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9.00 ECOLOGICAL RISK ASSESSMENT

An Ecological Risk Assessment (ERA) was conducted for the OU2 Study Area to evaluate whether significant adverse impacts to the natural community may have occurred due to exposure to contaminants migrating from the OU1 Landfill, or if there is a significant risk of adverse impacts in the future. The ERA generally followed the approach outlined in EPA guidance documents including: Ecological Risk Assessment Guidance for Superfund (EPA, 1997); Framework For Ecological Risk Assessments (EPA, 1992c); and the ECO Update series (specifically, EPA 1991b, and 1992b and 1992d).

9.10 PROBLEM FORMULATION

Problem formulation is the process by which the goals, scope and focus of an ERA are established. This section presents a description of the types of contaminants present at Central Landfill, their likely sources and potential contaminant migration pathways. Characteristics of potentially affected habitat areas are described, and concentrations of contaminants detected in these areas are screened to develop a list of Contaminants of Potential Ecological Concern (COPEC) for each area. Finally, information regarding the fate and transport, bioavailability, exposure pathways, and ecotoxicology for the COPECs are evaluated to develop assessment and measurement endpoints which were used in the Analysis and Risk Characterization processes.

9.11 Nature and Extent of Contaminants

This section presents a description of the identified and potential sources of contaminants at the landfill, and the types of contaminants detected in the OU2 Study Area. It presents a conceptual model for contaminant migration pathways from the OU1 Landfill to the OU2 Study Area, and addresses contaminant fate and transport issues. This model is based on information collected during the OU1 RI and OU2 RI studies. Section 9.15.1.2 presents a discussion of the exposure pathways for ecological receptors within the different exposure areas of OU2.

Primary contaminant source areas and types of contaminants disposed of at the landfill are discussed in detail in Sections 2.00 and 8.00 of the OU1 RI Report; Section 1.30 of this OU2 RI also presents a history of waste disposal at Central Landfill. Special Waste and Hazardous Waste materials that were, or may have been, disposed of at Central Landfill include septic waste liquids, septic sludge, industrial waste water treatment plant sludge, industrial solvent wastes, corrosive wastes, acid wastes, water soluble oils, cyanide plating wastes and other sludges. Disposal of Special Wastes and Hazardous Wastes was reportedly limited to the Hot Spot area and ceased in 1979. The final cap for this area was completed in

December 1999. Based on historic records of disposal at the landfill and contaminants detected during the OU1 and OU2 RIs, types of contaminants that may have migrated from the site include volatile organic compounds (VOCs), semi-volatile compounds (SVOCs), polychlorinated biphenyls (PCBs), pesticides and inorganics including metals and nitrogen and phosphorus nutrients.

Three primary potential routes of contaminant migration from the landfill to the surrounding environment have been identified; groundwater migration, surface water transport and fugitive dust. Each is considered briefly in the following paragraphs.

Volatile organic compounds (VOCs) and some relatively mobile metals (e.g., zinc, iron, chromium) are expected to migrate from the landfill as dissolved constituents in groundwater. However, most metals, SVOCs and pesticides have a higher affinity for particulates, and this will limit migration with groundwater. Metals, VOCs and SVOCs could be expected to migrate away from the landfill as dissolved or particulate-bound contaminants in surface water; however, due to volatilization, VOCs would not be expected to persist in surface water or with particulates deposited on the landscape. Metals, VOCs, and SVOCs may also migrate significant distances as particulate bound constituents on fugitive dust; however, VOCs would not be expected to be found in deposited particulates.

Since the time that disposal of Special Wastes and Hazardous Wastes was terminated (1979), these wastes have been covered with greater than 20 feet of septage, refuse and clean fill. Therefore, for the past twenty years, migration of contaminants from OU1 to OU2 is expected to have been largely limited to migration with groundwater flow. Migration of contaminants via surface water runoff or via windblown fugitive dust are expected to have largely ceased in the late 1970s or early 1980s. Therefore, migration of SVOCs, PCBs and pesticides, less mobile metals, and phosphorus from OU1 to the OU2 Study Area is expected to be very limited.

