc production, three facilities), both the CT and HE cancer risks exceed 10^{-5}, and the HE hazard index exceeds 1.0. Since these values are also reduced to acceptable levels under Option 1, additional regulatory benefits are realized by that option through reduced cancer risks.

In the case of CT mean-concentration risk estimates, the fully-recycled materials account for one-half (three of the six facilities) at which risks are reduced from above 10^{-5} to below this level under Option 1. Thus, from the point of view of CT cancer risk reduction, Options 1 and 2 differ substantially. The actual difference may be greater, since constituent concentration data were available for only two of the 13 fully-recycled materials. The fully recycled materials account for a smaller, but still substantial, proportion (three of eight facilities) at which HE cancer risks are reduced to from above to below levels of regulatory concern under Option 1.

In the case of noncancer hazard quotients, the fully-recycled materials account for a far smaller proportion of the total numbers of facility/waste stream combinations where CT hazard quotients are reduced to acceptable levels. Of the 71 facilities where mean-concentration CT hazard indices are reduced from above 1.0 to below 1.0 (under Option 1), the fully-recycled materials account for only four facilities. Where the number of HE mean-concentration hazard indices above 1.0 is decreased by 51 under Option 1, the corresponding decrease under Option 2 is nearly the same, at 44 facility/waste stream combinations.

5.4.3 Qualitative Benefits Assessment for Regulatory Options

The benefits quantified in **Section 5.4.2** represent only a fraction of the total benefits expected from the four regulatory options. Non-quantified benefits include the following:

- Risk reductions from treated waste streams for which EPA does not have concentration data.
- Risk reductions from wastes treated to less than the UTS levels post-LDR.
- Risk reductions from exposure pathways besides ingestion of contaminated groundwater (e.g., ingestion of contaminated drinking water from surface runoff, ingestion of food grown using contaminated water, inhalation of particulates).
- Risk reductions from receptors besides adult humans, such as children, other sensitive individuals, and non-human receptors and ecosystems.
- Increase in non-use values (i.e., "existence value").

Several of these benefits are listed in **Exhibit 5-6**. Potential additional risk reductions associated with post-LDR treatment of wastes to below UTS levels are addressed in the sample-specific screening risk analysis discussed in **Section 5.5**.

5.4.4 Key Uncertainties and Limitations of the Mean-Concentration Benefits Assessment

This section identifies potential major sources of uncertainties in the mean-concentration benefits assessment. The sources of uncertainty are discussed qualitatively in **Section 5.4.4.1**, and the results of sensitivity analyses of the benefits assessments to variability in constituent concentrations (a major source of uncertainty) are summarized in **Section 5.4.4.2**.

5.4.4.1 Major Sources of Uncertainty in the Mean-Concentration Benefits Assessment

Since the mean-concentration benefits analysis is based on the results of the mean-concentration screening risk assessment, the major sources of uncertainty in the risk analysis (see **Section 5.3.5**) are also major sources of uncertainty in the benefits analysis. Consequently, the no better than order-of-magnitude precision of the risk estimates carries over into the benefits estimates, limiting the precision with which facility-waste stream combinations can be classified into categories with regard to disposal risk. A major source of uncertainty in the mean-concentration benefits estimates that is most amenable to quantitative sensitivity analysis is the known variability in toxic constituent concentrations within waste streams. This issue is addressed in **Section 5.4.4.2** below.

EXHIBIT 5-6

Graphic Not Available.

As discussed in **Section 5.1.1**, there is also uncertainty associated with the procedures used to estimate the number of facilities managing hazardous mineral processing wastes. These estimates serve as the other main input to the benefits analysis. However, these methods affect only the CT benefits estimates, since the unadjusted number of facilities from the survey data are used to estimate benefits in the HE case. Thus, the accuracy of the estimates of numbers of facilities generating wastes are ultimately limited by the quality of the survey data.

Because of the uncertainty in the screening risk estimates and in the estimates of the numbers of facilities generating wastes, the shifts in the distributions of facilities at different risk levels shown in **Exhibits 5-4 and 5-5** must be interpreted cautiously. Any given waste (and the corresponding group of facilities) might easily be misclassified with regard to its level of risk by an order of magnitude or more. There is also uncertainty associated with the estimated numbers of facilities at any given risk level under both the baseline and regulatory options. However, the use of consistent assumptions about exposures and risks across the entire universe of wastes helps to assure that the ranking of the waste streams by risks is relatively consistent, and that the aggregate shifts in risks from the baseline case to post-LDR conditions are reasonably represented.

5.4.4.2 Sensitivity Analysis of Mean-Concentration Benefits Estimates to Variability in Constituent Concentrations

As noted in **Section 5.3**, the variability of constituent concentrations within some of the waste streams is quite high, and the waste stream concentration data for some streams are very heterogeneous with regard to the presence or absence of specific constituents, and with regard to the relative concentrations of the constituents. Thus, uncertainty in the pre-LDR concentration data has been identified as a major potential source of uncertainty in the screening risk assessment, and by extension, in the benefits assessment. To investigate the potential magnitude of the impact of constituent concentration variability on the mean-concentration screening risk and benefits estimates, sensitivity analyses were performed.

