PHASE I VOLUNTARY SITE INVESTIGATION REPORT
VOLUME II - HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE
95 AMES STREET
ROCHESTER, NEW YORK

Prepared for:

Combustion Engineering, Inc.
501 Merritt 7
Norwalk, CT 06851

Prepared by:

ABB Environmental Services, Inc.
1175 John Street
West Henrietta, NY 14586

NOVEMBER 1996

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NEW YORK

TABLE OF CONTENTS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NEW YORK

TABLE OF CONTENTS
(Continued)

TABLE OF CONTENTS
(Continued)

TABLE OF CONTENTS
(Continued)

LIST OF FIGURES

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NEW YORK

TABLE OF CONTENTS
(Continued)

LIST OF TABLES

TABLE OF CONTENTS
(Continued)

LIST OF APPENDICES

1.0 INTRODUCTION

ABB Environmental Services, Inc. (ABB-ES) has prepared this human health risk assessment (HHRA) for the former Taylor Instruments facility at 95 Ames Street (Ames Street Site), Rochester, Monroe County, New York as part of the Phase I Voluntary Site Investigation for the Ames Street Site ("the Site".) The Voluntary Site Investigation Report consists of the Site Investigation (SI) Report (Volume I) and the HHRA (Volume 2). An ecological risk assessment was not performed as Part of the Phase I Voluntary Site Investigation because there is no significant habitat, surface water or sediment on or adjacent to the Site.

As described in the Phase I Work Plan, the primary purpose of the HHRA is to provide information needed to develop risk-based Quality Goals (QGs) for soil and groundwater at the Site, and to then use that information to develop those goals. Quality Goals are numerical concentrations which are protective for health risks associated with potential exposures to site-related contaminants in environmental media at the Site. The QGs developed in this HHRA are compared to soil and groundwater data from the Site in Section 5 of Volume I.

The HHRA consisted of several components, including data evaluation and summarization, preliminary screening, and risk-based concentration (RBC) development. The RBC development consisted of toxicity assessment, exposure assessment, calculation of RBCs, and uncertainty evaluation. Collectively, these components were used to generate information used to develop the QGs presented in Section 5. In summary, the

purpose of the data evaluation component was to present the analytical data for the Site and discuss the methods used to select the data used in the HHRA. The preliminary screening step identified chemicals of potential concern (CPCs) in soil and groundwater through a comparison of analytical data sets to NYSDEC standards. Risk-based concentrations were then developed for all CPCs based on toxicity and exposure data evaluated in the toxicity assessment and exposure assessment components. From the various RBCs calculated for each CPC, a single RBC was selected as a screening RBC to determine which CPCs required QGs. QGs were then developed in Section 5 for site-specific target risk levels of $1x10^{-6}$ for carcinogens and a hazard index of 1.0 for non-carcinogens using exposure assumptions appropriate to several different site development scenarios. The subsections that follow describe the approach and results for each of these components.

The HHRA was conducted using guidance presented in the Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Standard #E1739-95; Nov. 1995), the Technical and Administrative Guidance Memorandum (TAGM): Determination of Soil Cleanup Objectives and Cleanup Levels (NYSDEC TAGM # HWR-94-4046, 1994), Site Assessment and Guidance for Petroleum Impacted Sites (review draft) (NYSDEC, 1995), Ambient Water Quality Standards and Guidance Values (NYSDEC 6NYCRR 703, 1993), and Risk Assessment Guidance for Superfund (USEPA, 1989). Additional references are listed at the end of Section 6.

The approach, components, and guidance used to complete the HHRA generally followed that described in the Phase I VSI Workplan, but included several additional elements intended to increase its usefulness as a basis for decision-making relative to the Ames

Street Site, and to incorporate evolving, relevant risk assessment techniques. These additional elements included:

- Development of Quality Goals for several future land development scenarios (based on the same technical approach). Evaluation of more than one future land use scenario will maximize flexibility available to the Site owners, developers, and regulatory agencies to arrive at mutually agreeable conditions for Site re-use.
- Combining long-established risk assessment technical guidance (e.g., NYSDEC TAGM # HWR-94-4046, 1994; USEPA, 1989), with a risk-based corrective action (RBCA) technical approach to developing RBCs (ASTM, 1995). The RBCA approach allowed the HHRA to maximize use of Site- and future use-specific data and assumptions. RBCA-type models are not only based on USEPA risk assessment technical approaches, but are currently issued as draft guidance by the state of New York (NYSDEC, 1995).
- Use of a wide range of mercury concentrations to characterize mercury speciation and bioavailability. The Workplan proposed an initial mercury screening level of 200 mg/kg above which speciation and bioavailability testing would occur. In practice, the HHRA evaluated the entire observed range of mercury concentrations (subject only to the limitations of the analytical methods); this provided more information about the mercury speciation and bioavailability in Site soils.

2.0 DATA EVALUATION

The purpose of the data evaluation was to evaluate the entire analytical database for the Site and organize it in a manner suitable for use in the quantitative HHRA. The individual steps involved in this process were conducted for the VSI (see Section 3 of Volume I), are briefly discussed below. These methods were used to develop data sets for soil and groundwater, as discussed in Subsections 2.1 and 2.2, respectively.

Sort Data by Medium. Data from the Phase I SI and from previous investigations conducted in 1993 and 1995 were compiled and sorted by environmental medium (i.e., soil and groundwater). All chemicals detected in at least one sample in each medium were identified.

Evaluate the Analytical Methods. A detailed discussion of the laboratory methods used for analysis of soil and groundwater samples is presented in Section 3 and Appendix D of Volume I. As described therein, the bulk of the Phase I (1996) analytical data were validated for use in the HHRA and meet the USEPA definition of definitive data. A portion of the pre-Phase I soil data (1993 and 1995) were analyzed using USEPA reference methods and, although not validated, still meet the USEPA definition of definitive data and were therefore used in the quantitative HHRA. Portions of the Phase I and pre-Phase I data sets that were determined to not meet the definition of definitive data were not used in the HHRA. These data are discussed qualitatively in Volume I, and included data associated with investigation of the "Class 4 Area", which featured some

very high mercury results (up to an estimated 52,000 mg/kg) that could not be judged definitive due to lack of associated QC data.

Evaluate Quantitation Limits. As described in Section 3 of Volume I, for all data used in the HHRA the reporting limit for organic compounds and inorganics was the sample quantitation limit (SQL), which is the practical quantitation limit (PQL) modified for sample moisture content and dilution factor. Analytical methods used during the Phase I VSI were selected so that their reporting limits were below the screening concentrations to ensure that contaminants of concern, from a risk or regulatory perspective, could be detected and quantified. Because analyses for some pre-Phase I samples were conducted prior to setting the Phase I DQOs, some SQLs for pre-Phase I data are above screening levels. Analyte-specific reporting limits that were above screening concentrations were identified so that limitations in the evaluation for those constituents could be identified and discussed. Limitations associated with these data are discussed in Sections 3 and 4 of Volume I.

Evaluate Qualified Data. Both the laboratory and data validators have assigned qualifiers to the analytical results. The results of the data validation are briefly discussed in Section 3 of Volume I. A more thorough discussion presented in Appendix D of Volume I. The validated data, with qualifiers, are presented in Appendix C of Volume I. All positive detections (whether they were unqualified or qualified with a "J") were considered detected concentrations for the HHRA. All non-detects (results qualified with a "U") were retained in the HHRA data set as samples without positive detections. If all sample results for a given analyte in a given medium were non-detections, then that analyte was not retained as a detected analyte for the purpose of the HHRA. Any sample

results with an "R" validation qualifier were eliminated from the HHRA data set because the QC review indicated that the result was unusable.

Compare Concentrations Detected in Samples to Concentrations Detected in Blanks. As discussed in Section 3 of Volume I, sample concentrations have been compared to concentrations in associated blanks to distinguish artifacts from actual presence of constituents in environmental samples. These comparisons were conducted as part of the data evaluation and validation processes. Those sample results considered artifacts of laboratory analyses and/or equipment decontamination are identified in Section 3 and Appendix D of Volume I and generally include chloroform and methyl ethyl ketone. Due to contamination of method blanks, cis-1,2-dichloroethene was rejected in all 1996 soil data. Limitations associated with this are discussed in Sections 3 and 4 of Volume I.

Evaluate TICs. Tentatively Identified Compounds or TICs (constituents for which both identity and concentration are uncertain) were reviewed during pre-Phase I investigations as discussed in the Phase I VSI Workplan (ABB-ES, 1996). As described therein, the number of TICs was small relative to the Target Analyte List (TAL) and Target Compound List (TCL) chemicals. There were no TICs identified in the 1996 SI off-site volatile organic compound (VOC) laboratory analyses. Therefore, TICs were not evaluated in the Phase I VSI data.

Develop a Data Set for Use in HHRA. The ultimate product of data evaluation and data summarization is a set of analytical data presented in a form that can be used in the HHRA. Analytical data sets for soil and groundwater are discussed in the subsections that

follow. The summarized data are presented in tables which contain (1) the identity of all compounds detected in at least one sample from any sampling event, (2) the ratio of the number of samples in which the analyte is detected to the total number of samples analyzed for the analyte (i.e., frequency of detection), (3) ranges of SQLs, (4) range of detected concentrations, and (5) mean concentrations. Results qualified as rejected were not included. In calculating the mean concentrations, one-half the SQL was used as the reported value for all results qualified as non-detect. For some constituents that had elevated reporting limits and low frequencies of detection, the calculated mean concentration may exceed the maximum detected concentration. These include 1,1,1-trichloroethane, trans-1,2-dichloroethene, and 1,3,5-trimethylbenzene in soil, and ethylbenzene, benzene, 1,3,5-trimethylbenzene, 1,2,4-trimethylbenzene, and n-butylbenzene in groundwater. The summarized data in the data summary tables were used in the preliminary screening process.

2.1 SOIL

Soil data were collected during the 1996 SI and previous investigations. As discussed in Sections 3 and 4 of Volume I, pre-Phase I soil data collected at the Site were generally analyzed for a broad spectrum of constituents. Soil samples collected during the Phase I VSI were analyzed only for the COCs identified in the Phase I VSI Workplan (ABB-ES, 1996); the soil data sets evaluated in this HHRA include the constituents identified as COCs for the VSI, as the Phase I VSI data did not suggest the presence of additional COCs.

Because the Site has been paved, all soil data were evaluated as a single exposure medium. Separate data sets were not developed for surface soil and subsurface soil. Any potential remedial decisions concerning Site soils will consider the current nature and extent of constituents that are selected as chemicals of concern at that time.

Soil data were evaluated by chemical class, including VOCs, inorganics and cyanide, and mercury. As discussed in detail in Section 3 of Volume I, data for constituents within each of these chemical classes were collected using various analytical protocols, each associated with specific levels of data quality. From the available data, the most appropriate data for use in this evaluation were selected based on the criteria described in Section 3 of Volume I. Only data meeting the definition of definitive data were used in the quantitative HHRA. Table 1 and the following subsections provide a summary of the sources of soil data used, with details of the evaluation process and selection rationale provided in Subsection 4.2 of Volume I. Any data which were not considered appropriate for this evaluation are identified below.

2.1.1 Volatile Organic Compounds

Volatile organic compounds (VOCs) data selected for use in this HHRA were from the on-site laboratory analyses of soil samples collected during the Phase I investigation and from off-site analysis of soil samples collected during previous investigations. As described in Section 4.2 of Volume I, these data all meet the definition of definitive data.

As discussed in Section 3 of the Volume I, VOC data from 1993 on-site laboratory analysis were determined to not be of suitable quality for quantitative use in the HHRA. These data were used qualitatively to identify investigation locations for the VSI and to supplement the understanding of conditions in certain areas of the Site. In addition, some data from 1995 were judged to not meet the definition of definitive data because they represent soil which was subsequently excavated and disposed of off-site. Because these samples are not representative of current site conditions, they were not determined to be definitive data and, therefore, were not included in the HHRA.

A summary of the soil data set is presented in Table 2. A total of 111 samples were collected and analyzed for VOCs, including 93 Phase I subsurface soil samples collected from depths between 4 and 25.8 feet; 2 shallow soil samples collected during 1993; and 16 subsurface soil samples collected during removals of tanks 1995. Table B-1, Appendix B, presents a list of samples included in the soil data set for VOCs.

The total number of samples used to calculate frequency of detection vary among the constituents because there was some rejected data for each analyte, and some constituents were not analyzed in all samples collected. For example, soil samples collected during the 1995 removal of tanks were analyzed for benzene, ethylbenzene, toluene, and xylenes (plus several other non-COC compounds) only, resulting in a higher number of total samples for these constituents.

2.1.2 Inorganics and Cyanide

Inorganics (chromium, lead, nickel, and zinc) and cyanide data selected for use in this HHRA were from the off-site laboratory analyses of soil samples collected during the Phase I investigation and previous investigations. Although on-site laboratory analyses for inorganics data were available, these data were not considered suitable for use in the HHRA. As described in Subsection 4.2 of Volume I, on-site laboratory analyses were performed using X-ray fluorescence (XRF) screening techniques. The detection limit for this analytical method was 100 mg/kg, which is well above the recommended soil cleanup levels in the TAGM (NYSDEC, 1994) for all of these constituents except lead, for which the TAGM value is 200 mg/kg. Comparison of the XRF data with off-site laboratory data from split samples indicated that many of the XRF detections reported as values near the XRF detection limit (e.g., 100 to 110 mg/kg) were not detected in off-site laboratory analyses. Off-site laboratory data have detection limits that are lower than TAGM recommended cleanup values. Therefore, only off-site laboratory data were included in the soil data set.

A summary of the HHRA soil data set is presented in Table 2. A total of 23 samples were collected and analyzed in an off-site laboratory for inorganics. These samples included 14 Phase I subsurface soil samples collected from depths between 4 and 22 feet, 7 pre-Phase I surface soil samples, and 2 pre-Phase I subsurface soil samples collected at depths of 2 and 6 feet. Table B-2, Appendix B, presents a list of samples included in the soil data set for inorganics and cyanide. There was some rejected data for each analyte and, therefore, the total number of samples in the frequencies of detection vary among constituents.

As indicated in Table 2, both total and hexavalent chromium were determined for each of the 17 samples. Comparison of total and hexavalent chromium results indicated that hexavalent chromium was detected in only 6 samples, and that the fraction of hexavalent chromium was not consistent among samples. In samples that contained hexavalent chromium, hexavalent chromium composed 6.4% to 66.6% of the total chromium detected. In one sample, the reported hexavalent chromium concentration was more than four times higher than the reported total chromium concentration. This is theoretically impossible, since there cannot be more hexavalent chromium than total chromium. Due to this analytical uncertainty and the lack of consistency in hexavalent chromium results, all chromium was treated as hexavalent chromium in the risk assessment. This provides a conservative evaluation, since hexavalent chromium (which is the more toxic form of chromium) comprised only a fraction of the total chromium detected.

2.1.3 Mercury

Mercury data selected for use in this HHRA were from the on-site laboratory analyses of soil samples collected during the Phase I investigation and off-site laboratory data collected during previous investigations. Specifically, the mercury data used in this HHRA consist of Phase I data from on-site laboratory analyses performed with a Leeman Analyzer, and pre-Phase I data from off-site laboratory analyses performed with cold vapor atomic absorption (CVAA).

Although Phase I and pre-Phase I on-site XRF screening data were available, these data were not considered suitable for use in the HHRA as they did not meet the definition of definitive data. As described in Subsection 4.2 of Volume I, the detection limit for the

XRF screening method was 100 mg/kg, which is well above the TAGM (NYSDEC, 1994) recommended soil cleanup value for mercury of 0.1 mg/kg. Comparison of Phase I off-site laboratory CVAA data with Phase I on-site Leeman Analyzer data from split samples (Subsection 3.1.4 of Volume I) indicated that there was a good quantitative agreement between the two analytical methods for the majority of samples, with soil heterogeneity judged to be the cause of the analytical methods for three samples to agree only on a qualitative basis. Since the Phase I mercury data set for Leeman Analyzer data is more comprehensive than the off-site laboratory data set, the on-site Leeman data were selected to represent Phase I investigation data. Because Leeman Analyzer analyses were not conducted during pre-Phase I investigations, and pre-Phase I XRF screening data are not suitable for use in the HHRA, off-site laboratory CVAA data were selected to represent pre-Phase I investigation data.

A summary of the HHRA soil data set is presented in Table 2. A total of 540 samples (not including field duplicates) were collected and were analyzed for mercury. These samples included 529 Phase I soil samples collected from depths between 2 and 29 feet and analyzed on-site with the Leeman Analyzer. A total of 11 pre-Phase I samples were collected and were analyzed in an off-site laboratory for mercury. These samples included 3 subsurface soil samples collected from depths between 3 and 8 feet, and 8 surface soil samples. Table B-3, Appendix B, presents a list of samples included in the soil data set for mercury.

2.1.4 Special Mercury Analyses

A total of fifteen soil samples collected from eleven sample locations were submitted for mercury headspace analyses, speciation analyses, and bioavailability testing. Fourteen of these samples contained mercury, as evidenced by detection during field analyses. One sample did not contain detectable mercury, and was submitted as a control sample. The results of these evaluations were used to develop a bioavailability adjustment factor (discussed at length later in this report) which was used to estimate the bioavailability of mercury in soils at the Site. All analyses and testing were conducted by PTI Environmental Services, Inc. (Boulder, CO) and its subcontractor, Columbia Analytical Services, Inc. (Kelso, WA.) The report submitted by PTI, presenting methods, results, and data quality assessments for these evaluations, is included as Appendix A.

2.1.4.1 Total Mercury and Headspace Analyses

Fifteen soil samples were submitted to PTI for total mercury analyses, mercury vapor headspace analyses, and fraction organic carbon (foc), carbonate, and acid volatile sulfide analyses. Total mercury, foc, carbonate and acid volatile sulfide analyses were performed following USEPA methods. Headspace analyses were conducted using a Jerome mercury vapor analyzer following a PTI-developed routine, as there is no USEPA approved method for headspace analysis. Total mercury analyses were conducted on soils sieved to two different size fractions: <2 mm and <250 mm. Results of the <250 mm fraction were used in the bioavailability evaluation because soil particles in this size fraction are most likely to adhere to skin (i.e., hands) and potentially be ingested through hand-mouth contact (PTI, 1996).

The results of these analyses are presented in Appendix A, Tables 1 through 4. Mercury vapor in headspace above soil samples ranged from 0.004 mg/m3 to 22.8 mg/m3 for samples with total measured mercury levels ranging from non-detectable to 3,110 mg/kg. These results suggest that volatile forms of mercury are present in Site soils; elemental mercury (the most volatile form of mercury) was well correlated (r2=0.72) with mercury headspace concentration (Appendix A, Figure 3). Total mercury ranged from 1.5 mg/kg to 12,000 mg/kg in the samples sieved to <2 mm, and 2.4 mg/kg to 5,850 mg/kg in the soils sieved to <250 mm. It was further concluded that results between the two size fractions were similar (Appendix A, Table 5); any inconsistencies between results for the two soil size fractions is attributable to heterogeneity of the Site soil samples rather than an unequal distribution of mercury between the two grain size fractions.

The above analyses were performed to support the overall bioavailability assessment and were not used directly in the quantitative HHRA.

2.1.4.2 Mercury Speciation Analyses

Fourteen soil samples sieved to <2 mm were submitted for sequential extraction analysis to determine the speciation of mercury in Site soils and to provide information for the bioavailability assessment. Mercury speciation in 10 Site soil samples was also evaluated by electron microprobe analysis (EMPA).

Sequential extraction mercury analyses were performed using CVAA. The results of these analyses are presented in Appendix A, Table 8. The mercury in Site soils was found to be composed on average of 0.3% organic mercury species (0.03% to 0.7%), 21% acid

soluble mercury species (0.2% to 65%), 63% elemental mercury (17% to 99%), and 15% mercuric sulfide (0.7% to 70%). For all but two samples, elemental mercury represented the largest fraction of mercury. These findings are consistent with the history of the Site, where elemental mercury was the form of mercury primarily used at the facilities. These findings are also supported by mercury vapor headspace analyses (representing volatile mercury), which correlated well (r2=0.72) with elemental mercury concentration (the primary volatile form of mercury) (Appendix A, Figure 3). As discussed in Subsection 4.4.1.2 and Appendix A, the buffering capacity of Site soils may have resulted in a low bias in the reported acid soluble species concentrations, and a high bias in the reported elemental species concentrations. For this reason, only soil samples with carbonate contents below 5% (which do not have a high buffering capacity) were considered for interpretation of the sequential extraction results. These limitations do not affect the results or interpretation of the bioavailability testing results. No pattern of source area-specific or soil type-specific speciation was identified. Therefore, the speciation results were applied to all soils throughout the Site.

EMPA results are summarized in Appendix A Table 14. The purpose of the EMPA analyses was to quantify the distribution of mercury mass among the various mercury-bearing mineral phases in Site soils. Results of the EMPA analyses were generally in agreement with the sequential extraction analyses. Distribution of the mercury mass among the various mercury-bearing particles showed that elemental mercury was the primary mercury species, representing on average 61% (18% - 98%) of the total mercury. Mercuric sulfide was found to represent approximately 24% (1% to 96%) of the total mercury, and mercuric chlorides and sulfates were found to represent approximately 15% (1% to 67%) of the total mercury. As discussed in detail in Appendix A and summarized

in Subsection 4.4.1.2, the EMPA analytical technique tends to underestimate the fraction of elemental mercury and mercuric chloride in soil samples. However, this analytical limitation does not affect the results or interpretation of the bioavailability assessment.

Similar to the headspace analyses, the speciation analyses are designed only to assist in corroborating the bioavailability testing and were not used directly in the quantitative HHRA.

2.1.4.3 Bioavailability Testing

Bioavailability refers to the amount of chemical that is absorbed into the bloodstream and is thereby available for biological interaction. Bioavailability testing was performed on 10 of the 14 samples submitted to PTI for speciation analyses. The samples submitted for bioavailability testing were selected to provide bioavailability information for a range of total mercury concentrations (2.4 mg/kg to 5,850 mg/kg) and for a range of conditions throughout the Site (i.e., soil type, source area, etc.). A description of the bioavailability test procedure is presented in Appendix A and summarized in Subsection 4.2.3.3. Results of the bioavailability testing are presented in Appendix A, Table 18, and summarized in Appendix A, Figure 6. From these data a bioavailability adjustment factor of 0.2 was derived. This adjustment factor indicates that mercury availability in Site soils is approximately 20% of the bioavailability of mercuric chloride (the mercury species on which the oral reference dose for mercury is based). The bioavailability adjustment factor of 20% is consistent with the mercury speciation results, which indicate that approximately 78% of the mercury in Site soils is present in insoluble, minimally bioavailable forms (i.e., elemental mercury and mercuric sulfide), and 22% of the mercury in these soils is present

in potentially bioavailable forms (i.e., acid soluble species and organic species). No pattern of source area-specific or soil type-specific bioavailability was identified. Therefore, the BAF of 0.2 was applied to all soils throughout the Site.

Data quality for the bioavailability testing was examined in Attachment A of Appendix A. Because bioavailability testing is a non-standard procedure, there is no specified procedure or standards for judging the resultant data quality. These data were considered suitable for use in the quantitative HHRA based on two factors:

- As described in Attachment A to Appendix A of this report, the quality of the analytical work (primarily total mercury analysis) performed by Columbia Analytical on the extracts from the bioavailability testing was determined to be acceptable. These data meet the definition of definitive data, similar to other analyses performed in off-site laboratories during Phase I and pre-Phase I investigations.
- As described in Appendix A, the bioavailability testing included a number of quality control procedures including replicate, duplicate and spike analysis. Evaluation of the resulting data, discussed in Appendix A and Section 4.4.1.2 of this report, indicate that the bioavailability data is of sufficient quality for quantitative use.

2.2 GROUNDWATER

Groundwater data were collected during the Phase I investigation and previous investigations. The groundwater data selected for this HHRA were from a round of samples collected from existing and newly installed temporary monitoring wells during Phase I (April 1996) and several samples from non-monitoring well locations collected during 1993 and 1995. These data reflect current site conditions and include the largest number of sampling points. All groundwater samples collected during the Phase I investigation and previous investigations were submitted to the off-site laboratory for analyses for VOCs, inorganics (chromium, lead, nickel, and zinc), and mercury. Table 1 provides a summary of the sources of groundwater data used in this HHRA. The details of the selection rationale are provided in Subsection 4.2 of Volume I, and are summarized below.

A total of 23 monitoring wells are located along the northern and eastern perimeter of the Site. These wells are in a downgradient flow direction from Site sources, and were selected for the HHRA perimeter well data set. Four wells located in the Site interior at specific areas known to be soil source areas were selected for evaluation as the interior well data set. Five samples collected in 1993 and 1995 from non-monitoring well locations (e.g., boreholes, terraprobes, and tank excavations) in the Site interior at specific areas known to be soil source areas were also included in the interior well data set because no monitoring well data were available for these Site locations. Three additional wells to the south (W-1, W-2, and W-6) upgradient of the potential source, do not exhibit site-related contamination, and were therefore not selected for use in the HHRA. Appendix B,

Tables B-4 and B-5 provide summaries of the samples included in the perimeter and interior well data sets, respectively.

2.2.1 Volatile Organic Compounds

Table 3 presents a summary of perimeter well groundwater data, and Table 4 presents a summary of interior well groundwater data. Volatile organic compounds were analyzed in each of the 23 perimeter wells and in each of the 4 interior wells. In addition, VOCs were analyzed in 4 non-monitoring well sampling locations during 1993 and 1995 investigations. There were no rejected data.

2.2.2 Inorganics and Cyanide

Table 3 presents a summary of perimeter well groundwater data, and Table 4 presents a summary of interior well groundwater data. Inorganics were analyzed in 8 of the 23 perimeter wells, and cyanide in 4 of the 23 perimeter wells, at well locations nearest and downgradient to potential source areas for the constituents. There were no rejected data. Only nickel and mercury were analyzed in the interior wells (Table 4) because these two constituents were interpreted to be the primary inorganic contaminants of concern. Analyses for total and dissolved nickel for were performed in 3 of the 4 interior wells. There were no rejected data.

2.2.3 Mercury

Table 3 presents a summary of perimeter well groundwater data, and Table 4 presents a summary of interior well groundwater data. Mercury analyses were performed for each of the 23 perimeter wells and for each of the 4 interior wells. Three of the interior wells were also analyzed for dissolved mercury. One non-monitoring well sample was collected from an open borehole during the 1993 investigation and analyzed for total mercury. There were no rejected data. No mercury speciation analyses or bioavailability testing were performed on groundwater samples.

3.0 PRELIMINARY SCREENING

The purpose of the preliminary screening step is to identify Chemicals of Potential Concern (CPCs) for the HHRA. Chemicals of potential concern are chemicals that are potentially site-related and which may pose risks of concern, for which data of sufficient quality are available for use in the HHRA. Constituents identified as CPCs were retained for development of risk-based concentrations (RBCs). The remaining constituents (i.e., those not selected as CPCs) are not considered to present an appreciable risk and, therefore, were not evaluated further. Chemicals of potential concern for soil and groundwater were selected as described below.

3.1 SOIL

Chemicals of potential concern in soil were identified by comparing the maximum concentrations of constituents included in the soil data set (Table 2) to soil cleanup objectives presented in the TAGM (NYSDEC, 1994) and site-specific screening concentrations developed by ABB-ES using the method specified in the TAGM. The TAGM values are analyte concentrations which represent generic soil cleanup levels for residential land use. Constituents with maximum reported concentrations below the TAGM values were not considered a significant threat to human health and/or the environment. Constituents with maximum reported concentrations exceeding the TAGM values were considered a potential threat to human health and/or the environment and, therefore, were selected as CPCs and included in the of the site-specific risk assessment.

3.1.1 Basis of Soil Cleanup Objectives in the TAGM

The recommended soil cleanup objectives published in the TAGM (NYSDEC, 1994) are based on several parameters, including a) the concentrations corresponding to an excess lifetime cancer risk (ELCR) of 1 in one-million (1x10-6) for Class A and B carcinogens, and 1 in 100,000 (1x10-5) for Class C carcinogens, based on a residential exposure; b) the concentrations corresponding to a hazard index (HI) of 1 for systemic toxicants, based on residential exposure for a child; c) the soil concentrations which are protective of groundwater quality for groundwater used as a source of drinking water (based on New York State drinking water standards); d) soil background values; and e) analytical detection limits. Criteria (a) and (b) were developed by NYSDEC using standard risk assessment equations, considering incidental ingestion exposures only. Criterion (c) was developed using the simple equilibrium partitioning model discussed in Subsection 3.1.2. TAGM soil cleanup objectives were developed by selecting the most stringent cleanup level using criteria a, b, and c for organic compounds, and criteria a, b, or d for inorganic constituents. This value was then compared to analytical detection limits, and the greater of the values was selected as the recommended residential soil cleanup objective published in the TAGM by NYSDEC.

3.1.2 Basis of Site-Specific Screening Concentrations Using the TAGM Approach

For most constituents, the generic TAGM recommended soil cleanup levels developed by NYSDEC (NYSDEC, 1994) are based on soil levels which are protective of groundwater quality because this criterion is associated with lower soil cleanup levels than the direct contact criteria. For organic constituents, this pathway is based on the assumption that

constituents may partition between soil and groundwater and, therefore, leach into groundwater. For inorganic constituents, it is assumed that substantial partitioning would not occur; NYSDEC recommends background soil values in lieu of a value based on partitioning. Because no site-specific soil background data are available for Site soils, the background values for eastern US soils published in the TAGM were used in this evaluation. In accordance with the TAGM, soil cleanup levels protective of groundwater quality for organic constituents are developed by using the following simple equilibrium partitioning model:

$$\textit{Allowable Soil Concentration} = foc \times Koc \times Cw \times CF$$

where: Allowable Soil Concentration is the cleanup level for this pathway
foc is the fraction of organic carbon in the soil
Koc is the soil:water partition coefficient
Cw is the New York Class GA Groundwater Standard (NYSDEC, 1993)
CF is a correction factor

The soil cleanup levels for this criterion, which are provided by NYSDEC in the TAGM, are based on default assumptions for organic carbon content (foc) of 1% and an attenuation correction factor (CF) of 100. For the Ames Street HHRA, the TAGM values used for preliminary screening were modified for site-specific foc by using the average foc value of 1.54% measured in Site soils. A modified CF of 10 was also used.

As described in the TAGM, the CF is applied to account for the amount of chemical that would not be expected to leach to groundwater as a result of natural fate and transport processes. However, the TAGM includes language cautioning against the use of the default CF of 100 if groundwater is shallow (e.g., <3 - 5 feet) with respect to the contaminated soil. In such situations, the TAGM recommends that a modified CF be used in order to provide a conservative evaluation. Available site data indicate that the sources of soil contamination may be within <3 - 5 feet of groundwater. Depth to groundwater varies seasonally at the Site, with an average annual depth to groundwater of approximately 6 feet. During the Phase I investigation, saturated soils were observed within 3 feet of the ground surface although the water table has never been measured this high. The Site is now paved in its entirety and this, in combination with the site-wide stormwater drainage system, is expected to produce a permanent lowering of the groundwater table. Therefore, although no guidance for modifying the CF is provided in the TAGM, a conservative CF of 10 was chosen for use at the Site, based on these site-specific conditions and professional judgment. Because site-specific foc and CF values were incorporated into the calculations for this pathway, the TAGM values for organic constituents used in this evaluation differ from those published in the TAGM (NYSDEC, 1994). Table D-1 (Appendix D) documents the calculations used to develop the adjusted TAGM values.

3.1.3 Screening Results

A comparison of maximum reported soil concentrations to Site-specific TAGM values is presented in Table 2. As indicated in Table 2, the maximum reported concentrations of all chemicals except 1,1,1-trichloroethane and trans-1,2-dichloroethene exceeded the adjusted

TAGM values. Therefore, all remaining constituents were retained as CPCs, and are identified in Table 5.

Chemicals of potential concern are constituents which were detected at concentrations that exceed conservative TAGM screening values. The TAGM values used in the preliminary screening represent extremely conservative values for CPC screening at the Site because they are based on protection of groundwater quality for groundwater used as drinking water and on direct-contact exposures to residents. As described in Subsection 4.2, neither of these exposure pathways exist under current and future land use at the Site; groundwater use and residential use of the Site will be prohibited by deed restrictions. Therefore, the TAGM values are based on exposure scenarios which will not exist at the Site. Because the TAGM values based on these exposure scenarios are protective for the potential future commercial/industrial uses that may occur at the Site, any constituents detected at concentrations which did not exceed TAGM values do not pose a significant risk under the future land use of the Site and thus do not require site-specific Quality Goals to be developed. Constituents identified as soil CPCs were carried forward for in-depth analysis and development of RBCs.

3.2 GROUNDWATER

As described in Subsection 2.2, separate groundwater data sets were developed for groundwater at the perimeter of the site and groundwater at the interior of the site. The screening of perimeter wells is described in Subsections 3.2.1 and 3.2.2. The screening of interior wells is described in Subsections 3.2.3 and 3.2.4.

3.2.1 Screening Criteria - Perimeter Wells

Perimeter well groundwater data were compared to New York Class GA groundwater standards (NYSDEC, 1993). These standards are intended to protect the quality of groundwater so that it can be used as a source of potable water. The groundwater outside the facility perimeter is not currently used as a source of potable water, nor is it anticipated to be used as such in the future.

3.2.2 Screening Results - Perimeter

Table 3 presents a summary of perimeter well groundwater data, and comparison of maximum reported groundwater concentrations to Class GA groundwater standards. The maximum reported concentrations of trichloroethene and mercury exceeded the Class GA groundwater standards. No other constituents were found to exceed groundwater standards. No Class GA groundwater standards were available for cis- or trans-dichloroethene and nickel. However, comparison of maximum reported concentrations of these constituents to federal maximum contaminant levels (MCLs) (USEPA, 1996a), indicated that cis- and trans-dichloroethene and nickel did not exceed drinking water standards.

3.2.3 Screening Criteria - Interior Wells

As discussed in Subsection 4.2, on-site groundwater is not currently, and will not be in the future, used as a potable source of water. The only exposure pathways which could possibly exist for on-site groundwater are inhalation of volatile compounds which may

migrate from groundwater to indoor air or ambient air. Since no standards or guidelines are available for screening this pathway, all volatile constituents detected in interior wells were retained as CPCs. In addition, because perimeter well data represent constituents which are present in on-site groundwater, volatile constituents detected in perimeter wells were retained as CPCs if they were detected at concentrations above Class GA standards. Non-volatile constituents were not retained as CPCs because there is no exposure pathway for these constituents.

3.2.4 Screening Results - Interior Wells

Table 5 presents a summary of the constituents identified as groundwater CPCs. As discussed in Subsection 3.2.3, groundwater CPCs include all volatile constituents detected in interior wells (Table 4) and all constituents detected in perimeter wells at concentrations above Class GA standards. Chromium, lead, nickel, zinc, and cyanide were not retained as CPCs because they are not volatile and, therefore, no exposure pathway to on-site groundwater exists for these constituents. Constituents identified as groundwater CPCs were carried forward for in-depth analysis and development of RBCs.

4.0 RISK-BASED CONCENTRATION DEVELOPMENT

The following sections of this report describe the methods used to develop risk-based concentrations (RBCs) for constituents identified as Chemicals of Potential Concern (CPCs). Risk-based concentrations were developed using guidance presented in Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Standard #1739-95; Nov. 1995; hereafter referred to as the ASTM Risk-Based Corrective Action [RBCA] standard). The NYSDEC RBCA guidance presented in Site Assessment and Guidance for Petroleum Impacted Sites (review draft) (NYSDEC, 1995) is based on the ASTM standard. Although both of these guidelines were developed for use at petroleum-contaminated sites, the principles and methods described are based on USEPA risk assessment methodologies discussed in Risk Assessment Guidance for Superfund (USEPA, 1989), and have been adopted in this HHRA for non-petroleum constituents. Subsection 4.1 presents a toxicity assessment of the constituents identified as CPCs.

Subsection 4.2 presents an exposure assessment, which identifies the current and future land uses of the Site, receptors, and potential soil and groundwater exposure pathways which consider the identified land uses and receptors. This section also presents the methods used to quantify exposures and RBCs. Subsection 4.3 presents RBCs for each of the CPCs for the exposure pathways and receptors identified in Subsection 4.2, and Subsection 4.4 presents an uncertainty discussion. Section 5 presents the selection of the Quality Goals (Qgs).

4.1 TOXICITY ASSESSMENT

The purpose of the toxicity assessment is to evaluate the evidence of potential adverse effects that may be associated with exposure to each CPC. With this information, a relationship between the extent of exposure and the likelihood or severity of adverse human health effects is developed. The toxicity assessment is developed in a two step process: hazard identification and dose-response assessment (USEPA, 1989).

4.1.1 Hazard Identification

Hazard identification is the process of determining whether exposure to an agent can cause an increase in the incidence or severity of a particular adverse health effect (e.g., lung cancer or birth defects) and whether that effect is likely to occur in humans. In this case, hazard is defined as any chemical, substance, or situation at a site that is capable of doing harm to human health. In most cases, the potential toxic effects associated with contaminants detected at hazardous wastes sites have already been identified. Consequently, the objectives of the hazard identification at the Site was to (1) identify which of the contaminants detected at the site are potential hazards, and (2) summarize their potential toxicity in brief narrative profiles. Those constituents selected as CPCs were deemed to present potential hazards. Narrative toxicity profiles for CPCs detected in Site media are presented in Appendix C.

4.1.2 Dose-Response Assessment

The objective of the dose-response assessment is to quantify the relationship between the intake, or dose, of a CPC and the likelihood that a toxic effect may result from exposure to that CPC. There are two major types of toxic effects evaluated in the HHRA: non-carcinogenic, and carcinogenic. Following USEPA guidance (USEPA, 1989), these two effects (non-carcinogenic and carcinogenic) are evaluated separately. Identified dose-response values are used to estimate the likelihood of adverse effects as a function of human exposure to a CPC.

There are two types of dose-response values: cancer slope factors (CSFs) for carcinogens, and reference doses (RfDs) for non-carcinogens. For many compounds, both types of values have been developed by USEPA because many compounds elicit both carcinogenic and non-carcinogenic (systemic) effects. In addition, because the toxicity and/or carcinogenicity of a compound can depend on the route of exposure (i.e., oral or inhalation), unique dose-response values have been developed for the oral and inhalation exposure routes.

The CSF is a chemical-specific toxicity value developed by the USEPA Carcinogen Risk Assessment Verification Endeavor (CRAVE), and is based upon the dose of a chemical and the probability of a carcinogenic response. CSFs have been developed for the oral and inhalation exposure routes.

The RfD is an estimate (with uncertainty spanning an order of magnitude or more) of a daily intake for the human population, including sensitive subpopulations, that is likely to be without appreciable risk of deleterious non-cancer effects during a lifetime. RfDs have been developed for the oral and inhalation exposure routes. For several chemicals, separate sets of RfDs have been developed for evaluating chronic and subchronic exposures. Chronic RfDs are typically used for evaluating exposures lasting more than 7 years. For evaluating exposures less than 7 years but more than 2 weeks (excavation worker), subchronic RfDs are appropriate. Because subchronic RfDs are intended to be used for evaluating exposures considerably shorter than chronic exposures, subchronic RfDs are often associated with higher dose-response values (i.e., tolerant to higher doses of chemical). There are no analogous reference values for evaluating acute exposures, i.e., those lasting less than 2 weeks. Because the commercial/industrial worker is assumed to be exposed for a chronic duration (i.e., 25 years), chronic RfDs were used. In accordance with the ASTM standard for Risk-Based Corrective Action (ASTM, 1995), chronic RfDs were used for the construction worker and utility worker, even though these receptors are assumed to be exposed for only 1 year or less (indicating that subchronic RfDs could be used). This provides an additional degree of conservatism in the evaluation s for the construction worker and utility worker.

Sources of Dose-Response Values. The primary source of dose-response values for the Ames Street HHRA was the USEPA Integrated Risk Information System (IRIS) (USEPA, 1996b), which is an on-line computer database containing health risk and USEPA regulatory information about specific chemicals. Health risk information is included on IRIS only after a comprehensive review of chronic toxicity data is conducted by work groups composed of USEPA scientists. If no information for a given CPC was found in

IRIS, the USEPA Health Effects Assessment Summary Tables (HEAST) (USEPA, 1995a; 1995b, 1995a; 1995b, 1995a; 1995b, 1995a; 1995b, 1995a; 1995b, 1995a; 1995b) were used as a source of information. If appropriate dose-response values were not located from either of these two sources, other USEPA sources (including past versions of IRIS and HEAST and the documents produced by the USEPA's National Center for Environmental Assessment (NCEA; formerly OHEA/ECAO) were consulted. In addition, dose-response values for several CPCs at the Site are published by NYSDEC (NYSDEC, 1995); these values were used when dose-response values were unavailable in other sources. The selected cancer and non-cancer dose-response values for CPCs identified in Site media are presented in the dose-response tables (Appendix C).

4.1.2.1 Mercury Dose-Response Values

For mercury, oral RfDs are available for methyl mercury (i.e., organic mercury) (1x10-4 mg/kg/day), and mercuric chloride (3x10-4 mg/kg/day). Based on the mercury speciation data collected from 14 Site soil samples, mercury on-site is, on average, only 0.3% organic mercury species (0.03% to 0.7%). The remainder of the mercury detected in Site soil is present as inorganic species, primarily elemental mercury (63%: range 17% to 99%), acid soluble mercury species (21%: range 0.2% to 65%), and mercuric sulfide (15%: range 0.7% to 70%). Elemental mercury differs from mercuric chloride and mercuric sulfide because it is not a compound (e.g., inorganic salt), but is the element in pure form. Elemental mercury is virtually non-bioavailable via the ingestion exposure route (see Subsection 4.2.3.3). According to Casarett and Doull's Toxicology: The Basic Science of Poisons, elemental mercury is generally thought to be of little toxicological significance via the oral route (Goyer, 1991). Given the speciation data (summarized in

Appendix A, Table 8), the oral RfD for inorganic mercury was selected for use in this HHRA.

An inhalation RfD is not available for mercury. As discussed in Subsection 4.2.4, RfDs are required for the calculation of RBCs. The absence of an inhalation RfD for mercury introduces a significant data gap into the HHRA because mercury is a volatile substance and there is concern that it may pose inhalation risks at the concentrations reported in soils at the Site. However, an inhalation reference concentration (RfC) is available for elemental mercury (USEPA, 1995a). RfCs are analogous to RfDs and are developed through a similar process. However, unlike RfDs, which represent a dosage (in mg/kg/day) at which adverse or deleterious effects are unlikely, RfCs represent air concentrations (in mg/m3) at which adverse or deleterious effects are unlikely (i.e., an air concentration corresponding to an HI = 1). Non-carcinogenic risks due to inhalation exposures are estimated by comparing the environmental air concentration to the inhalation RfC. The mercury RfC of 3x10-4 mg/m3 represents a lifetime air exposure concentration of elemental mercury which is protective against adverse effects. As described in Subsection 4.2.4.2, the receptor lifetime average air concentration was compared to the mercury RfC.

4.1.2.2 Toxicity Assessment for Lead

Currently, there are no suitable dose-response values for assessing the risks associated with exposure to lead. The USEPA Superfund program has issued a directive that sets soil lead cleanup levels at 400 mg/kg for residential soils USEPA, 1994, 1994, 1994,

1994, 1994, 1994, 1994). This value was used as the soil screening RBC for lead in Site soils, as described in Subsection 4.3.

4.2 HUMAN HEALTH EXPOSURE ASSESSMENT

The exposure assessment is conducted to evaluate the pathways by which humans are potentially exposed, estimate the magnitude of actual and/or potential human exposure, and the frequency and duration of exposure. This process involves several steps: (1) characterization of the exposure setting in terms of physical characteristics and the populations that may potentially be exposed to site-related chemicals, (2) identification of potential exposure pathways, and (3) quantification of exposure for each population in terms of the amount of chemical either ingested, inhaled, or absorbed through the skin from all exposure pathways. This assessment process was performed for both current and assumed future site conditions.

4.2.1 Characterization of Exposure Setting

In characterizing the exposure setting of the HHRA, the physical attributes and demographics of the area near the site are identified. Details pertaining to the physical setting of the Rochester, New York area are discussed in Subsection 2.1 of Volume I.

The physical setting is characterized in terms of the following attributes: climate, meteorology, geology, vegetation, soil type, groundwater, and surface water. This information was gathered from previous investigations and additional information collected during this SI. The information generated from this analysis aids in defining the

physical mechanisms that control or influence how people could be exposed at the site, and the processes which may control the fate and transport of contaminants.

Demographics are characterized for (1) the populations residing or working near the site, (2) the activity patterns of residents and/or workers, and (3) if any exist, the locations of potentially sensitive subgroups. Key to this activity was determining current and foreseeable future land use of the sites and surrounding areas (e.g., residential, commercial and industrial, and recreational). Sources for this information included the following: (1) site visits to the Ames Street site, (2) previous investigations, (3) information generated during the SI, and (4) maps and photographs.

The land in the vicinity of the Ames Street site is best characterized as mixed residential and light industrial. The site, located within the city of Rochester, is bounded on the south by West Avenue, the west by Hague Street, east by Ames Street, and to the north by Conrail railroad tracks. Rochester Gas and Electric operates a facility on the west side of Hague Street, whereas the areas south of West Avenue and east of Ames Street are predominantly residential.

The Site occupies approximately 14 acres. Within the past year, all buildings on the site except one unoccupied building (which will remain through the VSI) have been demolished and the building materials have been disposed of off-site. The Site has been re-graded with clean fill from an off-site source to a slope of no more than 3% and paved. The only persons who are present at the site on a regular basis are ABB-ES field investigation personnel and associated subcontractors. Because the site is paved, there are no direct-contact exposures to soil for these receptors. In addition, the pavement prevents

migration of vapors to ambient air in an substantial quantities, thereby preventing inhalation exposures to volatile compounds. The groundwater beneath the site is not used as a water source at the site. The only exposures that could potentially occur under current site conditions are associated with subsurface utility maintenance; these exposures occur very infrequently.

The future use of the site is anticipated to either remain as it is currently (i.e., vacant and unused) or, more likely, be an industrial or commercial property associated with passive uses such as parking lots and landscaped areas. Under these conditions, commercial or industrial workers would occupy the site. Future uses of the site will be controlled by deed restrictions that specify how the Site may be used under various future commercial/industrial development scenarios, as described in Section 5.

4.2.2 Identification of Exposure Pathways

The purpose of this step in the exposure assessment is to identify all pathways through which individuals may be exposed to site-related contaminants through current and foreseeable future land use. A complete exposure pathway requires four elements: a source or mechanism of chemical release, a transport or retention medium, a point of potential human contact with the contaminated source or medium, and a route of exposure at the point of contact (USEPA, 1989). In some cases, the source of release may be the point of contact, such as direct contact with a spill area.

Potential exposure pathways were determined by first identifying all sources of contamination and the receiving media (i.e., soil and groundwater). Once sources were identified, relevant fate and transport mechanisms were evaluated to predict distant and/or future exposures. Exposure points and exposure routes (i.e., ingestion, inhalation, and dermal absorption) were identified by determining the areas where individuals may potentially come in contact with contaminated media and the likely mechanisms of exposure. All exposure pathways that have these four elements (i.e., a source or mechanism of release, a transport or retention medium, an exposure point where contact can occur, and an exposure route at the point of contact) are considered complete pathways.

The conceptual model (Figure 1) presents the general fate and transport processes that have resulted in contamination of the exposure media. The model also presents a conceptual summary of the potential exposure pathways by which individuals could be exposed to contamination detected in the various exposure media.

For the Site, the sources of contamination are associated with historic manufacturing and related processes at the facility. As described in detail in Section 2 of Volume I, key operations carried out within the facilities occupying the Site included mercury-filled and other instrument manufacturing, electroplating, solvent degreasing, and machining. These processes were conducted for extended periods throughout the early and mid 1900's, and some processes remained until the facility closure in 1992. The activities that were conducted at the site formed the basis of identifying Areas of Concern (AOCs) for investigation during the Phase I VSI, as described in Section 2 of Volume I.

Subsurface soil are the receiving and source media for contaminants associated with operations at the Site. Individuals could be exposed directly to contaminants in the soil through incidental soil ingestion and/or dermal contact with the soil. Additionally, individuals may be exposed to contamination that has migrated from the source areas to various other exposure media. The potential mechanisms for migration of contaminants from soil to other media include volatilization, particulate suspension, and percolation. Although contaminants can also migrate through erosion, runoff, and groundwater discharge to surface water and sediment, the Site is paved and stormwater is directed to subsurface drains and then to the off-site combined sewer system. There is no surface water or sediment present at or adjacent to the Site.

Soil contaminants can theoretically migrate into air through volatilization (primarily volatile organic compounds (VOCs) and mercury) or wind erosion (all constituents). Individuals may be exposed to these contaminants through inhalation of wind-borne particulates, or inhalation of volatiles migrating from the surface to ambient air. Volatile constituents may also migrate from subsurface soils through foundations into indoor air. Some constituents, primarily soluble VOCs and inorganics, can theoretically migrate from soil to groundwater, and be transported in groundwater. From groundwater, constituents may adsorb to subsurface soil in the groundwater vadose zone. Volatile constituents in groundwater may volatilize through the soil overburden into ambient air, or through foundations into indoor air. Individuals can theoretically be exposed to contaminants in groundwater through inhalation of migrated vapor, ingestion of groundwater used as drinking water, and dermal contact and inhalation (volatiles only) during bathing and dishwashing activities.

Based on the conceptual site model (Figure 1), the potential exposure media at the Site include air (ambient and indoor), soil, and groundwater. Exposure routes to these media include inhalation, ingestion, and dermal contact. Only complete exposure pathways were selected for quantitative evaluation in the HHRA (Figure 1). Table 6 presents a summary of the exposure pathways selected for quantitative evaluation. Justifications for the exclusion of other potential pathways are provided in Table 6 and summarized below.

4.2.2.1 Current Land Use

Under current land use, the only exposures to contaminated media at the Site that could possibly occur are to utility workers who may infrequently excavate soils in order to repair utilities. Utility workers may be exposed to contamination in soils via incidental ingestion, dermal contact, and inhalation of wind-borne particulates and vapors during excavation and repair activities. Utility workers could also theoretically be exposed by the inhalation exposure route to vapors migrating from soils and groundwater to ambient air. As indicated in Table 6, these exposure pathways were selected for quantification in this HHRA.

Although ABB-ES personnel and associated subcontractors are frequently at the site, and trespassers could theoretically occur at the site, direct-contact exposures cannot occur because the site is paved. In addition, the pavement and lack of buildings on-site prevent migration of potentially substantial concentrations of volatile compounds to ambient air and thereby prevents inhalation exposures to these contaminants. Therefore, under current site conditions, there are no exposure pathways to contaminated media for receptors other than utility workers.

Groundwater beneath and downgradient of the Site is not used as a source of public or private potable water. Abutters to the Site are supplied with remote public water sources. Therefore, there are no direct-contact exposures to off-site receptors, although there may be potential for indirect contact exposures to groundwater constituents via inhalation of volatile compounds which may migrate from groundwater to indoor air.

4.2.2.2 Future Land Use

The anticipated future land use of the Site is commercial (e.g., retail stores) or industrial (e.g., manufacturing). Deed restrictions will prevent both residential use of the site and building designs which include subsurface structures (e.g., basements). Slab-on-grade structures will be used. In addition, a deed restriction will prevent potable and non-potable use of the groundwater at the site. The future facilities at the site will continue to be served by public water supply, and off-site receptors will continue to be supplied with remote public water sources. Potential exposures to off-site receptors (e.g., site visitors) are not quantified in this assessment because the exposure frequency and duration would be much lower than for a commercial/industrial worker (i.e., employee) at the site.

Based on discussions with consultants, developers, and City of Rochester Economic Development officials, under all reasonably foreseeable future commercial/industrial land uses, areas of the Site will be variously occupied by buildings, pavement, and landscaping. These development scenarios would substantially reduce the potential for any commercial/industrial worker direct-contact exposures or inhalation exposures to volatiles migrating from subsurface soil or groundwater to ambient (outdoor) air. Therefore, it is unlikely that any commercial or industrial workers or site visitors would be exposed to

soils via direct contact, particularly at the intensity evaluated in this HHRA. However, direct contact and vapor inhalation exposures were evaluated for commercial/industrial workers that are assumed to work outdoors and be exposed to contaminated soils in order to provide perspective for remedial decision-making (Subsection 5.1.1). Commercial/industrial workers that work indoors may be exposed to VOCs migrating from subsurface soil and groundwater to indoor air. Inhalation exposures to volatiles from this pathway would be more substantial than exposures to volatiles migrating to ambient air. As described in Subsection 4.4.3, these exposure scenario provides a conservative assessment.

Construction workers may be exposed to contamination in soils via incidental ingestion, dermal contact, and inhalation of wind-borne particulates and vapors during excavation and construction activities. Construction workers could also theoretically be exposed by the inhalation exposure route to vapors migrating from soils and groundwater to ambient air. As indicated in Table 6, these exposure pathways were selected for quantification in this HHRA. Because construction workers will not work inside a building (that they constructed) for more than a very short period, exposures to volatiles in indoor air were not evaluated for the construction worker.

In areas of the site that are left undeveloped (i.e., unchanged from the current use as a paved lot), or should the entire site be left undeveloped, the only receptors that may potentially be exposed to soils are utility workers who may infrequently be required to repair or maintain utilities beneath the Site. Utility workers may be exposed to contamination in soils via incidental ingestion, dermal contact, and inhalation of wind-borne particulates and vapors during excavation activities. Utility workers could also

theoretically be exposed by the inhalation exposure route to vapors migrating from soils and groundwater to ambient air. As indicated in Table 6, these exposure pathways were selected for quantification in this HHRA.

As indicated in Figure 1, another potential exposure pathway at the Site is the leaching of contaminants from soil to groundwater (all contaminants), where direct contact exposures to contaminants that have leached from soil could theoretically occur through potable and non-potable use of the groundwater and contaminant migration to ambient air and indoor air (VOCs and mercury). This exposure pathway was evaluated in the preliminary screening step (Section 3) because it is one of the criteria upon which TAGM values are based. However, exposures to soil and groundwater contaminants would not be as substantial through this pathway as they would be through either direct contact with soil or direct migration of soil and groundwater volatiles to air. Therefore, this exposure pathway was not selected for quantitative evaluation in this HHRA. In addition, the migration of soil contaminants to groundwater used as drinking water is not a complete exposure pathway because groundwater is not currently, nor will be in the future, used as source of drinking water.

4.2.3 Quantification of Exposure

Once complete exposure pathways are selected for evaluation, the final step of the exposure assessment is to quantify exposure (i.e., intake) for each pathway. This quantification process involves developing exposure scenarios and calculating intakes to estimate the total amount of contaminants that a hypothetical receptor may ingest, dermally absorb, and/or inhale from each exposure pathway. These exposure scenarios

are based on several variables, that can be grouped into population-, assessment-, and chemical-related variables. The ultimate goal of this step, as defined in USEPA guidance, is to identify the combination of these exposure variables or parameters that results in the most intense level of exposure that may "reasonably" be expected to occur under current and future site conditions (USEPA, 1989; ASTM, 1995). This is performed for every complete exposure pathway selected for evaluation. The resulting exposure scenarios are referred to as the Reasonable Maximum Exposure (RME) for each exposure pathway.

The following exposure scenarios were identified as the RME scenario for each pathway, and were selected for quantitative evaluation:

- Direct soil contact (ingestion, dermal, inhalation of dust and vapor)
  - commercial/industrial worker, construction worker, utility worker
- Inhalation of vapors from subsurface soil to indoor air
  - commercial/industrial worker
- Inhalation of vapors from surface soil to ambient air
  - commercial/industrial worker
- Inhalation of vapors from groundwater to indoor air
  - commercial/industrial worker

- Inhalation of vapors from subsurface soil to ambient air
  - construction worker and utility worker

- Inhalation of vapors from groundwater to ambient air
  - construction worker and utility worker

Subsection 4.2.4 describes the methods used to quantify exposures for these RME scenarios.

4.2.3.1 Population-Related Variable

Population-related variables describe the characteristics of a hypothetical individual receptor within each potentially exposed population. Hypothetically exposed populations were identified through analysis of exposure setting and exposure pathway information (see Subsection 4.2.2). Population-related variables include contact rates, such as exposure frequencies and ingestion rates, and physical characteristics of human bodies, such as body weights and surface areas. The population-related variables for the receptors evaluated in this HHRA (i.e., commercial/industrial worker, construction worker, and utility worker) were the Tier 1 Default RME Factors recommended in the ASTM RBCA standard (ASTM, 1995). These values are presented in Table 7. For the utility worker, the construction worker exposure values were used for all parameters except exposure frequency and exposure duration, which are described in Subsection 4.2.3.2.

4.2.3.2 Assessment-Related Variable

The assessment-related variable involved in exposure quantification is the averaging time. Averaging time reflects the duration of exposure and depends on the type of effect being evaluated. Exposure intake during a defined interval (e.g., a lifetime) is averaged over the entire period, resulting in an estimate of average daily intake.

There are generally two types of adverse health effects evaluated in HHRA: carcinogenic effects and non-carcinogenic effects. According to USEPA and ASTM guidance, the averaging time for carcinogenic effects is assumed to be a 70-year lifetime (USEPA, 1989, ASTM, 1995). This averaging time is used to evaluate carcinogenic effects for all receptors, regardless of the length of the receptor-specific exposure period. The averaging times for non-carcinogenic effects are equivalent to the duration of exposure and may vary depending on the nature of exposure. There is a wide range of possible estimates, from a day to a lifetime. However, based on USEPA guidance, exposure durations for non-carcinogenic effects can roughly be categorized into one of three periods: (1) chronic exposures of 7 years to a lifetime; (2) subchronic exposures of 2 weeks to 7 years; and (3) acute exposures of less than 2 weeks (USEPA, 1989). The length of the exposure period depends on the potentially exposed population and the characteristics of exposure. For the commercial/industrial worker, the exposure period was assumed to be 25 years, representing the upper bound estimate of employment duration at a single location. For the construction worker, the exposure period was assumed to be one year, representing a conservative estimate of the duration of most excavation projects. For the utility worker, the exposure period was assumed to be one-

month, representing a conservative estimate of the duration of a utility repair or maintenance project.

4.2.3.3 Chemical-Related Variable

The chemical-related variable is the chemical and physical data that are specific to each CPC. These data are used to describe chemical fate and transport characteristics and toxicity characteristics. Chemical fate and transport characteristics are used to model chemical movement among various exposure media, which provide the basis for calculating chemical intakes. Chemical-specific data used in this HHRA included variables such as Henry's law constant, soil:water partition coefficient, and air and water diffusion coefficients. Physical data included parameters such as depth to groundwater, soil porosity, and soil water content. The chemical and physical data used to model chemical fate and transport were generally based on default values provided by ASTM (1995), but were substituted for in some cases by site-specific data or default values recommended by NYSDEC (1995). These data are documented in Appendix D, Table D-5 and discussed further in Subsection 4.4.3.2. Chemical-specific toxicity characteristics are discussed in Subsection 4.1.

Relative Absorption Factors. The incidental ingestion and dermal contact components of the direct-contact equations (described in Subsection 4.2.4.1) incorporate a relative absorption factor (RAF) parameter. The RAF is incorporated to compensate for the differences in oral and dermal absorption efficiency between the exposure medium at the site (i.e., soil) and the exposure medium in which the analyte was administered during the toxicity study upon which the dose-response value was based (e.g., water). This

adjustment is performed because, for many chemicals, USEPA bases dose-response values on applied (i.e., administered) doses rather than absorbed doses. Therefore, if absorption differs between the exposure medium evaluated in the risk evaluation and the exposure medium upon which the dose-response value is based, the administered dose calculated in the exposure assessment and the administered dose representing the dose-response value are not directly comparable. Because it is the absorbed dose which is available for biological interaction, risk estimates may be over- or under-estimated if differences in absorption are not accounted for.

ASTM recommends oral absorption factors of 100% for all constituents and dermal absorption factors of 0.5% for VOCs, but does not provide dermal absorption factors for inorganic constituents (ASTM, 1995). However, USEPA Region IV suggests a default dermal absorption factor for inorganics of 0.1% (USEPA, 1995c). For all constituents except mercury, the NYSDEC and USEPA Region IV default absorption factors were used in this HHRA in the absence of other information. For mercury, a site-specific oral bioavailability factor was developed as described below, and dermal absorption data were obtained from the literature and used to calculate a dermal intake, as described in Subsection 4.2.4.1.

Mercury Bioavailability Adjustment. Because mercury has been identified as a primary contaminant at the Site, and because substantial evidence exists that mercury bioavailability varies with mercury species, a site-specific oral bioavailability factor has been developed for mercury in the Site soils. The bioavailability factor was developed by the ABB-ES subcontractor PTI Environmental Services, Inc. (Boulder, CO) using 10 soil samples collected from various depths and locations at the Site (see Subsection 2.1.4.3).

Methods and results of the bioavailability testing are summarized below, and method limitations associated with this approach are discussed in Subsection 4.4.1.2. Details of the methods and results of the bioavailability test are presented in the PTI report Development of a Bioavailability Adjustment Factor for Mercury in Soils at the Ames Street Site, Rochester, New York, included in its entirety as Appendix A.

As discussed previously, bioavailability factors (BAFs) and RAFs are used to adjust exposure equations such that the calculated intake from an environmental medium is comparable to the intake upon which the dose-response value is based. Therefore, applying BAFs or RAFs, the absorbed dose of toxicant associated with a toxicological response is evaluated against the absorbed dose of toxicant that may be received from exposure to an environmental medium. For mercury, toxicological data indicate that the solubility and bioavailability of certain mercury species such as elemental mercury and mercuric sulfide are minimal (e.g., less than 1%), whereas the bioavailability and solubility of other species such as mercuric chloride are substantially higher (e.g., more than 20%). The oral RfD is based on studies in which laboratory animals were exposed to mercuric chloride dissolved in water. It is appropriate to directly compare intakes of soluble mercury species to this oral RfD. To compare intakes of less bioavailable mercury species with this oral RfD without adjustment for bioavailability would result in overestimates of potential risk.

To provide data for use in adjusting the oral intake dose, bioavailability testing was conducted. The bioavailability test is an in vitro assay which simulates gastrointestinal digestion by a human child. Soil sieved to <250 mm is introduced to an extraction vessel which contains fluid resembling the conditions of a fasted stomach. The system is

incubated for one hour, during which extract samples collected at 30 minutes and one hour are analyzed for mercury. The system is then adjusted to conditions resembling the small intestine and incubated for an additional 4 hours (to result in a total of 5 hours incubation time). Two additional extract samples collected at 3 hours and 5 hours are analyzed for mercury. The ratio of mercury concentration present in the sample extract to the total mercury concentration present in the soil sample is interpreted as the bioavailable fraction. A positive control, consisting of mercuric chloride dissolved in water, is evaluated to establish the "baseline" conditions upon which the oral RfD was based.

The *in vitro* assay actually provides an evaluation of bioaccessibility rather than bioavailability. Bioaccessibility refers to the amount of constituent that is available for potential absorption into the bloodstream, whereas bioavailability refers to the amount of constituent which is actually absorbed into the bloodstream. Because mercury in soil must be in a soluble form in order for it to be absorbed into the blood stream, the evaluation of soluble mercury in the in vitro assay provides an estimate of bioaccessibility. This is used to conservatively represent an upper-bound estimate of potential bioavailability.

Results of the bioavailability testing for individual samples are presented in Appendix A, Table 18 and summarized in Appendix A, Figure 6. The average bioavailability in the intestinal phase was 14% (2% to 24%) for all soil samples tested. A general increase in bioavailability occurred over time during the in vitro assay, with mercury becoming more bioavailable in the intestinal phase than during the stomach phase. As discussed in Appendix A, the bioavailability test used an upper bound intestinal residence time to provide a conservative evaluation of potential bioavailability. Mercury bioavailability tended to decrease with increasing mercury concentrations (r2=0.57) (Appendix A, Figure

8), and it is believed this was due to an increasing fraction of total mercury present as elemental mercury. This was evidenced by the soil samples associated with the lowest bioavailability, which contained 99% elemental mercury (as determined by the sequential extraction results). As described in Appendix A, these findings are consistent with the mercury speciation results (Appendix A, Table 8), which indicated approximately 78% of the mercury is present in minimally bioavailable species (63% elemental mercury, 15% mercuric sulfide), and only 22% is present as species with greater potential bioavailability (e.g., mercuric chloride and organic mercury).

Because the relationship between increasing mercury concentration and decreasing bioavailability was observed, the bioavailability adjustment factor was developed for soils with lower mercury concentrations because they had higher bioavailability. In addition, since there were no correlations between the bioavailability results and the sample locations within the site (Appendix A), a single BAF was developed for use with all soils at the site. Using the linear correlation presented in Appendix A, Figure 8, a soil mercury bioavailability of 16% was selected for the calculation of the BAF. This mercury bioavailability corresponds to Site soils containing between 0 and 520 mg/kg mercury. Soils containing mercury within this concentration range do not present a risk greater than a hazard index of 1 (which is the NYSDEC acceptable non-cancer risk level) to future commercial/industrial workers, under the very conservative default assumption that the mercury is 100% bioavailable. The BAF was calculated by dividing the mercury bioavailability for Site soils (16%) by the mercury bioavailability of mercuric chloride in water (the mercury species and exposure medium which the mercury oral dose response value is based on). The bioavailability of mercuric chloride in water was determined to be 78% in the positive control sample tested in the in vitro assay. The resulting BAF is 0.2.

The mercury bioavailability factor was applied as the RAF to the incidental ingestion intake equation described in Subsection 4.2.4.1. The approach to developing and using the BAF to adjust oral mercury exposure estimates in this HHRA is consistent with the BAF adjustments approved at other sites where mercury was a CPC for which Records of Decision (RODs) have been signed, including the Almaden Quick Silver County Park in Los Gatos, CA (CDM, 1992), Lower East Fork Poplar Creek (Oak Ridge Department of Energy Plant) (DOE, 1995), Oakridge, TN, and the Carson River Mercury Site, Lyon/Churchill County, NV (USEPA, 1995d). As described in Appendix A and Subsection 4.4.1.2, the bioavailability factor derived in the in vitro assay is unlikely to underestimate the bioavailability of mercury in Site soils.

4.2.4 Calculation of Exposures and Risk-Based Concentrations

This section describes the methods used to quantify exposures and calculate risk-based concentrations for each of the RME scenarios. The final RBCs are presented and discussed in Subsection 4.3. As discussed previously, the purpose of this HHRA is to develop Quality Goals (QGs) based on RBCs. The approach for developing RBCs requires that chemical intakes be quantified for a specific level of risk. The intake associated with a fixed level of risk can then be used to develop medium-specific chemical concentrations which correspond to the chosen level of fixed risk, or target risk. This is referred to as a RBC. These two components are addressed using a single equation which calculates a pathway-specific chemical concentration that is associated with a chemical intake corresponding to a fixed level of risk. In accordance with NYSDEC guidance (NYSDEC, 1995), the target risks upon which screening RBCs for each CPC were

developed were an ELCR of 1x10-6 for carcinogenic effects, and a HI of 1 for non-carcinogenic effects.

Exposures were quantified using the approach described in the ASTM RBCA standard (ASTM, 1995). As indicated previously, the NYSDEC RBCA guidance is based on the ASTM guidance.