9.11.1 Migration With Groundwater and Surface Water

All of the surface water, and most of the groundwater from the active portion of the landfill drains to the lower portion of the Cedar Swamp Brook watershed. A number of surface water bodies are in the surface water drainage area of the landfill, including the lower, channelized portion of Cedar Swamp Brook, the Quarry Stream (former alignment), Sedimentation Ponds 2, 3, and 4 (Sedimentation Pond 2 is an impoundment of Cedar Swamp Brook; Sedimentation Pond 1 has been eliminated since the OU2 sampling rounds; see Section 9.12.1) the Upper Simmons Reservoir, and the Lower Simmons Reservoir. These waterbodies, with perhaps the exception of Sedimentation Pond 4 and the Lower Simmons Reservoir, also receive groundwater from OU1. These areas have the greatest potential for significant ecological impact from the landfill. To facilitate discussion, these waterbodies are referred to collectively as the "Central Landfill (CLF) Drainage Area."

Cedar Swamp Brook originates in a series of long narrow wetlands located in the forested area west of the active portion of the OU1 Landfill. This upper portion of the Cedar Swamp Brook watershed is referred to in this report as "Cedar Swamp Brook Headwaters". A small impoundment (approximately 50-foot diameter) was excavated at the point where Cedar Swamp Brook emerges from the woodland. This impoundment is referred to as the "Swimming Hole."

The Almy Reservoir, which is located about 1/2 mile northeast of the OU1 Landfill, also receives a small portion of the groundwater flow from OU1 (see below), however, it is outside of the surface water drainage area for the landfill. Wetlands located between the landfill and the Almy also have the potential to receive groundwater (but not surface water) from OU1. The wetland area between the landfill and Almy Reservoir are referred to as the "Almy Watershed."

9.11.1.1 Quarry Stream, Cedar Swamp Brook, Upper Simmons Reservoir and Sedimentation Ponds

Because of the drainage patterns at the landfill, and groundwater flow directions, the channelized portion of Cedar Swamp Brook (i.e., through the landfill area), Sedimentation Ponds 2 and 3, and the Upper Simmons Reservoir have the highest potential for receiving landfill derived contaminants via surface water and groundwater. GZA estimates that approximately 140,000 gallons per day (gpd) (97 percent) of the groundwater that passes beneath the OU1 Landfill area discharges to the Upper Simmons Reservoir (OU1 RI Report; Section 7.43). This 140,000 gpd includes the groundwater flow which passes beneath the Hot Spot and represents approximately 12 percent of the estimated average base flow of 1.2 million gpd[1] for the entire watershed to the Upper Simmons Reservoir (OU1 RI Report; Section 3.24). All stormwater drainage from the OU1 Landfill enters Cedar Swamp Brook (directly or via Quarry Stream or the sedimentation ponds), and then flows to the Upper Simmons Reservoir. Sedimentation from the landfill to the reservoir has been a significant problem in the past. During the period that hazardous wastes were disposed at the landfill, stormwater runoff is expected to have been a significant contaminant migration pathway.

For these reasons, Cedar Swamp Brook, the Upper Simmons Reservoir, Quarry Stream, and Sedimentation Ponds 2, 3, and 4 are considered the primary receptors of OU1 Landfill derived contaminants. Extensive sampling and analysis of sediments from the Upper Simmons Reservoir has been performed, and two data reports have been submitted to EPA ("Upper Simmons Reservoir Sediment Study Phase I Report/Phase II Work Plan, Operable Unit 2, Remedial Investigation - Task 1," February, 1992 [Upper Simmons Reservoir Phase I] and; "Upper Simmons Reservoir Phase II Report, Operable

[1] Base flow into the Upper Simmons Reservoir was estimated based on flow measurements at the weir which separates the Upper and Lower Simmons Reservoirs.

Unit 2, Remedial Investigation - Task 1, Central Landfill, Johnston, Rhode Island," July, 1993 [Upper Simmons Reservoir Phase II]). Additional sampling of surface water and sediment from the Cedar Swamp Brook, Upper Simmons Reservoir, Lower Simmons Reservoir, the Quarry Stream, and sedimentation ponds was performed in support of the OU2 RI. A screening level ecological risk assessment was performed on the Upper Simmons Reservoir Phase I and Phase II data (Upper Simmons Reservoir Screening Report; see Section 1.15). This screening level assessment was used to focus the supplemental sampling of Upper Simmons Reservoir sediments on analyses of PCBs and pesticides. The scope of the additional sampling and analysis was outlined in the OU2 SAP (see Section 1.15).