The exact methods used and the results of the sensitivity analyses of the mean-concentration screening risk and benefits assessments are described in detail in **Appendix J**. First, potential "swing" waste streams were identified. Swing wastes are defined as those waste streams for which a relatively small change in estimated pre-LDR and/or post-LDR cancer risk or noncancer hazard quotients would results in changes in the benefits estimates for the regulatory options. These swings would take the form of facility-waste stream combinations moving from just above to just below levels of regulatory concern, and vice verse. Potential swing wastes were identified, and the impacts of specified changes in risk estimates (plus or minus a factor of two or a factor of ten) on the benefits estimates were evaluated. The results of this analysis are summarized in Exhibit 2 of **Appendix J**.

Next, the potential variability in risk and hazard index estimates associated with the observed variability in pre-LDR constituent concentrations was investigated. The variabilities in the concentrations of key constituents (the constituents driving the risk estimates) were evaluated for potential swing wastes identified in the previous step, and for the waste streams with the highest pre-LDR risks, using concentration data from the waste stream data base. Risks were calculated for the maximum and minimum constituent concentrations, as well as for the mean concentrations, and the impacts on the benefits estimates (facility-waste streams moving above or below levels of regulatory concern) were evaluated. The results of this analysis are summarized in Exhibits 4 and 5 of **Appendix J**.

The overall findings of the sensitivity analysis of the mean-concentration benefits assessment can be summarized as follows:

- Numerous potential swing waste streams were identified. These wastes were generated at large numbers of facilities, and their pre- or post-LDR mean concentration risk estimates were close enough to levels of regulatory concern (just above or just below) so that small changes in risk estimates could result in large changes in the numbers of facilities above or below levels of concern.

- If risk estimates for the waste streams were off by as little as a factor of two (over- or under-estimated), substantial changes in the estimated benefits of the regulatory options would occur. For example, the number of facility-waste stream combinations estimated to have pre-LDR CT cancer risks less than 10^{-5} under Option 1 in **Exhibit 5-5a** (30), could range from as low as 26 to as high as 36 if the risk estimates were incorrect by a factor of two or less. Similarly, the number of facility-waste stream combinations with pre-LDR HE hazard quotients less than 1.0 in **Exhibit 5-5a** (6) could range from as low as 1 to as high as 41 if the hazard quotients were off by a factor of two. The estimated numbers of facility-waste stream combinations with post-LDR risk and hazard quotient estimates below levels of concern were also found to be very sensitive to small changes in the underlying risk estimates (see Exhibit 3 of **Appendix J**).

- The actual variability in the pre-LDR concentrations of the key constituents in bulk concentration and leachate samples from many waste streams was very large. The variability of key constituent concentrations (the ratios of minima to maxima) often exceeded two orders magnitude. For a large number of waste streams, the variability in concentration data was such that wastes could easily "swing" from above to below risk levels of regulatory concern, depending upon which concentration values were used (Exhibit 5 of **Appendix J)**.

The results of the sensitivity analysis suggest that using the mean concentrations of the key constituents in the waste streams as the sole measure of release concentration for risk assessment purposes may have resulted in benefits assessments different from the results that would be obtained if other concentration measures were used that better capture the variability in these concentrations. Thus, an alternative to the mean-concentration approach for assessing release concentrations was indicated.

Because the concentration data for many waste streams were so highly variable and heterogeneous, simply using another measure of central tendency (such as the median or geometric mean) would also not capture the variations in risk estimates that might occur. Thus, risks and benefits were re-calculated from the raw concentrations data, using sample-specific constituent concentrations and constituent concentrations from single facilities, in order to provide an alternative measure of regulatory benefits. This analysis is discussed below in **Section 5.5**.

5.5 SAMPLE-SPECIFIC SCREENING RISK AND BENEFITS ESTIMATION

This section provides a description of the methods used to perform the sample-specific screening risk and benefits analysis (the second of the two approaches), and a summary of the results of those analyses. As discussed in **Section 5.4.4.2**, sensitivity analyses indicated that there was a substantial degree of uncertainty associated with the use of mean-concentration data for estimating pre-LDR risks, and with the use of the UTS concentrations for estimating post-LDR risks. The analyses and results described below address these concerns.

5.5.1 Methodology of Sample-Specific Screening Risk Assessment and Benefits Analysis

There are four main differences between the sample-specific analysis and the mean-concentration analyses described above. The first major difference is both the addition and removal of several waste streams. Wastes were added after a re-evaluation of the chemical concentration data indicated that sufficient data were available to support screening risk and benefits analyses (see **Section 5.5.1.1**); wastes were removed after re-working the assumptions concerning recycling. The next two differences are in the methods used in the pre-LDR risk estimation for the various waste streams and in the approaches used to estimate the numbers of waste stream-facility combinations falling into different pre-LDR risk categories. The fourth and final difference relates to how the post-LDR release concentrations are estimated for use in the risk and benefits assessments. The following sections discuss these issues in turn.

