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8.00 HUMAN HEALTH RISK ASSESSMENT

This public human health risk assessment (PHRA) is the component of the OU2 RI report that evaluates potential risks to identified human receptors associated with exposure to constituents which have migrated from the OU1 landfill to the surrounding OU2 Study Area. The risk assessment evaluates the possible effects of contaminant concentrations found in the OU2 Study Area on human health and the environment under baseline conditions; i.e., assuming no remedial action is performed. It was designed to facilitate development of response objectives in the FS by identifying those characteristics of the site which may pose significant risks to public health.

This baseline risk assessment was performed in accordance with the revised Draft Work Plan for Baseline Risk Assessment of Operable Unit 2 (GZA, 1995f) and the current EPA Region I risk assessment guidance. Other key guidance documents used for the human health risk assessment are as follows: (1) EPA Region I Risk Updates and referenced Region III Risk Based Concentrations (RBCs); (2) Dermal Exposure Assessment, Principals and Applications, Interim Report, (EPA, 1992b); (3) Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors," (EPA, 1991); (4) EPA Exposure Factors Handbook, (EPA, 1997); and, (5) Risk Assessment Guidance for Superfund ("RAGS"), Volume I: Human Health Evaluation Manual (Part A), (EPA, 1989e).

As stated above, we completed this risk assessment using EPA methodology. The listed guidance documents were used to develop exposure assumptions which were then used with site-specific exposure concentrations to calculate risk estimates. The EPA believes that this is an appropriate approach for estimating public health and environmental risks. However, the scientific community generally understands that the cumulative result of EPA's conservative assumptions and approach to risk characterization frequently overestimates actual risks.

The methodology used for evaluating the human health risks posed by constituents present in the OU2 Study Area consisted of the following steps:

- Hazard Identification;
- Exposure Assessment;
- Dose-Response Assessment;
- Risk Characterization; and,
- Uncertainty Analysis.

These risk assessment components are summarized below:

- Hazard Identification

  The objective of the hazard identification is to summarize the nature and extent of contamination at the site, to delineate zones of contamination, and to select Constituents of Concern (COCs) for evaluation in the PHRA. COCs were

selected by reviewing and evaluating the distribution, prevalence, and characteristics of detected analytes and selecting the constituents that have inherent toxic effects that are likely to be of concern from a human health perspective. These COCs are considered in the remainder of the study to assess potential site risks. The PHRA is based on soil, groundwater, surface water, and sediment data primarily collected by GZA during sampling rounds 1 and 2 of the OU2 RI. In addition, the PHRA considers the sediment and surface water data collected in 1998 to supplement the RI data.

- Exposure Assessment

The exposure assessment identifies pathways and routes by which human receptors may contact COCs. Potential human receptors are identified based on characteristics of the site and surrounding areas, under both current and anticipated future conditions. In accordance with the Work Plan, concentrations of COCs at points of exposure were estimated based on environmental quality data provided in Section 6.00 and contaminant migration information described in Sections 5.00 and 7.00. Combinations of possible receptor populations and exposure scenarios were then screened based on the potential for a complete exposure pathway. In general, we utilized high-end exposure parameters in conjunction with the estimated exposure point concentrations (EPCs) to estimate average daily doses (and risks) associated with the reasonable maximum exposure (RME) expected to occur. The central tendency exposures were evaluated only for pathways that had a risk approaching or exceeding EPA risk limits using the reasonable maximum exposure (RME) scenario. EPCs for both scenarios (i.e., central tendency and RME) were calculated in accordance with applicable EPA guidance.

- Dose-Response Assessment

In the dose-response assessment, the relationship between the potential dose of each compound and the potential occurrence of adverse health effects is defined and discussed. Toxicity values for evaluation of potential noncarcinogenic (threshold) and carcinogenic (nonthreshold) effects were presented. Reference Doses (RfDs), Reference Concentrations (RfCs), Unit Risks and Cancer Slope Factors (CSFs) were obtained from EPA sources, when available. GZA used additional sources or derived appropriate toxicity values for COCs, which do not currently have established EPA toxicity values.

- Risk Characterization

In the risk characterization step, results of the dose-response assessment and the exposure assessment were combined to estimate the potential for the occurrence of adverse health effects. Both carcinogenic and noncarcinogenic risks were estimated for each selected exposure scenario.

Where applicable, concentrations of COCs were compared to ARARs and other criteria.

- Uncertainty Analysis

  In the uncertainty analysis, the major sources of uncertainty introduced at each step of the risk assessment are discussed.

The PHRA is organized as follows: Section 8.10 consists of the hazard identification, Section 8.20 contains the exposure assessment; Section 8.30 contains the toxicity assessment and Section 8.40 contains the public health risk characterization. An uncertainty analysis is presented at the end of each of these sections to facilitate an understanding of the key variables contributing to the uncertainty present in each risk assessment step. Section 8.50 contains a summary of the risk assessment findings and conclusions.

8.10 HAZARD IDENTIFICATION

This component of the risk evaluation provides a summary and review of site data, describes the nature and extent of contamination and describes the criteria used for selection of COCs for the PHRA. In addition, this section includes comparison of site data to background data. These comparisons helped to delineate contamination at or migrating from the site.

Data summaries are segregated by medium (surface soils, groundwater, sediment and surface water). The data summaries are organized to facilitate the PHRA and are presented on Tables 8.12-1 through 8.12-8. These summaries include a listing of analytes detected, frequency of detection, minimum and maximum concentrations detected, location of maximum detection, and the arithmetic mean concentration. The data focuses on the most recent two rounds of sampling, and the supplemental surface water and sediment sampling.

This section also presents, by medium, the screening process used to select the final COCs. In accordance with EPA methodology, we considered frequency of detection, detected concentrations relative to relevant screening criteria, laboratory qualifiers, and toxicity to select the COCs.

8.11 Review of Site Investigation Data

Section 6.00 provides a detailed description of the nature and extent of contamination observed during the OU2 RI. In addition, relevant findings from previous studies are presented.

8.12 Summary of OU2 RI Data

As previously described, the risk assessment focused on data collected during the OU2 RI. We believe that this data is most indicative of current and potential future site conditions, and have a high degree of confidence in this independently validated data set.

The following paragraphs are brief summaries of the constituents detected in each medium, presenting only the data that was considered in the human health risk characterization. Rather than discuss individual results, information for groups of compounds is presented as well as any general trends in the data, such as an unusually elevated concentration of a particular constituent or the appearance of multiple constituents in a particular location. Constituents that contributed the most to cumulative risks, such as arsenic and manganese, are highlighted. For a more comprehensive list of constituents that contributed to cumulative risks, see Table 8.43-2 and refer to Section 8.40 of this report.

Both site and background data were collected during the RI to evaluate the extent of off-site migration of site-related contaminants from the OU1 landfill to the OU2 Study Area, and specifically to assess continuing migration via groundwater, the Quarry Stream, and Cedar Swamp Brook. Migration of contaminants from the OU1 Hot Spot Area via surface water runoff or windblown fugitive dust is expected to have ceased as of the early-1980s when the chemical wastes in this area were reportedly covered by septage sludge and refuse. The potentially impacted media in the OU2 Study Area are groundwater, surface water, sediments, and surficial soils.

Wastes which were disposed of in the OU1 landfill fall into the general categories of municipal solid wastes, industrial wastes, and septage wastes, and include septic waste liquids, septic sludge, industrial wastewater treatment plant sludge, industrial solvent wastes, corrosive wastes, acid wastes, water soluble oils, cyanide plating wastes and other industrial sludges. Based on historic records of disposal at the landfill and contaminants detected during the OU1 RI, types of contaminants that may have migrated from the landfill include VOCs, SVOCs, pesticides and inorganics including metals, and nitrogen and phosphorus nutrients. Only one sample of the 87 OU1 samples analyzed for PCBs detected aroclors, with detected concentration less than 2 ppb.

In particular, bedrock groundwater is a major migration pathway for VOC contamination because trenches cut to or into bedrock were used to dispose of liquid industrial wastes in the OU1 area Hot Spot (an approximately 1 acre area in the eastern-central section of the OU1 landfill), where significant VOC contamination has been found. However, the levels of VOCs found in groundwater in the OU2 Study Area do not contribute significantly to risks calculated, as discussed in Section 8.40. Metals and SVOCs may have migrated from OU1 via the groundwater and streams and persist in surface water and sediments. Metals and SVOCs may also be transported via fugitive dust and deposited on OU2 surficial soils; however, levels of contaminants in soils, as described below, are not significant sources of risks.

Background sample SED95-46 was eliminated from consideration because of elevated levels of certain constituents likely associated with disposal practices at the M.E. Adams CERCLIS Site (see Figure 2-1).

A summary of the site data in each medium and the associated background set is presented in Tables 8.12-1 through 8.12-8, corresponding to the section numbers.

8.12.1 Surficial Soil

Soil samples were collected from locations downwind of the waste disposal areas. Downwind locations include areas believed to be unexcavated and affected only by eolian deposition from the landfill activities. Eleven (surficial) soil samples (SS95-01 through SS95-11) were collected from the site in December 1995 and analyzed for VOCs, SVOCs, PCBs, pesticides, and metals. Six additional soil samples were collected in October 1996; one in each location of soil samples SS95-02 and SS95-09 and two additional confirmatory samples located within 10 feet of each of these locations (SS95-02A and B, and SS95-09A and B). The samples collected in the second sampling round were analyzed only for metals. Results from samples collected from the same location in both sampling rounds were averaged together (refer to Table 8.12-1).

Five VOCs were detected at low concentrations relative to quantitation limits and low frequencies of detection. Fifteen SVOCs, mostly PAHs, were detected at relatively low concentrations (<0.8 ppm) with the exception of one high concentration (24 ppm) of bis(2-ethylhexyl)phthalate in soil sample SS95-04. Soil sample SS95-04 is located on the southern portion of the site. This maximum concentration of bis(2-ethylhexyl)phthalate detected is more than an order of magnitude greater than the concentrations of bis(2-ethylhexyl)phthalate detected in other soil samples. Concentrations of six pesticides (<0.15 ppm) were detected at low frequencies of detection.

Twenty metals were detected at individual maximum concentrations ranging from 0.52 to 20,600 ppm. Two metals, aluminum and beryllium, were detected at concentrations consistent with background concentrations (see Section 8.14); five others (calcium, iron, magnesium, potassium, and sodium) are considered essential nutrients and were consequently eliminated from consideration in the risk characterization (see Section 8.13). Arsenic was only detected in three of fifteen soil samples at concentrations ranging from 8.1 to 9.78 ppm. The highest concentration of arsenic was detected in soil sample SS95-09 located east of the Phase I area. Manganese was detected in all 15 soil samples analyzed at concentrations ranging from 24.3 to 556 ppm. The highest concentration of manganese was also detected in soil sample SS95-09.

8.12.2 Background Surficial Soil

Two surficial soil samples, SS95-12 and SS95-13 were collected from the western portion of the OU2 Study Area in areas that are considered not affected by the OU1 landfill (i.e., background areas) in December 1995. These samples were analyzed for VOCs, SVOCs, PCBs, pesticides, and metals.

Two VOCs, 1,1,1-trichloroethane and 1,1-dichloroethane, were detected at low concentrations (<0.02 ppm) in soil sample SS95-12. Three SVOCs, bis(2-ethylhexyl)phthalate, fluoranthene, and pyrene, were detected at individual concentrations ranging from 0.055 to 0.098 ppm in soil sample SS95-13. Bis(2-

ethylhexyl)phthalate was also detected in soil sample SS95-12 at a concentration of 0.051 ppm. These are all low concentrations relative to detection limits and common anthropogenic background conditions.

Low concentrations of two pesticides, 4,4'-DDE and 4,4'-DDT, were detected in soil sample SS95-13. Sixteen metals were detected at individual maximum concentrations ranging from 0.26 to 18,600 ppm. Arsenic was only detected in soil sample SS95-12 at a concentration of 9.6 ppm. Manganese was detected in both soil samples at concentrations of 85.7 and 215 ppm in soil samples SS95-13 and SS95-12, respectively.

8.12.3 Site Groundwater

The Operable Unit 1 (OU1) Remedial Investigation identified the Central Landfill, and specifically the Hot Spot, as a significant source of groundwater contamination. This study evaluated the 154-acre Phase I, II and III Landfills (the Site) as well as the surrounding operations and borrow areas resulting in an OU1 Study area of approximately 610 acres. The Upper Simmons Reservoir, and to a much lesser extent, the Almy Reservoir, were identified as the primary discharge points for groundwater transporting this contamination. The OUI risk assessment established that consumption of groundwater from within close proximity of the unlined Phase I landfill posed an unacceptable risk to human health. Based on these findings, the OU1 Feasibility Study and Consent Decree focused on contaminant source control consisting primarily of landfill capping, and groundwater containment and treatment. Attainment of Federal Safe Drinking Water Act Maximum Contaminant Levels (MCLs) at the toe-of-slope of the landfill was established as the performance standard for the source control remedy.

Public drinking water supplies have been made available to residents and businesses throughout the area of the landfill. As such, the objectives of the Operable Unit 2 (OU2) Remedial Investigation and Human Health Risk Assessment with respect to groundwater are to: 1) evaluate the potential extent of the off-site groundwater contaminant plume, as addressed in Section 7.10; 2) identify any current groundwater users within the study area, as discussed in Sections 3.11 and 6.30; and 3) estimate the potential risks to human health posed by use of groundwater from these identified supply wells.

To address this human health objective, risks were only evaluated at identified points of existing groundwater use (i.e., supply wells) which are potentially impacted by contamination emanating from Central Landfill, as determined by the hydrogeologic studies presented in Section 7.1 above. This list consists of only one well designated RW43/275 also known as the former Denney well. When earlier drafts of this risk assessment were prepared, the owner of the Denney property denied RIRRC access to the well for sampling. However, that property was sold in 2000, and the new owners allowed the water to be sampled on December 19, 2000. Consequently, whereas earlier versions of this risk assessment relied on a groundwater model to estimate contaminant concentrations in the Denney well, this final version relies on actual chemical data for a

sample from the Denney well. It should be noted that RIRRC is currently taking actions to connect the Denney residence to municipal water supply as per an agreement with the current property owner. All of the identified former and current water supply wells are shown on Figure 2-1. This figure also indicates which wells were sampled for chemical analysis by GZA during the OU2 RI. Sampled supply wells are also shown on the contaminant contour plans, Figures 6-1 through 6-5. Figures 7-1 and 7-2 show which wells (monitoring and water supply) had contaminant concentrations exceeding MCLs and which wells are potentially impacted by/downgradient of Central Landfill.

Five additional supply wells (RW31/004, RW43/070, RW43/167, RW31/012 and RW43/244) were considered in the human health risk assessment. At the time of our sampling (February 1996 or March 1997) it was our understanding that the wells at 43/070, 43/167, and 43/275 were actively being used for potable water supply purposes; the well at 43/244 was only being used for irrigation; and the wells at 31/004 and 31/012 were not being used because these were summer cottages that had not been occupied since the early to mid 90's. RW43/244 is located on RIRRC property and reportedly is used only for irrigation and public water is used for potable purposes. Samples collected from four of these residential wells (RW31/004, RW43/070, RW43/167, and RW43/244) were used as the exposure point concentrations for exposure to groundwater as drinking water at these locations, although none of these residential wells are hydraulically connected to the landfill. Since these wells are not hydraulically connected to the Central Landfill, this evaluation is presented in Appendix E-1 for informational purposes only. These wells are also shown on the referenced figures. The owners of the well at plot 31/012 (RW31/012) would not grant permission for sampling this location. However, this well which is on an unoccupied lot, is adjacent to RW31/004, and is not hydraulically connected to the Central Landfill.