9.11.1.2 Cedar Swamp Brook Headwaters and the Swimming Hole

No surface water flows from the landfill area to the Swimming Hole or Cedar Swamp Brook Headwaters were observed during the wetland delineation field work conducted in the summer and fall of 1993. Based on the groundwater contour plans included as Figures 5-1 through 5-5, groundwater flow from this area is west to east, from Cedar Swamp toward the landfill area. Therefore, there is little potential for migration of contaminants from the landfill to the Swimming Hole or Cedar Swamp. A limited amount of additional sampling and analysis of surface water and sediments from these areas was performed to confirm that contaminated groundwater does not discharge to these areas. These data were evaluated and it was determined that, with one exception (SED95-46, which was impacted by the M.E. Adams site) they are representative of background conditions.

9.11.1.3 Almy Reservoir and Associated Wetlands

Approximately 4,000 gpd (3 percent) of the groundwater that passes beneath the OU1 Landfill discharges to the Almy Reservoir (OU1 RI Report; Section 7.43). This component of groundwater flow varies seasonally; recent capping of the eastern portion of the Phase I Landfill, which contributes flow to the Almy, may have reduced the average flow. Because groundwater flows toward the Almy Reservoir, the reservoir and associated wetlands between the reservoir and the landfill may potentially receive contaminants migrating with groundwater.

Sampling and analysis of surface water and sediments within the Almy Reservoir and Almy Watershed wetlands was performed to look for contaminants that may be attributable to groundwater from the OU1 Landfill. However, based on groundwater data collected during the OU1 RI, contaminant levels in groundwater flowing toward the Almy Reservoir are much lower than those detected in groundwater flowing toward the Upper Simmons Reservoir, and are generally below MCLs.

There is a wide vegetated buffer between the landfill and the Almy Reservoir, and no surface water flow from the landfill toward the Almy or associated wetlands has been observed. Therefore, these areas are unlikely to have received landfill contaminants via surface water flow. Because of the lack of potential for contaminant transport via surface water runoff and the lower contaminant levels in groundwater as compared to those in water flowing toward the Upper Simmons Reservoir, the potential for significant contaminant migration from the OU1 Landfill to the Almy Reservoir and, to wetlands between the landfill and the Almy is relatively low.

9.11.2 Fugitive Dust

Under present conditions, landfill wastes are covered with soils or other approved alternate cover materials (e.g., construction and demolition debris screenings, dredge spoils, etc.) on a daily basis. Dust control measures are practiced in the active portions of the landfill. Therefore, concerns regarding contaminant migration via fugitive dust are limited to historic conditions when hazardous wastes were still being received at the landfill (up to 1979).

Migration of contaminants via fugitive dust may potentially affect upland as well as wetland and aquatic habitats. Surficial soil samples in undisturbed portions of the upland habitats surrounding the landfill were analyzed for VOCs, metals, SVOCs, and PCB/Pesticides for comparison with background levels to determine if fugitive dust may have been a significant migration pathway.

Surficial soil samples were collected from wooded areas surrounding the active portion of the landfill. The landfill property supported agriculture prior to being used as a soil mining site, and subsequently a landfill. The areas from which the soil samples were collected could have been contaminated by those agricultural practices; the most likely type of contamination would have been from the use of pesticides.

9.12 Ecological Characterization of OU2

GZA delineated wetlands and performed a detailed ecological site characterization for the OU2 Study Area during the summer of 1993. Results of this work were presented to EPA in a report titled "Central Landfill Ecological Characterization, Operable Unit 2 Remedial Investigation - Task 2, Johnston, Rhode Island" (Ecological Characterization) which was submitted to EPA in June 1994. That report included detailed descriptions of wetland and upland habitat areas within the OU2 Study Area, a habitat map, wildlife and aquatic species lists, an existing wetlands delineation plan, an historic wetland map, Wetland Evaluation Technique (WET 2) probability ratings for wetland functions and values, and a discussion of contaminant migration pathways. Figure 3-2 of this report presents the wetland delineation and habitat map developed during GZA's Task 2 study.

Brief descriptions of habitat characteristics which are pertinent to contaminant migration and the identification of ecological receptors and exposure pathways are presented below. Also included, when appropriate, is information about site history, changes that have occurred at the landfill since the Ecological Characterization and chemical data sample collection, and future plans for development within the exposure areas.

9.12.1 Active Landfill Areas Within OU2, and Former Landfill Operation Areas

This section presents descriptions of habitats within the active areas of the landfill, and areas that have formerly been used for soil borrow. However, before this, we describe landscape changes that have occurred since the Ecological Characterization field work and since surface water and sediment samples were collected in support of the OU2 RI.