4.2.4.1 Direct-Contact Exposures

Direct-contact exposures to soil were calculated for the commercial/industrial worker, construction worker, and utility worker exposure scenarios. The equations used to calculate intake are those presented in the ASTM RBCA standard (ASTM, 1995)(ASTM, 1995)(ASTM, 1995)(ASTM, 1995)(ASTM, 1995)81(ASTM,1995.

The equation for calculating RBCs for non-carcinogenic CPCs is as follows:

$$RBC_s = \frac{TR * BW * AT * 365}{EF * ED[(SF_o * 10^{-}6(IR_s * RAF_o + SA * M * RAF_d)) + (SF_i * IR_a (VF_{ss} + VF_p))]}$$

$$RBC_s = \frac{THI * BW * AT * 365}{EF * ED\left[\frac{10^{-6} * (IR_{soil} * RAF_o + SA * M * RAF_d)}{Rfd_o} \quad \frac{(IR_a * (VF_{ss} + VF_p))}{RfD_i}\right]}$$

where:

RBCs	=	Risk-based concentration soil [mg/kg]
SFo	=	Chemical-specific oral cancer slope factor [(mg/kg-day)-1]
IRs	=	Soil ingestion rate [mg/day]
RAFo	=	Chemical-specific oral relative absorption factor [---]
SA	=	Skin surface area [cm2/day]
RAFd	=	Chemical-specific dermal relative absorption factor [---]
M	=	Soil to skin adherence factor [mg/cm2]
TR	=	Target cancer risk (---)
BW	=	Body weight (kg)
AT	=	Averaging time (yr)
EF	=	Exposure frequency (day/yr)
ED	=	Exposure duration (yr)
SFi	=	Chemical-specific inhalation cancer slope factory [(mg/kg-day)-1)]
IRa	=	Inhalation rate (m3/day)
VFss	=	Volatilization factor - volatiles (mg/m3)(Appendix, Table D-5.2)
VFp	=	Volatilization factor - particulates (mg/m3) (Appendix, Table D-5.2)
RfDo	=	Chemical-specific oral reference dose [mg/kg-day]
RfDi	=	Chemical-specific inhalation reference dose [mg/kg-day]
THI	=	Target hazard index for individual constituents [--]

The direct-contact exposure equation accounts for incidental ingestion, dermal contact, and inhalation of fugitive dusts and vapors emitted from the surface to ambient air. For this exposure scenario, the cumulative intake from all exposure routes corresponds to the target risk (therefore the intake associated with a single exposure route, such as ingestion, will be below the target risks). Fugitive dust and vapor emissions were estimated using the modeling techniques presented in Appendix D, Table D-5. Calculations of the RBCs for direct-contact exposures, for the commercial/industrial worker, construction worker, and utility worker are presented in Appendix D, Tables D-2, D-3, and D-9 respectively.

Dermal exposure estimate for mercury. Because mercury is a primary contaminant of concern at the Site, dermal absorption data were obtained from the literature and used to develop a dermal exposure estimate. The dermal absorption data came from a study conducted by Hursh, et al (1989), who evaluated the percutaneous absorption of elemental mercury vapor in five human subjects. Following 35 minute exposures (average) of 382 cm2 areas of forearm skin (average) to mercury vapor concentrations of 1.61 ng/cm3 (average), the average uptake rate was calculated to be 0.024 ng/cm2-min/(ng/cm3), and the average maximum systemic level of mercury was measured at 40% of the mercury concentration which was deposited on the skin following exposures to mercury vapor. Using these data, Hursh et al (1989) determined that dermal exposure to mercury vapor contributed approximately 2.6% of the exposure which would be received from inhalation exposure to mercury vapor.

Using the approach presented by Hursh et al (1989) to estimate dermal exposure, the dermal exposure can be estimated for the site-specific conditions in this HHRA using the following equation:

$$DE = UR * SA * ET * VC * CF$$

where:

DE	=	Dermal Exposure (mg/workday)
UR	=	average Uptake Rate [0.024 (ng/cm2)/(ng/cm3)]
SA	=	exposed skin Surface Area of receptor (3160 cm2; see Table 7)
ET	=	Exposure Time (8 hours per workday)
VC	=	Vapor Concentration (20 ng/cm3)
CF	=	Conversion Factor (60 minutes/hour)

The vapor concentration used in this assessment (20 ng/cm3) represents the approximate saturation vapor concentration of mercury at the elevation of the Site. The saturation vapor concentration was chosen because it represents the maximum vapor concentration to which skin could be exposed. The saturation concentration may be achieved if mercury droplets were introduced into clothing, shoes, and socks. Applying these inputs to the above equation results in a dermal dose of 0.728 mg/day. This value represents the upper bound on dermal exposures of mercury vapor regardless of what the soil elemental mercury concentration is. This value was incorporated into the direct contact equation above by substituting it for the terms $SA * M * RAF$.

4.2.4.2 Inhalation-Indoor Air

Inhalation exposures to vapors migrated from soil and groundwater to indoor air were calculated for the commercial/industrial worker exposure scenario. Exposure calculations for these pathways included two components: 1) calculation of indoor air exposures corresponding to the target risk, and 2) calculation of the indoor air vapor concentrations from soil and groundwater. The equations used to calculate indoor air exposures are those presented in the ASTM RBCA standard (ASTM, 1995). The equation for calculating exposures to carcinogenic volatile CPCs is as follows:

$$RBC_a = \frac{TR * BW * AT * 365}{IR * ED * EF * SF_i}$$

The equation for calculating exposures to non-carcinogenic volatile CPCs is as follows:

$$RBC_a = \frac{THI * BW * AT * 365 * RfD_i}{IR * ED * EF}$$

where:

RBCa	=	Risk-based concentration in air [mg/m3]
TR	=	Target cancer risk for individual constituents [--]
BW	=	Body weight [kg]
AT	=	Averaging time [years]
IR	=	Inhalation rate [m3/day]
ED	=	Exposure duration [years]

EF	=	Exposure frequency [days/year]
SFi	=	The chemical-specific inhalation slope factor [(mg/kg-day)-1]
RfDi	=	The chemical-specific inhalation reference dose [mg/kg-day]
THI	=	Target hazard index for individual constituents [--]

Calculation of indoor air concentrations for the commercial/industrial worker is presented in Appendix D, Table D-4. These concentrations were then compared to the estimated vapor concentrations in indoor air that may result from vapor migration from soil or groundwater, to obtain RBCs for soil and groundwater.

For soil, the equation for calculating RBCs based on vapor migration to indoor air is as follows:

$$RBC_s = \frac{RBC_a}{VF_{sesp}}$$

For groundwater, the equation for calculating RBCs for vapor migration to indoor air is as follows:

$$RBC_w = \frac{RBC_a}{VF_{wesp}}$$

where:

RBC_s	=	Risk-based concentration for inhalation of vapors from subsurface soils [mg/kg-soil]

RBC_a = Risk-based concentration for inhalation of air [mg/m^3-air] (Appendix D, Table D-4)

VF_{sesp} = Volatilization factor from subsurface soil to enclosed space (indoor) air [(mg/m^3-air)/(mg/kg-soil)] (Appendix D, Table D-5.3)

RBC_w = Risk-based concentration for inhalation of vapors from groundwater [mg/L-H_2O]

VF_{wesp} = Volatilization factor for groundwater to enclosed space (indoor air) [(mg/m^3 air)/(mg/L-water)(Appendix D, Table D-5.3).

The vapor emissions were estimated using the modeling techniques presented in Appendix D, Table D-5. Calculations of the RBCs for commercial/industrial worker inhalation exposures to VOCs migrated from soil and groundwater are presented in Appendix D, Table D-6.

Air exposure estimate for mercury. As discussed in Subsection 4.1.2.1, an inhalation RfD is not available for mercury and, therefore, the inhalation exposure estimates for mercury were based on an RfC. Unlike the RfD, which represents an exposure dose (in mg/kg/day), the RfC represents a life-time exposure (i.e., 24 hours per day, 365 days per year) air concentration (in mg/m3) corresponding to a target risk of HI=1. Therefore, when using the RfC, air exposure concentrations corresponding to a fixed risk are not calculated as an air concentration corresponding to a dose, as is done when using RfDs. Rather, the air exposure concentration is simply modified to represent the exposure variables of the receptor evaluated, as shown in the following equation:

$$WPAC = EAC \times \frac{ET_{receptor} * ED_{receptor}}{ET_{RfC} * ED_{RfC}}$$

where:

WPAC	=	Workplace Air Concentration for receptor exposure conditions
EAC	=	Environmental Air Concentration representing conditions upon which the RfC was based (calculated in Appendix D, Table D-5)
ET_{RfC}	=	daily Exposure Time upon which the RfC is derived (24 hours/day)
EF_{RfC}	=	yearly Exposure Frequency upon which the RfC is derived (365 days)
$ET_{receptor}$	=	daily Exposure Time of receptor (8 hours/day for commercial/industrial, construction, and utility worker)
$EF_{receptor}$	=	yearly Exposure Frequency of receptor (250 days for commercial/industrial or construction worker; 22 days for utility worker)

Using the volatile emission concentrations for mercury developed in Appendix D, Table D-5 and the above exposure parameters, workplace adjusted target air concentrations were obtained for the construction worker and commercial/industrial worker. These air concentrations were used as the indoor and outdoor air concentrations for these receptors, and were compared to mercury RfC.

4.2.4.3 Inhalation-Ambient Air

Inhalation exposures to vapors migrated from subsurface soil and groundwater to ambient air were calculated for the construction worker and utility worker exposure scenarios. In addition, inhalation exposures to vapors migrated from groundwater to ambient air were calculated for the commercial/industrial worker. Exposure calculations for these pathways included two components: 1) calculation of ambient air exposures corresponding to the target risk, and 2) calculation of the ambient air vapor concentrations from soil and groundwater. The equations used to calculate ambient air exposures for the construction and utility workers are those presented in the ASTM RBCA standard (ASTM, 1995), and are the same as those used to calculate air exposures for the commercial/industrial worker.

Calculation of ambient air concentrations for the construction worker and utility worker are presented in Appendix D, Tables D-7 and D-10, respectively. The mercury air exposure estimate for these receptors was developed as described in Subsection 4.2.4.2. These concentrations were then compared to the estimated vapor concentrations in ambient air that may result from vapor migration from soil or groundwater, to obtain RBCs for soil and groundwater.

For soils, the equations for calculating RBCs based on vapor migration to ambient air are as follows:

$$RBC_s = \frac{RBC_a}{VF_{samb}}$$

$$RBC_s = \frac{RBC_a}{VF_{ss}}$$

For groundwater, the equation for calculating RBCs for vapor migration to ambient air is as follows:

$$RBC_w = \frac{RBC_a}{VF_{wamb}}$$

where:

RBC_s	=	Risk-based concentration for inhalation of vapors from subsurface soils [mg/kg-soil]
RBC_a	=	Risk-based concentration for inhalation of air [mg/m^3-air] (Appendix D, Tables D-4, D-7 and D-10)
VF_{samb}	=	Volatilization factor from subsurface soil to ambient (outdoor) air [(mg/m^3-air)/(mg/kg-soil)] (Appendix D, Table D-5.3)
VF_{ss}	=	Volatilization factor from surface soil to ambient (outdoor) air [(mg/m^3-air)/(mg/kg-soil)] (Appendix D, Table D-5.2)

RBC_w = Risk-based concentration for inhalation of vapors from groundwater [mg/l-H_2O]

VF_{wamb} = Volatilization factor from groundwater to ambient air [(mg/m^3-air)/(mg/L-water)] (Appendix D, Table D-5.3)

The vapor emissions were estimated using the modeling techniques presented in Appendix D, Table D-5. Calculations of the RBCs for construction worker, utility worker, and commercial/industrial inhalation exposures to VOCs migrated from soil and groundwater are presented in Appendix D, Tables D-6, D-8 and D-11, respectively.

4.3 RISK-BASED SCREENING CONCENTRATIONS

This section presents the risk-based screening concentrations for CPCs in Site soil and groundwater. As described in Subsection 4.2.3, RBCs were developed for three receptors, the commercial/industrial worker, construction worker, and utility worker for direct-contact exposures to soil, and inhalation exposures to vapors migrated from soil and groundwater. Risk-based screening concentrations were developed for a target cancer risk of 1x10-6 and a target non-cancer HI of 1 in accordance with NYSDEC (1995) guidance. These RBCs are used in Section 5 as screening RBCs for soil and groundwater, to identify those CPCs which, under very conservative assumptions, will need Quality Goals. As discussed in Section 5, QGs were then developed.

4.3.1 Soil

Risk-based screening concentrations for soil are presented in Tables 8 through 10. Risk-based screening concentrations for the future commercial/industrial worker are presented for direct-contact exposures to soil and inhalation exposures to volatile chemicals which may migrate from subsurface soil to indoor air or surface soil to outdoor air, for a cancer risk level of 1x10-6 and a non-cancer risk of HI = 1 (Table 8). Risk-based screening concentrations for the future construction worker and utility worker are presented for direct-contact exposures to soil and inhalation exposures to volatile chemicals which may migrate from subsurface soil to outdoor air, for a cancer risk level of 1x10-6 and a non-cancer risk of HI = 1 (Tables 9 and 10, respectively).

Risk-based screening concentrations for each CPC were developed for each exposure pathway and risk endpoint appropriate for that CPC. Risk-based screening concentrations for direct contact exposures, which include the soil ingestion, dermal contact, and inhalation of volatile and fugitive dust exposure routes, were developed for all CPCs, with the exceptions noted below. Risk-based screening concentrations for inhalation exposures to volatiles which may migrate from subsurface soil to indoor air or outdoor air were developed only for volatile CPCs because this exposure pathway is not applicable to non-volatile constituents (e.g., inorganics and cyanide). Risk-based screening concentrations for non-cancer effects were developed for all CPCs for which non-cancer dose-response values were available. Inhalation non-cancer screening RBCs for the volatile migration pathway were not developed for trichloroethene and tetrachloroethene because inhalation RfDs were unavailable for these CPCs. For this same reason, the direct-contact non-

cancer screening RBCs for these four CPCs do not include risks from volatile and fugitive dust inhalation exposures.

As indicated in Tables 8 through 10, there are only two VOC CPCs (tetrachloroethene and trichloroethene) and three inorganic CPCs (chromium, nickel, and lead) which are considered potentially carcinogenic. Trichloroethene and tetrachloroethene are potentially carcinogenic via the oral (ingestion) and inhalation exposure routes. Cancer-based screening RBCs for both the direct contact and volatile migration exposure pathways were developed for these CPCs. Chromium and nickel are considered potentially carcinogenic by the inhalation exposure route, and not the oral (ingestion) exposure route. Therefore, for these two CPCs, direct contact screening RBCs for cancer effects are based on cancer risks for the fugitive dust exposure pathway only, not the ingestion and dermal pathways. Because these two CPCs are not volatile, no screening RBCs were developed for the volatile migration pathway.

As discussed in Subsection 4.1.2.2, the lead value presented in Tables 8 through 10 is not a RBC, but rather the USEPA OSWER lead cleanup value (USEPA, 1994). This value represents a protective concentration for children exposed by direct contact to lead in soil, including oral, dermal, and fugitive dust exposures. Because the OSWER residential lead cleanup value is based on residential exposures to children, the value is not applicable to the future commercial/industrial use of the Site. A site-specific lead value based on biokinetic modeling for adult workers would be substantially higher than the residential lead soil cleanup value. Therefore, although the OSWER residential value is presented as a screening value for lead, it will not be used as the soil QG for lead at the Site.

The screening RBCs presented for each CPC represent the range of screening RBCs that may potentially be selected as basis of the Quality Goals. Subsection 4.4 (Uncertainty Evaluation) examines and evaluates the uncertainties associated with the various components of the HHRA. Section 5 (Selection of Quality Goals) identifies a single screening RBC for each receptor land use scenario (from the screening RBCs presented in this section) for comparison to soil data to determine which soil CPCs may require the development of Quality Goals.

4.3.2 Groundwater

Risk-based screening concentrations for groundwater are presented in Tables 11 through 13. As discussed in Subsection 4.2, the only complete groundwater exposure pathways are inhalation of vapors from volatile constituents which may migrate from groundwater to indoor or ambient air. Commercial/ industrial worker screening RBCs are presented for inhalation exposures to indoor air, for a cancer risk level of 1x10-6 and a non-cancer risk level of HI = 1. Construction worker and utility worker screening RBCs are presented for inhalation exposures to outdoor air, for a cancer risk level of 1x10-6 and a non-cancer risk level of HI = 1.

For each groundwater CPC, screening RBCs were developed for the risk endpoint appropriate for that CPC. Screening RBCs for inhalation exposures to volatiles which may migrate from groundwater to indoor air or outdoor air were developed only for volatile CPCs; non-volatile constituents (e.g., inorganics and cyanide) were not selected as CPCs for groundwater because no exposure pathway for these constituents is present at the Site (Subsection 3.2.4).

Risk-based screening concentrations for non-cancer effects were developed for all CPCs for which non-cancer inhalation dose-response values were available. Non-cancer screening RBCs were not developed for vinyl chloride, trichloroethene, tetrachloroethene, and benzene because inhalation RfDs were unavailable for these CPCs. As indicated in Tables 11 through 13, these four CPCs are the only VOCs which are considered potentially carcinogenic via the inhalation exposure route. Cancer-based screening RBCs were developed for these four CPCs. In addition, no screening RBCs were developed for trimethylbenzenes, n-butylbenzene, or dichloroethene because no inhalation dose-response values are available to evaluate indirect inhalation exposures to these CPCs in groundwater.

The screening RBCs presented for each CPC represent the range of RBCs that may potentially be selected as basis of the QGs. As indicated in Tables 11 through 13, the calculated screening RBCs for some CPCs exceed water solubility limits. Subsection 4.4 (Uncertainty Evaluation) examines and evaluates the uncertainties associated with the various components of the HHRA. Section 5 (Selection of Quality Goals) identifies a single screening RBC for each receptor land use scenario (from the screening RBCs presented in this section) for comparison to groundwater data to determine which groundwater CPCs may require the development of Quality Goals.

4.4 UNCERTAINTY ANALYSIS

The interpretation and application of the risk-based concentrations (RBCs) developed in this HHRA should be performed with the understanding that the RBCs are conservative values resulting from multiple layers of conservative assumptions inherent in the risk

assessment process. The majority of "uncertainties" identified and referred to in this HHRA relate to the use of conservative assumptions in lieu of site-specific data. Quantitative estimates of exposure that correspond to acceptable levels of risk are based on numerous "uncertainties", most of which are conservative assumptions intended to be protective of human health. As such, RBCs are conditional estimates given a series of conservative assumptions about exposure and toxicity. It is unlikely that any RBCs developed in this HHRA underestimate risk.

A thorough discussion of all potential sources of uncertainty in this risk assessment is not feasible. In general, sources of uncertainty can be categorized into those associated with data evaluation, toxicity assessment, and exposure assessment. Together, these uncertainties characterize the conservative assumptions in the calculated RBCs, and provide information for formulating risk management decisions.

4.4.1 Uncertainties Associated with Data Evaluation

The analytical data presented in this HHRA were used for three purposes: 1) to identify Chemicals of Potential Concern (CPCs), 2) to determine the speciation of mercury in Site soils, and 3) to develop a site-specific bioavailability adjustment factor for mercury. Uncertainties associated with these data are discussed in the following subsections.

4.4.1.1 Data Used to Identify Chemical of Potential Concern

The data used to identify CPCs represent analytical soil data from the Phase I VSI and previous investigations in 1995 and 1993. As discussed in Section 3 of Volume I, these

data were judged to represent definitive data. Data that do not represent definitive data and therefore not included in the HHRA were qualitatively discussed in Volume I. However, excluding these data did not result in a misrepresentation of Site conditions for the purposes of CPC selection and subsequent QG development.

4.4.1.2 Data Used to Estimate Mercury Speciation and Bioavailability

Mercury Speciation Data. Mercury speciation analyses were conducted with 14 soil samples collected from the Site. These data were used to provide information concerning the chemical speciation of mercury in soils at the site, and to provide independent qualitative validation of the bioavailability testing results. As discussed in Subsection 4.1, the speciation of mercury has a bearing on the toxicity and the bioavailability of mercury. Organic mercury species are generally regarded as more toxic and more bioavailable than inorganic mercury species, whereas elemental mercury is generally associated with low toxicity and minimal-bioavailability via the oral exposure route. Therefore, it is important to determine the mercury speciation in a given exposure medium in order to provide a more accurate characterization of the potential risks associated with exposure to that medium. Mercury speciation in Site soils was determined by two methods: 1) sequential extraction, and 2) electron microprobe analysis (EMPA).

The performance of the sequential extraction technique was evaluated by analyzing site soil samples spiked with known fractions of the various mercury species evaluated. As discussed in Appendix A, extraction of the spiked Site soils was complicated by the presence of high carbonate content in several Site soil samples. A comparison between the carbonate content of Site soils and the mercury concentrations in the acid-soluble extract

(Appendix A, Figure 2) showed that acid-soluble mercury was only detected at substantial concentrations in soils with carbonate concentrations below 5%. It was hypothesized by PTI that the elevated carbonate content (between 0.8 and 13.9 weight percent) of Site soils had a buffering effect on the acid-soluble mercury extraction step of the sequential extraction analysis, thereby inhibiting the amount of acid-soluble mercury that could be released in the extraction. PTI validated this hypothesis by performing an extraction of a spiked reference soil which contained 0.005% carbonate. Mercury recoveries (including acid-soluble mercury) for this sample were within acceptable ranges.

The effects of the high carbonate contents of Site soils on the sequential extraction results are to potentially under-recover acid-soluble mercury species and to over-recover elemental mercury (because acid-soluble mercury may remain in the sample after the acid-extraction is performed and thereby be "counted" as elemental mercury). For these reasons, the mercury speciation results that were interpreted and discussed in the PTI report (and this HHRA) were based on the sequential extraction results for Site soils with less than 4% carbonate content. It is important to note that the effects of high soil carbonate encountered with the speciation analyses have no bearing on the results of the bioavailability testing. The extractions performed in the bioavailability testing were monitored for buffering effects; the proper extraction pH was maintained for the duration of the test, thereby ensuring that all acid-soluble mercury in Site soils was extracted in the *in vitro* test.

The performance of the EMPA was evaluated by analysis of two samples spiked with equal portions of mercuric chloride, mercuric sulfide, and elemental mercury. Results for both of the spiked samples indicated that elemental mercury and mercuric chloride may

have been lost during analysis, resulting in under-estimation of these species and over-estimation of mercuric sulfide. PTI attributed the loss of mercuric chloride and elemental mercury to a volatilizing effect during the EMPA, a theory which is plausible given the three- to five-order of magnitude differences in vapor pressure applied during the EMPA (10^{-8} Torr) and the vapor pressure of elemental mercury (10^{-3} Torr) and mercuric chloride (10^{-5} Torr). As with the sequential extraction results, these speciation data have no bearing on the bioavailability test performance or test results. EMPA data are only used as a complimentary method of determining mercury speciation in Site soils.

The sequential extraction results for the eight Site soils with 4% or lower carbonate content indicated that the majority of mercury was found in the elemental fraction (63% average), with almost none in the organic fraction (0.3% average). The balance of mercury was found in the acid soluble fraction (21% average) and the mercuric sulfide fraction (15% average). Results of the EMPA analyses, showing distribution of the mercury mass among the various mercury-bearing particles, indicated that elemental mercury was the primary mercury species (61% average), with mercuric sulfide (24% average) and mercuric chlorides and sulfates (15% average) composing the remainder of the mercury mass. These findings were consistent with the data for mercury vapor headspace analyses, which indicated a good correlation ($r^2 = 0.72$) between mercury headspace and elemental mercury, the most volatile form of mercury in Site soils.

A confounding factor in the mercury analyses of Site soils was associated with sample heterogeneity. Variability in relative percent differences (RPDs) and some inconsistencies between duplicates were found during comparisons of the total mercury released during sequential extraction of the Site soil samples to the total mercury determined in the <2 mm

soil particle fractions. Comparisons of total mercury concentrations measured in soils sieved to <2 mm to soils sieved to <250 mm also identified some variability in RPDs. Problems with analytical techniques were ruled out because triplicate analyses of single samples/extracts were very consistent, and laboratory QA/QC results were within acceptable limits. In addition, mercury was not found to be associated with any particular grain size or mineral fraction; mercury was not unequally distributed between the <2 mm soil fraction or the <250 mm soil fraction. As discussed in Appendix A, soil heterogeneity presents a problem with mercury analyses because elemental mercury often flows within a soil, forming discrete beads. As discussed in Appendix A, visible beads of mercury were observed in two Site soil samples. Although determined to not be associated with any particular grains size of mineral fraction, the presence of even one bead of elemental mercury in the relatively small aliquots of sample (i.e., 7.5 grams) used for analytical techniques can cause variable results when conducting duplicate analyses of the same soil sample. Limitations associated with sample heterogeneity were also identified in field and laboratory analyses of Site soil samples (Subsection 3.3.4 of Volume I).

Bioavailability Test Data. Mercury bioavailability testing was conducted on 10 soil samples collected from the Site. Results of the bioavailability testing were used to determine the potential bioavailability of mercury in Site soils. As discussed in Subsection 4.2.3.3, a bioavailability adjustment factor was derived from the bioavailability testing data. This bioavailability adjustment factor was used to adjust the oral intake exposure estimates for site-specific exposure conditions.

The bioavailability test performed by PTI is an in vitro assay which seeks to model soil digestion by a human child. The test is based on the principle that mercury species which are soluble are also potentially bioavailable - a mechanism which is supported by the scientific literature. The amount of mercury that is "digested" into solution in the in vitro assay is the amount of mercury which is soluble, and assumed potentially bioavailable. As discussed in Appendix A, the in vitro test methodology has been verified using *in vivo* studies with laboratory animals (e.g., monkeys), and has been used by PTI to estimate bioavailability of soils contaminated with arsenic, lead, and mercury. The results of the mercury bioavailability testing for Site soils were consistent with the mercury bioavailability results obtained by PTI at other sites.

Bioavailability Test Performance. Performance of the bioavailability test was evaluated by performing triplicate bioavailability tests on one sample (BS-69 4-6 ft), performing a mass-balance evaluation for this same sample, and testing a spike sample (plus three replicates) consisting of a mercuric chloride solution.

Results of the bioavailability test triplicates are presented in Appendix A, Table 18 and Figure 7. Reproducibility among the triplicate analyses was very good; bioavailability results did not differ by more than 1% among the three samples. This indicates that precision was good, and that the assay could reliably extract potentially bioavailable mercury.

The mass-balance evaluation, summarized in Appendix A, Table 16, was conducted to determine the amount and potential sources of mercury loss that may occur during the *in vitro* assay. Results of the mass-balance evaluation indicated that percent recovery was

32% (range 22% to 39%) among the triplicate analyses. These data suggested that a mercury loss occurred in the test system. An evaluation by PTI of the potential sources of mercury loss concluded that the loss occurred during analysis of the post-extraction soil sample. Because the amount of the post-extraction soil sample was very small (i.e., approximately 1 gram), there was insufficient sample to perform analytical mercury analyses and percent solids determination with separate aliquots of sample. Therefore, the sample was air-dried over a period of several days to determine percent solids, and then analyzed for total mercury concentration. It was concluded by PTI that the air-drying step vaporized volatile forms of mercury (e.g., elemental mercury) that remained in the sample following the *in vitro* extraction. In another *in vitro* test performed by PTI, where the extracted sample was not treated in this manner, mercury mass-balance recoveries were good (e.g., 121% recovery among triplicate analyses). Therefore, the loss of mercury observed in the mass-balance analysis was attributed to the air-drying procedure.

The source of mercury loss identified in the mass balance evaluation did not affect bioavailability testing results with Site soils. Unlike the mass-balance evaluations, bioavailability in Site soils was evaluated by comparing the mass of mercury in the sample extract (i.e., the potentially bioavailable fraction) with the mass of mercury in the pre-extracted sample. Therefore, the process which the mercury loss was attributed to in the mass-balance evaluations (i.e., analysis of post-extraction soil) did not occur during analysis of the Site samples. In other words, the mass balance determination was used *only* to evaluate the performance of the *in vitro* test system, and the primary limitation associated with the mass-balance evaluation was associated with a step that is not performed in the *in vitro* tests for Site soils. As a result, the uncertainties associated with the mass balance are limited in scope to interpretation of the mass balance results. Since

results for the bioavailability test triplicate analyses of sample BS-69 (4-6 ft) replicated very well, they provide independent confirmation of the successful performance of the bioavailability test.

Analysis of the mercuric chloride solution (and the three replicates) indicated that an average of 78% (35% to 102%) of the mercuric chloride was recovered (Appendix A, Table 19). As described below, this bioavailability estimate was used in the BAF calculation to represent the bioavailability of mercuric chloride in water, which is the mercury species and dosing medium upon which the mercuric chloride oral dose-response value is based on. A mercuric chloride bioavailability of less than 100% results in calculation of a higher BAF (see below) and, therefore, results in a more conservative assessment.

Mercury Bioavailability Test Results. The bioavailability test results were used to develop a bioavailability adjustment factor. The bioavailability adjustment factor was based on data which are likely to overestimate the potential bioavailability of mercury in Site soil to humans.

The bioavailability test results showed a trend in increased bioavailability with increased test duration, as the largest percent bioavailability results were associated with the extracts collected following 5 hours of incubation (collection time of final extracts) (Appendix A, Table 18 and Figure 6). The bioavailability adjustment factor was based on these data. However, the soluble mercury measurements obtained for the 5 hour incubation conservatively represent the potential bioavailability in humans. As discussed in Appendix A, the intestinal transit time in humans is generally no more than 4 hours, with

average times in children of 3.5 hours. Therefore, since the modeled intestinal transit time in the in vitro assay was 4 hours, and mercury bioavailability was greatest at the 4 hour measurement, using the bioavailability data obtained for the 5 hour incubation (1 hour stomach phase, plus 4 hours intestinal phase) provides an upper-bound estimate of the intestinal transit time in humans. In addition, there is evidence that intestinal absorption of inorganic mercury occurs in the first two-fifths of the small intestine (representing one-half the small intestine transit time), providing further evidence that the bioavailability measurement obtained for the 4 hour intestinal incubation provides a conservative estimate of bioavailability in humans (PTI, 1996).

The redox conditions present in the in vitro tests of Site soils may also have contributed to conservative estimates of bioavailability. As discussed in Appendix A, the redox conditions measured in the in vitro tests were considerably more oxidizing than redox conditions in a fasted human stomach. Mercury species such as elemental mercury and mercuric sulfide tend to be more stable and more insoluble (i.e., less bioavailable) under reducing conditions. As shown in Appendix A, Figure 9, the redox conditions of a fasted human stomach would render elemental mercury more stable and less soluble than those present in the in vitro test performed with Site soils. Therefore, it is likely that more elemental mercury and mercuric sulfide were solubilized in the in vitro assay than would be in a human stomach.

In addition, the bioavailability testing was conducted using soil sieved to <250 um. Use of this soil size range provides a conservative estimate of bioavailability because soil grains in this size range are the most likely sizes to adhere to skin (e.g., hands) and be ingested (Duggan and Inskip, 1985), and bioavailability appears to increase with decreasing particle

size. Experiments with humans (Chaney et al., 1989) and laboratory animals (Barltrop and Meek, 1979) have demonstrated that bioavailability increases as particle size decreases. As much as a five-fold increase in gastrointestinal absorption has been observed as particle size is decreased from 197 um to 6 um. Therefore, the bioavailability results for the <250 um soil fraction provide a conservative estimate of the potential bioavailability of larger grained soils.

The mercury bioavailability factor was developed by dividing the average bioavailability of mercury in Site soils (16%) by the average bioavailability of mercuric chloride measured in the mercuric chloride spike samples (78%). The value used to represent the mercury bioavailability in Site soils (i.e., 16%) was selected using a conservative approach. Using the linear correlation presented in Figure 8 (Appendix A), PTI selected the soil bioavailability that corresponded to the total mercury concentration that would not pose an unacceptable risk to commercial/industrial workers (i.e., mercury concentration corresponding to HI = 1) if a mercury bioavailability of 100% was conservatively assumed. This mercury concentration (520 mg/kg) corresponded to a soil bioavailability of 16% (Appendix A, Figure 8). As described in Appendix A, mercury bioavailability decreased with increasing mercury concentration ($r^2 = 0.57$; Figure 8, Appendix A). It was concluded by PTI that this occurred because the fraction of elemental mercury, which is relatively non-bioavailable, increased with increasing total mercury concentration. Therefore, the bioavailability for soils with total mercury in the 0-520 mg/kg range is considerably higher than the bioavailability of soils with higher total mercury concentrations because the soils with higher total mercury are associated with a higher fraction of non-bioavailable elemental mercury. Therefore, the bioavailability factor

developed for the Site overestimates the bioavailability of soils with total mercury concentrations above 520 mg/kg.

Summary. In summary, the bioavailability test provides a conservative estimate of the oral bioavailability of mercury in Site soils. The BAF test method and results are independent of the mercury speciation results. The speciation results are used in a qualitative sense to validate the BAF. Although there are some limitations associated with the mercury speciation analytical methods, the majority of the uncertainty is associated with determining the ratio of elemental to inorganic forms of mercury, not the ratio of non-organic to organic forms. The mercury speciation results demonstrate that there is very little (i.e., less than 1%) organic mercury in Site soil, and that the majority of non-organic mercury in Site soil is in the elemental form. The weight-of-evidence from sequential extraction analyses, EMPA, headspace analyses, and visual observations together with the bioavailability test results indicate the majority of mercury in Site soils is in non-organic forms with limited bioavailability, thereby supporting a mercury bioavailability adjustment factor of less than 1. It is appropriate to apply this BAF to all soils at the Site because the distribution of mercury among various minerals and grain sizes does not appear to be different, and the bioavailability of the <250 um fraction is expected to be greater than the bioavailability of larger soil particles.

The approach used to develop the BAF is consistent with bioavailability assessment approaches and adjustment factors used at other Sites for which Records of Decision (RODs) have been signed, including the Alameda Quicksilver County Park (CDM, 1992) and the Lower East Fork Poplar Creek (USEPA, 1995). For example, the mercury BAF derived for the Alameda Quicksilver County Park site was 0.3, and the BAF derived for

the Lower East Fork Poplar Creek site was 0.1. Based upon these site-specific BAFs, the resulting action levels at these two sites were increased more than three-fold over the action levels that would have been calculated assuming a mercury bioavailability of 100%. In addition, had the action levels at these two sites been based on the ASTM (1995) default exposure parameters for the commercial/industrial worker that were used in this HHRA, the action levels would have been consistent with the direct contact soil QG for mercury proposed for the Ames Street Site.

4.4.2 Uncertainties Associated with Toxicity Assessment

For this HHRA, uncertainties associated with the toxicity assessment can be grouped into two general areas: 1) uncertainties associated with the methods used to develop dose-response values, and 2) uncertainties associated with dose-response values for CPCs evaluated in this HHRA.

4.4.2.1 General Dose-Response Value Uncertainties

Toxicity information for many chemicals is very limited, leading to varying degrees of uncertainty associated with calculated toxicity values obtained from USEPA's IRIS and HEAST data bases. General sources of uncertainty for calculating toxicity factors include extrapolation from animal to human populations, low to high dose extrapolation, short-term to long-term exposures, interspecies sensitivity variation, extrapolation from subchronic to chronic no observed adverse effect level (NOAEL), extrapolation from lowest observed adverse effect level (LOAEL) to NOAEL, amount of data supporting the

toxicity factors (i.e., inadequate studies), consistency of different studies for the same chemical, and responses of various species to equivalent doses.

The identification of human carcinogens and non-carcinogens, based on animal data, is a primary source of uncertainty in the use of toxicity values. It is not certain that the identification of carcinogenic activity in an animal species means that carcinogenic activity in humans will occur. In some cases, the metabolic processes involved in carcinogenic activity in a particular organ in animals may not exist in humans. Available evidence indicates that there are a limited number of substances that are classified as human carcinogens (USEPA Class A substances).

The use of toxicity measures (e.g., RfDs and CSFs) introduces additional uncertainties. These parameters are generally based on animal studies, many of which are performed at high doses relative to the site-specific exposures that potentially could occur. These data require interpretation and/or extrapolation in the low dose area of the dose-response curve. The CSFs used in the risk assessment generally represent a "high end" estimate. The CSFs are the 95 percent UCL on the actual slope derived from the scientific data and, therefore, are likely overestimates of the potency.

4.4.2.2 Site-Specific Dose-Response Value Uncertainties

Toxicity data for inhalation exposures are limited, particularly for non-cancer effects. Non-cancer inhalation dose-response values for trichloroethene, benzene, tetrachloroethene, cadmium, chromium, nickel, dichloroethene, zinc, and cyanide are not available. As a result, non-cancer risks associated with potential inhalation exposures to these CPCs cannot be quantitatively evaluated and total risks were, therefore, underestimated. However, inhalation exposures typically contribute substantially lower CPC intakes than do ingestion exposures (i.e., approximately 1%). With this in mind, it is unlikely that quantitative evaluation of inhalation non-cancer risks would substantially increase the total risk estimate (i.e., combined ingestion, dermal, and inhalation) for a given receptor. For evaluation of inhalation exposures to VOCs that have migrated from soil or groundwater to air, lack of inhalation dose-response values presents a significant data gap since only inhalation exposures are evaluated for these pathways.

In this HHRA, trimethylbenzenes, n-butylbenzene, and 1,2-dichloroethene are CPCs in groundwater for which no inhalation dose-response values (cancer and non-cancer) are available. Therefore RBCs for inhalation exposures could not be quantified for these CPCs. As discussed above, this represents a data gap since the inhalation pathway is the only exposure pathway evaluated for these CPCs. However, trimethylbenzenes are close structural analogs to xylenes, and n-butylbenzene is a close structural analog to ethylbenzene. Therefore, although not quantified in this HHRA, the potential adverse effects, dose-response characteristics, and RBCs for trimethylbenzenes and n-butylbenzene are likely to be similar to xylenes and ethylbenzene, respectively.

No dose-response data are available in IRIS or HEAST for lead. Therefore, QGs could not be developed for lead. In the absence of site-specific QGs, the OSWER lead screening value for residential soils (USEPA, 1994) was used as the screening RBC. However, this value has been developed as a screening level for lead soil exposures to children in residential settings, and is therefore based on exposures that will not occur at the Site. As a result, the OSWER screening value is not applicable as a Quality Goal for soils at the Site. In the absence of a published screening value for commercial/industrial sites, a site-specific screening level could be developed using site-specific lead biokinetic uptake modeling.

As discussed in Subsection 4.1, uncertainty factors are applied during development of non-cancer dose-response values (i.e., RfDs) to account for uncertainties associated with the toxicity study upon which the RfD is based, and extrapolation of data. Therefore, uncertainty factors provide a downward adjustment of the measured dose which corresponds to the effect endpoint evaluated in the toxicity study. Uncertainty factors for the majority of the CPCs evaluated in this HHRA range between 100 and 3000 for oral RfDs, and 300 and 1000 for inhalation RfDs. Only cadmium, mercury, and zinc have uncertainty factors below 100. These elevated uncertainty factors indicate that the chemical dose which produced the effect endpoint in the toxicity study is greater (e.g., 100 to 3000 times greater) than the published reference dose used in the RBC calculations, resulting in a very conservative evaluation of potential risk. For chemicals with RfDs that incorporate large uncertainty factors, there is considerable uncertainty associated with interpreting whether low-level risks (e.g., HIs equal to 1 to 5) pose a realistic threat of adverse effects.

4.4.3 Uncertainties Associated with Exposure Assessment

Uncertainties associated with exposure assessment can be categorized into two general areas: 1) uncertainties associated with exposure scenario assumptions, and 2) uncertainties associated with exposure modeling.

4.4.3.1 Uncertainties Associated with Exposure Scenarios

The exposure scenarios selected for evaluation in this HHRA were based on the intended future use of the Site, and were quantified using default exposure parameters published by ASTM (1995) which are likely to overestimate exposures. The probable future use of the Site is a commercial/industrial facility. Under this land use, buildings will likely be constructed at the Site, and commercial/industrial workers will use the Site daily. The contact rates for the commercial/industrial worker published by ASTM (1995) assume that a commercial/industrial worker will incur direct-contact exposures to soil every workday for 25 years. These assumptions are likely to overestimate the potential exposures to commercial/industrial workers. Based on discussions with consultants, developers, and City of Rochester economic development officials, under all reasonably foreseeable industrial or commercial/industrial uses the Site will remain essentially entirely paved with some parts covered by buildings. In addition, areas that are not paved are expected to continue to be landscaped, thereby reducing the potential for any direct-contact exposures to commercial/industrial workers. The QGs developed for the commercial/industrial outdoor worker are applicable only to accessible soils, that is soils 0 to 15.24 cm (NYSDEC, 1995), since this receptor is not assumed to be exposed to soils below that depth. Likewise, if there are no accessible soils (i.e., the site is paved), then QGs for this

receptor are not required, as pavement would prevent complete exposure pathways. Under these exposure conditions, there would be no direct-contact exposure pathway for the outdoor commercial/industrial worker, and RBCs for direct-contact could be based on the construction worker.

In the absence of health and safety protection equipment, construction workers will likely be exposed to soil via direct-contact exposures if buildings are constructed or excavation occurs at the site. Likewise, utility workers are likely to be exposed infrequently to site media during sub-surface utility repair or maintenance work in either a future commercial/industrial Site use or under a no further development (i.e., current conditions) Site use. The construction worker and utility worker exposure scenarios were evaluated to help determine whether health and safety protection equipment will be necessary during excavation and construction at the Site.

The commercial/industrial worker and construction worker exposure scenarios are not protective for residential exposures, particularly for children. However, under the expected future commercial/industrial use of the site, residential exposures will not occur; an deed restriction will prevent residential land use of the Site. Because the majority of the Site will likely remain paved (in areas where buildings are not constructed), direct-contact exposures to soil will not occur to any children that may visit the Site. In addition, the exposure scenarios selected for evaluation in this HHRA do not include current or future site visitors and trespassers. However, the commercial/industrial worker exposure scenario provides a conservative assessment of potential trespasser and visitor exposures because the frequency and duration of the commercial/industrial worker exposure (i.e., 250 days per year for 25 years) exceeds what would be expected for any trespassing or

site visit activities. In addition, because the Site is currently paved, no direct-contact exposures occur to receptors other than a utility worker. For areas of the site that remain largely paved (as most are expected to), direct-contact exposures will continue to be prevented.

4.4.3.2 Uncertainties Associated with Exposure Modeling

General Modeling Uncertainties. Soil and groundwater RBCs for inhalation exposures to vapors that may migrate from soil and groundwater to air were estimated by developing a risk-based air concentration for each CPC and comparing it to the CPC air concentration resulting from vapor migration. Uncertainties associated with risk-based air concentrations are associated with toxicity and exposure scenario assumptions (Subsections 4.4.2 and 4.4.3, respectively). Uncertainties associated with vapor migration modeling are discussed here.

The vapor migration models used in this HHRA are those presented in *Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites* (ASTM E1739.95, 1995). These models are based on conservative assumptions about chemical fate and transport in environmental media. In addition, the models incorporate many parameters that describe physical site conditions (e.g., soil moisture content, soil bulk density, depth to contamination, vadose zone thickness), chemical properties (e.g., Henry's law constant, diffusion coefficients, soil:water partition coefficients), and exposure conditions (e.g., building air exchange rate, above-surface wind speed). Each of these parameters are usually associated with ranges of possible values. The default values published by ASTM

(1995) generally incorporate the parameter values that will result in the most conservative evaluation.

For this HHRA, NYSDEC (1995) values and site-specific values were used in the fate and transport models when they differed from the ASTM values. Specifically, the NYSDEC (1995) value for the lower depth of surficial soils (15.24 cm) was used instead of the ASTM (1995) value of 100 cm. In addition, appropriate site-specific data were used when available. Site-specific data used in the fate and transport models included soil moisture content, depth to soil contamination, depth to groundwater, and thickness of capillary fringe and vadose zone. For each of these parameters, average site values were used in the models. Use of site-specific physical data reduces uncertainty associated with fate and transport modeling. The use of ASTM (1995) default values for parameters which had no site-specific data represents an uncertainty, as it is unknown how accurately the default values represent conditions at the Site.

Chemical-specific data were obtained from USEPA sources (USEPA, 1986; 1988; 1993) or were developed using estimation methods provided in Lyman, et al (1990). There is some degree of uncertainty associated with these chemical-physical data. Many of the parameters such as Henry's law constant, soil:water partition coefficient, and air and water dispersion coefficients are estimated or derived; published values differ among various sources. The models are particularly sensitive to Henry's law constant and soil:water partition coefficients. Therefore, efforts were made to use the values most consistently reported in the literature (adjusted for subsurface temperature of 10 C), and to use consistent approaches for estimating values when necessary. Nonetheless, all chemical-

specific values should be regarded as estimates. These values are most useful for evaluating how constituents may act relative to each other and to the environment.

The crack factor and building air exchange rate variables produce the greatest amount of variability in fate and transport estimates. The building air exchange rate represents the number of air volumes within the building that are changed within a given time period. The default value published by ASTM (1995) is 0.8 building air exchanges per hour. For comparison, USEPA reports typical air exchange rates in residences to range from 0.5 to 1.5, with lower values reported for energy-efficient structures (Air/Superfund National Technical Guidance Study Series. Assessing Potential Impacts for Superfund Sites; 1992). However, the air exchange rates for a commercial/industrial building that is fitted with high volume ventilation systems, cargo access doors, and doors which are frequently opened for access to the building, would likely be higher than the values reported for residences and the default air exchange rate used in the fate and transport models. The New York State building codes reference ASHRAE standards for commercial/industrial building ventilation. However, these standards are based on building occupancy, which is a future site-specific variable that cannot be accounted for in this HHRA. Therefore, the ASTM (1995) default air exchange rate of 0.8 changes per hour was used in the fate and transport models, although this value is likely to be low compared to the building ventilation rate that will likely be engineered in future commercial/industrial buildings at the Site.

The crack factor is used to estimate the floor area through which volatile constituents could migrate from soil to indoor air. Specifically, the crack factor represents the percent area of impermeable floor (e.g., a cement slab) which is "cracked" and thus able to allow

vapors to permeate the floor. A crack factor of 100%, for example indicates that the entire floor area is permeable to vapor migration (e.g., a dirt floor or plank-over-dirt construction), whereas a crack factor of 0% indicates that the entire floor is impermeable to vapor migration (e.g., in the case of an uncracked cement slab with no spaces between the interior walls and the slab). The ASTM (1995) default crack factor is 1%, for which the technical basis is not defined. For comparison, USEPA reports crack areas for buildings with slab floor construction to range from 0.01% to 0.1% (Air/Superfund National Technical Guidance Study Series. Assessing Potential Impacts for Superfund Sites; 1992). The crack factor selected for the fate and transport models used in this HHRA is 0.1%, based on the volume change of normal weight cement, which ranges between 0.01% and 0.08% (Portland Cement Association (1979), American Concrete Institute (1980)). This represents a conservative estimate of cracking, however, because concrete slabs are typically installed with control joints to prevent uncontrolled cracking. The cracking that would result from control joints installed in an 6 inch thick floor slab would be approximately 0.03% (American Concrete Institute (YR)), which is considerably lower than the upper value for concrete volume change (0.08%). If it was assumed that the maximum floor slab crack area was represented by engineered control joints (0.03%) in addition to the maximum value for concrete volume change of 0.08%, an upper estimate of the crack factor would be 0.1%.

In summary, the fate and transport models used in this HHRA provide order-of-magnitude estimates of potential vapor migration from soil and groundwater sources to air. Modeled indoor air concentrations, in particular, are subject to interpretation because of the many variables which are included in the models, many of which are conservative non-site-specific values intended to provide upper estimates of vapor migration.

Uncertainties Associated With Modeling Mercury Exposures. Mercury exposures to air were modeled using the same techniques that were used for volatile organic compounds (VOCs). Chemical-specific variables, including Henry's law constant, soil:water partition coefficient, and air diffusion coefficient were obtained from a publication issued by the Gas Research Institute (GRI, 1994). However, modeling mercury vapor migration with the fate and transport models used in this HHRA presents an uncertainty because mercury does not behave like a VOC.

Therefore, in addition to the uncertainties associated with these fate and transport models, there is uncertainty associated with application of these models to mercury. However, based on historical mercury air monitoring data, the vapor migration estimates and corresponding soil and groundwater mercury RBCs for the vapor migration pathway appear to be conservative. Indoor air monitoring data for mercury were collected at the Ames Street site facilities between 1986 and 1990 in areas where elemental mercury was used in manufacturing. The environmental conditions in many of the areas monitored included visible elemental mercury droplets on the floors, in cracks between floor boards, in the ceilings, and in ventilation ducting. With the exceptions of monitoring data collected just after elemental mercury spills, air concentrations at these locations ranged from lower than the instrument detection limit of 0.005 mg/m^3 to just under the OSHA limit of 0.05 mg/m^3. Air monitoring data generally showed highest concentrations at the floor level (where mercury droplets were often present between and underneath floor boards), with decreasing concentrations at waist and head levels.

The environmental conditions under which this air monitoring took place are not representative of the conditions which would be present in new buildings constructed at the Site. The only source for indoor mercury vapor under future site conditions would be mercury vapor in the soil beneath the buildings. As discussed previously, the mercury soil vapor would have to migrate through cracks in the building floor slab in order to reach indoor air. Clearly then, given the location of mercury contamination in Site soils, and the pathways which would need to be in place for mercury vapor move from soil to indoor air, indoor air mercury vapor concentrations would be significantly lower than the air concentrations measured in buildings with free mercury in the floors and ceilings.

5.0 QUALITY GOALS

This section presents the soil and groundwater Quality Goals (QGs) for the Ames Street Site. The QGs developed in this section of the HHRA are the proposed remedial goals for the Site. Subsection 5.1 describes the technical approach used to select soil and groundwater QGs. Soil and groundwater QGs for various land development scenarios are presented in Subsections 5.2 through 5.4. These sections provide the rationale for the QG selection.

5.1 TECHNICAL APPROACH FOR DEVELOPING SOIL AND GROUNDWATER QUALITY GOALS

Quality Goals for soil and groundwater were developed by reviewing the screening risk-based concentrations (RBCs) for each exposure scenario presented in Subsection 4.3, and applying risk-management principles to identify a single receptor-specific screening RBC for each Chemical of Potential Concern (CPC) for each land development scenario. Quality Goals were then developed for each land development scenario for CPCs present in soil and groundwater at concentrations above the screening RBCs.

The approach used for developing QGs involved several steps, including:

- selection of exposure scenario upon which QGs may be based,
- identification of screening RBCs,
- identification of CPCs that exceed the screening RBCs, and
- selection of final Quality Goals.

5.1.1 Selection of Exposure Scenarios

In this HHRA, screening RBCs for soil and groundwater were developed for a future commercial/industrial worker exposure scenario, a future construction worker exposure scenario, and a utility worker exposure scenario. Each receptor exposure scenario corresponds to the types of exposures that could occur under the various current and future Site land use conditions. RBCs for each of these exposure scenarios were developed for multiple exposure pathways in order to evaluate soil and groundwater contamination with respect to the types of exposures that may occur for each land use condition.

According to the ASTM RBCA standard (ASTM, 1995), the lowest screening RBCs among all exposure pathways and receptors evaluated should be selected as the basis of the remedial goals. However, this approach is valid only if the land use conditions and exposure pathways evaluated remain in place, unmodified from the conditions for which the exposures were quantified. If, for example, the land use conditions or exposure conditions were modified from the conditions upon which the lowest RBCs were based, those RBCs would no longer be applicable to the exposure conditions at the Site.

Therefore, to provide soil QGs that provide flexibility for remedial and risk management decision-making, soil QGs were developed for three future land development scenarios:

- Commercial/Industrial Development without Engineering Controls
- Commercial/Industrial Development with Engineering Controls
- No Further Development

Each land development scenario is associated with specific receptors and exposure pathways. The soil and groundwater QGs for each of these land development scenarios is based on the lowest, or most sensitive, receptor- and exposure pathway-specific RBCs that are applicable for that scenario. Arising from these three scenarios is a fourth "focused development" scenario. Under this land use scenario, different QGs for the same CPC would be applied to different portions of the Site, depending on the specific development scenario for that portion of the Site (i.e., with or without engineering controls, no further development). Table 14 provides a summary of the receptors and exposure pathways that are appropriate for consideration as the basis of QGs for each land use. These are discussed in the following subsections.

5.1.1.1 Commercial/Industrial Development with No Engineering Controls

As discussed throughout this HHRA, the anticipated future land use of the Site is commercial/industrial. Combustion Engineering, the site owner, has committed to placing deed restrictions on the Site which prohibit the construction of occupiable sub-grade structures (i.e., basements will not be constructed), any use of on-site groundwater, and

development of the site for any uses other than commercial/industrial, including residential, school, day-care center, etc.

Under this land development scenario, future commercial/industrial employees who work indoors at the Site in new buildings constructed on floor slabs (i.e., without occupiable basements) could be exposed to volatile CPCs (VOCs and mercury) that may migrate from soil and groundwater, through a building floor slab, to indoor building air. These receptors would not be exposed to non-volatile CPCs (e.g., inorganics and cyanide) because a direct contact exposure pathway does not exist for the indoor commercial/industrial employee. However, commercial/industrial employees who work outdoors at the Site could be exposed to CPCs through direct contact with surface soil or through inhalation of volatiles that may migrate from groundwater to ambient (outdoor) air, although these exposure pathways would only be complete if the Site was not paved. If deed restrictions were in place to prevent direct contact exposures to contaminated soils, these exposure pathways would not be complete.

Under a future commercial/industrial development scenario, construction workers could be exposed during excavation activities and construction of new buildings to soils via direct contact and inhalation of volatile CPCs, and to groundwater via inhalation of volatile CPCs.

The receptor and exposure pathways that are appropriate for this land development scenario are summarized in Table 14. From the screening RBCs for each of these receptor scenarios, a single screening RBC will be selected as the basis of the Quality Goal for the CPC, as described in Subsection 5.1.2.

5.1.1.2 Commercial/Industrial Development with Engineering Controls

As discussed throughout this HHRA, the anticipated future land use of the Site is commercial/industrial. A comparison of the commercial/industrial worker screening RBCs (Table 8) shows that RBCs based on inhalation of vapors that migrate to indoor air are typically lower than RBCs based on direct contact exposures. This indicates that the inhalation exposure pathway, under the conditions modeled in this HHRA, is associated with greater exposures to volatile CPCs than the direct contact exposure pathway. However, if exposures from the indoor air inhalation exposure pathway were reduced sufficiently, this pathway would become insignificant when compared to potential exposures from the direct contact exposure pathway. One method for minimizing potential exposures to soil and groundwater CPCs is to place engineering controls that restrict volatile migration to building air in new buildings constructed at the Site and restrict direct-contact exposures.

Under this land development scenario, future commercial/industrial employees who work indoors at the Site in new buildings constructed on floor slabs with engineered vapor controls would not be exposed to volatile CPCs (VOCs and mercury) that may otherwise migrate from soil and groundwater, through a building floor slab, to indoor building air. Likewise, these receptors would not be exposed to non-volatile CPCs (e.g., inorganics and cyanide) either because a direct contact exposure pathway does not exist for the indoor commercial/industrial employee. Therefore, under a development scenario that employs engineered building vapor controls, no complete exposure pathways for the indoor commercial/industrial worker are present; no QGs are required for this receptor. Commercial/industrial workers who work outdoors at the Site would not be exposed to

soil via direct contact or to groundwater vapors via inhalation because the Site would be paved, thereby preventing these exposure pathways.

Under a future commercial/industrial development scenario, construction workers could be exposed during excavation activities and construction of new buildings to soils via direct contact and inhalation of volatile CPCs, and groundwater via inhalation of volatile CPCs.

The receptor and exposure pathways that are appropriate for this land development scenario are summarized in Table 14. From the screening RBCs for each of these receptor scenarios, a single screening RBC will be selected as the basis of the Quality Goal for each CPC, as described in Subsection 5.1.2.

5.1.1.3 No Further Development (Current Conditions)

As discussed throughout this HHRA, the anticipated future land use of the Site is commercial/industrial. However, it is unlikely that the entire 13 acre Site will be redeveloped and covered with buildings. Portions of the Site may remain paved, undisturbed from the current conditions. The potential receptor exposures to soil and groundwater at a portion of the Site that remains undeveloped would be different than those associated with portions of the Site that are developed. Because buildings would not be constructed in areas of the Site that remain undeveloped, construction workers would not be exposed to Site soil or groundwater, and indoor commercial/industrial workers would not occur. Likewise, because the Site is presently paved, areas that are not developed would remain paved, thereby preventing direct contact and volatile

inhalation exposures to outdoor commercial/industrial workers. In summary, under the no further development scenario, there are no exposure pathways to soil and groundwater for indoor or outdoor commercial/industrial workers or outdoor construction workers.

The only receptor that could potentially be exposed to Site soil or groundwater under the no further development scenario (which represents current Site conditions) is a utility worker performing repairs or maintenance to utilities beneath the Site. Under these conditions, utility workers could be exposed during excavation activities and "trench work" to soils via direct contact and inhalation of volatile CPCs, and groundwater via inhalation of volatile CPCs.

The receptor and exposure pathways that are appropriate for this land development scenario are summarized in Table 14. From the screening RBCs for the receptor scenario, a single screening RBC will be selected as the basis of the Quality Goal for each CPC, as described in Subsection 5.1.2.

5.1.2 Identification of Screening Risk-Based Concentrations

For each receptor evaluated for a given development scenario, screening RBCs for all appropriate exposure pathways and effect-endpoints were reviewed to identify a single receptor-specific screening RBC for each CPC. The selected screening RBCs for each CPC were used as the basis of the Quality Goals, as described in Subsection 5.1.4.

As discussed in Subsection 4.3, the screening RBCs for cancer effects were calculated for the conservative NYSDEC and USEPA lowest acceptable cancer risk of $1x10^{-6}$, and the RBCs based on non-cancer effects were calculated for the conservative NYSDEC and USEPA non-cancer risk threshold of HI = 1. According to NYSDEC guidance (NYSDEC, 1995), the lowest RBC for cancer and non-cancer endpoints should be selected as the remedial goal. For each of the exposure pathways evaluated for a given receptor, the lower of the cancer-based RBC and non-cancer-based RBC for these cancer and non-cancer risk levels was chosen as the screening RBC for each CPC. From these RBCs, the lowest screening RBC among all exposure pathways for a given receptor was selected as the screening RBC to be used as the basis of the QG.

5.1.3 Identification of Chemicals of Potential Concern that Exceed Screening Risk-Based Concentrations

Once screening RBCs for soil and groundwater were identified for each exposure scenario, a data comparison was done in order to determine which CPCs may be present in Site soil and groundwater at concentrations above the screening RBCs. Chemicals of potential concern that exceeded screening RBCs were identified by comparing the maximum reported CPC concentration to the screening RBC. If the maximum CPC concentration exceeded the screening RBC, the CPC was considered to be potentially present in Site soil or groundwater at concentrations at which a QG was necessary.

5.1.4 Selection of Quality Goals

For the CPCs that were reported at maximum concentrations greater than the selected receptor-specific screening RBC, the screening RBC was selected as the QG. For the remaining CPCs, no QGs were required.

5.2 SELECTION OF QUALITY GOALS

The selection of soil Quality Goals was performed using the approach described in Subsection 5.1. Quality Goals were developed for three land development scenarios: Commercial/Industrial Development with No Engineering Controls, Commercial/Industrial Development with Engineering Controls, and No Further Development. The selection of soil and groundwater QGs for each of these land development scenarios is discussed in the following subsections.

5.2.1 Commercial/Industrial Development with No Engineering Restrictions

Quality Goals for this land development scenario are based on the following conditions:

- The future property use is restricted to commercial/industrial use.
- Restrictions against subgrade structures and use of on-site groundwater are in place and are being complied with.
- Volatile CPCs (i.e., volatile organic compounds (VOCs) and mercury) migrate from subsurface soil and/or groundwater sources and establish equilibrium concentrations in soil pore spaces beneath the slab floor of a building overlying

the soil source area. Volatiles migrate through cracks in the slab floor and establish equilibrium concentrations in air inside the building. Commercial/industrial employees working indoors are exposed to the indoor air via the inhalation exposure route. There are no specific engineering controls installed to prevent or reduce migration of vapors to air inside the building, thereby rendering the vapor migration pathway "unrestricted". Indoor commercial/industrial workers are not exposed to soil or groundwater via direct contact or volatile migration to outdoor air.

- There are no specific controls (including paving) to prevent commercial/industrial workers from contacting soil via the incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor exposure routes, and to groundwater via the vapor inhalation exposure route.
- It is assumed that because portions of the Site will be developed, construction workers will be exposed to soil via the incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor exposure routes, and to groundwater via the vapor inhalation exposure route.
- The soil direct contact soil QGs for mercury incorporate a site-specific oral bioavailability adjustment factor of 0.2.

5.2.1.1 Soil Quality Goals - No Engineering Controls

Tables 15 (indoor commercial/industrial worker), 16 (outdoor commercial/industrial worker), and 17 (outdoor construction worker) present the soil CPCs and their corresponding maximum reported concentrations, the lowest cancer or non-cancer based RBC for each CPC (i.e., the screening RBC), the pathway and target risk for which the

screening RBCs were developed, and an indication of whether the maximum reported CPC concentrations exceeded the screening RBC.

As indicated in Table 15, the screening RBCs for the indoor commercial worker are based on indoor air inhalation of volatiles (i.e., volatile migration pathway for VOCs and mercury). Screening RBCs for direct-contact exposures or outdoor inhalation exposures are not applicable to this receptor exposure scenario because there are no complete direct-contact exposure pathways. Therefore, the indoor commercial worker is not exposed to non-volatile CPCs (i.e., CPCs which cannot volatilize and migrate through floor slab cracks to building air). As indicated in Table 15, the maximum reported concentrations of trichloroethene, tetrachloroethene, and mercury exceeded the screening RBCs based on indoor commercial/industrial worker exposures to indoor building air.

Table 16 presents the selection of QGs for the outdoor commercial/industrial worker. As summarized in Table 14, screening RBCs based on direct-contact exposure were applicable for this receptor exposure scenario. The maximum reported concentrations of trichloroethene, tetrachloroethene, lead, and mercury exceeded the screening RBCs for these CPCs (Table 16). For trichloroethene, tetrachloroethene, and mercury the screening RBC was selected as the Quality Goal. QGs for the outdoor commercial/industrial worker are only applicable to accessible soils; that is soils to a depth of 15.24 cm (NYSDEC, 1995) that are not paved. The lead screening RBC, which is the OSWER residential soil cleanup value, was not adopted as the QG because this value is inappropriate for application as a cleanup goal at the Site. Therefore, no QG was developed. for lead.

Table 17 presents the selection of QGs for the outdoor construction worker. As summarized in Table 14, screening RBCs based on direct-contact exposure and outdoor inhalation exposures to VOCs and mercury that have migrated from subsurface soil to ambient air were applicable for this receptor exposure scenario. With the exception of trichloroethene (for which the RBC was based on inhalation exposure), screening RBCs based on direct contact were identified as the lowest RBCs and, therefore, were selected as the basis of QGs (Table 17). The maximum reported concentrations of trichloroethene, lead, and mercury exceeded the screening RBCs for these CPCs (Table 17). For trichloroethene and mercury, the screening RBC was selected as the Quality Goal. These QGs are applicable in the absence of a worker health and safety plan that controls worker exposures to soil. The lead screening RBC, which is the OSWER residential soil cleanup value, was not adopted as the QG because this screening value is inappropriate for application as a cleanup goal at the Site. Therefore, no QG was developed. for lead.

5.2.1.2 Groundwater Quality Goals - No Engineering Controls

Tables 18 (indoor commercial/industrial worker), 19 (outdoor commercial/industrial worker), and 20 (outdoor construction worker) present the groundwater CPCs and their corresponding maximum reported concentrations, the lowest cancer or non-cancer based RBC for each CPC (i.e., the screening RBC), the pathway and target risk for which the screening RBCs were developed, and an indication of whether the maximum reported CPC concentrations exceeded the screening RBC.

As indicated in Table 18, the screening RBCs for the indoor commercial worker are based on indoor air inhalation of volatiles (i.e., volatile migration pathway for VOCs and mercury); this is the only exposure pathway to groundwater for this receptor. As indicated in Table 18, the maximum reported concentrations of trichloroethene and vinyl chloride exceeded the screening RBCs based on indoor commercial/industrial worker exposures to indoor building air. The screening RBCs for these CPCs were selected as the QGs.

Table 19 presents the selection of QGs for the outdoor commercial/industrial worker. The only exposure pathway to groundwater for this receptor is inhalation exposure to volatile CPCs that have migrated from groundwater to ambient air. As indicated in Table 19, no maximum reported groundwater concentrations exceeded screening RBCs and, therefore, no QGs were calculated.

Table 20 presents the selection of QGs for the outdoor construction worker. As summarized in Table 14, screening RBCs based on outdoor inhalation exposures to VOCs and mercury that have migrated from groundwater to ambient air were applicable for this receptor exposure scenario. As shown in Table 20, screening RBCs were not exceeded by the maximum detected concentrations of any groundwater CPC and, therefore, no QGs were developed.

5.2.2 Commercial/Industrial Development with Engineering Restrictions

Quality Goals for this land development scenario are based on the following conditions:

- The future property use is restricted to industrial or commercial use.
- Restrictions against subgrade structures and use of on-site groundwater are in place and are being complied with.
- Volatile CPCs (i.e., VOCs and mercury) migrate from subsurface soil sources and establish equilibrium concentrations in soil pore spaces beneath the slab floor of a building overlying the soil source area. There are specific engineering controls present to prevent or reduce migration of vapors to air inside the building, thereby rendering the vapor migration pathway "restricted". Volatile migration through the slab floor is restricted or prevented to the extent that vapor concentrations in air inside the building are lower than the indoor inhalation screening RBCs. Indoor commercial/industrial workers are not exposed to volatiles that could migrate from subsurface sources to building air, nor are they exposed to surface soils via direct contact or volatile inhalation.
- Outdoor commercial/industrial workers do not contact soil via the incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor exposure routes, nor groundwater via the vapor inhalation exposure route. There are specific deed restrictions in place requiring that the Site be paved, thereby preventing exposures to outdoor commercial/industrial workers.

- It is assumed that because portions of the Site will be developed, construction workers will be exposed to soil via the incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor exposure routes, and to groundwater via the vapor inhalation exposure route.
- The soil direct contact soil QGs for mercury incorporate a site-specific oral bioavailability adjustment factor of 0.2.

5.2.2.1 Soil Quality Goals - With Engineering Controls

Table 17 (outdoor construction worker) presents the soil CPCs and their corresponding maximum reported concentrations, the lowest cancer or non-cancer based RBC for each CPC (i.e., the screening RBC), the pathway and target risk for which the screening RBCs were developed, and an indication of whether the maximum reported CPC concentrations exceeded the screening RBC.

As discussed previously, there are no complete exposure pathways to soil for an indoor commercial worker if engineered building vapor controls are in place and, therefore, QGs were not developed for this receptor scenario. Likewise, there are no complete exposure pathways for an outdoor commercial worker if deed restrictions are in place that prevent exposure to accessible soil (i.e., by requiring that the majority of the Site remain paved or covered by slab-on-grade buildings). Therefore, no QGs were developed for this receptor scenario.