5.5.1.1 Addition and Removal of Waste Streams

As described in **Section 5.1.3**, many of the potentially hazardous mineral processing waste streams analyzed for cost were not initially analyzed for risk because the Agency did not have adequate constituent concentration data. EPA therefore investigated several options for filling this data gap. One option involved estimating (interpolating) constituent concentrations using waste streams with concentration data as proxies. However, EPA rejected this approach because of the significant–but unmeasurable–differences known to exist between the mineral processing waste streams. For example, waste streams with concentration data may be likely to have higher concentrations of constituents than waste streams without data because of "selection bias" toward data on the more contaminated waste streams. This bias is inherent to the type of data compilation used in this analysis, yet the degree of the bias is unknown and therefore would be difficult to correct during interpolation.

A second option, which subsequently was implemented, involved modifying the decision criteria used previously for determining the acceptability for analysis of the concentration data (see **Section 5.1.1**). That is, rather than using only leachate data for LNWW and NWW and only bulk concentration data for WW, EPA used a hierarchical approach for each waste stream, which employed the following steps:

(1) Apply the previous criteria (i.e., use bulk data for WW, leachate data for LNWW and NWW in the risk assessment) where possible, thereby obtaining the most relevant concentration data available for each waste stream.

(2) If this first step indicates that concentration data of the preferred type is not available for the waste, but concentration data of another type is available (e.g., bulk data for NWW), then

- for WW, use the leachate concentrations;
- for LNWW, use the bulk concentrations; and
- for NWW, use 5 percent of the bulk concentrations based on the dilution that occurs during the TCLP (see **Section 3.3** for a more detailed discussion on this approach).

If, after these steps, no concentration data is available for the waste stream, then it is not included in the quantitative screening risk analysis, similar to the approach in the mean-concentration screening risk and benefits assessment. This approach resulted in nine waste streams being added to the quantitative risk assessment.

EPA also slightly adjusted the analytical assumptions concerning recycled wastes. That is, rather than assuming that 20 percent of the high-probability (i.e., "Y") fully recycled wastes are treated and disposed, zero percent was assumed. This new assumption resulted in several waste streams being removed from the quantitative portion of the analysis, including the fully recycled waste streams analyzed in Option 1 (which thus eliminated any differences between Options 1 and 2 in the quantitative analysis).

The end result of these changes is the addition of six waste streams to the analysis, bring the total number of waste streams quantitatively analyzed to 42.

5.5.1.2 Sample-Specific Pre-LDR Screening Risk Estimation Methodology

Under this approach, pre-LDR screening risk estimates are developed for each individual waste sample from each waste stream, rather than using the mean constituent concentrations derived from all of the samples from that waste stream. Thus, for each waste stream, there are now "N" risk estimates, where "N" is the number of chemical analyses reported in the constituent concentration data base.

The waste stream constituent data used to develop sample-specific risk estimates is precisely the same data as those used to derive mean-concentration screening risk estimates, with the addition of data on six more waste streams. Approximately 220 sample results are included in the data base for the 42 waste streams now addressed in the benefits analysis. As a first step in the sample-specific screening risk analysis, facility identity information from the original data sources was recombined with the constituent data, so that the extent of intra- and inter-facility variability in constituent concentrations could be investigated, as discussed below. Some of the data, however, are confidential business information (CBI), and therefore the Agency could not disclose the name or location of these facilities. Other data also come from sources that did not identify the facility (although the sources are reliable; e.g., effluent guidelines documents). Some of these unidentified facilities may be the same as some of the identified facilities, which might lead to "double counting" during the sample aggregation step described in **Section 5.5.1.4**. The Agency did not assess this possibility comprehensively, however, because this potential overlap is believed to be either relatively minor or nonexistent based on the likelihood that facilities that provided CBI data–the majority of the unidentified facilities–always provided CBI data, and facilities that provided publicly accessible data always provided publicly accessible data. Facility identity data were available for the vast majority of the samples, confirming the low likelihood of overlap or double counting of samples. The complete data base of facility ID and concentration data is attached as **Appendix K**.

Similar to the mean-concentration approach, cancer risk estimates and estimates of noncancer hazard quotients were developed for each waste sample. Risks and hazard quotients were estimated using exactly the same assumptions about environmental fate and transport, exposure conditions, and constituent toxicity as were used in the mean-concentration screening risk assessment (see **Section 5.2**). Central tendency (CT) and high end (HE) risk estimates were derived for groundwater exposures to toxic constituents in each waste sample using the same CT or HE dilution-attenuation factor (DAF) values as were used in that analysis.

Since not all samples of the various waste streams were analyzed for the same suite of analytes, the risk or hazard quotient results within a waste stream may be driven by different analytes in different samples. (For example, if arsenic and beryllium were not analyzed for in a sample, no cancer risk estimate is developed for that sample, even if these analytes are present in other samples of the same waste stream.) Similarly, non-detect ("ND") results (analytical results where a given analytes is analyzed for, but not present at levels that can be detected) are always included in the sample-specific screening risk calculations at a concentration equal to one-half the detection limit, irrespective of whether the analyte was ever detected in any sample of that waste stream. Thus, there is the potential that risk results for some wastes may be driven by analytes which are never reported as being present in that waste stream, although this is unlikely, given the low levels of risk usually associated with such low concentrations.