Landscape Changes Since OU2 Field Work

Ongoing landfill operations, including development of new phases of the landfill build-out, have significantly altered the landscape as compared to the time when the Ecological Characterization was performed, and since surface water and sediment samples were collected in support of the OU2 RI. Changes that are pertinent to the ERA are described below.

Cedar Swamp Brook

RIRRC is currently constructing the Phase IV Landfill within the former southwestern borrow area. As part of this project, RIRRC relocated a section of Cedar Swamp Brook to flow south of Phase IV (i.e., closer to Shun Pike). Because of the elevation of bedrock in the area of the relocated channel, and the grade necessary for the stream bed, a large section of the new stream channel flows through a deep cut in the bedrock. However, the cut was made wide enough to accommodate riparian wetland areas adjacent to the stream, therefore, for the purposes of this ERA, wildlife usage of Cedar Swamp Brook under existing conditions was assumed to be similar to the use observed in 1993 during the Ecological Characterization field work.

In addition to the Cedar Swamp Brook relocation, a sedimentation pond which was present in 1995 and 1996 (referred to as Sedimentation Pond 1), and for which chemical data was collected, was filled in, and replaced with other sedimentation ponds.

Quarry Stream

At the time of the OU2 RI surface water and sediment sampling, RIRRC was in the process of relocating the Quarry Stream to the west of its early tract. Portions have been reworked since the collection of OU2 RI samples, including the installation of a sedimentation pond in the lower reach of the stream near the confluence of Cedar Swamp Brook. For the purposes of this ERA, we have assumed that the condition of the Quarry Stream is essentially the same as it was during the Ecological Characterization field work and during surface water and sediment collection in 1995 and 1996. We believe this assumption is conservative.

Sedimentation Ponds

As mentioned above, Sedimentation Pond 1 has been eliminated due to preparation of the Phase IV Landfill cell.

Since the time that OU2 RI samples were collected, Sedimentation Ponds 2 and 3 were dredged, so sediments which were sampled and analyzed have been removed. However, for the purposes of this ERA, we have assumed that sediments that collect in Sedimentation Ponds 2 and 3 in the future, and surface water in the ponds will be comparable to, or less contaminated than the samples collected in 1995 and 1996. Therefore, use of chemical data from the 1995 and 1996 sampling events is representative of existing conditions, and may be a conservative representation. Note, that none of the existing sedimentation ponds was present during the period of hazardous waste disposal from 1976 and 1979.

Upper Simmons Reservoir

Landfill-derived sediments were dredged from the Upper Simmons Reservoir in 1996. Based on a post-dredge survey performed by Coast Line Engineering of Marion, Massachusetts, and GZA's field observations made during collection of samples for toxicity tests in May 1998; landfill derived sediments appear to have been almost completely removed from the main body of the reservoir (i.e., south of sample location SW/SED96-06 as shown on Figure 3-1), however, measurable thicknesses of landfill derived sediments remain in, or have been redeposited in the North Basin of the Upper Simmons Reservoir (i.e., the area north of sample location SW/SED95-06, which will be referred to as the "North Basin of the USR" throughout this risk assessment). A description of the dredging program is presented below.

Dredging of the Upper Simmons Reservoir was conducted between July 1996, and December 1996. Dredging of sediment from the reservoir included the entire reservoir area from the southern dam to and including portions of the northern delta.

Mobil Dredging of Chester, PA performed the dredging operations utilizing an Ellicott Model 370 Cutterhead dredge supplied with a 12"x10" centrifugal pump and 300 hp booster pump. Dredge spoils material was pumped from the Upper Simmons Reservoir to three stilling basins located on landfill property. Approximately 60 percent of the dredged material was pumped to the first settling basin (Settling Basin 2A located on property commonly known as the "Anderson Property") with the remaining 40 percent in settling basins 1A and 1B located in the area formerly known as the "Southwest Borrow Area". Total capacity of the three stilling basins was estimated at 296,000 cubic yards. Dredge operations were conducted to maximize pumping time by switching between basins to allow sufficient settling time within the stilling basins to achieve proper discharge turbidity levels. The dredging contractor was held to an upper limit of 10 NTU for discharge of waters back into the reservoir. This limit was achieved over a 5 to 7 day settling time within the stilling basins. No coagulants were required throughout the project to aid in settling.