We sampled five inactive residential wells in the OU2 Study Area (Lots 31/002, 31/014, 31/017, 43/007, and 43/036). Two of these wells (31/002, 43/007) are in the area where groundwater may have been contaminated by the OU1 Landfill when the wells were pumping. We did not evaluate potential risks at these locations because the water is not currently being utilized and it is unlikely that it will be utilized in the future. This is because one of the locations (31/002) is connected to public water and the other location is a vacant lot, which is owned by RIRRC.

Groundwater samples were collected from twenty-one monitoring well locations at or proximate to the site between December 1995 and March 1997; however, as described above, results from 19 of these wells were considered in the PHRA. To be conservative, we utilized chemical testing results from 19 of the 21 monitoring wells (exclusive of MW-J and WE87-4 which are on or in very close proximity to the OU1 landfill) and each of the 10 sampled residential supply wells when selecting contaminants of concern for the groundwater risk evaluation. Groundwater wells that were sampled during Round 1 and Round 2 of the OU2 RI were considered to be representative of current and future conditions. Multiple samples collected at individual wells over time were averaged.

Typically, the nineteen detected VOCs were present at low concentrations (<0.01 ppm) and at low frequencies of detection. The highest concentration of total VOCs was detected in monitoring well MW95-48S.

Eight SVOCs were detected at low concentrations relative to detection limits and low frequencies (i.e., typically only one detect out of 24 analyzed samples). The highest concentration of total SVOCs was detected in monitoring well RW31002. Four pesticides were detected in groundwater at individual maximum concentrations ranging from 0.000009 to 0.00002 ppm. Twenty-two (total) metals were detected at individual maximum concentrations ranging from 0.0001 to 551 ppm. Arsenic was detected in fourteen of nineteen monitoring wells at concentrations ranging from 0.002 to 0.028 ppm; the highest concentration was detected in RW31002. Concentrations of manganese ranged from 0.033 to 12.4 ppm with the highest concentration in RW43036 located.

No VOCs, SVOCs, PCBs, or pesticides were detected in the four active residential wells sampled as part of the original RI. Ten metals were detected in one or more of these wells; four metals, aluminum, barium, beryllium, and copper, were considered to be present at or below background levels (see Section 8.14). Arsenic was detected in three of four residential wells at concentrations ranging from 0.0026 to 0.0084 ppm. Manganese was detected in the same three wells at concentrations ranging from 0.074 to 0.18 ppm. Neither of these metals was detected in residential well RW43070. Zinc was detected in residential wells RW31004 and RW43167 at concentrations of 0.22 and 1.0 ppm, respectively. With respect to well 43/275 (the former Denney well) sampled in December 2000, no SVOCs, PCBs or pesticides were detected. Six metals (beryllium, calcium, magnesium, manganese, potassium and sodium) were detected. Manganese was detected at 0.27 and 0.30 ppm and beryllium was detected at 0.0046 and 0.0052 ppm.

8.12.4 Background Groundwater

Two groundwater samples were collected from each of the four monitoring wells, WE87-8, WE85-16, WE87-17 and WE85-18, between January and September 1996. These samples were analyzed for VOCs, SVOCs, PCBs, pesticides, and metals. These wells are not hydraulically effected by the site. Concentrations of constituents detected in these wells are considered representative of background concentrations for the OU2 Study Area. The concentrations in multiple samples collected from individual wells were averaged and considered as representative of the concentration for that location. These concentrations were subsequently averaged to provide background concentrations for the OU2 Study Area.

Carbon disulfide was detected at concentrations of 0.002 and 0.006 ppm in groundwater wells WE87-17 and WE87-8, respectively. Bis(2-ethylhexyl)phthalate was detected at a concentration of 0.002 ppm in monitoring wells WE87-8 and WE87-16. No other VOCs or SVOCs were detected in background groundwater samples. Nineteen total metals were detected at individual maximum concentrations ranging from 0.0005 to 9.36

ppm. Arsenic was detected in only one of the four background monitoring wells (WE87-17) at a concentration of 0.004 ppm. Concentrations of manganese ranging from 0.020 to 0.069 ppm were detected in all four background groundwater samples with the highest concentration detected in groundwater sample WE87-8.

8.12.5 Site Sediment

Sediment samples were collected from the Almy and Upper Simmons Reservoirs, Cedar Swamp Brook, Quarry Stream, associated wetland areas, and the four landfill Sedimentation Ponds. Forty-five sediment samples were considered in our evaluation, and these samples were analyzed for VOCs, SVOCs, pesticides/PCBs, and metals. The sediment samples used are believed to be representative of current, post-dredging conditions. Data from Rounds 1 and 2, and the supplemental sampling conducted in 1998, were used for most of the site; however, earlier data (e.g., 1993) was relied upon to characterize sediments in the Upper Simmons Reservoir data because the upper inorganic layer was dredged in 1996 and those samples are no longer present. Refer to Section 9 and GZA's Upper Simmons Reservoir Screening Level Risk Assessment (GZA, 1995e) for more details regarding the dredging of the Upper Simmons Reservoir. Multiple (i.e., "duplicate") samples from the same location were averaged together to represent the concentration at that location.

Fifteen individual VOCs were detected in sediment samples at relatively low frequencies of detection and low concentrations (<0.350 ppm). Twenty-eight SVOCs were detected in sediment samples at individual maximum concentrations ranging from 0.039 to 23.5 ppm. The maximum detected concentration for many of the identified SVOCs, including benzo(a)anthracene, benzo(a)pyrene, and benzo(b)fluoranthene, was found in sediment sample SED95-40 located in Sedimentation Pond No. 3. The highest concentration of di-n-octylphthalate, 23.5 ppm, detected in sediment sample SED95-43, located in the Upper Simmons Reservoir, is more than an order of magnitude greater than the other concentrations of di-n-octylphthalate detected in this medium.

Ten pesticides were detected at low prevalence (<25 percent) and at individual maximum concentrations ranging from 0.005 to 0.087 ppm. The highest concentrations of 4,4'-DDD and 4,4'-DDE was located in sediment sample SED95-32 collected in the Almy Reservoir watershed. Three PCB aroclors (1232, 1242, 1254) were detected at maximum concentrations ranging from 0.080 to 0.087 ppm, and at very low frequencies of detection (<6 percent). Twenty-four metals were detected at individual maximum concentrations ranging from 0.6 to 37,100 ppm. Manganese was detected in forty-four of forty-five locations at concentrations ranging from 17 to 13,900 ppm with the three highest concentrations (13,900, 11,100, and 8,670 ppm) detected at sediment sample locations SED95-02 and SED95-01 in the Lower Simmons Reservoir, and SED95-28 in the Almy Reservoir watershed.

8.12.6 Background Sediment

Seven sediment samples were collected from locations that are not impacted by the site (i.e., background areas) between December 1995 and October 1996. These samples were analyzed for VOCs, SVOCs, PCBs, pesticides, and metals.

Four VOCs were detected at individual maximum concentrations ranging from 0.005 to 0.081 ppm. Concentrations of twelve SVOCs with individual maximum concentrations ranging from 0.055 to 2.60 ppm were detected in background sediment samples. The highest background concentrations of many of the SVOCs including benzo(a)anthracene and benzo(a)pyrene was detected in sediment sample SED96-49 collected in a wetland stream channel between the Oak Swamp Reservoir and the Almy Reservoir. Two pesticides, 4,4'-DDD and 4,4'-DDE were detected at concentrations of 0.012 and 0.017, respectively, in sediment sample SED95-39 from Oak Swamp Reservoir. Nineteen (total) metals were detected at individual maximum concentrations ranging from 3.8 to 21,500 ppm. Manganese was detected in all seven background sediment samples at concentrations ranging from 10 to 654 ppm. Arsenic was detected in five sampling locations at concentrations ranging from 0.9 to 8.8 ppm. The highest background manganese and arsenic levels were detected in SED95-38, located in Oak Swamp Reservoir.

8.12.7 Surface Water

Surface water quality data generated during Round 1 and Round 2 of the OU2 RI were considered to be representative of current conditions. Surface water samples were collected from thirty-eight locations in December 1995 and October 1996. Five additional samples were collected in 1998 from the Upper and Lower Simmons Reservoir. The second round of surface water samples had unusually elevated detection limits for metals and water quality parameters due to high levels of inorganics in the equipment blanks which erroneously consisted of "organic free" and not "deionized" water. This error should not impact sample quality and is discussed in greater detail in Section 6. We conservatively instructed the validator to recheck the "UJs" associated with the equipment blank problem and change them to "J*". By doing this, certain compounds that may actually not be in surface water may have been retained as surface water COCs. We believe this approach is reasonable and conservative. Multiple samples collected from the same location were averaged together to represent the concentration at that location. These samples were analyzed for VOCs, SVOCs, PCBs, pesticides, and metals.

Eighteen VOCs were detected infrequently in surface water samples at low concentrations (<0.060 ppm). Likewise, low concentrations (<0.020 ppm) of seven SVOCs were detected at relatively low frequencies of detection (<25 percent). Four pesticides (4,4'-DDT, aldrin, delta-BHC, and Endosulfan I) were detected in three or fewer of forty-three samples tested, at concentrations less than 1 ppb. Twenty-three metals were detected in surface water samples at individual maximum concentrations ranging from 0.0006 to 181 ppm. Arsenic was detected at concentrations ranging from 0.001 to

0.003 ppm, with the highest concentration detected in a sample from SW95-02 located in the Almy Reservoir watershed. Manganese was detected in thirty-seven of the thirty-eight samples analyzed at concentrations ranging from 0.009 to 30 ppm; the highest concentration was detected in surface water sample SW95-19 located in Cedar Swamp Brook.

8.12.8 Background Surface Water

Surface water samples located in areas not affected by the site (i.e., background locations) were collected from eight locations in December 1995 and October 1996. Multiple samples collected from the same location were averaged together to represent the concentration at that location. The samples were analyzed for VOCs, SVOCs, PCBs, pesticides, and metals.

Low concentrations of four VOCs and two SVOCs were detected at low frequencies in the surface water background samples. Two pesticides, aldrin and delta-BHC were detected at 0.000024 and 0.000011 ppm in SW95-30. Twenty metals were detected at individual maximum concentrations ranging from 0.0003 to 14.4 ppm. Arsenic was detected in four of eight samples at concentrations ranging from 0.002 to 0.003 ppm. Manganese was detected in seven of eight samples at concentrations ranging from 0.012 to 0.083 ppm.

8.13 Selection of Contaminants of Concern

Contaminants of Concern (COCs) for the PHRA are a subset of the contaminants detected in the various media. COCs are selected to focus the risk characterization on those contaminants, which pose the greatest potential hazard for public health. In accordance with the Work Plan, a selection procedure was followed to identify the COCs based on various criteria. These criteria, described below, are based on published EPA guidance (EPA, 1989e) and guidance from regional EPA personnel. The criteria consisted of:

- comparing maximum values to established standards or guidelines;
- evaluating frequency of detection;
- determining if the compound is an essential nutrient; and
- determining if the compound is a laboratory induced contaminant.

Several different standards and guidelines were used in this process depending on the medium. Table 8.13-1 lists these screening criteria. For soil and sediment, the comparison value is based on EPA Region III RBCs (updated May 8, 2001) for residential exposure. The actual RBC values for non-carcinogens were reduced by a factor of 10 per EPA guidance; no adjustment was made to the RBC values for carcinogens. For groundwater, three different types of standards were considered: EPA Region III RBCs for tap water, EPA MCLs for groundwater, and Rhode Island Groundwater Quality Standards. If there were more than one applicable standard for a contaminant, the minimum value was

selected for comparison. For surface water two different criteria were evaluated: the EPA Ambient Water Quality Criteria (AWQC) for fish consumption and a modified RBC, adjusted for incidental ingestion of water during swimming (refer to Table 8.13-5 for adjustment calculations). Again, the minimum of these two values was used to screen for COC selection. We conservatively used the RBC values for residential soil ingestion as a sediment screening criteria.

The frequency of detection criteria is based on EPA guidance (EPA, 1989e) and EPA Region I. The accepted criteria is that if the frequency of detection is less than 5 percent it may be dropped from further quantitative evaluation.

The essential nutrient criteria, based on EPA guidance (EPA, 1989e) applied to: calcium, iron, magnesium, potassium, and sodium. Some or all of these compounds are present in soil, groundwater, sediment, and surface water and are not considered COCs.

No compounds were deleted because they were considered laboratory induced contaminants.

Tables 8.13-2 through 8.13-5 present the site maximum value, frequency of detection, method detection limit, and the appropriate guideline or standard for each media: soil, groundwater, sediment and surface water. Compounds shown in bold are considered COC and are listed in the following table.

Media	Contaminants of Concern		
	Volatile Organic Compounds	**Semi-Volatile Organic Compounds & Pesticides**	**Metals**
Soil		Benzo(a)pyrene, Benzo(g,h,i)perlyene, and phenanthrene.	aluminum, arsenic, barium, manganese, and zinc.
Groundwater	1,4-dichlorobenzene, benzene, carbon tetrachloride, chlorobenzene, chloroform, cis-1,2-dichloroethene, trichloroethene, and vinyl chloride.	bis(2-ethylhexyl) phthalate, phenanthrene, aldrin, and dieldrin.	aluminum, arsenic, barium, beryllium, cadmium, copper, iron, lead, manganese, nickel, and thallium.
Sediment		Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perlyene, And phenanthrene.	aluminum, arsenic, beryllium, cadmium, lead, manganese, thallium, and vanadium.
Surface Water (current)	Tetrachloroethene	Aldrin	arsenic, beryllium, lead, manganese, mercury, and thallium.

8.14 Identification of Site Related Contaminants of Concern

The contaminants of concern, as identified in the previous section, were reviewed to determine if on-site concentrations were within the range of background concentrations, in accordance with the Work Plan. Comparison to background is not considered a COC screening criterion; however, this screening techniques indicates whether or not background risks should be characterized separately from site-related risks. Consequently, a comparison of on-site concentrations to background concentrations was done for all four media: soil, sediment, groundwater, and surface water, and for two classes of compounds: SVOCs and metals. VOCs and pesticides are considered to be of anthropogenic origin thus they were not included in the comparison to background levels. The location of the background samples for each medium is described in Section 6.00. These locations are shown on Figure 3-1.

This analysis required the identification of a representative concentration for on-site and background locations and the selection of the appropriate method of comparison for the two groups of data. The following procedure was used for each analyte in every media. The first step was to evaluate the samples in the appropriate comparison data sets. If three or fewer samples were analyzed, the maximum concentration was selected for comparison. If the data set contained greater than three samples but more than 50 percent of these samples were identified as being below the method detection limit, the maximum concentration again was selected for comparison. If the data set contained greater than three samples and the majority were above the method detection limit, then the shape of the distribution, i.e., normal, log-normal, or non-normal was determined.

The second step was to compare the location specific and background levels. If the maximum value was identified as the comparison concentration for either data set (e.g., background), than it was simply compared to the maximum value from the other data set (e.g., OU2 Study Area) to determine if the on-site concentration was greater or less than background. If both on-site and background data sets were considered normally distributed, then the t-test was used for the comparison. If only one of the data sets was normally distributed, then the non-parametric Kruskal-Wallis test was used for comparison. The Kruskal-Wallis test is a non-parametric analysis of variance by ranks. In this application, it is similar to the Mann-Whitney test. Similar to the t-test, the Kruskal-Wallis test involves the calculation of a test statistic and the comparison of this statistic with a critical value at a specified significance level. A full description of the test is included in Appendix E-2, as are the results of these statistical tests. The Monitor System ™ software was used to identify the shape of the distributions and perform the t-test and Kruskal-Wallis test. Table 8.14-1 summarizes the comparison values and the tests performed by analyte for each medium.