Table 17 presents the selection of QGs for the outdoor construction worker. As summarized in Table 14, screening RBCs based on direct-contact exposure and outdoor inhalation exposures to VOCs and mercury that have migrated from subsurface soil to ambient air were applicable for this receptor exposure scenario. With the exception of trichloroethene (for which the RBC was based on inhalation exposure), screening RBCs based on direct contact were identified as the lowest RBCs and, therefore, were selected as the basis of QGs (Table 17). The maximum reported concentrations of trichloroethene, lead, and mercury exceeded the screening RBCs for these CPCs (Table 17). For trichloroethene and mercury, the screening RBC was selected as the Quality Goal. These QGs are applicable in the absence of a worker health and safety plan that controls worker exposures to soil. The lead screening RBC, which is the OSWER residential soil cleanup value, was not adopted as the QG because this screening value is inappropriate for application as a cleanup goal at the Site. Therefore, no QG was developed for lead.

5.2.2.2 Groundwater Quality Goals - With Engineering Controls

Table 20 (outdoor construction worker) presents the groundwater CPCs and their corresponding maximum reported concentrations, the lowest cancer or non-cancer based RBC for each CPC (i.e., the screening RBC), the pathway and target risk for which the screening RBCs were developed, and an indication of whether the maximum reported CPC concentrations exceeded the screening RBC.

As discussed previously, there are no complete exposure pathways to groundwater for an indoor commercial worker if engineered building vapor controls are in place and, therefore, QGs were not developed for this receptor scenario. Likewise, there are no

complete exposure pathways for an outdoor commercial worker if deed restrictions are in place that prevent vapor migration from groundwater to ambient air (i.e., by requiring that the majority of the Site remain paved or covered by slab-on-grade buildings). Therefore, no QGs were developed for this receptor scenario.

Table 20 presents the selection of QGs for the outdoor construction worker. As summarized in Table 14, screening RBCs based on outdoor inhalation exposures to VOCs and mercury that have migrated from groundwater to ambient air were applicable for this receptor exposure scenario. As shown in Table 20, screening RBCs were not exceeded by the maximum detected concentrations of any groundwater CPC and, therefore, no QGs were developed.

5.2.3 No Further Development (Current Conditions)

Quality Goals for this land development scenario are based on the following conditions:

- The future property use is undeveloped. The property remains unchanged from current conditions; no construction activities will take place.
- Pavement remains in place to prohibit direct contact or volatile inhalation exposures to outdoor commercial/industrial workers.
- There are no specific controls to prevent utility workers from contacting soil via the incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor exposure routes, and to groundwater via the vapor inhalation exposure route.

- The soil direct contact soil QG for mercury incorporates a site-specific oral bioavailability adjustment factor of 0.2.
- The asphalt cover will be removed only for utility work to take place. Soil will be placed back into the excavation or moved to an appropriate off-site disposal site, and the asphalt will be replaced when the utility work is complete.

5.2.3.1 Soil Quality Goals - No Further Development

Table 21 (utility worker) presents the soil CPCs and their corresponding maximum reported concentrations, the lowest cancer or non-cancer based RBC for each CPC (i.e., the screening RBC), the pathway and target risk for which the screening RBCs were developed, and an indication of whether the maximum reported CPC concentrations exceeded the screening RBC.

As discussed previously, there are no complete exposure pathways to soil for all receptors except a utility worker and, therefore, QGs were developed only for this receptor scenario. Table 21 presents the selection of QGs for the outdoor utility worker. As summarized in Table 14, screening RBCs based on direct-contact exposure and outdoor inhalation exposures to VOCs and mercury that have migrated from subsurface soil to ambient air were applicable for this receptor exposure scenario. The maximum reported concentration of lead exceeded the screening RBC for this CPC (Table 21). The lead screening RBC, which is the OSWER residential soil cleanup value, was not adopted as the QG because this screening value is inappropriate for application as a cleanup goal at the Site. Therefore, no QG was developed for lead.

5.2.3.2 Groundwater Quality Goals - No Further Development

Table 22 (outdoor utility worker) presents the groundwater CPCs and their corresponding maximum reported concentrations, the lowest cancer or non-cancer based RBC for each CPC (i.e., the screening RBC), the pathway and target risk for which the screening RBCs were developed, and an indication of whether the maximum reported CPC concentrations exceeded the screening RBC.

As discussed previously, there are no complete exposure pathways to groundwater for all receptors except a utility worker and, therefore, QGs were developed only for this receptor scenario. Table 22 presents the selection of QGs for the outdoor utility worker. As summarized in Table 22, no screening RBCs were exceeded by the maximum detected concentrations of groundwater CPCs and, therefore, no QGs were developed.

6.0 PHASE I HHRA SUMMARY AND CONCLUSIONS

This section summarizes the approach and results of the HHRA for the Ames Street Site Phase I Voluntary Site Investigation. Conclusions of the HHRA and recommendations are also provided.

6.1 SUMMARY

The HHRA was performed to conservatively evaluate the human health risks that may exist at the Ames Street Site and to develop site-specific Quality Goals (QGs) for Chemicals of Potential Concern (CPCs) in soil and groundwater at the Site. The recommended QGs are human health risk-based concentrations (RBCs) for CPCs identified in Site soil and groundwater at levels above conservative screening values. Quality Goals represent concentrations which do not pose risks of concern for potential exposures to Site soil and groundwater under the exposure scenarios and land uses evaluated in this HHRA. The Quality Goals developed in this HHRA are compared to the SI soil and groundwater data findings in Section 5 of Volume I. The evaluations presented in that section form the basis for recommended remedial decisions for the Site.

The HHRA was performed in three steps:

- Identification of CPCs in soil and groundwater
- Development of screening RBCs for the soil and groundwater CPCs
- Development of Quality Goals based upon screening RBCs

Site-related soil CPCs were identified by comparing the maximum constituent concentrations reported in pertinent soil data from the 1996 SI and site investigations conducted in 1993 and 1995 to NYSDEC TAGM (1994) recommended soil cleanup values that were adjusted for site-specific conditions. Groundwater CPCs were identified by comparing the maximum constituent concentrations measured in perimeter well groundwater data collected during the 1996 SI to New York Class GA groundwater standards. For soil and groundwater, all volatile constituents which exceeded the comparison standards were selected as CPCs. In addition, all Site-related volatile constituents reported in interior well groundwater data from the Phase I SI were retained as CPCs because no appropriate standards were available to evaluate indirect exposures to this groundwater. The approach used to select CPCs was conservative for Site soil and groundwater because the standards used in the CPC selection were based on conservative land use conditions which do not and will not occur at the Site. For this reason, the site-specific TAGM values and Class GA groundwater standards were not considered as potential site-specific QGs of the Site. Instead, Quality Goals based on site-specific exposure conditions and land uses were developed.

For all soil and groundwater CPCs, screening risk-based concentrations were developed for exposure scenarios that may occur during future land use of the Site. The only exposures to soil and groundwater under current land use are associated with infrequent below-ground utility maintenance and repair work; there are no exposures to site visitors or trespassers. The future use of the Site will be commercial/industrial property. Deed restrictions will prevent residential use of the Site (including use of the site for schools, day-care centers, etc.), potable and non-potable use of groundwater at the Site, and construction of occupiable subsurface structures (e.g., basements). Given these land use

restrictions, the only receptors which would potentially be present at the Site would be commercial and/or industrial workers, construction workers, utility workers, trespassers, and site visitors. Because commercial/industrial workers and construction workers are assumed to be potentially exposed to Site media for much longer periods of time than trespassers or visitors (i.e., incur greater exposures to Site media), RBCs were not developed for trespassers or site visitors. Risk-based concentrations were developed for the following exposure pathways:

- Direct contact exposures to soil (commercial/industrial worker, construction worker, and utility worker)
- Inhalation exposures to volatiles that have migrated from subsurface soil to indoor building air (commercial/industrial worker) or ambient (outdoor) air (construction worker and utility worker)
- Inhalation exposures to volatiles that have migrated from groundwater to indoor building air (commercial/industrial worker) or ambient (outdoor) air (construction worker, commercial/industrial worker, and utility worker)

From the screening RBCs developed for the various exposure pathways and receptors, single screening RBCs for each CPC in each medium were selected as the basis of Quality Goals for each land development scenario evaluated. Chemicals of Potential Concern with maximum reported concentrations above screening RBCs were retained for development of Quality Goals.

Quality Goals for soil and groundwater were developed for three land development scenarios:

- Commercial/industrial development with no engineering controls
- Commercial/industrial development with engineering controls
- No further development (current conditions)

A fourth development scenario will apply QGs for two or more of the above scenarios to various areas of the Site to support a focused development scenario.

Soil and groundwater QGs for these land development scenarios are presented in Tables 23 and 24, respectively. Discussions pertaining to how the soil and groundwater QGs for each land development scenario relate to the interpreted areas of soil and groundwater source areas are presented in Section 5 of Volume I.

QGs for commercial/industrial development with no engineering controls are based on the following assumptions:

- A building constructed on a concrete slab six inches in thickness overlies a subsurface soil and/or groundwater source of volatile CPCs (i.e., Volatile organic compounds (VOCs) and mercury) of ubiquitous concentration and of equal size to the floor area of the overlying building.
- Vapors of volatile CPCs migrate from the subsurface source, through cracks in the floor slab equaling in area 0.1% of the total slab area, and establish equilibrium conditions in indoor building air.

- An indoor commercial/industrial worker is exposed by the inhalation route to the indoor air 8 hours per day, 250 days per year, for 25 years. This receptor is not exposed to soil outdoors via direct contact or volatile inhalation.
- An outdoor commercial/industrial worker is exposed to accessible Site soil (soil 0 to 15.24 cm that is not paved) by incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor emissions 250 days per year, for 25 years, and by inhalation to volatiles in groundwater that may migrate to ambient (outdoor) air for the same duration.
- A construction worker is exposed to Site soil by incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor emissions 250 days per year, for 1 year.
- A construction worker is exposed to Site groundwater by inhalation of vapor emissions, 250 days per year, for 1 year.
- A deed restriction is in place to prevent use of on-site groundwater for potable and non-potable uses, thereby preventing direct-contact exposures to all receptors.

QGs for the commercial/industrial development with engineering controls are based on the following assumptions:

- A building constructed on a concrete slab six inches in thickness overlies a subsurface soil and/or groundwater source of volatile CPCs (i.e., VOCs and mercury) of ubiquitous concentration and of equal size to the floor area of the overlying building.

- Vapors of volatile CPCs that may migrate from the subsurface source are restricted from entering the indoor building air by implementation of specific engineering controls as mandated in a deed restriction. There are therefore no exposures to soil or groundwater for indoor commercial/industrial workers.
- A deed restriction is in place requiring that the Site be paved to prevent outdoor commercial/industrial worker exposures to accessible Site soil (soil 0 - 15.24 cm) by incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor emissions, or to groundwater via inhalation of volatiles that may migrate to ambient (outdoor) air.
- A construction worker is exposed to Site soil by incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor emissions 250 days per year, for 1 year.
- A construction worker is exposed to Site groundwater by inhalation of vapor emissions, 250 days per year, for 1 year.
- A deed restriction is in place to prevent use of on-site groundwater for potable and non-potable uses, thereby preventing direct-contact exposures to all receptors.

QGs for the no further development scenario are based on the following assumptions:

- A deed restriction is in place to prevent the current conditions at the Site from being altered; the Site will remain paved and undeveloped (i.e., no construction of buildings).

- The Site will be restored to current conditions (i.e., a paved lot) if below-pavement soils are temporarily exposed for utility work; soil excavated for utility work will be placed back into the excavation or relocated to an appropriate off-site disposal site.
- A utility worker is exposed to Site soil by incidental ingestion, dermal contact, and inhalation of fugitive dust and vapor emissions, and to Site groundwater via vapor inhalation, 5 days per week for 1 month.
- A deed restriction is in place to prevent use of on-site groundwater for potable and non-potable uses, thereby preventing direct-contact exposures to all receptors.

The soil and groundwater QGs were developed in accordance with standard risk assessment technical approaches described by ASTM (1995) and USEPA (1989), and are based on conservative default exposure parameters developed by ASTM (1995) that are unlikely to result in underestimation of risk.

The direct-contact RBCs for mercury incorporate a site-specific bioavailability adjustment factor. This factor was developed to reduce uncertainty associated with the oral bioavailability of mercury in Site soils. The BAF was used to adjust the receptor-specific estimated oral intake of mercury in soils to be consistent with the oral bioavailability of mercuric chloride, on which is the mercury species which the mercury oral dose-response value is based. The overall oral bioavailability of mercury in Site soils was found to be approximately 15%. Compared to the oral bioavailability of mercuric chloride, which was found to be approximately 78%, mercury in Site soils has low bioavailability. The resulting calculated bioavailability adjustment factor for mercury in Site soils is 0.2, or

20% of the oral bioavailability of mercuric chloride. Due to conservative approaches and assumptions incorporated into development of the BAF, this BAF is unlikely to underestimate the bioavailability of mercury in Site soils. The BAF is consistent with bioavailability assessment approaches and adjustment factors used at other Sites for which Records of Decision (RODs) have been signed, including the Alameda Quicksilver County Park (CDM, 1992) and the Lower East Fork Poplar Creek (USEPA, 1995).

The limited oral bioavailability of mercury in Site soils is consistent with the mercury chemical speciation in Site soils. More than 78% of the mercury in Site soils was found to be present as elemental mercury (average 63%) and mercuric sulfide (average 15%), which are mercury species associated with extremely limited bioavailability and low toxicity through the oral exposure route. Only 22% of the mercury in Site soils was found to be present in potentially bioavailable forms, including acid-soluble mercury species (21% average) and organic mercury (less than 1% average). Organic mercury, representing the smallest fraction of total mercury in Site soils, is a more toxic form of mercury via the oral route than elemental mercury, which is the mercury species representing the largest fraction of total mercury.

The soil and groundwater QGs developed for the Site represent the range of possible remedial goals for various land development scenarios. The focused development scenario is based on applying the QGs discussed above, with their accompanying land-use restrictions, to various areas of the Site in order to balance future Site development with risk management and future Site owner interest considerations. For each land development scenario, the lowest soil and groundwater QG for each chemical of concern (COC) is protective for future exposures to Site soil and groundwater, respectively, under the

assumed exposure conditions for each land development scenario. Under these exposure conditions, potential exposures to Site soil or groundwater would not result in unacceptable risks if the soil and groundwater CPCs that exceed QGs were reduced in concentration to meet the Qgs.

6.2 CONCLUSIONS

The conclusions of this HHRA are as follows.

1. The HHRA developed a set of Quality Goals (QGs) appropriate to the Ames Street Site's likely commercial/industrial future use. Because the Site's future use may feature a mix of commercial/industrial activities and/or land uses (e.g., some portions occupied by buildings, some portions simply paved) a number of use-based QGs were developed which are appropriate for application at particular site locations based on the location's use and use restrictions.
2. There are no substantial Site visitor or trespasser exposures to Site soil and groundwater under current land use conditions. Based on a comparison of Quality Goals to soil and groundwater data, there does not appear to be any threat to human health associated with the soil and on-site groundwater at the Ames Street Site under current land use (i.e., vacant and secured land), and existing data suggest there is low potential for a significant off-site human health risk due to the Ames Street site. The only exposures to Site soil and groundwater that could potentially occur under current land use conditions, or under the assumptions for the no further development land use scenario, are to a utility worker engaged in infrequent below-ground utility work. Although lead

concentrations exceed the screening RBC, the RBC is clearly overprotective and no significant threat to a utility worker is believed to exist. This can be insured through the imposition of reasonable site Health and Safety Plans addressing such work.

3. Comparison of site data to QGs indicates that there while there are no unacceptable human health risks associated with the site in its current condition, or under a no further development land use, unacceptable risks may be present under the future commercial/industrial development scenarios unless some remediation is undertaken.

4. Soil direct contact Quality Goals for mercury incorporate a site-specific bioavailability adjustment factor of 0.2. This factor is consistent with the BAFs developed at other sites for which RODs have been signed, and the mercury speciation results for Site soils, which are consistent with the historical use of the Site, indicate that up to 90% of the mercury in Site soils is in the elemental form, with most of the remainder in acid-soluble and mercuric sulfide forms, and less than 1% in organic forms.

5. The QGs developed in the HHRA are conservative. For inhalation exposures, there is some uncertainty associated with the vapor migration fate and transport model inputs such as floor slab crack factor and air exchange rate. The inputs selected for use are believed to overestimate indoor air volatile CPC concentrations and are thus conservative. Collection of actual soil gas data would be the most expedient way to reduce the model's conservativeness if this is deemed desirable.

For direct contact exposures there is some uncertainty associated with the BAF developed for mercury. Bioaccessibility test results exhibited good general agreement with the known forms of mercury historically used at the Site, the results of Site-specific speciation analyses, the scientific literature and results from other sites with similar mercury contamination. Some data quality limitations - believed to be associated primarily with sample heterogeneity and the inherent difficulty in measuring and working with elemental mercury - were observed. Because the exposure parameters used by the RBCA models likely overestimate the intensity of potential direct contact exposures that may occur under the future use of the Site, the QGs developed in this HHRA are considered adequately health-protective and unlikely to underestimate potential risks to the evaluated receptors.

6. It was not possible to develop a direct contact exposure QG for lead under any of the site use scenarios due to a lack of an accepted dose-response value for this compound. The screening RBC of 400 mg/kg is overprotective for the current and potential future commercial/industrial uses of the Site (i.e., it is based on residential exposures to children), and should not be used as an actual Quality Goal. Promulgation of an accepted dose-response value for lead within the time frame for Site redevelopment is believed to be unlikely.

   As discussed in Section 5 of Volume I, the presence of lead concentrations above the screening is strongly correlated with elevated mercury levels. Because many methods designed to mitigate mercury exposure are expected to mitigate lead exposure, there is a significant possibility that mitigation of lead will occur as a consequence of mercury mitigation. It is therefore concluded that the best way to address the lack of a lead QG is to perform a re-evaluation of lead-

associated risk following design or implementation of mercury mitigation measures. If at this point a QG is still required, efforts will probably need to focus on employment of one of several available biokinetic models for lead uptake.

7. It was not possible to develop 1, 2 4-trimethylbenzene, 1, 3, 5-trimethylbenzene, and n-butylbenzene which were all detected in a pre-Phase I sample. Although trimethylbenzenes and n-butylbenzenes were not COCs during Phase I, they are strongly associated with gasoline which is believed to have been released at only the single site location represented by the sample. As discussed in Section 5 of Volume I, because the associated soil concentration is below STARS criteria for leaching potential for these compounds, the gasoline-impacted area (former Tank 2) is believed to be limited. The impacted area is entirely contained within a larger area of trichloroethene-impacted soil and groundwater; it is considered unlikely these compounds represent a current or future human health risk, and the lack of a screening RBC and QG is not considered significant.

8. It was not possible to develop a groundwater-based inhalation screening RBC or QG for 1,2-dichloroethene. Due to this CPC's potential to exist over a greater area of the Site (i.e., in association with trichloroethene), and at relatively high concentrations in samples from the two trichloroethene source areas, it is not possible to confidently evaluate it's potential implications with respect to future human health risk at this time. Because mitigation of trichloroethene-related risks will likely also address 1,2-dichloroethene, it is concluded that performing a residual risk analysis subsequent to design or implementation of trichloroethene mitigative measures is the best way to address lack of a 1,2-dichloroethene Quality Goal. If these mitigative measures do not effectively address 1,2-

dichloroethene, additional efforts will likely need to focus on developing an appropriate dose-response value for 1,2-dichloroethene.

6.3 RECOMMENDATIONS

Recommendations relative to potential human health risks at the Ames Street Site are as follows.

1. If Site is to be used for commercial/industrial development, the QGs developed by the HHRA should mitigate human health risks (e.g., by adopting them as remedial goals, developing mitigative measures, etc.).
2. Because QG development occurred under a set of baseline use assumptions (e.g., type of land use, no groundwater use), use of the QGs must be proceeded by application of the necessary legal and other mechanisms necessary to ensure these assumptions continue to be true for the life of QG application.
3. Because all QGs are equally protective of human health, the selection of the specific QGs adopted for a particular Site location should be left to the owner/developer, based on a strategic Site development plan.
4. The option on the part of the Site owner/developer to collect soil gas data in order to refine the QGs for the volatile migration pathway should be preserved.
5. A residual risk analysis should be performed for lead and 1,2-dichloroethene following design or completion of measures intended to address mercury and trichloroethene impacts, respectively. Should these analyses indicate a continued human health risk, a biokinetic model (or other appropriate method) should be utilized in order to develop an appropriate QG for lead, and development of an

appropriate dose-response value (for use with the RBCA-based groundwater vapor migration model) should be undertaken for 1,2-dichloroethene.

REFERENCES

American Concrete Institute (ACI) ; 1980; Control of Cracking in Concrete Structures. ACI Report 224-R-80. 1980.

American Society for Testing and Materials, 1995; Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites, E1739-95, November.

Barltrop, D., Meek, F.; 1979; Effect of Particle Size on Lead Absorption from the Gut. Arch. Env. Health; 34:280-285

Camp Dresser & McKee, Inc. (CDM), 1992; Final Report. Risk Assessment, Almaden Quicksilver County Park. Camp Dresser & McKee, Inc., Denver, CO.

Chaney, R.L, Mielke, H.W., Sterrett, S.B., 1989; Speciation, Mobility and Bioavailability of Soil Lead. Environ. Geochem. Health; 11:105-129.

Department of Energy (DOE), 1995; Record of Decision for Lower East Fork Poplar Creek. DOE/OR/02-1270&D1. U.S. Department of Energy, Office of Environmental Restoration and Waste Management. Prepared by Jacobs ER Team, Oak Ridge, TN.

Duggan, M.J., Inskip, M.J.; 1985; Childhood exposure to lead I surface dust and soil: A community health problem; Public Health Rev; 13:1-54.

Gas Research Institute (GRI), 1994; User's Manual for HgSCREEN. A Risk-Based Screening Model for Mercury Contaminated Sites. July, 1994.

Goyer, R.A. 1996; Toxic effects of metals. In: Casarett and Doull's Toxicology: The Basis Science of Poisons. Fourth edition. C.D. Klaassen, M.O. Amdur, and J. Doull(eds.) McGraw-Hill.

Howard, P.H. 1990; Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Vol. I and II. Lewis Publishers, Inc. Chelsea, MI.

Hursh, John B,. T.W. Clarkson,. E.F. Miles and L.A. Goldsmith, 1989; Archives of Environmental Health; 44(2), March/April.

Lyman, Warren J., W.F. Rheel and D.H. Rosenblatt, 1990; Handbook of Chemical Property Estimation Methods: Environmental Behavior of Organic Compounds; American Chemical Society, Washington, D.C., 1990.

New York State of Department of Environmental Conservation (NYSDEC), 1993: Ambient Water Quality Standards and Guidance Values; Division of Water, Albany, New York, October

NYSDEC, 1994; Division Technical and Administrative Guidance Memorandum: Determination of Soil Cleanup Objectives and Cleanup Levels; Division of Hazardous Waste Remediation, January.

NYSDEC, 1995; Site Assessment and Closure Guidance for Petroleum Impacted Sites; Division of Spills Management, September.

Portland Cement Association (PCA); 1979. Design and Control of Concrete Mixtures. Twelfth Edition.

PTI Environmental Services, Inc. 1996; Development of a Bioavailability Adjustment Factor for Mercury in Soils at the Ames Street Site, Rochester, New York. PTI Environmental Services, Boulder, Colorado, July.

USEPA, 1986; Superfund Public Health Evaluation Manual: Office of Emergency and Remedial Response, EPA/540/1-86/086, Washington, D.C., October.

USEPA, 1988; Superfund Exposure Assessment Manual: Office of Remedial Response, EPA/540/1-88/001, Washington, D.C., April.

USEPA, 1989; Risk Assessment Guidance for Superfund Volume 1 Human Health Evaluation Manual (Part A): Office of Emergency and Remedial Response, EPA/540/1-89/002, Washington, D.C., December (interim final).

USEPA, 1990, Code of Federal Regulations, Title 40, Part 300, National Oil and Hazardous Substances Pollution Contingency Plan: Federal Register, March 8.

USEPA, 1992; Air/Superfund National Technical Guidance Study Series. Assessing Potential Impacts for Superfund Sites. Office of Air Quality Planning and Standards. Research Triangle Park, NC. EPA-451/R-92-002. September, 1992.

USEPA, 1993; Superfund Chemical Data Matrix. March, 1993.

USEPA, 1994; Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities: memorandum from Elliott P. Laws, Assistant Administrator, Office of Solid Waste and Emergency Response, OSWER Directive 9355.4-12,Washington, D.C.

USEPA, 1995a; Health Effects Summary Tables (HEAST), Annual Update: Office of Solid Waste and Emergency Response, EPA 540-R-95-036, PB94-921199, May.

USEPA, 1995b; HEAST Supplement November, 1995.

USEPA, 1995c; Supplemental Guidance to RAGS: Region IV Bulletins. Exposure Assessment. Human Health Risk Assessment Bulletin No. 3. USEPA Region IV Waste Management Division, Atlanta. November, 1995.

USEPA, 1995d; Carson River Mercury Superfund Site: "EPA Selects Cleanup Actions for Soils" USEPA Region IX, San Francisco, CA.

USEPA, 1996a; Drinking Water Regulations and Health Advisories: Office of Water, Washington, D.C, May.

USEPA, 1996b; Integrated Risk Information System (IRIS): on-line database search, February.

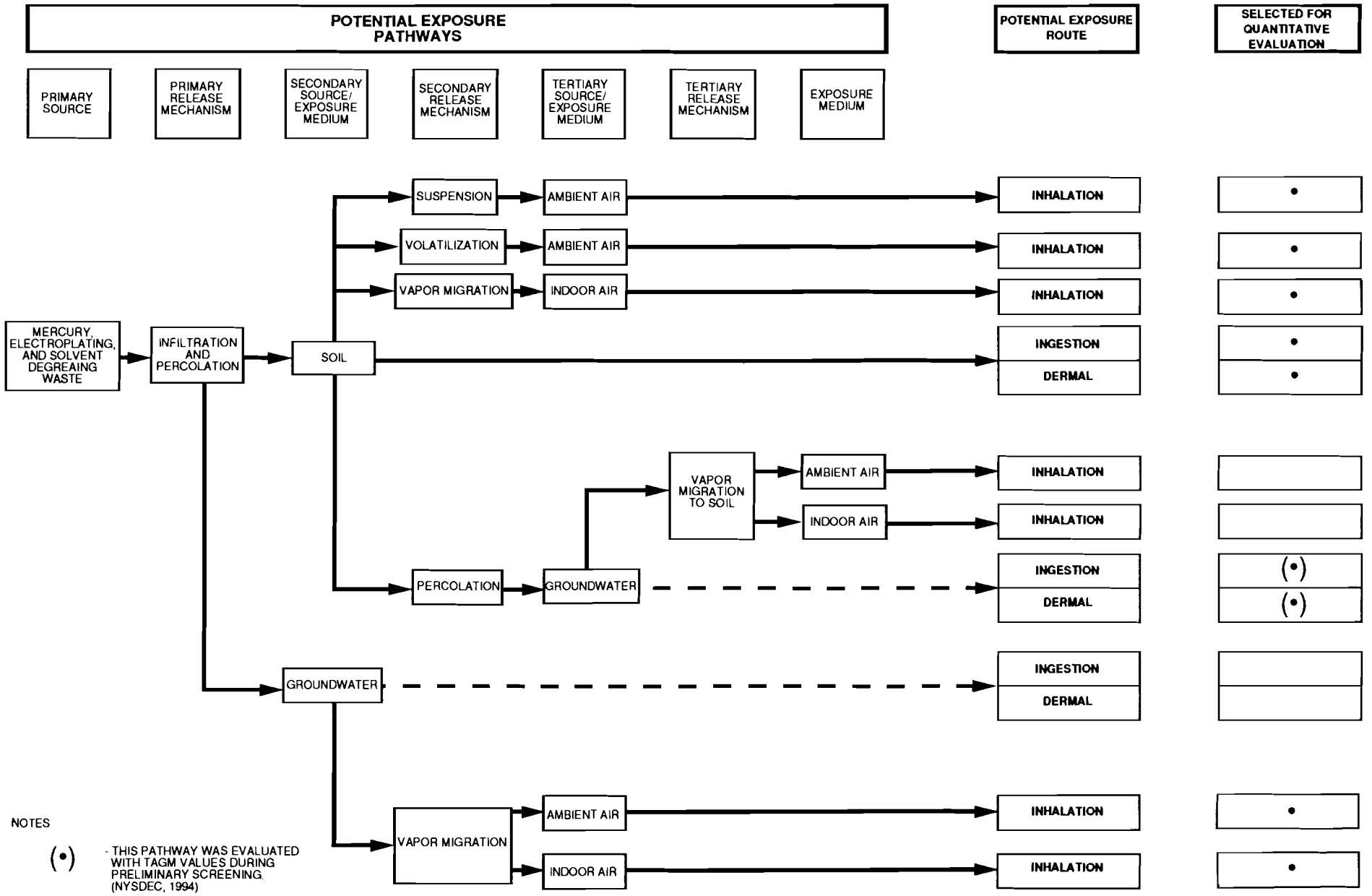


ABB Environmental Services, Inc.

FIGURE 1
POTENTAIL EXPOSURE PATHWAYS FOR HUMAN RECEPTORS
HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NEW YORK

7198.23

TABLE 1
SUMMARY OF SOURCES FOR DATA INCLUDED IN SOIL AND GROUNDWATER DATA SETS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

INVESTIGATION AND DATE	SOIL			GROUNDWATER		
	VOCs	Inorganics and Cyanide	Mercury	VOCs	Inorganics and Cyanide	Mercury
Pre-phase I Investigation (1993)	X [b]	X [b]	X [b]	X [b]	X [b]	X [b]
Tank 11 and 12 (1995)	X [b]	NA	NA	X [b]	NA	NA
Phase I Site Investigation (1995/1996)	X [a]	X [b]	X [c]	X [b]	X [b]	X [b]

Notes:
[a] = On-site laboratory analyses
[b] = Off-site laboratory analyses
[c] = On-site analyses with Leeman Analyzer
NA = Not analyzed

TABLE 2
SELECTION OF CHEMICALS OF POTENTIAL CONCERN IN SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

Chemical	Range of SQLs	Frequency of Detection	Detected Concentration Minimum	Detected Concentration Maximum	Mean of all Samples [a]	Screening Value [b]	Chemical of Poential Concern? [c]
Volatile Organic Compounds (ug/kg) [d]							
1,1,1-Trichloroethane	1 - 28,000	3 / 95	1.6	2.90	305	76	NO
4-Methyl-2-pentanone	21 - 56,000	9 / 95	19	13,000	668	39	YES
trans-1,2-Dichloroethene	5 - 7,000	1 / 93	14	14	49	32	NO
Ethylbenzene	1 - 230,000	13 / 111	1.6	62,000	2,121	780	YES
Tetrachloroethene	1 - 28,000	23 / 95	1.1	28,000	462	470	YES
Toluene	1 - 27,000	10 / 111	1.4	1,800,000	20,798	240	YES
Trichloroethene	1 - 28,000	71 / 95	1.5	280,000	6,719	110	YES
Xylenes - total [e]	1 - 2,800	18 / 111	1.8	1,830,000	23,961	NA	YES
Inorganic Analytes (mg/kg) [f]							
Cadmium	0.524 - 0.573	3 / 15	4.18	177	13.2681	1	YES
Chromium	NA	17 / 17	3.81	27	9.9856	10	YES
Hexavalent Chromium	0.539 - 0.595	6 / 17	0.846	33	3.5296	ND	YES
Cyanide	0.268 - 0.293	4 / 14	0.292	37	2.94	ND	YES
Lead	8.2 - 11.5	8 / 23	34	36,000	1710.7337	200	YES
Nickel	NA	14 / 14	4.87	344	43.3396	13	YES
Zinc	NA	14 / 14	12.3	483	81.2821	20	YES
Mercury [g]	0.1 - 8.5	281 / 540	0.036	12,800	59.2732	0.1	YES

NOTES:
[a] Mean of all samples is the arithmetic mean concentration, using 1/2 the detection limit as the analytical result for samples in which the analyte was reported as not detected.
[b] Screening value is the recommended soil cleanup value reported in the TAGM (NYSDEC, 1994), adjusted for site-specific TOC (1.54%) and Correction Factor (10).
[c] Chemical of Potential Concern (COCs) are those chemicals with maximum detected concentrations greater than the screening value. COCs were retained for development of site-specific Quality Goals.
[d] Samples included in this data set are identified in Appendix B, Table B-1.
[e] Sum of concentrations for o-, m-, and p-isomers.
[f] Samples included in this data set are identified in Appendix B, Table B-2.
[g] Samples included in this data set are identified in Appendix B, Table B-3.
NA = Not Applicable

TABLE 3
SUMMARY OF PERIMETER WELL DATA AND COMPARISON TO CLASS GA GROUNDWATER STANDARDS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

Chemical	Range of SQLs	Frequency of Detection	Detected Concentration Minimum	Detected Concentration Maximum	Mean of all Samples [a]	Class GA Groundwater Standard [b]	Exceeds Class GA Standard ? [c]
Volatile Organic Compounds (ug/L)							
Acetone	9 - 17	1 / 23	43	43	6.3478	50	NO
Cis-1,2-Dichloroethene	2.4 - 14	5 / 23	4.3	26	4.2783	70 [d]	NO
Toluene	1.7 - 1.7	1 / 23	2.1	2.1	0.9044	5	NO
Trichloroethene	3 - 11	14 / 23	3.1	4100	436.3304	5	YES
trans-1,2-Dichloroethene	2.4 - 2.4	2 / 23	9.9	12	2.0478	100 [d]	NO
Inorganic Analytes (ug/L)							
Chromium,total	10 - 10	1 / 8	26.4	26.4	7.675	50	NO
Lead,total	20 - 20	1 / 8	21.7	21.7	11.4625	25	NO
Nickel,total	15 - 15	1 / 8	30.2	30.2	10.3375	100 [d]	NO
Zinc,total	30 - 30	1 / 8	100	100	25.625	300	NO
Cyanide	5 - 5	2 / 4	5	28	9.2	100	NO
Mercury,total	0.2 - 0.9	5 / 23	0.23	67.7	3.1161	2	YES

NOTES:
Samples included in this data set are identified in Appendix B, Table B-4.
[a] Mean of all samples is the arithmetic mean concentration, using 1/2 the detection limit as the analytical result for samples in which the analyte was reported as not detected.
[b] Class GA values from "Ambient Water Quality and Guidance Values". Divison of Water. NYSDEC. October, 1993., except as noted.
[c] Comparison of maximum detected groundwater concentration to Class GA standard.
[d] Class GA standard unavailable; value is the federal maximum contaminant level from "Drinking Water Standards and Health Advisories" (USEPA, 1996).

TABLE 4
SUMMARY OF INTERIOR WELL DATA

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

Chemical	Range of SQLs	Frequency of Detection	Detected Concentration		Mean of all Samples[a]
			Minimum	Maximum	
Volatile Organic Compounds (ug/L)					
1,1,1-Trichloroethane	2.5 - 2.5	1 / 6	8.4	9	3.11
Ethylbenzene	1.3 - 1,000	2 / 8	35	94	105
Toluene	1.7 - 1,000	3 / 8	2.5	18,000	2,592
Trichloroethene	3 - 3	3 / 6	20	32,000	5,392
Vinyl Chloride	1 - 1	1 / 8	12.5	12.5	3.5
Xylenes,total	3.7 - 2,000	3 / 8	710	12,000	2,257
Benzene	1 - 1,000	1 / 8	10	10	64
Tetrachloroethene	3 - 3	2 / 6	22	31	9.8
1,2-Dichloroethene (total)	3 - 3	2 / 6	550	1,600	359
1,3,5-Trimethylbenzene	20 - 1,000	1 / 2	300	300	400
1,2,4-Trimethylbenzene	20 - 1,000	1 / 2	170	170	335
n-Butylbenzene	20 - 1,000	1 / 2	96	96	298
Inorganic Analytes (ug/L)					
Nickel, Dissolved	15 - 15	1 / 3	18.4	18.5	11.2
Nickel,total	15 - 15	1 / 3	19.3	22.9	12
Mercury, Dissolved	0.2 - 0.2	2 / 3	0.36	3.64	1.37
Mercury,total	0.2 - 0.2	2 / 5	12.2	107	23.8

NOTES:
Samples included in this data set are identified in Appendix B, Table B-5.
[a] Mean of all samples is the arithmetic mean concentration, using 1/2 the detection limit as the analytical result for samples in which the analyte was reported as not detected.

TABLE 5
SUMMARY OF CPCs FOR SOIL AND ON-SITE GROUNDWATER

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	SOIL	GROUNDWATER
1,1,1-Trichloroethane		X
1,2-Dichloroethene (total)		X
1,2,4-Trimethylbenzene		X
1,3,5-Trimethylbenzene		X
4-Methyl-2-pentanone	X	
Benzene		X
Ethylbenzene	X	X
n-Butylbenzene		X
Tetrachloroethene	X	X
Toluene	X	X
Trichloroethene	X	X
Vinyl chloride		X
Xylenes	X	X
Cadmium	X	
Chromium	X	
Nickel	X	
Lead	X	
Zinc	X	
Cyanide	X	
Mercury	X	X

TABLE 6
SUMMARY OF POTENTIAL EXPOSURE PATHWAYS AND RECEPTORS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

LAND USE	MEDIUM	RECEPTOR	EXPOSURE ROUTES	SELECTED FOR EVALUATION?	RATIONALE
CURRENT	Soils	Trained site investigators/workers	Ingestion Dermal Inhalation (dust and/or vapors)	No No No	Current workers are operating under a health and safety plan.
		Utility Workers	Ingestion Dermal Inhalation (dust and/or vapors outdoors)	Yes Yes Yes	Utilities workers may be exposed to soil during infrequent utility maintenance/repair activities.
		Trespassers (area residents)	Ingestion Dermal Inhalation (dust and/or vapors outdoors)	No No No	Access to the site is limited by a fence and ground surfaces are paved/covered by buildings.
	Groundwater	Area residents	Ingestion Dermal Inhalation of vapors	No No No	Neighboring residences are served by remote public water supply.
		Site investigators/workers	Ingestion Dermal Inhalation of vapors	No No No	There is no use of groundwater for potable or non-potable uses on site.
		Utility Workers	Ingestion Dermal Inhalation of vapors	No No Yes	There is no use of groundwater for potable or non-potable uses on site. Workers may be exposed to vapors migrating into subsurface excavations.

TABLE 6
SUMMARY OF POTENTIAL EXPOSURE PATHWAYS AND RECEPTORS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

LAND USE	MEDIUM	RECEPTOR	EXPOSURE ROUTES	SELECTED FOR EVALUATION?	RATIONALE
CURRENT (CON'T)	On-site Surface Water/Sediment	Trespassers, Site workers	Ingestion Dermal Inhalation	No No No	No surface water/sediment present on site.
FUTURE	Soils	On-site Residents	Ingestion Dermal Inhalation (dust and/or vapors)	No No No	Property will be industrial or commercial use with virtually total paving/building coverage. Residential use will be prevented by a deed restriction.
		Area Residents and site visitors	Ingestion Dermal Inhalation (dust and/or vapors)	No No No	Property will be industrial or commercial use with virtually total paving/building coverage. Potential on-site worker exposures would be greater than off-site receptor exposures.
		Construction Workers	Ingestion Dermal Inhalation (dust and/or vapors outdoors)	Yes Yes Yes	During and after site redevelopment construction workers are potentially exposed to impacted soils.
		Indoor Industrial/Commercial Workers	Ingestion Dermal Inhalation (vapors indoors)	No No Yes	After redevelopment on-site workers could potentially be exposed to vapors that migrate to indoor building air.
		Outdoor Industrial/Commercial Workers	Ingestion Dermal Inhalation (dust and/or vapors outdoors)	Yes Yes Yes	After redevelopment on-site workers could potentially be exposed to impacted soils and vapors that migrate to ambient air.

TABLE 6
SUMMARY OF POTENTIAL EXPOSURE PATHWAYS AND RECEPTORS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

LAND USE	MEDIUM	RECEPTOR	EXPOSURE ROUTES	SELECTED FOR EVALUATION?	RATIONALE
FUTURE (CON'T)	Groundwater	On-site residents	Ingestion Dermal Inhalation of vapors	No No No	Property will be industrial or commercial use with virtually total paving/building coverage. Residential use will be prevented by a deed restriction.
		Industrial/Commercial Workers and Construction Workers	Ingestion Dermal Inhalation of vapors	No No Yes	Deed restriction will prohibit use of site groundwater for potable or non-potable purposes. On-site workers may be potentially exposed to vapors that migrate to indoor and outdoor air.
		Area residents	Ingestion Dermal Inhalation of vapors	No No No	Neighboring residences are served by remote public water supply; the Phase I VSI addresses on-site characterization and exposure.

TABLE 7
EXPOSURE PARAMETERS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETER	Units	Value
Averaging Time - Carcinogen	yr	70
Averaging Time - Noncarcinogen (equals exposure duration) :		
On-site Commercial/Industrial Worker	yr	25
Construction Worker	yr	1
Utility Worker	month	1
Body Weight		
Adult receptors	kg	70
Exposure Duration		
On-site Commercial/Industrial Worker	yr	25
Construction Worker	yr	1
Utility Worker	month	1
Exposure Frequency		
On-site Commercial/Industrial Worker	days/yr	250
Construction Worker	days/yr	250
Utility Worker	days/month	22
Soil ingestion rate		
On-site Commercial/Industrial Worker	mg/day	58.6
Construction Worker/Utility Worker	mg/day	58.6
Daily Indoor Inhalation Rate		
Commercial/Industrial Workers	m^3/day	20
Daily Outdoor Inhalation		
Commercial/Industrial Worker, Construction Worker, Utility Worker	m^3/day	20
Soil skin adherence factor	mg/cm^2	0.5
Oral relative absorption factor	---	1
Dermal relative absorption factor (volatiles)	---	0.5
Dermal relative absorption factor (PAHS)	---	0.005
Skin surface area for dermal contact with soil		
Adult receptors	cm^2	3160

TABLE 7
EXPOSURE PARAMETERS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETER	Units	Value
Target Hazard Quotient for individual constituents	---	1
Target Excess Individual Lifetime Cancer Risk	---	1×10^{-6}

NOTES:
Values presented are TIER I Default Parameters from "Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites" ASTM Standard E1739-95 (November, 1995).
Construction worker values were used for utility worker, with the exception of exposure frequency, exposure duration, and averaging time. For these parameters, values chosen are conservative estimates.
--- = unitless

TABLE 8
SUMMARY OF SCREENING RISK BASED CONCENTRATIONS (RBCs) FOR THE COMMERCIAL WORKER - SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC [a]	RBC - COMMERCIAL WORKER (mg/kg) [b, c]			
	Direct Contact (cancer)	Direct Contact (non-cancer)	Indoor Inhalation of Volatiles (cancer)	Indoor Inhalation of Volatiles (non-cancer)
4-Methyl-2-pentanone	NA	3,800	NA	1,200
Ethylbenzene	NA	11,000	NA	20,000
Toluene	NA	10,000	NA	2,400
Trichloroethene	31	720	2.7	ND
Xylenes	NA	19,000	NA	3,800
Tetrachloroethene	6.5	1200	19	NA
Cadmium	ND	370	NA	NA
Chromium	150,000	2,500	NA	NA
Nickel	>	3,300	NA	NA
Lead	400 [d]	400 [d]	NA	NA
Zinc	NA	340,000	NA	NA
Cyanide	NA	19,000	NA	NA
Mercury	NA	2,500	NA	4,100

NOTES:
[a] Contaminants of Potential Concern identified in Table 5.
[b] Developed using approach described in "Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites" (ASTM Standard #E1739-95; November 1995).
[c] For screening purposes the target cancer risk level was set at an excess lifetime cancer risk of $1x10^{-6}$ for carcinogens, and the target hazard index was set at 1 for non-carcinogens.
[d] This value is the OSWER cleaup value for lead contaminated soils from "Revised Interim Soil Lead Cleanup Guidance for CERCLA Sites and RCRA Corrective Action Facilities (OSWER 9355.4-12)
> = Calculated RBC is greater than 1 million mg/kg.
NA = Not Applicable.
ND = No dose-response data available for calculating RBC.

TABLE 9
SUMMARY OF SCREENING RISK BASED CONCENTRATIONS (RBCs) FOR THE CONSTRUCTION WORKER – SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC [a]	RBC – CONSTRUCTION WORKER (mg/kg) [b, c]			
	Direct Contact (cancer)	Direct Contact (non-cancer)	Outdoor Inhalation of Volatiles (cancer)	Outdoor Inhalation of Volatiles (non-cancer)
4-Methyl-2-pentanone	NA	940	NA	2,500
Ethylbenzene	NA	7,000	NA	41,000
Toluene	NA	3,200	NA	5,000
Trichloroethene	750	720	140	ND
Xylenes	NA	4,000	NA	8,000
Tetrachloroethene	160	1200	990	NA
Cadmium	ND	370	NA	NA
Chromium	38	2,500	NA	NA
Nickel	1,900	3,300	NA	NA
Lead	400 [d]	400 [d]	NA	NA
Zinc	NA	340,000	NA	NA
Cyanide	NA	19,000	NA	NA
Mercury	NA	2,500	NA	8,600

NOTES:
[a] Contaminants of Potential Concern identified in Table 5.
[b] Developed using approach described in "Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites" (ASTM Standard #E1739-95; November 1995).
[c] For screening purposes the target cancer risk level was set at an excess lifetime cancer risk of $1x10^{-6}$ for carcinogens, and the target hazard index was set at 1 for non-carcinogens.
[d] This value is the OSWER cleaup value for lead contaminated soils from "Revised Interim Soil Lead Cleanup Guidance for CERCLA Sites and RCRA Corrective Action Facilities (OSWER 9355.4-12)
NA = Not Applicable.
ND = No dose-response data available for calculating RBC.

TABLE 10
SUMMARY OF SCREENING RISK BASED CONCENTRATIONS (RBCs) FOR THE UTILITY WORKER – SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC [a]	RBC – UTILITY WORKER (mg/kg) [b, c]			
	Direct Contact (cancer)	Direct Contact (non-cancer)	Outdoor Inhalation of Volatiles (cancer)	Outdoor Inhalation of Volatiles (non-cancer)
4-Methyl-2-pentanone	NA	11,000	NA	28,000
Ethylbenzene	NA	80,000	NA	460,000
Toluene	NA	36,000	NA	56,000
Trichloroethene	34,000	8,200	19,000	ND
Xylenes	NA	46,000	NA	91,000
Tetrachloroethene	20,000	14,000	140,000	ND
Cadmium	ND	4,200	NA	NA
Chromium	5,200	29,000	NA	NA
Nickel	250,000	38,000	NA	NA
Lead	400 [d]	400 [d]	NA	NA
Zinc	NA	>	NA	NA
Cyanide	NA	220,000	NA	NA
Mercury	NA	28,000	NA	97,000

NOTES:
[a] Contaminants of Potential Concern identified in Table 5.
[b] Developed using approach described in "Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites" (ASTM Standard #E1739-95); November 1995).
[c] For screening purposes the target cancer risk level was set at an excess lifetime cancer risk of $1x10^{-6}$ for carcinogens, and the target hazard index was set at 1 for non-carcinogens.
[d] This value is the OSWER cleaup value for lead contaminated soils from "Revised Interim Soil Lead Cleanup Guidance for CERCLA Sites and RCRA Corrective Action Facilities (OSWER 9355.4-12)
NA = Not Applicable.
ND = No dose-response data available for calculating RBC.
> = Calculated RBC is greater than 1 million mg/kg.

TABLE 11
SUMMARY OF SCREENING RISK BASED CONCENTRATIONS (RBCs) COMMERCIAL WORKER – GROUNDWATER

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC [a]	RISK BASED CONCENTRATION (mg/L) [b, c]			
	Indoor Inhalation of Volatiles (cancer)	Indoor Inhalation of Volatiles (non–cancer)	Outdoor Inhalation of Volatiles (cancer)	Outdoor Inhalation of Volatiles (non–cancer)
Vinyl chloride	0.009	ND	2.0	ND
Trichloroethene	2.3	ND	440	ND
Ethylbenzene	NA	(2,100)	NA	(400,000)
Toluene	NA	(820)	NA	(150,000)
Xylenes	NA	(670)	NA	(120,000)
1,1,1–Trichloroethane	NA	1,100	NA	(220,000)
Tetrachloroethene	3.9	ND	(820)	ND
Benzene	0.72	ND	130	ND
1,2–Dichloroethene (total)	NA	ND	NA	ND
1,3,5–Trimethylbenzene	ND	ND	ND	ND
1,2,4–Trimethylbenzene	ND	ND	ND	ND
n–Butylbenzene	ND	ND	ND	ND
Mercury (total)	NA	(0.71)	NA	(130)

NOTES:
[a] Contaminants of Potential Concern identified in Table 5.
[b] Developed using approach described in "Standard Guide for Risk–Based Corrective Action Applied at Petroleum Release Sites" (ASTM Standard #E1739–95; November 1995).
[c] For screening purposes the target cancer risk level was set at an excess lifetime cancer risk of $1x10^{-6}$ for carcinogens, and the target hazard index was set at 1 for non–carcinogens.
NA = Not Applicable.
ND = No Dose Response–Data Available
( ) = calculated RBC exceeds the water solubility limit for this compound.

TABLE 12
SUMMARY OF SCREENING RISK BASED CONCENTRATIONS (RBCs) FOR THE CONSTRUCTION WORKER – GROUNDWATER

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC [a]	RBC (mg/L) [b, c] – CONSTRUCTION WORKER	
	Outdoor Inhalation of Volatiles (cancer)	Outdoor Inhalation of Volatiles (non-cancer)
Vinyl chloride	49	ND
Trichloroethene	(11,000)	ND
Ethylbenzene	NA	(400,000)
Toluene	NA	(150,000)
Xylenes	NA	(120,000)
1,1,1-Trichloroethane	NA	(220,000)
Tetrachloroethene	(20,000)	ND
Benzene	(3,300)	ND
1,2-Dichloroethene (total)	NA	ND
1,3,5-Trimethylbenzene	ND	ND
1,2,4-Trimethylbenzene	ND	ND
n-Butylbenzene	ND	ND
Mercury (total)	NA	(130)

NOTES:
[a] Contaminants of Potential Concern identified in Table 5.
[b] Developed using approach described in "Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites" (ASTM Standard #E1739-95; November 1995).
[c] For screening purposes the target cancer risk level was set at an excess lifetime cancer risk of $1x10^{-6}$ for carcinogens, and the target hazard index was set at 1 for non-carcinogens.
NA = Not Applicable.
ND = No dose-response data available for calculating RBC
( ) = Calculated RBC exceeds the water solubility limit for this CPC.

TABLE 13
SUMMARY OF SCREENING RISK BASED CONCENTRATIONS (RBCs) FOR THE UTILITY WORKER – GROUNDWATER

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC [a]	RBC (mg/L) [b, c] – UTILITY WORKER	
	Outdoor Inhalation of Volatiles (cancer)	Outdoor Inhalation of Volatiles (non–cancer)
Vinyl chloride	6,800	ND
Trichloroethene	>	ND
Ethylbenzene	NA	>
Toluene	NA	>
Xylenes	NA	>
1,1,1–Trichloroethane	NA	>
Tetrachloroethene	>	ND
Benzene	(450,000)	ND
1,2–Dichloroethene (total)	NA	ND
1,3,5–Trimethylbenzene	ND	ND
1,2,4–Trimethylbenzene	ND	ND
n–Butylbenzene	ND	ND
Mercury (total)	NA	(1,500)

NOTES:
[a] Contaminants of Potential Concern identified in Table 5.
[b] Developed using approach described in "Standard Guide for Risk–Based Corrective Action Applied at Petroleum Release Sites" (ASTM Standard #E1739–95; November 1995).
[c] For screening purposes the target cancer risk level was set at an excess lifetime cancer risk of $1x10^{-6}$ for carcinogens, and the target hazard index was set at 1 for non–carcinogens.
NA = Not Applicable.
ND = No dose–response data available for calculating RBC.
> = Calculated RBC exceeds 1 million mg/L.
( ) = Calculated RBC exceeds the water solubility limit for this CPC.

TABLE 14
SUMMARY OF RECEPTOR EXPOSURE SCENARIOS APPLICABLE TO FUTURE LAND DEVELOPMENT SCENARIOS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

LAND DEVELOPMENT SCENARIO	RECEPTOR EXPOSURE SCENARIO EVALUATED AS THE BASIS OF QUALITY GOALS			
	Commercial Worker - Outside	**Commercial Worker - Inside**	**Utility Worker**	**Construction Worker**
Development with No Engineering Restrictions	1) Direct contact (accessible soils) 2) Outdoor inhalation of groundwater volatiles	1) Indoor inhalation of soil and groundwater volatiles	Not Applicable	1) Direct contact (all soils) 2) Outdoor inhalation of soil and groundwater volatiles
Commercial Development with Engineering Restrictions	Not Applicable	Not Applicable	Not Applicable	1) Direct contact (all soils) 2) Outdoor inhalation of soil and groundwater volatiles
No Further Development (Current Conditions)	Not Applicable	Not Applicable	1) Direct contact (all soils) 2) Outdoor inhalation of soil and groundwater volatiles	Not Applicable

TABLE 15
SELECTION OF QUALITY GOALS FOR SOIL – COMMERCIAL WORKER – INDOOR

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM CPC CONCENTRATION [a] (mg/kg)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
		Screening RBC [b] (mg/kg)	Pathway/Effect		Screening Target Risk [c]	Quality Goal Required? [d]	Quality Goal [d] (mg/kg)	Quality Goal Target Risk
Trichloroethene	280	2.7	Inhalation	c	$1x10^{-6}$	YES	2.7	$1x10^{-6}$
Toluene	1,800	2,400	Inhalation	nc	HI = 1	NO	NA	
Xylenes	1,830	3,800	Inhalation	nc	HI = 1	NO	NA	
Ethylbenzene	62	20,000	Inhalation	nc	HI = 1	NO	NA	
4-Methyl-2-pentanone	13	1,200	Inhalation	nc	HI = 1	NO	NA	
Tetrachloroethene	28	19	Inhalation	c	$1x10^{-6}$	YES	19	$1x10^{-6}$
Cadmium	177	NA						
Chromium	27	NA						
Nickel	344	NA						
Lead	36,000	NA						
Zinc	483	NA						
Cyanide	36.5	NA						
Mercury	12,800	4,100	Inhalation	nc	HI = 1	YES	4,100	HI = 1

NOTES:
[a] Phase I Data (maximum concentrations) from the data set described subsection 2.1.
[b] Lowest risk-based concentration for indoor inhalation of volatiles for commercial/industrial worker presented in Table 8.
[c] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[d] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
c = carcinogenic effects
nc = non-carcinogenic effects

TABLE 16
SELECTION OF QUALITY GOALS FOR SOIL – COMMERCIAL WORKER – OUTDOOR

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM CPC CONCENTRATION [a] (mg/kg)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
		Screening RBC [b] (mg/kg)	Pathway/Effect		Screening Target Risk [c]	Quality Goal Required? [d]	Quality Goal [d] (mg/kg)	Quality Goal Target Risk
Trichloroethene	280	31	Direct contact	c	$1x10^{-6}$	YES	31	$1x10^{-6}$
Toluene	1,800	10,000	Direct contact	nc	HI = 1	NO	NA	
Xylenes	1,830	19,000	Direct contact	nc	HI = 1	NO	NA	
Ethylbenzene	62	11,000	Direct contact	nc	HI = 1	NO	NA	
4-Methyl-2-pentanone	13	3,800	Direct contact	nc	HI = 1	NO	NA	
Tetrachloroethene	28	6.5	Direct contact	c	$1x10^{-6}$	YES	6.5	$1x10^{-6}$
Cadmium	177	370	Direct contact	nc	HI = 1	NO	NA	
Chromium	27	2,500	Direct contact	nc	HI = 1	NO	NA	
Nickel	344	3,300	Direct contact	nc	HI = 1	NO	NA	
Lead	36,000	400 [e]	Direct contact	nc	NA	YES	ND	
Zinc	483	340,000	Direct contact	nc	HI = 1	NO	NA	
Cyanide	36.5	19,000	Direct contact	nc	HI = 1	NO	NA	
Mercury	12,800	2,500	Direct contact	nc	HI = 1	YES	2,500	HI = 1

NOTES:
[a] Phase I Data (maximum concentrations) from the data set described subsection 2.1.
[b] Lowest risk-based concentration for direct contact for commercial/industrial worker presented in Table 8.
[c] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[d] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
[e] This value is the OSWER residential cleaup value for lead contaminated soils from "Revised Interim Soil Lead Cleanup Guidance for CERCLA Sites and RCRA Corrective Action Facilities" (OSWER 9355.4-12).
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
ND = Not Developed because sufficient dose--response data were unavailable for developing a site-spefic QG.
c = carcinogenic effects
nc = non-carcinogenic effects

TABLE 17
SELECTION OF QUALITY GOALS FOR SOIL – CONSTRUCTION WORKER

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM CPC CONCENTRATION [a] (mg/kg)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
		Screening RBC [b] (mg/kg)	Pathway/Effect		Screening Target Risk [c]	Quality Goal Required? [d]	Quality Goal [d] (mg/kg)	Quality Goal Target Risk
Trichloroethene	280	140	Inhalation	c	$1x10^{-6}$	YES	140	$1x10^{-6}$
Toluene	1,800	3,200	Direct contact	nc	HI = 1	NO	NA	
Xylenes	1,830	4,000	Direct contact	nc	HI = 1	NO	NA	
Ethylbenzene	62	7,000	Direct contact	nc	HI = 1	NO	NA	
4-Methyl-2-pentanone	13	940	Direct contact	nc	HI = 1	NO	NA	
Tetrachloroethene	28	160	Direct contact	c	$1x10^{-6}$	NO		
Cadmium	177	370	Direct contact	nc	HI = 1	NO	NA	
Chromium	27	38	Direct contact	c	$1x10^{-6}$	NO	NA	
Nickel	344	1,900	Direct contact	c	$1x10^{-6}$	NO	NA	
Lead	36,000	400 [e]	Direct contact	nc	NA	YES	ND	
Zinc	483	340,000	Direct contact	nc	HI = 1	NO	NA	
Cyanide	36.5	19,000	Direct contact	nc	HI = 1	NO	NA	
Mercury	12,800	2,500	Direct contact	nc	HI = 1	YES	2,500	HI = 1

NOTES:
[a] Phase I Data (maximum concentrations) from the data set described subsection 2.1.
[b] Lowest risk-based concentration for outdoor inhalation of volatiles or direct contact for construction worker presented in Table 9.
[c] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[d] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
[e] This value is the OSWER residential cleaup value for lead contaminated soils from "Revised Interim Soil Lead Cleanup Guidance for CERCLA Sites and RCRA Corrective Action Facilities" (OSWER 9355.4-12).
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
ND = Not Developed because sufficient dose-response data were unavailable for developing a site-spefic QG.
c = carcinogenic effects
nc = non-carcinogenic effects

TABLE 18
SELECTION OF QUALITY GOALS FOR GROUNDWATER – COMMERCIAL WORKER – INDOOR

HUMAN HEALTH ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM PERIMETER WELL CONCENTRATION [a] (mg/L)	MAXIMUM ON-SITE CONCENTRATION [b] (mg/L)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
			Screening RBC [c] (mg/L)	Pathway/Effect		Screening Target Risk [d]	Quality Goal Required? [e]	Quality Goal [e] (mg/L)	Quality Goal Target Risk
Vinyl chloride	ND	0.0125	0.009	Inhalation	c	$1x10^{-6}$	YES	0.009	$1x10^{-6}$
Trichloroethene	4.1	32	2.3	Inhalation	c	$1x10^{-6}$	YES	2.3	$1x10^{-6}$
Benzene	ND	0.01	0.72	Inhalation	c	$1x10^{-6}$	NO	NA	
Ethylbenzene	ND	0.094	(2,100)	Inhalation	nc	HI = 1	NO	NA	
Toluene	0.0021	18	(820)	Inhalation	nc	HI = 1	NO	NA	
Xylenes	ND	12	(670)	Inhalation	nc	HI = 1	NO	NA	
1,1,1-Trichloroethane	ND	0.009	1,100	Inhalation	nc	HI = 1	NO	NA	
Tetrachloroethene	ND	0.031	3.9	Inhalation	c	$1x10^{-6}$	NO	NA	
1,2-Dichloroethene (total)	0.038 [f]	1.6	NE					NE	
1,3,5-Trimethylbenzene	ND	0.3	NE					NE	
1,2,4-Trimethylbenzene	ND	0.17	NE					NE	
n-Butylbenzene	ND	0.096	NE					NE	
Mercury (total)	0.0677	0.107	(0.71)	Inhalation	nc	HI = 1	NO	NA	

NOTES:
[a] Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in perimeter wells.
[b] Phase I and pre-Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in interior sample locations.
[c] Lowest risk-based concentration for indoor inhalation commercial/industrial worker presented in Table 11.
[d] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[e] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
[f] Value is the sum of cis and trans isomers.
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
ND = Not detected.
NE = Not Evaluated. Appropriate dose-response data were not available to develop RBCs and, therefore, QGs were not calculated.
nc = non-carcinogenic effects
( ) = calculated RBC exceeds the water solubility limit for this CPC.

TABLE 19
SELECTION OF QUALITY GOALS FOR GROUNDWATER – COMMERCIAL WORKER – OUTDOOR

HUMAN HEALTH ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM PERIMETER WELL CONCENTRATION [a] (mg/L)	MAXIMUM ON-SITE CONCENTRATION [b] (mg/L)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
			Screening RBC [c] (mg/L)	Pathway/Effect		Screening Target Risk [d]	Quality Goal Required? [e]	Quality Goal [e] (mg/L)	Quality Goal Target Risk
Vinyl chloride	ND	0.0125	2	Inhalation	c	$1x10^{-6}$	NO	NA	
Trichloroethene	4.1	32	440	Inhalation	c	$1x10^{-6}$	NO	NA	
Benzene	ND	0.01	130	Inhalation	c	$1x10^{-6}$	NO	NA	
Ethylbenzene	ND	0.094	(400,000)	Inhalation	nc	HI = 1	NO	NA	
Toluene	0.0021	18	(150,000)	Inhalation	nc	HI = 1	NO	NA	
Xylenes	ND	12	(120,000)	Inhalation	nc	HI = 1	NO	NA	
1,1,1-Trichloroethane	ND	0.009	(220,000)	Inhalation	nc	HI = 1	NO	NA	
Tetrachloroethene	ND	0.031	(820)	Inhalation	c	$1x10^{-6}$	NO	NA	
1,2-Dichloroethene (total)	0.038 [f]	1.6	NE					NE	
1,3,5-Trimethylbenzene	ND	0.3	NE					NE	
1,2,4-Trimethylbenzene	ND	0.17	NE					NE	
n-Butylbenzene	ND	0.096	NE					NE	
Mercury (total)	0.0677	0.107	(130)	Inhalation	nc	HI = 1	NO	NA	

NOTES:
[a] Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in perimeter wells.
[b] Phase I and pre-Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in interior sample locations.
[c] Lowest risk-based concentration for outdoor inhalation commercial/industrial worker presented in Table 11.
[d] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[e] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
[f] Value is the sum of cis and trans isomers.
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
ND = Not detected.
NE = Not Evaluated. Appropriate dose-response data were not available to develop RBCs and, therefore, QGs were not calculated.
nc = non-carcinogenic effects
( ) = calculated RBC exceeds the water solubility limit for this CPC.

TABLE 20
SELECTION OF QUALITY GOALS FOR GROUNDWATER – CONSTRUCTION WORKER

HUMAN HEALTH ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM PERIMETER WELL CONCENTRATION [a] (mg/L)	MAXIMUM ON-SITE CONCENTRATION [b] (mg/L)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
			Screening RBC [c] (mg/L)	Pathway/Effect		Screening Target Risk [d]	Quality Goal Required? [e]	Quality Goal [e] (mg/L)	Quality Goal Target Risk
Vinyl chloride	ND	0.0125	49	Inhalation	c	$1x10^{-6}$	NO	NA	
Trichloroethene	4.1	32	(11,000)	Inhalation	c	$1x10^{-6}$	NO	NA	
Benzene	ND	0.01	(3,300)	Inhalation	c	$1x10^{-6}$	NO	NA	
Ethylbenzene	ND	0.094	(400,000)	Inhalation	nc	HI = 1	NO	NA	
Toluene	0.0021	18	(150,000)	Inhalation	nc	HI = 1	NO	NA	
Xylenes	ND	12	(120,000)	Inhalation	nc	HI = 1	NO	NA	
1,1,1-Trichloroethane	ND	0.009	(220,000)	Inhalation	nc	HI = 1	NO	NA	
Tetrachloroethene	ND	0.031	(20,000)	Inhalation	c	$1x10^{-6}$	NO	NA	
1,2-Dichloroethene (total)	0.038 [f]	1.6	NE					NE	
1,3,5-Trimethylbenzene	ND	0.3	NE					NE	
1,2,4-Trimethylbenzene	ND	0.17	NE					NE	
n-Butylbenzene	ND	0.096	NE					NE	
Mercury (total)	0.0677	0.107	(130)	Inhalation	nc	HI = 1	NO	NA	

NOTES:
[a] Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in perimeter wells.
[b] Phase I and pre-Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in interior sample locations.
[c] Lowest risk-based concentration for outdoor inhalation construction worker presented in Table 12.
[d] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[e] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
[f] Value is the sum of cis and trans isomers.
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
ND = Not detected.
NE = Not Evaluated. Appropriate dose-response data were not available to develop RBCs and, therefore, QGs were not calculated.
nc = non-carcinogenic effects
( ) = calculated RBC exceeds the water solubility limit for this CPC.