Sediment thicknesses within the reservoir ranged from 6 inches to 5 feet, with the thickest sediments being located within the northern delta. Cutting depths below surface water ranged from 4 to 16 feet based on the pre-dredge survey and a pool elevation of 292.55 MSL (NGVD 29). Dredge operations worked in a southerly direction from the northern delta to the existing dam. It was apparent during the month of November, that additional material within the northern delta still required removal. Further dredging efforts within the north central part of the delta were limited due to bedrock elevations. Dredging on the northeastern side of the delta was extended to within three quarters of the original limit. Dredging in this area was also limited due to large boulders and/or bedrock. Materials dredged within this area included not only the silty landfill-derived sediment but also a good quantity of sand and gravel deposits. Based on Post-Dredge surveys, approximately 221,000 cubic yards of material was removed from the reservoir.

Coast Line Engineering of Marion, MA was hired by the contractor to perform the necessary pre-dredge and post-dredge surveys as well as the pre-dredge core sampling to confirm levels of sediment. A dredge operational plan was prepared for RIRRC in accordance with the specifications. The plan includes the pre-dredge survey results and coring information. Post-dredge surveys of the reservoir were supplied at the end to confirm removal quantity. The pre-dredge survey plan is presented herein as Figure 9-1. The post-dredge survey is presented as Figure 9-2, along with post-dredge cross sections (Figures 9-3 and 9-4).

As mentioned, however, based on field observations in May of 1998, a relatively thin layer of landfill-derived sediment appear to have been left in, or redeposited in the North Basin of USR. Sediment samples for the toxicity tests performed in 1998 (see Section 9.25) were collected with a petite ponar grab sampler which samples to a depth of about 4 to 5 inches. Sample SED98-52 consisted entirely of landfill-derived sediment, therefore, landfill-derived sediments in that area appear to be 5 inches thick or more. Sample SED98-51 consisted mainly of landfill-derived sediment, however original bottom sediments were observed in the bottom of the dredge indicating that the landfill derived sediments in this area were about 4 to 5 inches thick. SED98-50, on the other hand, was composed of original organic sediments, with just a "dusting" of landfill-derived sediment on the surface. This indicates that essentially all of the landfill-derived sediment has been removed from that area, and presumably from the rest of the reservoir down stream (south) of this location.

Dredging performed in the Upper Simmons Reservoir affected usability of data collected from landfill-derived sediment. This issue is discussed in detail in Section 9.13.1.

9.12.1.1 Channelized Portion of Cedar Swamp Brook

As shown on Figure 3-1, the forested area to the north and west of the OU2 Study Area contain the headwaters of Cedar Swamp Brook. Cedar Swamp Brook is formed by several tributary streams which mirror the dendritic pattern formed by several small valleys within the forest west of, and in the western portion of the OU2 Study Area. As Cedar Swamp Brook emerges from the forest it flows through a small man-made pond (approximately 50 feet in diameter), locally identified as the Swimming Hole which was excavated into the natural grade at the edge of the forested area. Below the Swimming Hole, the grade of the land has been lowered significantly as a result of mining, and Cedar Swamp Brook has been channelized from the Swimming Hole to Sedimentation Pond 2 near the entrance to the landfill facility. As mentioned above, a portion of Cedar Swamp Brook was recently re-routed to flow to the south of Phase IV Landfill cell which is currently under construction.

Stormwater drainage from the OU1 Landfill enters Cedar Swamp Brook, and then flows to the Upper Simmons Reservoir. Stormwater runoff either enters Cedar Swamp Brook directly, or via discharge from one of several sedimentation ponds or via the Quarry Stream. Along with the Upper Simmons Reservoir and Sedimentation Ponds 2 and 3, Cedar Swamp Brook is expected to receive groundwater which migrated beneath the OU1 Landfill. Furthermore, groundwater interceptor drains beneath the Phase II Landfill (the Phase II Groundwater Diversion System) may have, at times, pick up leachate from the landfill and discharge it into Cedar Swamp Brook directly. Note, at the time of the OU2 field work, the outlet to this drain flowed into Cedar Swamp Brook via a small channel at sampling station SW95-19. As part of the Phase IV Landfill liner construction, this drainage system was connected to an on-site pre-treatment facility, which then discharges into the Cranston Rhode Island municipal waste water treatment plant. Because of its position as the primary recipient of stormwater and groundwater from the landfill, Cedar Swamp Brook has the highest potential for adverse effects from contamination from the OU1 Landfill, along with Sedimentation Ponds 2 and 3 and the Upper Simmons Reservoir (see below). During the period that hazardous wastes were exposed as surficial material, stormwater runoff from the waste cells was expected to be a significant contaminant migration pathway to Cedar

Swamp Brook. Under current conditions, groundwater is likely to be an ongoing migration pathway from the unlined Phase I Landfill.