Table 8.14-2 presents the results of the site to background comparison. For each medium and analyte the comparison values are presented (either the maximum concentrations or the test statistic values) along with a notation indicating whether the comparison showed that the on-site concentration was consistent with the background

concentration. Across all media, on-site SVOCs were always greater than background concentrations; however, on-site beryllium concentrations were consistent with background in all media. In surface water, beryllium was the only metal consistent with background. For the other media, additional metals consistent with background are as follows: soil- only aluminum; sediment- aluminum, arsenic, cadmium, lead, and vanadium; and groundwater aluminum and barium.

Table 8.14-3 summarizes the contaminants of concern by media with an additional indication of which on-site metal concentrations were consistent with background concentrations.

While lead in groundwater is listed as a COC and was not consistent with background, we did not quantitatively evaluate exposure to lead in groundwater and surface water. In lieu of performing a quantitative evaluation of lead exposure and risk, we qualitatively evaluated the sampling data. The USEPA Action Level for the lead is 0.015 mg/L from the National Primary Drinking Water Regulations. The maximum detected concentration of lead (0.0028 mg/L) in surface water is significantly less than the action level. Additionally, lead was only detected in seven percent of the dissolved, surface water samples obtained for the Site, indicating that lead is not present at significant concentration or frequency in surface water to constitute quantitative evaluation.

Eleven wells located in the OU2 area, that are not on RIRRC property were sampled and analyzed for lead. Lead was found above a detection limit of 0.001 mg/L in four of these groundwater samples. Total lead concentrations exceeded the MCL of 0.015 mg/L in two samples (wells RW 31/002 and MW 97-ML10A). The dissolved lead concentration in the samples from MW 97-ML10A was below the method detection limit. Consequently, we believe that total lead concentrations are associated with sediments in the sample. The dissolved lead in the sample from well RW 31/002 was the same as the total lead concentration (0.055 mg/L). The source of this lead is not known but, based on other water quality testing, we do not attribute it to migration from the Central Landfill.

8.15 Applicable or Relevant and Appropriate Requirements and Standards

In accordance with the Work Plan, Federal and State Applicable or Relevant and Appropriate Requirements and Standards (ARARS) were compiled for indicator compounds. Measured concentrations of COCs in different media were compared to available federal and state standards and criteria, where applicable and appropriate. These standards and criteria were used to evaluate concentrations of COC in specific media. Federal ARARs that were reviewed for standards and criteria included EPA MCLs and Drinking Water Health Advisories established under the Safe Drinking Water Act; EPA AWQC; and the EPA risk-based lead standard. GZA also reviewed Rhode Island Groundwater Quality Standards, the Rhode Island Permissible Exterior Soil/Dust Standard RIDOH, 1992, and Rules and Regulations for the Investigation and Remediation of Hazardous Material Releases (Remediation Regulations). The Method 1 soil criteria presented in the last regulation, pertaining to acceptable levels of contaminants in soil, are not considered applicable to this site because a full cumulative health

risk assessment was performed. The cumulative risk approach used at this site is consistent with the Method 3 Remedial Objectives and approach discussed in Section 8.04 of the Rhode Island Remediation Regulations.

Comparisons of measured concentrations of COCs to the identified standards and criteria for soils and groundwater are described in Section 8.45. Refer to the Ecological Risk Characterization (Section 9.00) for a discussion of ecological ARARs for surface water and sediment.

8.16 Toxicity Summaries for Human Health Contaminants of Concern

Toxicological summaries were developed or compiled for COCs for the public health risk evaluation. The toxicity summaries provide, in nontechnical language, summary information on mechanisms of toxic action, acute and chronic noncarcinogenic effects, and potential carcinogenicity from human and animal studies, as well as data on chemical and physical properties, and transport and fate processes. Toxicity profiles for site COCs are presented in Appendix E-3. These profiles provide general information and do not relate directly to potential effects associated with exposures at the site. The dose-response values used in this risk assessment are discussed in Section 8.30 and are listed in Tables 8.31-1 through 8.32-1.

The constituents that contributed the most to calculated human health risks at the site are manganese and arsenic. Manganese is a ubiquitous element that at low concentrations is essential for normal human physiologic function. Arsenic is present in the environment in low concentrations. Inorganic forms of arsenic are more toxic than organic forms. This information must be taken into account in the quantitative risk characterization and is discussed further in Section 8.30.

8.17 Discussion of Uncertainty

There are general and specific sources of uncertainty in the hazard assessment for this site. The general sources of uncertainty include adequacy of the site characterization and quality and treatment of the analytical data. An example of a factor regarding the adequacy of site characterization is the assumption that atmospheric deposition was the primary mechanism for the transport of soil contaminants off-property. Thus, soil samples were only collected from downwind locations. If this is an incorrect assumption and there were other types of transport which would create a different distribution of contaminants (e.g., run-off) then the site may not be adequately characterized.

An example of the treatment of the analytical data is how the second round of surface water samples were validated. There were unusually elevated detection limits for metals due to high levels of inorganics in the equipment blank which erroneously consisted of "organic free" and not "deionized" water. We conservatively instructed the validator to recheck the "UJs" associated with equipment blank problem and change them to "J*". By doing this, certain compounds that may not actually be in surface water may be presented

as surface water COCs. We also incorporated reported results which were quantified at levels below the Contract Required Quantification Limits (CRQLs) but above the instrument detection limits (IDLs). These validator qualified "Js" increase the level of uncertainty, however, the magnitude of this uncertainty is generally small because "J" qualified concentrations are generally very low. However, if the particular contaminant is highly toxic, and/or the compound was falsely identified as being in a particular suite of samples, uncertainties resulting from the use of "J" values could be significant. It is difficult to quantify the impact of these sources of uncertainty on the hazard identification, however, in general, decisions were made to include rather exclude compounds when there was a question.

Specific sources of uncertainty relate to how the constituents of concern were selected. In this risk assessment, the constituents of concern were selected based on a protocol of comparing maximum concentrations to established, conservative, standards or guidance, e.g., the RBCs developed by US EPA, Region III; evaluating the frequency of detection; and determining if a contaminant is an essential nutrient. The use of site maximum concentrations is a conservative screening approach which captures some compounds as COC that may have much lower mean concentrations and may have low frequencies of detection. The standards and guidelines used for comparison are based on a residential exposure scenario which assumes a higher level of exposure than that which is considered plausible for this site. It is highly unlikely that this selection process eliminates any compounds, which may be a significant source of risk.

The elimination of compounds with low frequencies of detection assumes that these are anomalies. If this assumption is erroneous this could contribute uncertainty to the hazard identification. Given the number of samples collected and analyzed it is considered unlikely that this is a large source of uncertainty.

An additional step in this hazard identification is the comparison of on-site concentrations to background concentrations. A total of seven metals (in four different media) were identified as occurring at concentrations on-site which were consistent with background concentrations. These compounds were not removed from the risk characterization process, however, their inclusion in the initial "subtotal" computation of the hazard index and incremental lifetime cancer risk overestimates the public health risk estimate.

8.20 EXPOSURE ASSESSMENT

The goal of the exposure assessment is to develop estimates of the extent to which people are exposed to contaminants in environmental media. Consequently, the exposure assessment identifies pathways and routes by which contaminants may reach potential human receptors so that the potential for adverse health effects can be estimated. While these exposure scenarios represent hypothetical people and activities, they are intended to reflect the physical condition of the Site and surrounding areas, as well as the activities that may occur in these areas. In this baseline assessment, both current and reasonably foreseeable (hypothetical) future exposures

are considered. In accordance with applicable guidance, only complete or potentially complete exposure pathways were evaluated.

Following U.S. EPA guidance (EPA, December 1989e), the assessment initially focused on the reasonable maximum exposure (RME) expected to occur, although central tendency exposures were evaluated for select scenarios. In addition, according to recent U.S. EPA Region I guidance (EPA, August 1994), the 95 percent Upper Confidence Limit (UCL) on the arithmetic mean was typically used as the concentration term. In cases where data sets were neither normally nor log-normally distributed, or where sample numbers were small, the maximum detected concentration was typically used.

The exposure assessment is the key portion of the risk assessment as it describes the physical setting of the Site (Section 8.21), identifies potential receptors (Section 8.22), and potential exposure pathways (Section 8.23). Concentrations of COCs at points of exposure are estimated based on available sampling data and modeling (Section 8.24) and chemical intake levels for each exposure scenario are estimated (Section 8.25). The final section (Section 8.26) discusses the key uncertainties in the exposure assessment.

8.21 Characterization of Exposure Setting

In this step, the exposure setting was characterized with respect to the general physical characteristics of the Site and the characteristics of the populations in the vicinity of the site. Site characteristics may include climate, vegetation, groundwater hydrology, and the location of nearby surface water bodies. Population characteristics may include location of potential exposure relative to the Site, activity patterns, and the presence of sensitive subpopulations. Characteristics of both the current population and potential future populations were considered. As a result, receptors were selected for evaluation in the risk assessment under both current and future uses, as described below.

8.21.1 Current Land Use

The Town of Johnston is a community of about 30,000, located in the north central portion of the State of Rhode Island. The CLF property is characterized by an irregularly shaped property line which reflects its border with numerous smaller, privately and publicly owned lots. The bordering areas are either undeveloped, residential (primarily single family residences) or commercial/industrial. Businesses operating in the vicinity of the site include municipal/commercial transfer stations and demolition debris recycling and disposal facilities, refuse hauling companies, various vehicle repair shops and hazardous waste/oil TSDF concerns. As reported in the July Residential Well Survey Report (GZA, 1997b), RIDEM had identified 38 locations near the site which had been the subject of some form of RIDEM evaluation, including six CERCLIS sites for which PAs and SIs had been performed.

Most of the CLF is located in the watershed of the Upper Simmons Reservoir, with a small portion in the watershed of the Almy Reservoir. Both of these water bodies are used for recreational purposes only (i.e., they are not drinking water supplies). The CLF is also

adjacent to the watershed of the Scituate Reservoir which is a water supply for the greater Providence area.

The area immediately surrounding the active landfill is composed of undeveloped, residential, and commercial/agricultural/industrial properties. The landfill property is partially fenced and vehicular entry is limited to secured roadways. In addition, two armed security guards patrol the landfill property 24 hours/day. RIRRC controls a 1,000-foot buffer zone around the landfill area and operations area (see Figure 2-1). The following major RIRRC operations are located primarily east-southeast of the landfill: a vehicle maintenance facility, materials recycling facility, and a landfill gas-to-energy plant. The remaining portion of the OU2 Study Area includes residential property acquired by RIRRC, primarily to the north and east along Bishop Hill Road, Central Avenue, and Scituate Avenue as well as several small businesses, located primarily to the south and east of the landfill. RIRRC has acquired most of the residences between the landfill's 1,000- and 2,000-foot buffer zones. RIRRC reported that 30 to 35 of these residences were rented to tenants in 1993 (CDM, 1993). When this report was written, the majority of these homes were being moved from the buffer zone, and it is estimated that the 20-25 homes currently rented will be decreased to 6-8 within the next few years. Homes along Simmons Lake Drive (in the hydrogeologically downgradient direction) have all been demolished. Business operations (not owned by RIRRC) in the OU2 Study Area identified during the OU2 RI included; a screw machine products manufacturer, a commercial welder, a demolition contractor, vehicle repair shop, refuse transfer station, C&P recycling operation, refuse hauling company, and a former hazardous waste/oil TSDF business.

The Upper and Lower Simmons Reservoirs, man-made impoundments, are located southeast of the landfill, and have been included within the boundaries of the OU2 Study Area. Groundwater beneath the landfill may reach the Upper Simmons Reservoir via two routes. Groundwater that flows into Cedar Swamp Brook and Quarry Stream ultimately drains into the Upper Simmons Reservoir. Groundwater also discharges directly to this reservoir. A small portion of the OU2 Study Area northeast of the landfill, including a portion of the Phase I waste cell, is located in the watershed of the Almy Reservoir, which has also been included as part of OU2. The Upper and Lower Simmons Reservoirs and the Almy Reservoir are classified as Class B surface waters by RIDEM, and hence, are limited to recreational use.

The Scituate Reservoir, which serves as Rhode Island's major public water supply (Class A), is located approximately 1.4 miles west of the western edge of the RIRRC property and 2.5 miles west of the OU1 landfill boundary. Studies undertaken during the OU1 RI demonstrated that groundwater beneath the landfill does not migrate to this reservoir.

The majority of the OU2 Study Area groundwater has been classified by RIDEM as GA, except for the area immediately surrounding and below the licensed landfill. The groundwater below the licensed waste cells is classified as GC - suitable for waste disposal activities. The area surrounding the active landfill has been classified as GB for distances of 100 feet in the upgradient direction, and the closest of the following in the downgradient direction: property boundary, surface water body or wetland, or 500 feet from the landfill

boundary. As shown on Figure 2-1, a number of small areas within OU2 have been classified as GA-non-attainment areas; many of these are located in the vicinity of industrial facilities not owned by RIRRC.

Approximately 300 private wells were located within a 1-mile radius of the Central Landfill, and have been tested by the RIDEM, U.S. EPA and/or RIRRC. RIRRC has provided public water to residential and commercial properties in the vicinity of the landfill that were likely to be affected; however, a small number of residences declined to connect to the public supply. GZA performed a well reconnaissance as part of this RI and identified 86 existing or suspected private water supply wells within the OU2 Study Area boundary. Sixty-four of these wells were located on properties owned by RIRRC, and 22 wells were identified on properties not owned by RIRRC. As described in the Residential Well Identification Survey Report (GZA, 1997b) and summarized in Section 3, nine residential wells were sampled by GZA.

8.21.2 Future Land Use

Land uses in the OU2 Study Area are not likely to change significantly. Landfill operations will continue and likely expand and include the reconstruction of Quarry Stream, relocation of Cedar Swamp Brook, and construction of a new 45-acre lined waste cell (designated Phase IV) to the south-southwest of the existing Phase I, II, and III areas. It is anticipated that the properties currently used as residences will remain residential. RIRRC properties within the 1,000-foot buffer zone will either remain undeveloped or be used for landfill-related purposes. Figures 2-2 and 2-3, which are based on the Town of Johnston Assessors records current as of April 1997, show zoning and general land usage. A review of these figures indicates that 15 residentially-zoned properties fall partially or wholly within the OU2 Study Area and are undeveloped and not owned by RIRRC. Further, recent observations suggest that development, including apparent industrial development, is taking place on formerly residential, undeveloped property. The Town of Johnston requires that all new construction be connected to the municipal water supply system where available. Due to the availability of municipal water, and the current groundwater classifications of the aquifer at and in the vicinity of the landfill, it is unlikely that developers will install private potable water supplies.

8.22 Fate and Transport of Constituents of Concern

Descriptions of the occurrence and distribution of contaminants are provided in Sections 6.00 and 8.10 of this report. The exploration locations for various environmental media are depicted on Figures 3-1 and 3-2. This section focuses upon leaching to, and migration of groundwater because of potential concerns regarding the use of the site aquifer and the migration of site groundwater to its ultimate discharge points, the Upper Simmons Reservoir and, to a much lesser extent, the Almy Reservoir.