TABLE 21
SELECTION OF QUALITY GOALS FOR SOIL – UTILITY WORKER

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM CPC CONCENTRATION [a] (mg/kg)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
		Screening RBC [b] (mg/kg)	Pathway/Effect		Screening Target Risk [c]	Quality Goal Required? [d]	Quality Goal [d] (mg/kg)	Quality Goal Target Risk
Trichloroethene	280	8,200	Direct contact	nc	HI = 1	NO	NA	
Toluene	1,800	36,000	Direct contact	nc	HI = 1	NO	NA	
Xylenes	1,830	46,000	Direct contact	nc	HI = 1	NO	NA	
Ethylbenzene	62	24,000	Inhalation	nc	HI = 1	NO	NA	
4-Methyl-2-pentanone	13	11,000	Direct contact	nc	HI = 1	NO	NA	
Tetrachloroethene	28	14,000	Direct contact	nc	HI = 1	NO		
Cadmium	177	4,200	Direct contact	nc	HI = 1	NO	NA	
Chromium	27	5,200	Direct contact	c	$1x10^{-6}$	NO	NA	
Nickel	344	38,000	Direct contact	nc	HI = 1	NO	NA	
Lead	36,000	400 [e]	Direct contact	nc	NA	YES	ND	
Zinc	483	>	Direct contact	nc	HI = 1	NO	NA	
Cyanide	36.5	220,000	Direct contact	nc	HI = 1	NO	NA	
Mercury	12,800	28,000	Direct contact	nc	HI = 1	NO	NA	

NOTES:
[a] Phase I Data (maximum concentrations) from the data set described subsection 2.1.
[b] Lowest risk-based concentration for outdoor inhalation of volatiles or direct contact for utility worker presented in Table 10.
[c] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[d] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
[e] This value is the OSWER cleaup value for lead contaminated soils from "Revised Interim Soil Lead Cleanup Guidance for CERCLA Sites and RCRA Corrective Action Facilities" (OWSER 9355.4-12).
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
ND = Not Developed because sufficient dose-response data were unavailable for developing a site-spefic QG.
c = carcinogenic effects
> = Calculated RBC is greater than 1 million mg/kg.
nc = non-carcinogenic effects

TABLE 22
SELECTION OF QUALITY GOALS FOR GROUNDWATER – UTILITY WORKER

HUMAN HEALTH ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CPC	MAXIMUM PERIMETER WELL CONCENTRATION [a] (mg/L)	MAXIMUM ON-SITE CONCENTRATION [b] (mg/L)	SCREENING RBC					RECOMMENDED QUALITY GOALS	
			Screening RBC [c] (mg/L)	Pathway/Effect		Screening Target Risk [d]	Quality Goal Required? [e]	Quality Goal [e] (mg/L)	Quality Goal Target Risk
Vinyl chloride	ND	0.0125	6,800	Inhalation	c	$1x10^{-6}$	NO	NA	
Trichloroethene	4.1	32	>	Inhalation	c	$1x10^{-6}$	NO	NA	
Benzene	ND	0.01	(450,000)	Inhalation	c	$1x10^{-6}$	NO	NA	
Ethylbenzene	ND	0.094	>	Inhalation	nc	HI = 1	NO	NA	
Toluene	0.0021	18	>	Inhalation	nc	HI = 1	NO	NA	
Xylenes	ND	12	>	Inhalation	nc	HI = 1	NO	NA	
1,1,1-Trichloroethane	ND	0.009	>	Inhalation	nc	HI = 1	NO	NA	
Tetrachloroethene	ND	0.031	>	Inhalation	c	$1x10^{-6}$	NO	NA	
1,2-Dichloroethene (total)	0.038 [f]	1.6	NE					NE	
1,3,5-Trimethylbenzene	ND	0.3	NE					NE	
1,2,4-Trimethylbenzene	ND	0.17	NE					NE	
n-Butylbenzene	ND	0.096	NE					NE	
Mercury (total)	0.0677	0.107	(1,500)	Inhalation	nc	HI = 1	NO	NA	

NOTES:
[a] Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in perimeter wells.
[b] Phase I and pre-Phase I data (maximum concentrations) from off-site laboratory analyses, for analytes detected in interior sample locations.
[c] Lowest risk-based concentration for outdoor inhalation construction worker presented in Table 13.
[d] NYSDEC default target risks presented in "Site Assessment and Closure Guidance for Petroleum Impacted Sites" Division of Spills Management, NYSDEC, September 24, 1995.
[e] Quality Goals are developed only for those CPCs that have a maximum reported concentration that exceeds the selected screening RBC.
[f] Value is the sum of cis and trans isomers.
ELCR = Excess lifetime cancer risk
HI = Hazard Index
NA = Not Applicable because the maximum CPC concentration is lower than the RBC.
ND = Not detected.
NE = Not Evaluated. Appropriate dose-response data were not available to develop RBCs and, therefore, QGs were not calculated.
nc = non-carcinogenic effects
( ) = calculated RBC exceeds the water solubility limit for this CPC.
> Calculated RBC exceeds 1 million mg/L.

TABLE 23
SUMMARY OF SOIL QUALITY GOALS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

LAND USE	SOIL QUALITY GOALS (mg/kg)				
	Receptor / CPC	Commercial Worker - Outside	Commercial Worker - Inside	Utility Worker	Construction Worker
Commercial Development with No Engineering Restrictions	Trichloroethene Tetrachloroethene Lead Mercury	31 6.5 400 [a] 2,500	2.7 19 NA 4,100	Not Applicable	140 (160) 400 [a] 2,500
Commercial Development with Engineering Restrictions	Trichloroethene Tetrachloroethene Lead Mercury	Not Applicable	Not Applicable	Not Applicable	140 (160) 400 [a] 2,500
No Further Development (Current Conditions)	Trichloroethene Tetrachloroethene Lead Mercury	Not Applicable	Not Applicable	(8,200) (14,000) 400 [a] (28,000)	Not Applicable

NOTES:
( ) = Maximum reported site concentration does not exceed screening risk-based concentration; screening risk-based concentration is presented for informational purposes.
[a] = The value presented is the OSWER cleanup value for residential soils (OSWER 9355.4-12). This value is not applicable to non-residential sites and, therefore, is not a Quality Goal for this Site. The value is presented for informational purposes only.

TABLE 24
SUMMARY OF GROUNDWATER QUALITY GOALS

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

LAND USE	GROUNDWATER QUALITY GOALS (mg/L)				
	Receptor / CPC	Commercial Worker - Outside	Commercial Worker - Inside	Utility Worker	Construction Worker
Commercial Development with No Engineering Restrictions	Vinyl Chloride Trichloroethene	(2) (440)	0.009 2.3	Not Applicable	(49) (11,000)
Commercial Development with Engineering Restrictions	Vinyl Chloride Trichloroethene	Not Applicable	Not Applicable	Not Applicable	(49) (11,000)
No Further Development (Current Conditions)	Vinyl Chloride Trichloroethene	Not Applicable	Not Applicable	(6,800) (>)	Not Applicable

NOTES:
( ) = Maximum reported site concentration does not exceed screening risk-based concentration; screening risk-based concentration is presented for informational purposes.
> = Calculated screening risk-based concentration exceeds 1 million mg/L.

APPENDIX A

DEVELOPMENT OF A BIOAVAILABILITY ADJUSTMENT FACTOR FOR MERCURY IN SOILS AT THE AMES STREET SITE ROCHESTER, NEW YORK

PTI
ENVIRONMENTAL SERVICES
4940 Pearl East Circle, Suite 300, Boulder, Colorado 80301

Development of a Bioavailability Adjustment Factor for Mercury in Soils at the Ames Street Site, Rochester, New York

Submitted to

ABB Environmental Services, Inc.
1175 John Street
West Henrietta, New York 14586

PTI Contract CA51-02-04

July 1996

CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACRONYMS AND ABBREVIATIONS

ABB	ABB Environmental Services Inc.
AVS	acid-volatile sulfide
BAF	bioavailability adjustment factor
BEI	backscatter electron image
CAS	Columbia Analytical Services
CVAA	cold vapor atomic absorption spectroscopy
EMPA	electron microprobe analysis
EPA	U.S. Environmental Protection Agency
PTI	PTI Environmental Services
QA/QC	quality assurance and quality control
RfD	reference dose
RPD	relative percent difference
RSD	relative standard deviation
TOC	total organic carbon

EXECUTIVE SUMMARY

PTI Environmental Services was retained by ABB Environmental Services, Inc. (ABB) to derive a bioavailability adjustment factor (BAF) for mercury in soil at the Ames Street site in Rochester, New York. The mercury BAF can be used to adjust soil mercury intake estimates used in the human health risk assessment, to develop revised site-specific soil remediation goals.

Mercury can occur in soils as different chemical species with varying solubility and bioavailability. Several species of mercury found in soils have been shown to be less bioavailable than mercuric chloride dissolved in drinking water, which forms the toxicological basis for the RfD. This difference suggests that the development of risk-based cleanup levels can be adjusted to account for the relative bioavailability of mercury species in soil. Therefore, a site-specific bioavailability study was undertaken to support the development of an alternative cleanup level for mercury in soils at the Ames Street site. The purpose of this study was to accurately characterize the mercury species present in site soils and to determine the solubility, or bioaccessibility, of mercury from soil using a physiologically based extraction test.

For the purpose of this study, bioaccessible mercury is defined as the fraction of mercury that is soluble in the gastrointestinal tract and is available for absorption, while bioavailable mercury is defined as the fraction of mercury that is absorbed into the bloodstream. Because mercury in soil must be solubilized in order to become bioavailable, mercury bioaccessibility is a precursor to, and provides an upper-bound estimate of, mercury bioavailability.

Mineralogical analyses and sequential extraction procedures were used to characterize organic mercury, acid-soluble mercury, elemental mercury, and mercuric sulfide in soils at the Ames Street site. The results from the mercury speciation work were used to support the results of the physiologically based extraction test in developing a mercury BAF. Fourteen soil samples from 10 locations across the site were analyzed to evaluate mercury speciation. Results of the sequential extraction of mercury from the site soil samples suggest that, on average, organic, acid-soluble, elemental, and mercuric sulfide forms of mercury account for 0.3, 21, 63, and 15 percent, respectively, of the mercury found in the samples. Results from the samples analyzed for mercury speciation using mineralogical techniques suggested that elemental mercury is the dominant form of mercury in site samples, with lesser amounts of acid-soluble and mercuric chloride mercury forms. However, analysis of quality assurance samples indicated that the sequential extraction technique underestimated the fraction of acid-soluble mercury in site soils, and the mineralogy technique underestimated the fraction of both acid-soluble and elemental mercury in site soils. Despite the limitations of each method, the sequential extraction and mineralogy data, when evaluated together, suggest that elemental mercury is the dominant mercury phase in the Ames Street soil samples, with lesser amounts of mercuric sulfide and

acid-soluble mercury forms. Because both elemental mercury and mercuric sulfide are mercury forms that have limited bioavailability, these data support a BAF of less than one for the Ames Street site.

Mercury bioavailability in soil was estimated using an *in vitro* test designed to emulate the chemistry and function of a child's gastrointestinal tract. This test determines the soluble, or bioaccessible, fraction of mercury from site soils considering gastrointestinal transit time, pH, and stomach and small-intestine conditions. Results of this study, performed on 10 soil samples from the site, indicate that the bioaccessible fraction of mercury from site soils generally increased in the small intestine over the 4-hour transit time, but remained much lower than that of the mercuric chloride spike. The low bioaccessibility of mercury from these soil samples is consistent with the mercury speciation study results, which found primarily mercury species with very limited solubility.

Based on the *in vitro* results, a BAF of 0.20 was calculated for the site. Because mercury bioaccessibility was observed to increase with decreasing soil mercury concentrations, the BAF was calculated for soil mercury concentrations in the 0–520 mg/kg range. Because the *in vitro* test was designed to produce conservative results, the BAF of 0.20 is expected to substantially overestimate average mercury bioavailability from the site soils.

1. INTRODUCTION

ABB Environmental Services, Inc. (ABB) retained PTI Environmental Services (PTI) to develop a bioavailability adjustment factor (BAF) for mercury in soil at the Ames Street site, Rochester, New York. The Ames Street site operated as an industrial manufacturing facility producing mercury-filled thermometers, barometers, and related instruments from 1904 until approximately 1965. These activities appear to have resulted in the deposition of mercury in site soils. This document describes the procedures used to determine a site-specific soil mercury BAF through the application of mercury speciation and physiologically based extraction studies, and presents the results and conclusions of these studies.

The EPA's RfD for inorganic mercury forms the toxicological basis for generic risk-based soil mercury cleanup levels. This RfD is based on a toxicological study of mercuric chloride that was dissolved in drinking water and administered to rats. However, other mercury species have different water solubility and therefore different bioavailability when ingested. For mercury in soils, oral bioavailability is dependent on the mineral form of mercury present and the physical and chemical characteristics of the soil matrix, which can limit dissolution of mercury in the gastrointestinal tract. Therefore, an alternative risk-based cleanup level that is site specific can be developed using EPA's RfD for inorganic mercury by adjusting the soil mercury intake factor to reflect the bioavailability of mercury in site soils relative to that of mercuric chloride dissolved in drinking water.

Mercury can occur in soil as different chemical species, with varying solubilities. It is assumed that only those mercury species that are dissolved during passage through the human gastrointestinal tract are absorbed. To evaluate this issue, PTI applied a two-phase approach to estimating a BAF for mercury in site soils, which can then be used to derive alternative risk-based cleanup levels. The first phase evaluates speciation of mercury in soil using adaptations of two techniques: sequential extraction and electron microprobe analyses. The second phase estimates site-specific bioavailability of mercury in soil using a physiologically based extraction test (*in vitro* test) that simulates metal dissolution in the human gastrointestinal tract.

For the purpose of this study, bioaccessible mercury is defined as the fraction of mercury that is soluble in the gastrointestinal tract and is available for absorption, while bioavailable mercury is defined as the fraction of mercury that is absorbed into the bloodstream. Because mercury in soil must be solubilized in order to become bioavailable, mercury bioaccessibility is a precursor to, and provides an upper-bound estimate of, mercury bioavailability.

Speciation data were used to indicate the predicted solubility of mercury in soil samples and provided support for the estimated bioavailability of mercury determined from the *in vitro* assay. The *in vitro* assay allowed quantification of the soluble, or bioaccessible,

fraction of mercury in onsite soils that was available for absorption from the gastrointestinal tract. Because mercury must be dissolved in order to be absorbed, the bioaccessible fraction can be considered a conservative upper-bound estimate of the oral bioavailability of mercury in site soils. The BAF for mercury in site soils was calculated as the bioaccessible fraction of mercury from onsite soils relative to the bioaccessible fraction of mercuric chloride dissolved in water. This approach is consistent with that used in previous evaluations of mercury bioavailability (CDM 1992; Barnett and Turner 1995). The results of these studies were used to estimate a BAF for mercury in soil at the Ames Street site.

2. EVALUATING MERCURY SPECIATION IN SOILS

Mercury can occur in soils as elemental mercury in liquid or vapor form, organic mercury compounds, mercuric chloride, or one of several different mineral species, including mercuric oxides, carbonates, and sulfides. In general, organic mercury, mercuric chloride, and elemental mercury in the vapor phase are very soluble and bioavailable; mercuric oxides and carbonates are less soluble; and liquid elemental mercury and mercuric sulfides are essentially insoluble and non-bioavailable after being ingested. The chemical form of mercury controls its mobility in the soil, its bioavailability when ingested, and its response to specific remedial actions. Therefore, an understanding of mercury mineral speciation in soils at the Ames Street site is critical for determining the bioavailability of mercury.

Mercury mineral speciation in soils was evaluated using both sequential extractions and electron microprobe analysis (EMPA).

2.1.1 Sequential Extraction Technique

Recently, several investigators have focused on developing sequential extraction procedures to quantitatively evaluate the speciation of mercury in soils (Revis et al. 1989; Miller 1993; Sakamoto et al. 1992). These procedures use stepwise extractions of a sample, where the individual steps are designed to extract specific mineral phases within a soil sample. Application of the procedures of each investigator to mercury-contaminated soils from Oak Ridge, Tennessee, showed mercury occurring predominantly as elemental mercury and mercuric sulfide minerals (Barnett et al. 1994). However, the relative proportions of the two species did not agree among procedures, indicating that the extractions were either not fully effective in removing specific mercury compounds or not fully specific in extracting individual mercury species. This problem is common to sequential extraction methods (Belzile et al. 1989). All the extraction techniques gave similar levels of organic mercury in soils. However, the method used by Miller (1993), developed by EPA, generally found much less elemental mercury and mercuric sulfide than the other two extraction procedures. The method used by Sakamoto et al. (1992) tended to have poor recovery for elemental mercury. The method used by Revis et al. (1993) produced higher recoveries of mercuric sulfide and elemental mercury, but does not include a procedure for mercuric oxides and carbonates (acid-soluble mercury). Given the drawbacks of all the methods, a procedure combining the most effective aspects of each was developed by PTI to produce the most useful and reliable results.

2.1.2 Electron Microprobe Analysis Technique

Electron microprobe analysis (EMPA) is an x-ray technique that provides direct visual evidence of the mercury mineral species present in soil. By bombarding a sample with a focused beam of electrons, and measuring the resulting x-rays that are emitted by the sample, the elemental composition of individual soil grains within the sample can be determined. In addition, the back-scatter image that reflects off the surface of the sample can be recorded photographically, documenting the morphology and composition of the mercury-bearing grains. The microprobe is used to determine the distribution of the specific mercury-bearing species in the soil and can be used to qualitatively, rather than quantitatively, confirm the visible amounts of these species. This information can then be used to assess the potential bioavailability of the metal in the soil (Davis et al. 1993).

Several limitations of the EMPA procedure must be considered when designing a study and evaluating the resulting data. They include:

- **Ubiquitous, low-concentration dispersion of mercury**—It is not possible to quantitatively account for the entire mass of mercury within a soil sample, because mercury may be distributed throughout the soil at low concentrations that are difficult to quantify (i.e., below the detection limit of the instrument). For example, a sample of clay could contain a detectable bulk mercury concentration, but the mercury concentration within each clay particle might be below the instrument detection limit of 0.5 wt percent.

- **Low bulk mercury concentration**—It is often difficult to detect mercury-bearing species in soils with very low bulk mercury concentrations (generally less than 100 mg/kg). At these low concentrations, it becomes statistically less likely that a mercury-bearing soil particle will be exposed at the surface of the polished sample. In these cases, it may not be possible to evaluate enough particles to be statistically representative of the entire population of mercury-bearing phases within the sample.

- **Mercury phase volatility**—Many mercury-bearing mineral phases are volatile at either room temperature or within the temperature-pressure ranges of the instrument. Most notable of these phases are organic and elemental mercury, which may be partially or wholly volatilized during sample preparation and analysis.

- **Liquid mercury**—Between –39 and 357 °C (–38 and 675 °F), mercury is a liquid with a very high specific gravity (13.5 g/cm^3), which frequently causes mercury to settle within a sample. Therefore, particular care must be used when removing subsamples to ensure that a representative sample is being analyzed. The EMPA sample preparation technique was modified to ensure that mercury would be evenly

dispersed throughout the EMPA sample (the technique is discussed below).

2.1.3 Comparative Study: Sequential Extraction vs. EMPA

To address the limitations described above, PTI conducted a coupled study of mercury mineral speciation, which combined both sequential extractions and EMPA. The purpose of using more than one method was to allow for data cross-checking and validation to ensure the reliability of study results. It was hoped that the combined approach would allow for better quantification of mercury species distribution, especially organic and elemental mercury, in soils. In addition, specific quality control samples were subjected to both procedures to further assess the limitations of both techniques (these samples are described in Sections 2.3.1.6 and 2.4.2).

2.2 SAMPLE COLLECTION AND CHARACTERIZATION

Samples were collected by ABB from the Ames Street site for the mercury speciation and bioavailability study. Fourteen samples were collected from varying depths at 10 locations. Samples to be included in PTI's study were selected by ABB to meet their investigation goals, and included 14 samples for sequential extraction analysis, and a subset of 10 of these samples for EMPA and *in vitro* analysis.

2.2.1 Mercury Vapor Analysis

Prior to performing sequential extractions and EMPA, the concentration of mercury vapor in the headspace of the sample containers was measured. The headspace concentrations were used to establish the relative amount of volatile mercury species in each sample.

To begin the analysis, the soil samples were transferred to 16-oz, amber-glass, wide-mouth septa-jars (I-chem certified 300 series) in the hood and allowed to warm to room temperature. The soil volumes were recorded to facilitate estimation of the headspace within the bottles. To measure the mercury vapor in the headspace, the bottles were agitated gently, and a needle attached to a 60-cm^3 plastic syringe was inserted through the septum lid and pumped twice to homogenize the sample. A total of 60 cm^3 of air was removed and injected into a Jerome® 431X mercury analyzer. To eliminate the effects of vacuum, 60 cm^3 of air was then pumped back into the sample bottles. The bottles were agitated again and placed aside to be reanalyzed once the initial measurement of mercury vapor in the headspace was taken for all the samples. All samples were then analyzed twice more, so a triplicate result could be obtained for each.

2.2.2 Sample Preparation

After measuring mercury vapor in the headspace, approximately 300 g (wet weight) of each soil sample was placed in a stainless-steel bowl in the fume hood to air dry until a constant weight was obtained (approximately 2 days). The air-drying procedure resulted in the loss of some elemental mercury, causing the subsequent speciation analyses to underestimate the fraction of elemental mercury in site soils. The air-dried samples were then weighed, desegregated, and sieved through a number 10, 2-mm stainless-steel sieve. Total mercury concentrations were determined for the 14 samples (<2 mm) by cold vapor atomic absorption spectrometry (CVAA) at Columbia Analytical Services (CAS) in Kelso, Washington (Method 7471, U.S. EPA 1991). The sieved soil was separated to provide 30 g for sequential extractions. A subset of 10 samples were then sieved through a number 60, <250-μm stainless-steel sieve and the resulting sample was split for analysis by EMPA (5 g) and the *in vitro* technique. Total mercury concentrations in the <250-μm samples were determined by CVAA as part of the *in vitro* testing.

In transferring the soil sample from the sampling jars to the stainless steel bowls, beads of elemental mercury were visible in soils from station BS42 (2–4 ft) and BS41 (6–8 ft). The beads were about the size of half a pinhead and were present throughout the sample. The field concentrations for total mercury in these two samples were elevated—1,100 ppm and 3,500 ppm, respectively—consistent with the observation of beads of elemental mercury. Although no other samples contained visible mercury, samples BS70 (0–2 ft) and SS3 (0–2 ft) contained visible pieces of broken thermometers.

2.3 SEQUENTIAL EXTRACTION STUDIES

Mercury speciation was determined using a sequential extraction assay to provide evidence of the various mercury species present in the site soils. The sequential extraction procedure exploits the differing chemical properties of the major categories of mercury-bearing soil phases, to selectively extract these phases during a stepwise chemical extraction procedure. These major phases were:

- Organic mercury
- Acid-soluble mercury, including carbonates, hydroxides, oxides, and chlorides
- Elemental mercury
- Mercuric sulfide.

The sequential extraction procedure for mercury is outlined in Figure 1. Using air-dried soil samples sieved to <2 mm, organic mercury was extracted with chloroform, and then with a sodium thiosulfate solution. The acid-soluble mercury species were then extracted using sulfuric acid. After extracting these two phases, elemental mercury was determined by the difference between a sample split that was roasted at 150 °C for 5 days and a non-roasted sample split. Mercuric sulfide was assumed to be the mercury remaining after roasting.

2.3.1 Methods

2.3.1.1 AVS, Carbonate, and Total Organic Carbon

In addition to sequential extractions, the <2-mm fractions of the 14 soil samples were also submitted for acid-volatile sulfide (AVS), carbonate, and total organic carbon (TOC) analyses. It is particularly important to determine if these phases are present in the site soils, as they often contain coprecipitated or adsorbed mercury (EPRI 1984). In addition, the carbonate content of soils may contribute to the soils' buffering capacity, which could potentially affect the efficiency of the acid-soluble mercury extraction step in the sequential extractions. These analyses were performed at CAS using EPA Method 376.3 for total AVS (U.S. EPA 1991) and ASTM Method D-4239 for TOC and carbonate.

2.3.1.2 Equipment Preparation

Equipment for the organic mercury extraction was prepared by washing four 250-mL glass separatory funnels and 20 50-mL Fisher-brand polypropylene centrifuge tubes in acid. The separatory funnels were then pre-rinsed with Mallinckroft Lot 4440 KJTP chloroform to remove any organic mercury. Once the centrifuge tubes were air-dried, they were tared on a balance, and 7.5 g of a soil sample (less than 2 mm) was weighed into them. Sixteen centrifuge tubes were prepared for the sequential extraction of mercury phases in soil (14 samples plus two duplicates). Extractions were performed in sets of four.

2.3.1.3 Organic Mercury Extraction

The organic mercury extraction began by adding 30 mL of Mallinckroft Lot 4440 KJTP chloroform to four of the centrifuge tubes and sealing them as specified by Sakamoto et al. (1992). The centrifuge tubes were placed in a wrist-action shaker for 3 minutes. Any organic mercury phases within the soil should dissolve into the chloroform during this step. To separate the remaining soil sample from the chloroform, the mixture was centri-

fuged for 3 minutes at 3,000 rpm. The chloroform was then decanted into a 250-mL separatory funnel. To ensure complete extraction of the organic mercury phase, the chloroform extraction was repeated with another 30 mL of chloroform, again decanting the chloroform extract into the separatory funnel. To extract the organic mercury into an aqueous phase for analysis, 10 mL of 0.01 M sodium thiosulfate was then added to the chloroform extracts in each of the separatory funnels. The funnels were hand-shaken for 3 minutes and allowed to settle. The more dense chloroform layer was discharged from the separatory funnel into a large bowl and allowed to evaporate. The remaining sodium thiosulfate solution, which contained any organic mercury that was present in the original soil sample, was collected in 10-mL plastic bottles, preserved with concentrated nitric acid (20 μL per 10 mL of sample), and sent to CAS for total mercury analysis by CVAA. The solid remaining in the centrifuge tubes was placed under the hood to air dry for the acid-soluble extraction step.

Several observations were noted during the organic mercury extraction. The chloroform extracts from samples BS46 (2–4 ft), BS69 (2–4 ft), BS69 (4–6 ft), and BS69 (6–8 ft) were brown, with visible dark brown particles suspended in the chloroform phase. The particles did not settle during centrifugation, indicating that substantial quantities of soil organic matter were extracted by the chloroform. Because of the coloration and suspended particles, the chloroform layer was discharged slowly until all solid particles were contained in the last 0.5 mL of chloroform, which was then discharged directly into the centrifuge (i.e., retained for the acid-soluble extraction step). A cream-colored slurry was present in samples BS42 (2–4 ft), BS41 (4–6 ft), BS42 (4–6 ft), BS46 (2–4 ft), and BS69 (2–4 ft) between the sodium thiosulfate aqueous solution and the chloroform phase, possibly an emulsion of organic compounds and the overlying aqueous phase. Because it was assumed that all of the mercury originally in the chloroform had been oxidized by the sodium thiosulfate solution, and subsequently dissolved into the aqueous phase, this emulsion was not considered an important phase into which mercury would be partitioned. Therefore, the sodium thiosulfate phase was slowly discharged into a 15-mL tube so that the slurry would remain on the sides of the separatory funnels. After the entire sodium thiosulfate layer was collected, the separatory funnels were rinsed with chloroform, to collect the slime and any residual soil particles, and discharged into the centrifuge tubes (i.e., retained for the acid-soluble extraction step).

2.3.1.4 Acid-Soluble Mercury Extraction

The extraction of acid-soluble mercury consisted of adding 15 mL of 0.1 M sulfuric acid to the residual air-dried residue from the organic extraction step (in the centrifuge tubes), shaking the tubes in the wrist-action shaker for 3 minutes, and centrifuging for 10 minutes as specified in Sakamoto et al. (1992). The sulfuric acid solution was then extracted from the centrifuge tube with a 10-mL plastic syringe and filtered through a Corning® disposable sterile syringe filter (25-mm syringe with 0.45-μm acrylic filter with cellulose acetate membrane). The syringe and filter were added to this step because there were minute particles in the hydrochloric acid solution. The hydrochloric acid extract was col-

lected in 15-mL, graduated plastic tubes, and the volume of recovered extract was recorded. The extracts were shipped to CAS for total mercury analysis by CVAA, and the solid sample was air dried in the hood before the next extraction step.

2.3.1.5 Elemental Mercury Extraction

For the elemental mercury determination, a 2.5-g split of air-dried sample from the previous extraction step was sent to CAS for total mercury analysis by CVAA. The mass of remaining sample was determined in tared, stainless-steel pans to the nearest 0.0001 g. Elemental mercury was volatilized from the sample by placing it in a 150 °C oven for 5 days. The mass of the post-roasted samples was determined to the nearest 0.0001 g before transferring the samples to 15-mL plastic tubes to be shipped to CAS for total mercury analysis by CVAA. The mercuric sulfide fraction was assumed to be all the mercury remaining in the sample following roasting. The elemental mercury fraction was determined as the difference between the pre- and post-roasting mercury concentration. The change in sample mass during roasting was compared to the analytically determined elemental mercury content.

2.3.1.6 Quality Assurance and Quality Control Procedures

To assess the specificity and completeness of each extraction step, QA/QC samples consisting of three mercury spikes and a blank were prepared and subjected to the sequential extraction procedure. The blank sample consisted of an empty acid-washed 50-mL polypropylene centrifuge tube. The blank was processed by following the extraction procedures described above without soil. The mercury-spiked samples were prepared using sample BS68 (6–8 ft), which did not contain detectable mercury (detection limit 0.1 mg/kg). A 7.5-g sample of this soil was spiked with 1,140 mg/kg elemental mercury (spike sample 1). This spiked sample was subjected to the first two extraction steps and then sent to CAS for total mercury analysis by CVAA, to assess the quantity of elemental mercury lost during the first two extraction steps. A second elemental mercury spike was then prepared by adding 5,460 mg/kg elemental mercury to approximately 7.5 g of the nondetect soil (spike sample 2). This spiked sample was roasted in the oven at 150 °C for five days and then sent to CAS for total mercury analysis by CVAA. Because this second elemental mercury spike was not subjected to the first two steps, any loss of mercury from this sample could be attributed to the roasting step alone. The third mercury spike was prepared by spiking a 25-g sample of the non-detect soil with 1,470 mg/kg mercury, consisting of 30, 52, and 18 percent mercuric sulfide (HgS), mercuric oxide (HgO), and mercuric chloride ($HgCl_2$), respectively (spike sample 3), resulting in a sample that was 70 percent acid-soluble mercury and 30 percent mercuric sulfide. The spiked sample was split into two subsamples, and each subsample was subjected to the sequential extraction procedure.

2.3.2 Results

2.3.2.1 Headspace and Total Mercury Concentrations

Mercury concentrations resulting from the headspace analyses are presented in Table 1, as individual measurements and as the average of the triplicate analyses. Average headspace mercury concentrations ranged from 0.01 to 22.8 mg/m^3. The limited variability across the triplicate data indicates good precision for the mercury headspace analyses. However, the difference between mercury headspace concentrations in one of the duplicate samples (BS21 [6–8 ft]) suggests that the two splits of this sample do not contain equivalent elemental mercury concentrations. Evaluation of equilibrium air-water partition coefficients for volatile mercury species (Table 2) indicates that elemental mercury is the only mercury species likely to yield a significant fraction of volatile mercury at room temperature. (Note that these partition coefficients do not take into account the effect of the soil matrix present in the sample jars.) Based on the equilibrium partition coefficients, the mercury concentration measured in the headspace should be due almost entirely to elemental mercury.

Total mercury concentrations (<2-mm size fraction) and percent moisture (range of 10–28 percent) for each of the 14 soil samples are also presented in Table 1. A comparison of total mercury in the <2-mm and <250-μm soil size fractions (Table 3) indicates that mercury concentrations in these two size fractions were generally similar; relative percent differences (RPDs) ranged from 3 to 69 percent. These RPD values most likely reflect sample heterogeneity, particularly with respect to elemental mercury, rather than actual differences between mercury concentrations in these two size fractions. Elemental mercury may occur as discrete beads—or nuggets—within a soil sample, causing difficulty in obtaining a homogeneous subsample for analysis of mercury concentration when the analytical laboratory obtains the 0.2-g subsample for analysis, as per Method 7471 (U.S. EPA 1991). This observation of sample heterogeneity is consistent with previous projects performed by PTI, where a nugget effect was observed for elemental mercury in soil. Results from the AVS, carbonate, and TOC analyses on the 14 samples are presented in Table 4.

2.3.2.2 Sequential Extraction Method Validation

For the soil spiked with 1,140 mg/kg mercury as elemental mercury (spike sample 1), 0.001 and 0.003 percent of the elemental mercury spike was extracted during the organic mercury and acid-soluble extraction steps, respectively, indicating that these extracts do not readily dissolve elemental mercury in the presence of the site soils (Table 5). Following these first two extraction steps, the sample was submitted for total mercury analysis. The results of this analysis indicated that the mercury concentration in this

sample increased by 151 percent (from 1,140 to 1,730 mg/kg) following this procedure (i.e., more mercury was recovered than was thought to be in this sample, based on the amount of elemental mercury that was added to the sample, and previous measurements of the mercury content of the unspiked sample).

This increase could be due to contamination from the apparatus, analytical error, or heterogeneities in the soil used to prepare this sample. Considering the minimal amounts of mercury present in the organic and acid-soluble extracts (Table 5), it is unlikely that contamination from the apparatus was the source of the 590-mg/kg increase observed in the elemental mercury-spiked sample. To determine if analytical error could account for the high spike recovery, the results of a triplicate analysis of a single site sample (<2 mm size fraction) were evaluated (data presented in Table 1). These analyses yielded results of 194, 312, and 268 mg/L (relative standard deviation of 23 percent), indicating that the analytical error may be the source of the observed increase. However, if sample heterogeneity is the source of the error, this would affect the performance of laboratory triplicate analysis, and it would be difficult to distinguish between these two possible sources of error. Because mercury often occurs as a liquid in soils, it is able to flow within the soil, often forming discrete beads, or nuggets, of mercury. This behavior leads to a heterogeneous distribution of mercury within the soil matrix. Consequently, mercury analyses in soils are often difficult to reproduce. Given that the internal laboratory QA/QC results were generally within control (see Appendix A), sample heterogeneity is the most likely explanation for the observed increase in mercury concentration in elemental mercury spike sample 1 (Table 5).

In addition to the elemental mercury spike sample described above, a second elemental mercury spike sample (spike sample 2) was prepared with the same mercury-free sample. This spiked sample was subjected to the roasting step only, resulting in a loss of 99.9 percent of the 5,460 mg/kg mercury spiked into the sample (Table 5). As it was the goal of this roasting step to drive off the elemental mercury, these data demonstrate that this step of the extraction procedure effectively removed elemental mercury from the sample.

Two additional standard samples were prepared by spiking the same mercury-free soil described above with 1,470 mg/kg mercury, where 70 percent of the mercury was added as acid-soluble mercury (HgO and $HgCl_2$) and 30 percent as mercuric sulfide (spike sample 3, Table 6). This sample was subjected to the sequential extraction procedure in duplicate. Thus, the known apportionment among the organic, acid-soluble, elemental, and mercuric sulfide fractions was 0, 70, 0, and 30 percent, respectively, of the total mercury mass. However, the sequential extraction of mercury among the fractions was 2, 7, 28, and 37 percent, respectively (Table 6).

Overall, only 74 percent of the mercury that was spiked into this sample was accounted for. Given the consistent results observed for the two replicate samples, the inability of the procedure to accurately apportion mercury among the known mercury phases may indicate that an unidentifiable systematic error is occurring during the extraction proce-

dure, the effect of which would be to potentially overestimate the proportion of mercury attributed to the elemental mercury fraction and to underestimate the acid-soluble fraction.

The inability of the procedure to extract spiked minerals during the appropriate extraction step was likely due to the high carbonate content of the site soils, which ranged between 0.88 and 13.9 weight percent (Table 4). The high alkalinity associated with the high carbonate content would likely buffer the acid-soluble extraction step, resulting in a higher pH in the extract, and a lower extraction efficiency. This conclusion is supported by a comparison between the carbonate content of the site soils and the concentration of mercury in the acid-soluble extract (Figure 2). Acid-soluble mercury concentrations above 30 mg/L in the acid-soluble extracts occurred only in those soils with carbonate contents below 4 percent.

To determine if a low-carbonate-content soil spiked with the same mineral phases would perform more predictably, a PTI internal standard soil, which contained 0.005 percent carbonate, was spiked following the same protocol described above. The results of this sequential extraction of mercury among the organic, acid-soluble, elemental, and mercuric sulfide fractions accurately determined all phases except elemental mercury, which was overestimated by 29 percent (Table 7). These data indicate that while the sequential extraction procedure yielded accurate estimates of the acid-soluble fraction when soil alkalinity was low, the high concentration of carbonate in the site soils likely resulted in a reduced acid-extractable fraction efficiency. Such a reduction would result in an underestimation of the amount of acid-extractable or soluble mercury species present in the soils.

In summary, the sequential extraction method validation results indicate that mercuric sulfide in the Ames Street site soils will be extracted during the appropriate extraction phase, but that acid-soluble forms of mercury (HgO and $HgCl_2$) will not be fully extracted during the acid extraction, and elemental mercury may be over-recovered, resulting in an underestimation and overestimation of the contribution of these mercury forms, respectively, in site soils.

2.3.3 Site Sample Results

Because the sequential extraction results are questionable for samples with elevated carbonate concentrations, only those results from site soils with less than 5 percent carbonate (Table 4) were used in evaluating the sequential extraction results (Table 8, data for samples with <5 percent carbonate in bold). Of the 8 soil samples with less than 5 percent carbonate subjected to the soil extraction procedure, none was found to have any appreciable amounts of mercury within the organic mercury fraction (average of 0.3 percent, Table 8). The 8 samples with total carbonate <5 percent tended to have greater fractions of acid-soluble mercury, with a maximum apportionment of 65 percent in sample BS69 (2–4 ft) (Table 8). The average acid-soluble fraction of mercury was 21 percent. The fraction of elemental mercury ranged from 17 to 99 percent (average of 63 percent) in

these 8 samples, and the fraction of mercuric sulfide ranged from 1 to 70 percent (average of 15 percent). In combination, the elemental mercury and mercuric sulfide accounted for an average of 78 percent of the mercury.

Identification of elemental mercury as a major mercury-bearing phase in these samples is consistent with the headspace analysis described above. The concentration of elemental mercury detected in the samples by sequential extraction was well correlated with the mercury concentration detected in the headspace of the sample (Figure 3, $r^2 = 0.72$). As discussed in Section 2.3.2.1, this observation is consistent with the relative volatility of the various mercury forms present in the Ames Street soils. These data are also in accord with previous site data, which indicate that the amount of mercury in headspace readings is proportional to the amount of visible elemental mercury in a sample, and provide further evidence that elemental mercury is a major mercury-bearing phase in these samples.

The presence of volatile mercury in the sample headspace analysis indicates that elemental mercury will have been lost during the air-drying step of sample preparation. This will have resulted in an underestimation of elemental mercury in the speciation analyses relative to the other (non-volatile) mercury species. However, because elemental mercury is a nonbioavailable mercury species, this underestimation provides an additional level of conservatism to the project results.

To determine if there was any obvious variation in sample mineralogy as a function of mercury concentration, the amount of mercury attributed to each mineral fraction in the 8 samples with <5 percent carbonate was plotted as a function of total mercury concentration (Figure 4). These data indicate that the fraction of elemental mercury increases as total mercury concentrations increase ($r^2 = 0.832$). As a result, these data suggest that mercury bioavailability should decrease at higher total mercury concentrations, due to the presence of more elemental mercury.

By summing the quantity of mercury that was released during the sequential extraction of the 14 soils tested, and comparing it to the total concentration of mercury in the <2-mm soil fraction, an assessment of the extraction efficiency can be made (Table 9). For 10 of the 14 samples tested, less mercury was recovered during the sequential extraction procedure than was determined to be in the sample through direct analysis of a <2-mm fraction of the sample (Table 9). In three of these samples, less than half of the total soil mercury was accounted for. This loss could be due to volatilization of mercury during the extraction procedure, sorption of mercury to the extraction apparatus, or sample heterogeneity. Insufficient data exist to distinguish definitively between these possible explanations. In the remaining four samples, between 6 and 124 percent additional mercury was extracted during the sequential extraction procedure, compared to the amount expected from the total mercury concentration data. As discussed in Section 2.3.2.2, the most likely explanation for these differences between total and extracted mercury concentrations is the heterogeneous nature of soils containing elemental mercury, and the difficulty in obtaining a representative subsample.

2.3.3.1 AVS, Carbonate, and Total Organic Carbon Results

Although no acid-volatile sulfide was detected (detection limit of 5 mg/kg) in any of the samples tested (Table 4), mercuric sulfide was detected consistently in the samples using the sequential extraction technique (Table 8). If the mercuric sulfide were present in the samples as a freshly precipitated, amorphous mercuric sulfide, it is likely that this mineral would be identified as a volatile sulfide (Di Toro et al. 1992). Alternatively, well-crystallized mercuric sulfide minerals (e.g., cinnabar) would not be dissolved by the acid-volatile sulfide (AVS) extraction procedure, which consists of leaching of the soil samples by a 1 M HCl solution (Di Toro et al. 1992). To determine if a sufficient quantity of mercuric sulfide was extracted during the sequential extraction procedures to be detected by the AVS procedure, the concentrations of mercury detected in each of the mercuric sulfide extracts (Table 10) were used to calculate an equimolar concentration of sulfide (assuming that the mercury sulfide mineral extracted was HgS). These data indicate that 4 of the 13 samples tested should have contained sufficient sulfide in the mercuric sulfide extract to be detected during the AVS procedure. Because no AVS was detected, it may be concluded that the majority of the mercuric sulfide occurs as relatively well-crystallized, and consequently relatively non-soluble, mercuric sulfide.

The total organic carbon content of the soils ranged between <0.05 and 6.00 percent. If organic mercury was a predominant mercury-bearing phase in these soils, it is likely that the total mercury content of soils would be correlated with the TOC content. However, the data do not suggest this (Figure 5). These data, together with the sequential extraction results, indicate that organic mercury is not a major mercury-bearing phase in the site soils.

As discussed in Section 2.3.2.2, the total carbonate concentrations for the site soils were relatively high, ranging between 0.88 and 13.2 weight percent (Table 4), with a geometric mean carbonate content of 6 percent. Although no site-specific data are available, it is reasonable to assume that the site soils are likely to be buffered by the carbonate, resulting in neutral soil pHs. Under these conditions, the major mercury-bearing phases identified by sequential extractions, elemental mercury and mercuric sulfide, are more likely to be soluble than at lower soil pHs (EPRI 1984).

2.4 ELECTRON MICROPROBE STUDIES

2.4.1 Methods

Polished sample pucks were prepared at the Laboratory for Geological Studies, University of Colorado, Boulder, for electron microprobe analysis by embedding 4 g of air-dried sample in epoxy within a sample mold to cure at room temperature. To minimize settling

of the liquid mercury within the epoxy substrate during curing, the epoxy was allowed to partially cure prior to addition of the sample. This resulted in a higher epoxy viscosity, which has been shown to eliminate the segregation of elemental mercury during curing (Drexler 1996). To minimize any loss of elemental mercury that may occur during polishing, a one-step sample-polishing step was employed, consisting of a 600-grit wet/dry abrasive paper stretched across a glass plate. All polishing was performed with the polished surface pointing up, thus minimizing the likelihood that beads of mercury would drop out of the sample when exposed. All polishing utilized water to avoid dissolution of any organic mercury species. Finally, sample pucks were placed in a carbon coater, where a thin layer of carbon was sputtered onto the surface of each puck.

Electron microprobe analysis was conducted on a JEOL® 8600 electron microprobe operating at 15 kV with a 20-nA specimen current and a 1-mm beam, according to the methods described in the work plan, as adapted for mercury mineral speciation. Quantitative mineralogic data were collected using wavelength-dispersive spectrometers and mineral standards and corrected using Phi Rho Z parameters. The mercury-bearing particles were identified using a combination of energy dispersive detection, wavelength-dispersive detection, and backscatter electron image (BEI) detection devices.

Initially, spectra were generated for each grain that allowed identification of all elements with an atomic mass greater than or equal to that of carbon. Subsequently, the elemental proportions were quantified by comparison with standard materials, and the mineral proportions were identified based on the equivalent molecular weight of the oxide. Therefore, the identifications provide quantitative stoichiometric ratios from which the mineral identity can be calculated. The relations among mercury-bearing species were established from BEI images and wavelength-dispersive/energy-dispersive analyses as necessary.

Individual mercury-bearing particles were analyzed (representing one point count each) until a minimum of 100 particles were evaluated, or 5 hours of machine time had been spent on the analysis. Point counts were made by traversing each sample from left to right and top to bottom in a grid pattern, with each vertical displacement moving only to the adjacent field of view. Magnification settings of 40× to 100× and 300× to 600× were used; the latter magnifications allowed for the analysis of the smallest identifiable (1–2 μm) mineral grains. The grain size of each mercury carrier was determined by measuring the dimension of the long axis, assuming that the sample area was proportional to its long axis. Percent composition of mercury phases in each sample was determined by summing the total area of all mercury grains and dividing the area for each phase by the total area.

2.4.2 Quality Assurance Sample

Analyses of elemental mercury in soils are often difficult to reproduce, because elemental mercury occurs as a dense liquid, often in isolated beads within a soil sample. This dense

liquid often flows within a sample, pooling in some spots and leaving other areas devoid of mercury. In addition, the vapor pressure of liquid mercury is very close to the operating pressure of the EMPA (10^{-8} Torr), increasing the potential for elemental mercury loss during sample analysis. To assess the accuracy with which the EMPA apportioned mercury between three major mercury phases, two quality control samples were prepared and analyzed.

The quality control samples consisted of a mercury-free soil (BS-68, 6–8 ft) spiked with mercury as elemental mercury, mercuric chloride, and mercuric sulfide at the concentrations indicated in Table 11. Given this composition, the EMPA results should have indicated that the mercury mass was distributed relatively evenly among each of the three phases. However, the results indicated that 87–93 percent of the mercury was associated with the mercuric sulfide phase, 5–11 percent was identified as elemental mercury, and 0–2 percent as mercuric chloride (Table 11). These results indicate that the elemental mercury and the mercuric chloride were lost from the sample, either during sample preparation or analysis. The difficulties in preparing samples and analyzing them for mercury using this method are discussed below.

Prior to analysis, the soil sample is prepared by placing a split of the sample into a photographic film canister, adding an epoxy resin, and then homogenizing the sample/epoxy mixture. Because elemental mercury occurs as a dense liquid, which would flow to the bottom of uncured epoxy, partially cured epoxy was used to minimize this effect (Drexler, personal communication, 1996). However, in the event that the epoxy did not cure sufficiently, the elemental mercury may have settled through the bottom of the epoxy-sample slurry, thus making this mercury phase unavailable for measurement by EMPA. Furthermore, during the EMPA, it is necessary to reduce the pressure surrounding the samples to 10^{-8} Torr, which is below the vapor pressure of elemental mercury (10^{-3} Torr) and mercuric chloride (10^{-5} Torr) at 25 °C (CRC 1985). Because the operating pressure of the electron microprobe is lower than that of these two phases, it is possible that they are volatilizing within the sample chamber. Lastly, when the electron beam of the instrument strikes epoxy-mounted sample material, the surface of the puck heats up, resulting in an increase in the vapor pressure of the minerals contained within the sample. All of these factors may have contributed to the poor spike recoveries for this sample.

2.4.3 Results

The purpose of the EMPA analyses was to quantify the distribution of mercury mass among the various mercury-bearing mineral phases in 10 soil samples from the site. The results from the spiked samples, discussed in Section 2.4.2, indicated that potential problems may be associated with measuring mercury mineralogy using this method. The results for the site samples are discussed below, and conclusions are drawn within the limitations of the quality control results.

The frequency of occurrence of each mercury-bearing mineral phase was calculated by summing the diameter of all the particles of that mineral phase, and dividing by the summed diameters of all mercury-bearing particles encountered within a sample. Results from this calculation are presented in Table 12. By multiplying the frequency distribution by the specific gravity and mercury content of each mineral phase (values provided in Table 13), the relative mercury mass distribution among the mercury-bearing minerals observed in a sample can be calculated (Table 14). Of the ten samples submitted for EMPA, three contained an insufficient quantity of mercury to be detected by the EMPA method. These included BS-21 6–8 ft, BS-69 6–8 ft, and BS-74 0–2 ft, which contained 6.5, 17.3, and 2.4 mg/kg total mercury (in the <250-μm fraction), respectively.

The mineralogy of the remaining seven samples varied with elemental mercury, mercuric chloride, and mercuric sulfide each appearing to be the predominant mineral phase in various samples (Table 14). However, given the results from the quality control sample, both mercuric chloride and elemental mercury may have been lost during the analysis, resulting in an under-estimation of the amount of these two phases, and a resulting over-estimation of mercuric sulfide. Because the vapor pressure of elemental mercury (10^{-3} Torr) is higher than that of mercuric chloride (10^{-5} Torr), elemental mercury would be lost more quickly than mercuric chloride during analysis in the electron microprobe. Even though the quality control results indicate that elemental mercury is lost during analysis, this phase was identified as being the dominant mercury mineral phase in three of the seven samples, indicating that so much elemental mercury was present in these samples, that despite volatilization during the EMPA elemental mercury was still present as a major mercury phase.

To confirm the EMPA results where elemental mercury was identified as the predominant mercury-bearing phase, two samples were subjected to a gravimetric separation technique (the gold panning method) to isolate elemental mercury. To determine if the gravimetric separation technique was an efficient method of isolating mercury, 100 g of a site soil containing relatively little mercury (BS-75 0–2 ft, containing 1.5 mg/kg mercury prior to spiking) was spiked with an additional 0.6492 g of elemental mercury (Table 15). The spiked sample was panned by an experienced technician, and the separated elemental mercury was transferred to a tared bottle to determine its mass. The gravimetric separation technique recovered 92 percent of the total mercury in the sample, indicating that this method is a viable approach to determining the elemental mercury content of a sample. Subsequently, a portion (100 g) of the bulk fraction (i.e., unsieved) of soil sample BS-41 6–8 ft, was evaluated by this technique for recovery of elemental mercury. The elemental mercury isolated from this sample was equivalent to 5,884 mg/kg elemental mercury (Table 15). The concentrations of elemental mercury in the <250-μm and <2-mm fractions of this sample were 5,850 and 12,000 mg/kg, respectively, indicating that one-half to all (49–101 percent) of the total mercury was recovered as elemental mercury. The discrepancy between mercury concentrations in the <2-mm and <250-μm particle size fractions of a given sample most likely result from sample heterogeneity, as discussed previously. The gravimetric separation requires a minimum of 100 g of sample material,

and the small sample sizes available precluded the application of this method to other site samples.

2.5 MERCURY SPECIATION CONCLUSIONS

The speciation results suggest that mercury in soil at the Ames Street site occurs primarily as elemental mercury, with lesser amounts of both mercuric sulfide and acid-soluble mercury forms. The sequential extraction results suggest that, on average, elemental mercury, mercuric sulfide, and acid-soluble forms of mercury accounted for 63, 15, and 21 percent, respectively, of mercury in the 8 samples for which reliable data were obtained. None of samples was found to contain any appreciable amounts of mercury in the organic mercury fraction (average of 0.3 percent). The results from the headspace analyses for mercury, and comparison of these data to the results of the sequential extractions, also suggest that elemental mercury is a dominant form of mercury in these samples. In addition, the air drying of samples prior to the speciation analyses invariably resulted in the loss of some elemental mercury, resulting in an underestimation of the fraction of elemental mercury in site soils. However, based on the results from the sequential extraction method validation samples, the sequential extraction technique tends to underestimate the fraction of acid-soluble mercury and overestimate the fraction of elemental mercury associated with soil from the Ames Street site. Therefore, the fraction of elemental mercury observed in the sequential extractions is likely an upper estimate.

The EMPA data also suggest that elemental mercury is the dominant phase in Ames Street soils, followed by mercuric sulfide and mercuric chloride. These data are generally consistent with the results of the sequential extraction analyses. However, it should be noted that the quality assurance sample results for EMPA indicate that this technique tends to underestimate the fraction of elemental mercury and mercuric chloride in soil samples. The gravimetric separation technique appears to provide a reliable measurement of elemental mercury mass, and indicated that one of the samples with elevated mercury concentration was composed largely of elemental mercury.

Despite their respective limitations, the sequential extraction and EMPA data, when evaluated together suggest the elemental mercury is the dominant mercury phase in the Ames Street soil samples, with lesser amounts of mercuric sulfide and acid-soluble mercury forms. Because both elemental mercury and mercuric sulfide are mercury forms that have limited bioavailability, these data support a BAF of less than unity for the Ames Street site.

3. *IN VITRO* BIOAVAILABILITY TESTING

In humans, an orally administered dose of a compound is seldom completely absorbed, and differences in the extent of absorption of orally administered compounds exist among different exposure media. For most compounds, the toxicity values derived by EPA are not adjusted to the absorbed dose (i.e., the dose-response evaluation is based on the administered dose). This procedure can lead to errors in assessing the risks of exposure to a particular chemical in a medium other than the one used in the toxicity or epidemiology studies on which the toxicity values are based. For example, the EPA oral toxicity value (or reference dose [RfD]) for inorganic mercury was derived from studies in which mercuric chloride dissolved in water was administered to laboratory animals. Most of the mercury at the Ames Street site is present in forms that are less soluble than mercuric chloride. Because absorption decreases with decreasing solubility, absorption of mercury from ingested site soils will be reduced compared to mercuric chloride. If these differences in mercury bioavailability are not accounted for, risks associated with ingestion of mercury in site soils will be overestimated. The bioavailability adjustment factor to correct for differences in absorption from different exposure media is termed the BAF. This fractional value is used to adjust the dose or intake so that it is expressed in the same terms as the doses used to generate the toxicity values.

Substantial evidence exists that mercury solubility and bioavailability vary with mercury species. Studies in rodents suggest that 10–20 percent of dissolved mercuric chloride is absorbed from single oral doses (Nielsen 1992). Several studies comparing tissue levels in rodents after single or repeated doses of mercuric chloride and mercuric sulfide have concluded that absorption of mercuric sulfide is much lower than absorption of mercuric chloride (Sin et al. 1983, 1992; Yeoh 1989). In 1993, EPA reviewed available studies on the toxicity and bioavailability of mercuric sulfide in response to a petition for a provisional mercuric sulfide RfD for the East Fork Poplar Creek site in Oak Ridge, Tennessee. At that time, EPA concluded that insufficient information was available to derive a separate RfD for mercuric sulfide, but they did note that comparison of relative tissue levels of mercury in animal studies suggested that mercuric sulfide was 30–80 times less bioavailable than mercuric chloride (DOE 1994). Thus, a relative BAF of 1/30 to 1/80 (0.03–0.01) may be appropriate when applying toxicity values for mercuric chloride to mercuric sulfide. Several studies also indicate that elemental mercury is poorly absorbed in the human gastrointestinal tract (Goyer 1996; Berlin 1979). Experiments on rats indicate that absorption may be as small as 0.01 percent of an oral exposure of elemental mercury (Bornmann et al. 1970, as cited in Friberg and Nordberg 1973). Little or no information is available on the oral absorption of other mercury compounds (e.g., mercuric carbonate, mercuric oxide) relative to mercuric chloride; however, other mercury species are also likely to be less bioavailable than mercuric chloride. The bioavailability of mercury species in soil also may be reduced because of interactions with soil constituents.

Thus, site-specific BAFs will vary, depending on the mix of mercury species present at the site and the composition of other soil constituents. Because a variety of mercury species are present in soils at the Ames Street site, site-specific mercury BAFs were determined based on a study of site soil samples.

For the purpose of this study, bioaccessible mercury is defined as the fraction of mercury that is soluble in the gastrointestinal tract and is available for absorption, while bioavailable mercury is defined as the fraction of mercury that is absorbed into the bloodstream. Because mercury in soil must be solubilized in order to become bioavailable, mercury bioaccessibility is a precursor to, and provides an upper-bound estimate of, mercury bioavailability.

To evaluate mercury bioaccessibility, an *in vitro* test was used (Ruby et al. 1996), which had previously been used to assay the bioavailability of lead and arsenic in soils. Experimental results from the application of this test to the evaluation of lead and arsenic bioaccessibility indicate that dissolution of arsenic and lead from soil is limited and that the *in vitro* test provides a useful, rapid screening-level estimate of maximum available arsenic and lead from soil relative to animal models (Ruby et al. 1993, 1996). For this study, the *in vitro* test was modified to provide a system appropriate for mercury bioaccessibility evaluation.

In vitro assays similar to the PTI test have been employed at several other sites to estimate site-specific bioavailability of mercury in soil. At the Almaden Quick Silver County Park in Los Gatos, California, the form of mercury present in site soils, which resulted from mining and ore processing (predominantly mercuric sulfide), was experimentally measured to be from 0.03 to 9.4 percent as soluble as mercuric chloride in a simulated gastrointestinal environment (CDM 1992). The Los Gatos site samples were tested using a leaching procedure designed to simulate the human gastrointestinal system: 200 mg of sample (sieved to less than 2 mm) was added to 480 mL of a pH-2.5 solution of dilute hydrochloric acid in 500-mL bottles, and the bottles were agitated for 4 hours to simulate conditions in the human stomach. The human intestine was simulated by adjusting the pH of the solution to 6.5 using sodium hydroxide, and agitating for an additional 4 hours. At the end of the simulated stomach and intestinal phases, aliquots of the solutions were filtered (0.45 μm) and analyzed for their mercury content. Based on the results of this *in vitro* assay, the Santa Clara County Parks and Recreation Department and California State regulatory authorities agreed to use a BAF of 0.3 for mercury in soils at the Los Gatos site.

An *in vitro* procedure nearly identical to the one described above was used to evaluate the solubility of mercury in soil samples collected at Oak Ridge National Laboratory in Tennessee (Barnett and Turner 1995). The experimental procedure was altered in that the soil samples were pulverized after sieving, and only the <180-μm size fraction was subjected to the leaching procedure. For 19 of the 20 samples, the mercury in soils was determined to be from 0.3 to 14.2 percent soluble (average of 3.2 percent). One sample, the only one with detectable mercury vapor in the sample headspace, contained 45.9 percent soluble

mercury by this *in vitro* method. Mercuric chloride was determined to be 100 percent soluble in the *in vitro* test system. Based on these analyses, EPA accepted a site-specific BAF of 0.1 for mercury in soils at the Oak Ridge site (DOE 1995).

For the Ames Street site samples, PTI performed bioaccessibility testing on soil samples that previously underwent mercury speciation analyses. It should be noted that the presence of elevated carbonate concentrations in the Ames Street soils, which caused difficulties during the sequential extraction analyses, will not affect the *in vitro* extractions, because pH values in the *in vitro* assay were monitored and maintained at a constant pH value. As described in the work plan, the soil samples were sieved to the <250-μm size fraction prior to the *in vitro* assay, because particles in this size range are most likely to adhere to children's hands and be ingested (Duggan and Inskip 1985). In addition, samples were extracted for one-hour under the conditions of the stomach phase to represent the gastric empyting time of a child.

In the following sections, the methods and results of the *in vitro* testing on site soils are presented, followed by those of soluble mercuric chloride spikes, which were evaluated in the *in vitro* assay because mercuric chloride is the basis for EPA's RfD for mercury. These results are followed by calculation of a site-specific mercury BAF, a discussion of the conservative aspects associated with the *in vitro* testing, and the study conclusions.

3.1 METHODS

The *in vitro* test is designed to determine the fraction of mercury that is solubilized and available for absorption in the gastrointestinal tract. The method is implemented in two phases, simulating the passage of ingested soil from the acidic environment of the stomach to the near-neutral conditions of the small intestine. Because of the concern for potential loss of volatile mercury from the reaction vessels, the experiment was performed in sealed containers. Argon gas was introduced into the reaction vessels at the beginning of the *in vitro* assay to purge them of atmospheric oxygen, mimicking the anoxic conditions present in the gastrointestinal tract. An activated carbon trap was placed on the inflowing argon gas to remove mercury potentially present in the inflowing gas.

The *in vitro* test methods are summarized below, and described in more detail in the work plan (ABB 1996). Several minor changes from the work plan are noted and discussed in Appendix A. The deviations are sufficiently minor that they did not appear to affect the outcome of the tests.

The *in vitro* extraction tests were performed by the following method. The stomach solution was prepared by adding the following compounds (all chemicals from Sigma Chemical Company, unless otherwise noted) to 1 L of deionized water (stirred continually on a stir plate):

1.25 g pepsin (50 mg, activity of 800–2,500 units/mg)

0.50 g citrate (Fisher Chemical Co.)
0.50 g malate (Aldrich Chemical Co.)
420 mL lactic acid (synthetic syrup 85 percent w/w)
500 mL acetic acid (97 percent w/w; Fisher Chemical Co.).

The pH of the stomach solution was adjusted to 2.5 by adding a measured volume of concentrated hydrochloric acid; 150 mL of the stomach solution was added to a 200-mL acrylic reaction vessel. The stomach solution was sparged with argon for 15 minutes to remove oxygen. At the end of the sparging procedure, the Eh of the stomach solution was measured and recorded, 1.5 g of soil sample was added, and the reaction vessel was sealed. The reaction vessel was submerged approximately half-way into a temperature-controlled water bath, heated to maintain a constant 37 °C in the reaction vessel. The soil/stomach solution was allowed to stand (no agitation) for 10 minutes, and then stirred with a plastic propeller stir rod mounted in a rheostat-controlled motor (Arrow Engineering Model 1750 motor on a rheostat setting of 2, resulting in approximately 150 rpm for the stir rod).

The pH was checked at 5-minute intervals and readjusted to pH 2.5 with hydrochloric acid, if necessary. Five-mL samples of the solution were collected at 30 and 60 minutes, using a stainless-steel hypodermic syringe to pierce the sampling septum. The 5-mL samples were centrifuged at approximately 2500 xg for 25 minutes, and the supernatant was decanted for analysis. After collecting the 60-minute sample, the solution was titrated to pH 7.0 (±0.2) by adding a 5-in. length of dialysis tubing containing approximately 2 g of $NaHCO_3$ to each reaction vessel.

The dialysis tubing was added to the solution by opening the vessels under an exhaust hood. While this procedure momentarily exposed the sample to atmospheric oxygen, it was decided that unsealing the reaction vessels would not adversely affect the test results, because only a small fraction of the mercury introduced into the system was measured as mercury vapor in the headspace of the reaction vessel. The dialysis tubes were added and the vessels re-sealed as quickly as possible to minimize contact of the reaction solution with atmospheric oxygen. The vessels were returned to the water bath, and the mechanical stirring was resumed. After titrating to pH 7.0, the dialysis tubing was removed by the inverse process. Before resealing the containers, 260 mg of bile salts and 75 mg of pancreatin was added to the solution, after having been dissolved in 10 mL of deionized water.

One hour after the reaction fluid reached equilibrium at pH 7.0, a 5-mL sample of the intestinal-phase solution was collected through the septum, using a stainless-steel hypodermic syringe. The samples were centrifuged and the supernatant decanted for analysis, as described above. Three hours after the equilibration to pH 7.0, the Eh and pH of the reaction fluid were measured and recorded, and the second (and final) intestinal-phase sample was collected. The final sample consisted of the entire volume of fluid remaining in the reaction vessel. This sample was collected by measuring the volume of contents from each reaction vessel in a graduated cylinder and pouring them into three centrifuge

tubes. (Three 50-mL centrifuge tubes were needed for each sample, because the volume of fluid remaining in each reaction vessel approached 150 mL.) The tubes were centrifuged, and the fluid contents were decanted and combined in the appropriate sample container for analysis. For the samples undergoing a mass balance evaluation (i.e., the triplicate analysis of sample BS69), the soil was collected from the centrifuge tubes after decanting, to analyze the post-extraction soil for mercury remaining in the sample.

Mass balance experiments were performed on only one site soil sample (analyzed in triplicate), rather than the two samples specified in the protocol. This modification was implemented because method development work, completed previously on other samples, indicated little potential for mercury to be lost from the test system by volatilizing, or adhering to the reaction vessel walls.

The *in vitro* samples of soil and extracts were shipped under chain of custody to Columbia Analytical Services (Kelso, Washington) for analysis, as described in the work plan. However, a modification was made to the sample preservation technique specified in the CVAA analytical procedure used for the aqueous samples (U.S. EPA 1991, Method 7470A). The *in vitro* small-intestinal-phase samples were not preserved with nitric acid before analysis, as this would cause the precipitation of proteins (along with the potential for loss of mercury). Because of the nature of the *in vitro* extraction process, the stomach-phase samples were already acidified with hydrochloric acid and were shipped for analysis in this manner.

The amount of volatile mercury in the headspace of the reaction vessels was measured at a minimum of two time points for each soil sample during the *in vitro* procedure. Typically, sampling was done during the stomach phase and again at the end of the intestinal phase, before removing the last fluid sample. The vapor sampling was performed by using a syringe to withdraw either 1 or 2 cm^3 of the gases in the headspace through the septum of the reaction vessel. The sample volume was adjusted (to either 1 or 2 cm^3), to collect a sample within the analytical range of the instrument. The sample in the syringe was injected into the septum of a T-shaped sampling apparatus attached to the mercury vapor analyzer (Jerome® Model 431X). One arm of the sampling apparatus contained a zero-air filter trap, which prevented the passage of mercury between ambient air and the analyzer. Usually, triplicate samples of the headspace vapor were collected, and the results were averaged for each time point.

3.2 RESULTS

Prior to conducting *in vitro* testing on all soil samples from the site, tests were performed to evaluate the *in vitro* system, including an evaluation of mercury mass balance (i.e., total recovery of mercury mass from all components of the test system—soil, fluid, and vapor). These results are presented below, as is an assessment of quality control data from blind samples submitted to evaluate laboratory performance for the *in vitro* test. Appendix A

contains an additional discussion of quality control information, including an evaluation of the analytical laboratory's internal procedures to ensure data quality.

3.2.1 Mass Balance and Quality Control Samples

Before analyzing soil samples for the purpose of developing a site-specific BAF, one site soil sample was evaluated in the assay, in triplicate, to determine the potential for loss of mercury during the test (e.g., from volatilization of mercury, or from it adhering to the reaction vessel walls).[1] The quantity of mercury present in the soil sample before the *in vitro* assay was estimated from a split of the soil sample, analyzed in triplicate, with the results averaged. Samples of the *in vitro* fluids (i.e., the stomach- and intestinal-phase extracts) and of headspace mercury were collected, as described in the work plan, to determine the concentration and mass of mercury present in each phase. After the assay, the remaining soil was analyzed to determine the post-extraction mercury mass. The values for the mass of mercury present in each phase were summed and compared to the initial mass of mercury introduced into the reaction vessel. As can be seen in the mass balance results presented in Table 16, recoveries for individual assays ranged from 22 to 39 percent, with the mean recovery of introduced mercury being 32 percent for the three assays. While these results could be interpreted to indicate a potential for mercury to be lost from the test system through volatilization or from binding to reaction vessel walls, experience with previous mercury *in vitro* assays indicates that loss from the system is not likely.

The mass of mercury vapor measured in the headspaces of the reaction vessels was 0.02 percent, or less, of the total mass of mercury in the reaction vessel. Virtually 100 percent of the spiked mercury was measured in either the fluid extract samples or remained in the soil samples after extraction, suggesting that very little of the mercury in the soil samples was lost to volatilization during the *in vitro* extraction procedure. Similarly, previous mercury *in vitro* assays have demonstrated the stability of mercury in the aqueous phases of the test. This stability was confirmed with a matrix spike submitted as a blind quality control sample, for which 75 percent of the spiked amount was recovered (Table 17). Also, the recovery of mercury from a soluble mercury spike analyzed in duplicate assays (mean recovery of 78 percent) indicates the stability of mercury in the *in vitro* test solutions (Section 3.2.3; Table 19).