In addition to calculating sample-specific risks for each waste stream, EPA compared constituent analyses and risk results within facilities, whenever more than one waste sample is available for that facility. In some cases, samples from a single facility are highly homogeneous (the same constituents are detected in most or all samples, and the variability in constituent concentrations is low). In these cases, facility-specific screening risk results are estimated using typical (arithmetic mean) of the risks. In other cases, the waste stream samples from a single facility are very heterogeneous (high variability in constituent composition and concentrations). In such cases, a simple averaging process is not appropriate, since any risk estimate based on a combination of concentrations in the different samples would seriously under- or over-estimate risks for some fraction of the wastes. In these cases, identifiable subsets of the samples at a given facility were identified with similar risks, and risks were calculated for each subset of the wastes, again using the mean of the risks. The sample-specific and facility-specific risks are used to derive benefit estimates, as discussed in **Section 5.5.1.4**.

5.5.1.3 Estimate of Post-LDR Concentrations for Screening Risk Assessment Methodology

As noted in **Section 5.4.4**, using the UTS levels for screening risk estimation inadequately reflects real-world levels. Therefore, values of half of the UTS levels have been used in the post-LDR sample-specific screening risk estimation, unless pre-LDR data show that the initial concentrations were below these levels, in which case the sample-specific data are used. The reason for choosing a level below the UTS is outlined as follows.

In *Test Methods for Evaluating Solid Waste* (U.S. EPA, 1986), federal guidance is given for compliance with a standard. One option to demonstrate compliance is to develop a sampling plan and conduct a waste analysis. This document suggests that an appropriate number of samples should be taken and the 80 percent confidence interval determined from the sampling data. If the upper bound of the 80 percent confidence interval is below the standard, then the waste is not hazardous. The important point to note is that the mean concentration of the hazardous constituent must be below the standard for the upper bound of the 80 percent confidence interval to be below the standard. While this method of compliance is federal RCRA guidance, the states or implementing agencies may impose stricter standards, which could, and often do, result in mean concentrations that are even further below the standard.

The question that remains is what level below the UTS is a reasonable estimate. A limited review of the constituent concentrations in the pre-LDR data indicates that this data is highly variable, within a given waste stream and within a facility (see **Appendix J**). The post-LDR leachate concentrations most likely reflect this variability to some extent.[18] Based on the variability seen in these data (sometimes several orders of magnitude for a given facility), it is clear that a large margin of error would be needed to assure that all of the waste meets the UTS targets. Given the costs to the regulated community of noncompliance and chemical analyses, it is reasonable to assume that wastes will be treated such that the mean concentrations of the constituents in the leachate from the stabilized waste will probably be significantly below the UTS levels.

Conversations with experts who either are in the stabilization industry or have worked with this industry have confirmed that ensuring compliance and minimizing analytical costs are primary concerns. However, no universal rule of thumb exists for treatment to a specific level below the standard. One contact, who was working with a stabilization company, stated that this company develops a formulation so that the constituent concentrations in the TCLP extract obtained in the laboratory are an order of magnitude below the standards. (See **Appendix L** for telephone logs of this and related conversations.) The theory is that the order of magnitude difference will allow ample room for error when the method is used in the field. In addition, another contact within the stabilization industry stated that every sample has to be below the standard for them to demonstrate compliance. Therefore, a safety margin below the standard is chosen based on the variability of the waste stream. He noted that the marginal cost of the material is cheap compared to the cost of noncompliance.

Based on guidance for meeting a standard, the variability present in the waste streams, and the information provided by industry experts, values of one-half the UTS appear to be reasonable and conservative estimates of post-LDR concentrations. The validity of this assumption is supported by examining the limited amount of data presented in *Best Demonstrated Available Technology Background Document for Mineral Processing Wastes* (U.S. EPA, 1995) (see **Appendix L** for summary of data). This document gives TCLP leachate concentrations for wastes before and after stabilization. Although these data are not all from mineral processing wastes, they do indicate that the majority of the constituent concentrations in the leachate from stabilized waste is less than one-half the UTS levels.

5.5.1.4 Sample-Specific Benefits Estimation Methodology

In the mean-concentration benefits assessment, the numbers of facilities included in the benefits analysis for each regulatory option and exposure scenario were estimated based on the total numbers of facilities generating the wastes, and on the degree of certainty that specific waste streams were, in fact, TC hazardous (see **Section 5.1**). In the sample-specific screening risk analysis, in contrast, the numbers of facilities included in the benefits assessment were still derived from the universe of generating facilities and on the certainty that the wastes are hazardous, but also based on the fraction of the samples (or facilities) with risk estimates in the various order-of-magnitude risk categories.

In the simplest (and very frequent) case, there is only one sample per waste stream, and thus the pre-LDR screening risk results for that waste sample are assumed to apply to all the HE waste stream-facility combinations generating the waste. For example, if the HE cancer risk estimate for the single sample of a waste stream were 5×10^{-5}, and the total regulated universe consisted of three generating facilities, all three facilities would fall into the category of facility-waste stream combinations with pre-LDR HE cancer risks between 10^{-5} and 10^{-4}. If the HE cancer risk were, in contrast, below 10^{-5}, then all three facilities would fall into the CT pre-LDR $<10^{-5}$ cancer risk category.