As part of the development of Phase IV of the landfill build out, the section of Cedar Swamp Brook from the Swimming Hole to a point about 2,000 feet southeast of the Swimming Hole was re-routed to the south. Much of this section of stream now flows through a deep cut in the bedrock, however, the bedrock cut was made about 40 feet wide to accommodate riparian wetlands along the stream channel. Downstream of the recently relocated stretch of the brook, Cedar Swamp Brook consists of a narrow channel (10 to 15 feet wide) with a slope of approximately 1 percent down to Sedimentation Pond 2. A wetland plant community dominated by cattails, rushes and sedges has developed along this portion of the stream channel. The substrate of the channelized portion of Cedar Swamp Brook is predominantly silt and organic sediments, with moderate percentages of fine to coarse sand.

Herons, egrets and ducks have been observed feeding in Cedar Swamp Brook. Wading predaceous birds were likely to be feeding on amphibians, small fish or benthic invertebrates; ducks were likely to be feeding on plant shoots, grains and fruits.

Despite the presence of these wildlife species, the channelized portion of Cedar Swamp Brook is expected to supply relatively poor aquatic habitat. The operation of the landfill results in large areas of unvegetated, or sparsely vegetated land; and numerous soil and "alternate cover material" stockpiles were present; and all of the haul roads beyond the scale house used by trucks and heavy equipment are unpaved. A buffer strip of vegetation is generally present along most of Cedar Swamp Brook, and other erosion and sedimentation controls are in place. However, the stream receives significant loads of sediment before reaching Sedimentation Pond 2. High sediment inputs and the artificial nature of these stream channels significantly reduce aquatic habitat quality.

In addition, a haul road to bring refuse to the Phase II and III areas runs parallel to the length of the channelized Cedar Swamp Brook, and truck and heavy equipment traffic is substantial between Phase IV, which is under construction, and the landfill entrance. The isolation of Cedar Swamp Brook from other vegetated habitat and the heavy human activity in the area significantly limits the value of this section of the stream as wildlife habitat.

9.12.1.2 Quarry Stream

The Quarry Stream originates from two small wetland areas located outside of the OU2 Study Area (north of the OU1 Landfill), and from Sedimentation Pond 4. The stream also receives discharges from other drainage swales along the north side of the Phase II and III Landfill areas. The Quarry Stream flows into Cedar Swamp Brook about 200 feet east (downstream) of the Swimming Hole. The lower 2,100 feet of the Quarry

Stream channel was moved in 1994 to the western side of the active landfill property. Prior to this relocation, the quarry stream flowed through the current footprint of the Phase II and III Landfill areas. Under current conditions, most of the Quarry Stream is essentially a ditch through soil and bedrock with very little development of vegetative cover or benthic habitat. A sedimentation pond was also constructed as an impoundment of the Quarry Stream at a point about 300 feet upstream from where the Quarry Stream meets Cedar Swamp Brook.

The substrate within the Quarry Stream is predominantly rock, with areas of silty sand and gravel. A short section of the channel (about 700 linear feet near Sedimentation Pond 4), which has been in place for several years, has developed a silty sediment substrate, and substrates of silt overlain by a well developed root mat. The area near Sedimentation Pond 4 has also developed a small (on the order of 2,000 to 3,000 square feet) cattail wetland.

As noted above, the Quarry Stream is essentially an earthen and bedrock ditch. Similar to Cedar Swamp Brook, most of the Quarry Stream is within close proximity to bare soil areas previously used for stockpiles of a variety of off-site alternate cover materials, and daily heavy truck and equipment traffic which is a necessary part of the operation of the landfill. The Quarry Stream is also largely isolated from more extensive vegetated habitats. The highly artificial nature of the Quarry Stream, along with significant inputs of particulates during storm events, the high degree of human activity, and its isolation from larger habitat areas all significantly limit the value of the Quarry Stream as wildlife habitat.

As part of the RIDEM permit for Phases II and III of the landfill, RIRRC is required to restore the Quarry Stream, and that work is on-going.