For the mass balance experiments, it is more likely that mercury recoveries of less than 100 percent can be explained by the small mass of soil that is available for laboratory analysis after the *in vitro* assay. The mass of post-extraction soil samples that feasibly could be collected was approximately 1 g (or less), wet weight. Because of the small sample size, there was insufficient soil available to split the samples for analysis of total

[1] The results for soluble mercury in the *in vitro* fluids for this sample (BS-69 [4–6 ft]) were also used in the calculations of percent bioaccessibility (Table 14), and to evaluate the reproducibility of the test (Figure 7).

mercury and determination of percent solids (to obtain a dry-weight mercury concentration value). Therefore, the analyses were performed sequentially on each individual post-extraction soil sample. On receipt by the analytical laboratory, the soil samples were first air-dried, to determine percent solids. Then the dried samples were analyzed for total mercury concentration. This additional drying procedure is a potential source of error in the interpretation of the mass balance results (the post-extraction drying procedure was not involved in any of the other *in vitro* assays). The soil samples were air-dried over a period of several days, and volatile forms of mercury (e.g., elemental mercury) that remain in the soil after the *in vitro* procedure could potentially volatilize before the sample is analyzed for total mercury. This lost mercury would not be accounted for in the mass balance calculations, possibly resulting in an underestimation of recovered mercury. It is likely that this drying procedure accounts for the low mercury recovery in the mass balance experiment.[2]

A review of laboratory quality assurance procedures (Appendix A) indicated that data from the *in vitro* analyses were acceptable and that the laboratory met requirements for accuracy, precision, and quality control. Laboratory analysis of a blind sample of the standard reference soil (NIST SRM 2711 Montana Soils) recovered 94 percent of the certified mean concentration of mercury, indicating good analytical accuracy (Table 17). Similarly, laboratory precision was good, with the results for a site soil sample submitted as a blind triplicate having a relative standard deviation (RSD) of 0.22 (data presented in Table 3). There was no evidence of mercury contamination during the *in vitro* assays, sample handling, or analysis, because blank matrix solutions were reported by the laboratory to have concentrations of mercury below method detection limits (Table 17).

3.2.2 Site Sample Results

Ten site soil samples that underwent mercury speciation analysis, were sieved to <250 μm for evaluation of bioaccessibility in the *in vitro* assay. Analysis of the samples revealed concentrations of total mercury ranging from 2.4 mg/kg to 5,850 mg/kg in the <250-μm fraction of the soil before *in vitro* extraction (Table 3). Approximately 1.5 g of each of these samples was then evaluated in the *in vitro* assay to determine the fraction of total mercury that was soluble under simulated gastrointestinal conditions.

Under the acidic conditions of the stomach phase, an average of 6 percent of the mercury was soluble at the 30-minute sampling point, and 8 percent at 1 hour (mean values of all 10 soil samples) (Figure 6). Under the near-neutral conditions of the intestinal phase, the average solubility initially decreased to 7 percent at 3.1 hours and then increased to 14

[2] In another study not reported here, the sample being tested was not air-dried, and the percent solids was estimated from another sample handled in a similar manner. In that case, mercury recovery in the soil was much higher, and overall mass balance was excellent (121 percent average recovery for triplicate analyses of a sample with similar mercury concentration and composition).

percent at 5.1 hours (Figure 6).[3] Results from the sample analyzed in triplicate in the assay, BS69 (4–6 ft), were averaged before being included in the calculations.

Because the solubility of mercury in the intestinal phase increased over time (Figure 6), the mercury bioaccessibility values resulting from the *in vitro* test are dependent on the length of time that soil spends in the intestinal phase of the assay. The available literature supports a small-intestinal transit time in humans of approximately 4 hours. As a result of peristaltic waves that move chyme (semi-fluid digested food material) along the small intestine, 3–5 hours is required for passage of chyme from the top of the small intestine to the entrance to the large intestine in adults (Guyton 1981). Studies of orocoecal transit time (from ingestion to the ileocecal valve at the entrance to the large intestine) in healthy children (n=7), after ingestion of a semi-solid meal, resulted in an average orocoecal transit time of 4.5 hours (Vajro et al. 1988). Subtracting a 1-hour stomach transit time from the above value results in a pediatric small intestinal transit time of 3.5 hours. It should be noted that orocoecal transit times following ingestion of a fluid meal are considerably shorter, approximately 60 minutes (Vajro et al. 1988; Murphy et al. 1988). Therefore, samples that are collected 4 hours into the small intestinal simulation represent a reasonable upper-bound approximation of small-intestinal transit times in children. Mercury bioaccessibility for the individual site soil samples ranged from 2 to 24 percent (Table 18). The average value was 14 percent (Figure 6).

The site sample analyzed in triplicate indicates the reproducibility of the *in vitro* assay results for an individual soil sample. The percent bioaccessibility results for Sample BS69 (4–6 ft) are presented in Figure 7. The mean value for each sampling time in the triplicate assay is depicted, as well as the individual data points for each *in vitro* test. The numerical results for the individual assays are presented in Table 18. The data exhibit good precision among the triplicate results.

As with the sample used for the mass balance tests, the results for the other nine soil samples indicate that only a very small percentage of the mercury present in the reaction vessels was detected in the vapor phase. For all 10 samples, on average, 0.15 percent of the total mercury was measured in the reaction vessel headspaces. Individual measurements ranged from 0.01 percent to 0.5 percent of the total as mercury vapor.

3.2.3 Mercuric Chloride Spike

To develop a site-specific BAF, the measured bioaccessibilities for mercury in site soils must be compared with the bioaccessibility of mercuric chloride, the soluble form of

---

[3] Sampling times are measured from the start of an *in vitro* assay and, for intestinal-phase extracts, vary slightly depending on the sodium bicarbonate titration time. Mean values were 3.1 hours (range of 2.8 hours to 3.3 hours) and 5.1 hours (range of 4.8 hours to 5.3 hours) for the first and second intestinal-phase samples, respectively.

mercury that is the basis of EPA's RfD for mercury. Four reaction vessels containing stomach solution were spiked with an aqueous solution of mercuric chloride, to attain a concentration of either 210 or 162 μg/L (as mercury). The quadruplicate *in vitro* tests were then treated in a fashion identical to the previous *in vitro* assays. On average, 78 percent of the mercuric chloride in the spikes was recovered in the *in vitro* extracts, with individual extract samples ranging from 35 percent to 102 percent recovered mercury (Table 19). It is unclear why two of the mercuric chloride spikes yielded percent recoveries lower than the other two samples (Table 19). For the purpose of the BAF calculation, discussed below, the bioaccessibility of mercuric chloride in the assay is defined to be 78 percent, the average value for all of the *in vitro* extract samples.

3.3 CALCULATION OF A BIOAVAILABILITY ADJUSTMENT FACTOR

No relation is apparent between the bioaccessibility of mercury and sample depth or location on the site; therefore, it is appropriate to calculate a single BAF for the entire site. As discussed above (Section 3.2.2), the mean bioaccessibility estimate for site soil samples is 14 percent. Likewise, on average, 78 percent of a mercuric chloride spike was measured to be bioaccessible in the *in vitro* assay (Section 3.2.3). Because the soluble mercury spike was less than 100 percent bioaccessible in the testing, the BAF is calculated by correcting the bioaccessibility estimate for soil samples for the recovery of the soluble mercury spike, to yield a relative bioavailability adjustment factor.

Evaluation of mercury bioaccessibility versus the concentration of mercury introduced into the assay indicates that mercury bioaccessibility decreases with increasing mercury concentration ($r^2 = 0.57$, $p < 0.01$; Figure 8). This relation appears to be due to the dominance of elemental mercury in the samples with the greatest total mercury concentrations. Note that the samples with 3,710 and 5,850 mg/kg mercury, which produced minimal bioaccessible mercury, both contained 99 percent elemental mercury, according to the sequential extraction results (Tables 3 and 8). Because mercury bioaccessibility increases with decreasing soil mercury concentrations, it is appropriate to apply a BAF relevant to the concentration range that does not pose an unacceptable risk of mercury exposure from direct contact under a commercial/industrial scenario. This mercury concentration is 520 mg/kg, based on direct-contact exposure to a commercial/industrial worker for a hazard index equal to 1, and assuming the default (non site-specific) bioavailability factor of 1. The linear correlation on Figure 8 indicates that the mercury bioaccessibility will be 0.16, on average, for soils with a mercury concentration in the range of 0 to 520 mg/kg (a BAF of 0.16 corresponds to a mercury concentration of 520 mg/kg). Thus, the BAF for the Ames Street site is calculated to be 0.20 (0.16 divided by 0.78).

The measured bioaccessibilities for the Ames Street site are similar to those obtained for soils containing mercuric sulfide and elemental mercury at the East Fork Poplar Creek site in Oak Ridge, Tennessee (Barnett and Turner 1995). As discussed above, the authors used an experimental *in vitro* technique slightly different from that employed here, and measured an average mercury bioaccessibility of 3.2 percent (range of 0.3 percent to 14.2

percent) for 19 of 20 samples. The twentieth sample exhibited 46 percent bioaccessibility; it was noted to have been collected near the mercury source and to be geochemically different from the other samples (Barnett and Turner 1995). Also, the authors of this study observed 100 percent recovery of a mercuric chloride spike in their test system. Based on these results, EPA accepted a BAF of 0.1 for mercury in soils at the site (DOE 1995).

3.4 DISCUSSION

The following section discusses features of the *in vitro* method that generally result in conservative estimates of bioaccessibility from the test data. These features include reduction-oxidation (redox) conditions and intestinal transit times.

3.4.1 Reduction-Oxidation Conditions

Because mercury solubility is highly dependent on redox conditions in the solubilizing fluid, it is important to ensure appropriate redox conditions in the test system. In general, both elemental mercury and mercuric sulfide will be more rapidly and completely dissolved under oxidizing conditions than under reducing conditions (Figure 9). As a result, appropriate redox control is important to reduce uncertainty regarding the experimental solubility of mercury. Although the stomach solution was sparged with argon at the beginning of each assay, and the reaction vessel was sealed for most of the testing time, the measured Eh in the reaction vessels was more oxidizing than in an actual mammalian gastrointestinal tract. The Eh values for *in vitro* solutions varied from +512 mV to +576 mV (average of +541 mV) at the beginning of the assays on site soils and from +336 mV to +409 mV (average of +387 mV) at the completion of those experiments (Table 18). These conditions are more oxidizing than those present in the human stomach, where an Eh of +308 mV (mean value) has been measured for the gastric fluid of healthy young volunteers who were fasting (n=132) (Mueller and Deistler 1987). It is important to note that the Eh reported for stomach fluids in fasting individuals is likely an upper-bound value. Researchers studying circadian rhythms in gastric juice reported that the reducing power of the fluids increased after meals (i.e., the Eh decreased), probably as a result of the stimulation of gastric secretions (De Flora et al. 1987). Therefore, Eh measurements taken in non-fasting individuals would likely be lower than +308 mV, the value for human gastric fluid reported above.

Similar redox conditions are reported for gastric fluids in other animals. In the forestomach of rabbits, fluids have been measured to have a Eh of +180 mV (Davis et al. 1992). Further along in the rabbit digestive tract, the conditions become more reducing, with average values of +20 mV reported for the duodenum and ileum (Davis et al. 1992). Also, redox potentials have been measured in the large intestines of swine, where Eh values of –173 mV and –214 mV are reported for the large-intestinal fluids in post-weanling piglets

and sows, respectively (Hornich and Chrastova 1981). These data are consistent with the trend of conditions becoming more reducing farther along in the digestive tracts of mammals, due to intestinal microbial activity, which produces a reducing environment. So, while no values could be located in the literature for the human small intestine, it is likely that the fluids present are more reducing than those of the human stomach (i.e., less than +308 mV, as reported above). Therefore, while attempts were made to create reducing conditions for the *in vitro* test similar to those of a human gastrointestinal tract, the experimental fluids, in fact, were more oxidizing than those present in a human digestive tract.

Because the *in vitro* fluids are more oxidizing than *in vivo* conditions, the experimental estimates of bioaccessibility are conservative. This occurs because species of mercury that would remain insoluble under more reducing conditions (e.g., mercuric sulfide and elemental mercury) are solubilized in the relatively more oxidizing fluids of the *in vitro* test. In examining an Eh-pH diagram for mercury (Figure 9), note that the Eh-pH conditions measured in the *in vitro* test are outside the stability field for mercuric sulfide and are near the edge of the stability field for elemental mercury, indicating that both of these compounds may be unstable in the *in vitro* test fluids. In contrast, these two compounds would be expected to be relatively stable under the actual Eh-pH conditions of the gastrointestinal tract (based on the values measured in rabbits [Davis et al. 1992]). Therefore, the experimentally determined bioaccessibility estimates include quantities of mercury that likely would not have been solubilized under the redox conditions present in the human gastrointestinal tract.

3.4.2 Intestinal Transit Time

In general, the measured bioaccessibility estimates for site soil samples increase with increasing time in the reaction fluids (Table 14, Figure 6), although several samples (e.g., samples BS41 [6–8 ft] and BS70 [0–2 ft]) exhibited a transient decline in mercury solubility at the beginning of the intestinal phase, associated with the increase in pH. The mean bioaccessibility estimates for the site soil samples reflect this trend (Figure 6). So, in theory, if the transit time in a human stomach and small intestine exceeded that used for the *in vitro* assay (i.e., 5.1 hours), the *in vitro* results would underestimate mercury solubility. However, a 5.1-hour transit time in the human gastrointestinal tract (1 hour in the stomach and 4.1 hours in the intestinal tract) is a conservative value, as discussed in Section 3.2.2. Furthermore, there is evidence that the gastrointestinal absorption of inorganic mercury occurs largely in the proximal jejunum (i.e., within the first two-fifths of the small intestine) (Nielsen et al. 1992). As a result, the intestinal transit time of importance for mercury absorption may be less than one-half the entire transit time. Therefore, the intestinal transit time used to calculate bioaccessibility in the *in vitro* assay provides a conservative estimate.

4. COMPARISON BETWEEN *IN VITRO* AND SPECIATION RESULTS

A trend in the data is readily apparent that would explain the solubility of mercury in the *in vitro* assay, relative to the species of mercury present in soil samples taken from the Ames Street site. In general, soil samples containing greater percentages of elemental mercury were observed to produce lower mercury bioaccessibility estimates ($r^2 = 0.43$) (Figure 10). So, while elemental mercury is near the edge of its stability field for the redox conditions of the *in vitro* test (Figure 9), it is not readily solubilized in the test system. This may be explained by reaction kinetics, with only those samples containing small particle sizes of elemental mercury (i.e., with a large ratio of surface area to mass) likely to oxidize and become solubilized within the time frame of the test. The reduced bioaccessibility observed in soil samples containing greater percentages of elemental mercury supports the data suggesting that elemental mercury is poorly absorbed from the gastrointestinal tract of animals and humans.

No relation was discerned among percent bioaccessible mercury and other parameters measured for the speciation analyses, including percent acid-soluble mercury, percent organic mercury, percent mercuric sulfide, percent TOC, or concentrations of AVS or total carbonate. Neither is a correlation apparent between the bioaccessibility of mercury and either sample depth or location on the site. Therefore, because *in vitro* bioaccessibility estimates for the site soil samples do not appear to vary spatially or vertically in the site soils, it is appropriate to calculate a single BAF for the entire site.

It should be noted that the *in vitro* assays were performed on the <250-μm soil fraction, while the <2-mm size fraction was used for the speciation studies. A comparison of the mercury concentrations in the two size fractions (Table 3) indicates that RPDs ranged from 3 to 69 percent. While useful information is gained by comparing the results of the *in vitro* and speciation studies, it is possible that for individual soil samples, the forms or relative amounts of the forms of mercury introduced into the *in vitro* assay are different from those measured in the speciation studies due to the fundamental heterogeneity of the mercury soil samples as previously discussed. Nevertheless, evaluation of the <250-μm soil fraction provides a conservative estimate of the mercury bioavailability in soils with larger grain sizes because soil grains <250-μm are more likely to be ingested than larger soil grains (soil grains larger than this do not appreciably adhere to skin [Duggan and Inskip 1985]), and soils <250-μm lend to greater bioavailability than larger soil grains (Chaney et al. 1989; Barltrop and Meek 1979). Therefore, application of the BAF developed in *in vitro* assays is appropriate to apply to soils throughout the entire site.

5. STABILITY OF MERCURY FORMS IN SOIL

This study was conducted to evaluate the forms of mercury present in soil at the Ames Street site, to develop a site-specific mercury BAF. Application of the results of this study is based on the premise that mercury forms in site soils will not alter appreciably in the near future, thereby changing the BAF. Because the bulk of mercury present at the site was probably deposited between 1904 and 1965 or so, it is reasonable to assume that elemental mercury was the form of mercury deposited in soils, and that it has been resident in soils for between 30 and 90 years. During this time, a portion of the mercury appears to have altered to the mercuric sulfide and acid-soluble mercury forms observed during this study.

The alteration of elemental mercury to other mercury forms in soil will eventually reach equilibrium, at which point, no net change in mercury speciation will occur in site soils. The acid-volatile sulfide data collected during this study suggest that there may be insufficient available sulfide in site soils for further formation of mercuric sulfide (Table 10). However, the stability constant for the formation of mercuric sulfide (log K_{sp} of –38.5, Di Toro et al. 1992) indicates that mercuric sulfide will be the preferred mercury soil alteration form in the presence of sufficient sulfide and reducing conditions. Reducing conditions may be present in the native soils at the Ames Street site, because they are covered with approximately 2 ft of fill material. With respect to further formation of acid-soluble mercury species, such as mercuric carbonates or chlorides, the EMPA data from this study indicate that mercuric chlorides and sulfates were the only acid-soluble mercury species found at any appreciable concentrations on the site (Table 14). Although it is difficult to predict the future formation of these species, data from the Oak Ridge, Tennessee, site and the Almaden Quicksilver County Park site in Los Gatos, California suggest that formation of mercuric chloride and sulfate is limited in soils (CDM 1992; Barnett and Turner 1995). Because negligible amounts of organic mercury were observed in this study, after 30 to 90 years of soil weathering time, it is highly unlikely that any appreciable amounts of organic mercury will form in site soils in the future. These data suggest that the mercury forms in soils at the Ames Street site are unlikely to change appreciably in the near future, and that mercuric sulfide would be the preferred soil alteration phase in the event of further alteration.

6. CONCLUSIONS

This study was conducted to identify the forms of mercury in soils, and to estimate a conservative relative bioavailability adjustment for mercury in soils at the Ames Street site.

An evaluation of soil mercury speciation in the Ames Street soil samples, based on sequential extraction and electron microprobe analyses, indicated that they contain elemental mercury as the dominant mercury form, with lesser amounts of mercuric sulfide and acid-soluble mercury species. However, both the mercury speciation techniques evinced limitations, with the sequential extractions underestimating the contribution of acid-soluble mercury forms, and the electron microprobe underestimating the fraction of elemental mercury and mercuric chloride. Despite these limitations, the mercury speciation results support the BAF derived from the *in vitro* assay, in that elemental mercury and mercuric sulfide, both mercury species with limited bioavailability, were observed to account for a majority of mercury in the Ames Street soil samples.

An *in vitro* assay was conducted on 10 site soils, to measure the mercury bioaccessibility from soil. The bioaccessibility estimate is likely conservative, because the Eh of the *in vitro* fluids was less reducing than actual conditions in the human digestive tract, and the intestinal transit time used in the *in vitro* test is likely greater than the small-intestinal transit time in humans. Therefore, some quantity of mercury was solubilized in the assay that would not be available for absorption in the human gastrointestinal tract.

The mean bioaccessibility estimate for mercury in site soil samples is 14 percent, and ranges from 2 to 24 percent. Because mercury bioaccessibility was observed to increase with decreasing mercury concentrations, a conservative BAF of 0.20 was selected as representative of site soils in the 0–520 mg/kg mercury concentration range.

7. REFERENCES

ABB. 1996. Final site investigation work plan. Ames Street site, Rochester, New York. Phase I. Appendices I and J. ABB Environmental Services, Inc., Rochester, NY.

Barltrop, D., Meek, F.; 1979; Effect of Particle Size on Lead Absorption from the Gut. Arch. Env. Health; 34:280-285

Barnett, M.O., L.A. Harris, R.R. Turner, T.J. Henson, R.E. Melton, and R.J. Stevenson. 1994. Characterization of mercury species in contaminated floodplain soils. Water Air Soil Pollut.

Barnett, M.O., and R.R. Turner. 1995. Bioavailability of mercury in East Fork Poplar Creek soils. Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, TN. Y-12 ER Report.

Belzile, N., P. Lecomte, and A. Tessler. 1989. Testing readsorption of trace elements during partial chemical extractions of bottom sediments. Environ. Sci. Technol. 23:1015–1020.

Berlin, M. 1979. Mercury. In: Handbook on the Toxicology of Metals. L. Friberg (ed.) Elsevier/North-Holland Biomedical Press.

Bornmann, G. G. Henke, H. Alfes, and H. Möllmann. 1970. Ueber die enterale resorption von metallischem quecksilber. [Intestinal absorption of metallic mercury.] Arch. Toxikol. 26:203-209 (not seen; as cited in Friberg and Nordberg [1973]).

CDM. 1992. Final report. Risk assessment, Almaden Quicksilver County Park, Volume II - Appendices. Appendix E. Camp Dresser & McKee, Inc., Denver, CO.

Chaney, R.L, Mielke, H.W., Sterrett, S.B. , 1989; Speciation, Mobility and Bioavailability of Soil Lead. Environ. Geochem. Health; 11:105-129

Davis, A., M.V. Ruby, and P.D. Bergstrom. 1992. Bioavailability of arsenic and lead in soils from the Butte, Montana, mining district. Environ. Sci. Technol. 26(3):461–468.

Davis, A., J.W. Drexler, M.V. Ruby, and A. Nicholson. 1993. Micromineralogy of mine wastes in relation to lead bioavailability, Butte, Montana. Environ. Sci. Technol. 27(7):1415– 1425.

DeFlora, S., G.S. Badolti, D. Serra, A. Picciotto, M.R. Magnolia, and V. Savarino. 1987. Cicadian reduction of chromium in the gastric environment. Mutation Research. 192:169–174.

Di Toro, D.M., J.D. Mahoney, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. Ankley. 1992. Acid volatile sulfide predicts toxicity of cadmium and nickel in sediments. Environ. Sci. Technol. 26(1):96–101.

DOE. 1994. Addendum to the East Fork Poplar Creek sewer line beltway remedial investigation report. Prepared by Science Application International Corporation, Oak Ridge, TN.

DOE. 1995. Record of Decision for Lower East Fork Poplar Creek. DOE/OR/02-1370&D1. U.S. Department of Energy, Office of Environmental Restoration and Waste Management. Prepared by Jacobs ER Team, Oak Ridge, TN.

Drexler, J.D. 1996. Personal communication (conversation with C. Sellstone, PTI Environmental Services, Boulder, CO). Department of Geological Services, University of Colorado, Boulder, CO.

Duggan, M.J., and M.J. Inskip. 1985. Childhood exposure to lead in surface dust and soil: A community health problem. Public Health Rev. 13:1–54.

EPRI. 1984. Chemical attenuation rates, coefficients, and constants in leachate migration. Vol. 1: A critical review. Prepared by Battelle, Pacific Northwest Laboratories, Richland, WA. EPRI EA-3356.

Friberg, L. and G. Nordberg. 1973. Inorganic Mercury – A toxicological and epidemiological appraisal. In: M.W. Miller and T.W. Clarkson (eds). Mercury Mercurials and Mercaptans. Charles C. Thomas, Springfield, IL.

Goyer, R.A. 1996. Toxic effects of metals. In: Casarett and Doull's Toxicology: The Basis Science of Poisons. Fifth edition. C.D. Klaassen, M.O. Amdur, and J.Doull (eds.) McGraw-Hill.

Guyton, A.C. 1981. Movement of food through the alimentary tract. In: Textbook of medical physiology, Sixth edition. W.B. Saunders Co., Philadelphia, PA. p. 792.

Hornich, M., and V. Chrastova. 1981. Redox potencial tlusteho streva prasete ve vztahu k dyzenterii prasat. [The redox potential of the large intestine in swine in relation to swine dysentery.] Vet Med. (Praha) 26(10):593–598. (Published in Czech). Abstract from "MedLine" On Line Database, National Library of Medicine.

Iverfeldt, A., and O. Lindquist. 1982. Distribution equilibrium of methylmercury chloride between water and air. Atmos. Environ. 16(12):2917–2925.

Lindquist, O., A. Jernelov, et al. 1983. Mercury in the Swedish environment: Global and local sources.

Miller, E.L. 1993. Speciation of mercury in soil. EPA Contract 68-CO-0049. U.S. Environmental Protection Agency, Quality Assurance and Methods Development Division, Environmental Monitoring Systems Laboratory, Office of Research and Development, Lockheed Environmental Sciences and Technology Company, Las Vegas, NV.

Moser, H.C., and A.F. Voigt. 1957. Dismutation of the mercurous dimer in dilute solutions. J. Am. Chem. Soc. 79:1837–1839.

Mueller, R.L., and M. Deistler. 1987. Dynamik der endogenen bakteriellen Nitritbilddung in Magen. 6. Mitteilung: Redoxpotential und Nitritbildung im Magensekret. [Dynamics of endogenuous bacterial nitrite formation in the stomach. 6. Redox potential and nitrite formation in stomach secretions.] Zentralbl Bakteriol Mikrobiol. Hyg. [B] 184(5):412-23.

Murphy, M.S., R. Nelson, and E.J. Eastham. 1988. Measurement of small intestinal transit time in children. Acta Paediatr. Scand. 77:802–806.

Nielsen, J.B., H.L. Andersen, and O. Andersen. 1992. Localization of gastrointestinal deposition of mercuric chloride studies *in vivo*. Pharmacol. Toxicol. 70(4):262–267.

Revis, N.W., T.R. Osborne, D. Sedgley, and A. King. 1989. Quantitative method for determining the concentration of mercury(II) in soils and sediments. Analyst 114:823-825.

Ruby, M.V., A. Davis, T.E. Link, R. Schoof, R.L. Chaney, G.B. Freeman, and P. Bergstrom. 1993. Development of an *in vitro* screening test to evaluate the *in vivo* bioaccessibility of ingested mine-waste lead. Environ. Sci. Technol. 27(13):2870–2877.

Ruby, M.V., A. Davis, R. Schoof, S. Eberle, and C.M. Sellstone. 1996. Estimation of lead and arsenic bioavailability using a physiologically based extraction test. Environ. Sci. Technol. 30(2):422–430.

Sakamoto, H.T., T. Tomiyasu, and N. Yonehara. 1992. Differential determination of organic mercury, mercury(II) oxide and mercury(II) sulfide in sediments by cold vapor atomic adsorption spectrometry. Anal. Sci. 8:35–39.

Sanemasa, I. 1975. The solubility of elemental mercury vapor in water. Bull. Chem. Soc. Jpn. 48(6):1795–1798.

Sin, Y.M., Y.F. Lim, and M.K. Wong. 1983. Uptake and distribution of mercury in mice from ingesting soluble and insoluble mercury compounds. Bull. Environ. Contam. Toxicol. 31(5):605–612.

Sin, Y.M, and Teh. 1992. Effects of long-term uptake of mercuric sulphide on thyroid hormones and glutathione in mice. Bull. Environ. Contam. Toxicol. 49(6):847–854.

U.S. EPA. 1991. Test methods for evaluating solid waste. Physical/chemical methods. SW-846. Third edition. U.S. Environmental Protection Agency, Washington, DC.

Vajro, P., G. Silano, D. Longo, A. Staiano, and A. Fontanella. 1988. Orocoecal transit time in healthy and constipated children. Acta Paediatr. Scand. 77:583–586.

Yeoh, T.S., H.S. Lee, and A.S. Lee. 1989. Gastrointestinal absorption of mercury following oral administration of cinnabar in a traditional Chinese medicine. Asia. Pac. J. Pharmacol. 4(2):69–73.

Figures and Tables

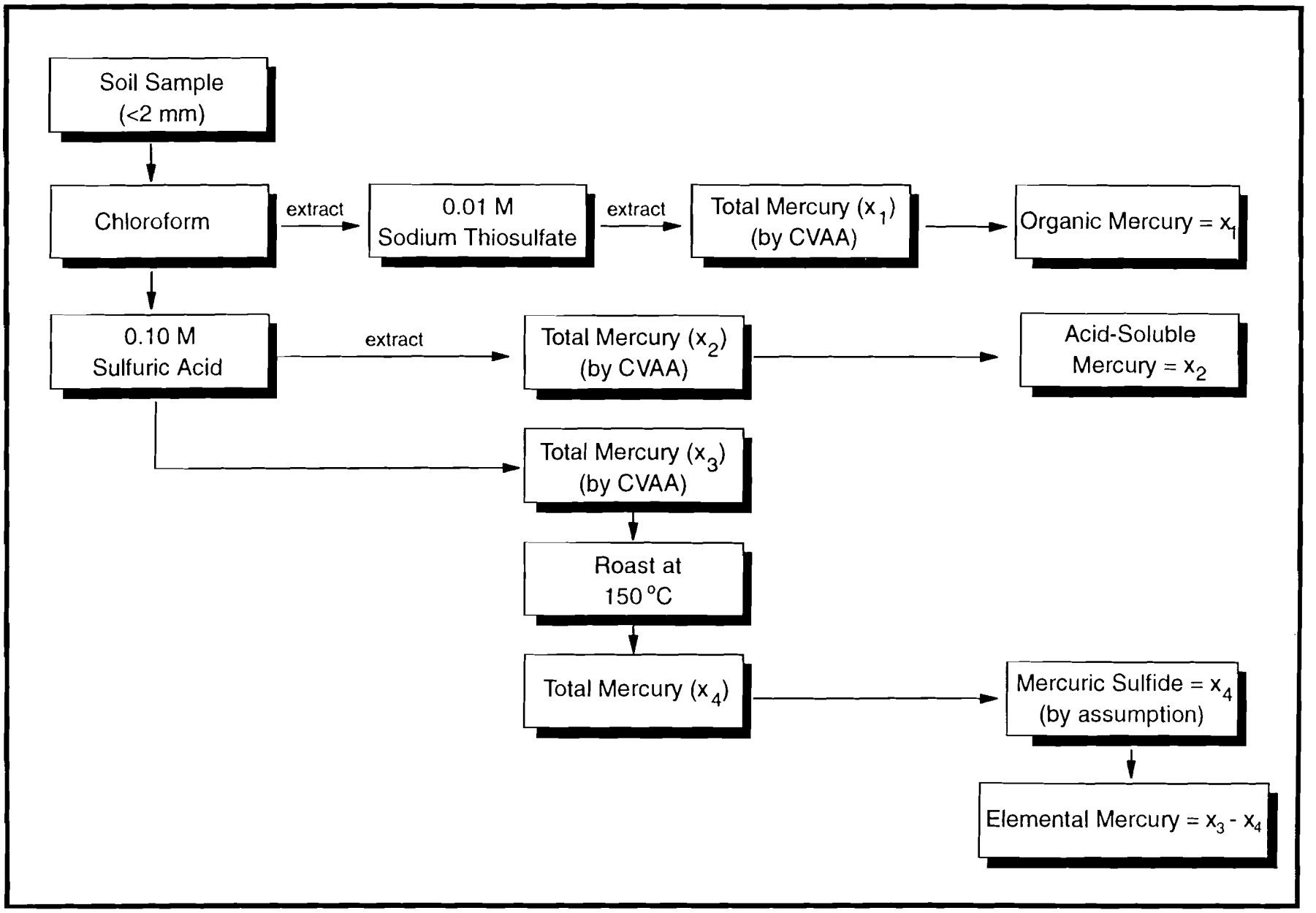


Figure 1. Sequential extraction of mercury phases in soil.

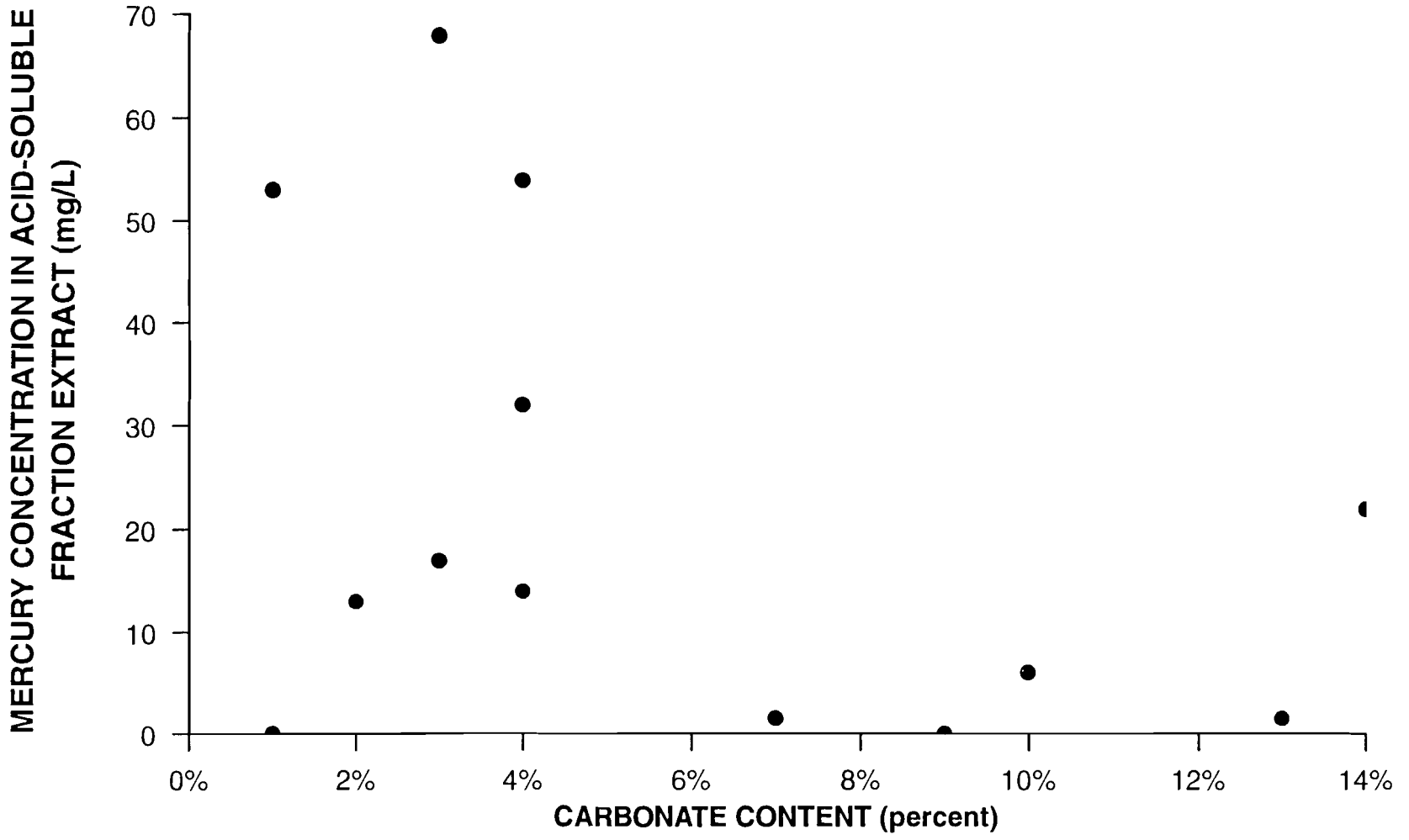


Figure 2. Carbonate concentration and fraction of mercury associated with the acid-soluble fraction of the sequential extraction procedure.

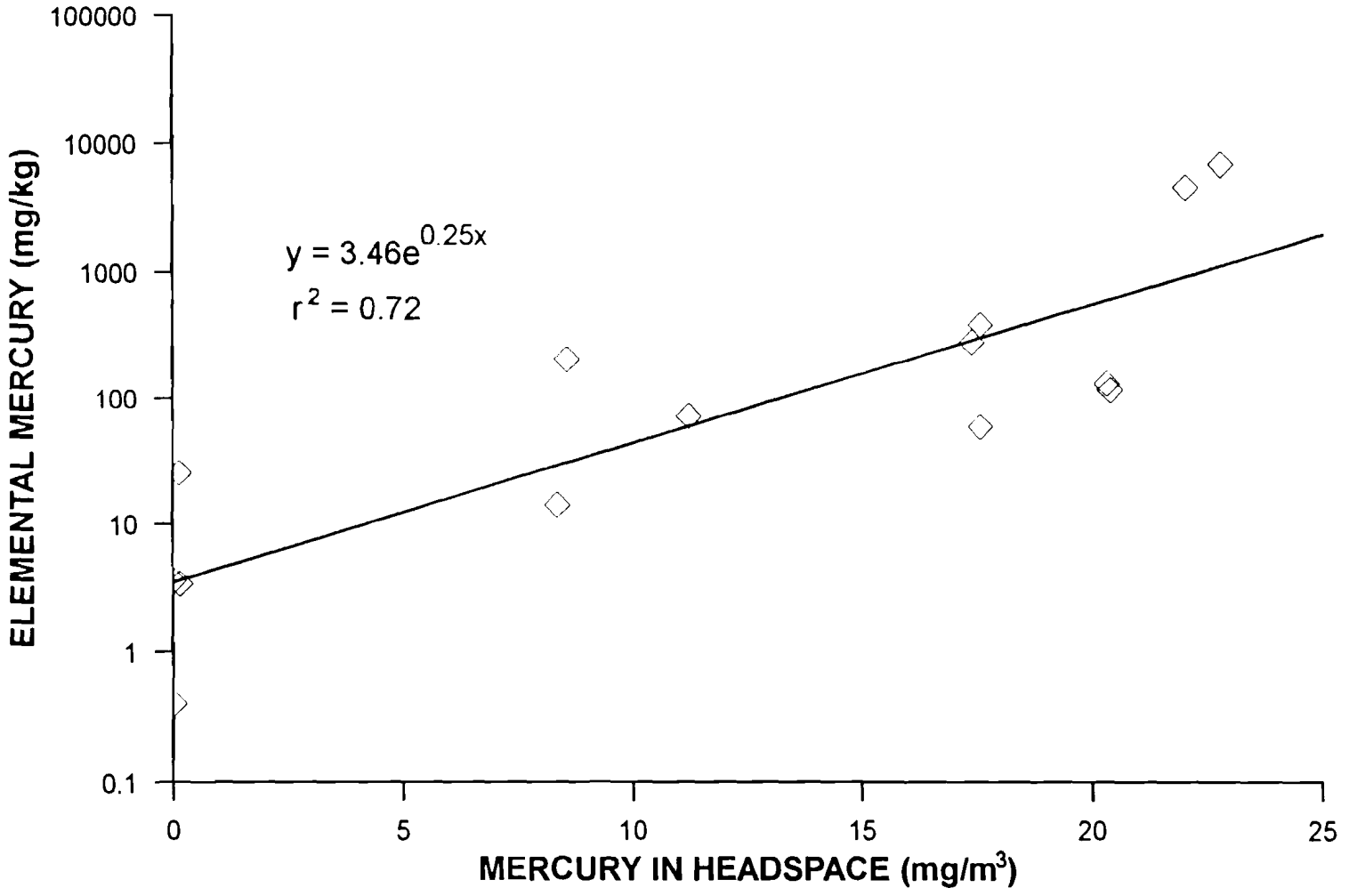


Figure 3. Average mercury concentration in headspace vs. concentration of elemental mercury (as determined by sequential extraction).

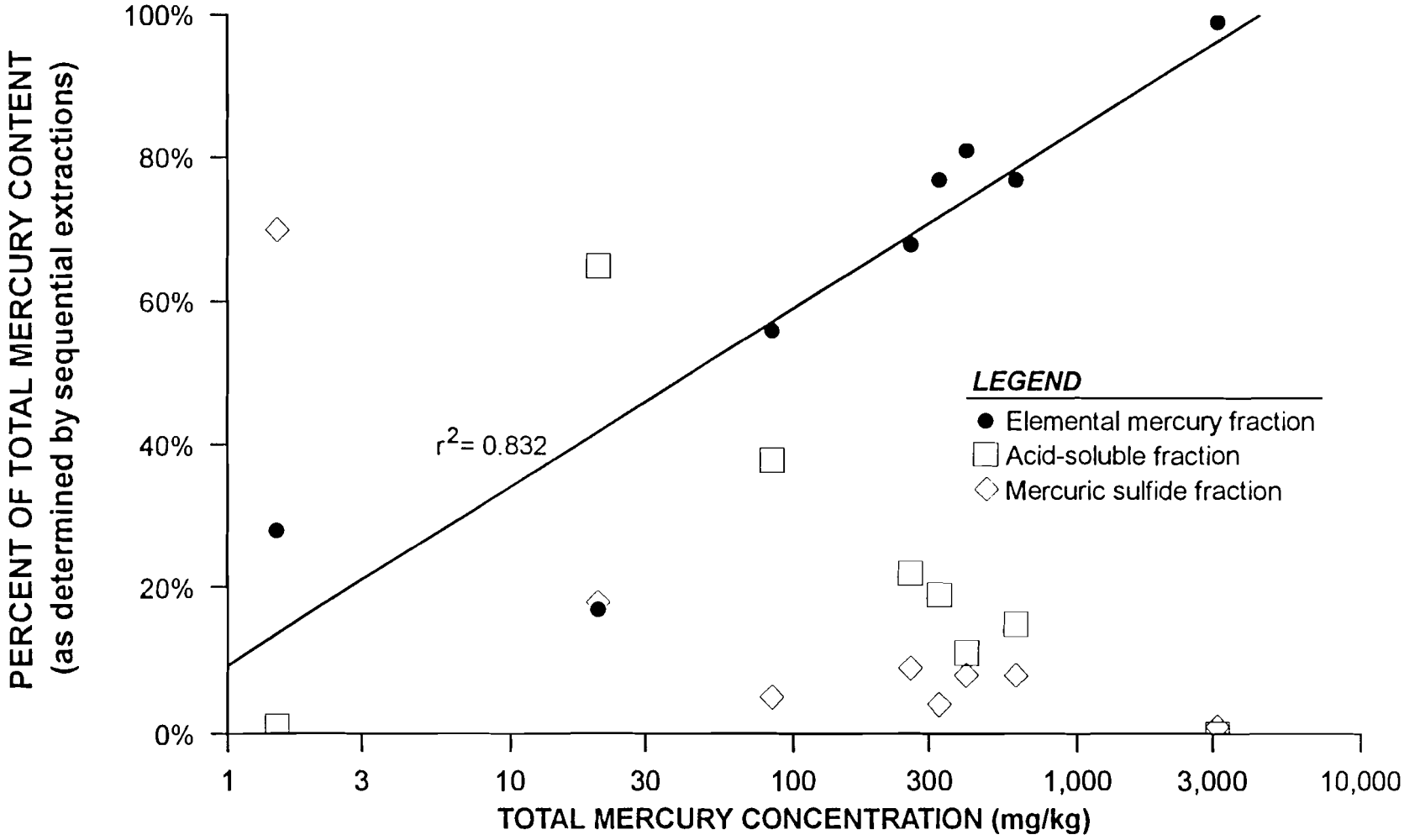


Figure 4. Sequential extraction results and mercury content of Ames Street site soils.

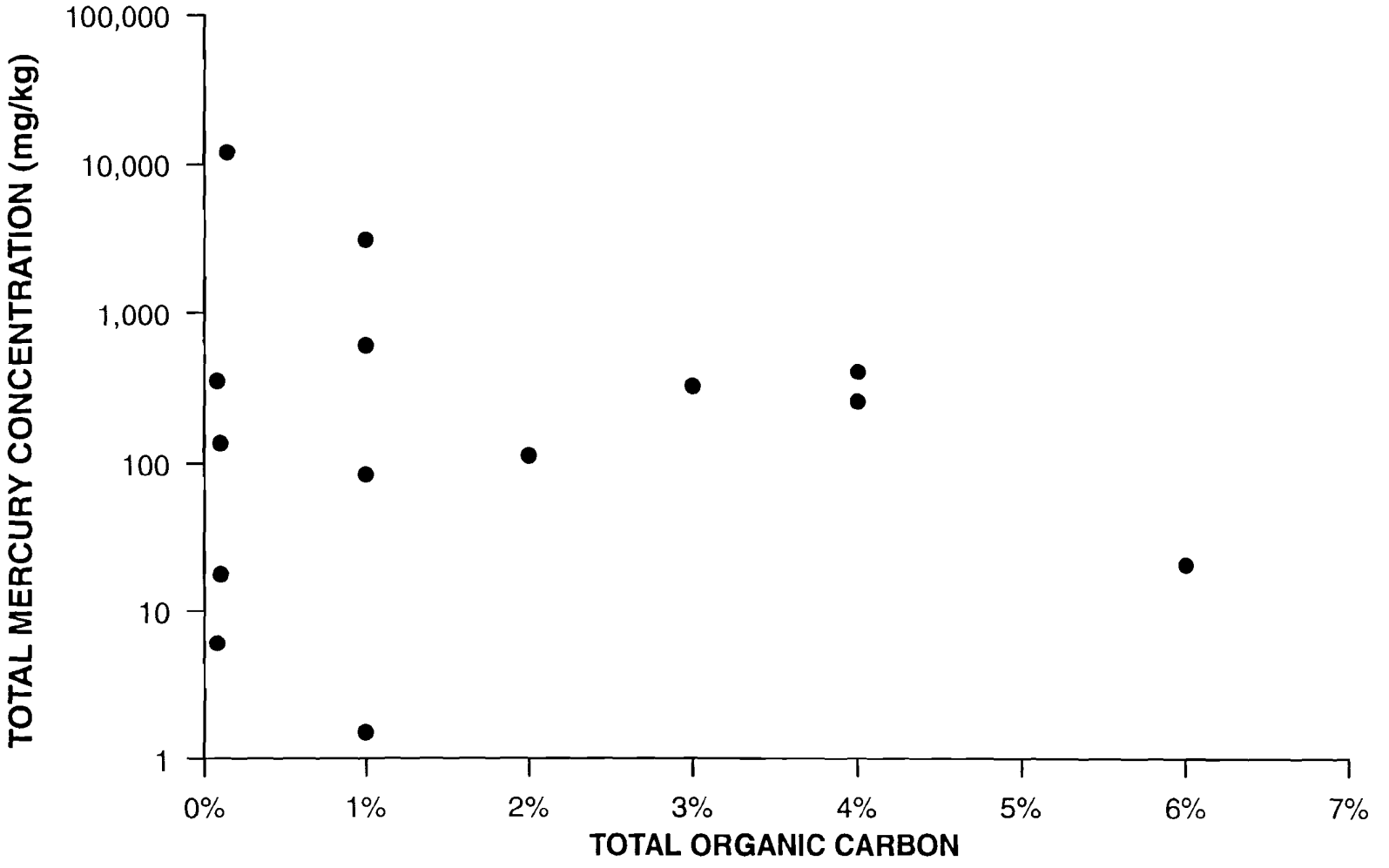


Figure 5. TOC and mercury concentration in Ames Street site soils.

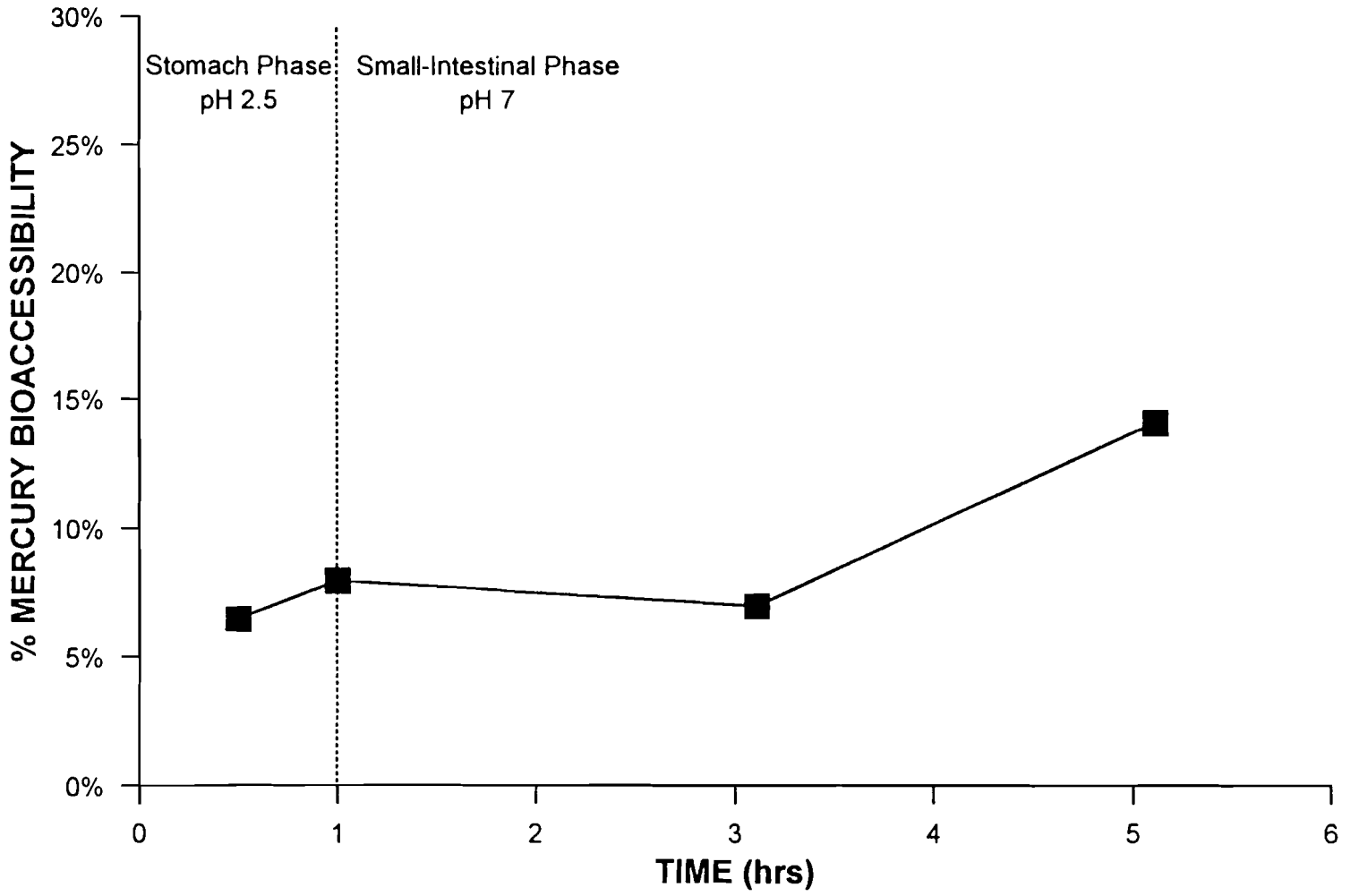


Figure 6. Average mercury bioaccessibility in 10 soil samples from the Ames Street site.

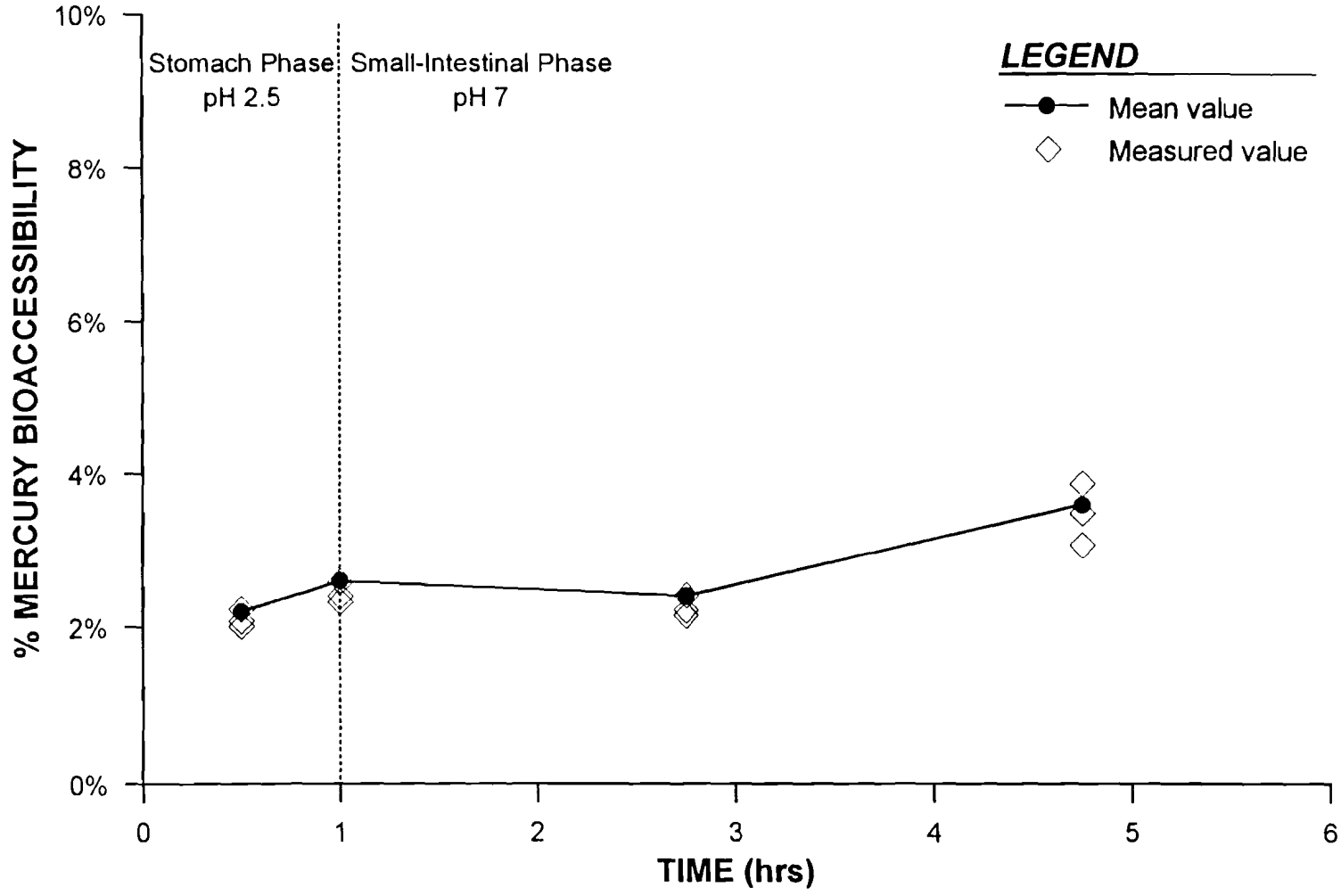


Figure 7. Triplicate mercury bioaccessibility results for soil sample BS69 (4-6 ft) from the Ames Street site.

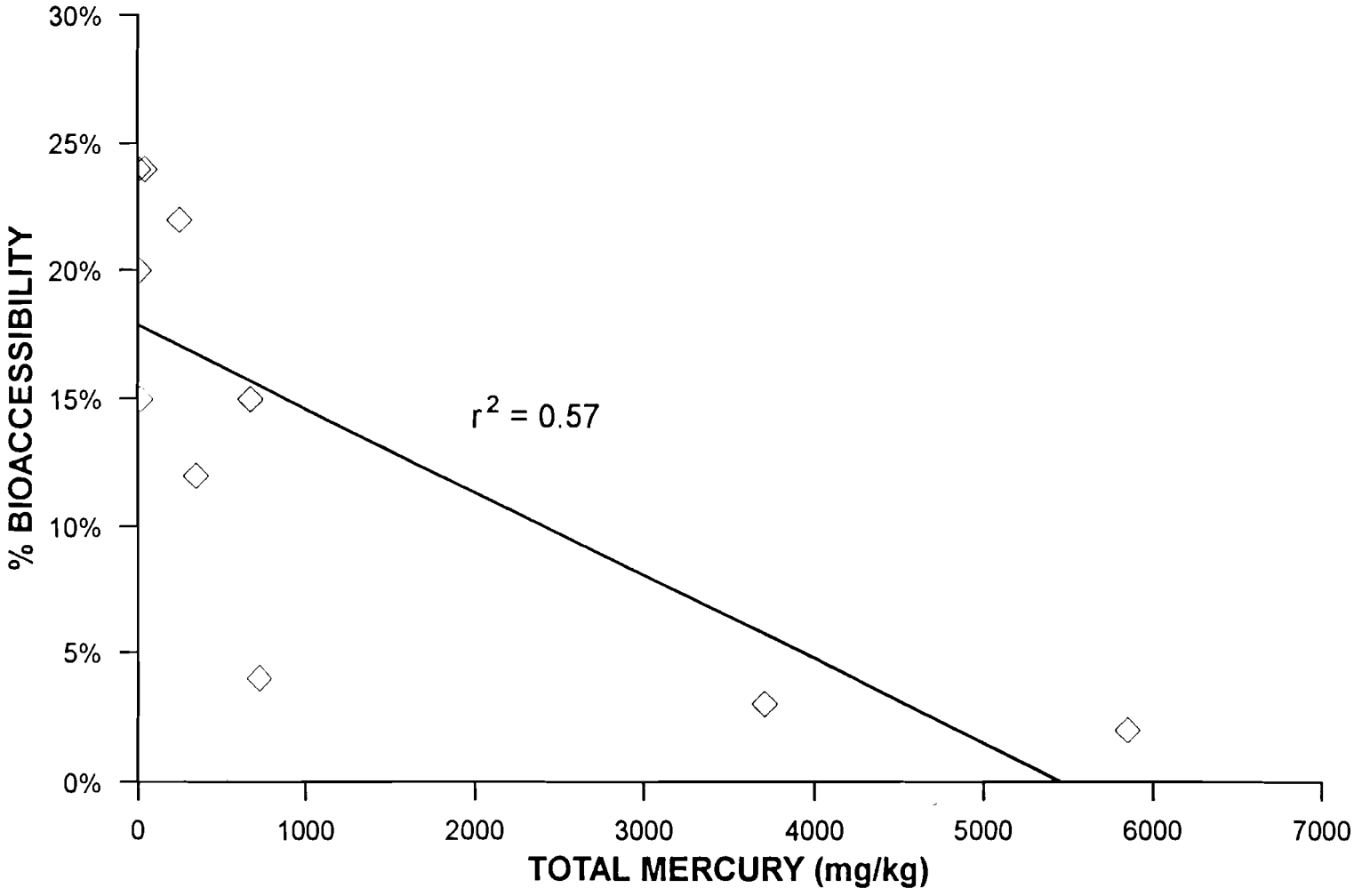
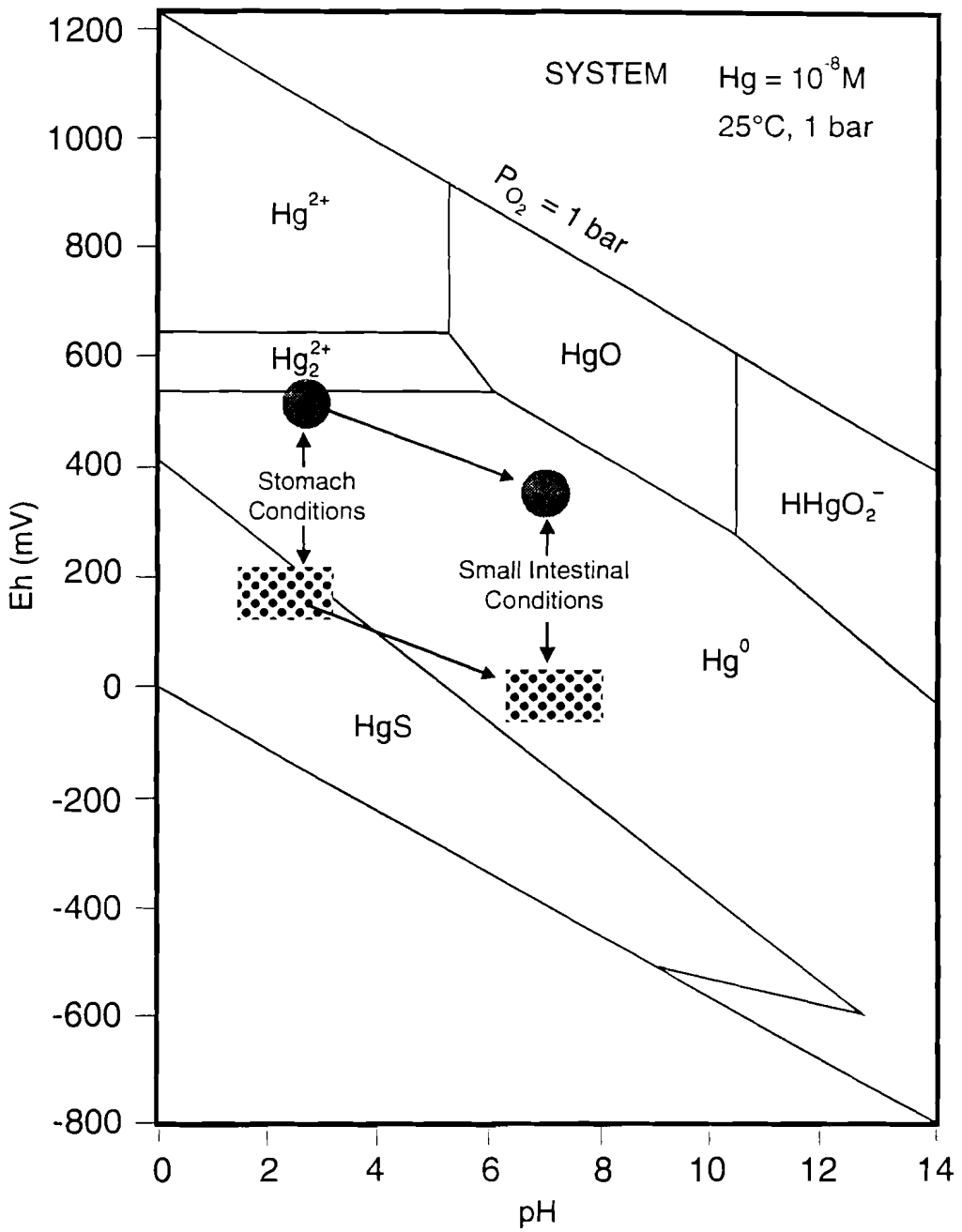


Figure 8. Percent bioaccessible mercury vs. total mercury concentration in the <250-µm size fraction.



LEGEND

- Conditions present in the in vitro test
- Gastro-intestinal Eh/pH conditions reported for animals and humans

Adapted from Brookins, D.G. 1988. *Eh-pH Diagrams for Geochemistry.* Springer-Verlag, New York.

Figure 9. Eh-pH diagram for mercury.

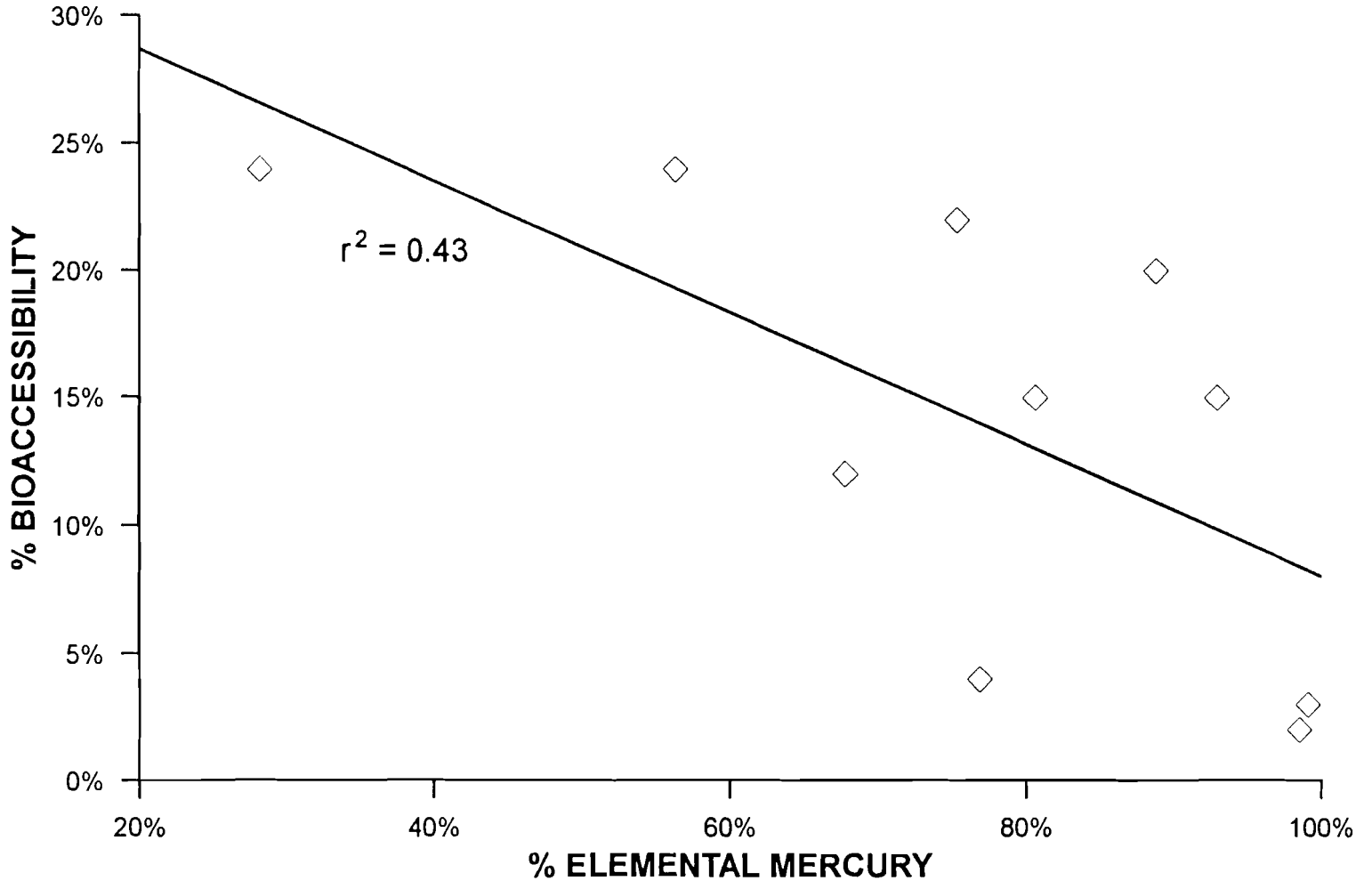


Figure 10. Percent bioaccessible mercury vs. percent elemental mercury (as determined by sequential extraction).

TABLE 1. MERCURY VAPOR AND SOIL MOISTURE DATA

Station	Upper Depth (ft)	Lower Depth (ft)	Total Mercury (mg/kg)[a]	Reading 1 Headspace Concentration (mg/m^3)	Reading 2 Headspace Concentration (mg/m^3)	Reading 3 Headspace Concentration (mg/m^3)	Average Concentration (mg/m^3)	Percent Moisture
BS21	6	8	6.0	0.32	0.25	0.22	0.26	10%
BS21	6	8	6.0	0.04	0.04	0.04	0.04	10%
BS41	4	6	349	21.5	18.3	21.3	20.4	15%
BS41	6	8	12000	23.5	19.7	22.8	22.0	11%
BS42	2	4	3110	23.6	20.4	24.0	22.7	14%
BS42	2	4	3110	24.9	20.3	23.2	22.8	14%
BS42	4	6	136	13.4	9.2	11.0	11.2	19%
BS46	2	4	326	21.5	17.9	21.5	20.3	23%
BS68	6	8	0.2 U	0.004	0.004	0.004 *U*	0.004	13%
BS69	2	4	20.4	0.01	0.01	0.004 *U*	0.01	28%
BS69	4	6	604	18.4	15.8	17.8	17.4	27%
BS69	6	8	17.9	8.5	7.8	8.8	8.4	12%
BS70	0	2	405	17.1	16.1	19.5	17.5	19%
BS72	4	6	84.0	0.15	0.12	0.13	0.13	20%
BS73	0	2	113	16.5	16.8	19.8	17.7	10%
BS73	0	2	113	16.5	16.5	19.3	17.4	10%
BS74	0	2	1.5	0.02	0.03	0.01	0.02	17%
SS03	0	2	258 [b]	8.4	8.0	9.3	8.6	19%

U = Not detected; value represents the detection limit.