Where multiple samples were available for a waste stream, the numbers of facility-waste stream combinations in the various risk categories are estimated according to the proportions of the individual facilities–represented by the samples–that fall into the same risk categories. For example, in the simplest case, if there are a total of four samples of a given waste stream in the data base, each from different facilities, and if two of four samples have hazard quotients less than 1.0 and the other two samples fall into the hazard quotient category 10 to 100, then half of the universe of facilities generating that waste stream would likewise be placed into those two risk categories in the benefits assessment.

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[18] If the leachate concentrations of the wastes input to the treatment process, for example, vary by plus-or-minus a factor of ten, this is a good estimate of the proportionate variability in concentration of the treated material, although it is possible to envision processes that would both decrease the level of variability (better mixing, ion- or pH-controlled solubility) and increase the level of variability (incomplete mixing, uneven stabilization).

A variation on this approach is applied wherever risks from different facilities differ widely and systematically (some facilities high, others low), and where there is a large disparity in the numbers of samples taken from the different facilities. In these cases, giving equal weight to all of the individual samples would results in a disproportionate weight being given in the benefits assessment to the risks results from the facility with the greatest number of samples. Where this occurs, pre-LDR facility-waste stream combinations were apportioned into risk categories using both facility-specific and sample-specific risk results, in the following manner. In cases where a relatively large number of samples come from the same facility, and where the screening risk results from that facility are similar, the risk results for all of the samples from that facility are averaged, and a facility-wide risk is calculated. This risk result is then treated as if it were a single sample-specific risk result when benefits are calculated. This avoids giving disproportionate weight to facilities with lots of samples and giving low weights to results from facilities with only one risk result. Assume, for example, eight samples are available for a given waste stream, four of which come from one facility, while the remaining four come from four other facilities. All four of the risk results for the one facility are quite similar and very high, while the risk results from all the other facilities are much lower. In this case, the weight of the risk results from the single facility is reduced, and the results from this facility are counted as if they were from one sample.[19] In the benefits analysis for this example, one-<u>fifth</u> of the universe of facilities (corresponding to the single facility where risks were higher) would thus be placed into the high-risk category, and the other four-fifths of the facilities (corresponding to the other four samples from four other facilities) would be placed into the pre-LDR lower-risk categories.

Where sample-specific screening risk results from a single facility are highly variable, these results are not combined, but are still counted as individual sampling results in the benefits assessment. Where a subset of samples from a facility are homogeneous, these samples are again treated together as a single result, analogous to the approach taken above. For example, in the sample-specific risk results for facility "3" for copper acid plant blowdown in **Appendix M**, where a single risk result for a facility with multiple samples is many orders of magnitude different from all the others, the sample is treated separately in the benefits analysis, and the remaining, more similar risk results are combined. In the example facility, five samples with relatively similar risk results were combined into one "adjusted facility", and the single outlying sample with cancer risks many orders of magnitude higher was treated as another "adjusted facility". The result was that both the low-risk and high-risk samples were reflected in the benefits analysis, and were given equal weights.

This approach of combining sample-specific screening risk results into facility-wide risk estimates, and breaking down some results within individual facilities into two or more strata, introduces a degree of uncertainty into the benefits assessment methodology, because it assigns varying weights to individual sample results, depending on other sampling results from the same facility. What EPA found, however, was that decisions regarding how to aggregate or subdivide samples from single facilities had relatively impact on the aggregate benefits estimates. As discussed in **Section 5.5.4**, the distribution of facility-waste stream combinations across risk categories was found to follow the distribution of risk results from the individual samples quite closely.

As noted in **Section 5.5.1.3**, the post-LDR screening risk assessment for each waste stream is based on the assumption that, irrespective of the starting constituent concentrations, post-LDR waste managers would all achieve average leachate concentrations equal to one-half the UTS levels (or lower, if data are available which support this finding). This assumption is consistent with the strategies that facilities will most likely adopt to assure compliance with the standard. Thus, individual sample concentrations do not enter into the post-LDR placement of waste stream-facility combinations into risk categories, unless they are below one-half the UTS. Instead, analogous to the approach that was taken in the mean-concentration screening risk estimate, all of facilities generating the waste stream are counted into the same risk category based on the risk level achieved at the post-LDR UTS concentrations divided by two.

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[19] Risks from multiple samples at a single facility are considered to be homogeneous if the maximum and minimum key constituent concentrations and risks are within one to two orders of magnitude of one another, and the constituent compositions are generally similar. Where facility-specific risks are calculated from multiple sample results from a single facility, the arithmetic mean risk result is used in the benefits assessment. The arithmetic mean result is used because, as will be discussed below, "outliers" (samples from a given facility with much higher or lower risks than the other samples from the same facility) are further segregated in the benefits analysis.

5.5.2 Sample-Specific Risk Assessment Results

EPA performed cancer risk and noncancer hazard quotient calculations for each of the individual sample results, using both HE and CT assumptions, as described in **Section 5.5.1**. Screening level risks were calculated for 42 waste streams; those that were included in the mean-concentration analysis, minus the two fully recycled materials, plus six additional waste streams for which concentration data were subsequently identified, as described in **Section 5.5.1.1**.