Migration of contaminants in groundwater from the OU1 landfill is the primary source of COCs in OU2 groundwater, rather than leaching from OU2 soils. In fact, the concentrations and extent of site-related COCs in surficial soils outside the areas of the active landfill[1] are minimal relative to the other environmental media. However, we did model fugitive dust emissions from these surficial soils, assuming mechanical disturbance of these soils while trespassers dirt biked over affected areas. With respect to volatilization of landfill-related COCs, historic and current air monitoring data collected at and proximate to the landfill[2] (upwind and downwind directions) indicates that the landfill is not a significant source of volatile air emissions.

The OU2 Study Area includes a surface water and groundwater divide which separates the watersheds of the Upper Simmons Reservoir and the Almy Reservoir. The vast majority of groundwater in OU2, as well as surface water in Quarry Stream and Cedar Swamp Brook, discharges into the Upper Simmons Reservoir, located approximately 1,100 feet southeast of the OU1 landfill. The remainder (approximately 4 percent) of OU1 landfill groundwater flows northeast to the Almy Reservoir. As groundwater flows toward these surface water bodies, it also feeds small wetlands located between the landfill and the reservoirs. Following discharge of OU1 groundwater into these wetlands and surface water bodies, the more soluble constituents mix and flow with surface water (with VOCs dissipating due to volatilization into ambient air), while the less soluble constituents (i.e., SVOCs, pesticides and metals) preferentially adsorb to sediments. Biota, including fish, may be exposed to constituents which have migrated with groundwater into these wetlands and reservoirs.

We considered the contaminant migration pathways described above in the development of the exposure profiles presented in the next subsection.

8.23 Identification of Potential Human Receptors and Exposure Pathways

Based on the nature and distribution of contaminants at the OU2 Study Area and the area characteristics, activities, and uses described above, we identified human receptors which may potentially be exposed to those contaminants. The following paragraphs summarize the potential receptors and exposure points that were evaluated in the risk assessment. Table 8.23-1 provides a matrix summary of this information.

Recent EPA guidance (EPA, 1994) recommends that: (1) the 95 percent UCL typically should be used as the sole concentration term and, (2) high end and central tendency exposure parameters should be used in conjunction with the 95 percent UCL to estimate the RME and

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[1] These soils are assumed to have been potentially impacted by fugitive dust from prior activities within the OU1 Hot Spot Area. Areas which were believed to be unexcavated since the time of disposal in the Hot Spot Area (i.e., 1976 - 1979) were targeted for sampling and analysis performed as part of the OU2 RI.

[2] Samples of ambient air have been collected from locations at the active face, upwind and downwind of the landfill. In addition, indoor and ambient air samples have been collected near downwind residences. Evaluation of this monitoring data indicates there is no statistically significant difference between upwind locations and downwind (residential) locations.

central tendency exposure, respectively. However, the selection of receptors for quantitative evaluation was based on the RME expected to occur, under both current and assumed future land-use conditions. The RME is defined as the highest exposure that is reasonably expected to occur at a site. The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures. EPA believes that by combining the 95 percent UCL with their default high-end exposure parameters, reasonable maximum risk will be calculated.

EPA's default high-end and central tendency exposure assumptions were not available for many of the receptors and exposure pathways evaluated in this assessment. Based on recommendations received from EPA, GZA estimated high-end exposure parameters for the identified human receptors in cases where no default parameters were available. Since site-specific considerations were used to develop many of these exposure assumptions, estimation of both central tendency and RME scenarios was unwarranted for the majority of exposure pathways. However, central tendency exposures were evaluated for one exposure route (consumption of groundwater as drinking water) since risks in excess of risk management criteria were estimated for the RME at select exposure points.

This portion of the exposure assessment involves the identification of pathways by which the potential receptors may be exposed. Exposure pathways are identified after consideration of the sources, releases, types and locations of contamination at the site, fate and transport mechanisms, and the location and activities of the potentially exposed populations. In order for an exposure pathway to be considered complete, the following elements must generally be identified:

- a source and mechanism of chemical release;
- a retention or transport medium (or media);
- a point of potential human contact (exposure point); and
- an exposure route (e.g., ingestion).

A complete exposure pathway, which links COCs in a site medium to a human receptor, consists of a source from which the chemical is released; mechanisms for transport by which a receptor receives a dose of the chemical; and, a point of human contact (the exposure point), with the opportunity for exposure by ingestion, inhalation or dermal absorption. Human exposure may be direct, i.e., the receptor contacts the COC in the affected medium such as air, water or soil, or may be indirect, involving exposure to chemicals from the site through the food chain, such as consumption of fish.

Complete and incomplete expsosure pathways were evaluated in order to determine potential receptors. An example of an incomplete pathway at this site is the inhalation of indoor air pathway. In order to evaluate this pathway, we first evaluated the groundwater data (the source). We compared the groundwater data to the RIDEM GB Groundwater Objectives as amended in August, 1996. These values are based on the potential for VOCs to volatilize and migrate from groundwater into indoor air and are protective of human health. Therefore, a comparison was made to these standards to determine whether further

quantitative evaluation of risk is necessary for this pathway. Table 8.23-2 derives Method 2 GB Groundwater Objectives for the COCs that do not have promulgated standards and compares the maximum detected groundwater concentration to its GB Objective. With only one exception, (for vinyl chloride) all of the maximum concentrations of VOCs were below promulgated or derived GB Objectives. Of the 24 wells sampled, two wells on the RIRRC property had levels of vinyl chloride that slightly exceed the derived GB Groundwater Objective.

Since two wells exceed the vinyl chloride standard, the potential for migration of groundwater was evaluated. The two wells (MW48 and MW48S- a well cluster) are located on RIRRC property, 175 feet downgradient of the toe of slope (HS) of the OU-1 plume. The surface water discharge point for these wells is located 250 feet downgradient of the well cluster. There are currently no buildings or structures, proximate to or downgradient of this cluster. A requirement of The OU1 Consent Decree is to place a deed restriction limiting further development at the RIRRC property. Hence, this undeveloped area will remain undeveloped in the future. Furthermore, due to the potential subsurface migration of landfill gas, RIRRC installs passive subslab ventilation systems on all buildings it constructs on-site, eliminating VOC migration into occupied on-property structures. As there are not and will not be buildings that may be impacted by the vinyl chloride, there is no exposure point for inhalation of indoor air and therefore no complete exposure pathway for groundwater volatilization into indoor air. Therefore, this exposure route was not evaluated further.

Only complete or potentially complete exposure pathways were quantitatively evaluated in this risk assessment; these exposure profiles are summarized in Table 8.23-1.

A summary of exposure points and exposure routes for those receptors selected for quantitative evaluation is presented below.

8.23.1 Off - Site Residents

Several different exposure pathways and exposure points were evaluated for off-site residents. Based on the likelihood of the same receptor encountering COCs in various media and locations, we divided this potential receptor population into two main subpopulations:

- Residential Well Users and,
- Recreators/Trespassers.

The potential exposures of these two receptor groups are discussed separately below.

Residential Well Users

Based on the presence of (suspected or confirmed) operational residential wells, and those residents' lack of connection to the public water supply (as evidenced by billing account information), nine residential wells were sampled during the OU2 RI. Five of the wells sampled are on properties owned by RIRRC plot/lot numbers (31/014, 31/017, 43/007, 43/036, and 43/244) and four wells (31/004, 43/167, 31/002 and 43/070) are located on privately owned properties. One of the on-property wells, 43/244, is connected to a public water supply, however well water is used for irrigation. Off-property well 31/002 has been disconnected; the well pump has been removed and the house connected to the public water supply. For informational purposes, we calculated risks associated with residential use of groundwater in the OU2 Study Area, due to non-OU1 sources, by considering water quality in samples collected from the residential wells referenced above. A discussion is provided in Appendix E-1.

As previously described, the residential well survey identified only one potentially active, private, drinking water supply well (the Denney residence at lot 43/plot 275) which may be affected by the OU1 landfill. A quantitative risk assessment for ingestion of groundwater from this well was performed. A groundwater sample was collected from the Denney property well on December 19, 2000 and the chemical data from that sample were used as EPCs for the residential receptor. The Denney property will be connected to municipal water in the near future.

An evaluation of exposures and risks associated with household water use was performed in accordance with current EPA Region I guidance. Typically, this evaluation includes assessment of exposures via consumption of tap water, dermal contact with water (e.g. during dishwashing or bathing) and inhalation of volatiles released from the water (e.g., while showering). For the purposes of this evaluation, we were instructed by the EPA to present quantitative risk estimates for only consumption of drinking water since guidance for evaluation of other pathways is pending. Current EPA practice is to double the VOC risks to account for inhalation risks, and quantitatively evaluate dermal contact risks, in accordance with current (draft) guidance. However, since the only site COCs evaluated for this pathway included metals, the inhalation risk is irrelevant. Similarly, the dermal risks associated with exposure to metals in water is negligible, since skin absorption and uptake of metals in aqueous soulution is very low. The potential contribution of the dermal contact with potable water pathway is further discussed in Section 8.43.1.

We evaluated both a "high end" exposure and a "central tendency" exposure for this receptor, to permit a greater understanding of the range of potential risks. The age range considered for the high end exposure included children and adults ages one through thirty years old, based on the 90th percentile for years spent at a single residence (i.e., 30 years). A child resident one through nine years of age was evaluated for the central tendency exposure, since the 50th percentile for years spent at a single residence is nine years (EPA, August 1994)

Based on the location of the residential wells, we believe it is very unlikely that the same residential receptors evaluated for household water use would also trespass onto the OU2 Study Area and recreate in the reservoirs on a frequent basis, as assumed for the other residential subpopulation. Hence, we have assumed that the "recreator/trespasser" is a distinct receptor subpopulation, as described below.

Recreator/Trespasser

As described in Section 8.21, a number of residences are located proximate to (or within) the OU2 Study Area. The majority of residential properties not owned by RIRRC are located near the Almy and to the north and west of the landfill. We assumed that local residents may contact COCs while both trespassing onto RIRRC property and recreating in the Almay or Upper Simmons Reservoirs.

We assumed that individuals who owned cottages around the reservoirs would engage in recreational activities such as swimming, boating and/or fishing, on a regular basis throughout the summer. Accordingly, we evaluated dermal absorption and incidental ingestion exposures to both surface water and sediment in the reservoirs. [3] Similar to the household water user described above, the "recreator" was assumed to be a child/adult 1 through 30 years of age. In accordance with the Work Plan, both current and estimated future exposures to surface water were evaluated, since an analysis of data trends and migration rates conducted during two earlier studies indicate that SVOCs and metals may not have achieved steady-state conditions. We did not assume that the same receptors would contact COCs in both the Almy and Upper Simmons, since an individual with a cottage around one reservoir would not also use/reside at the other reservoir. Due to sediment loading from several sources, the aesthetic quality of the Upper Simmons Reservoir is frequently not good, which discourages recreational use of this water body. For this reason we believe our exposure assumptions for this reservoir are very conservative.

In accordance with the Work Plan, the need for quantitative assessment of risks associated with consumption of fish from the reservoirs (primarily, the Upper Simmons) was based on the results of the fish survey conducted by Environmental Science Services, Inc. (ESS). In July and August 1993, ESS personnel completed a fish sampling program in the Upper Simmons Reservoir and the Lower Simmons Reservoir. A total of 19 fish (11 bass and 8 pumpkin seed) were caught from the Upper Simmons and 9 fish (5 bass and 4 pumpkin seed) were caught from the Lower Simmons. Since fish tissue analysis was only performed on the two species caught (i.e., bass and pumpkin seed), information on other species of fish that may live in either of these reservoirs is unknown. Data on length, weight, location caught and species for the fish caught were recorded.

[3] Dermal absorption of metals in surface water was conservatively evaluated using the permeability coefficient (kp) for water, 0.001 cm/hr, and 100% absorption. This approach is considered conservative and appropriate because metals are the major COC in surface water and to not evaluate them would mean this pathway would not be quantitatively evaluated.

A total of 11 samples were analyzed; six from the Upper Simmons and five from the Lower Simmons. Eight of these samples were composite samples made up of two or more fish of the same species and from the same reservoir. The remaining three samples were of individual fish.

Each fish tissue sample was analyzed for six metals including cadmium, chromium, copper, lead, mercury, and zinc. All six samples analyzed from the Upper Simmons Reservoir contained zinc, with levels ranging from 4.2 to 5.6 ppm. One sample (individual bass) also contained mercury at a concentration of 0.4 ppm. No other metals were detected in Upper Simmons samples. Zinc was detected in all fish tissue samples collected from the Lower Simmons Reservoir at concentrations ranging from 2.1 to 4.5 ppm. No other metals were detected in Lower Simmons samples.

A previous study by ESS, Trace Metals in Panfish from Rhode Island Freshwaters (ESS 1988), indicates that copper and zinc are biologically required by fish for proper functioning of enzymes. However, it does not present the concentration needed for proper function. In the 1988 study, ESS analyzed 82 fish caught by local fishermen in freshwater bodies across the state of Rhode Island. The average concentration of zinc and mercury in these fish are as follows:

Zinc:	31.2 ppm
Mercury:	2.63 ppm.

Both of these average concentrations are much higher than those found in the Upper and Lower Simmons Reservoir. Furthermore, the U.S. Food and Drug Administration considers 1 ppm of mercury in fish tissue to be an acceptable level. Rhode Island does not have a separate standard.

With respect to the COCs selected for human health, neither mercury or zinc were retained as COCs for sediment, and only mercury was retained as a surface water COC.

In summary, the data collected as part of the fish survey indicate that elevated levels of site COCs have not accumulated in fish. Regardless, EPA has requested that a quantitative evaluation of this pathway be performed. Fish tissue concentrations estimated in the Ecological Risk Assessment (Section 9.00 of this report) were used to evaluate this pathway.

Exposures and risks for the distinct "recreators" at either the Almy Reservoir or the Upper Simmons Reservoir were summed with exposures and risks for an older child receptor (7 through 15 years of age) assumed to trespass onto landfill property. This age group was selected since the presence of fencing and 24 hour security guards, as well as packs of "wild" (stray) dogs, would preclude trespassing by younger children. In reality, the likelihood of routine trespassing by children is minimal, because trespassing has rarely been observed by landfill personnel, and in those circumstances, it has primarily

been by older teens dirt biking at the perimeter of the landfill property. When trespassers have been identified, they were immediately escorted off of landfill property. Trespassers were assumed to dermally contact/incidentally ingest surficial soil and inhale soil-derived dust while dirt biking, walking, or recreating on the landfill property. These receptors were also assumed to be exposed to sediments in Cedar Swamp Brook, Quarry, Stream, and the four Sedimentation Ponds via dermal contact and incidental ingestion. Trespassers were also assumed to dermally contact surface water in these exposure points while wading/playing, since the water is too shallow for swimming.

8.23.2 Facility Workers

Based on observations and information provided by RIRRC representatives, we believe that the most probable exposure for facility (Landfill) workers to OU1 contaminants in environmental media would occur during the infrequent trash collection from Cedar Swamp Brook, Quarry Stream, and surrounding wetland areas. Therefore, we performed an evaluation of exposures to sediments (via dermal contact and incidental ingestion) and surface water (via dermal contact only) by facility workers collecting trash from these exposure points. According to RIRRC, the "Swimming Hole" formerly located in Cedar Swamp Brook is no longer used for recreational purposes and there is no frequent maintenance that involves contact with sediment or surface water in these sedimentation ponds. However, to be protective, we assumed that sedimentation ponds would be dredged on an annual basis, as required in RIRRC's operating plan, and that limited exposure to pond sediment and surface water would occur. Specifically, we assumed that facility workers would dermally contact and incidentally ingest sediment, and dermally contact surface water while cleaning and maintaining the large excavators used to dredge the ponds.