U = Not detected; value represents the detection limit. Detection limit was calculated by dividing the sensitivity of the Jerome mercury analyzer by the dilution factor.

[a] Mercury concentration measured in the <2-mm size fraction.

[b] Value is the average of triplicate analyses. Actual values were 194, 312, and 268 mg/kg.

TABLE 2. MERCURY PARTITION COEFFICIENTS AT ROOM TEMPERATURE (25 °C)

Mercury Species	Air-Water (v/v)	Reference
Hg^0	0.32	Moser and Voigt (1957) Sanemasa (1975)
CH_3HgCl	1.9×10^{-5}	Iverfeldt and Lindquist (1982)
$HgCl_2$	2.9×10^{-8}	Lindquist et al. (1983)
$Hg(OH)_2$	3.2×10^{-6}	Lindquist et al. (1983)

TABLE 3. COMPARISON OF TOTAL MERCURY IN SOIL SIZE FRACTIONS

(All units mg/kg unless otherwise noted)

Station	Upper Depth (ft)	Lower Depth (ft)	<2-mm Size Fraction Total Mercury	<250-μm Size Fraction Total Mercury	Relative Percent Difference
BS21	6	8	6.0	6.5	8%
BS41	4	6	349	246	35%
BS41	6	8	12000	5850	69%
BS42	2	4	3110	3710	18%
BS42	4	6	136		
BS46	2	4	326		
BS69	2	4	20.4		
BS69	4	6	604	723 [b]	18%
BS69	6	8	17.9	17.3	3%
BS70	0	2	405	667	49%
BS72	4	6	84.0	42.1	66%
BS73	0	2	113		
BS74	0	2	1.5	2.4	46%
SS03	0	2	258 [a]	344	29%

[a] Values are the average of triplicate analyses. Actual values were 194, 312, and 286 mg/kg.

[b] Values are the average of triplicate analyses. Actual values were 826, 538, and 805 mg/kg.

TABLE 4. ACID-VOLATILE SULFIDE, CARBONATE, AND TOTAL ORGANIC CARBON RESULTS

Station	Upper Depth (ft)	Lower Depth (ft)	Acid-Volatile Sulfide (mg/kg as S^{2-})	Total Carbonate (% as CO_3)	Total Organic Carbon (% as C)
BS21	6	8	5 UJ	9.24	0.08
BS41	4	6	5 UJ	9.66	0.14
BS41	6	8	5 UJ	7.01	0.08
BS42	2	4	5 UJ	2.41	1.19
BS42	4	6	5 UJ	13.2	0.14
BS46	2	4	5 UJ	4.17	2.72
BS68	6	8	5 UJ	8.23	0.05 U
BS69	2	4	5 UJ	4.00	6.00
BS69	4	6	5 UJ	4.27	1.20
BS69	6	8	5 UJ	9.00	0.06
BS70	0	2	5 UJ	1.19	3.82
BS72	4	6	5 UJ	2.76	1.46
BS73	0	2	5 UJ	13.9	2.35
BS74	0	2	5 UJ	0.88	1.10
SS03[a]	0	2	5 UJ	3.16	4.18

U = Not detected; value represents detection limit.

J = Estimated as qualified during data validation.

[a] Values are the average of triplicate analyses.

TABLE 5. CALCULATED DISTRIBUTION OF MERCURY AMONG MINERAL PHASES OF A SOIL SPIKED WITH ELEMENTAL MERCURY

Target Phase	Expected Distribution	Observed Distribution	
		Spike Sample 1[a]	Spike Sample 2[b]
Organic mercury	0%	0.001%	-[d]
Acid-soluble mercury	0%	0.003%	-[d]
Elemental mercury	100%	151%[c]	99.95%
Mercuric sulfide	0%	-[c]	0.05%

[a] Sample BS-68 (6-8 ft) spiked with 1,140 mg/kg elemental mercury.

[b] Sample BS-68 (6-8 ft) spiked with 5,460 mg/kg elemental mercury.

[c] The soil sample was analyzed for total mecury content following the first two extraction steps.

[d] This sample was not subjected to the first two extraction steps.

TABLE 6. SEQUENTIAL EXTRACTION RESULTS OF SPIKED AMES STREET SOIL

	Concentration of Each Phase Added to the Spiked Sample as Mercury (mg/kg)	Measured Concentration in Each Phase of Spike Sample 3		Theoretical Distribution of Mercury Among Phases	Observed Distribution of Mercury Among Phases[a]
		Replicate 1 (mg/kg)	Replicate 2 (mg/kg)		
Organic Mercury	0	33.5	37.6	0%	2%
Acid-Soluble Mercury	1,020	108	82.4	70%	7%
Elemental Mercury	0	320	500	0%	28%
Mercuric Sulfide	444	568	530	30%	37%
Total	1,466	1,030	1,150	100%	74% [b]

[a] These are mean values of the two replicates.

[b] Because the total mercury recovered was less than 100%, it is likely that mercury was lost during the extraction procedure.

TABLE 7. SEQUENTIAL EXTRACTION RESULTS OF SPIKED INTERNAL SOIL SAMPLE

	Concentration of Each Phase Added to the Spiked Sample as Mercury (mg/kg)	Measured Concentration in Each Phase		Theoretical Distribution of Mercury Among Phases	Observed Distribution of Mercury Among Phases
		Replicate 1 (mg/kg)	Replicate 2 (mg/kg)		
Organic Mercury	0	4.94	7.19	0%	2%
Acid-Soluble Mercury	222	229	186	69%	64%
Elemental Mercury	0	102.4	82	0%	29%
Mercuric Sulfide	100	93.6	114	31%	32%
Total	322	430	389	100%	127%

TABLE 8. SEQUENTIAL EXTRACTION RESULTS OF AMES STREET SOIL SAMPLES

Station	Upper Depth (ft)	Lower Depth (ft)	Total Mercury (mg/kg)	Organic Mercury (mg/kg)	Organic Mercury Percent[a]	Acid-Soluble Mercury (mg/kg)	Acid-Soluble Mercury Percent	Elemental Mercury (mg/kg)	Elemental Mercury Percent	Mercuric Sulfide (mg/kg)	Mercuric Sulfide Percent
BS21	6	8	6	0.027 UB	0.7%	0.004	0.1%	3.4	89%	0.4	10%
BS41[a]	4	6	349	0.359	0.2%	1.5	0.9%	118	75%	36.9	24%
BS41	6	8	12000	0.907	0.02%	6.0	0.1%	4647	99%	63.4	1%
BS42	**2**	**4**	**3110**	**1.80**	**0.03%**	**13**	**0.2%**	**6960**	**99%**	**50**	**0.7%**
BS42	4	6	136	0.108	0.1%	1.5	2%	73	83%	13.2	15%
BS46	**2**	**4**	**326**	**0.184**	**0.1%**	**32**	**19%**	**133**	**77%**	**6.40**	**4%**
BS69	**2**	**4**	**20.4**	**0.120**	**0.6%**	**14**	**65%**	**3.6**	**17%**	**3.8**	**18%**
BS69	**4**	**6**	**604**	**0.687**	**0.2%**	**54**	**15%**	**281**	**77%**	**29.5**	**8%**
BS69	6	8	17.9	0.131	0.8%	0.07	0.5%	14	93%	0.9	6%
BS70	**0**	**2**	**405**	**1.560**	**0.3%**	**53**	**11%**	**388**	**81%**	**38**	**8%**
BS72	**4**	**6**	**84**	**0.101**	**0.2%**	**17**	**38%**	**26**	**56%**	**2.4**	**5%**
BS73	0	2	113	0.455	0.5%	22	22%	60	60%	18	18%
BS74	**0**	**2**	**1.5**	**0.007**	**0.5%**	**0.01 UB**	**1%**	**0.40**	**28%**	**1.0**	**70%**
SS03	**0**	**2**	**258** [b]	**2.267**	**0.7%**	**68**	**22%**	**206**	**68%**	**28**	**9%**
Average[c]					**0.3%**		**21%**		**63%**		**15%**

Bolded data represent samples with carbonate of < 5 percent.

UB = Qualified as not detected because analyte was detected in the blank.

[a] Percent of mercury recovered during sequential extraction analysis.

[b] Value is the average of triplicate analyses.

c Average of bolded values

TABLE 9. COMPARISON OF TOTAL MERCURY CONCENTRATIONS WITH RECOVERED MERCURY

Station	Upper Depth (ft)	Lower Depth (ft)	Total Mercury in < 2-mm Size Fraction (mg/kg)	Cumulative Mercury Recovered During the Sequential Extraction (mg/kg)	Percent Lost	Percent Gained
BS21	6	8	6.0	3.83	36%	
BS41[a]	4	6	349	157	55%	
BS41	6	8	12000	4717	61%	
BS42	2	4	3110	7024		126%
BS42	4	6	136	87.6	36%	
BS46	2	4	326	172	47%	
BS69	2	4	20.4	21.6		6%
BS69	4	6	604	365	40%	
BS69	6	8	17.9	15.4	14%	
BS70	0	2	405	481		19%
BS72	4	6	84.0	45.5	46%	
BS73	0	2	113	99.9	12%	
BS74	0	2	1.5	1.4	7%	
SS03[a]	0	2	258 [b]	304		18%

[a] Values are the average of duplicate analyses.

[b] Value is the average of triplicate analyses.

TABLE 10. COMPARISION BETWEEN ACID-VOLATILE SULFIDE AND SEQUENTIAL EXTRACTION RESULTS

Station	Upper Depth (ft)	Lower Depth (ft)	Mercuric Sulfide (mg/kg as Hg)[a]	S^{2-} Content of Mercuric Sulfide (mg/kg as S^{2-})	Acid-Volatile Sulfide (mg/kg as S^{2-})
BS21	6	8	0.4	0.06	5 UJ
BS41[b]	4	6	36.9	5.90	5 UJ
BS41	6	8	63.4	10.13	5 UJ
BS42	2	4	50	7.99	5 UJ
BS42	4	6	13.2	2.11	5 UJ
BS46	2	4	6.40	1.02	5 UJ
BS69	2	4	3.8	0.61	5 UJ
BS69	4	6	29.5	4.71	5 UJ
BS69	6	8	0.9	0.14	5 UJ
BS70	0	2	38	6.07	5 UJ
BS72	4	6	2.4	0.38	5 UJ
BS73	0	2	18	2.88	5 UJ
BS74	0	2	1.0	0.16	5 UJ
SS03[c]	0	2	28	4.48	5 UJ

U = Not detected; value represents detection limit.

J = Estimated as qualified during data validation.

[a] The mercuric sulfide content was determined in the sequential extraction procedure.

[b] Value is the average of duplicate analyses.

[c] Value is the average of triplicate analyses.

TABLE 11. EMPA RESULTS FOR AMES STREET SOIL SPIKED WITH ELEMENTAL MERCURY, MERCURIC CHLORIDE, AND MERCURIC SULFIDE

(All units mg/kg unless otherwise noted)

	Concentration of Each Phase Added to the Spiked Sample		Theoretical Distribution of Mercury Among Phases		Observed Distribution of Mercury Among Phases	
	Spike 1	Spike 2	Spike 1	Spike 2	Spike 1	Spike 2
Elemental mercury	555	630	33%	26%	5%	11%
Mercuric chloride	579	625	34%	35%	0%	2%
Mercuric sulfide	554	828	33%	39%	93%	87%
Other	0	0	0%	0%	2% [a]	0.1% [a]
Total	1,688	2,082	100%	100%	100%	100%

[a] 2% of the quatifiable mercury was found to be associated with the mercuric sulfate phase.

TABLE 12. FREQUENCY OF OCCURRENCE OF MERCURY-BEARING PARTICLES

	BS-41		BS-42	BS-69	BS-70	SS-03	BS-72
Depth of sample (ft):	4-6	6-8	2-4	4-6	0-1	0-1	4-6
Mercury concentration (mg/kg):	246	5850	3710	723	667	344	42
No. of particles counted:	70	161	98	66	100	54	16
Chlorides and Sulfates							
Mercuric chloride	4%	86%	5%	3%	14%	2%	6%
Mercuric iron sulfate					1%		
Mercuric sulfate				29%	1%	4%	
Oxides							
Manganese oxide	1%						
Mercuric iron oxide (leaded)					2%		
Iron oxide						4%	
Sulfides							
Mercuric sulfide (stannous)					14%		
Mercuric sulfide	1%			39%	60%	59%	94%
Metals and alloys							
Elemental mercury	93%	14%	95%	29%	8%	28%	
Bronze						4%	

TABLE 13. MEASURED MERCURY CONCENTRATION AND ESTIMATED SPECIFIC GRAVITY OF MERCURY PHASES FOUND WITHIN THE EMPA SAMPLES

Mineral Phase	Average Mercury Concentration (Percent)	Specific Gravity	Source of Specific Gravity Estimate	Mineral Phases Used To Estimate Specific Gravity
Chlorides and sulfates				
Mercury chloride	74	6	CRC 1985[b]	Average of all mecury chloride species listed in CRC 1985.
Iron sulfate	18	3.2	Klein and Hurlbut 1985	Jarosite
Mercury sulfate	29	6.4	CRC 1985	$HgSO_4 \cdot 2H_2O$
Oxides				
Manganese oxide	4.1	4.0	CRC 1985	Estimated from $Fe_2O_3 \cdot cH_2O$ (amorphous), Fe_2O_3, and MnO_2
Iron oxide	4.1	3.0	CRC 1985	$Fe_2O_3 \cdot cH_2O$ (amorphous)
Iron oxide (leaded)	9.1	3.0	CRC 1985	$Fe_2O_3 \cdot cH_2O$ (amorphous)
Sulfides				
Mercury sulfide	86	8.1	Klein and Hurlbut 1985	Cinnabar
Mercury sulfide (stannous)	23	5.9	CRC 1985	Cinnabar and SnS
Metals and alloys				
Mercury	100	13.6	Klein and Hurlbut 1985	Elemental mercury
Bronze	11	7.5	Estimated	Average of values found via Internet search

[a] Klein, C. and C. Hurlbut. 1985. Manual of mineralogy. John Wiley and Sons, New York. p. 596.

[b] CRC 1985. CRC Handbook of chemistry and physics. CRC Press, Boca Raton, FL.

TABLE 14. DISTRIBUTION OF RELATIVE MERCURY MASS AMONG MERCURY-BEARING PARTICLES

	BS-41		BS-42	BS-69	BS-70	SS-03	BS-72	Average[a]
Depth of sample (ft):	4-6	6-8	2-4	4-6	0-2	0-2	4-6	
Mercury concentration (mg/kg):	246	5850	3710	723	667	344	42	
No. of particles counted:	70	161	98	66	100	54	16	
Chlorides and Sulfates								
Mercuric chloride	1%	67%	2%	2%	10%	1%	4%	14%
Mercuric iron sulfate					0%			0%
Mercuric sulfate				14%	1%	2%		1%
Oxides								
Manganese oxide	0%						0%	0%
Mercuric iron oxide (leaded)					0%		0%	0%
Iron oxide						0%	0%	0%
Sulfides								
Mercuric sulfide (stannous)					3%			0%
Mercuric sulfide	1%			35%	68%	51%	96%	24%
Metals and alloys								
Elemental mercury	98%	33%	98%	50%	18%	46%		61%

[a] The average was calculated by intergrating all EMPA data into one data set.

TABLE 15. RESULTS OF THE GRAVIMETRIC SEPARATION OF SITE SOILS

Sample ID	Mass of Mercury in Spike (g)	Mass of Soil Panned	Mass of Mercury Recovered (g)	Mercury Concentration Recovered (mg/kg)	Expected Mercury Concentration (mg/kg)	Percent of Mercury Recovered
BS-75 0-2 ft [a]	0.6492	100.77	0.5942	5,897	6,444	92%
BS-41 6-8 ft	0	83.54	0.4916	5,884	5,850 [b]	101%
BS-41 6-8 ft					12,000 [c]	49%

[a] Site soil with minimal mercury concentration, which was spiked with elemental mercury.

[b] Mercury concentration measured in the <250-μm size fraction of sample BS-41 6-8 ft.

[c] Mercury concentration measured in the <2-mm size fraction of sample BS-41 6-8 ft.

TABLE 16. MERCURY MASS BALANCE RESULTS FOR TRIPLICATE *IN VITRO* SAMPLES

Sample ID	Upper Depth (ft)	Lower Depth (ft)	Mass of Soil (g)	Mass of Mercury in Pre-Extraction Soil (mg)	Mass of Mercury in Post-Extraction Soil (mg)	Mass of Mercury in Fluid Phase (mg)	Mass of Mercury in Headspace[a] (mg)	Percent Recovery
BS69	4	6	1.505	1.088	0.384	0.036	0.00015	39%
BS69	4	6	1.505	1.088	0.202	0.041	0.00016	22%
BS69	4	6	1.507	1.089	0.327	0.045	0.00015	34%

[a] The average value of all headspace samples collected for each reaction vessel during an assay.

TABLE 17. MERCURY *IN VITRO* SPIKE AND BLANK RESULTS

Matrix	Spike Concentration (μg/L)[b]	Analytical Result (μg/L)[b]	Spike Recovery
Soil[a]	6.25 ± 0.19[c] mg/kg	5.9 mg/kg	94%
Intestinal solution	20	15	75%
Stomach solution (blank)	0	1 U	NA
Intestinal solution (blank)	0	1 U	NA

NA = Not applicable.

[a] NIST Standard Reference Material (SRM) 2711 Montana Soils

[b] Units μg/L unless otherwise noted.

[c] Percent recovery based on the mean value of 6.25 mg/kg.

TABLE 18. MERCURY *IN VITRO* RESULTS FOR AMES STREET SAMPLES

Sample ID	Time (hrs)	pH (s.u.)	Eh (mV)	Mercury Conc. of Substrate (mg/kg)	Mass of Soil Tested (g)	Conc. in Extract (mg/L)	Volume of Extract (L)	Calculated Mass of Mercury in Soil Mass Tested (mg)	Calculated Mass of Mercury in Extract (mg)	Mercury Bioaccessibility[a]
BS21 6-8 ft	0.5	2.6	573	6.5	1.5093	0.002	0.150	0.010	0.0003	3%
BS21 6-8 ft	1	2.3		6.5	1.5093	0.003	0.150	0.010	0.0005	5%
BS21 6-8 ft	3.3	6.7		6.5	1.5093	0.005	0.150	0.010	0.0008	8%
BS21 6-8 ft	5.3	7.1	406	6.5	1.5093	0.013	0.150	0.010	0.0020	20%
BS41 4-6 ft	0.5	2.8	576	246	1.4971	0.645	0.150	0.368	0.0968	26%
BS41 4-6 ft	1	2.5		246	1.4971	0.840	0.150	0.368	0.1260	34%
BS41 4-6 ft	3.3	6.8		246	1.4971	0.304	0.150	0.368	0.0456	12%
BS41 4-6 ft	5.3	7.0	405	246	1.4971	0.531	0.150	0.368	0.0797	22%
BS41 6 8 ft	0.5	2.5	528	5850	1.5039	1.24	0.150	8.798	0.1860	2%
BS41 6-8 ft	1	2.4		5850	1.5039	1.28	0.150	8.798	0.1920	2%
BS41 6-8 ft	3.1	6.7		5850	1.5039	0.665	0.150	8.798	0.0998	1%
BS41 6-8 ft	5.1	7.0	378	5850	1.5039	1.12	0.149	8.798	0.1669	2%
BS42 2-4 ft	0.5	2.4	538	3710	1.5051	1.09	0.150	5.584	0.1635	3%
BS42 2 4 ft	1	2.5		3710	1.5051	1.26	0.150	5.584	0.1890	3%
BS42 2-4 ft	3.1	7.0		3710	1.5051	0.77	0.150	5.584	0.1155	2%
BS42 2-4 ft	5.1	7.1	345	3710	1.5051	1.18	0.145	5.584	0.1711	3%
BS69 4-6 ft A	0.5	2.4	512	723 [b]	1.5047	0.146	0.150	1.088	0.0219	2%
BS69 4-6 ft A	1	2.6		723 [b]	1.5047	0.169	0.150	1.088	0.0254	2%
BS69 4-6 ft A	2.8	6.8		723 [b]	1.5047	0.156	0.150	1.088	0.0234	2%
BS69 4-6 ft A	4.8	NA	394	723 [b]	1.5047	0.237	0.141	1.088	0.0334	3%
BS69 4-6 ft B	0.5	2.3	513	723 [b]	1.5048	0.162	0.150	1.088	0.0243	2%
BS69 4-6 ft B	1	2.6		723 [b]	1.5048	0.188	0.150	1.088	0.0282	3%
BS69 4 6 ft B	2.8	6.7		723 [b]	1.5048	0.175	0.150	1.088	0.0263	2%
BS69 4-6 ft B	4.8	NA	400	723 [b]	1.5048	0.292	0.130	1.088	0.0380	3%
BS69 4-6 ft C	0.5	2.5	513	723 [b]	1.5068	0.151	0.150	1.089	0.0227	2%
BS69 4-6 ft C	1	2.5		723 [b]	1.5068	0.174	0.150	1.089	0.0261	2%
BS69 4-6 ft C	2.8	6.9		723 [b]	1.5068	0.161	0.150	1.089	0.0242	2%
BS69 4-6 ft C	4.8	NA	400	723 [b]	1.5068	0.320	0.132	1.089	0.0422	4%
BS69 6 8 ft	0.5	2.6	556	17.3	1.5004	0.004	0.150	0.026	0.0006	2%
BS69 6 8 ft	1	2.6		17.3	1.5004	0.003	0.150	0.026	0.0005	2%
BS69 6-8 ft	3	7.0		17.3	1.5004	0.017	0.150	0.026	0.0026	10%
BS69 6 8 ft	5	7.1	398	17.3	1.5004	0.026	0.146	0.026	0.0038	15%

TABLE 18. (cont.)

Sample ID	Time (hrs)	pH (s.u.)	Eh (mV)	Mercury Conc. of Substrate (mg/kg)	Mass of Soil Tested (g)	Conc. in Extract (mg/L)	Volume of Extract (L)	Calculated Mass of Mercury in Soil Mass Tested (mg)	Calculated Mass of Mercury in Extract (mg)	Mercury Bioaccessibility[a]
BS70 0-2 ft	0.5	2.5	534	667	1.5051	0.865	0.150	1.004	0.1298	13%
BS70 0-2 ft	1	2.6		667	1.5051	1.14	0.150	1.004	0.1710	17%
BS70 0-2 ft	3.1	6.7		667	1.5051	0.610	0.150	1.004	0.0915	9%
BS70 0 2 ft	5.1	6.8	336	667	1.5051	1.08	0.143	1.004	0.1544	15%
BS72 4-6 ft	0.5	2.6	551	42.1	1.5136	0.006	0.150	0.064	0.0009	1%
BS72 4-6 ft	1	2.5		42.1	1.5136	0.008	0.150	0.064	0.0012	2%
BS72 4-6 ft	3	6.7		42.1	1.5136	0.025	0.150	0.064	0.0038	6%
BS72 4-6 ft	5	7.2	387	42.1	1.5136	0.107	0.145	0.064	0.0155	24%
BS74 0-2 ft	0.5	2.6	530	2.4	1.5083	0.001 U	0.150	0.004	0.0002	4%
BS74 0-2 ft	1	2.4		2.4	1.5083	0.001 U	0.150	0.004	0.0002	4%
BS74 0-2 ft	3	7.0		2.4	1.5083	0.003	0.150	0.004	0.0005	12%
BS74 0-2 ft	5	7.1	381	2.4	1.5083	0.006	0.145	0.004	0.0009	24%
SS03 0-2 ft	0.5	2.4	572	344	1.5014	0.249	0.150	0.516	0.0374	7%
SS03 0-2 ft	1	2.5		344	1.5014	0.280	0.150	0.516	0.0420	8%
SS03 0-2 ft	3.3	6.8		344	1.5014	0.259	0.150	0.516	0.0389	8%
SS03 0-2 ft	5.3	7.0	409	344	1.5014	0.425	0.148	0.516	0.0629	12%

[a] Values rounded to whole numbers.

[b] Values are the average of triplicate analyses.

TABLE 19. *IN VITRO* MERCURIC CHLORIDE SPIKE RESULTS

Sample ID	Time (hrs)	pH (s.u.)	Eh (mV)	Concentration of $HgCl_2$ Spike (μg/L)	Concentration in Extract (μg/L)	Mercury Bioaccessibility[a]
$HgCl_2$ Spike	0.5	2.4	476	210	137	65%
$HgCl_2$ Spike	1	2.3		210	156	74%
$HgCl_2$ Spike	3	7.1		210	98	47%
$HgCl_2$ Spike	5	7.3	390	210	89	42%
$HgCl_2$ Spike Dup	0.5	2.4	483	210	190	90%
$HgCl_2$ Spike Dup	1	2.3		210	185	88%
$HgCl_2$ Spike Dup	3	7.0		210	73	35%
$HgCl_2$ Spike Dup	5	7.1	362	210	143	68%
$HgCl_2$ Spike Trip	0.5	2.5	518	162	156	96%
$HgCl_2$ Spike Trip	1	2.4		162	165	102%
$HgCl_2$ Spike Trip	3	6.7		162	144	89%
$HgCl_2$ Spike Trip	5	7.0	438	162	145	90%
$HgCl_2$ Spike Quad	0.5	2.5	520	162	156	96%
$HgCl_2$ Spike Quad	1	2.4		162	160	99%
$HgCl_2$ Spike Quad	3	6.8		162	134	83%
$HgCl_2$ Spike Quad	5	7.1	447	162	142	88%

[a] Recovery of $HgCl_2$ spike was 78 percent based on average of all available data.

Appendix A

Mineralogy Determination Using Microprobe Analysis

QUALITY ASSURANCE REVIEW SUMMARY—MERCURY AND CONVENTIONAL ANALYTES IN SOIL AND SOIL EXTRACTS FOR THE AMES STREET SITE INVESTIGATION

A quality assurance review of laboratory data was completed for the following analytes:

- Mercury for 79 soil and 104 soil extract samples
- Acid-volatile sulfide (AVS), carbonate, and total organic carbon (TOC) in 17 soil samples, and total solids in 3 soil samples.
- TOC in three field blank samples.

All data are acceptable for the uses identified in the Site Investigation Work Plan (ABB 1996). Qualifiers have been added to some of the accepted results to indicate minor irregularities in the analyses that could affect the bias or precision of the reported value. Data qualifiers and their meanings are indicated in Table A-1. Analytical results and associated data qualifiers for the natural samples are presented in Tables 3, 7, and 14.

For the purposes of this data quality review, all samples not prepared by the analytical laboratory are designated as field samples. Field samples discussed in this data quality review summary include those collected by ABB and those prepared in the PTI Laboratory (Boulder, Colorado), which were submitted as blind quality control samples to the analytical laboratory.

SUMMARY OF QUALIFIED DATA

A total of 180 results for mercury and 57 results for conventional analytical parameters were reported by the analytical laboratory for this study. Of the 180 mercury results, 79 were reported from soil samples and 101 from soil extract samples. All but 3 of the 57 results for conventional analytes were from soil samples; these 3 TOC results were obtained from analyses of field blanks associated with the soil samples. Of the 237 total results, 170 mercury and 36 conventional analytical results were reported at concentrations above the method detection limits and 10 mercury and 21 conventional results were reported as undetected (the method detection limit was reported by the laboratory with a U qualifier). During the quality assurance review, 6 mercury and 17 AVS results were qualified as estimated (*J*), and 3 mercury results that were initially stated as detected were

restated as undetected (*U*). Appropriate descriptors and descriptor values were added to the qualifiers.

SAMPLE DIGESTION GROUPS

The data discussed in this reported comprised 12 sample digestion groups (SDGs). Of the 12 SDGs, 5 contained data for soil samples, and 7 contained data for soil extract samples. The data packages for these SDGs contained all documentation and data necessary to conduct the quality assurance review. The samples in each SDG and the analyses performed on each sample are summarized in Tables A-2 and A-3.

DATA QUALITY ASSESSMENT

The results for quality control procedures employed during sample analysis are discussed below, including data on completeness, holding times, analytical methods, instrument performance, bias, and precision. Data quality was assessed in terms of method-specific control limits. U.S. Environmental Protection Agency (EPA) functional guidelines (U.S. EPA 1994) were used for additional guidance during the quality assurance review. Results from quality control samples employed by the laboratory are summarized in Table A-4.

Completeness

The results reported by the laboratory were 100-percent complete and met the project DQO. No data were rejected during the quality assurance review.

Holding Times

Analyses for all mercury and conventional analytes were performed within the acceptable method-specified control limits.

Analytical Methods

All mercury analyses were completed according to EPA Method 7471 for solids, and EPA Method 7470 for liquids (U.S. EPA 1991), without modification. Conventional analytes were determined by the following methods: AVS by EPA draft method (Draft Analytical Method for Determination of Acid Volatile Sulfide in Sediment [U.S. EPA 1991]), and carbonate and TOC by ASTM methods D513-82 and D4129-82, respectively, as modified for soil samples (ASTM 1988). Analysis of TOC in the field blank samples

was performed by EPA Method 415.1 (U.S. EPA 1983). All analyses were performed by Columbia Analytical Services, Inc., Kelso, Washington.

The laboratory performed the analyses without modification, with the exception of the AVS analysis. This analysis should be performed on wet samples, because acid digestion is required to liberate the sulfide prior to analysis. However, because of the experimental design of this study, AVS was performed on soil samples that were dried prior to receipt at the laboratory. Drying soil samples prior to digestion may slightly reduce any unreacted sulfide presence in the wet soil, thus creating a slight negative bias in the results. However, during the quality assurance review, all AVS data were qualified as estimated, due to a low matrix spike recovery result (negative bias), and no additional data qualification is thought to be necessary to address the effect on these data of predrying the soils.

Instrument Performance

The results for the initial and continuing calibrations associated with the sample analyses are provided in Table A-4 and are described below. No changes in instrument performance were indicated during any analytical sequence that would have resulted in the degradation of data quality.

Initial Calibration

The initial calibrations completed for all mercury and conventional analyses met the criteria for acceptable performance and frequency of analysis.

Initial and Continuing Calibration Verification

The initial and continuing calibration verifications for all mercury and conventional analyses met the criteria for acceptable performance and frequency of analysis.

Initial and Continuing Calibration Blanks

The initial and continuing calibration blanks met the criteria for acceptable performance and frequency of analysis. No contamination was observed in any initial or continuing calibration blanks.

Method Blank Analyses

No target metals or conventional analytes were detected in the method blanks.

Accuracy

The accuracy of the analytical results is evaluated in the following sections in terms of analytical bias (laboratory control sample and matrix spike recoveries) and precision (laboratory duplicates).

Laboratory Control Sample Recoveries

The recoveries for all laboratory control samples (blank spikes), and the frequency of analysis, met the criteria for acceptable performance. Laboratory control sample recoveries for mercury and conventional analytes are summarized in Table A-4.

Matrix Spike Recoveries

The recoveries for the matrix spike samples, and the frequency of analysis, met the criteria for acceptable performance, with the exception of AVS, which had a recovery value of 49 percent (Table A-5). AVS results for all associated samples were qualified as estimated (*JS49*), and may exhibit a low bias. Matrix spike recoveries are summarized in Table A-5.

Precision

The results for all duplicate sample analyses, and the frequency of analysis, met the criteria for acceptable performance. A control limit of 20 relative percent difference (RPD) for soil extracts and 35 RPD for soil samples was used to evaluate the precision of the data. One matrix spike recovery result (SDG No. 1), although not a formal duplicate sample, demonstrated poor precision. The spiked result (1320 mg/kg) differed from the original sample result (826 mg/kg) by an RPD of 46 (Table A-5). Because the spike concentration (0.5 mg/kg) was so low compared to the sample concentration, the matrix spike sample is comparable to a laboratory duplicate. However, this sample is one of a triplicate set that were submitted to the laboratory to evaluate field sampling precision. Because the mercury result for this sample is within 35 RSD of the two split sample results (BS69 [4–6 ft] and BS69 [4–6 ft] dup.), no action was taken for the high RPD value of the sample and matrix spike sample result. A summary of all duplicate results for mercury and conventional analytes is summarized in Table A-6.

Analyte Quantification and Method Detection Limits

The calculations for analyte quantification and method detection limits were acceptable for all target analytes.

FIELD QUALITY CONTROL

The field quality control samples included field blanks, field replicate samples, and a reference material sample.

Field blanks were prepared and submitted with the soil and soil extract samples. One set of field blanks was submitted with the soil samples (one distilled water, one bottle, and one equipment blank [Table A-7]). Two additional blanks were submitted to the laboratory, one each for the soil and soil extract samples. Mercury results in the blanks were below the detection limit, with the exception of mercury in sample ASW020, which had a concentration of 5 μg/L (Table A-7). Results of the associated natural samples were compared to five times this amount, and any results below 25 μg/L were restated as undetected (*UBF5*). Results for two samples met this criterion (BS21 [6–8 ft], and BS74 [0–2 ft]; Table A-7) and were restated as undetected during the quality assurance review.

Field duplicates were analyzed for mercury only. Due to the inherent heterogeneity of soil samples, RPD (or relative standard deviation [RSD] when multiple split samples are analyzed) values of 50 or below are considered acceptable for field split samples. The RSD values for field triplicates (data in Table 3) were all below 50 percent, and therefore, were acceptable.

One reference material sample was submitted with the soil samples. This reference material (NIST 2711) was obtained from National Institute of Standards and Technology. The recovery of the reference material (94 percent) was acceptable (Table 12).

A summary of the results of the data quality control checks is presented in Table A-8.

REFERENCES

ABB. 1996. Final site investigation work plan. Ames Street site, Rochester, New York. Phase I. Appendices I and J. ABB Environmental Services, Inc., Rochester, New York.

ASTM. 1988. Annual book of ASTM standards. Vol. 04.08: Soil and rock, building stones; Geotextiles. American Society for Testing and Materials, Philadelphia, PA.

U.S. EPA. 1983. Methods for chemical analysis of water and wastes. EPA/600/4-79-020. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1991. Test methods for evaluating solid waste. Physical/chemical methods. SW-846. Third Edition (revised methods). U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

TABLE A-1. DEFINITIONS OF DATA FLAGS, QUALIFIERS, AND DESCRIPTIONS FOR INORGANIC DATA

Type	Description	Value
Flag[a]		
N	Laboratory spike sample results outside control limits	--
U	Reported result is less than instrument detection limit	--
*	Laboratory duplicate results outside control limits	--
C	Sample results qualified because of interference (graphite furnace atomic absorption [GFAA] analytical spike or inductively coupled plasma [ICP] serial dilution)	--
M	Duplicate injection precision for GFAA analysis outside control limits	--
W	Postdigestion spike for GFAA outside control limits	--
|	Correlation coefficient for method of standard additions (MGA) for GFAA less than 0.995	--
S	The reported value was determined by MSA	
Qualifier		
R[b]	Rejected	--
U[b]	Undetected	--
J[b]	Estimated	--
A[c]	Justified as enforcement quality data	
Description[d]		
5%	Qualified because matrix spike control limits are exceeded	Percent recovery of matrix spike
5X	Qualified because frequency of matrix spike sample analysis is not satisfied	No descriptor value
D%	Qualified because duplicate relative percent difference (RFD) control limits are exceeded	RPD of duplicate analysis
DX	Qualified because frequency of duplicate sample analysis is not satisfied	No descriptor value
E%	Qualified because ICP serial dilution control limits are exceeded	Percent difference of ICP serial dilution
EX	Qualified because frequency of ICP serial dilution is not satisfied	No descriptor value
HT	Qualified because holding time is exceeded	Holding time in days

TABLE A-1 (cont.)

Type	Description	Value
MC	Qualified because correlation coefficient of MSA results is less than 0.995	Correlation coefficient of MSA
L%	Qualified because laboratory control sample (LCS) control limits are exceeded	Percent recovery of LCS
LX	Qualified because frequency of LCS analysis is not satisfied	No descriptor value
I%	Qualified because of ICP interference check sample (ICS) results	Percent recovery of ICS
IX	Qualified because frequency of analysis of ICP ICS is not satisfied	No descriptor value
GS	Qualified because GFAA analytical spike result control limits are exceeded	Analytical spike percent recovery
BC	Qualified because of calibration blank results	Calibration blank value
BP	Qualified because of laboratory blank results	Laboratory blank value
BX	Qualified because frequency of preparation blank analysis is not satisfied	No descriptor value
B	Qualified because of field or laboratory blank results	No descriptor value
K	Qualified because of negative blank results	Absolute value of the negative blank result
C%	Qualified because of instrument calibration (i.e., initial calibration verification, continuing calibration verification, frequency of calibration)	Percent recovery of continuing calibration verification or initial calibration verification
CX	Qualified because frequency of analysis of calibration samples is not satisfied	No descriptor value
CC	Qualified because correlation coefficient of instrument calibration is exceeded	Correlation coefficient
CL	Qualified because linear range of calibration is exceeded	No descriptor value
EU	Qualified because of an unexplained interference	No descriptor value
Q	Qualified because of other QC violators	No descriptor value

[a] Defined in U.S. EPA. 1988. *Contract Laboratory Program statement of work. Inorganic analysis, multi-media, multi-concentration*. July 1988. SOW. No. 788. U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Las Vegas, NV. (Flags are assigned by the laboratory.)

[b] Defined in U.S. EPA. (1985). *Laboratoyr data validation: functional guidelines for evaluating inorganic analyses.* U.S. Environmental Protection Agency, Washington, DC. Also defined in Viar & Co. (eds). 1988 (revision). *Laboratory data validation: functional guidelines for evaluating inorganics analyses.* Prepared by U.S. Environmental Protection Agency Work Group. Prepared for the U.S. Environmental Protection Agency, Hazardous Site Evaluation Division, Washington, DC.

[c] Justified as enforcement quality data as defined in Administrative Order on Consent.

[d] Defined in MDHES. 1990. *Clark Fork Data System reference.* Montana Department of Health and Environmental Sciences, Solid and Hazardous Waste Bureau. Montana State Library Natural Resource Information System. Helena, MT. (The descriptions provide the data user with information concerning the qualification of data.)

TABLE A-2. SAMPLE DESCRIPTIONS FOR EACH SOIL SAMPLE DELIVERY GROUP

Job No.	Sample Digestion Group Number	Sampling Date	Analyte	Method	Description
K9602361	1	4/18/96	Mercury	EPA 7471	In vitro soil samples
K9602556	2	4/18/96	Mercury	EPA 7471	In vitro soil samples
K9602527	3	4/29/96	Mercury	EPA 7471	Sequential extraction soil samples
K9602669	4	4/29/96	Mercury	EPA 7471	Sequential extraction soil samples
K9602433	5	4/18/96	Carbonate	ASTM D513-82M	Sequential extraction samples
K9602433	5	4/18/96	Acid-volatile sulfide	EPA Draft Aug. 1991	Sequential extraction samples
K9602433	5	4/18/96	Carbon, total organic	ASTM D4129-82 M	Sequential extraction samples
K9602433	5	4/18/96	TOC	EPA 415.1	Sequential extraction samples
K9602433	5	4/18/96	Mercury	EPA 7471	Sequential extraction samples

TABLE A-3. SAMPLE DESCRIPTIONS FOR EACH AQUEOUS SAMPLE DELIVERY GROUP

Job No.	Sample Digestion Group Number	Sampling Date	Analyte	Method	Description
K9602360	6	4/18/96	Mercury	EPA 7470	In vitro aqueous samples
K9602554	7	4/25/96	Mercury	EPA 7470	In vitro aqueous samples
K9602552	8	4/25/96	Mercury	EPA 7470	In vitro aqueous samples
K9602359	9	4/23/96	Mercury	EPA 7470	In vitro aqueous samples
K9603249	10	6/3/96	Mercury	EPA 7470	In vitro aqueous samples
K9602434	11	4/19/96	Mercury	EPA 7470	Sequential extraction aqueous samples
K9602529	12	4/23/96	Mercury	EPA 7470	Sequential extraction aqueous samples

TABLE A-4. LABORATORY CONTROL SAMPLE RESULTS

Sample ID	Units	Analyte	Reference Value	Percent Recovery	QC Limits	True Value
Lab Control	mg/kg	Mercury	3.8		1.47-5.15	
ICV	µg/L	Mercury	5.14	103		5.00
CCV1	µg/L	Mercury	5.03	101		5.00
CCV2	µg/L	Mercury	5.02	100		5.00
CCV3	µg/L	Mercury	5.00	100		5.00
CCB1	µg/L	Mercury	0.001 U			
CCB2	µg/L	Mercury	0.001 U			
CCB3	µg/L	Mercury	0.001 U			
Lab Control	mg/kg	Mercury	3.19		1.47-5.15	
ICV	µg/L	Mercury	5.30	106		5.00
CCV1	µg/L	Mercury	5.01	100		5.00
CCV2	µg/L	Mercury	4.82	96		5.00
CCV3	µg/L	Mercury	4.74	95		5.00
CCV4	µg/L	Mercury	4.69	94		5.00
CCV5	µg/L	Mercury	4.84	97		5.00
CCV6	µg/L	Mercury	4.64	93		5.00
CCV7	µg/L	Mercury	4.53	91		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
CCB5	µg/L	Mercury	0.1 U			
CCB6	µg/L	Mercury	0.1 U			
CCB7	µg/L	Mercury	0.1 U			
Lab Control	µg/L	Mercury	4.9	98		5.00
ICV	µg/L	Mercury	5.13	103		5.00
CCV1	µg/L	Mercury	5.06	101		5.00
CCV2	µg/L	Mercury	5.02	100		5.00
CCV3	µg/L	Mercury	4.94	99		5.00
CCV4	µg/L	Mercury	4.78	96		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
Lab Control	µg/L	Mercury	4.78	96	80-120%	
ICV	µg/L	Mercury	5.25	105		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
CCB5	µg/L	Mercury	0.1 U			
CCV1	µg/L	Mercury	5.1	102		5.00
CCV2	µg/L	Mercury	4.96	99		5.00
CCV3	µg/L	Mercury	5.00	100		5.00
CCV4	µg/L	Mercury	4.92	98		5.00
CCV5	µg/L	Mercury	4.96	99		5.00
Lab Control	µg/L	Mercury	5.1	102	80-120%	
ICV	µg/L	Mercury	5.25	105		5.00
CCV1	µg/L	Mercury	5.1	102		5.00
CCV2	µg/L	Mercury	4.96	99		5.00
CCV3	µg/L	Mercury	5.00	100		5.00
CCV4	µg/L	Mercury	4.92	98		5.00
CCV5	µg/L	Mercury	4.96	99		5.00
CCV6	µg/L	Mercury	4.91	98		5.00
CCV7	µg/L	Mercury	4.91	98		5.00

TABLE A-4. (cont.)

Sample ID	Units	Analyte	Reference Value	Percent Recovery	QC Limits	True Value
CCV8	µg/L	Mercury	4.95	99		5.00
CCV9	µg/L	Mercury	4.97	99		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
CCB5	µg/L	Mercury	0.1 U			
CCB6	µg/L	Mercury	0.1 U			
CCB7	µg/L	Mercury	0.1 U			
CCB8	µg/L	Mercury	0.1 U			
CCB9	µg/L	Mercury	0.1 U			
Lab Control	µg/L	Mercury	4.85	97	80-120%	5.00
ICV	µg/L	Mercury	4.84	97		5.00
CCV1	µg/L	Mercury	5.04	100		5.00
CCV2	µg/L	Mercury	5.11	96		5.00
CCV3	µg/L	Mercury	5.12	95		5.00
CCV4	µg/L	Mercury	5.08	94		5.00
CCV5	µg/L	Mercury	5.11	97		5.00
CCV6	µg/L	Mercury	4.95	99		5.00
CCB1	µg/L	Mercury	0.5 U			
CCB2	µg/L	Mercury	0.5 U			
CCB3	µg/L	Mercury	0.5 U			
CCB4	µg/L	Mercury	0.5 U			
CCB5	µg/L	Mercury	0.5 U			
Lab Control	µg/L	Mercury	4.84	97	80-120%	5.00
ICV	µg/L	Mercury	5.19	104		5.00
CCV1	µg/L	Mercury	5.05	101		5.00
CCV2	µg/L	Mercury	4.80	96		5.00
CCV3	µg/L	Mercury	4.76	95		5.00
CCV4	µg/L	Mercury	5.13	103		5.00
CCV5	µg/L	Mercury	5.05	101		5.00
CCV6	µg/L	Mercury	5.03	101		5.00
CCV7	µg/L	Mercury	4.98	100		5.00
CCV8	µg/L	Mercury	4.90	98		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
CCB5	µg/L	Mercury	0.1 U			
CCB6	µg/L	Mercury	0.1 U			
CCB7	µg/L	Mercury	0.1 U			
CCB8	µg/L	Mercury	0.1 U			
Lab Control	µg/L	Mercury	3.12		1.47-5.15	
ICV	µg/L	Mercury	5.13	103		5.00
CCV1	µg/L	Mercury	4.98	100		5.00
CCV2	µg/L	Mercury	4.91	98		5.00
CCV3	µg/L	Mercury	4.97	99		5.00
CCV4	µg/L	Mercury	4.99	100		5.00
CCV5	µg/L	Mercury	5.03	101		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
CCB5	µg/L	Mercury	0.1 U			
Lab Control	µg/L	Mercury	3.30		1.47-5.15	

TABLE A-4. (cont.)

Sample ID	Units	Analyte	Reference Value	Percent Recovery	QC Limits	True Value
ICV	µg/L	Mercury	5.10	102		5.00
CCV1	µg/L	Mercury	5.18	104		5.00
CCV2	µg/L	Mercury	5.37	107		5.00
CCV3	µg/L	Mercury	5.31	106		5.00
CCV4	µg/L	Mercury	5.27	105		5.00
CCV5	µg/L	Mercury	5.36	107		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
CCB5	µg/L	Mercury	0.1 U			
Lab Control	%	Carbonate	62.8	105		60.0
Lab Control	%	Carbonate	59.5	99		60.0
CCV1	%	Carbonate	103	103		100
CCV2	%	Carbonate	101	101		100
CCV1	%	Carbonate	101	101		100
CCV2	%	Carbonate	101	101		100
CCV3	%	Carbonate	101	101		100
CCB1	%	Carbonate	0.005 U			
CCB2	%	Carbonate	0.005 U			
CCB1	%	Carbonate	0.005 U			
CCB2	%	Carbonate	0.005 U			
CCB3	%	Carbonate	0.005 U			
Lab Control	mg/kg	Acid volatile sulfide	0.41	82		0.51
CCV1	mg/kg	Acid volatile sulfide	0.67	99		0.68
CCV2	mg/kg	Acid volatile sulfide	0.66	97		0.68
CCV3	mg/kg	Acid volatile sulfide	0.66	97		0.68
CCB1	mg/kg	Acid volatile sulfide	0.05 U			
CCB2	mg/kg	Acid volatile sulfide	0.05 U			
CCB3	mg/kg	Acid volatile sulfide	0.05 U			
Lab Control	%	TOC	0.62	100		0.62
Lab Control	%	TOC	0.63	102		0.62
CCV2	%	TOC	19.9	100		20.0
CCV3	%	TOC	20.0	100		20.0
CCV1	%	TOC	20.1	100		20.0
CCV2	%	TOC	20.3	102		20.0
CCV3	%	TOC	20.1	100		20.0
CCB2	%	TOC	0.05 U			
CCB3	%	TOC	0.05 U			
CCB1	%	TOC	0.05 U			
CCB2	%	TOC	0.05 U			
CCB3	%	TOC	0.05 U			
Lab Control	mg/L	TOC	9.9	85		11.6
Lab Control	mg/L	TOC	27.4	94		29.0
CCV1	mg/L	TOC	24.1	96		25.0
CCV2	mg/L	TOC	23.8	95		25.0
CCB1	mg/L	TOC	0.5 U			
CCB2	mg/L	TOC	0.5 U			
Lab Control	mg/kg	Mercury	3.27		1.47-5.15	
ICV	µg/L	Mercury	5.30	106		5.00
CCV1	µg/L	Mercury	5.01	100		5.00
CCV2	µg/L	Mercury	4.82	96		5.00
CCV3	µg/L	Mercury	4.74	95		5.00
CCV4	µg/L	Mercury	4.69	94		5.00
CCV5	µg/L	Mercury	4.84	97		5.00

TABLE A-4. (cont.)

Sample ID	Units	Analyte	Reference Value	Percent Recovery	QC Limits	True Value
CCB1	µg/L	Mercury	0.5 U			
CCB2	µg/L	Mercury	0.5 U			
CCB3	µg/L	Mercury	0.5 U			
CCB4	µg/L	Mercury	0.5 U			
CCB5	µg/L	Mercury	0.5 U			
Lab Control	µg/L	Mercury	4.6	92		5.00
ICV	µg/L	Mercury	4.77	95		5.00
CCV1	µg/L	Mercury	5.06	101		5.00
CCV2	µg/L	Mercury	4.93	99		5.00
CCV3	µg/L	Mercury	4.92	98		5.00
CCV4	µg/L	Mercury	4.83	97		5.00
CCV5	µg/L	Mercury	4.71	94		5.00
CCB1	µg/L	Mercury	0.1 U			
CCB2	µg/L	Mercury	0.1 U			
CCB3	µg/L	Mercury	0.1 U			
CCB4	µg/L	Mercury	0.1 U			
CCB5	µg/L	Mercury	0.1 U			

U = Not detected; value represents detection limit.
ICV = Initial calibration verification
CCB = Continuing calibration blank
CCV = Continuing calibration verification

TABLE A-5. MATRIX SPIKE SAMPLE RESULTS

Sample ID	Units	Analyte	Spike Concentration	Sample Concentration	Spiked Sample Concentration	Percent Recovery
S94801	mg/kg	Mercury	0.5	1320	826	63
S94819	mg/kg	Mercury	0.46	5850	4760	81
W94805	µg/L	Mercury	1.0	237	242	102
W94825	µg/L	Mercury	10	1090	1130	103
W94823	µg/L	Mercury	10	1120	1120	99
W94715	µg/L	Mercury	10	81	82	90
W94739	µg/L	Mercury	1	741	778	105
S94754	mg/kg	Mercury	0.46	7010	9470	135
S94774	mg/kg	Mercury	0.5	50.0	40.2	80
S95752	%	Carbonate	1.60	2.41	4.00	99
S95774	%	Carbonate	2.00	4.27	6.32	103
S95752	mg/kg	Acid-volatile sulfide	50.5	5 U	25	49
S95752	%	TOC	2.40	1.19	3.54	98
S95772	%	TOC	2.85	6.00	8.76	97
W94708	mg/L	TOC	25	0.5 U	24.9	100
AIW071	µg/L	Mercury	10	145	156	101

U = Not detected; value represents detection limit.

TABLE A-6. LABORATORY DUPLICATE SAMPLE RESULTS

Sample ID	Units	Analyte	Sample Result	Duplicate Sample Result	Average	RPD/RSD
S94801	mg/kg	Mercury	826	1060	943	25
S94819	mg/kg	Mercury	5850	6000	5925	2
W94805	µg/L	Mercury	237	238	238	<1
W94821	µg/L	Mercury	1280	1280	1280	<1
W94823	µg/L	Mercury	1120	1120	1120	<1
W94715	µg/L	Mercury	81	83	82	2
W94738	µg/L	Mercury	482	471	477	2
S94754	mg/kg	Mercury	7010	5970	6490	16
S94774	mg/kg	Mercury	50.0	40.1	45	22
S95752	%	Carbonate	2.41	2.36	2.39	2
S95774	%	Carbonate	4.27	4.31	4.29	<1
S95752	mg/kg	Acid-volatile sulfide	5 U	5 U	5 U	--
S95752	%	TOC	1.19	1.18	1.19	<1
S95772	%	TOC	6.00	6.04	6.02	<1
W94708	mg/L	TOC	0.5 U	0.5 U	0.5 U	--
S95752	mg/kg	Mercury	3110	3140	3125	<1
S95752	mg/kg	Mercury	3110	2400	2755	26
AIW071	µg/L	Mercury	145	140	143	4

U = Not detected; value represents detection limit.

TABLE A-7. MERCURY SEQUENTIAL EXTRACTION BLANK RESULTS

Matrix	Blank Concentration (μg/L)
Deionized Water	0.5 U
Equipment Blank	0.5 U
Bottle Blank	0.02 U[a]
Blank (Step 1)	5
Blank (Step 2)	1 U

U = Not detected; value represents detection limit.

[a] Units in μg.

TABLE A-8. SUMMARY OF QUALITY CONTROL CHECKS

Quality Control Check	Status	Comment
Completeness	Acceptable	
Holding times	Acceptable	
Analytical methods	Acceptable	
Instrument performance		
Initial calibration	Acceptable	
Initial and continuing calibration verification	Acceptable	
Initial and continuing calibration blanks	Acceptable	
Method blanks	Acceptable	
Bias		
Laboratory control samples	Acceptable	
Matrix spike samples	Acceptable: 17 AVS results affected	See Matrix Spike Recoveries section
Precision	Acceptable	
Quantification of results	Acceptable	
Detection limits	Acceptable	
Field quality control samples	Acceptable; 3 mercury effected	See Field Quality Control section
OVERALL ASSESSMENT	**Acceptable**	**No data rejected during the quality assurance review**

APPENDIX B
IDENTIFICATION OF SAMPLES INCLUDED IN ANALYTICAL DATA SETS

TABLE B-1
SUMMARY OF SAMPLES INCLUDED
IN IN VOC DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
11/12-01; TANK		0	04-Dec-95	VOA	ug/kg
11/12-02; TANK		0	04-Dec-95	VOA	ug/kg
11/12-03; TANK		0	04-Dec-95	VOA	ug/kg
11/12-04; TANK		0	04-Dec-95	VOA	ug/kg
11/12-05; TANK		0	04-Dec-95	VOA	ug/kg
11/12-06; TANK		0	04-Dec-95	VOA	ug/kg
17/19-08-01B;TANK		8	01-Jan-96	VOA	ug/kg
17/19-08-02B;TANK		8	01-Jan-96	VOA	ug/kg
17/19-01;TANK		1	08-Dec-95	VOA	ug/kg
17/19-02;TANK		1	08-Dec-95	VOA	ug/kg
17/19-03;TANK		1	08-Dec-95	VOA	ug/kg
S-TANK1-04		1	29-Nov-95	VOA	ug/kg
S-TANK1-05		1	29-Nov-95	VOA	ug/kg
S-TANK-COMP.		1	29-Nov-95	VOA	ug/kg
BS0108XX		8	11-Mar-96	VOA	ug/kg
BS0110XX		10	11-Mar-96	VOA	ug/kg
BS0308XX		8	13-Mar-96	VOA	ug/kg
BS0320XX		20	13-Mar-96	VOA	ug/kg
BS0408XX		8	13-Mar-96	VOA	ug/kg
BS0412XX		12	13-Mar-96	VOA	ug/kg
BS0412XX	DUP	12	13-Mar-96	VOA	ug/kg
BS0420XX		20	13-Mar-96	VOA	ug/kg
BS0508XX		8	12-Mar-96	VOA	ug/kg
BS0522XX		22	12-Mar-96	VOA	ug/kg
BS0604XX		4	12-Mar-96	VOA	ug/kg
BS1006XX		6	17-Mar-96	VOA	ug/kg
BS1020XX		20	17-Mar-96	VOA	ug/kg
BS1210XX		10	16-Mar-96	VOA	ug/kg
BS1220XX		20	16-Mar-96	VOA	ug/kg
BS1308XX		8	16-Mar-96	VOA	ug/kg
BS1316XX		16	16-Mar-96	VOA	ug/kg
BS1406XX		6	15-Mar-96	VOA	ug/kg
BS1416XX		16	15-Mar-96	VOA	ug/kg
BS1506XX		6	15-Mar-96	VOA	ug/kg
BS1512XX		12	15-Mar-96	VOA	ug/kg
BS1608XX		8	15-Mar-96	VOA	ug/kg
BS1614XX		14	15-Mar-96	VOA	ug/kg
BS2106XX		6	26-Mar-96	VOA	ug/kg
BS2118XX		18	26-Mar-96	VOA	ug/kg
BS2908XX		8	18-Mar-96	VOA	ug/kg
BS2922XX		22	18-Mar-96	VOA	ug/kg

TABLE B-1
SUMMARY OF SAMPLES INCLUDED
IN IN VOC DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS3012XX		12	18-Mar-96	VOA	ug/kg
BS3022XX		22	18-Mar-96	VOA	ug/kg
BS3906XX		6	26-Mar-96	VOA	ug/kg
BS3918XX		18	26-Mar-96	VOA	ug/kg
BS4408XX		8	27-Mar-96	VOA	ug/kg
BS4412XX		12	27-Mar-96	VOA	ug/kg
BS4420XX		20	27-Mar-96	VOA	ug/kg
BS4506XX		6	27-Mar-96	VOA	ug/kg
BS4508XX		8	27-Mar-96	VOA	ug/kg
BS4516XX		16	27-Mar-96	VOA	ug/kg
BS4616XX		16	28-Mar-96	VOA	ug/kg
BS4628XX		28	28-Mar-96	VOA	ug/kg
BS4710XX		10	28-Mar-96	VOA	ug/kg
BS4710XX	DUP	10	28-Mar-96	VOA	ug/kg
BS4716XX		16	28-Mar-96	VOA	ug/kg
BS4904XX		4	29-Mar-96	VOA	ug/kg
BS4924XX		24	29-Mar-96	VOA	ug/kg
BS5006XX		6	29-Mar-96	VOA	ug/kg
BS5016XX		16	29-Mar-96	VOA	ug/kg
BS5025XX		25.1	29-Mar-96	VOA	ug/kg
BS5104XX		4	31-Mar-96	VOA	ug/kg
BS5108XX		8	31-Mar-96	VOA	ug/kg
BS5118XX		18	31-Mar-96	VOA	ug/kg
BS5210XX		10	31-Mar-96	VOA	ug/kg
BS5218XX		18	31-Mar-96	VOA	ug/kg
BS5220XX		20	31-Mar-96	VOA	ug/kg
BS5224XX		24	31-Mar-96	VOA	ug/kg
BS5310XX		10	01-Apr-96	VOA	ug/kg
BS5318XX		18	01-Apr-96	VOA	ug/kg
BS5324XX		24.4	01-Apr-96	VOA	ug/kg
BS5324XX	DUP	24.4	01-Apr-96	VOA	ug/kg
BS5410XX		10	30-Mar-96	VOA	ug/kg
BS5412XX		12	30-Mar-96	VOA	ug/kg
BS5426XX		25.5	30-Mar-96	VOA	ug/kg
BS5502XX		2	31-Mar-96	VOA	ug/kg
BS5512XX		12	31-Mar-96	VOA	ug/kg
BS5520XX		20	31-Mar-96	VOA	ug/kg
BS5710XX		10	30-Mar-96	VOA	ug/kg
BS5710XX	DUP	10	30-Mar-96	VOA	ug/kg
BS5722XX		22	30-Mar-96	VOA	ug/kg
BS5818XX		18	30-Mar-96	VOA	ug/kg

TABLE B-1
SUMMARY OF SAMPLES INCLUDED
IN IN VOC DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS5818XX	DUP	18	30-Mar-96	VOA	ug/kg
BS5918XX		18	01-Apr-96	VOA	ug/kg
BS6018XX		18	30-Mar-96	VOA	ug/kg
BS6112XX		12	31-Mar-96	VOA	ug/kg
BS6116XX		16	31-Mar-96	VOA	ug/kg
BS6118XX		18	31-Mar-96	VOA	ug/kg
BS6122XX		22	31-Mar-96	VOA	ug/kg
BS6218XX		18	31-Mar-96	VOA	ug/kg
BS6312XX		12	31-Mar-96	VOA	ug/kg
BS6314XX		14	31-Mar-96	VOA	ug/kg
BS6318XX		18	31-Mar-96	VOA	ug/kg
BS6322XX		22	31-Mar-96	VOA	ug/kg
BS6418XX		18	01-Apr-96	VOA	ug/kg
BS6514XX		14	01-Apr-96	VOA	ug/kg
BS6518XX		18	01-Apr-96	VOA	ug/kg
BS7016XX		16	02-Apr-96	VOA	ug/kg
BS7022XX		22	02-Apr-96	VOA	ug/kg
BS7508XX		8	09-Apr-96	VOA	ug/kg
BS7512XX		12	09-Apr-96	VOA	ug/kg
BS7608XX		8	09-Apr-96	VOA	ug/kg
BS7610XX		10	09-Apr-96	VOA	ug/kg
BS7620XX		20	09-Apr-96	VOA	ug/kg
BS7708XX		8	09-Apr-96	VOA	ug/kg
BS7712XX		12	09-Apr-96	VOA	ug/kg
BS7716XX		16	09-Apr-96	VOA	ug/kg
BS7720XX		20	09-Apr-96	VOA	ug/kg
BS7808XX		8	09-Apr-96	VOA	ug/kg
BS7812XX		12	09-Apr-96	VOA	ug/kg
BS7816XX		16	09-Apr-96	VOA	ug/kg
BS7820XX		20	09-Apr-96	VOA	ug/kg
BS7820XX	DUP	20	09-Apr-96	VOA	ug/kg
42SSXX1		6	05-May-93	VOA	ug/kg
43SSXX6		5	05-May-93	VOA	ug/kg