The complete risk assessment results are provided in **Appendix M**. The results of these calculations are summarized in **Exhibits 5-7a** and **5.7b**. Altogether, pre-LDR cancer risks were calculated for 126 individual samples, and noncancer hazard quotients were calculated for 217 samples, the difference being that, when waste sampling had taken place, the carcinogenic constituents were not analyzed for in the laboratory (or they did not pass QA/QC).

5.5.2.1 Sample-Specific Risk Results for the No Prior Treatment Baseline

The distribution of the cancer risk results for all of the wastes streams are totaled in the bottom row of **Exhibit 5-7a**. For pre-LDR, 72 samples had CT cancer risks less than 10^{-5}, with 54 having cancer risk results greater than this value. A total of 21 samples had cancer risk results greater than 10^{-3}, with one sample (copper acid plant blowdown) showing a cancer risk approaching 1.0. The HE pre-LDR risk estimates are, as expected, considerably higher than the CT values. Only 23 samples had HE pre-LDR cancer risks less than 10^{-5}, while 47 samples showed HE cancer risks greater than 10^{-3}.

EXHIBIT 5-7

Graphic Not Available.

The waste samples with the highest risks were concentrated in a relatively few industries. As noted above, the highest CT risk was associated with one sample of copper acid plant blowdown. Samples with CT cancer risks in the range of 10^{-2} to 10^{-1} were beryllium spent barren filtrate streams (5 samples), berryllium chip treatment wastewater (one sample), and lead process wastewater (2 samples). Risks in the range from 10^{-3} to 10^{-2} were beryllium spent barren filtrate, bertrandite thickener slurry (3 samples each), copper acid plant blowdown and spent bleed electrolyte (2 samples each), and zinc process wastewater and spent surface impoundment liquids (one sample each).

Under HE assumptions, cancer risks for additional waste streams to those just mentioned also fell into the upper cancer risk ranges. Additional waste streams with one or more samples greater than 10^{-2} included antimony autoclave filtrate (8 samples), elemental phosphorous furnace scrubber blowdown (one sample), lead surface impoundment waste liquids (1 sample), tantalum/columbium process wastewater (3 samples), titanium and titanium dioxide leach liquor (1 sample), and zinc process wastewater (2 samples).

Ninety-five of the 217 samples had CT pre-LDR noncancer hazard quotients less than 1.0. Only 28 samples had CT hazard quotients in the three highest hazard quotient ranges (100 to greater than 10,000), with the remainder falling in between. In the HE pre-LDR case, 34 samples had hazard quotients less than 1.0, and the number of samples with hazard quotients greater than 100 increased to 93.

In the CT case, the waste streams with the highest hazard quotients (greater than 100) included antimony autoclave filtrate (6 samples), copper acid plant blowdown (one sample), elemental phosphorous furnace scrubber blowdown (one sample), lead process wastewater (4 samples, tantalum/columbium process wastewater (2 samples), and five different zinc industry wastes (13 samples total). Under HE assumptions, the hazard quotients for many of the samples increased; 19 waste streams had one or more samples with pre-LDR HE hazard quotients between 100 and 1,000, 10 had one or more samples with hazard quotients between 1,000 and 10,000, and seven waste streams had one or more samples with hazard quotients greater than 10,000.

As expected, these risks follow the pattern seen in the mean-concentration risk assessment, with a few differences. For example, a few high-risk samples from several of the waste streams move into higher categories than the corresponding mean-concentration risk assessment. Also, low-risk samples show up in this analysis for some waste streams that had high mean-concentration risks. This is discussed in more detail in **Section 5.5.4**.

5.5.2.2 Post-LDR Sample-Specific Risk Results

Under Options 1 and 2, the distribution of sample-specific risk results shifts substantially towards lower risk categories, as would be expected if all of the wastes above one-half UTS were treated so as to reduce leachate concentrations to those levels. This shift in risks is documented in **Appendix M**, and summarized in **Exhibits 5-7a** and **5-7b**.

Under CT assumptions, post-LDR sample-specific[20] cancer risks all fall below 10^{-4}; 79 samples had CT risks less than 10^{-5}, and 47 samples had CT cancer risks in the range between 10^{-5} and 10^{-4}. Under HE assumptions, 32 samples had post-LDR cancer risks less than 10^{-5}, 31 had cancer risks between 10^{-5} and 10^{-4}, 63 had cancer risk between 10^{-4} and 10^{-3}, and none had risks greater than 10^{-3}. As noted previously, the post-LDR cancer risk estimates are driven primarily by the assumption that all post-LDR leachate concentrations will be reduced to one-half the UTS concentrations; thus, there is no particular pattern of risk reduction associated with specific industries or waste characteristics.

Under CT assumptions, all of the 217 waste samples had post-LDR hazard quotients less than 1.0. Under HE assumptions, most (146) of the samples had post-LDR hazard quotients less than one, and the remainder (71 samples) had hazard quotients between 1.0 and 10.