Unlike the trespasser (see previous subsection and subsection 8.25.1), the facility worker is unlikely to be exposed to contaminants in on-site soils (see Section 6.10) and associated fugitive dust. Soils within the active portions of the landfill property are largely comprised of newly disposed refuse and clean soils from on-site borrow areas or brought in as landfill cover material. Furthermore, facility workers have no reason and are not expected to go into the wooded areas surrounding the landfill. It is from these areas that the soils data were collected. Consequently, for this potential receptor, exposure to surficial soils in undisturbed upland areas of OU2 (outside the active landfill areas) were not evaluated.

8.24 Estimation of Exposure Point Concentrations

Based on recent EPA guidance (EPA, 1992e), GZA employed an iterative approach for development of EPCs. The stepwise process used to calculate EPCs for surficial soil, sediments, and surface water is presented graphically on Figure 8.24-1. The preliminary steps and equations used are described on the following page:

1. The first summary statistic calculated was the frequency of detection. The numerator included all surface water locations (using temporal averages of Round 1 and 2 data and supplemental sediment and surface water data) or sediment and soil sampling points with a detect; the denominator included all sampling locations analyzed.

2. We eliminated all non-detects, represented as one-half the sample quantitation limit (SQL), which were greater than the maximum detected/estimated concentration (EPA, May 1989). This occurred for approximately 3 percent of individual analyte results for groundwater and surface water, 9 percent of individual analyte results in soil, and 14 percent of analyte results in sediment.

3. We used the maximum concentration detected as the EPC if the number of detects was less than 3.

4. For constituents which did not meet the criteria listed in Step 3, we used software titled "The Monitor System" ™ (Entech Systems, Portland, Maine), to determine how the data were distributed (i.e., normal, lognormal, non-normal). Although we included one-half the detection limit to represent concentrations reported as non-detected, we did not include samples which had been eliminated based on elevated SQLs (step 2).

5. As described in the Supplemental Guidance to RAGS: Calculating the Concentration Term, (EPA, 1992e), the appropriate UCL equation for constituents in each area was selected dependent upon the data distribution of most constituents in that area. Specifically, the guidance states that the t-statistic should be used to calculate the UCL of the arithmetic mean for a normal distribution, using the following formula:

$$UCL = \bar{x} + \left[(t * s) / \sqrt{n})\right]$$

where:

UCL	=	upper confidence limit
$\bar{x}$	=	mean of the untransformed data
s	=	standard deviation of the untransformed data
t	=	student statistic (e.g. from table published in Gilbert 1987)
n	=	number of samples

The UCL of the arithmetic mean for lognormal distributions was calculated using the following formula on transformed data:

$$UCL = e^{(\bar{x} + (0.5 * s^2) - sH/\sqrt{n-1})}$$

where:

UCL	=	upper confidence limit
e	=	constant (base of the natural log, equal to 2.718)
$\bar{x}$	=	mean of the transformed data
s	=	standard deviation of the transformed data
H	=	H-statistic (e.g. from table published in Gilbert 1987)
n	=	number of samples

The majority of EPCs presented on Table 8.24-1 were estimated using this approach, however, for constituent data which was non-normally distributed, we used the higher of the maximum concentration or the UCL. The sample locations used to estimate these EPCs are presented on Table 8.24-2. The EPCs based on the UCL approach described above included soil, sediment and current surface water. To determine the EPCs for drinking water, well 43/275 (the Denney well) was sampled in December 2000. The results of these analyses were used as EPCs for drinking water. The methods used to estimate EPCs for fish ingestion, future surface water and fugitive dust are provided below.

8.24.1 Future Surface Water EPCs for the Almy and Upper Simmons Reservoirs

Potential future surface water conditions due to contaminant migration from the OU1 landfill to the Almy Reservoir were estimated based on data collected from seven wells (designated WE85-5, -6A, -6B, WE87-ML1A and-ML1B, MW91-31A, and MW91-31B). These wells are located upgradient of the reservoir and in close proximity to the northeastern landfill toe of slope which is at least seasonally within the Almy Reservoir watershed (as discussed in Section 5.00).

GZA used these wells to estimate the average annual flux of contaminants entering the Almy watershed from the OU1 landfill. Consistent with the assumption and procedures outlined in Section 9.00 of the OU1-RI report (GZA 1993), we ignored the mitigating effects of advection, dispersion, degradation, and attenuation and estimated contaminant concentrations based solely on dilution. Our EPC calculations are based on the annual mass flux of contaminants entering the Almy watershed (calculated by the average concentrations observed in samples from these seven wells, multiplied by the infiltration into the contributory portion of the landfill) and diluted by areal recharge in the Almy watershed.

The calculations of the initial average concentrations were complicated by the lack of CLP data from monitoring wells MW90-31A and MW90-31B. Quarterly environmental monitoring data indicate that samples from these wells consistently show the highest concentrations of organic contaminates. To be conservative, we sought a method to

include these sampling results although the sampling data had not had the most stringent QA/QC procedures or independent data validation.

Due to the lack of pertinent OU2 CLP data, the CLP data collected from the two rounds of OU1 Remedial Investigation sampling from wells WE85-5, WE85-6A, and -6B, and WE87-Ml1A and -ML1B were averaged. Then, the quarterly environmental sampling data from all seven wells were averaged from August 1990 to May 1991 (the first complete year after the installation of the MW90-31 cluster). The two data sets were compared and the highest average concentration for each detected constituent was retained for further analysis.

The infiltration contributing to flow from the landfill to the Almy watershed was estimated at 11,000 GPD (or approximately 32 million pounds per year). The baseflow to the Almy Reservoir was estimated to be 1 million gallons per day (or 3.3 billion pounds per year). This results in a dilution factor of approximately 100 when considering the Almy reservoir.

Note that VOC migration to the Almy Reservoir seems to be in steady state, therefore, we could estimate future risk due to VOCs in surface water based on currently observed concentrations. However, inorganics, SVOCs, PCBs, and pesticides which may be sourced by the OU1 landfill, may continue to increase in locations downgradient of the landfill until the OU1 cap is complete.

For the Upper Simmons Reservoir EPCs, we assumed the concentrations for groundwater impacted by the OU1 landfill discharging the Uppers Simmons Reservoir (see Figure 5-5) could be estimated by using the average concentration of constituents detected in samples from MW95-47, -47S, -48, -48S, and MW95-53, MW97-ML9A, -ML9B, and ML9C from OU2 sampling Rounds 1 and 2. For more complete coverage, these samples were also averaged with the OU1 Round 2 sampling data from wells MW-B, MW-B1, MW-C, MW-C1, and WE87-ML4A, -ML4B, and -ML4C.

A dilution factor of 17 was used to estimate the future surface water EPCs for the Upper Simmons Reservoir. The calculation of this dilution factor is presented in Section 5.30 of this report, and is described briefly below.

8.24.2 Fish Tissue EPCs

Concentrations of contaminants in fish were estimated to evaluate potential risk to recreational receptors who may ingest fish from the Upper Simmons Reservoir. Concentrations in fish tissue are derived in Section 9.00 within the Ecological Risk Assessment. Concentrations of organic contaminants in fish of the Upper Simmons Reservoir were estimated using the Gobas Model. The Gobas Model allows estimation of organic chemical concentrations in the tissues of aquatic organisms based on chemical specific parameters (e.g., molecular weight, Henry's Law constant, octanol-water partitioning coefficients), and site-specific measurements or assumptions (e.g., trophic relationships among organisms, concentrations in sediment, surface water, and suspended

solids; suspended solid concentrations, TOC concentrations in sediment and suspended solids tissue, water temperature and pH, etc.). This model was used to estimate whole body fish tissue concentrations in the Upper Simmons Reservoir. Whole body concentrations are appropriate for ecological risk assessments because most receptors will consume the entire fish. However, since organic contaminants tend to concentrate in fatty tissue which humans don't typically eat (e.g., the skin or organs) use of whole body estimates is conservative for estimating human exposures.

Concentrations of inorganic contaminants in fish tissue were estimated by multiplying average surface water concentrations by bioconcentration factors presented in EPA documents[4], or, in the case of COCs which were detected in sediment but not surface water, by multiplying average sediment concentrations by sediment to fish BCFs calculated using data for freshwater streams in the Puget Sound watershed[5]. Table 8.24-1 presents the EPCs for fish consumption.

8.24.3 Fugitive Dust EPCs

A combination of two predictive models was used to develop EPCs for fugitive dust exposures to trespassers during dirt biking or other recreational activities. These estimated contaminant-specific concentrations in ambient air served as the EPCs for potential inhalation exposures to fugitive dust. The input soil concentrations were estimated based on either the 95 percent UCL or maximum contaminant concentrations from data obtained during the OU2 RI. GZA assumed the dirt bike riders could generate dust plumes from a large area on CLF property over a one-hour duration of riding. The emission rate for mechanical suspension from dirt bikes was obtained using the published emission factors from the EPA AP-42 for unpaved roads. The model incorporates variables such as weight of vehicle, number of wheels, surficial soil contaminant concentration, soil type, mean wind speed, moisture content of soil, and vegetative cover to calculate an emission rate for fugitive dust. This emission rate was then used in a conservative air dilution model ("Near Field Box") which utilizes site windspeed and area over which the dust is generated to predict ambient air concentrations of fugitive dust.

The AP-42 unpaved road emission factors were derived using vehicles with more than four wheels; however, based on discussions with representatives from Research Triangle Park (RTP), in the absence of a specific emission factor for dirt bike riders, this use is considered acceptable. Details of the fugitive dust modeling are presented in Appendix E-4.

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[4] EPA, 1998. Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities - Volume Two [Peer Review Draft]. Office of Soil Waste and Emergency Response. EPA 530-D-98-001B, and EPA, 1986. Superfund Public Health Evaluation Manual. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response. EPA/600/R-93/187A and b. NTIS No. PB94-174778,

[5] MacCoy, D.E., and R.W. Black, 1998. Organic Compounds and trace Elements in Freshwater Streambed Sediment and Fish from the Puget Sound Basin. U.S.G.S. Fact Sheet 105-98/1/1998. url = http://www.dwatcm.wr.usgs.gov/pugt/fs.105-98.html

8.25 Estimation of Chemical Intakes

In this step, chemical-specific exposures or "doses" are calculated, in units of mg/kg-day, for each exposure pathway identified. As described in the beginning of the exposure assessment, (EPA, 1994) requires that at least one exposure estimate be based on RME which is defined as "the highest exposure that is reasonably expected to occur at a site." Although current guidance indicates that both "central tendency" and "high end" exposures be evaluated, central tendency parameters were typically not available for the scenarios we evaluated. Consequently, we focused on the evaluation of RME for the identified receptors, with the exception of the residential drinking water exposures. As previously stated, both the central tendency and "high end" exposures were evaluated for the residential well users.

The following paragraphs define the equations used to calculate exposure dose and describe the parameters that are incorporated into these equations including exposure factors, constituent-specific variables and conversion factors. The U.S. EPA Exposure Assessment Group defines exposure as the amount of materials in contact with an organism and available for absorption. The material which reaches the organism's (receptor's) absorption barrier (such as skin, lung, or gastrointestinal tract) is referred to as the applied dose (for ingestion and inhalation exposures), while the absorbed (or internal) dose (for dermal exposures) is defined as the amount of material which actually crosses the receptor's exchange boundary.

We calculated exposure doses (and average daily exposures) for each quantitatively evaluated exposure scenario. Exposure doses were calculated as the amount of constituent taken into the body per unit body weight per unit time (mg/kg-day), and were based on conservative exposure assumptions and factors developed in accordance with previously-referenced federal guidelines. We used a combination of mean and upper bound values for various parameters to provide a conservative estimate of the potential risks for the identified receptors. In addition, we selected the most sensitive receptors (in this case, the youngest age group likely to be exposed) to evaluate potential risks for an identified sub-population.

The following sections discuss the receptor-specific assumptions and chemical-specific variables incorporated into the calculation of exposure factors for the receptors evaluated: residential well users, trespassers/recreators and facility workers. For inhalation exposures, average daily exposures (ADEs) were calculated, instead of average daily doses (ADDs), by normalizing fugitive dust EPCs with averaging times only (see below).

The general equations used to estimate Average Daily Dose (ADD), Lifetime Average Daily Dose (LADD), Average Daily Exposure (ADE), and Lifetime Average Daily Exposure (LADE) are:

$$\text{ADD or LADD} = \frac{\text{Total amount of COC contacted/ingested} * \text{Absorption Adjustment Factor}}{\text{Body Weight} * \text{Averaging Period}}$$

$$\text{ADE or LADE} = \frac{\text{Time-weighted exposure concentration for airborne contaminants}}{\text{Averaging Period}}$$

The ADD/LADD equation was used to evaluate the risk of adverse effects to the identified receptors (trespasser/recreator, residential well user and facility worker) resulting from:

- dermal contact and incidental ingestion exposures to soil or sediment;
- dermal contact and incidental ingestion exposures to surface water;
- ingestion of fish; and,
- ingestion of groundwater as drinking water.

The ADE/LADE equation was used to evaluate potential risks posed by inhalation of fugitive dust by trespassers.

8.25.1 Receptor-Specific Parameters

Receptor-specific exposure variables incorporated in the equations used to calculate ADDs, ADEs, LADDs, or LADEs may include variables such as body weight, skin surface area, water, soil or fish ingestion rate, frequency of exposure, duration of the exposure event, duration of the exposure period, absorption adjustment factor (AAF), and averaging period. Refer to Tables 8.25-1 through 8.25-5 for definitions of these terms and the exposure assumptions, exposure equations, and receptor-specific exposure factors for each quantitatively evaluated exposure scenario. These exposure assumptions were based on both conservative EPA default or recommended values as well as site-specific information. Key exposure assumptions which deviate from the default assumptions are discussed below.

Residential Well Users

As presented on Table 8.25-1 and 8.25-2, both high end (consistent with RME) and central tendency exposures were evaluated for this receptor group. For the high end exposures, the 90th percentile for total tap water intake (both sexes) was used as the ingestion rate for children 1 through 10 (1.3 liters/day) and 11 through 19 years of age (1.7 liters/day), as reported by Ershow and Cantor Exposure Factors Handbook, (EPA, 1997). Total tap water is defined as "all water from the household tap consumed directly as a beverage or used to prepare foods and beverages." This demographically representative study, which is cited as a "key study," contains the intake rates recommended by EPA for most age categories. EPA's "high end" default value of 2 liters/day was used as the intake rate for adults 20-30 years of age (EPA, 1994). This ingestion rate corresponds well with the Ershow and Cantor data, and is approximately at the 84th percentile of the relevant distribution. The receptor evaluated for this exposure was 1-30 years old (including 1-10, 11-19, and 20-30-year old subcategories),

consistent with use of the 30 year exposure duration, and the most potentially sensitive subpopulation (assumed to include the youngest potential receptors).

For the central tendency exposure (refer to Table 8.25-2), an exposure duration of 9 years was used, based on the 50th percentile for years spent at a single residence (EPA, 1997). Similar to the high end exposure, we selected the youngest subpopulation for this time period (1-9 years in age) based on the assumption that the intake rate: body weight ratios would be highest, and that they would exhibit the highest relative sensitivity to chemicals. An intake rate of 0.7 liters per day was used for the (1-9 years of age) child resident receptor, based on the 50th percentile reported for that age range in the Ershow and Cantor study. No other parameters varied from the high end to the central tendency exposure evaluations.