TABLE B-2					
SUMMARY OF SAMPLES INCLUDED					
IN INORGANICS AND CYANIDE DATA SET - SOIL					
HUMAN HEALTH RISK ASSESSMENT					
AMES STREET SITE					
ROCHESTER, NY					
smpl_id	stype	depth	s_date	type	
23SSXX1		1	08-May-93	INOR	mg/kg
34SSXX1		1	06-May-93	INOR	mg/kg
34SSXX4		2	06-May-93	INOR	mg/kg
34SSXX8		1	08-May-93	INOR	mg/kg
42SSXX1		6	05-May-93	INOR	mg/kg
BS2508XX		0	18-Mar-96	INOR	mg/kg
BS2512XX		0	18-Mar-96	INOR	mg/kg
BS2606XX		0	18-Mar-96	INOR	mg/kg
BS2608XX		0	19-Mar-96	INOR	mg/kg
BS2608XX	DUP	0	19-Mar-96	INOR	mg/kg
BS2704XX		0	19-Mar-96	INOR	mg/kg
BS2708XX		0	19-Mar-96	INOR	mg/kg
BS2806XX		0	19-Mar-96	INOR	mg/kg
BS2812XX		0	19-Mar-96	INOR	mg/kg
BS2908XX		0	18-Mar-96	INOR	mg/kg
BS2916XX		0	18-Mar-96	INOR	mg/kg
BS3008XX		0	18-Mar-96	INOR	mg/kg
BS3012XX		0	18-Mar-96	INOR	mg/kg
BS3108XX		0	18-Mar-96	INOR	mg/kg
BS3112XX		0	18-Mar-96	INOR	mg/kg
KTSSXX5		1	28-Sep-93	INOR	mg/kg
KTSSXX7		1	28-Sep-93	INOR	mg/kg
LTSSXX1		1	08-May-93	INOR	mg/kg
LTSSXX2		1	08-May-93	INOR	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
02SSXX1X1		1	10-May-93	Hg	mg/kg
23SSXX1X1		1	08-May-93	Hg	mg/kg
34SSXX8 X1		1	08-May-93	Hg	mg/kg
42SSXX1X6		6	05-May-93	Hg	mg/kg
42SSXX3X3		3	05-May-93	Hg	mg/kg
BS0104XX		4	11-Mar-96	Hg	mg/kg
BS0106XX		6	11-Mar-96	Hg	mg/kg
BS0108XX		8	11-Mar-96	Hg	mg/kg
BS0110XX		10	11-Mar-96	Hg	mg/kg
BS0110XX	DUP	10	11-Mar-96	Hg	mg/kg
BS0112XX		12	11-Mar-96	Hg	mg/kg
BS0114XX		14	11-Mar-96	Hg	mg/kg
BS0116XX		16	11-Mar-96	Hg	mg/kg
BS0118XX		18	11-Mar-96	Hg	mg/kg
BS0120XX		20	11-Mar-96	Hg	mg/kg
BS0122XX		22	11-Mar-96	Hg	mg/kg
BS0124XX		24	11-Mar-96	Hg	mg/kg
BS0126XX		25.5	11-Mar-96	Hg	mg/kg
BS0204XX		4	12-Mar-96	Hg	mg/kg
BS0206XX		6	12-Mar-96	Hg	mg/kg
BS0208XX		8	12-Mar-96	Hg	mg/kg
BS0210XX		10	12-Mar-96	Hg	mg/kg
BS0212XX		12	12-Mar-96	Hg	mg/kg
BS0214XX		14	12-Mar-96	Hg	mg/kg
BS0216XX		16	12-Mar-96	Hg	mg/kg
BS0218XX		18	12-Mar-96	Hg	mg/kg
BS0220XX		20	12-Mar-96	Hg	mg/kg
BS0222XX		22	12-Mar-96	Hg	mg/kg
BS0224XX		24	12-Mar-96	Hg	mg/kg
BS0225XX		24.8	12-Mar-96	Hg	mg/kg
BS0304XX		4	13-Mar-96	Hg	mg/kg
BS0306XX		6	13-Mar-96	Hg	mg/kg
BS0308XX		8	13-Mar-96	Hg	mg/kg
BS0310XX		10	13-Mar-96	Hg	mg/kg
BS0312XX		12	13-Mar-96	Hg	mg/kg
BS0314XX		14	13-Mar-96	Hg	mg/kg
BS0314XX	DUP	14	13-Mar-96	Hg	mg/kg
BS0316XX		16	13-Mar-96	Hg	mg/kg
BS0318XX		18	13-Mar-96	Hg	mg/kg
BS0320XX		20	13-Mar-96	Hg	mg/kg
BS0320XX		20.2	13-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS0604XX		4	12-Mar-96	Hg	mg/kg
BS0606XX		6	12-Mar-96	Hg	mg/kg
BS0610XX		10	12-Mar-96	Hg	mg/kg
BS0612XX		12	12-Mar-96	Hg	mg/kg
BS0614XX		14	12-Mar-96	Hg	mg/kg
BS0614XX	DUP	14	12-Mar-96	Hg	mg/kg
BS0616XX		16	12-Mar-96	Hg	mg/kg
BS0618XX		18	12-Mar-96	Hg	mg/kg
BS0620XX		20	12-Mar-96	Hg	mg/kg
BS0621XX		20.8	12-Mar-96	Hg	mg/kg
BS0704XX		4	13-Mar-96	Hg	mg/kg
BS0706XX		6	13-Mar-96	Hg	mg/kg
BS0708XX		8	13-Mar-96	Hg	mg/kg
BS0710XX		10	13-Mar-96	Hg	mg/kg
BS0712XX		12	13-Mar-96	Hg	mg/kg
BS0714XX		14	13-Mar-96	Hg	mg/kg
BS0716XX		16	13-Mar-96	Hg	mg/kg
BS0718XX		18	13-Mar-96	Hg	mg/kg
BS0720XX		20	13-Mar-96	Hg	mg/kg
BS0722XX		22	13-Mar-96	Hg	mg/kg
BS1002XX		2	17-Mar-96	Hg	mg/kg
BS1004XX		4	17-Mar-96	Hg	mg/kg
BS1006XX		6	17-Mar-96	Hg	mg/kg
BS1008XX		8	17-Mar-96	Hg	mg/kg
BS1010XX		10	17-Mar-96	Hg	mg/kg
BS1012XX		12	17-Mar-96	Hg	mg/kg
BS1014XX		14	17-Mar-96	Hg	mg/kg
BS1018XX		18	17-Mar-96	Hg	mg/kg
BS1020XX		20	17-Mar-96	Hg	mg/kg
BS1021XX		20.7	17-Mar-96	Hg	mg/kg
BS1202XX		2	16-Mar-96	Hg	mg/kg
BS1204XX		4	16-Mar-96	Hg	mg/kg
BS1206XX		6	16-Mar-96	Hg	mg/kg
BS1208XX		8	16-Mar-96	Hg	mg/kg
BS1210XX		10	16-Mar-96	Hg	mg/kg
BS1212XX		12	16-Mar-96	Hg	mg/kg
BS1216XX		16	16-Mar-96	Hg	mg/kg
BS1218XX		18	16-Mar-96	Hg	mg/kg
BS1218XX	DUP	18	16-Mar-96	Hg	mg/kg
BS1220XX		20	16-Mar-96	Hg	mg/kg
BS1222XX		21.5	16-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS1304XX		4	16-Mar-96	Hg	mg/kg
BS1306XX		6	16-Mar-96	Hg	mg/kg
BS1308XX		8	16-Mar-96	Hg	mg/kg
BS1310XX		10	16-Mar-96	Hg	mg/kg
BS1312XX		12	16-Mar-96	Hg	mg/kg
BS1314XX		14	16-Mar-96	Hg	mg/kg
BS1316XX		16	16-Mar-96	Hg	mg/kg
BS1318XX		18	16-Mar-96	Hg	mg/kg
BS1320XX		20	16-Mar-96	Hg	mg/kg
BS1702XX		2	14-Mar-96	Hg	mg/kg
BS1704XX		4	14-Mar-96	Hg	mg/kg
BS1706XX		6	14-Mar-96	Hg	mg/kg
BS1708XX		8	14-Mar-96	Hg	mg/kg
BS1708XX	DUP	8	14-Mar-96	Hg	mg/kg
BS1710XX		10	14-Mar-96	Hg	mg/kg
BS1712XX		12	14-Mar-96	Hg	mg/kg
BS1714XX		14	14-Mar-96	Hg	mg/kg
BS1718XX		18	14-Mar-96	Hg	mg/kg
BS1720XX		20	14-Mar-96	Hg	mg/kg
BS1722XX		22	14-Mar-96	Hg	mg/kg
BS1723XX		23	14-Mar-96	Hg	mg/kg
BS1802XX		2	17-Mar-96	Hg	mg/kg
BS1808XX		8	17-Mar-96	Hg	mg/kg
BS1810XX		10	17-Mar-96	Hg	mg/kg
BS1812XX		12	17-Mar-96	Hg	mg/kg
BS1814XX		14	17-Mar-96	Hg	mg/kg
BS1816XX		16	17-Mar-96	Hg	mg/kg
BS1818XX		18	17-Mar-96	Hg	mg/kg
BS1820XX		20	17-Mar-96	Hg	mg/kg
BS1822XX		22	17-Mar-96	Hg	mg/kg
BS1823XX		23.1	17-Mar-96	Hg	mg/kg
BS1902XX		2	17-Mar-96	Hg	mg/kg
BS1904XX		4	17-Mar-96	Hg	mg/kg
BS1904XX	DUP	4	17-Mar-96	Hg	mg/kg
BS1906XX		6	17-Mar-96	Hg	mg/kg
BS1908XX		8	17-Mar-96	Hg	mg/kg
BS1910XX		10	17-Mar-96	Hg	mg/kg
BS1912XX		12	17-Mar-96	Hg	mg/kg
BS1914XX		14	17-Mar-96	Hg	mg/kg
BS1916XX		16	17-Mar-96	Hg	mg/kg
BS1916XX	DUP	16	17-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS1918XX		18	17-Mar-96	Hg	mg/kg
BS1920XX		20	17-Mar-96	Hg	mg/kg
BS1922XX		22	17-Mar-96	Hg	mg/kg
BS2002XX		2	18-Mar-96	Hg	mg/kg
BS2004XX		4	18-Mar-96	Hg	mg/kg
BS2006XX		6	18-Mar-96	Hg	mg/kg
BS2008XX		8	18-Mar-96	Hg	mg/kg
BS2010XX		10	18-Mar-96	Hg	mg/kg
BS2012XX		12	18-Mar-96	Hg	mg/kg
BS2014XX		14	18-Mar-96	Hg	mg/kg
BS2016XX		16	18-Mar-96	Hg	mg/kg
BS2018XX		18	18-Mar-96	Hg	mg/kg
BS2020XX		20	18-Mar-96	Hg	mg/kg
BS2022XX		22	18-Mar-96	Hg	mg/kg
BS2023XX		23.4	18-Mar-96	Hg	mg/kg
BS2102XX		2	26-Mar-96	Hg	mg/kg
BS2104XX		4	26-Mar-96	Hg	mg/kg
BS2106XX		6	26-Mar-96	Hg	mg/kg
BS2108XX		8	26-Mar-96	Hg	mg/kg
BS2110XX		10	26-Mar-96	Hg	mg/kg
BS2110XX	DUP	10	26-Mar-96	Hg	mg/kg
BS2112XX		12	26-Mar-96	Hg	mg/kg
BS2114XX		14	26-Mar-96	Hg	mg/kg
BS2116XX		16	26-Mar-96	Hg	mg/kg
BS2118XX		18	26-Mar-96	Hg	mg/kg
BS2120XX		20	26-Mar-96	Hg	mg/kg
BS2121XX		20.7	26-Mar-96	Hg	mg/kg
BS2202XX		2	26-Mar-96	Hg	mg/kg
BS2204XX		4	26-Mar-96	Hg	mg/kg
BS2204XX	DUP	4	26-Mar-96	Hg	mg/kg
BS2206XX		6	26-Mar-96	Hg	mg/kg
BS2208XX		8	26-Mar-96	Hg	mg/kg
BS2210XX		10	26-Mar-96	Hg	mg/kg
BS2212XX		12	26-Mar-96	Hg	mg/kg
BS2214XX		14	26-Mar-96	Hg	mg/kg
BS2216XX		16	26-Mar-96	Hg	mg/kg
BS2218XX		18	26-Mar-96	Hg	mg/kg
BS2220XX		20	26-Mar-96	Hg	mg/kg
BS2222XX		22	26-Mar-96	Hg	mg/kg
BS2302XX		2	29-Mar-96	Hg	mg/kg
BS2304XX		4	29-Mar-96	Hg	mg/kg

TABLE B-3
SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS2306XX		6	29-Mar-96	Hg	mg/kg
BS2308XX		8	29-Mar-96	Hg	mg/kg
BS2310XX		10	29-Mar-96	Hg	mg/kg
BS2312XX		12	29-Mar-96	Hg	mg/kg
BS2312XX	DUP	12	29-Mar-96	Hg	mg/kg
BS2314XX		14	29-Mar-96	Hg	mg/kg
BS2316XX		16	29-Mar-96	Hg	mg/kg
BS2318XX		18	29-Mar-96	Hg	mg/kg
BS2322XX		22	29-Mar-96	Hg	mg/kg
BS2324XX		24	29-Mar-96	Hg	mg/kg
BS2326XX		25.8	29-Mar-96	Hg	mg/kg
BS2404XX		4	29-Mar-96	Hg	mg/kg
BS2406XX		6	29-Mar-96	Hg	mg/kg
BS2408XX		8	29-Mar-96	Hg	mg/kg
BS2408XX	DUP	8	29-Mar-96	Hg	mg/kg
BS2410XX		10	29-Mar-96	Hg	mg/kg
BS2412XX		12	29-Mar-96	Hg	mg/kg
BS2414XX		14	29-Mar-96	Hg	mg/kg
BS2416XX		16	29-Mar-96	Hg	mg/kg
BS2418XX		18	29-Mar-96	Hg	mg/kg
BS2420XX		20	29-Mar-96	Hg	mg/kg
BS2422XX		22	29-Mar-96	Hg	mg/kg
BS2424XX		23.5	29-Mar-96	Hg	mg/kg
BS2902XX		2	18-Mar-96	Hg	mg/kg
BS2906XX		6	18-Mar-96	Hg	mg/kg
BS2908XX		8	18-Mar-96	Hg	mg/kg
BS2910XX		10	18-Mar-96	Hg	mg/kg
BS2912XX		12	18-Mar-96	Hg	mg/kg
BS2914XX		14	18-Mar-96	Hg	mg/kg
BS2916XX		16	18-Mar-96	Hg	mg/kg
BS2918XX		18	18-Mar-96	Hg	mg/kg
BS2920XX		20	18-Mar-96	Hg	mg/kg
BS2922XX		22	18-Mar-96	Hg	mg/kg
BS3002XX		2	18-Mar-96	Hg	mg/kg
BS3002XX	DUP	2	18-Mar-96	Hg	mg/kg
BS3004XX		4	18-Mar-96	Hg	mg/kg
BS3006XX		6	18-Mar-96	Hg	mg/kg
BS3008XX		8	18-Mar-96	Hg	mg/kg
BS3010XX		10	18-Mar-96	Hg	mg/kg
BS3012XX		12	18-Mar-96	Hg	mg/kg
BS3014XX		14	18-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS3016XX		16	18-Mar-96	Hg	mg/kg
BS3016XX	DUP	16	18-Mar-96	Hg	mg/kg
BS3018XX		18	18-Mar-96	Hg	mg/kg
BS3020XX		20	18-Mar-96	Hg	mg/kg
BS3022XX		22	18-Mar-96	Hg	mg/kg
BS3024XX		23.7	18-Mar-96	Hg	mg/kg
BS3102XX		2	18-Mar-96	Hg	mg/kg
BS3104XX		4	18-Mar-96	Hg	mg/kg
BS3106XX		6	18-Mar-96	Hg	mg/kg
BS3108XX		8	18-Mar-96	Hg	mg/kg
BS3110XX		10	18-Mar-96	Hg	mg/kg
BS3112XX		11.5	18-Mar-96	Hg	mg/kg
BS3202XX		2	25-Mar-96	Hg	mg/kg
BS3204XX		4	25-Mar-96	Hg	mg/kg
BS3206XX		6	25-Mar-96	Hg	mg/kg
BS3208XX		8	25-Mar-96	Hg	mg/kg
BS3210XX		10	25-Mar-96	Hg	mg/kg
BS3212XX		12	25-Mar-96	Hg	mg/kg
BS3214XX		14	25-Mar-96	Hg	mg/kg
BS3216XX		15.7	25-Mar-96	Hg	mg/kg
BS3216XX	DUP	15.7	25-Mar-96	Hg	mg/kg
BS3302XX		2	25-Mar-96	Hg	mg/kg
BS3304XX		4	25-Mar-96	Hg	mg/kg
BS3306XX		6	25-Mar-96	Hg	mg/kg
BS3308XX		8	25-Mar-96	Hg	mg/kg
BS3310XX		10	25-Mar-96	Hg	mg/kg
BS3312XX		12	25-Mar-96	Hg	mg/kg
BS3314XX		14	25-Mar-96	Hg	mg/kg
BS3316XX		16	25-Mar-96	Hg	mg/kg
BS3318XX		18	25-Mar-96	Hg	mg/kg
BS3320XX		20	25-Mar-96	Hg	mg/kg
BS3320XX		20.2	25-Mar-96	Hg	mg/kg
BS3320XX	DUP	20.2	25-Mar-96	Hg	mg/kg
BS3404XX		4	19-Mar-96	Hg	mg/kg
BS3406XX		6	19-Mar-96	Hg	mg/kg
BS3408XX		8	19-Mar-96	Hg	mg/kg
BS3410XX		10	19-Mar-96	Hg	mg/kg
BS3410XX	DUP	10	19-Mar-96	Hg	mg/kg
BS3412XX		12	19-Mar-96	Hg	mg/kg
BS3414XX		14	19-Mar-96	Hg	mg/kg
BS3416XX		16	19-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS3418XX		18	19-Mar-96	Hg	mg/kg
BS3420XX		20	19-Mar-96	Hg	mg/kg
BS3502XX		2	19-Mar-96	Hg	mg/kg
BS3504XX		4	19-Mar-96	Hg	mg/kg
BS3506XX		6	19-Mar-96	Hg	mg/kg
BS3508XX		8	19-Mar-96	Hg	mg/kg
BS3508XX	DUP	8	19-Mar-96	Hg	mg/kg
BS3510XX		10	19-Mar-96	Hg	mg/kg
BS3512XX		12	19-Mar-96	Hg	mg/kg
BS3514XX		14	19-Mar-96	Hg	mg/kg
BS3516XX		16	19-Mar-96	Hg	mg/kg
BS3518XX		18	19-Mar-96	Hg	mg/kg
BS3520XX		20	19-Mar-96	Hg	mg/kg
BS3522XX		21.7	19-Mar-96	Hg	mg/kg
BS3602XX		2	27-Mar-96	Hg	mg/kg
BS3604XX		4	27-Mar-96	Hg	mg/kg
BS3606XX		6	27-Mar-96	Hg	mg/kg
BS3608XX		8	27-Mar-96	Hg	mg/kg
BS3610XX		10	27-Mar-96	Hg	mg/kg
BS3612XX		12	27-Mar-96	Hg	mg/kg
BS3614XX		14	27-Mar-96	Hg	mg/kg
BS3614XX	DUP	14	27-Mar-96	Hg	mg/kg
BS3616XX		16	27-Mar-96	Hg	mg/kg
BS3618XX		18	27-Mar-96	Hg	mg/kg
BS3620XX		20	27-Mar-96	Hg	mg/kg
BS3622XX		22	27-Mar-96	Hg	mg/kg
BS3704XX		4	17-Mar-96	Hg	mg/kg
BS3706XX		6	17-Mar-96	Hg	mg/kg
BS3708XX		8	17-Mar-96	Hg	mg/kg
BS3708XX	DUP	8	17-Mar-96	Hg	mg/kg
BS3710XX		10	17-Mar-96	Hg	mg/kg
BS3710XX	DUP	10	17-Mar-96	Hg	mg/kg
BS3712XX		12	17-Mar-96	Hg	mg/kg
BS3712XX	DUP	12	17-Mar-96	Hg	mg/kg
BS3714XX		14	17-Mar-96	Hg	mg/kg
BS3716XX		16	17-Mar-96	Hg	mg/kg
BS3716XX	DUP	16	17-Mar-96	Hg	mg/kg
BS3718XX		18	17-Mar-96	Hg	mg/kg
BS3720XX		20	17-Mar-96	Hg	mg/kg
BS3720XX	DUP	20	17-Mar-96	Hg	mg/kg
BS3804XX		4	18-Mar-96	Hg	mg/kg

TABLE B-3
SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS3806XX		6	18-Mar-96	Hg	mg/kg
BS3808XX		8	18-Mar-96	Hg	mg/kg
BS3810XX		10	18-Mar-96	Hg	mg/kg
BS3812XX		12	18-Mar-96	Hg	mg/kg
BS3814XX		14	18-Mar-96	Hg	mg/kg
BS3816XX		16	18-Mar-96	Hg	mg/kg
BS3818XX		18	18-Mar-96	Hg	mg/kg
BS3820XX		20	18-Mar-96	Hg	mg/kg
BS3822XX		22	18-Mar-96	Hg	mg/kg
BS3824XX		24	18-Mar-96	Hg	mg/kg
BS4002XX		2	27-Mar-96	Hg	mg/kg
BS4004XX		4	27-Mar-96	Hg	mg/kg
BS4006XX		6	27-Mar-96	Hg	mg/kg
BS4008XX		8	27-Mar-96	Hg	mg/kg
BS4010XX		10	27-Mar-96	Hg	mg/kg
BS4010XX	DUP	10	27-Mar-96	Hg	mg/kg
BS4012XX		12	27-Mar-96	Hg	mg/kg
BS4016XX		16	27-Mar-96	Hg	mg/kg
BS4018XX		18	27-Mar-96	Hg	mg/kg
BS4020XX		20	27-Mar-96	Hg	mg/kg
BS4102XX		2	28-Mar-96	Hg	mg/kg
BS4102XX	DUP	2	28-Mar-96	Hg	mg/kg
BS4104XX		4	28-Mar-96	Hg	mg/kg
BS4106XX		6	28-Mar-96	Hg	mg/kg
BS4108XX		8	28-Mar-96	Hg	mg/kg
BS4110XX		10	28-Mar-96	Hg	mg/kg
BS4112XX		12	28-Mar-96	Hg	mg/kg
BS4114XX		14	28-Mar-96	Hg	mg/kg
BS4116XX		16	28-Mar-96	Hg	mg/kg
BS4118XX		18	28-Mar-96	Hg	mg/kg
BS4118XX	DUP	18	28-Mar-96	Hg	mg/kg
BS4120XX		20	28-Mar-96	Hg	mg/kg
BS4202XX		2	28-Mar-96	Hg	mg/kg
BS4204XX		4	28-Mar-96	Hg	mg/kg
BS4206XX		6	28-Mar-96	Hg	mg/kg
BS4208XX		8	28-Mar-96	Hg	mg/kg
BS4208XX	DUP	8	28-Mar-96	Hg	mg/kg
BS4210XX		10	28-Mar-96	Hg	mg/kg
BS4212XX		12	28-Mar-96	Hg	mg/kg
BS4214XX		14	28-Mar-96	Hg	mg/kg
BS4216XX		16	28-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS4218XX		18	28-Mar-96	Hg	mg/kg
BS4220XX		20	28-Mar-96	Hg	mg/kg
BS4302XX		2	28-Mar-96	Hg	mg/kg
BS4304XX		4	28-Mar-96	Hg	mg/kg
BS4306XX		6	28-Mar-96	Hg	mg/kg
BS4306XX	DUP	6	28-Mar-96	Hg	mg/kg
BS4308XX		8	28-Mar-96	Hg	mg/kg
BS4310XX		10	28-Mar-96	Hg	mg/kg
BS4312XX		12	28-Mar-96	Hg	mg/kg
BS4314XX		14	28-Mar-96	Hg	mg/kg
BS4316XX		16	28-Mar-96	Hg	mg/kg
BS4318XX		18	28-Mar-96	Hg	mg/kg
BS4504XX		4	27-Mar-96	Hg	mg/kg
BS4506XX		6	27-Mar-96	Hg	mg/kg
BS4508XX		8	27-Mar-96	Hg	mg/kg
BS4508XX	DUP	8	27-Mar-96	Hg	mg/kg
BS4510XX		10	27-Mar-96	Hg	mg/kg
BS4512XX		12	27-Mar-96	Hg	mg/kg
BS4516XX		16	27-Mar-96	Hg	mg/kg
BS4520XX		20	27-Mar-96	Hg	mg/kg
BS4524XX		24	27-Mar-96	Hg	mg/kg
BS4528XX		28	27-Mar-96	Hg	mg/kg
BS4604XX		4	28-Mar-96	Hg	mg/kg
BS4606XX		6	28-Mar-96	Hg	mg/kg
BS4608XX		8	28-Mar-96	Hg	mg/kg
BS4610XX		10	28-Mar-96	Hg	mg/kg
BS4612XX		12	28-Mar-96	Hg	mg/kg
BS4614XX		14	28-Mar-96	Hg	mg/kg
BS4616XX		16	28-Mar-96	Hg	mg/kg
BS4616XX	DUP	16	28-Mar-96	Hg	mg/kg
BS4618XX		18	28-Mar-96	Hg	mg/kg
BS4620XX		20	28-Mar-96	Hg	mg/kg
BS4622XX		22	28-Mar-96	Hg	mg/kg
BS4624XX		24	28-Mar-96	Hg	mg/kg
BS4626XX		26	28-Mar-96	Hg	mg/kg
BS4628XX		28	28-Mar-96	Hg	mg/kg
BS4629XX		29	28-Mar-96	Hg	mg/kg
BS4804XX		4	29-Mar-96	Hg	mg/kg
BS4806XX		6	29-Mar-96	Hg	mg/kg
BS4808XX		8	29-Mar-96	Hg	mg/kg
BS4810XX		10	29-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS4812XX		12	29-Mar-96	Hg	mg/kg
BS4812XX	DUP	12	29-Mar-96	Hg	mg/kg
BS4814XX		14	29-Mar-96	Hg	mg/kg
BS4816XX		16	29-Mar-96	Hg	mg/kg
BS4818XX		18	29-Mar-96	Hg	mg/kg
BS4820XX		20	29-Mar-96	Hg	mg/kg
BS4822XX		22	29-Mar-96	Hg	mg/kg
BS4824XX		24	29-Mar-96	Hg	mg/kg
BS4826XX		26	29-Mar-96	Hg	mg/kg
BS4904XX		4	29-Mar-96	Hg	mg/kg
BS4906XX		6	29-Mar-96	Hg	mg/kg
BS4908XX		8	29-Mar-96	Hg	mg/kg
BS4910XX		10	29-Mar-96	Hg	mg/kg
BS4912XX		12	29-Mar-96	Hg	mg/kg
BS4914XX		14	29-Mar-96	Hg	mg/kg
BS4916XX		16	29-Mar-96	Hg	mg/kg
BS4918XX		18	29-Mar-96	Hg	mg/kg
BS4920XX		20	29-Mar-96	Hg	mg/kg
BS4922XX		22	29-Mar-96	Hg	mg/kg
BS4924XX		24	29-Mar-96	Hg	mg/kg
BS4924XX	DUP	24	29-Mar-96	Hg	mg/kg
BS5104XX		4	31-Mar-96	Hg	mg/kg
BS5108XX		8	31-Mar-96	Hg	mg/kg
BS5112XX		12	31-Mar-96	Hg	mg/kg
BS5116XX		16	31-Mar-96	Hg	mg/kg
BS5118XX		18	31-Mar-96	Hg	mg/kg
BS5118XX	DUP	18	31-Mar-96	Hg	mg/kg
BS5120XX		20	31-Mar-96	Hg	mg/kg
BS5124XX		24	31-Mar-96	Hg	mg/kg
BS5204XX		4	31-Mar-96	Hg	mg/kg
BS5208XX		8	31-Mar-96	Hg	mg/kg
BS5212XX		12	31-Mar-96	Hg	mg/kg
BS5212XX	DUP	12	31-Mar-96	Hg	mg/kg
BS5218XX		18	31-Mar-96	Hg	mg/kg
BS5222XX		22	31-Mar-96	Hg	mg/kg
BS5226XX		26	31-Mar-96	Hg	mg/kg
BS5304XX		4	01-Apr-96	Hg	mg/kg
BS5306XX		6	01-Apr-96	Hg	mg/kg
BS5308XX		8	01-Apr-96	Hg	mg/kg
BS5312XX		12	01-Apr-96	Hg	mg/kg
BS5314XX		14	01-Apr-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS5314XX	DUP	14	01-Apr-96	Hg	mg/kg
BS5316XX		16	01-Apr-96	Hg	mg/kg
BS5320XX		20	01-Apr-96	Hg	mg/kg
BS5322XX		22	01-Apr-96	Hg	mg/kg
BS5602XX		2	29-Mar-96	Hg	mg/kg
BS5604XX		4	29-Mar-96	Hg	mg/kg
BS5606XX		6	29-Mar-96	Hg	mg/kg
BS5608XX		8	29-Mar-96	Hg	mg/kg
BS5610XX		10	29-Mar-96	Hg	mg/kg
BS5610XX	DUP	10	29-Mar-96	Hg	mg/kg
BS5612XX		12	29-Mar-96	Hg	mg/kg
BS5614XX		14	29-Mar-96	Hg	mg/kg
BS5616XX		16	29-Mar-96	Hg	mg/kg
BS5620XX		20	29-Mar-96	Hg	mg/kg
BS5622XX		22	29-Mar-96	Hg	mg/kg
BS5624XX		23.9	29-Mar-96	Hg	mg/kg
BS5704XX		4	30-Mar-96	Hg	mg/kg
BS5710XX		10	30-Mar-96	Hg	mg/kg
BS5712XX		12	30-Mar-96	Hg	mg/kg
BS5714XX		14	30-Mar-96	Hg	mg/kg
BS5716XX		16	30-Mar-96	Hg	mg/kg
BS5718XX		18	30-Mar-96	Hg	mg/kg
BS5720XX		20	30-Mar-96	Hg	mg/kg
BS5720XX	DUP	20	30-Mar-96	Hg	mg/kg
BS5722XX		22	30-Mar-96	Hg	mg/kg
BS5724XX		24	30-Mar-96	Hg	mg/kg
BS5725XX		25	30-Mar-96	Hg	mg/kg
BS5802XX		2	30-Mar-96	Hg	mg/kg
BS5804XX		4	30-Mar-96	Hg	mg/kg
BS5806XX		6	30-Mar-96	Hg	mg/kg
BS5808XX		8	30-Mar-96	Hg	mg/kg
BS5810XX		10	30-Mar-96	Hg	mg/kg
BS5812XX		12	30-Mar-96	Hg	mg/kg
BS5814XX		14	30-Mar-96	Hg	mg/kg
BS5816XX		16	30-Mar-96	Hg	mg/kg
BS5818XX		18	30-Mar-96	Hg	mg/kg
BS5820XX		20	30-Mar-96	Hg	mg/kg
BS5820XX	DUP	20	30-Mar-96	Hg	mg/kg
BS5821XX		21.2	30-Mar-96	Hg	mg/kg
BS5904XX		4	01-Apr-96	Hg	mg/kg
BS5910XX		10	01-Apr-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS5912XX		12	01-Apr-96	Hg	mg/kg
BS5914XX		14	01-Apr-96	Hg	mg/kg
BS5916XX		16	01-Apr-96	Hg	mg/kg
BS5918XX		18	01-Apr-96	Hg	mg/kg
BS5920XX		20	01-Apr-96	Hg	mg/kg
BS5922XX		22	01-Apr-96	Hg	mg/kg
BS5924XX		23.7	01-Apr-96	Hg	mg/kg
BS6002XX		2	30-Mar-96	Hg	mg/kg
BS6004XX		4	30-Mar-96	Hg	mg/kg
BS6008XX		8	30-Mar-96	Hg	mg/kg
BS6010XX		10	30-Mar-96	Hg	mg/kg
BS6012XX		12	30-Mar-96	Hg	mg/kg
BS6012XX	DUP	12	30-Mar-96	Hg	mg/kg
BS6014XX		14	30-Mar-96	Hg	mg/kg
BS6014XX	DUP	14	30-Mar-96	Hg	mg/kg
BS6016XX		16	30-Mar-96	Hg	mg/kg
BS6018XX		18	30-Mar-96	Hg	mg/kg
BS6020XX		20	30-Mar-96	Hg	mg/kg
BS6022XX		22	30-Mar-96	Hg	mg/kg
BS6102XX		2	31-Mar-96	Hg	mg/kg
BS6104XX		4	31-Mar-96	Hg	mg/kg
BS6106XX		6	31-Mar-96	Hg	mg/kg
BS6108XX		8	31-Mar-96	Hg	mg/kg
BS6110XX		10	31-Mar-96	Hg	mg/kg
BS6112XX		12	31-Mar-96	Hg	mg/kg
BS6114XX		14	31-Mar-96	Hg	mg/kg
BS6116XX		16	31-Mar-96	Hg	mg/kg
BS6118XX		18	31-Mar-96	Hg	mg/kg
BS6118XX	DUP	18	31-Mar-96	Hg	mg/kg
BS6120XX		20	31-Mar-96	Hg	mg/kg
BS6122XX		22	31-Mar-96	Hg	mg/kg
BS6202XX		2	31-Mar-96	Hg	mg/kg
BS6204XX		4	31-Mar-96	Hg	mg/kg
BS6204XX	DUP	4	31-Mar-96	Hg	mg/kg
BS6206XX		6	31-Mar-96	Hg	mg/kg
BS6208XX		8	31-Mar-96	Hg	mg/kg
BS6210XX		10	31-Mar-96	Hg	mg/kg
BS6212XX		12	31-Mar-96	Hg	mg/kg
BS6214XX		14	31-Mar-96	Hg	mg/kg
BS6218XX		18	31-Mar-96	Hg	mg/kg
BS6302XX		2	31-Mar-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS6304XX		4	31-Mar-96	Hg	mg/kg
BS6306XX		6	31-Mar-96	Hg	mg/kg
BS6306XX	DUP	6	31-Mar-96	Hg	mg/kg
BS6308XX		8	31-Mar-96	Hg	mg/kg
BS6312XX		12	31-Mar-96	Hg	mg/kg
BS6314XX		14	31-Mar-96	Hg	mg/kg
BS6316XX		16	31-Mar-96	Hg	mg/kg
BS6318XX		18	31-Mar-96	Hg	mg/kg
BS6320XX		20	31-Mar-96	Hg	mg/kg
BS6322XX		22	31-Mar-96	Hg	mg/kg
BS6402XX		2	01-Apr-96	Hg	mg/kg
BS6404XX		4	01-Apr-96	Hg	mg/kg
BS6406XX		6	01-Apr-96	Hg	mg/kg
BS6406XX	DUP	6	01-Apr-96	Hg	mg/kg
BS6408XX		8	01-Apr-96	Hg	mg/kg
BS6410XX		10	01-Apr-96	Hg	mg/kg
BS6412XX		12	01-Apr-96	Hg	mg/kg
BS6414XX		14	01-Apr-96	Hg	mg/kg
BS6416XX		16	01-Apr-96	Hg	mg/kg
BS6418XX		18	01-Apr-96	Hg	mg/kg
BS6420XX		20	01-Apr-96	Hg	mg/kg
BS6422XX		22	01-Apr-96	Hg	mg/kg
BS6502XX		2	01-Apr-96	Hg	mg/kg
BS6504XX		4	01-Apr-96	Hg	mg/kg
BS6506XX		6	01-Apr-96	Hg	mg/kg
BS6508XX		8	01-Apr-96	Hg	mg/kg
BS6512XX		12	01-Apr-96	Hg	mg/kg
BS6514XX		14	01-Apr-96	Hg	mg/kg
BS6516XX		16	01-Apr-96	Hg	mg/kg
BS6518XX		18	01-Apr-96	Hg	mg/kg
BS6518XX	DUP	18	01-Apr-96	Hg	mg/kg
BS6520XX		20	01-Apr-96	Hg	mg/kg
BS6522XX		22	01-Apr-96	Hg	mg/kg
BS6604XX		4	03-Apr-96	Hg	mg/kg
BS6606XX		6	03-Apr-96	Hg	mg/kg
BS6606XX	DUP	6	03-Apr-96	Hg	mg/kg
BS6608XX		8	03-Apr-96	Hg	mg/kg
BS6610XX		10	03-Apr-96	Hg	mg/kg
BS6614XX		14	03-Apr-96	Hg	mg/kg
BS6616XX		16	03-Apr-96	Hg	mg/kg
BS6616XX	DUP	16	03-Apr-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS6618XX		18	03-Apr-96	Hg	mg/kg
BS6620XX		20	03-Apr-96	Hg	mg/kg
BS6620XX	DUP	20	03-Apr-96	Hg	mg/kg
BS6702XX		2	08-Apr-96	Hg	mg/kg
BS6704XX		4	08-Apr-96	Hg	mg/kg
BS6706XX		6	08-Apr-96	Hg	mg/kg
BS6708XX		8	08-Apr-96	Hg	mg/kg
BS6708XX	DUP	8	08-Apr-96	Hg	mg/kg
BS6710XX		10	08-Apr-96	Hg	mg/kg
BS6802XX		2	08-Apr-96	Hg	mg/kg
BS6804XX		4	08-Apr-96	Hg	mg/kg
BS6806XX		6	08-Apr-96	Hg	mg/kg
BS6808XX		8	08-Apr-96	Hg	mg/kg
BS6808XX	DUP	8	08-Apr-96	Hg	mg/kg
BS6810XX		10	08-Apr-96	Hg	mg/kg
BS6902XX		2	01-Apr-96	Hg	mg/kg
BS6904XX		4	01-Apr-96	Hg	mg/kg
BS6906XX		6	01-Apr-96	Hg	mg/kg
BS6908XX		8	01-Apr-96	Hg	mg/kg
BS7002XX		2	02-Apr-96	Hg	mg/kg
BS7004XX		4	02-Apr-96	Hg	mg/kg
BS7006XX		6	02-Apr-96	Hg	mg/kg
BS7008XX		8	02-Apr-96	Hg	mg/kg
BS7010XX		10	02-Apr-96	Hg	mg/kg
BS7014XX		14	02-Apr-96	Hg	mg/kg
BS7016XX		16	02-Apr-96	Hg	mg/kg
BS7018XX		18	02-Apr-96	Hg	mg/kg
BS7020XX		20	02-Apr-96	Hg	mg/kg
BS7022XX		22	02-Apr-96	Hg	mg/kg
BS7104XX		4	02-Apr-96	Hg	mg/kg
BS7106XX		6	02-Apr-96	Hg	mg/kg
BS7108XX		8	02-Apr-96	Hg	mg/kg
BS7110XX		10	02-Apr-96	Hg	mg/kg
BS7112XX		12	02-Apr-96	Hg	mg/kg
BS7112XX	DUP	12	02-Apr-96	Hg	mg/kg
BS7114XX		14	02-Apr-96	Hg	mg/kg
BS7116XX		16	02-Apr-96	Hg	mg/kg
BS7116XX	DUP	16	02-Apr-96	Hg	mg/kg
BS7118XX		18	02-Apr-96	Hg	mg/kg
BS7118XX	DUP	18	02-Apr-96	Hg	mg/kg
BS7202XX		2	02-Apr-96	Hg	mg/kg

TABLE B-3

SAMPLES INCLUDED IN MERCURY DATA SET - SOIL

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	depth	s_date	type	
BS7204XX		4	02-Apr-96	Hg	mg/kg
BS7206XX		6	02-Apr-96	Hg	mg/kg
BS7208XX		8	02-Apr-96	Hg	mg/kg
BS7302XX		2	08-Apr-96	Hg	mg/kg
BS7304XX		4	08-Apr-96	Hg	mg/kg
BS7306XX		6	08-Apr-96	Hg	mg/kg
BS7308XX		8	08-Apr-96	Hg	mg/kg
BS7310XX		10	08-Apr-96	Hg	mg/kg
BS7402XX		2	09-Apr-96	Hg	mg/kg
BS7404XX		4	09-Apr-96	Hg	mg/kg
BS7406XX		6	09-Apr-96	Hg	mg/kg
BS7408XX		8	09-Apr-96	Hg	mg/kg
BS7410XX		10	09-Apr-96	Hg	mg/kg
BS7410XX	DUP	10	09-Apr-96	Hg	mg/kg
KT23SSXX1		1	08-May-93	Hg	mg/kg
KTSSXX1X8		8	28-Sep-93	Hg	mg/kg
LTSSXX1X0		0	08-May-93	Hg	mg/kg
LTSSXX1X1		1	08-May-93	Hg	mg/kg
LTSSXX2X0		0	08-May-93	Hg	mg/kg
LTSSXX2X1		1	08-May-93	Hg	mg/kg

TABLE B-4

SAMPLES INCLUDED IN PERIMETER WELL DATA SET

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	s_date	type		
MWW4XXXX		13-Apr-96	HG	UG/L	
MWW4XXXX		13-Apr-96	LLVC	UG/L	
MWW4XXXX		13-Apr-96	VOA	UG/L	
MWW5XXXX		12-Apr-96	HG	UG/L	
MWW5XXXX		12-Apr-96	LLVC	UG/L	
MWW5XXXX		12-Apr-96	VOA	UG/L	
TW01XXXX		14-Apr-96	HG	UG/L	
TW01XXXX		14-Apr-96	LLVC	UG/L	
TW01XXXX		14-Apr-96	VOA	UG/L	
TW02XXXX		14-Apr-96	HG	UG/L	
TW02XXXX		14-Apr-96	LLVC	UG/L	
TW02XXXX		14-Apr-96	VOA	UG/L	
TW03XXXX		14-Apr-96	HG	UG/L	
TW03XXXX		14-Apr-96	LLVC	UG/L	
TW03XXXX		14-Apr-96	VOA	UG/L	
TW03XXXX	DUP	14-Apr-96	HG	UG/L	
TW03XXXX	DUP	14-Apr-96	LLVC	UG/L	
TW03XXXX	DUP	14-Apr-96	VOA	UG/L	
TW04XXXX		14-Apr-96	HG	UG/L	
TW04XXXX		14-Apr-96	LLVC	UG/L	
TW04XXXX		14-Apr-96	VOA	UG/L	
TW05XXXX		15-Apr-96	HG	UG/L	
TW05XXXX		15-Apr-96	INOR	MG/L	
TW05XXXX		15-Apr-96	LLVC	UG/L	
TW05XXXX		15-Apr-96	VOA	UG/L	
TW06XXXX		15-Apr-96	HG	UG/L	
TW06XXXX		15-Apr-96	INOR	MG/L	
TW06XXXX		15-Apr-96	LLVC	UG/L	
TW06XXXX		15-Apr-96	VOA	UG/L	
TW06XXXX	DUP	15-Apr-96	HG	UG/L	
TW06XXXX	DUP	15-Apr-96	INOR	MG/L	
TW06XXXX	DUP	15-Apr-96	LLVC	UG/L	
TW06XXXX	DUP	15-Apr-96	VOA	UG/L	
TW07XXXX		15-Apr-96	HG	UG/L	
TW07XXXX		15-Apr-96	INOR	MG/L	
TW07XXXX		15-Apr-96	LLVC	UG/L	
TW07XXXX		15-Apr-96	VOA	UG/L	
TW08XXXX		14-Apr-96	HG	UG/L	
TW08XXXX		14-Apr-96	INOR	MG/L	
TW08XXXX		14-Apr-96	LLVC	UG/L	

TABLE B-4

SAMPLES INCLUDED IN PERIMETER WELL DATA SET

HUMAN HEALTH RISK ASSESSMENT

AMES STREET SITE

ROCHESTER, NY

smpl_id	stype	s_date	type		
TW08XXXX		14-Apr-96	VOA	UG/L	
TW09XXXX		14-Apr-96	HG	UG/L	
TW09XXXX		14-Apr-96	LLVC	UG/L	
TW09XXXX		14-Apr-96	VOA	UG/L	
TW11XXXX		14-Apr-96	HG	UG/L	
TW11XXXX		14-Apr-96	INOR	UG/L	
TW11XXXX		14-Apr-96	LLVC	UG/L	
TW11XXXX		14-Apr-96	VOA	UG/L	
TW12XXXX		13-Apr-96	HG	UG/L	
TW12XXXX		13-Apr-96	LLVC	UG/L	
TW12XXXX		13-Apr-96	VOA	UG/L	
TW13XXXX		13-Apr-96	HG	UG/L	
TW13XXXX		13-Apr-96	INOR	UG/L	
TW13XXXX		13-Apr-96	LLVC	UG/L	
TW13XXXX		13-Apr-96	VOA	UG/L	
TW14XXXX		13-Apr-96	HG	UG/L	
TW14XXXX		13-Apr-96	LLVC	UG/L	
TW14XXXX		13-Apr-96	VOA	UG/L	
TW15XXXX		13-Apr-96	HG	UG/L	
TW15XXXX		13-Apr-96	INOR	UG/L	
TW15XXXX		13-Apr-96	LLVC	UG/L	
TW15XXXX		13-Apr-96	VOA	UG/L	
TW16AXXX		15-Apr-96	HG	UG/L	
TW16AXXX		15-Apr-96	LLVC	UG/L	
TW16AXXX		15-Apr-96	VOA	UG/L	
TW16XXXX		15-Apr-96	HG	UG/L	
TW16XXXX		15-Apr-96	INOR	UG/L	
TW16XXXX		15-Apr-96	LLVC	UG/L	
TW16XXXX		15-Apr-96	VOA	UG/L	
TW17XXXX		11-Apr-96	HG	UG/L	
TW17XXXX		11-Apr-96	INOR	UG/L	
TW17XXXX		11-Apr-96	LLVC	UG/L	
TW17XXXX		11-Apr-96	VOA	UG/L	
TW18AXXX		15-Apr-96	HG	UG/L	
TW18AXXX		15-Apr-96	INOR	UG/L	
TW18AXXX		15-Apr-96	LLVC	UG/L	
TW18AXXX		15-Apr-96	VOA	UG/L	
TW18XXXX		15-Apr-96	HG	UG/L	
TW18XXXX		15-Apr-96	INOR	UG/L	
TW18XXXX		15-Apr-96	LLVC	UG/L	

TABLE B-4				
SAMPLES INCLUDED IN PERIMETER WELL DATA SET				
HUMAN HEALTH RISK ASSESSMENT				
AMES STREET SITE				
ROCHESTER, NY				
smpl_id	stype	s_date	type	
TW18XXXX		15-Apr-96	VOA UG/L	
TW19XXXX		11-Apr-96	HG UG/L	
TW19XXXX		11-Apr-96	INOR UG/L	
TW19XXXX		11-Apr-96	LLVC UG/L	
TW19XXXX		11-Apr-96	VOA UG/L	
TW20XXXX		15-Apr-96	HG UG/L	
TW20XXXX		15-Apr-96	LLVC UG/L	
TW20XXXX		15-Apr-96	VOA UG/L	

TABLE B-5
SAMPLES INCLUDED IN INTERIOR WELL DATA SET

HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

smpl_id	stype	s_date	type	
MW00XXXX		11-Apr-96	HG	UG/L
MW00XXXX		11-Apr-96	INOR	UG/L
MW00XXXX		11-Apr-96	LLVC	UG/L
MW00XXXX		11-Apr-96	VOA	UG/L
MWW3XXXX		12-Apr-96	HG	UG/L
MWW3XXXX		12-Apr-96	LLVC	UG/L
MWW3XXXX		12-Apr-96	VOA	UG/L
TW69XXXX		11-Apr-96	HG	UG/L
TW69XXXX		11-Apr-96	INOR	UG/L
TW69XXXX		11-Apr-96	LLVC	UG/L
TW69XXXX		11-Apr-96	VOA	UG/L
TW74XXXX		11-Apr-96	HG	UG/L
TW74XXXX		11-Apr-96	INOR	UG/L
TW74XXXX		11-Apr-96	LLVC	UG/L
TW74XXXX		11-Apr-96	VOA	UG/L
TW74XXXX	DUP	11-Apr-96	HG	UG/L
TW74XXXX	DUP	11-Apr-96	INOR	UG/L
TW74XXXX	DUP	11-Apr-96	LLVC	UG/L
TW74XXXX	DUP	11-Apr-96	VOA	UG/L
W-TANK-2-01		22-Nov-95	VOA	UG/L
W-TANK-11/12-01		5-Dec-95	VOA	UG/L
34TGW693		10-May-93	VOA	UG/L
LTSW193		08-May-93	HG	UG/L
43GW693		09-May-93	VOA	UG/L
43GW693		09-May-93	HG	UG/L

APPENDIX C
TOXICITY PROFILES AND DOSE-RESPONSE TABLES

SHORT TOXICITY PROFILES

1,1,1-Trichloroethane. 1,1,1-Trichloroethane is a man-made, chlorinated volatile organic chemical. It is used as a solvent for paints, as a cleanser to remove grease and oil, and is contained in household spot removers, aerosol sprays, and glues.

1,1,1-Trichloroethane is extensively absorbed through the gastrointestinal tract, the lungs, and to a lesser extent, through the skin. Like many chlorinated volatile compounds, 1,1,1-trichloroethane depresses the central nervous system and impairs coordination, equilibrium and judgment in both humans and animals, when exposed by any route. Additionally, exposures to high concentrations of 1,1,1-trichloroethane may produce adverse cardiovascular effects, including arrhythmias and decreased blood pressure. Reproductive and developmental toxicity have not been reported in human epidemiological studies. Evidence for or against the potential carcinogenicity of 1,1,1-trichloroethane in humans and animals has not been established.

References:

Agency for Toxic Substances and Disease Registry (ATSDR), 1993. "Toxicological Profile for 1,1,1-Trichloroethane"; Agency for Toxic Substances and Disease Registry, U.S. Public Health Service. October, 1993.

MADEP, 1992. "Risk Assessment Shortform Residential Exposure Scenario, Version 1.6"; Policy #WSC/ORS-142-92; Office of Research and Standards and the Bureau of Waste Site Cleanup, Boston, MA; September 1992.

1,2-Dichloroethene. 1,2-Dichloroethene is a volatile organic compound which exists as cis- and trans-isomers. The commercially used material is usually a mixture of the two isomers. In the past, it was used as a general inhalation anesthetic. It is currently used as an extraction solvent or as a component of dyes, perfume oils, waxes, resins, and plastics. It is also used as an intermediate in the synthesis of polymers.

1,2-Dichloroethene is absorbed by all routes of administration. Distribution is rapid and, due to its lipophilic nature, occurs to all organ systems. It is extensively metabolized to dichloroacetaldehyde and chloroacetic acids which are excreted primarily through urine.

Dermal contact to 1,2-dichloroethene may result in defatting of the skin and dermatitis. Exposure to airborne 1,2-dichloroethene causes irritation to eyes, mucous membranes and the upper respiratory tract. Systemically, the trans-isomer is believed to be more toxic than the cis-isomer. However, both have been reported to produce central nervous system depression and toxicity to liver and lungs. No data on the reproductive toxicity of 1,2-dichloroethene exists. Both isomers have tested negative for mutagenicity in vitro tests. Cancer effects have not been studied in humans or animals.

References:
Agency for Toxic Substances and Disease Registry (ATSDR), 1990. "Toxicological Profile for 1,2-Dichloroethene"; Agency for Toxic Substances and Disease Registry, U.S. Public Health Service, February 1990.

Mycroft, F.J., Jones, J.R., and Olson, K.R. 1990. Environmental and Occupational Toxicology. In: Poisoning and Drug Overdose. Ed. K.R. Olson. Appleton & Lange, CT. p. 397.

4-Methyl-2-Pentanone (Methyl isobutyl ketone). Methyl isobutyl ketone (MIBK) has been used as a solvent for coatings, adhesives, cements, and in the lacquer and varnish industries. Acute exposure to MIBK has produced irritation to mucous membranes and conjunctiva while prolonged exposure to high concentrations has caused CNS depression. Chronic exposure studies in animals indicate the liver and the kidney as target organs of MIBK. MIBK can potentiate the neurotoxicity of other solvents such as n-hexane, methyl ethyl ketone, and ethyl butyl ketone. MIBK has been classified by USEPA as a class D carcinogen, not classifiable as to human carcinogenicity.

References:
Agency for Toxic Substances and Disease Registry (ATSDR), 1990. "Toxicological Profile for 4-Methyl-2-Pentanone"; Agency for Toxic Substances and Disease Registry, U.S. Public Health Service, October, 1990.

Benzene. Benzene is an organic solvent that is both naturally occurring and produced from petroleum. Benzene is used in the synthesis of many industrial chemicals and pharmaceuticals, for the extraction of fats and oils, in the manufacture of explosives, in is a major component of petroleum based fuels such as gasoline.

Benzene is readily absorbed orally, moderately absorbed by inhalation, and poorly absorbed dermally. Its toxic actions are primarily a result of its metabolites, which are largely produced in the liver, and to some extent, in the bone marrow. Acute exposure to benzene has produced central nervous system depression in humans and animals. Chronic exposures have produced adverse liver effects and hematological toxicity, including aplastic anemia and leukemia. Available evidence does not suggest that benzene is teratogenic in humans or animals. There is sufficient evidence of benzene-induced carcinogenicity in humans via inhalation exposure, placing it in USEPA weight-of-evidence group A, human carcinogen.

References:
MADEP, 1992. "Risk Assessment Shortform Residential Exposure Scenario, Version 1.6"; Policy #WSC/ORS-142-92; Office of Research and Standards and the Bureau of Waste Site Cleanup, Boston, MA; September 1992

n-Butylbenzene. Butylbenzene is constituent of automotive gasoline. No toxicological data for this compound was identified in the literature. However, butylbenzenes is a close structural analog to ethylbenzene. Therefore, potential adverse effects and dose-response characteristics of butylbenzene is likely to be similar to those of ethylbenzene.

Ethylbenzene. Ethylbenzene is a naturally occurring and synthetically produced volatile hydrocarbon which is used in the manufacture of styrene and other plastics, and in gasoline, which contains approximately 2% ethylbenzene.

Ethylbenzene is readily absorbed through inhalation, oral, and dermal routes, and is distributed throughout the body. Exposures to ethylbenzene have been associated with central nervous system depression in humans, and liver, kidney, and hematopoietic system toxicity in laboratory animals. No evidence of carcinogenicity has been reported in human epidemiological studies, and animal evidence is equivocal.

References:
MADEP, 1992. "Risk Assessment Shortform Residential Exposure Scenario, Version 1.6"; Policy #WSC/ORS-142-92; Office of Research and Standards and the Bureau of Waste Site Cleanup, Boston, MA; September 1992.

Toluene. Toluene is a component of gasoline, paint, lacquers, adhesives, rubber, and printing ink. It is also used as a solvent and industrial degreaser. Acute and chronic exposures of toluene vapor to humans cause eye irritation, headache, nausea, and CNS effects. Acute exposures to very high concentrations can cause narcosis. Similar toxic effects have been observed in both humans and animals. The major target organ in animals is the CNS, with adverse effects including impaired motor abilities, narcosis, tremors, changes in levels of brain neurotransmitter, and morphological changes. Evidence from animal studies indicate that toluene is a developmental toxicant. Toluene is not thought to be a human carcinogen and has been classified as group D.

References:
Agency for Toxic Substances and Disease Registry (ATSDR), 1989. "Toxicological Profile for Toluene"; Agency for Toxic Substances and Disease Registry, U.S. Public Health Service, 1989.

Tetrachloroethene (Perchlorethene). Tetrachloroethene is a man-made volatile chlorinated solvent that is used extensively in the textile and dry cleaning industries as a cleanser and degreaser. Tetrachloroethene is also used as a degreaser in the electronics and metal industry. Since tetrachloroethene effectively cleans and decreases without adversely affecting what is being cleansed, tetrachloroethene is used extensively in a multitude of commercially available cleansers.

Tetrachloroethene is nearly completely absorbed via the inhalation and oral routes; dermal exposure represents a minor pathway. Oral and inhalation exposure to tetrachloroethene in humans and animals indicates that the liver, kidney, and nervous system are target organs. Long-term exposures to tetrachloroethene produced proliferative changes in the mouse livers, renal nephropathy in animals and occupationally exposed workers, and

irreversible nervous system damage in laboratory animals. Additionally, an increased incidence of menstrual disorders and spontaneous abortions have been observed in women occupationally exposed to tetrachloroethene in the dry cleaning business. Epidemiological data in humans is insufficient to make conclusions regarding the potential carcinogenicity of tetrachloroethene. However, tetrachloroethene has produced hepatic cancer in laboratory animals exposed orally and by inhalation. Therefore, the USEPA has placed tetrachloroethene in weight-of-evidence group B2, probable human carcinogen.

References: .
Agency for Toxic Substances and Disease Registry (ATSDR), 1991. "Toxicological Profile for Tetrachloroethene"; Agency for Toxic Substances and Disease Registry, U.S. Public Health Service. October, 1991

Trichloroethene. Trichloroethene is a man-made chlorinated solvent that is used extensively in industry as a metal decreasing agent. Trichloroethene is also used in dry cleaning and as a solvent in paints and adhesives.

Several human deaths and acute neurotoxic effects have been attributed to oral and inhalation exposure to trichloroethene. In animals, oral and inhalation exposure to trichloroethene has produced neurotoxic effects, including behavioral changes. Trichloroethene is also associated with renal toxicity. Additionally, inhalation and oral exposures to trichloroethene in animals have produced lung, liver, and testicular cancers. Epidemiological data in humans is insufficient to conclude whether trichloroethene is a human carcinogen. However, studies on trichloroethene metabolism suggest that it is metabolized similarly in humans and laboratory animals. Therefore, the USEPA has placed trichloroethene in weight-of-evidence group B2, probable human carcinogen.

References:
MADEP, 1992. "Risk Assessment Shortform Residential Exposure Scenario, Version 1.6"; Policy #WSC/ORS-142-92; Office of Research and Standards and the Bureau of Waste Site Cleanup, Boston, MA; September 1992.

Trimethylbenzene. Trimethylbenzenes are constituents of automotive gasoline. No toxicological data for 1,2,4- or 1,3,5-trimethylbenzene were identified in the literature. However, trimethylbenzenes are close structural analogs to xylene. Therefore, potential adverse effects and dose-response characteristics of trimethylbenzenes are likely to be similar to those of xylenes.

Vinyl Chloride. Most of the vinyl chloride produced in the United States is used in the manufacture of polyvinyl chloride and other vinyl polymers. Because vinyl chloride is a gas the only significant route of exposure is inhalation. It is highly flammable. Acute exposure to vinyl chloride causes CNS depression. Several epidemiologic studies have found associations between occupational exposure and impaired liver function to vinyl chloride. Symptoms of liver disease associated with occupational exposure include pain, hepatomegaly, portal hypertension, and thrombocytopenia. Carcinogenicity studies by inhalation and oral routes in rats, mice, and hamsters resulted in liver angiosarcomas in all animals tested. Vinyl chloride workers are at increased risk for developing liver

angiosarcomas, brain, skin, and lung tumors, and tumors of the lymph and blood-forming systems. Vinyl chloride is classified in group A, a human carcinogen.

References:
Clayton, George D. and Florence E. Clayton, editors, 1981. Patty's Industrial Hygiene and Toxicology, 3rd Revised Edition; John Wiley & Sons; New York.

Integrated Risk Information System (IRIS), 1993. United States Environmental Protection Agency.

Xylene. Xylene is a volatile organic compound that is generally composed of a mixture of the meta, ortho, and para isomers. Xylene are used as solvents, in paints, thinners, cleaners, degreasers, and as a component in gasoline.

Xylene are absorbed by oral, inhalation, and dermal exposures, and distribute to all tissues, particularly those with high fat contents. All three isomers produce similar effects, although the potency with which various effects are produced may vary from effect to effect with each isomer. In both humans and animals, xylene exposure has been associated with central nervous system depression, impaired learning and memory, and tremors. In humans, inhalation of xylene may produce prolonged respiratory tract inflammation and edema. In laboratory animals, exposures to xylene have produced adverse reproductive effects, including increased fetal death rate and retarded development. There is no evidence of carcinogenicity in humans or animals.

References:
MADEP, 1992. "Risk Assessment Shortform Residential Exposure Scenario, Version 1.6"; Policy #WSC/ORS-142-92; Office of Research and Standards and the Bureau of Waste Site Cleanup, Boston, MA; September 1992.

Cadmium. Cadmium is commonly used in electroplating and galvanizing due to its non-corrosive properties. It is a local respiratory tract irritant following inhalation exposure to cadmium dust or fumes. Acute exposure to cadmium dust/fumes may produce an acute chemical pneumonitis. Acute oral exposure to cadmium results in nausea, vomiting, salivation, abdominal pain, cramps, and diarrhea. Chronic exposure to cadmium results in osteomalacia and osteoporosis (Itai-Itai disease) secondary to renal damage. The USEPA has classified cadmium as a B1 carcinogen via inhalation based on epidemiological data from Japan and China.

References:
Amdur, Mary O., John Doull, Curtis D. Klaassen, 1991. Toxicology: The Basic Science of Poisons, 4th edition; Pergamon Press, Inc. New York.

Integrated Risk Information System (IRIS), 1993. United States Environmental Protection Agency.

Chromium. Chromium has been used in plating for corrosion resistance and decorative purposes, in the manufacture of alloys, and in printing, dying, and photography. The toxicity of chromium depends upon its valence state. Hexavalent chromium is more toxic via inhalation than trivalent chromium. The effects of inhalation exposure to hexavalent chromium include ulcers of the upper respiratory tract, nasal inflammation, perforation of the nasal septa and lung cancer. Most trivalent chromium compounds are inactive in short-term genotoxicity assays. Trivalent chromium compounds have not been found to be carcinogenic by any route of exposure, and are generally associated with minimal toxicity. There is epidemiological evidence of an association between chromium and lung cancer. The USEPA has classified hexavalent chromium as an Class A, human carcinogen, by the inhalation route.

References:
Amdur, Mary O., John Doull, Curtis D. Klaassen, 1991. Toxicology: The Basic Science of Poisons, 4th edition; Pergamon Press, Inc. New York.

Integrated Risk Information System (IRIS), 1993. United States Environmental Protection Agency.

Cyanide. Cyanide is commonly found in pesticides, fumigants, and photographic solutions. Symptoms of exposure include palpitations, shortness of breath, pain over the heart, vertigo, involuntary eye movements, cyanosis, and left-sided blindness. Longer term exposures has resulted in CNS, thyroid gland, and cardiovascular effects. Epidemiological studies have also found thyroid abnormalities in exposed workers. Animal studies have demonstrated that CNS and cardiac symptoms result from cyanide exposure. There is no available data indicating that cyanide has any carcinogenic effects.

References:
Amdur, Mary O., John Doull, Curtis D. Klaassen, 1991. Toxicology: The Basic Science of Poisons, 4th edition; Pergamon Press, Inc. New York.

Lead. Lead is used as a component in storage batteries and was widely used in gasoline and paints. It is the most ubiquitous toxic metal in the environment. The most serious effects of chronic exposure are encephalopathy, renal damage, and changes in the hematopoietic system, which is the most sensitive indicator of lead exposure. Peripheral nerve dysfunction is observed in adults at blood lead levels of 30 to 50 mg/dL-blood. The nervous systems of children are reported to be affected at levels of 15 mg/dL-blood (Benignus and others, 1981). Chronic lead exposure by workers through inhalation has

resulted in statistically significant increases in tumors. Oral exposures of lead salts in animals has been shown to increase tumor formation.

References:
Amdur, Mary O., John Doull, Curtis D. Klaassen, 1991. Toxicology: The Basic Science of Poisons, 4th edition; Pergamon Press, Inc. New York.

Benignus, V.A., Otto, D.A., Muller, K.E., Seiple, K.J., 1981. "Effects of Age and Body Lead Burden on CNS Function in Young Children. II:EEG Spectra." Electroencephalograph. Clin. Neurophysiol. 52:240-248.

Mercury. Mercury has been used in the past for medicinal purposes, however, occupational exposure to mercury can occur during mining, smelting, chloralkali production, and in the manufacturing of mercury containing products. There are three forms in which mercury can exist: elemental, inorganic, and organic. The chemical form of mercury and route of exposure determine toxicity. Elemental mercury causes behavioral effects and other nervous system damage if the vapor is inhaled. However, elemental mercury causes minimal toxicity if ingested; this is primarily attributable to the extremely low bioavailability of elemental mercury in the gastrointestinal tract. Ingestion of inorganic mercury salts may produce kidney damage. Inorganic mercury is soluble and can be absorbed in the gastrointestinal tract, making inorganic mercury bioavailable via the ingestion route. Organic mercury compounds target the central nervous system, and are bioavailable via the oral exposure route. Most organic mercury compounds that contaminate the environment can produce a toxic neuroencephalopathy (paresthesias, ataxia, spasticity, tremor, mental status changes, learning defects and neurasthenic symptoms). Some organic mercury compounds readily break down in the body to inorganic compounds and thus produce toxicity similar to that produced by inorganic mercury compounds. Some studies have indicated that mercury is genotoxic. It has not been classified as to its carcinogenicity by the USEPA.

References:
Amdur, Mary O., John Doull, Curtis D. Klaassen, 1991. Toxicology: The Basic Science of Poisons, 4th edition; Pergamon Press, Inc. New York.

Nickel. Nickel is used in metal alloys designed for high stress applications, and in electroplating operations. Toxic effects of nickel occur mainly through the inhalation route. Nickel is emitted into the air through fossil fuel combustion, incinerators, metallurgy, chemical manufacturing, cement manufacturing, and nickel recovery. Nickel exposure can cause dermatitis and eczema-like lesions at high exposure levels likely to occur only in the work place. The major adverse effects observed in humans are dermatitis, chemical pneumonitis, and lung and nasal cancers. Rats fed nickel in their diets showed no adverse effects on infertility, gestation, viability, or lactation in a three generation study. Occupational studies indicate that nickel is a carcinogen via inhalation. However, there is no evidence that it is a carcinogen via ingestion or dermal exposure.

References:

United States Environmental Protection Agency (USEPA), 1985. "Health Effect Assessment Document for Nickel." Office of Research and Development. Office of Emergency and Remedial Response.

Zinc. The most common use of zinc is as a protective coating for other metals. Zinc is emitted into the air during mining, refining, manufacturing, and combustion of zinc-containing materials. Zinc is an essential trace element and is found in many foods. Zinc compounds are of relatively low toxicity via ingestion, however, ingestion may result in gastrointestinal distress and diarrhea. Metal fume fever results from occupational inhalation of fumes of zinc oxide whose symptoms include chills, fever, and profuse sweating. There is no evidence that zinc is a carcinogen.

References:

Amdur, Mary O., John Doull, Curtis D. Klaassen, 1991. Toxicology: The Basic Science of Poisons, 4th edition; Pergamon Press, Inc. New York.

DOSE-RESPONSE TABLES

Table C-1
Oral Dose-Response Data
for Carcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Weight of Evidence	Oral Slope Factor (mg/kg/day)(-1)	Source	Test Species	Exposure Route	Tumor Type	Study Source
VOLATILES							
1,1,1-Trichloroethane	D	NE					
1,2-Dichloroethene (total)	D	NE					
4-Methyl-2-Pentanone	D	NE					
Benzene	A	2.9e-02	IRIS	Human	Inhalation	Leukemia	IRIS
1,2,4-Trimethylbenzene	D	NE					
1,3,5-Trimethylbenzene	D	NE					
n-Butylbenzene	D	NE					
Ethylbenzene	D	NE					
Tetrachloroethene	B2	5.2e-02	(1)				
Toluene	D	NE					
Trichloroethene	B2	1.1e-02	(1)				
Vinyl Chloride	A	1.9+00	HEAST		Oral-diet	Liver, lung	HEAST
Xylene (total)	D	NE					
Zinc	D	NE					

Table C-1
Oral Dose-Response Data
for Carcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Weight of Evidence	Oral Slope Factor (mg/kg/day)(-1)	Source	Test Species	Exposure Route	Tumor Type	Study Source
INORGANICS							
Cadmium	D	NE					
Chromium	D	NE					
Cyanide	D	NE					
Lead	B2	NE					
Mercury	C	NE	(2)				
Nickel	D	NE					

Notes:
ND = No Data
NE = Not Evaluated
Integrated Risk Information System (IRIS) on-line database search, current as of June 1996.
Health Effects Assessment Summary Tables (HEAST), current as of November 1995.
(1) This value was provided by the Environmental Criteria and Assessment Office (ECAO) of the USEPA in response to a specific request.
(2) Classification is based on mercury as mercuric chloride.