Again, these results parallel to some extent the results of the mean-concentration risk assessment described in **Section 5.4**. A large proportion of the sample-specific risks shift from relatively high pre-LDR

[20] Note that since all of the post-LDR concentrations are reduced to one-half the UTS levels, these risks are not truly "sample-specific" any more, unless the pre-LDR concentrations already meet the UTS levels.

cancer risk values to lower-risk categories post-LDR. All of the pre-LDR sample-specific cancer risks greater than 10^{-4} under CT conditions and greater than 10^{-3} under HE assumptions are shifted to less than these values post-LDR, but the number of sample-specific risk estimates less than the level of regulatory concern of 10^{-5} is only slightly increased post-LDR compared to pre-LDR under conditions. Similarly, all pre-LDR CT hazard quotients greater than 1.0, and all pre-LDR HE hazard quotients greater than 10 are reduced below these value post-LDR. In contrast to the cancer risk results, the proportions of sample-specific hazard quotients (both CT and HE) less than 1.0 increase substantially post-LDR.

5.5.3 Sample-Specific Benefits Assessment Results

The results of the sample-specific screening risk assessment were used to develop estimates of the regulatory benefits of Options 1 and 2, analogous to the procedure used in the mean-concentration risk assessment (see **Section 5.5.1.4**). The results of the sample-specific risk assessment were distributed across the CT and HE estimates of the numbers of facilities generating each waste stream, to derive estimates of the number of waste stream-facility combinations with cancer risks or hazard quotients falling into the specified risk categories under baseline and post-LDR conditions. The results of this analysis are summarized in **Exhibits 5-8a** and **5-8b** (and illustrated in **Exhibit 5-8c**).

The benefits of Options 1 and 2, in terms facility-waste stream combinations at different risk levels, are shown in **Exhibit 5-8a**. Pre-LDR, 68 of the 117 facilitty-waste stream combinations at which wastes with carcinogenic constituents are generated are distributed into the CT risk category below the regulator level of concern of 10^{-5}. The 49 remaining facility-waste stream combinations have been placed into risk categories above this level, with 16 facilities falling into the risk categories above 10^{-3}. Post-LDR, the number of facility-waste stream combinations with CT cancer risks less than 10^{-5} increases only slightly to 71. However, all of the remaining facility-waste stream combinations are reduced to CT cancer risk levels less between 10^{-5} and 10^{-4}.

Under HE assumptions, only 43 of 140 total facility-waste stream combinations have been placed into the pre-LDR risk category below 10^{-5}, and a larger proportion (51 of 140) fall into

EXHIBIT 5-8

Graphic Not Available.

the risk categories above 10^{-3}. Post-LDR, the number of facility-waste stream combinations below 10^{-5} increases to 50, but, as was the case for the CT cancer risks, the number of facility-waste stream combinations in the high-risk categories (greater than 10^{-3}) falls to zero. The bulk of the facility-waste stream combinations (90) fall into the HE post-LDR risk categories between 10^{-5} and 10^{-3}.

Pre-LDR, CT sample-specific hazard quotient values less than the levels of regulatory concern (1.0), have been assigned to 73 of 139 facility-waste stream combinations.[21] A total of 10 combinations fall into the three highest-risk categories (hazard quotients greater than 100), and the remainder (56) are placed into pre-LDR risk categories with hazard quotients between 1 and 100 (**Exhibit 5-8b**). Post-LDR, consistent with the sample-specific hazard quotient results, all of the facility-waste stream combinations are placed into the risk category with hazard quotients values below 1.0. Thus under CT assumptions, the noncancer health benefit of Options 1 and 2 is to reduce the hazard quotients for all of the facility-waste stream combinations below levels of regulatory concern.

Pre-LDR, the facility-waste stream combinations are much more broadly distributed across HE hazard quotient categories. Only 28 facility-waste stream combinations have hazard quotients less than 1.0, and 62 have been placed into the high-risk categories with hazard quotients greater than 100. Under Options 1 and 2, the number of facility-waste stream combinations below the level of regulatory concern is increased to 108, and all of the remaining combinations (65) fall into the next higher category (hazard quotients between 1.0 an 10). Thus, there is also a substantial noncancer health benefit, even under HE assumptions.

5.5.4 Comparison of Mean-Concentration and Sample-Specific Benefits Assessment Results

As noted previously, one of the major reasons for performing the sample-specific risk and benefits assessments was to determine how such results would differ from the mean-concentration approach. As noted in **Section 5.1**, because of the lack of knowledge of constituent concentration as a function of waste volume, and the lack of constituent concentration data for unsampled facilities, any approach to estimating industry-wide risks and risk reduction benefits is subject to a large degree of uncertainty. The sample-specific benefits assessments provide a useful comparison to the mean-concentration approach because it uses a different approach to estimating waste stream risks and risk reduction across facilities, waste streams, and commodities.