Recreator

Table 8.25-3 presents the exposure assumptions used to evaluate the exposures to sediment and surface water for the receptor assumed to recreate in the Almy or Upper Simmons Reservoirs. We assumed that this child/adult receptor (aged 1-31 years, again based on the assumed 30 year residence time) would access and swim in either of these reservoirs 71 times per year, or five times per week during the summer months. This is a conservative assumption, since most of the cottages around the Upper Simmons Reservoir have been demolished or abandoned, and typical weather patterns would likely preclude such frequent swimming. We assumed that these receptors would be exposed to surface water for one hour per swimming event. Although there are no beach areas surrounding the reservoirs, we did assume that these local residents would contact sediment while either accessing the reservoirs for swimming, fishing or playing/walking along the shoreline. Body parts assumed to be exposed to sediment include hands, feet and legs. To determine surface area, we used the surface area for the whole body and the percentage surface area of the individual body parts assumed to be exposed (i.e., hands, feet, and legs) presented in the EPA Exposure Factors Handbook (EPA, 1997). The average whole body surface area was then multiplied by the percent surface area for each body part assumed exposed to calculate the surface area for the exposed body parts. Then, a sediment adherence factor of 1.4 mg/cm^2 (Driver et al., 1989) was employed. The sediment adherence factor used is higher than the 1 mg/cm^2 ("high end") default adherence factor presented in RAGS (EPA, 1989e), and is based on the highest adherence factor (correlated with the smallest soil particle size) reported in the Driver et al. study. The daily sediment ingestion rate is an age-weighted rate based on the mean soil ingestion rates recommended by Region I EPA (EPA, 1994) of 200 mg/day for children 1-6 years old , and 100 mg/day for older children and adults. Furthermore, due to the likelihood of swimming following contact with sediment, it is unlikely that hand to mouth activity will result in significant sediment intake levels. We also assumed that this receptor may fish in the vicinity of the Site one day per week during the summer months, i.e., fourteen times per year. We assumed that they would catch and ingest fish 50% of the instances that they fish, i.e., seven times per year. The fish ingestion rate for the child is a weighted average of the mean quantity of fish consumed per eating occasion for 1-8 year olds, as presented in the Exposure Factors Handbook (EPA, 1997). The overall mean quantity of fish consumed per meal was

used for the child/adult receptor. As indicated on Table 8.25-3, this corresponds to approximately 68 grams/meal for children, and 117 grams/meal for children/adults.

Trespasser

As presented on Table 8.25-4, it was assumed that children age 7 through 15 years may be exposed to chemical constituents in soil, sediment and surface water in the OU2 landfill areas, and fugitive dust assumed to be emitted from unpaved (upland) landfill areas. Children in this age range were considered the most likely and most sensitive subpopulation of possible trespassers based on their higher intake/body weight ratio relative to adults, and the evidence of dirt biking activities (while site access is limited, trespassers of this age can find/create paths through the woods). We conservatively assumed that trespassers would access these Site areas one day per week during the seven non-winter months of April though November (or 31 days per year) for a period of nine years (from age 7 through 15 years). The assumed frequency of exposure far exceeds the actual likely exposure, based on the observations of landfill security. The nine-year exposure duration was based on the 50th percentile for years at an individual residence and the duration of the ages being considered. The duration of exposure to fugitive dust and surface water was assumed to be 1 hour per event, based on the likelihood of security personnel catching and removing trespassers, and/or packs of stray dogs frightening them away. For dermal contact exposures to soil and sediment, the body parts assumed to be exposed include feet, arms, and hands. Consistent with the "recreator", with whom risks for this receptor were summed, a soil/sediment adherence factor of 1.4 mg/cm^2 was used.

Facility Workers

Table 8.25-5 presents the exposure assumptions used to calculate chronic and lifetime average daily doses for the landfill worker receptor. In general, there is minimal potential for this receptor to encounter environmental contaminants in OU2, since the vast majority of their time is spent working on the active phases of the landfill, where wastes are covered daily. Nonetheless, we conservatively assumed that these workers would contact sediments and surface water in the sedimentation ponds, Quarry Stream and Cedar Swamp Brook. Based on discussions with RIRRC, we assumed that workers would contact sediments and surface water in both the stream and the brook six times per year, during occasional trash collection in these surface water bodies. Workers were assumed to contact surface water in the brook and stream two hours per trash collection event, and body parts assumed to be exposed to sediment and surface water at these exposure points conservatively included hands, forearms, lower legs, and feet. In all likelihood, workers would wear rubber boots and protective gloves when engaging in this activity thus minimizing contact.

With respect to exposure to COCs in the sedimentation ponds, an exposure frequency of 10 days per year was applied, based on the estimated amount of time to dredge the four sedimentation ponds on an annual basis. Again, this is a very conservative assumption, since these ponds have not been dredged on an annual basis, and such routine maintenance is not warranted. We assumed that worker forearms and hands could be exposed

to sediments and surface water during cleaning/pressure washing of the excavator equipment used to dredge the ponds, and that this cleaning event would last 2 hours per day.

For evaluation of dermal contact with sediments in the streams, brooks, and the sedimentation ponds, sediment adherence factors based on the Kissel et al. (1996) study (presented in the Exposure Factors Handbook, EPA, 1997c) were used. This is cited as the "key" study and is recommended for use by EPA. In this study, dermal adherence to various body parts was related to soil characteristics and to specific activities. The "groundskeeper" was selected as the activity which best approximated the exposure scenario of concern. The mean soil loadings for Groundskeeper Nos. 4 (used to evaluate adherence to feet only) and 5 were selected, based on the long hourly duration of their work (4.25 to 8 hours), the clothing worn (short sleeve shirts, intermittent use of gloves), and the body parts exposed.

8.25.2 Chemical-Specific Parameters

A number of chemical-specific parameters are required when performing an exposure assessment. These are summarized below.

Dermal Permeability Coefficient - The permeability coefficient (Kp) is a key parameter used in estimating dermal absorption of chemicals in water. Kp (cm/hour) represents the dermal permeability for a chemical from an aqueous vehicle through the skin. Experimentally measured or estimated Kp values were used for compounds in aqueous media using data and equations provided by the EPA in their dermal exposure assessment guidance (EPA 1992b). These Kp values are incorporated into the equation used to calculate the dose for dermal contact with surface water. Since the permeability coefficient is used to calculate the amount of chemical penetrating the skin, the resulting dose is an absorbed dose. Table 8.25-6 presents the permeability coefficients used in this assessment, as well as the source for each chemical-specific value.

Absorption Adjustment Factors - Chemical- and media-specific AAFs were applied to the calculated subchronic ADDs and LADDs to make them compatible with the relevant dose-response criteria to be used in the risk characterization section. AAFs are necessary to account for differences in the absorption of a chemical in a given environmental medium relative to that in the dose-response study. Absorption differences can result from matrix attenuation effects as well as differences in the route of administration (oral versus dermal exposures). As requested by EPA, GZA used only the upper end of the experimentally derived absorption values for dermal contact with soil/sediment AAFs (EPA, personal communication, August 20, 1997). Dermal soil/sediment AAFs have only been derived for three compounds that have been detected at the site; arsenic, cadmium, and benzo(a)pyrene. To be conservative, we applied the dermal soil/sediment AAF for benzo(a)pyrene to all carcinogenic PAHs and also used the two generic defaults, one for SVOCs and one for inorganics provided by EPA.

Additionally, AAFs can be used to convert an applied exposure dose to an absorbed dose, in cases where a dose-response value is based on absorbed dose. Finally, for dermal exposure to water, an absorbed dose is calculated. Thus, the AAF can be used to convert the dose-response value to an absorbed dose.

EPA has not published AAFs; however, a methodology to derive AAFs is contained in RAGS (EPA, 1989). When not available for a given chemical of concern, GZA derived the appropriate AAFs using EPA methodology. GZA derived all of the AAFs for dermal exposure to water. Table 8.25-7 presents the AAFs used in this risk assessment. Specific information used to calculate AAFs for each compound are provided in Appendix E-5.

8.26 Discussion of Uncertainty

The exposure assessment focuses on the identification of exposure profiles and the quantification of exposure for an individual who would maximally contact site related COC. Generally, conservative exposure assumptions regarding exposure point concentrations, frequency, and duration of exposure are used to estimate the maximum risk. In general, these conservative assumptions can overestimate the risk from exposure and contribute to the overall uncertainty of the risk characterization. The areas of the exposure assessment which have the potential to introduce the greatest uncertainty are: (1) estimation of EPCs, including statistics and modeling; (2) characterization of current and future land uses and exposure pathways; and (3) calculation of exposure doses through the use of receptor-specific and chemical-specific parameters.

In this exposure assessment, there are several specific sources of uncertainty which should be noted. These relate to: modeling EPCs, using 90th percentile exposure factors, and summing exposure (and thus risks) from several different sources for a single individual which may be unrealistic. For the EPCs, modeling was conducted to estimate fugitive dust concentrations due to dirt bike activity, to estimate fish tissue concentrations in the Upper Simmons Reservoir, and to predict future surface water concentrations in the Almy and Upper Simmons Reservoirs.

U.S. EPA has developed empirical models to predict fugitive dust emissions from different types of vehicles traveling on dirt roads, however, no model has been developed for dirt bikes. The empirical equation used to predict the emission rate from such vehicles is based on that for a truck; a dirt bike does not meet the size and weight applicability requirements for this model. It is not known whether the empirical model used here under or over estimates the emission rate. Another source of uncertainty in the calculation of exposure to fugitive dust is the use of a simple box model to estimate air concentrations. This type of model overestimates concentrations relative to a dispersion model which would use representative meteorological data.

As previously described, it is likely that use of the fish tissue concentrations estimated in the ecological risk assessment is very conservative for evaluation of human health risks. This is because the majority of contaminants will accumulate in the fatty portions of the fish, such as skin and organs, and these portions are typically not eaten by people.

With respect to surface water EPCs, it is possible that surface water concentrations in the Almy and Upper Simmons Reservoirs may not have reached equilibrium with the OU1 groundwater plume for some constituents. To assess future risks from this media, concentrations were modeled. The model was based on our estimates of hydraulic conditions, estimated transport rates, and observed temporal variations in contaminated air concentrations. This simplified model provides an upper bound estimate of the probable annual flux of contaminants and actual long-term concentrations will be less than calculated based on infiltration dilution alone.

In our calculation of receptor-specific exposure factors for the quantitatively evaluated scenarios, we incorporated assumptions which are at the upper end of the range of values available. Receptor-specific parameters, such as contact rates, selected for the PHRA were obtained from EPA guidance, which is intended to err on the side of the protection of human health. Even frequencies and durations were selected according to EPA guidance when it was available. These assumptions were often from the upper end of the distribution for that exposure parameter (e.g., 90th percentile). For example, for the local resident profile we assumed an exposure period of 30 years which represents the 90th percentile for residing at a single residence. The duration of exposure to surface water and sediment are also high-end estimates; GZA assumed local residents would be exposed five days/week during the summer season (Memorial Day through Labor Day). Combining several high-end parameter estimates in the calculation of the exposure results in an estimate which represents exposure at the very extreme end of the distribution (e.g., >95th percentile).

There are also uncertainties associated with the permeability coefficients used to estimate dermal exposure to chemicals in water. Permeability coefficients were calculated using experimental data, when available; in the absence of experimental data, these values were derived through the use of regression equations relating octanol-water partition coefficients to measured permeability coefficients for several compounds. Experimental conditions may be different from exposure conditions evaluated in the risk assessment, and the equations fail to account for several chemical-specific issues affecting permeability such as molecular size. However, there is generally good agreement between measured and estimated values presented by EPA (EPA, 1992b). In order to minimize the uncertainties inherent in estimating dermal absorption, GZA used measured permeability coefficients when available. Although uncertainty is introduced through the use of default and estimated values, these values tend to be conservative, leading to overestimates of risk. When the AAFs and permeability coefficients are calculated using more reliable chemical-specific data, the uncertainty in the risk estimates is reduced relative to when the default or estimated values are used.

In summary, based on health-protective assumptions made in both estimation of EPCs and exposure profiles (receptor selection, exposure assumptions, etc.), the risks presented in this report are likely to be overestimates of any actual risks associated with exposure to contaminants at the site.

8.30 TOXICITY ASSESSMENT

The toxicity assessment portion of the risk assessment provides information which relates chemical-specific exposure to anticipated health effects. Toxicity information is used to quantitatively characterize the relationship between the dose of a chemical and the incidence of adverse health effects in an exposed population. This information is obtained from published literature describing epidemiologic or toxicologic studies involving a particular chemical. Based on the dose-response relationship, dose-response criteria have been derived by the EPA for evaluation of noncarcinogenic and carcinogenic effects. Toxicological information for site COCs were obtained from EPA's on-line toxicological database (IRIS) (EPA, 1995e) in August 1997. The dose-response criteria presented on IRIS are used in the risk characterization to estimate the probability of, or potential for adverse health effects to occur in humans at estimated exposure levels.

EPA requires that the potential for adverse health effects be evaluated following acute, subchronic, and chronic exposures, when appropriate. In the current risk characterization, there are no constituents present at the site that are acutely toxic; thus an evaluation of acute health effects was not warranted. No subchronic evaluations were performed at this site because the chronic pathways that were evaluated (at the same exposure points and at the same concentrations) were considered more likely to occur at this site. It is more conservative to calculate risk estimates based on chronic exposures because the chronic dose-response values, for a given dose, will result in a higher risk. Typically, chronic values are lower than subchronic values and therefore, for a given dose, will result in higher risk. Furthermore, because chronic exposures occur over a greater length of time, there is less attenuation factored into the cancer risk estimates which are averaged over a lifetime.

8.31 Threshold Toxicity Values

For noncarcinogenic health effects, it is believed that a threshold level exists at or below which no adverse health effects would be expected. This dose or threshold is called a No-Observed-Adverse-Effect Level (NOAEL). The lowest dose at which an adverse effect occurs is identified as a Lowest-Observed-Adverse-Effect Level (LOAEL). EPA generates dose-response values for noncarcinogenic effects, called RfDs, by applying uncertainty factors to a No-Observed-Adverse-Effect Level (NOAEL) or to a Lowest-Observed-Adverse-Effect Level (LOAEL), obtained from studies of dose-response relationships. The purpose of these uncertainty factors is to establish exposure levels that are health protective even for sensitive receptors such as the elderly. Uncertainty factors are used as appropriate, to account for interspecies variability between humans and other mammals used in the dose-response studies; use of a NOAEL derived from a subchronic rather than a chronic study; uncertainty when extrapolating from LOAELs to NOAELs; and variation in the sensitivity of the human

population. A modifying factor is an additional factor used in the estimation of allowable levels that allows for "professional judgment" regarding confidence in the studies.

The chronic RfD, which may incorporate modifying factors and uncertainty factors, is a conservative estimate of an average daily exposure level for humans, below which no adverse noncarcinogenic health effects are expected to occur over long periods of exposure. The units of the RfD are mg/kg-day (mg chemical/kg body weight per day). The RfD presented in IRIS for manganese accounts for total oral intake of manganese and has been adjusted for use in this risk assessment. EPA Region I adjusted the RfD for dietary contribution, resulting in a non-dietary reference dose of 0.07 for soil exposure and an RfD of 0.024 for drinking water exposure (USEPA, 1999). The RfD of 0.14 was used for fish ingestion, as it represents dietary intake.