Weight of Evidence (route-specific):
A = Human carcinogen
B = Probable human carcinogen (B1 = limited human evidence; B2 = sufficient human evidence)
C = Possible human carcinogen
D = Not classifiable as to human carcinogenicity

Table C-2
Inhalation Dose-Response Data
for Carcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Weight of Evidence	Inhalation Slope Factor (mg/kg/day)(-1)	Source	Inhalation Unit Risk ($\mu g/m^3$)(-1)	Source	Test Species	Exposure Route	Tumor Type	Study Source
VOLATILES									
1,1,1-Trichloroethane	D	NE		NE					
1,2-Dichloroethene (total)	D	NE							
4-Methyl-2-Pentanone	D	NE		NE					
1,2,4-Trimethylbenzene	D	NE							
1,3,5-Trimethylbenzene	D	NE							
n-Butylbenzene	D	NE							
Benzene	A	2.9e-02	HEAST	8.3e-06	IRIS	Human	Inhalation	Leukemia	IRIS
Ethylbenzene	D	NE		NE					
Tetrachloroethene	B2	2.0e-03	(1)	5.8e-07	(1)				
Toluene	D	NE		NE					
Trichloroethene	B2	6.0e-03	(1)	1.7e-06	(1)				
Vinyl Chloride	A	3.0e-01	HEAST	8.4e-05	HEAST	Rat	Inhalation	Liver	HEAST

Table C-2
Inhalation Dose-Response Data
for Carcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Weight of Evidence	Inhalation Slope Factor (mg/kg/day)(-1)	Source	Inhalation Unit Risk ($\mu g/m^3$)(-1)	Source	Test Species	Exposure Route	Tumor Type	Study Source
Xylene (total)	D	NE		NE					
INORGANICS									
Cadmium	B1	NE		1.8e-03	IRIS	Human	Inhalation	Lung	IRIS
Chromium	A	4.1e+01	HEAST (2)	1.2e-02	IRIS (2)	Human	Inhalation	Lung	IRIS
Cyanide	D	NE		NE					
Lead	D	NE		NE					
Mercury	C	NE	(3)	NE					
Nickel	A	8.4e-01	HEAST (4)	2.4e-04	IRIS (4)	Human	Inhalation	Lung	IRIS
Zinc	D	NE		NE					

Notes:
NE = Not Evaluated
Integrated Risk Information System (IRIS) on-line database search, current as of June 1996.
Health Effects Assessment Summary Tables (HEAST), current as of November 1995.
(1) This value was provided by the Environmental Criteria and Assessment Office (ECAO) of the USEPA in response to a specific request.
(2) The toxicity values for chromium are based on chromium VI.
(3) Classification is based on mercury as mercuric chloride.
(4) The toxicity values for nickel are based on nickel refinery dust.
Weight of Evidence (route-specific):
A = Human carcinogen
B = Probable human carcinogen (B1 = limited human evidence; B2 = sufficient human evidence)
C = Possible human carcinogen
D = Not classifiable as to human carcinogenicity

Table C-3
Oral Dose-Response Data
for Noncarcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Chronic		Subchronic		Study Type	Confidence Level	Critical Effect	Test Animal	Uncertainty Factor	Study Source
	Oral RfD (mg/kg-day)	Source	Oral RfD (mg/kg-day)	Source						
VOLATILES										
1,1,1-Trichloroethane	3.5e-02	(5)	ND							
1,2-Dichloroethene (total)	9.0e-03	IRIS (6)	9.0e-03	HEAST (6)	Orak-drinking water	Medium	Hepatic lesions	Rat	1,000 $H.A.L._3$	IRIS
4-Methyl-2-Pentanone	8.0e-02	HEAST	8.0e-01	HEAST	Oral-gavage	Low	Liver/kidney toxicity	Rat	3000 H,A,S,D	HEAST
1,2,4-Trimethylbenzene	ND									
1,3,5-Trimethylbenzene	ND									
n-Butylbenzene	ND									
Benzene	3.0e-04	(1)	ND							
Ethylbenzene	1.0e-01	IRIS	ND		Oral-gavage	Low	Liver, kidney toxicity	Rat	1000 H,A,S	IRIS
Toluene	2.0e-01	IRIS	2.0e+00	HEAST	Oral-gavage	Medium	Changes in liver, kidney weight	Rat	1000 H,A,S	IRIS
Tetrachloroethene	1.0e-02	IRIS	1.0e-01	HEAST	Oral-gavage	Medium	Hepatotoxicity	Mouse	1000 H,A,S	IRIS
Trichloroethene	6.0e-03	(1)	ND							
Vinyl Chloride	ND		ND							
Xylene (total)	2.0e+00	IRIS	ND		Oral-gavage	Medium	Hyperactivity, decreased weight	Rat	100 H,A	IRIS

Table C-3
Oral Dose-Response Data
for Noncarcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Chronic		Subchronic		Study Type	Confidence Level	Critical Effect	Test Animal	Uncertainty Factor	Study Source
	Oral RfD (mg/kg-day)	Source	Oral RfD (mg/kg- day)	Source						
INORGANICS										
Cadmium Food	1.0e-03	IRIS	ND		Oral-diet	High	Proteinuria	Human	10 H	IRIS
Drinking Water	5.0e-04	IRIS	ND		Oral-drinking water	High	Proteinuria	Human	10 H	IRIS
Chromium	5.0e-03	IRIS (2)	2.0e-02	HEAST (2)	Oral-drinking water	Low	No effects observed	Rat	500 H,A,S	IRIS
Cyanide	2.0e-02	IRIS	2.0e-02	HEAST	Oral-diet	Medium	No effects observed	Rat	100 H,A	IRIS
Lead	ND		ND							
Inorganic Mercury	3.0e-04	IRIS (5)	3.0e-03	HEAST (5)	Oral-diet	High	Autoimmune effects	Rat	1000 H,A,S,L	IRIS
Nickel	2.0e-02	IRIS (4)	2.0e-02	HEAST (4)	Oral-diet	Medium	Decreased body, organ weights	Rat	300 H,A,D	IRIS
Zinc	3.0e-01	IRIS	3.0e-01	HEAST	Oral-diet supplement	Medium	Decrease in ESOD activity	Human	3 S	IRIS

Table C-3
Oral Dose-Response Data
for Noncarcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Chronic		Subchronic		Study Type	Confidence Level	Critical Effect	Test Animal	Uncertainty Factor	Study Source
	Oral RfD (mg/kg-day)	Source	Oral RfD (mg/kg-day)	Source						

Notes:
ND = No Data
NA = Not Applicable
Integrated Risk Information System (IRIS) on-line database search, current as of June 1996.
Health Effects Assessment Summary Tables (HEAST), current as of November 1995.
Environmental Criteria and Assessment Office (ECAO) of the USEPA in response to a specific request.
(1) This value was provided by the Environmental Criteria and Assessment Office (ECAO) of the USEPA in response to a specific request.
(2) The toxicity values for chromium are based on chromium VI.
(3) The ingestion RfD values for nickel are based on nickel, soluble salts.
(4) This mercury value is based on mercuric chloride.
(5) Value is an EPA-NLBA Regional Support Provisional Value published in "Risk-Based Concentration Table, January-June 1996" USEPA Region III (May 30, 1996).
(6) Values for 1,1-dichloroethene used as surrogates for 1,2-dichloroethene.
Uncertainty factors:
H = Variation in human sensitivity
A = Animal to human extrapolation
S = Extrapolation from subchronic to chronic NOAEL
L = Extrapolation from LOAEL to NOAEL
D = Inadequate data
M = Modifying factor

Table C-4
Inhalation Dose-Response Data for Noncarcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Chronic		Subchronic		Study Type	Confidence Level	Critical Effect	Test Animal	Uncertainty Factor	Study Source
	RfC ($\mu g/m^3$)	Source	RfC ($\mu g/m^3$)	Source						
VOLATILES										
1,1,1-Trichloroethane	ND	(4)	ND							
1,2-Dichloroethene (total)	ND		ND							
4-Methyl-2-Pentanone	8.0e+01	HEAST (1)	8.0e+02	HEAST (2)	Inhalation	Low	Liver, kidney effects	Rat	1000 H,A,S	HEAST
1,2,4-Trimethylbenzene	ND									
1,3,5-Trimethylbenzene	ND									
n-Butylbenzene	ND									
Benzene	ND		ND							
Ethylbenzene	1.0e+03	IRIS (5)	ND		Inhalation	Low	Developmental toxicity	Rat/ rabbit	300 H,A,S	IRIS
Tetrachloroethene	ND		ND							
Toluene	4.0e+02	IRIS (6)	ND		Inhalation	Medium	Neurological effects	Human	300 H,L,D	IRIS
Trichloroethylene	ND		ND							
Vinyl Chloride	ND		ND							
Xylene (total)	ND	(7)	ND							
INORGANICS										
Cadmium	ND		ND							

Table C-4
Inhalation Dose-Response Data
for Noncarcinogenic Effects

Human Health Risk Assessment
Ames Street Site
Rochester, NY

Chemical	Chronic		Subchronic		Study Type	Confidence Level	Critical Effect	Test Animal	Uncertainty Factor	Study Source
	RfC ($\mu g/m^3$)	Source	RfC ($\mu g/m^3$)	Source						
Chromium	ND		ND							
Cyanide	ND		ND							
Lead	ND		ND							
Elemental Mercury	3.0e-01	HEAST (3)	3.0e-01	HEAST (3)	Inhalation	Low	Neurotoxicity	Human	30 H,D	HEAST
Nickel	ND		ND							
Zinc	ND		ND							

Notes:
ND = No Data
NA = Not Applicable
Integrated Risk Information System (IRIS) on-line database search, current as of February 1996.
Health Effects Assessment Summary Tables (HEAST), current as November 1995.
(1) 4-Methyl-2-pentanone has a chronic inhalation RfD of 2.0e-02 mg/kg-day (HEAST).
(2) 4-Methyl-2-pentanone has a subchronic inhalation RfD of 2.0e-01 mg/kg-day (HEAST).
(3) The mercury RfC values were developed specifically for elemental mercury.
(4) 1,1,1-Trichloroethane has a chronic inhalation RfD of 2.9E-01 mg/kg/day which was withdrawn from IRIS.
(5) Ethylbenzene has a chronic inhalation RfD of 2.9E-01 mg/kg/day published by NYSDEC (1995).
(6) Toluene has a chronic inhalation RfD of 1.1E-01 mg/kg/day published by NYSDEC (1995).
(7) Xylene (mixed) has a chronic inhalation RfD of 8.6E-02 mg/kg/day published by NYSDEC (1995).
Uncertainty factors:
A = Animal to human extrapolation
H = Variation in human sensitivity
S = Extrapolation from subchronic to chronic NOAEL
L = Extrapolation from LOAEL to NOAEL
D = Inadequate data
M = Modifying factor

APPENDIX D
RISK-BASED CONCENTRATION CALCULATIONS

TABLE D-1
CALCULATION OF SITE-SPECIFIC TAGM VALUES

LCHTAGM1

HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
GA GROUNDWATER STANDARD	GA	chemical-specific	mg/L	NYSDEC, 1994
WATER AND SOIL PARTITION COEFFICIENT	Koc	chemical-specific	(cm^3/g)	[a]
CORRECTION FACTOR	CF	10	unitless	NYSDEC, 1994
FRACTION ORGANIC CARBON IN SOIL	foc	0.0154	unitless	Site-specific [b]

NYSDEC, 1994. Technical and Administrative Guidance Memorandum (TAGM). Determination of Soil Cleanup Objectives and Cleanup Levels (Revised). January 24, 1994.

USEPA, 1986. Superfund Public Health Evaluation Manual. Oswer Directive 9285.4-1. October, 1986. EPA/540/1-86/060

USEPA, 1993. Superfund Chemical Data Matrix. March 9, 1993.

Howard, Phillip H. "Handbook of Environmental Fate and Exposure Data for Organic Chemicals" Vol I and II. 1990.

[a] Average of Koc values reported in USEPA (1986, 1993), and Howard (1990)

[b] Average TOC concentration in soils collected for laboratory analyses.

EQUATIONS

$TAGM_{adj}$ (mg/kg) = GA x CF x Koc x foc

TABLE D-1, continued
CALCULATION OF SITE-SPECIFIC TAGM VALUES

HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

CALCULATIONS

COMPOUND	GA GROUNDWATER (mg/L)	Koc (cm^3/g)	$TAGM_{adj}$ (mg/kg)
ORGANICS			
Trichloroethene	5.00E-03	1.42E+02	1.09E-01
Tetrachloroethene	5.00E-03	6.10E+02	4.70E-01
Ethylbenzene	5.00E-03	1.01E+03	7.79E-01
Toluene	5.00E-03	3.15E+02	2.43E-01
Xylene (total)	5.00E-03	6.26E+02	4.82E-01
1,1,1-Trichloroethane	5.00E-03	9.90E+01	7.62E-02
trans-1,2-Dichloroethene	5.00E-03	4.10E+01	3.16E-02
4-Methyl-2-pentanone	5.00E-02	5.00E+00	3.85E-02

ND = No data available

TABLE D–2
RISK BASED SCREENING LEVEL – SURFICIAL SOIL DIRECT CONTACT
COMMERCIAL WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

CMSSRBCA	14–Nov–96

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
TARGET CANCER RISK	TR_c	1E–06	unitless	NYSDEC, 1995
TARGET NON–CANCER RISK	TR_{nc}	1	unitless	NYSDEC, 1995
INGESTION RATE	IRs	58.6	mg/day	ASTM, 1995
INHALATION RATE	IRa	20	m^3/day	ASTM, 1995
ADHERENCE FACTOR	M	0.5	mg/cm^2	ASTM, 1995
SURFACE AREA EXPOSED	SA	3,160	cm^2/day	ASTM, 1995
SOIL TO AIR VOLATILIZATION	VFss	chemical–specific	(mg/m^3–air)/(mg/kg–soil)	Calculated [a]
SOIL TO AIR PARTICULATES	VFp	2.30E–12	(mg/m^3–air)/(mg/kg–soil)	Calculated [a]
CONVERSION FACTOR	CF	1.00E–06	kg/mg	ASTM, 1995
BODY WEIGHT	BW	70	kg	ASTM, 1995
EXPOSURE FREQUENCY	EF	250	days/year	ASTM, 1995
EXPOSURE DURATION	ED	25	years	ASTM, 1995
AVERAGING TIME				
CANCER	AT	70	years	ASTM, 1995
NONCANCER	AT	25	years	ASTM, 1995

NYSDEC, 1995. Site Assessment and Closure Guidance for Petroleum Impacted Sites (Review Draft). Division of Spills Management. September 24, 1995.
ASTM, 1995. Emergency Standard Guide for Risk–Based Corrective Action Applied at Petroleum Release Sites (ASTM Stnd" E1739–95)
[a] Calculation documented in Table D–5.

EQUATIONS

$$RBSL_{cancer}\ (mg/kg) = \frac{TR_c \times BW \times AT \times 365\ days/yr}{EF \times ED \times [(CSF_o \times CF \times (IR_s \times RAF_o + SA \times M \times RAF_d)) + (CSF_i \times IR_a \times (VF_{ss} + VF_p))]}$$

$$RBSL_{non-cancer}\ (mg/kg) = \frac{TR_{nc} \times BW \times AT \times 365\ days/yr}{EF \times ED \times [(CF \times (IR_s \times RAF_o + SA \times M \times RAF_d))/RfD_o) + (IR_a \times (VF_{ss} + VF_p))/RfD_i)]}$$

Note: For noncarcinogenic effects AT = ED
RBSL = Risk Based Screening Level
CSF = Cancer Slope Factor
RfD = Reference Dose

TABLE D-2 , continued
RISK BASED SCREENING LEVEL – SURFICIAL SOIL DIRECT CONTACT
COMMERCIAL WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

CMSSRBCA	14-Nov-96

CARCINOGENIC EFFECTS

COMPOUND	VFss (mg/m^3-air)/ (mg/kg-soil)	ORAL RAF (unitless)	DERMAL RAF (unitless)	ORAL CSF (mg/kg-day)^-1	INHALATION CSF (mg/kg-day)^-1	RBSL TOTAL (mg/kg-soil)
ORGANICS						
Trichloroethene	5.87E-05	1	0.5	1.1E-02	6.0E-03	3.1E+01
Tetrachloroethene	3.88E-05	1	0.5	5.2E-02	2.0E-03	6.5E+00
Cadmium	NA	1	0.14	ND	ND	
Chromium	NA	1	0.09	ND	4.1E+01	1.5E+05
Nickel	NA	1	0.35	ND	8.4E-01	7.4E+06
Lead	NA	1	0.006	ND	ND	

ND = No data available
Oral RAFs and Dermal RAFs for volatiles and semivolatiles from NYSDEC, 1995. Dermal RAFs for inorganics from MADEP Residental Shortform (1992)

TABLE D-2, continued
RISK BASED SCREENING LEVEL - SURFICIAL SOIL DIRECT CONTACT
COMMERCIAL WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

CMSSRBCA | 14-Nov-96

NONCARCINOGENIC EFFECTS

COMPOUND	VFss (mg/m^3-air)/ (mg/kg-soil)	ORAL RAF (unitless)	DERMAL RAF (unitless)	ORAL RfD (mg/kg-day)	INHALATION RfD (mg/kg-day)	RBSL TOTAL (mg/kg-soil)
ORGANICS						
Trichloroethene	5.87E-05	1	0.5	6.0E-03	ND	7.2E+02
Tetrachloroethene	3.88E-05	1	0.5	1.0E-02	ND	1.2E+03
Cadmium	NA	1	0.14	1.0E-03	ND	3.7E+02
Chromium	NA	1	0.09	5.0E-03	ND	2.5E+03
Nickel	NA	1	0.35	2.0E-02	ND	3.3E+03
Ethylbenzene	1.74E-05	1	0.5	1.0E-01	2.9E-01	1.1E+04
Toluene	3.05E-05	1	0.5	2.0E-01	1.1E-01	1.0E+04
Xylenes (total)	2.14E-05	1	0.5	2.0E+00	8.6E-02	1.9E+04
Lead	NA	1	0.006	ND	ND	
Zinc	NA	1	0.02	3.0E-01	ND	3.4E+05
1,1,1-Trichloroethane	6.74E-05	1	0.5	3.5E-02	2.9E-01	3.5E+03
4-Methyl-2-Pentanone	1.60E-05	1	0.5	8.0E-02	2.0E-02	3.8E+03
Cyanide	NA	1	0.03	2.0E-02	ND	1.9E+04
Mercury	1.13E-07	0.2	ND	3.0E-04	ND	2.5E+03

ND = No Data
Oral RAFs and Dermal RAFs for volatiles and semivolatiles from NYSDEC, 1995. Dermal RAFs for inorganics from MADEP Residential Shortform (1992)

CWSSRBCA	14-Nov-96

TABLE D-3
RISK BASED SCREENING LEVEL - SOIL DIRECT CONTACT
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
TARGET CANCER RISK	TR_c	1E-06	unitless	NYSDEC, 1995
TARGET NON-CANCER RISK	TR_{nc}	1	unitless	NYSDEC, 1995
INGESTION RATE	IRs	58.6	mg/day	ASTM, 1995
INHALATION RATE	IRa	20	m^3/day	ASTM, 1995
ADHERENCE FACTOR	M	0.5	mg/cm^2	ASTM, 1995
SURFACE AREA EXPOSED	SA	3,160	cm^2/day	ASTM, 1995
SOIL TO AIR VOLATILIZATION	VFss	chemical-specific	(mg/m^3-air)/(mg/kg-soil)	Calculated [a]
SOIL TO AIR PARTICULATES	VFp	2.30E-07	(mg/m^3-air)/(mg/kg-soil)	Calculated [a]
CONVERSION FACTOR	CF	1.00E-06	kg/mg	ASTM, 1995
BODY WEIGHT	BW	70	kg	ASTM, 1995
EXPOSURE FREQUENCY	EF	250	days/year	ASTM, 1995
EXPOSURE DURATION	ED	1	years	ASTM, 1995
AVERAGING TIME				
CANCER	AT	70	years	ASTM, 1995
NONCANCER	AT	1	years	ASTM, 1995

NYSDEC, 1995. Site Assessment and Closure Guidance for Petroleum Impacted Sites (Review Draft). Division of Spills Management. September 24, 1995.
ASTM, 1995. Emergency Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Stnd. E1739-95)
[a] Calculation documented in Table D-5.

EQUATIONS

$$RBSL_{cancer}\ (mg/kg) = \frac{TR_c \times BW \times AT \times 365\ days/yr}{EF \times ED \times [(CSF_o \times CF \times (IR_s \times RAF_o + SA \times M \times RAF_d)) + (CSF_i \times IR_a \times (VF_{ss} + VF_p))]}$$

$$RBSL_{non-cancer}\ (mg/kg) = \frac{TR_{nc} \times BW \times AT \times 365\ days/yr}{EF \times ED \times [(CF \times (IR_s \times RAF_o + SA \times M \times RAF_d))/RfD_o) + (IR_a \times (VF_{ss} + VF_p))/RfD_i)]}$$

Note: For noncarcinogenic effects: AT = ED
RBSL = Risk Based Screening Level
CSF = Cancer Slope Factor
RfD = Reference Dose

TABLE D-3, continued
RISK BASED SCREENING LEVEL - SOIL DIRECT CONTACT
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

CWSSRBCA	14-Nov-96

CARCINOGENIC EFFECTS

COMPOUND	VFss ((mg/m^3-air)/(mg/kg-soil))	ORAL RAF (unitless)	DERMAL RAF (unitless)	ORAL CSF ((mg/kg-day)^-1)	INHALATION CSF ((mg/kg-day)^-1)	RBSL TOTAL (mg/kg-soil)
ORGANICS						
Trichloroethene	2.94E-04	1	0.5	1.1E-02	6.0E-03	7.5E+02
Tetrachloroethene	1.94E-04	1	0.5	5.2E-02	2.0E-03	1.6E+02
Cadmium	NA	1	0.14	ND	ND	
Chromium	NA	1	0.09	ND	4.1E+01	3.8E+01
Nickel	NA	1	0.35	ND	8.4E-01	1.9E+03
Lead	NA	1	0.006	ND	ND	

ND = No data available
Oral RAFs and Dermal RAFs for volatiles and semivolatiles from NYSDEC, 1995. Dermal RAFs for inorganics from MADEP Residental Shortform (1992)

TABLE D-3, continued
RISK BASED SCREENING LEVEL - SOIL DIRECT CONTACT
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

CWSSRBCA	14-Nov-96

NONCARCINOGENIC EFFECTS

COMPOUND	VFss (mg/m^3-air)/(mg/kg-soil)	ORAL RAF (unitless)	DERMAL RAF (unitless)	ORAL RfD (mg/kg-day)	INHALATION RfD (mg/kg-day)	RBSL TOTAL (mg/kg-soil)
ORGANICS						
Trichloroethene	2.94E-04	1	0.5	6.0E-03	ND	7.2E+02
Tetrachloroethene	1.94E-04	1	0.5	1.0E-02	ND	1.2E+03
Cadmium	NA	1	0.14	1.0E-03	ND	3.7E+02
Chromium	NA	1	0.09	5.0E-03	ND	2.5E+03
Nickel	NA	1	0.35	2.0E-02	ND	3.3E+03
Ethylbenzene	8.73E-05	1	0.5	1.0E-01	2.9E-01	7.0E+03
Toluene	1.52E-04	1	0.5	2.0E-01	1.1E-01	3.2E+03
Xylenes (total)	1.07E-04	1	0.5	2.0E+00	8.6E-02	4.0E+03
Lead	NA	1	0.006	ND	ND	
Zinc	NA	1	0.02	3.0E-01	ND	3.4E+05
1,1,1-Trichloroethane	3.37E-04	1	0.5	3.5E-02	2.9E-01	2.1E+03
4-Methyl-2-Pentanone	9.75E-05	1	0.5	8.0E-02	2.0E-02	9.4E+02
Cyanide	NA	1	0.03	2.0E-02	ND	1.9E+04
Mercury	5.68E-07	0.2	ND	3.0E-04	ND	2.5E+03

ND = No Data
Oral RAFs and Dermal RAFs for volatiles and semivolatiles from NYSDEC, 1995. Dermal RAFs for inorganics from MADEP Residential Shortform (1992)

TABLE D-4
RISK BASED SCREENING LEVEL – AMBIENT AIR
COMMERICAL WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
TARGET CANCER RISK	TR_c	1E-06	unitless	NYSDEC, 1995
TARGET NON-CANCER RISK	TR_{nc}	1	unitless	NYSDEC, 1995
INHALATION RATE	IR	20	m^3/day	ASTM, 1995
BODY WEIGHT	BW	70	kg	ASTM, 1995
EXPOSURE FREQUENCY	EF	250	days/year	ASTM, 1995
EXPOSURE DURATION	ED	25	years	ASTM, 1995
AVERAGING TIME				
CANCER	AT	70	years	ASTM, 1995
NONCANCER	AT	25	years	ASTM, 1995

NYSDEC, 1995. Site Assessment and Closure Guidance for Petroleum Impacted Sites (Review Draft). Division of Spills Management. September 24, 1995.

ASTM, 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Stnd. E1739-95)

EQUATIONS

$$\mathbf{RBSL_{cancer}\ (mg/m^3) = \frac{TR_c \times BW \times AT \times 365\ days/yr}{IR \times ED \times EF \times CSF}}$$

$$\mathbf{RBSL_{non-cancer}\ (mg/m^3) = \frac{TR_{nc} \times BW \times AT \times 365\ days/yr \times RfD}{IR \times ED \times EF}}$$

Note:

For noncarcinogenic effects: AT = ED

RBSL = Risk Based Screening Level
CSF = Cancer Slope Factor
RfD = Reference Dose

TABLE D-4, continued
RISK BASED SCREENING LEVEL - AMBIENT AIR
COMMERICAL WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

CARCINOGENIC EFFECTS

COMPOUND	INHALATION CSF (mg/kg-day)^-1	RBSL (mg/m^3)
ORGANICS		
Benzene	2.90E-02	4.93E-04
Trichloroethene	6.00E-03	2.38E-03
Tetrachloroethene	2.00E-03	7.15E-03
Vinyl chloride	3.00E-01	4.77E-05

ND = No data available

TABLE D-4, continued
RISK BASED SCREENING LEVEL - AMBIENT AIR
COMMERICAL WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

NONCARCINOGENIC EFFECTS

COMPOUND	INHALATION RfD (mg/kg-day)	RBSL (mg/m^3)
ORGANICS		
1,2-Dichloroethene	ND	
1,1,1-Trichloroethane	2.90E-01	1.48E+00
1,2,4-Trimethylbenzene	ND	
1,3,5-Trimethylbenzene	ND	
4-Methyl-2-Pentanone	2E-02	1.18E-01
Benzene	ND	
n-Butylbenzene	ND	
Ethylbenzene	2.90E-01	1.48E+00
Tetrachloroethene	ND	
Toluene	1.10E-01	5.62E-01
Trichloroethene	ND	
Vinyl chloride	ND	
Xylene (total)	8.60E-02	4.39E-01

ND = No data available

TABLE D-5

This Table presents the chemical fate and transport calculations which were used to estimate chemical transfer from soil and groundwater to ambient air and indoor air. The equations used for modelling these exposure pathways are based on the approach described in *Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites* (ASTM Standard E1739-95; November, 1995).

Specifically, equations for the following exposure pathways are presented in this Table:

- Vapor emissions from surface soil to ambient air
- Particulate emissions from surface soil to ambient air
- Vapor migration from subsurface soil sources to indoor air
- Vapor migration from subsurface soil sources to ambient air
- Vapor migration from groundwater sources to indoor air
- Vapor migration from groundwater sources to ambient air

The equations used to model chemical transfer for these exposure pathways are summarized below. The parameters used in the calculations are presented in Table D-5.1. The calculations are documented in Tables D-5.2 and D-5.3.

A) Vapor emissions from surface soil to ambient air

The equations used to estimate the vapor concentration in ambient air resulting from surface soil vapor emissions are as follows:

$$VF_{ss} = \frac{W * \rho_s * d}{U_a * \delta_a * \tau} * CF_1$$

and

$$VF_{ss} = \frac{2W \rho_s}{U_{air} * L_{air} * T} * \left[\frac{D_{ws}^{eff} * H}{\pi [\theta_{ws} + K_s * \rho_s + H * \theta_{as}] T} \right]^{1/2} * CF_1$$

The parameters used in these equations are described in Table D-5.1. The products of these equations, presented in Table D-5.2, are used as the values for quantifying direct-contact exposures to vapor emissions in the direct contact exposure estimates (Tables D-2,

D-3, and D-9). Specifically, the equation which results in the highest emission rate, and thus produces the lowest RBC, is used in the direct-contact equations.

B) Particulate emissions from surface soil to ambient air

The equation used to estimate the particulate concentration in ambient air resulting from surface soil particulate emissions is as follows:

$$VF_p = \frac{P_e * W}{U_a * \delta_a} * CF_1$$

The parameters used in this equation are described in Table D-5.1. The products of this equation, presented in Table D-5.2, are used as the values for quantifying direct-contact exposures to particulate emissions in the direct contact exposure estimates (Tables D-2, D-3, and D-9).

C) Vapor migration from subsurface soil to indoor air

The equation used to estimate the vapor concentration in indoor air resulting from vapor migration from subsurface soil sources is as follows:

$$VF_{sesp} = \frac{\frac{H * \rho_s}{[\,\theta_{ws} + K_s * \rho_s + H * \theta_{as}\,]} * \left[\frac{D_s^{eff} / L_s}{ER * L_B}\right]}{1 + \left[\frac{D_s^{eff} / L_s}{ER * L_B}\right] + \left[\frac{D_s^{eff} / L_s}{(D_{crack}^{eff} / L_{crack})\, n}\right]} * CF_1$$

where:

$$D_{crack}^{eff} = D^a * \frac{\theta_{acrack}^{3.33}}{\theta_T^{2.0}} + D^w * \frac{1}{H} * \frac{\theta_{wcrack}^{3.33}}{\theta_T^{2.0}}$$

and

$$D_s^{eff} = D^a * \frac{\theta_{as}^{3.33}}{\theta_T^{2.0}} + D^w * \frac{1}{H} * \frac{\theta_{ws}^{3.33}}{\theta_T^{2.0}}$$

The parameters used in these equations are described in Table D-5.1. The products of

these equations, presented in Table D-5.3, are used as the values for quantifying the vapor concentrations in indoor air that may result from volatile emissions from subsurface soil sources. These values are compared to risk-based concentrations for indoor air (Table D-4), to develop risk-based concentrations in soil for this pathway (Table D-6).

D) Vapor migration from subsurface soil to ambient air

The equation used to estimate the vapor concentration in ambient air resulting from vapor migration from subsurface soil sources is as follows:

$$VF_{samb} = \frac{H * \rho_s}{(\theta_{ws} + K_s * \rho_s + H * \theta_{as}) * \left(1 + \frac{U_a * \delta_a * L_s}{D_s^{eff} * W}\right)} * CF_1$$

The parameters used in these equations are described in Table D-5.1. The products of these equations, presented in Table D-5.3, are used as the values for quantifying the vapor concentrations in ambient air that may result from volatile emissions from subsurface soil sources. These values are compared to risk-based concentrations for ambient air (Tables D-7 and D-10), to develop risk-based concentrations in soil for this pathway (Tables D-8 and D-11).

E) Vapor migration from groundwater to indoor air

The equation used to estimate the vapor concentration in indoor air resulting from vapor migration from groundwater sources is as follows:

$$VF_{wesp} = \frac{H * \left[\frac{D_{ws}^{eff} / L_{GW}}{ER * L_B}\right]}{1 + \left[\frac{D_{ws}^{eff} / L_{GW}}{ER * L_B}\right] + \left[\frac{D_{ws}^{eff} / L_{GW}}{(D_{crack}^{eff} / L_{crack})\, n}\right]} * CF_2$$

where:

$$D_{ws}^{eff} = (h_{cap} + h_v) * \left[\frac{h_{cap}}{D_{cap}^{eff}} + \frac{h_v}{D_{crack}^{eff}}\right]^{-1}$$

and

$$D_{cap}^{eff} = D^{a} * \frac{\theta_{acap}^{3.33}}{\theta_{T}^{2.0}} + D^{w} * \frac{1}{H} * \frac{\theta_{wcap}^{3.33}}{\theta_{T}^{2.0}}$$

The parameters used in these equations are described in Table D-5.1. The products of these equations, presented in Table D-5.3, are used as the values for quantifying the vapor concentrations in indoor air that may result from volatile emissions from groundwater sources. These values are compared to risk-based concentrations for indoor air (Table D-4), to develop risk-based concentrations in groundwater for this pathway (Table D-6).

F) Vapor migration from groundwater to ambient air

The equation used to estimate the vapor concentration in ambient air resulting from vapor migration from groundwater sources is as follows:

$$VF_{wamb} = \frac{H}{1 + \left[\frac{U_a * \delta_a * L_{GW}}{W * D_{ws}^{eff}}\right]} * CF_2$$

The parameters used in these equations are described in Table D-5.1. The products of these equations, presented in Table D-5.3, are used as the values for quantifying the vapor concentrations in ambient air that may result from volatile emissions from groundwater sources. These values are compared to risk-based concentrations for ambient air (Tables D-7 and D-10), to develop risk-based concentrations in groundwater for this pathway (Tables D-8 and D-11).

TABLE D--5.1
PARAMETERS FOR CALCULATION OF VOLATILIZATION FACTORS FOR SUBSURFACE SOIL AND GROUNDWATER
HUMAN HEALTH RISK ASSESSMENT
ROCHESTER, NY

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
HENRY'S LAW CONSTANT	H	chemical-specific	dimensionless	[a,h]
SOIL BULK DENSITY	Ps	1.70E+00	g-soil/cm^3-soil	ASTM, 1995
WATER CONTENT VADOSE ZONE SOILS	θ_{ws}	0.12	cm^3-water/cm^3-soil	Site-specific [b]
AIR CONTENT VADOSE ZONE SOILS	θ_{as}	0.26	cm^3-air/cm^3-soil	ASTM, 1995
SOIL-WATER PARTITION COEFFICIENT	Ks	chemical-specific	cm^3-water/g-soil	USEPA, 1986 [a,c]
DEPTH TO SUBSURFACE SOIL SOURCES	L_s	91	cm	Site-specific [d]
ENCLOSED SPACE VOL./INFILT. AREA RATIO	L_b	300	cm	ASTM, 1995
DEPTH TO GROUNDWATER	L_{gw}	198	cm	Site-specific [e]
ENCLOSED SPACE AIR EXCHANGE RATE	ER	0.00023	sec^{-1}	ASTM, 1995
ENCLOSED SPACE WALL THICKNESS	L_{crack}	15	cm	ASTM, 1995
AREAL FRACTION OF CRACKS IN WALLS	N	0.0008	cm^2-crack/cm^2-total	[i]
DIFFUSION COEFFICIENT IN SOIL	D_s	chemical-specific	cm^2/sec	ASTM, 1995 [f]
DIFFUSION COEFFICIENT THROUGH CRACKS	D_{crack}	chemical-specific	cm^2-crack/cm^2-total	ASTM, 1995 [f]
DIFFUSION COEFFICIENT: SOIL AND GROUNDWATER	D_{ws}	chemical-specific	cm^2/sec	ASTM, 1995 [f]
WIND SPEED ABOVE GROUND MIXING ZONE	U_a	225	cm/sec	ASTM, 1995
AMBIENT AIR MIXING ZONE HEIGHT	L_a	200	cm	ASTM, 1995
WIDTH OF SOURCE AREA PARALLEL TO GW FLOW	W	1500	cm	ASTM, 1995
DIFFUSION COEFFICIENT IN AIR	D^a	chemical-specific	cm^2/sec	USEPA, 1988 [a]
DIFFUSION COEFFICIENT IN WATER	D^w	chemical-specific	cm^2/sec	Lyman, et al. (1990)
SOIL POROSITY IN IMPACTED ZONE	θ_T	0.38	cm^3/cm^3 soil	ASTM, 1995
THICKNESS OF CAPILLARY FRINGE	h_{cap}	25	cm	Site-specific [g]
THICKNESS OF VADOSE ZONE	h_v	127	cm	Site-specific
DIFFUSION THROUGH CAPILLARY FRINGE	D_{cap}	chemical-specific	cm^3/cm^3	ASTM, 1995 [f]
AIR CONTENT CAPILLARY FRINGE	θ_{acap}	0.038	cm^3-air/cm^3-soil	ASTM, 1995
WATER CONTENT CAPILLARY FRINGE	θ_{wcap}	0.342	cm^3-water/cm^3-soil	ASTM, 1995
AIR CONTENT IN WALL CRACKS	θ_{acrack}	0.26	cm^3-air/cm^3-tot.vol.	ASTM, 1995
WATER CONTENT WALL CRACKS	θ_{wcrack}	0.12	cm^3-water/cm^3-tot.vol.	ASTM, 1995
PARTICULATE EMISSION RATE - COMMERCIAL WORKER	P_e	6.90E-14	g/cm^3-sec	ASTM, 1995
PARTICULATE EMISSION RATE - CONSTRUCTION WORKER	P_e	6.90E-09	g/cm^3-sec	ASTM, 1995
LOWER DEPTH OF SURFICIAL SOIL ZONE	d	15.24	cm	NYSDEC, 1995
AVERAGING TIME FOR VAPOR FLUX	T	receptor-specific	sec	ASTM, 1995
VOLATILIZATION FACTOR - SUBSURFACE SOIL:INDOOR AIR	VF_{sesp}	chemical-specific	mg/m^3 per mg/kg	ASTM, 1995 [f]
VOLATILIZATION FACTOR - SUBSURFACE SOIL:AMBIENT AIR	VF_{samb}	chemical-specific	mg/m^3 per mg/kg	ASTM, 1995 [f]
VOLATILIZATION FACTOR - GROUNDWATER:INDOOR AIR	VF_{wesp}	chemical-specific	mg/m^3 per mg/L	ASTM, 1995 [f]
VOLATILIZATION FACTOR - GROUNDWATER:AMBIENT AIR	VF_{wamb}	chemical-specific	mg/m^3 per mg/L	ASTM, 1995 [f]
VAPOR EMISSION - SURFACE SOIL:AMBIENT AIR	VF_{ss}	chemical-specific	mg/m^3	ASTM, 1995 [f]
PARTICULATE EMISSION - SURFACE SOIL:AMBIENT AIR	VF_p	chemical-specific	mg/m^3	ASTM, 1995 [f]
CONVERSION FACTOR 1	CF_1	1.0E+03	cm^3-kg/m^3-g	ASTM, 1995
CONVERSION FACTOR 2	CF_2	1.0E+03	L/m^3	ASTM, 1995

NOTES:

ASTM, 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Standard E1739-95)
NYSDEC, 1995. Site Assessment and Closure Guidance for Petroleum Impacted Sites (Review Draft). Division of Spills Management. September 24, 1995.
USEPA, 1986. Superfund Public Health Evaluation Manual. Oswer Directive 9285.4-1. October, 1986. EPA/540/1-86/060
USEPA, 1988. Superfund Exposure Assessment Manual. Oswer Directive 9285.5-1. April, 1988. EPA/540/1-88/001
USEPA, 1993. Superfund Chemical Data Matrix. March 9, 1993.
Howard, Phillip H. "Handbook of Environmental Fate and Exposure Data for Organic Chemicals" Vol I and II. 1990.
Lyman, et al. (1990). Handbook of Chemical Property Estimation Methods. American Chemical Society, Washington. Chpt. 17-5.
Values calculated using the following equation: $D_{BW} = 13.26E-5/N_W \text{^} 1.14 \times V_B \text{^} 0.589$, where $N_W = 1.307$ at 10C and V_B from Lyman (1990).
[a] For mercury, values were obtained from "User's Manual for HgSCREEN. A Risk-Based Screening Model for Mercury-Contaminated Sites (GRI, 1994).
[b] Average water content in soils collected for laboratory analyses.
[c] Ks = Koc x foc; foc = 0.0154 (site-specific average TOC). Koc values represent the average of values reported in USEPA (1986) and those calculated using a standard fate and transport algorithm (Koc = Kow x 0.63).
[d] Average depth of sources at Site.
[e] Average annual water table.
[f] Calculated below.
[g] Value for soils composed of medium sand from Todd (1980); value used to represent sandy soil conditions at Site.
[h] Average of Henry's law constant values reported in USEPA (1986), USEPA (1993), and Howard (1990), adjusted for subsurface temperature of 10 C.
[i] Value represents the maximum volume changes for a concrete floor (Portland Cement Association, 12th Ed); see text Section 4.
NOTE: All equations presented in the text accompanying this Table.

D-5.2
CALCULATION OF VOLATILIZATION FACTORS FOR SURFACE SOIL
HUMAN HEALTH RISK ASSESSMENT
AMES STREET SITE
ROCHESTER, NY

CHEMICAL	T – com.wrkr (sec)	T – con.wrkr (sec)	$VF_{ss}A$ – com.wrk (mg/m^3)	$VF_{ss}A$ – con.wrk (mg/m^3)	VF_p – com. wrk. (mg/m^3)	VF_p – con. wrk. (mg/m^3)	$VF_{ss}B$ – com.wrk	$VF_{ss}B$ – con.wrk
1,1,1-Trichloroethane	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	6.74E-05	3.37E-04
4-Methyl-2-Pentanone	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	1.60E-05	8.01E-05
1,2-Dichloroethene	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	6.97E-05	3.49E-04
Ethylbenzene	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	1.74E-05	8.73E-05
Toluene	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	3.05E-05	1.52E-04
Trichloroethene	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	5.87E-05	2.94E-04
Vinyl chloride	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	2.01E-04	1.00E-03
Xylene (total)	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	2.14E-05	1.07E-04
Tetrachloroethene	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	3.88E-05	1.94E-04
Mercury	7.88E+08	3.15E+07	1.10E-06	2.74E-05	2.30E-12	2.30E-07	1.13E-07	5.68E-07
Cadmium	7.88E+08	3.15E+07	NA	NA	2.30E-12	2.30E-07	NA	NA
Chromium	7.88E+08	3.15E+07	NA	NA	2.30E-12	2.30E-07	NA	NA
Cyanide	7.88E+08	3.15E+07	NA	NA	2.30E-12	2.30E-07	NA	NA
Lead	7.88E+08	3.15E+07	NA	NA	2.30E-12	2.30E-07	NA	NA
Nickel	7.88E+08	3.15E+07	NA	NA	2.30E-12	2.30E-07	NA	NA
Zinc	7.88E+08	3.15E+07	NA	NA	2.30E-12	2.30E-07	NA	NA

Note: The VFss "A" and VFss "B" value selected for calculation of RBCs is the one which results in the lower RBC.

D-5.3
CALCULATION OF VOLATILIZATION FACTORS FOR SUBSURFACE SOIL AND GROUNDWATER
HUMAN HEALTH RISK ASSESSMENT
ROCHESTER, NY

CHEMICAL	H (unitless)	Ks (cm^3/g)	D^a (cm^2/sec)	D^w (cm^2/sec)	D_{cap} (cm^3/cm^3)	D_s (cm^2/sec)	D_{crack} (cm^2 crack/cm^2 total)	D_{ws} (cm^2/sec)	VF_{sesp} – indoor $(mg/m^3)/(mg/kg)$	VF_{samb} – outdoor $(mg/m^3)/(mg/kg)$	VF_{wesp} – indoor $(mg/m^3)/(mg/L)$	VF_{wamb} – outdoor $(mg/m^3)/(mg/L)$
1,1,1–Trichloroethane	5.7E–01	2.0E+00	7.50E–02	5.99E–06	1.17E–05	5.85E–03	5.85E–03	7.06E–05	1.17E–03	5.63E–04	1.36E–03	6.77E–06
4–Methyl–2–Pentanone	3.4E–03	8.0E–02	7.38E–02	5.39E–06	3.18E–04	5.77E–03	5.77E–03	1.51E–03	9.83E–05	4.73E–05	1.44E–05	8.63E–07
1,2–Dichloroethene	3.3E–01	1.0E+00	7.44E–02	7.07E–06	1.38E–05	5.81E–03	5.81E–03	8.28E–05	1.29E–03	6.23E–04	8.43E–04	4.60E–06
Ethylbenzene	3.3E–01	1.6E+01	6.27E–02	5.00E–06	1.10E–05	4.90E–03	4.90E–03	6.64E–05	7.59E–05	3.65E–05	6.96E–04	3.69E–06
Toluene	2.7E–01	5.0E+00	7.37E–02	5.48E–06	1.34E–05	5.75E–03	5.75E–03	8.05E–05	2.34E–04	1.13E–04	6.89E–04	3.72E–06
Trichloroethene	4.2E–01	2.0E+00	7.64E–02	6.23E–06	1.27E–05	5.96E–03	5.96E–03	7.67E–05	8.88E–04	4.27E–04	1.05E–03	5.42E–06
Vinyl chloride	1.7E+00	6.0E–01	1.01E–01	8.61E–06	1.40E–05	7.88E–03	7.88E–03	8.45E–05	1.09E–02	5.25E–03	5.17E–03	2.42E–05
Xylene (total)	2.9E–01	1.0E+01	6.74E–02	5.00E–06	1.21E–05	5.26E–03	5.26E–03	7.25E–05	1.14E–04	5.50E–05	6.61E–04	3.54E–06
Tetrachloroethene	8.4E–01	9.0E+00	6.97E–02	5.61E–06	1.03E–05	5.44E–03	5.44E–03	6.20E–05	3.76E–04	1.81E–04	1.82E–03	8.77E–06
Benzene	2.4E–01	1.0E+00	8.19E–02	6.10E–06	1.55E–05	6.39E–03	6.39E–03	9.32E–05	1.05E–03	5.05E–04	6.81E–04	3.76E–06
1,2,4–Trimethylbenzene	ND	ND	ND	ND								
1,3,5–Trimethylbenzene	ND	ND	ND	ND								
n–Butylbenzene	ND	ND	ND	ND								
Mercury	4.7E–01	1.0E+04	1.14E–01	1.58E–05	2.12E–05	8.90E–03	8.90E–03	1.27E–04	3.19E–07	1.54E–07	1.86E–03	1.02E–05

TABLE D-6
RISK BASED SCREENING LEVEL - VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
COMMERCIAL WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
RISK BASED SCREENING LEVEL - AIR; CARCINOGENIC	$RBSL_{cancer}$	chemical-specific	mg/kg	Calculated [a]
RISK BASED SCREENING LEVEL - AIR; NON-CARCINOGENIC	$RBSL_{non-cancer}$	chemical-specific	mg/kg	Calculated [a]
SUBSURFACE SOIL VOLATILIZATION TO INDOOR AIR	VF_{soil}	chemical-specific	$(mg/m^3)/(mg/kg)$	Calculated [b]
GROUNDWATER VOLATILIZATION TO INDOOR AIR	$VF_{groundwater}$	chemical-specific	$(mg/m^3)/(mg/L)$	Calculated [b]
SURFACE SOIL VOLATILIZATION TO OUTDOOR AIR	VF_{ss}	chemical-specific	$(mg/m^3)/(mg/kg)$	Calculated [b]

ASTM, 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM stnd. E1739-95).
[a] Calculated in Table D-4.
[b] Calculated in Table D-5.

EQUATIONS

$$RBSL_{soil}\ (mg/kg) = \frac{RBSL_{air}}{VF_{soil\ -\ indoor}}$$

$$RBSL_{groundwater}\ (mg/kg) = \frac{RBSL_{air}}{VF_{groundwater\ -\ indoor}}$$

Note:
For noncarcinogenic effects: AT = ED
Calculations are repeated for cancer and non-cancer effects
RBSL = Risk Based Screening Level

TABLE D-6, continued
RISK BASED SCREENING LEVEL - VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
COMMERCIAL WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

CARCINOGENIC EFFECTS

COMPOUND	RBSL AMBIENT AIR (mg/m^3)	VF SOIL (indoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (indoor) (mg/m^3)/(mg/L)	VF SOIL (outdoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (outdoor) (mg/m^3)/(mg/L)	RBSL SOIL (indoor) (mg/kg)	RBSL GROUNDWATER (indoor) (mg/L)	RBSL SOIL (outdoor) (mg/kg)	RBSL GROUNDWATER (outdoor) (mg/L)
ORGANICS									
Trichloroethene	2.38E-03	8.88E-04	1.05E-03	NA	5.42E-06	2.68E+00	2.27E+00		4.39E+02
Vinyl chloride	4.77E-05	1.09E-02	5.17E-03	NA	2.42E-05	4.38E-03	9.23E-03		1.97E+00
Tetrachloroethene	7.15E-03	3.76E-04	1.82E-03	NA	8.77E-06	1.90E+01	3.93E+00		8.15E+02
Benzene	4.93E-04	1.05E-03	6.81E-04	NA	3.76E-06	4.70E-01	7.24E-01		1.31E+02

ND = No data available

TABLE D-6, continued
RISK BASED SCREENING LEVEL - VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
COMMERCIAL WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

NONCARCINOGENIC EFFECTS

COMPOUND	RBSL AMBIENT AIR (mg/m^3)	VF SOIL (indoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (indoor) (mg/m^3)/(mg/L)	VF SOIL (outdoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (outdoor) (mg/m^3)/(mg/L)	RBSL SOIL (indoor) (mg/kg)	RBSL GROUNDWATER (indoor) (mg/L)	RBSL SOIL (outdoor) (mg/kg)	RBSL GROUNDWATER (outdoor) (mg/L)
ORGANICS									
1,1,1-Trichloroethane	1.48E+00	1.17E-03	1.36E-03	NA	6.77E-06	1.26E+03	1.09E+03		2.19E+05
4-Methyl-2-Pentanone	1.18E-01	9.83E-05	1.44E-05	NA	8.63E-07	1.20E+03	8.16E+03		1.36E+05
1,2-Dichloroethene	ND	1.29E-03	8.43E-04	NA	1.41E-05	0.00E+00	0.00E+00		0.00E+00
Ethylbenzene	1.48E+00	7.59E-05	6.96E-04	NA	3.69E-06	1.95E+04	2.13E+03		4.02E+05
Toluene	5.62E-01	2.34E-04	6.89E-04	NA	3.72E-06	2.40E+03	8.16E+02		1.51E+05
Trichloroethene	ND	8.88E-04	1.05E-03	NA	5.42E-06	0.00E+00	0.00E+00		0.00E+00
Vinyl chloride	ND	1.09E-02	5.17E-03	NA	2.42E-05	0.00E+00	0.00E+00		0.00E+00
Xylene	4.39E-01	1.14E-04	6.61E-04	NA	3.54E-06	3.85E+03	6.65E+02		1.24E+05
Tetrachloroethene	ND	3.76E-04	1.82E-03	NA	8.77E-06	0.00E+00	0.00E+00		0.00E+00
Benzene	ND	1.05E-03	6.81E-04	NA	3.76E-06	0.00E+00	0.00E+00		0.00E+00
1,2,4-Trimethylbenzene	ND	ND	ND	NA	ND				
1,3,5-Trimethylbenzene	ND	ND	ND	NA	ND				
n-Butylbenzene	ND	ND	ND	NA	ND				
Mercury	3.00E-04	7.28E-08	4.25E-04	NA	2.32E-06	4.12E+03	7.06E-01		1.29E+02

NOTE: The ambient air concentration for mercury is the RfC, and the VFsoil and VFgroundwater values have been modified for the worker exposure parameters, as described in the text.

TABLE D-7
RISK BASED SCREENING LEVEL – AMBIENT AIR
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
TARGET CANCER RISK	TR_c	1E-06	unitless	NYSDEC, 1995
TARGET NON-CANCER RISK	TR_{nc}	1	unitless	NYSDEC, 1995
INHALATION RATE	IR	20	m^3/day	ASTM, 1995
BODY WEIGHT	BW	70	kg	ASTM, 1995
EXPOSURE FREQUENCY	EF	250	days/year	ASTM, 1995
EXPOSURE DURATION	ED	1	years	ASTM, 1995
AVERAGING TIME				
CANCER	AT	70	years	ASTM, 1995
NONCANCER	AT	1	years	ASTM, 1995

NYSDEC, 1995. Site Assessment and Closure Guidance for Petroleum Impacted Sites (Review Draft). Division of Spills Management. September 24, 1995.

ASTM, 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Stnd. E1739-95)

EQUATIONS

$$RBSL_{cancer}\ (mg/m^3) = \frac{TR_c \times BW \times AT \times 365\ days/yr}{IR \times ED \times EF \times CSF}$$

$$RBSL_{non-cancer}\ (mg/m^3) = \frac{TR_{nc} \times BW \times AT \times 365\ days/yr \times RfD}{IR \times ED \times EF}$$

Note:

For noncarcinogenic effects: AT = ED

RBSL = Risk Based Screening Level
CSF = Cancer Slope Factor
RfD = Reference Dose

TABLE D-7, continued
RISK BASED SCREENING LEVEL - AMBIENT AIR
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

CARCINOGENIC EFFECTS

COMPOUND	INHALATION CSF (mg/kg-day)^-1	RBSL (mg/m^3)
ORGANICS		
Benzene	2.90E-02	1.23E-02
Trichloroethene	6.00E-03	5.96E-02
Tetrachloroethene	2.00E-03	1.79E-01
Vinyl chloride	3.00E-01	1.19E-03

ND = No data available

TABLE D-7, continued
RISK BASED SCREENING LEVEL – AMBIENT AIR
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

NONCARCINOGENIC EFFECTS

COMPOUND	INHALATION RfD (mg/kg-day)	RBSL (mg/m^3)
ORGANICS		
1,2-Dichloroethene	ND	
1,1,1-Trichloroethane	2.90E-01	1.48E+00
1,2,4-Trimethylbenzene	ND	
1,3,5-Trimethylbenzene	ND	
4-Methyl-2-Pentanone	2E-02	1.18E-01
Benzene	ND	
n-Butylbenzene	ND	
Ethylbenzene	2.90E-01	1.48E+00
Tetrachloroethene	ND	
Toluene	1.10E-01	5.62E-01
Trichloroethene	ND	
Vinyl chloride	ND	
Xylene (total)	8.60E-02	4.39E-01

ND = No data available

TABLE D-8
RISK BASED SCREENING LEVEL - VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
RISK BASED SCREENING LEVEL - AIR; CARCINOGENIC	$RBSL_{a\,cancer}$	chemical-specific	mg/kg	Calculated [a]
RISK BASED SCREENING LEVEL - AIR; NON-CARCINOGENIC	$RBSL_{a\,non-cancer}$	chemical-specific	mg/kg	Calculated [a]
SUBSURFACE SOIL VOLATILIZATION TO OUTDOOR AIR	VF_{soil}	chemical-specific	$(mg/m^3)/(mg/kg)$	Calculated [b]
GROUNDWATER VOLATILIZATION TO OUTOOR AIR	$VF_{groundwater}$	chemical-specific	$(mg/m^3)/(mg/L)$	Calculated [b]

ASTM, 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM stnd. E1739-95).
[a] Calculated in Table D-7.
[b] Calculated in Table D-5.

EQUATIONS

$$RBSL_{soil}\ (mg/kg) = \frac{RBSL_{air}}{VF_{soil\ -\ indoor}}$$

$$RBSL_{groundwater}\ (mg/kg) = \frac{RBSL_{air}}{VF_{groundwater\ -\ indoor}}$$

Note:
For noncarcinogenic effects: AT = ED
Calculations are repeated for cancer and non-cancer effects
RBSL = Risk Based Screening Level

TABLE D-8, continued
RISK BASED SCREENING LEVEL – VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

CARCINOGENIC EFFECTS

COMPOUND	RBSL AMBIENT AIR (mg/m^3)	VF SOIL (indoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (indoor) (mg/m^3)/(mg/L)	VF SOIL (outdoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (outdoor) (mg/m^3)/(mg/L)	RBSL SOIL (indoor) (mg/kg)	RBSL GROUNDWATER (indoor) (mg/L)	RBSL SOIL (outdoor) (mg/kg)	RBSL GROUNDWATER (outdoor) (mg/L)
ORGANICS									
Trichloroethene	5.96E-02	NA	NA	4.27E-04	5.42E-06			1.39E+02	1.10E+04
Vinyl chloride	1.19E-03	NA	NA	5.25E-03	2.42E-05			2.27E-01	4.92E+01
Tetrachloroethene	1.79E-01	NA	NA	1.81E-04	8.77E-06			9.89E+02	2.04E+04
Benzene	1.23E-02	NA	NA	5.05E-04	3.76E-06			2.44E+01	3.27E+03

ND = No data available

TABLE D-8, continued
RISK BASED SCREENING LEVEL - VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
CONSTRUCTION WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

NONCARCINOGENIC EFFECTS

COMPOUND	RBSL AMBIENT AIR (mg/m^3)	VF SOIL (indoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (indoor) (mg/m^3)/(mg/L)	VF SOIL (outdoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (outdoor) (mg/m^3)/(mg/L)	RBSL SOIL (indoor) (mg/kg)	RBSL GROUNDWATER (indoor) (mg/L)	RBSL SOIL (outdoor) (mg/kg)	RBSL GROUNDWATER (outdoor) (mg/L)
ORGANICS									
1,1,1-Trichloroethane	1.48E+00	NA	NA	5.63E-04	6.77E-06			2.63E+03	2.19E+05
4-Methyl-2-Pentanone	1.17E-01	NA	NA	4.73E-05	8.63E-07			2.47E+03	1.35E+05
1,2-Dichloroethene	ND	NA	NA	6.23E-04	4.60E-06			0.00E+00	0.00E+00
Ethylbenzene	1.48E+00	NA	NA	3.65E-05	3.69E-06			4.06E+04	4.02E+05
Toluene	5.62E-01	NA	NA	1.13E-04	3.72E-06			4.97E+03	1.51E+05
Trichloroethene	ND	NA	NA	4.27E-04	5.42E-06			0.00E+00	0.00E+00
Vinyl chloride	ND	NA	NA	5.25E-03	2.42E-05			0.00E+00	0.00E+00
Xylene	4.39E-01	NA	NA	5.50E-05	3.54E-06			7.99E+03	1.24E+05
Tetrachloroethene	ND	NA	NA	1.81E-04	8.77E-06				
Benzene	ND	NA	NA	5.05E-04	3.76E-06				
1,2,4-Trimethylbenzene	ND	NA	NA	ND	ND				
1,3,5-Trimethylbenzene	ND	NA	NA	ND	ND				
n-Butylbenzene	ND	NA	NA	ND	ND				
Mercury	3.00E-04	NA	NA	3.51E-08	2.32E-06			8.55E+03	1.29E+02

NOTE: The ambient air concentration for mercury is the RfC, and the VFsoil and VFgroundwater values have been modified for the worker exposure parameters, as described in the text.

UWSSRBCA	14-Nov-96

TABLE D-9
RISK BASED SCREENING LEVEL - SOIL DIRECT CONTACT
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
TARGET CANCER RISK	TR_c	1E-06	unitless	NYSDEC, 1995
TARGET NON-CANCER RISK	TR_{nc}	1	unitless	NYSDEC, 1995
INGESTION RATE	IRs	58.6	mg/day	ASTM, 1995 [a]
INHALATION RATE	IRa	20	m^3/day	ASTM, 1995 [a]
ADHERENCE FACTOR	M	0.5	mg/cm^2	ASTM, 1995 [a]
SURFACE AREA EXPOSED	SA	3,160	cm^2/day	ASTM, 1995 [a]
SOIL TO AIR VOLATILIZATION	VFss	chemical-specific	(mg/m^3-air)/(mg/kg-soil)	Calculated [b]
SOIL TO AIR PARTICULATES	VFp	2.30E-07	(mg/m^3-air)/(mg/kg-soil)	Calculated [b]
CONVERSION FACTOR	CF	1.00E-06	kg/mg	ASTM, 1995 [a]
BODY WEIGHT	BW	70	kg	ASTM, 1995 [a]
EXPOSURE FREQUENCY	EF	22	days/year	Assumption [c]
EXPOSURE DURATION	ED	0.083	years	Assumption [c]
AVERAGING TIME				
CANCER	AT	70	years	ASTM, 1995 [a]
NONCANCER	AT	0.083	years	Assumption [c]

NYSDEC, 1995. Site Assessment and Closure Guidance for Petroleum Impacted Sites (Review Draft). Division of Spills Management. September 24, 1995.
ASTM, 1995. Emergency Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Stnd. E1739-95)
[a] The construction values were used for the utility worker.
[b] Calculation documented in Table D-5.
[c] EF = 5 days per week for one month; ED = one month.

EQUATIONS

$$RBSL_{cancer}\ (mg/kg) = \frac{TR_c \times BW \times AT \times 365\ days/yr}{EF \times ED \times [(CSF_o \times CF \times (IR_s \times RAF_o + SA \times M \times RAF_d)) + (CSF_i \times IR_a \times (VF_{ss} + VF_p))]}$$

$$RBSL_{non-cancer}\ (mg/kg) = \frac{TR_{nc} \times BW \times AT \times 365\ days/yr}{EF \times ED \times [(CF \times (IR_s \times RAF_o + SA \times M \times RAF_d))/RfD_o) + (IR_a \times (VF_{ss} + VF_p))/RfD_i)]}$$

Note: For noncarcinogenic effects: AT = ED
RBSL = Risk Based Screening Level
CSF = Cancer Slope Factor
RfD = Reference Dose

TABLE D-9, continued
RISK BASED SCREENING LEVEL - SOIL DIRECT CONTACT
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

UWSSRBCA	14-Nov-96

CARCINOGENIC EFFECTS

COMPOUND	VPss (mg/m^3-air)/ (mg/kg-soil)	ORAL RAF (unitless)	DERMAL RAF (unitless)	ORAL CSF (mg/kg-day)^-1	INHALATION CSF (mg/kg-day)^-1	RBSL TOTAL (mg/kg-soil)
ORGANICS						
Trichloroethene	2.94E-04	1	0.5	1.1E-02	6.0E-03	3.4E+04
Tetrachloroethene	1.94E-04	1	0.5	5.2E-02	2.0E-03	2.0E+04
Cadmium	NA	1	0.14	ND	ND	
Chromium	NA	1	0.09	ND	4.1E+01	5.2E+03
Nickel	NA	1	0.35	ND	8.4E-01	2.5E+05
Lead	NA	1	0.006	ND	ND	

ND = No data available
Oral RAFs and Dermal RAFs for volatiles and semivolatiles from NYSDEC, 1995. Dermal RAFs for inorganics from MADEP Residential Shortform (1992)

TABLE D-9, continued
RISK BASED SCREENING LEVEL - SOIL DIRECT CONTACT
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

UWSSRBCA | 14-Nov-96

NONCARCINOGENIC EFFECTS

COMPOUND	VPss ((mg/m^3-air)/(mg/kg-soil))	ORAL RAF (unitless)	DERMAL RAF (unitless)	ORAL RfD (mg/kg-day)	INHALATION RfD (mg/kg-day)	RBSL TOTAL (mg/kg-soil)
ORGANICS						
Trichloroethene	2.94E-04	1	0.5	6.0E-03	ND	8.2E+03
Tetrachloroethene	1.94E-04	1	0.5	1.0E-02	ND	1.4E+04
Cadmium	NA	1	0.14	1.0E-03	ND	4.2E+03
Chromium	NA	1	0.09	5.0E-03	ND	2.9E+04
Nickel	NA	1	0.35	2.0E-02	ND	3.8E+04
Ethylbenzene	8.73E-05	1	0.5	1.0E-01	2.9E-01	8.0E+04
Toluene	1.52E-04	1	0.5	2.0E-01	1.1E-01	3.6E+04
Xylenes (total)	1.07E-04	1	0.5	2.0E+00	8.6E-02	4.6E+04
Lead	NA	1	0.006	ND	ND	
Zinc	NA	1	0.02	3.0E-01	ND	3.9E+06
1,1,1-Trichloroethane	3.37E-04	1	0.5	3.5E-02	2.9E-01	2.4E+04
4-Methyl-2-Pentanone	9.75E-05	1	0.5	8.0E-02	2.0E-02	1.1E+04
Cyanide	NA	1	0.03	2.0E-02	ND	2.2E+05
Mercury	5.68E-07	0.2	ND	3.0E-04	ND	2.8E+04

ND = No Data
Oral RAFs and Dermal RAFs for volatiles and semivolatiles from NYSDEC, 1995. Dermal RAFs for inorganics from MADEP Residential Shortform (1992)

TABLE D-10
RISK BASED SCREENING LEVEL – AMBIENT AIR
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
TARGET CANCER RISK	TR_c	1E-06	unitless	NYSDEC, 1995
TARGET NON-CANCER RISK	TR_{nc}	1	unitless	NYSDEC, 1995
INHALATION RATE	IR	20	m^3/day	ASTM, 1995 [a]
BODY WEIGHT	BW	70	kg	ASTM, 1995 [a]
EXPOSURE FREQUENCY	EF	22	days/year	ASTM, 1995 [b]
EXPOSURE DURATION	ED	0.083	years	ASTM, 1995 [b]
AVERAGING TIME				
CANCER	AT	70	years	ASTM, 1995 [a]
NONCANCER	AT	0.083	years	ASTM, 1995 [b]

NYSDEC, 1995. Site Assessment and Closure Guidance for Petroleum Impacted Sites (Review Draft). Division of Spills Management. September 24, 1995.

ASTM, 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM Stnd. E1739-95)

[a] The construction worker values were used for the utility worker.

[b] EF = 5 days per week for one month; ED = one month.

EQUATIONS

$$RBSL_{cancer}\ (mg/m^3) = \frac{TR_c \times BW \times AT \times 365\ days/yr}{IR \times ED \times EF \times CSF}$$

$$RBSL_{non-cancer}\ (mg/m^3) = \frac{TR_{nc} \times BW \times AT \times 365\ days/yr \times RfD}{IR \times ED \times EF}$$

Note:

For noncarcinogenic effects: AT = ED

RBSL = Risk Based Screening Level

CSF = Cancer Slope Factor

RfD = Reference Dose

TABLE D-10, continued
RISK BASED SCREENING LEVEL – AMBIENT AIR
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

CARCINOGENIC EFFECTS

COMPOUND	INHALATION CSF (mg/kg-day)^-1	RBSL (mg/m^3)	
ORGANICS			
Benzene	2.90E-02	1.69E+00	
Trichloroethene	6.00E-03	8.16E+00	
Tetrachloroethene	2.00E-03	2.45E+01	
Vinyl chloride	3.00E-01	1.63E-01	

ND = No data available

TABLE D-10, continued
RISK BASED SCREENING LEVEL – AMBIENT AIR
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

NONCARCINOGENIC EFFECTS

COMPOUND	INHALATION RfD (mg/kg-day)	RBSL (mg/m^3)
ORGANICS		
1,2-Dichloroethene	ND	
1,1,1-Trichloroethane	2.90E-01	1.68E+01
1,2,4-Trimethylbenzene	ND	
1,3,5-Trimethylbenzene	ND	
4-Methyl-2-Pentanone	2E-02	1.34E+00
Benzene	ND	
n-Butylbenzene	ND	
Ethylbenzene	2.90E-01	1.68E+01
Tetrachloroethene	ND	
Toluene	1.10E-01	6.39E+00
Trichloroethene	ND	
Vinyl chloride	ND	
Xylene (total)	8.60E-02	4.99E+00

ND = No data available

TABLE D-11
RISK BASED SCREENING LEVEL - VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

EXPOSURE PARAMETERS

PARAMETER	SYMBOL	VALUE	UNITS	SOURCE
RISK BASED SCREENING LEVEL - AIR; CARCINOGENIC	$RBSL_{cancer}$	chemical-specific	mg/kg	Calculated [a]
RISK BASED SCREENING LEVEL - AIR; NON-CARCINOGENIC	$RBSL_{non\ cancer}$	chemical-specific	mg/kg	Calculated [a]
SUBSURFACE SOIL VOLATILIZATION TO OUTDOOR AIR	VF_{soil}	chemical-specific	$(mg/m^3)/(mg/kg)$	Calculated [b]
GROUNDWATER VOLATILIZATION TO OUTOOR AIR	$VF_{groundwater}$	chemical-specific	$(mg/m^3)/(mg/L)$	Calculated [b]

ASTM, 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites (ASTM stnd. E1739-95).
[a] Calculated in Table D-10.
[b] Calculated in Table D-5.

EQUATIONS

$$RBSL_{soil}\ (mg/kg) = \frac{RBSL_{air}}{VF_{soil\ -\ indoor}}$$

$$RBSL_{groundwater}\ (mg/kg) = \frac{RBSL_{air}}{VF_{groundwater\ -\ indoor}}$$

Note:
For noncarcinogenic effects: AT = ED
Calculations are repeated for cancer and non-cancer effects
RBSL = Risk Based Screening Level

TABLE D-11, continued
RISK BASED SCREENING LEVEL – VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT – AMES STREET SITE
ROCHESTER, NY

CARCINOGENIC EFFECTS

COMPOUND	RBSL AMBIENT AIR (mg/m^3)	VF SOIL (indoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (indoor) (mg/m^3)/(mg/L)	VF SOIL (outdoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (outdoor) (mg/m^3)/(mg/L)	RBSL SOIL (indoor) (mg/kg)	RBSL GROUNDWATER (indoor) (mg/L)	RBSL SOIL (outdoor) (mg/kg)	RBSL GROUNDWATER (outdoor) (mg/L)
ORGANICS									
Trichloroethene	8.16E+00	NA	NA	4.27E-04	5.42E-06			1.91E+04	1.51E+06
Vinyl chloride	1.63E-01	NA	NA	5.25E-03	2.42E-05			3.11E+01	6.75E+03
Tetrachloroethene	2.45E+01	NA	NA	1.81E-04	8.77E-06			1.35E+05	2.79E+06
Benzene	1.69E+00	NA	NA	5.05E-04	3.76E-06			3.35E+03	4.49E+05

ND = No data available

TABLE D-11, continued
RISK BASED SCREENING LEVEL - VOLATILIZATION FROM SUBSURFACE SOIL AND GROUNDWATER TO INDOOR AND OUTDOOR AIR
UTILITY WORKER
HUMAN HEALTH RISK ASSESSMENT - AMES STREET SITE
ROCHESTER, NY

NONCARCINOGENIC EFFECTS

COMPOUND	RBSL AMBIENT AIR (mg/m^3)	VF SOIL (indoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (indoor) (mg/m^3)/(mg/L)	VF SOIL (outdoor) (mg/m^3)/(mg/kg)	VF GROUNDWATER (outdoor) (mg/m^3)/(mg/L)	RBSL SOIL (indoor) (mg/kg)	RBSL GROUNDWATER (indoor) (mg/L)	RBSL SOIL (outdoor) (mg/kg)	RBSL GROUNDWATER (outdoor) (mg/L)
ORGANICS									
1,1,1-Trichloroethane	1.68E+01	NA	NA	5.63E-04	6.77E-06			2.98E+04	2.48E+06
4-Methyl-2-Pentanone	1.34E+00	NA	NA	4.73E-05	8.63E-07			2.83E+04	1.55E+06
1,2-Dichloroethene	ND	NA	NA	6.23E-04	4.60E-06			0.00E+00	0.00E+00
Ethylbenzene	1.68E+01	NA	NA	3.65E-05	3.69E-06			4.60E+05	4.56E+06
Toluene	6.39E+00	NA	NA	1.13E-04	3.72E-06			5.65E+04	1.72E+06
Trichloroethene	ND	NA	NA	4.27E-04	5.42E-06			0.00E+00	0.00E+00
Vinyl chloride	ND	NA	NA	5.25E-03	2.42E-05			0.00E+00	0.00E+00
Xylene	4.99E+00	NA	NA	5.50E-05	3.54E-06			9.07E+04	1.41E+06
Tetrachloroethene	ND	NA	NA	1.81E-04	8.77E-06				
Benzene	ND	NA	NA	5.05E-04	3.76E-06				
1,2,4-Trimethylbenzene	ND	NA	NA	ND	ND				
1,3,5-Trimethylbenzene	ND	NA	NA	ND	ND				
n-Butylbenzene	ND	NA	NA	ND	ND				
Mercury	3.00E-04	NA	NA	3.09E-09	2.05E-07			9.71E+04	1.46E+03

NOTE: The ambient air concentration for mercury is the RfC, and the VFsoil and VFgroundwater values have been modified for the worker exposure parameters, as described in the text.