The results of the mean-concentration and sample specific benefits analyses are compared in **Exhibits 5-9a** (cancer risks) and **5.9b** (noncancer risks). These graphs display the proportions of risk estimates that fall within the order-of magnitude risk ranges used in the benefits analysis. Risk estimates are displayed for (1) facility-waste stream combinations estimates derived using the mean-concentration approach, as shown in **Exhibit 5-5**, (2) individual samples from each waste stream, as shown in **Exhibit 5-7**, and (3) facility-waste stream combinations estimated using the sample-specific risk values, as shown in **Exhibit 5-8**. The results of this analysis are displayed in terms of the proportions of the samples and facility-waste stream combinations falling into the various categories (instead of numbers) because the numbers of samples and facility-waste stream

[21] The numbers of CT and HE facility-waste stream combinations included in the benefits analysis differ because smaller numbers of facilities were identified as having carcinogens present in wastes than the numbers of facilities having noncarcinogens. For example (see **Appendix M**), several wastes from the lead industry (4 CT and HE facilities) and several waste streams from the titanium/titanium dioxide industry (4 CT and 7 HE facilities) were not analyzed for carcinogenic constituents, and thus are not included among the facility-waste streams combinations counted in the cancer risk reduction benefits tabulation.

EXHIBIT 5-9

Graphic Not Available.

combinations are different across the three analyses, and normalizing them all to a single scale better shows how the overall distribution of the results compares across risk categories.

It can be seen from **Exhibit 5-9a** that there are substantial differences in the distribution of facility-waste stream combination risks and sample-specific risk results calculated using the different methods. In particular, the proportion of facility-waste stream combinations for which CT risk estimates are below 10^{-5} is substantially lower when the mean-concentration approach is used than when the sample-specific risk results are used (31 percent, or 30 of 98, compared to 58 percent, or 68 of 117). The mean concentration approach also places a larger proportion of the facility-waste stream combinations in the 10^{-3} to 10^{-2} category than does the sample-specific method. Overall, the mean-concentration distribution of facility-waste stream combination by CT cancer risks is skewed toward higher risks than the comparable distribution calculated using the sample-specific risk method.

A similar situation is seen for post-LDR CT cancer risks. The mean-concentration benefits analysis places a larger proportion of the facility-waste stream combinations in the risk category above 10^{-5} (36 of 98) than does the sample-specific benefits assessment (71 of 117), although both analyses predict a large reduction (decrease to zero) in the number of facility-waste stream combinations in the high cancer risk categories.

It is also interesting to note that, while the mean-concentration benefits assessment predicts generally higher proportions of higher risks in both the pre-LDR and post-LDR CT risk categories, the magnitude of the benefits of Options 1 and 2, measured in terms of the incremental number of facility-waste stream combinations moving below the level of regulatory concern is similar for the two methods. Using mean-concentration risk results, the estimated number of facility-waste stream combinations below 10^{-5} CT cancer risk increases from 30 to 36, while using the sample-specific risk results, the number of combinations below this level increases from 68 to 71.

The same general pattern of more facility-waste stream combinations falling into higher-risk categories when calculated using the mean-concentration methods is also seen for the HE cancer risk estimates, as shown in the lower panel of **Exhibit 5-9a**, in both the pre-LDR and post-LDR cases. This pattern is not unexpected because, when using the mean concentration risk estimation method, sampling results with very high risks could be expected to "swamp" lower risk results from the same waste stream.

The pattern of larger proportions of mean-concentration facility-waste streams in higher risk categories is somewhat less pronounced in the case noncancer hazard index calculations, as shown in **Exhibit 5-9b**. For example, the proportion of facility-waste stream combinations with CT pre-LDR hazard indices less than 1.0 is 48 percent (65 of 136) for the mean-concentration approach, and 53 percent (73 of 139) for the sample-specific approach. The proportions in the higher CT hazard quotient ranges are also closer than was the case for cancer risks. Both methods of risk and benefits calculation place all of facility-waste stream combinations in the lowest post-LDR CT hazard quotient category.

Under HE assumptions, the mean-concentration approach again places a smaller proportion of facility-waste stream combinations in the pre-LDR category with hazard quotients less than 1.0 (6 of 157, or 4 percent), compared to the sample-specific approach (28 of 174, or 16 percent). The proportions of facility-waste stream combinations in the higher HE pre-LDR hazard quotients is also increased compared to the sample-specific results. The post-LDR HE hazard quotient distributions of facility-waste stream combinations calculated using the two methods differ in that only about 36 percent (57 of 157) of the mean-concentration facility-waste stream combinations end up below levels of regulatory concern, while 62 percent of the sample-specific combinations (108 of 173) are placed in this category. Thus, the benefits of Options 1 and 2, in terms of facility-waste stream combinations moving from high noncancer risk to lower risk categories under HE assumptions are somewhat higher for the sample-specific procedure than when the mean-concentration methodology is used.

On the whole, the sample-specific risk and benefits analyses confirm the suspicion that smaller proportions of facility-waste stream combinations would be placed into high risk categories than was the case when the mean-concentration approach was used. This tendency does not, however, translate into a simple consistent bias of one method over the other in terms of the magnitude of benefits that have been estimated. As seen in this analysis, while pre-LDR HE and CT cancer risks are generally higher using the mean-concentration approach than are found using the sample-specific approach, the changes in risks, as measured by the numbers of facility-waste stream combinations moving to lower cancer risk categories, are very comparable between the two methods. Likewise, the results of the benefits assessment calculated using CT hazard quotients are very nearly the same for the two methods. The benefits estimate, when HE hazard quotients are the indicators of risk, is slightly higher using the sample-specific approach than that which is obtained using the mean-concentration method.