RfCs are inhalation exposure concentrations to which daily exposure of a human population, including sensitive populations, is likely to be without appreciable effects. EPA considers it to be inappropriate to convert agency-approved RfCs into RfDs. Since RfCs generally measure toxic effects at the point of entry (i.e., the lungs or nasal tissues), they cannot be normalized according to body weight, which would be necessary to convert to an RfD. In addition, uptake of chemicals in the lungs is dependent upon an equilibrium between the blood and air in the lungs. The equilibrium partitioning approach is not dependent on body weight or on inhalation rates. These issues have prompted EPA to derive an alternate risk assessment methodology for inhalation exposures, substituting average daily exposures for ADDs. EPCs, in units of mg/m^3 or ug/m^3, are normalized to the averaging period before comparison with RfCs. Tables 8.31-1 and 8.31-2 present the dose-response values for oral ingestion and inhalation noncarcinogenic effects, respectively.

8.32 Cancer Slope Factors and Unit Risks

In contrast to the dose-response assessment of noncarcinogenic effects, carcinogens are assumed to act without a threshold. Thus, we assumed that there is some level of cancer risk associated with every nonzero dose. The dose-response assessment for chemicals suspected of being human carcinogens includes a weight-of-evidence classification and a cancer slope factor (CSF). The weight-of-evidence classification indicates the likelihood that a compound is a human carcinogen based on the quality of evidence from human and animal studies and other supportive information such as mutagenic effects or structure-activity data. The CSF is an estimate of the cancer-causing potency of a substance.

CSFs are derived by the EPA's Carcinogen Assessment Group (CAG) using the linearized multistage model (for animal data) to extrapolate from high experimental doses to low environmental doses. The dose-response curve indicates the relationship between the dose of a particular chemical and the probability of obtaining cancer over a lifetime. Use of a CSF assumes that the calculated dose received is expressed as a lifetime average.

The EPA utilizes the 95 percent UCL of the slope of the dose-response curve from the multistage model (the CSF), expressed in units of $(mg/kg\text{-}day)^{-1}$, in risk characterizations to estimate the incremental lifetime cancer risk (ILCR). The ILCR is an upper bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a suspected carcinogen. As opposed to those derived from animal studies, CSFs derived from human data are not 95 percent upper bound estimates, nor are the resulting risk estimates.

The Unit Risk is the 95 percent UCL of the mean ILCR estimated to result from lifetime exposure to an agent if it is present at a unit concentration (in the air at a concentration of 1 ug/m^3 or in the drinking water at a concentration of 1 ug/l). These values are used in lieu of a CSF when an estimate of a lifetime average concentration of a contaminant is available. Table 8.32-1 presents the dose-response summary for evaluation of cancer endpoints for both oral ingestion and inhalation exposures.

8.32.1 Relative Potency Factors

For evaluation of carcinogenic PAHs in any medium, the U.S. EPA's Environmental Criteria and Assessment Office (ECAO) recommends that a comparative or relative potency approach be used (EPA, 1993b). As stated in the August 1994 Risk Updates guidance provided by EPA Region I, Region I has adopted this policy, which is described below.

There are 7 PAH compounds the EPA classified as Group B2: Probable Human Carcinogens (EPA, 1993b). Four of these listed below are COCs:

- Benzo(a)anthracene (COC)
- Benzo(a)pyrene (BaP) (COC)
- Benzo(b)fluoranthene (COC)
- Benzo(k)fluoranthene
- Chrysene (COC)
- Dibenz(a,h)anthracene
- Indeno(1,2,3-cd)pyrene

Exposure of laboratory animals to these PAHs has been associated with local and systemic carcinogenic effects following various routes of administration. Many of these studies, however, do not provide a suitable basis for development of dose-response criteria. A suitable oral cancer bioassay has been performed by Neal and Rigdon (1967) for BaP. The EPA (EPA, 1991) utilized this study to derive a CSF of 7.3 $(mg/kg\text{-}day)^{-1}$ for benzo(a)pyrene, the only PAH for which a CSF has been derived.

GZA utilized the relative potency approach recommended by ECAO and adopted by EPA Region I, to evaluate the 4 carcinogenic PAHs detected at the site: benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, and chrysene. This method involves the derivation of a numerical estimate of the relative carcinogenic potency of a

specific PAH by comparison with that of benzo(a)pyrene in a manner similar to that utilized for chlorinated dibenzodioxins and dibenzofurans (EPA, 1989b).

Relative potency factors or RPFs for the Group B2 carcinogenic PAHs are presented in Table 8.32-1. These PAH-specific RPFs were multiplied by the CSF for BaP to obtain adjusted CSFs for each of the carcinogenic PAHs, which were then used as the PAH-specific carcinogenic dose-response values in the risk assessment.

GZA

8.33 Summary of Dose-Response Values

EPA-derived dose-response criteria for both noncancer and cancer endpoints were obtained from EPA's IRIS database (EPA, 1995e), or from the 1997 Update of EPA's Health Effects Assessment Summary Tables (EPA, 1997b) when not available in IRIS. For several chemicals, no RfDs and/or RfCs are presented in the 1997 HEAST and a footnote indicates that the values are available from the Superfund Health Risk Technical Support Center (TSC). The TSC was contacted and GZA was provided with several provisional dose-response values. GZA was instructed not to use the old values presented in the previous HEAST documents but to use the new values when available. Footnotes on Tables 8.31-1, 8.31-2 and 8.32-1 indicate which dose response values were based on EPA provisional values.

Finally, to ensure the currency of the references from which dose-response values were obtained, we sent Margaret McDonough (Risk Assessment Project Manager, EPA Region I) a list of constituents for which dose-response information was currently unavailable. Ms. McDonough indicated that there was no additional dose-response information for those constituents.

The EPA has not derived dose-response values for lead in IRIS. In accordance with recent guidance from EPA, the concentration of lead in soil was compared to the EPA risk-based standard of 400 ppm and the Rhode Island Permissible Exterior Soil/Dust Standard (refer to Section 8.44.3), both of which are ARARs. Further, the concentration of lead in groundwater was compared to federal and state drinking water standards. Based on the low concentrations of lead present in site soils, use of the biokinetic uptake model was not necessary.

The EPA has not derived dose-response criteria for all of the PAH compounds. For evaluation of noncancer endpoints, the RfD for naphthalene was used as a default value for benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, chrysene, and phenanthrene.

For some COCs, dose-response values may not be available for both inhalation and oral exposures. Such chemicals were only quantitatively evaluated in the pathways for which EPA-derived dose-response values were available. GZA did not perform route-to-route extrapolation in order to use oral RfDs with inhalation exposures, or vice versa.

8.34 Dose-Response Uncertainties

The following indicate the major sources of uncertainty associated with toxicity values, as indicated in EPA guidance:

- Extrapolation of dose-response information from effects observed at high doses to predict adverse effects at low levels anticipated for human contact with environmental contaminants;
- Using toxicity information from short-term exposure studies to predict effects on long-term exposures (and vice-versa);
- Using dose-response information from animal studies to predict human effects; and
- Using toxicity information based on homogeneous animal populations or healthy human populations to predict the effects likely to be observed in the general population which includes individuals with varying sensitivities (EPA, 1989e).

The dose-response values used in the calculation of HIs and ILCR estimates are conservative values. The chronic RfDs, RfCs and the CSFs used in this PHRA were generally based on studies provided by EPA (when available). Because RfDs are derived using a number of uncertainty factors and are developed to protect sensitive populations, the actual dose associated with a health effect is likely to be higher than the RfD established by the EPA for most groups in the general population. The CSFs are derived based on the upper 95 percent confidence limit, and assume that no threshold level exists for exposure to carcinogens. Thus, although a great deal of uncertainty is introduced in this step, through EPA's use of conservative methods in deriving dose-response criteria for carcinogens and noncarcinogens, the dose-response criteria err on the health-protective side. Dose-response values for each COC are listed in Tables 8.31-1 through 8.32-1 and, when derived by EPA, discussed in more detail in IRIS (EPA, 1995e).

The dietary RfD of 1.4E-1 mg/kg-day for manganese is conservative because it is unclear if there are toxic effects associated with manganese at concentrations above 10 mg/day. In fact, some individuals may be ingesting larger amounts of manganese with no apparent toxic effects. Toxic effects have only been observed at concentrations 3 to 4 times the 10 mg/day level. The use of the adjusted manganese RfDs for soil and drinking water exposures is also conservative, since the types of exposures these modified RfDs are protective of (e.g. potential effects in neonates, etc.) aren't actually occurring at the site.

Risks associated with the levels of arsenic in site media may be over estimated because the levels of organic versus inorganic arsenic at the site have not been quantified. It is conservative to assume, as we did, that all arsenic at the site is in the inorganic form which is the most toxic form of arsenic. Further, the critical toxic effect that the RfD is

based on is a skin disease which is not life threatening and only increases in incidence with longer periods of exposure. Most receptors are likely to only live in one area for a short period of time and not be exposed to the arsenic for a long enough time to exhibit the critical effect.

8.40 RISK CHARACTERIZATION

In the risk characterization step, results of the exposure and dose-response assessments were combined in order to estimate the potential for adverse human health effects to occur from hypothetical exposure to the chemicals of potential concern at the site. Potential subchronic and chronic noncarcinogenic effects and potential carcinogenic effects were evaluated for each receptor, as appropriate.

The risk characterization for the Central Landfill site focused on reasonably conservative scenarios for current and future exposures under current conditions of chemical distribution and modeled future distributions where appropriate. For each identified human receptor, cumulative cancer and noncancer risks were calculated. To evaluate potential noncancer effects, hazard quotients (HQs) were calculated by comparing estimated exposure doses to compound-specific RfDs. Compound specific HQs are then summed to estimate HIs for a particular pathway. To evaluate potential cancer effects, probabilities that an individual will develop cancer over a lifetime of exposure (or "incremental lifetime cancer risks") were estimated by applying compound-specific CSFs to estimated lifetime exposure doses.
For receptors exposed to mixtures of chemicals, or through multiple exposure pathways at the site, pathway-specific risk estimates were summed to calculate a cumulative receptor noncancer risk (cumulative HI)[6] and a cumulative receptor cancer risk (cumulative ILCR) by receptor. In addition, estimated EPCs for constituents in affected media were compared to ARARs.

The following paragraphs provide general information regarding the methodology used to calculate the HIs and ILCR estimates used to evaluate noncarcinogenic and carcinogenic risks.

8.41 Estimation of Incremental Lifetime Cancer Risk

In this step, the ILCR from exposure to the chemicals of potential concern (Group A, B1, B2, and C) was estimated by multiplying the CSF by the LADD. The ILCR is an estimate of the risk of contracting cancer over and above the background cancer rate, during a lifetime. It is an extreme upper bound estimate, due to the use of conservative exposure assumptions, 95 percent UCL concentrations, and 95 percent upper bound CSFs (in the case of animal data).

[6]The term "noncancer risk limit", used by EPA represents a ratio of the potential dose received by a receptor to an "acceptable" dose for protection of human health. Consequently, although EPA terms the HI a "non-cancer risk estimate" and states that an HI of 1.0 is their "acceptable non-cancer risk limit", they are not referring to "risk" in terms of probability, as with the carcinogenic risk estimates.

The ILCR equation may be expressed as follows:

$$ILCR = LADD\ (mg/kg\text{-}day) \times CSF\ (mg/kg\text{-}day)^{-1}$$

The total ILCR for each pathway was calculated by summing the ILCR for each chemical evaluated under a given pathway. The total ILCR for each receptor was then calculated by summing the pathway-specific ILCRs relevant to a given receptor. However, as described in the risk summary and conclusions (Section 8.50), two constituents, arsenic and benzo(a)pyrene, accounted for the greatest proportion of the cumulative receptor ILCR estimates.

8.42 Estimation of Noncancer Hazard Index

To evaluate noncarcinogenic risks, an HQ was calculated. The HQ, the ratio of the receptor's quantified exposure to the "acceptable" level of exposure, provides a general indication of whether exposures are likely to result in adverse health effects, but does not represent the severity of effects associated with an exposure. To evaluate the noncarcinogenic effects for each COC, the estimated ADD (or ADE, for the inhalation pathway) was divided by the appropriate RfD (or RfC) to yield an HQ:

$$HQ_{oral}\ and\ HQ_{dermal} = ADD\ (mg/kg\text{-}day)/RfD\ (mg/kg\text{-}day)$$
$$HQ_{inhalation} = ADE\ (mg/m^3)/RfC\ (mg/m^3)$$

For multiple chemical exposures, single HQs were summed to yield an HI for an individual exposure pathway. A cumulative HI for each receptor was derived by summing the HIs for each exposure pathway.

As with the evaluation of cancer risk, the total HI for each pathway and for each receptor was calculated. As a screening approach, the total HI was calculated by summing the HI for each chemical. This conservatively assumes that each chemical of concern acts on the same target organ, although it is not actually the case. For the purposes of this assessment, GZA calculated a cumulative HI which assured simple additivity; that is, the HQs for each COC were summed. However, as described in the conclusions (Section 8.50), two constituents, manganese and arsenic, accounted for the majority of the cumulative receptor HIs.

8.43 Comparison of Results to Risk Management Criteria

The EPA uses certain benchmarks for the management of risks associated with the potential carcinogenic and noncarcinogenic effects of chemicals of concern evaluated in human health risk assessments. For potential carcinogenic effects, the EPA has developed a target risk range of 1×10^{-6} to 1×10^{-4}. This corresponds to a lifetime probability of getting cancer, over and above the background rate, of between 1 in a million and 1 in ten thousand.

For potential noncarcinogenic effects, a benchmark HI of 1.0 is used for comparison. If the HI is less than 1.0, then adverse health effects are not expected to occur, even in sensitive subpopulations. If the HI exceeds 1.0, adverse health effects are possible. A HI greater than 1.0 does not, however, indicate that an adverse health effect will definitely occur. If the total HI for a receptor exceeded 1.0, then a more refined analysis should be performed wherein target-organ specific HIs are calculated.

8.43.1 Summary of Hazard Indices and Incremental Lifetime Cancer Risks

A summary of total noncarcinogenic HI estimates and total ILCR estimates for each of the identified receptors are presented in Table 8.43-1. The table presents the results of the comparison between both calculated HIs and the EPA "risk" limit of one (1), and between calculated ILCRs and EPA's acceptable risk range of 1×10^{-6} to 1×10^{-4}.

Risks for both noncancer and cancer effects are summed for three categories; total, site related, and background risks. The total risks are risks associated with all constituents that were considered COCs for that media. Site related risks include risks from only constituents that are considered to be site related (i.e., that were detected at concentrations greater than background). In accordance with the Work Plan, concentrations of constituents relative to background could not be considered a COC screening criterion, but were used to segregate "background" risks from site-related risks. Background risks are the cumulative risks from constituents that are considered to be present at concentrations consistent with background. Although constituents that were detected at or below background concentrations are constituents that are ubiquitous to the site and surrounding areas and not present due to activities at the landfill, per EPA guidance, these values are presented to show the potential risks associated with all constituents that are present in the OU2 Study Area. Thus, the total risks are the risks that the receptors may potentially have but the site-related risks reflect the portion of risks potentially associated with chemicals related to site contamination. The following sections summarize the cumulative risks associated with each of the identified receptors included in the quantitative risk assessment.

Cumulative Receptor Risk Estimates

As presented on Tables 8.43-1 and 8.43-2, the total site-related and background risks for each receptor were compared to EPA risk management criteria, and an analysis of the relative contribution of each pathway and risk "drivers" is presented.

Residential Well Users

For the drinking water receptors, two risk scenarios are presented; one considering high end exposure and the other considering the central tendency.

The estimated risks associated with ingestion of groundwater from a potable supply well located on Lot 43/275 (based on measured EPCs) using high-end exposure estimates were summed to yield a total HI of 8.5×10^{-1}, a site-related HI of 7.0×10^{-1}, and a background related HI of 1.5×10^{-1}. The current evaluation for groundwater ingestion at the Denney Residence indicates that the non-cancer and cancer risks associated with site-related constituents are below/within the EPA risk limits. The high-end exposure, site-related HI for groundwater ingestion at the Denney residence is 0.7, which driven essentially entirely by manganese detected in drinking water. Dermal risks associated with groundwater contact at the Denney residences would also be driven by manganese. It is our opinion that under typical circumstances, dermal exposures to metallic contaminants present negligible risks, and it is our understanding that EPA generally agrees. Therefore, dermal exposures to tap water at the Denney residences do not warrant a quantitative evaluation.

The HI, which would result from a very conservative exposure assumption, can be quickly estimated. The reference dose for ingestion of manganese (0.024 mg/kg bw/day) is approximately 34 percent of the reference dose to be used for dermal contact with manganese (0.07 mg/kg bw/day). Expressed another way, ingested manganese is about 3 times more toxic than an equivalent does of manganese absorbed through the skin. This is not even taking into account the low permeability coefficient for manganese in water traveling through the skin. If one were to make the very conservative assumption that residential use of water at the Denney residence results in a dermal exposure to manganese which is equivalent to the ingestion pathway, the resultant HI can be estimated by increasing the ingestion HI by one-third (i.e., the HI of 0.7 would become 0.9). This is still below the risk limit of 1.0. Given this conservative estimate and the likelihood that the Denney residence will soon be hooked up to municipal water by RIRRC, a quantitative evaluation of dermal exposure is not warranted.

The groundwater ingestion risks for the central tendency exposure also did not exceed EPA's non-cancer limit of 1.0. The high-end ILCR estimate is 3.6×10^{-4} which exceeds the upper limit of the EPA target range (i.e., 1×10^{-4}), however this excess cancer risk is due solely to beryllium, which is not a site-related COC. Thus, this excess cancer risk is a background condition for this area. The central tendency ILCR estimate is 9.0×10^{-5}, which is within the EPA's target range; again, this excess cancer risk is due solely to beryllium, and is thus a background condition.

Recreator/Trespasser

The other local resident receptor groups (the recreator and trespasser) was evaluated under four separate scenarios. It was assumed that a recreator living near the Almy Reservoir or the Upper Simmons Reservoir may trespass into the OU2 Study Area but would not access or use the reservoir that they did not reside near. Therefore, we evaluated two separate "recreator" receptors; one that frequents the Almy Reservoir and one that goes to the Upper Simmons Reservoir. Both of these receptors were also assumed to trespass onto RIRRC's property. For each of these receptors, the risks were calculated under current conditions and under future conditions.

For local resident (recreator/trespasser) exposure to the Almy Reservoir and the site under current conditions using high-end exposure estimates, the cumulative risks yield a total HI of 0.49, a site-related HI of 0.31 and a background related HI of 0.18 and a total ILCR estimate of 5×10^{-5}, a site-related ILCR estimate of 4×10^{-6} and a background related ILCR of 4×10^{-5}. Under future conditions using high-end exposure estimates, the cumulative risks yield a total HI of 0.48, a site-related HI of 0.30 and a background related HI of 0.18 and a total ILCR estimate of 5×10^{-5}, a site-related ILCR estimate of 3×10^{-6} and a background related ILCR of 4×10^{-5} for this receptor. The noncancer and cancer risks for current and future exposure to the Almy Reservoir are approximately the same. None of the risks exceed EPA limits. The majority of noncancer risk is associated with the concentrations of manganese detected in sediment from the Almy Reservoir and in surface water from Cedar Swamp Brook and Quarry Stream. For cancer risk estimates, the majority of risks are associated with carcinogenic PAHs in sediment from the Almy Reservoir and Sedimentation Ponds 2 and 3.

For the receptor assumed to reside near the Upper Simmons Reservoir (under current conditions) using high-end exposure estimates, the cumulative site-related risks are 0.22 and 2×10^{-5} for the HI and ILCR estimates, respectively. Similar to the Almy recreator/trespasser, the background related ILCR estimate exceeds the ILCR for site-related chemicals. Under future conditions using high-end exposure estimates, the cumulative (site-related) risks are 0.05 (HI) and 1×10^{-5} (ILCR). The future site-related noncancer risks estimated for exposure to the Upper Simmons Reservoir are approximately one-fourth the risks of current exposure. Cancer risks are the same for both current and future conditions. None of the estimated risks exceed EPA risk management criteria. The majority of noncancer risk is associated with the concentrations of manganese detected in surface water from the Upper Simmons Reservoir and Cedar Swamp Brook and Quarry Stream. For cancer risk estimates, the most significant contribution is from carcinogenic PAHs in sediment from the Upper Simmons Reservoir.

Facility Worker

For facility worker exposure to chemicals in sediment and surface water in Cedar Swamp Brook, Quarry Stream and the four Sedimentation Ponds using high-end exposure estimates, the cumulative site-related risks are 0.001 and 6×10^{-8} for the HI and ILCR, respectively. None of the risks estimated for this receptor exceed EPA limits. The majority of noncancer risk is associated with the concentrations of manganese detected in surface water from Cedar Swamp Brook and Quarry Stream. For cancer risk estimates, the majority of risks are associated with carcinogenic PAHs in sediment from Sedimentation Ponds 2 and 3. As with the recreator/trespasser, cancer risks associated with constituents present at background levels exceed (by more than a factor of 3) site-related cancer risk estimates. The background cancer risks for both the facility worker and recreator/trespasser receptors was largely associated with arsenic and beryllium in sediment.

Pathway-Specific Risk Estimates

Tables 8.44-1 and 8.44-2 present the exposure risks estimated for the one residential well user under high-end and central tendency exposure estimates. Tables 8.44-3 through 8.44-11 present the pathway-specific risks for the trespasser receptor. Lastly, Tables 8.44-12 through 8.44-23 present the risks for the receptors assumed to recreate in the Almy and Upper Simmons Reservoirs and Tables 8.44-24 through 8.44-29 present the pathway-specific risks for the facility worker.

8.44 Comparison of Ambient Concentrations to ARARs

8.44.1 Comparison of Tap Concentrations to Drinking Water Standards

One sample of groundwater from the Denney well was analyzed for COCs. The only groundwater COCs detected were beryllium and manganese. Beryllium was detected at a concentration of 0.0052 mg/l, which slightly exceeds its MCL of 0.004 mg/l. However, as previously described, beryllium is not a site-related contaminant. Manganese was the only other COC detected and did not exceed its MCL.

8.44.2 Comparison of Measured Surface Water Concentrations to Ambient Water Quality Criteria for Protection of Human Health (Fish Consumption)

To evaluate current and future impacts to humans consuming (edible) fish from the Almy Reservoir, Upper Simmons Reservoir, Sedimentation Ponds, Quarry Stream, or Cedar Swamp Brook, current measured surface water COCs were compared to AWQC for the protection of human health with respect to fish consumption; for the metallic COCs, we compared maximum detected dissolved metals concentration to the AWQC (refer to Table 8.45-1).

Arsenic exceeds its AWQC for fish consumption in the Almy Reservoir, the Sedimentation Ponds, and the Quarry Stream and Cedar Swamp Brook. In addition, aldrin exceeds its AWQC in the Sedimentation Ponds.

Among these waterbodies the Upper Simmons Reservoir and the Almy Reservoir are the only waterbodies where fishing is know to occur. As discussed in detail in Section 9.11, the Upper Simmons Reservoir is clearly the primary habitat receptor of landfill contaminants because of its relatively high habitat value, and because the landfill comprises a significant proportion of its watershed. None of the identified Upper Simmons Reservoir surface water COCs exceed their AWQC for fish consumption. These comparisons indicate that the landfill has not adversely affected the edibility of fish in the Upper Simons Reservoir, and corroborate the fish tissue data collected from the Upper Simmons by ESS (see Section 8.23.1). Furthermore, a conservative, quantitative evaluation of consumption of fish from the Upper

Simmons Reservoir was performed. As indicated in the previous subsection of this report, no excess risks were identified for this pathway.

As also discussed in Section 9.11, there are no surface water inputs from the landfill to the Almy Reservoir, and only a small percentage of the landfill's groundwater discharges to the Almy Reservoir. Furthermore, the groundwater component which flows toward the Almy Reservoir is relatively clean. Therefore, the landfill has relatively little potential to have significantly impacted the Almy Reservoir. Although the arsenic may present a potential risk for humans consuming fish from the Almy, it is unlikely that the landfill contributed significantly to the elevated arsenic values.

The water bodies located within the active portion of the landfill (Quarry Stream, the channelized portion of Cedar Swamp Brook, and the sedimentation ponds) do not support a recreational fishery under current conditions. Therefore, there is not a complete exposure pathway for human exposure to COCs via fish consumption, and the risks suggested by the exceedances of the AWQC are not a significant concern under current conditions. The stream channels may support recreational fisheries in the future, however, the likelihood of any particular fisherman regularly eating fish from these streams is remote. In addition, such improvements are expected to result from ongoing efforts to hydrologically separate the landfill storm water and groundwater from the stream channels, and to improve habitat characteristics of the channelized portion of the brook. These efforts will improve water quality within the brook. Based on these considerations, human consumption of fish from the waterbodies within he active portion of the landfill does not present a significant risk of harm.

It should be noted that an AWQC is not available for lead. Based on the low concentrations of lead detected in soil (maximum concentration of 145 ppm), groundwater (maximum total lead concentration of 0.055 ppm), and surface water, (maximum lead concentration of 0.009 ppm), adverse health effects are not anticipated. A discussion of the AWQC relevant to the ecological receptors is contained in Section 9.00 of this report.

AWQCs were also not available for beryllium and manganese. However, beryllium is not a site-related contaminant, and dissolved beryllium was not detected in the Upper Simmons Reservoir, which presents the greatest potential for human exposure via fish consumption. Although the landfill likely significantly increases the concentrations of manganese in surface water, this is likely due to geochemical effects of the landfill (i.e., release of naturally occurring manganese from soils and bedrock due to altered redox in groundwater) rather than past disposal practices. Therefore, the landfill is not expected to present a significant risk to humans consuming fish due to beryllium or manganese.

8.44.3 Comparison of Soil Concentrations to Lead Standards

In accordance with guidance received from EPA Region I, the 95 percent UCL concentration on the mean of lead in surface soil was compared to the EPA risk-based standard of 400 ppm and the more stringent Rhode Island Permissible Exterior Soil/Dust Standard of 150 ppm. The 95 percent UCL concentration of the mean of lead,

85 ppm, was well below both standards (refer to Table 8.45-2). Even the maximum detected lead concentration (145 ppm) did not exceed these residential standards. Consequently, adverse impacts to human health associated with exposures to lead in soil are not anticipated.

8.45 Discussion of Uncertainty

In the Risk Characterization Section, the results of the Dose-Response Assessment and the Exposure Assessment were combined, as was the uncertainty introduced in each of these steps. Again, through the use of conservative assumptions and both average and maximum concentrations, in conjunction with 95 percent upper bound CSFs, calculated cancer risk estimates are much greater than 95 percent UCL values. Thus, actual health risks are expected to be much lower than those calculated in the PHRA.

Important sources of uncertainty in the risk characterization include:

- The equal weight given to Groups A, B1, B2 and C carcinogens in estimation of ILCR;
- The equal weight given to contaminants whose RfDs have different confidence levels in estimating noncarcinogenic HIs; and
- The assumption of simple additivity of ILCRs and HIs across site contaminants.

8.50 SUMMARY AND CONCLUSIONS

The human health risk assessment was completed to characterize potential health risks under baseline conditions (i.e., assuming no remediation is performed) and to help evaluate whether or not remedial response actions are warranted. Total and receptor-specific HIs and ILCRs were compared to EPA benchmarks and acceptable risk limits. In addition, concentrations of COCs in groundwater and surface water were compared to ARARs, as was the concentration of lead in soil.

A number of on-site and off-site receptors and exposure scenarios were evaluated in this PHRA. On-site receptors included facility workers; off-site receptors considered in our evaluation included children and adult local residents assumed to use groundwater as drinking water and (separate) local residents assumed to engage in recreational activities in the Almy or Upper Simmons Reservoirs, ("recreators") and assumed to trespass onto the landfill property located within the OU2 Study Area ("trespassers").

Pathway-specific risks for each receptor group were summed so that the cumulative receptor risks could be compared to EPA criteria. Three separate pathway and cumulative risks are presented for each receptor as presented below:

- Total Risks: these risks reflect calculated HIs and ILCR estimates associated with exposure to all detected COCs within each medium;

- Site-related Risks: these risks reflect calculated HIs and ILCR estimates associated with exposure to COCs which were assumed to be site-related based on the collected information and analytical data, and their concentrations relative to measured background concentrations: and,

- Background Risks: these risks reflect calculated HIs and ILCR estimates associated with exposure to COCs which are present at levels consistent with measured background concentrations.

For the purposes of this risk assessment, the site-related risks were the focus of comparisons to EPA criteria. In accordance with EPA guidance, the risks estimated for VOCs associated with ingestion of drinking water were doubled to account for household exposure to water via dermal contact and inhalation. The site-related, non-cancer and cancer receptor risks calculated for residential well users at the one active residential well (the Denney well at Lot 43/275) potentially impacted by contaminants derived from the OU1 Landfill were below applicable EPA risk limits, under both central tendency and high-end condition. The carcinogenic target risk range was exceeded at the Denney well, however, this was due entirely to beryllium, which is not a site-related contaminant, as discussed in Section 8.14. Also, it should be noted that the RIRRC is currently undertaking activities to provide municipal water to the Denney residence. None of the calculated hazard indices or incremental lifetime cancer risk estimates for other exposure pathways exceeded EPA's non-cancer limit of 1.0 or carcinogenic target risk range.

None of the concentrations of surface water COCs measured in the Upper Simmons Reservoir exceeded EPA AWQC (for fish consumption). The Upper Simmons Reservoir is the only recreational fishery with a significant potential to be impacted by the landfill. However, based on the lack of AWQC exceedances and based on fish tissue data for metals, the landfill has not adversely affected the edibility of the fish in the Reservoir. Furthermore, the conservative quantitative assessment of potential fish consumption risks indicated no excess health risks are anticipated. Maximum concentrations of dissolved arsenic in Almy Reservoir, the Sedimentation Ponds, Cedar Swamp Brook and the Quarry Stream exceeded its AWQC for fish consumption, and aldrin in the Sedimentation Ponds exceeded its AWQC. However, the Sedimentation Ponds, Quarry Stream and Cedar Swamp Brook do not provide a recreational fishery, therefore this is not a complete exposure pathway. It is unlikely that the landfill contributed significantly to the elevated arsenic concentrations detected in the Almy Reservoir, therefore the risk suggest by this exceedance is not attributable to the landfill.

The concentrations of lead measured in OU2 Study Area soils was below federal and state criteria which are protective of residential exposures to children.

Based on these risk estimates, we conclude that site-related COCs present in the OU2 Study Area media, i.e., groundwater, soil, surface water and sediments, do not pose excess risks based on the exposure scenarios evaluated in this assessment. Consequently, we believe a condition of No Significant Risk for the one Residential Well user receptor, Recreator/Trespasser receptor, and the Facility Worker receptors has been demonstrated to exist under baseline conditions. We also note that groundwater beneath portions of the OU2 Study Area that are downgradient of the OU1 landfill is not suitable for use as a drinking water supply (refer to Sections 6 and 7), and not all of these properties are currently owned by RIRRC.

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