9.12.1.3 Sedimentation Ponds

Data were collected from four sedimentation ponds in support of the OU2 RI, but as mentioned above, one of those ponds (Sedimentation Pond 1) is no longer in existence. Pond 4 is to the north of the landfill, and Pond 3 is near the southeast corner of the Phase I cell. Pond 2 is an impoundment of Cedar Swamp Brook near the entrance to the facility. The ponds were constructed to detain stormwater and remove suspended particulates from the water column before the water flows to Upper Simmons Reservoir.

Ponds 2 and 3 support small areas of cattail marsh around their perimeters, and each of these has at least some shallow areas that allow wading birds to feed. Sedimentation Pond 4 is lined with rip rap, with relatively steep banks and deep water, and is generally not good habitat for wading birds to feed.

The sensitivity of benthic and aquatic species to chemical contaminants is often directly correlated to their sensitivity to physical disturbances, such as high suspended particulate loads and high sedimentation rates. The frequent input of particulates related to general landfill operations likely prevents the establishment of the most sensitive benthic and aquatic receptors. In addition, the bottom sediments of these ponds are dredged and placed in the landfill on a regular basis; Sedimentation Ponds 2 and 3 were last dredged in the summer of 1998. These factors combine to make the sedimentation ponds poor habitat for aquatic organisms. During sediment sampling of Ponds 2 and 3, GZA observed high densities of oligocheate worms of the family Tubificidae. Tubificid worms are one of a few groups of organisms that can tolerate low oxygen conditions, and often develop high densities in deep water sediments where the hypolimnion (bottom of water column) of lakes become anoxic. Tubificids also often dominate sediments of polluted water bodies where inputs of nutrients, particulates or organic contaminants contribute to sediments and/or water columns with high oxygen demand. Therefore, the presence of these high densities of tubificid worms support the assumption that the detention basins provide low quality aquatic habitat.

Given the physical characteristics of Ponds 2 and 3, the likely presence of some aquatic species to act as a food base for wildlife species, and the fact that predaceous wading shore birds have been observed feeding in the ponds, it is likely that wildlife species will be exposed to contaminants in the sedimentation ponds. However, given the intent of the Sedimentation Ponds, and the likelihood that the physical stress from sedimentation is likely to be a greater problem for aquatic life compared to the potential toxicity of contaminants, aquatic life is not considered to be a significant receptor for the Sedimentation Ponds. Given the rip rap lined banks, and steep slopes, Sedimentation Pond 4 is likely to be a less important feeding area for wading birds or other avian or mammalian wildlife.

9.12.1.4 Upland Habitat Areas

As mentioned above in reference to activities occurring near Cedar Swamp Brook and the Quarry Stream, the upland areas to the west and southwest of the OU1 Landfill are very active with truck and heavy equipment traffic, soil excavation activities for daily cover material, stockpiling of soil, and formerly sludge compost and other alternate cover materials. In addition, the area to the southeast of the waste cells, near the entrance to the facility is occupied by buildings, parking areas, roads and a refuse transfer area for residential disposal of waste and recyclable materials. These areas provide poor wildlife habitat because of the lack of cover and food, and because of extensive human activity.

The surface of the landfill, and a former soil borrow area to the east and north support a herbaceous field which developed following landfill and earth moving activities. Compared to other upland areas within the active portion of the landfill property, the field areas provides stable habitats with far less human activity. Therefore, these field areas are the most important upland habitat within the active or former operations area of the landfill. Plantable soils placed in these areas generally consist of composted septic sludge and recycled organic material (e.g., wood chips). Note that composted septage and wood chips, known as "Billy Mix," is no longer used at the Landfill. Therefore, if surficial contaminants were present in the soil previously, they have likely been buried by this organic material, and the potential for wildlife to be exposure to landfill contaminants in these areas is low.

GZA

9.12.2 Upper Simmons and Lower Simmons Reservoirs

Essentially all stormwater from the landfill and surrounding facilities enters Cedar Swamp Brook, then flows through a culvert under Shun Pike and into the Upper Simmons Reservoir. In addition, most of the groundwater from below the OU1 Landfill discharges to the Upper Simmons Reservoir. For these reasons, the Upper Simmons Reservoir has a potential for receiving contamination from the landfill. However, it should be noted that several other sources of chemical contaminants are present within the watershed of the Upper Simmons, and these sites could account for a portion of the contamination detected in the Upper Simmons Reservoir surface water and sediments. As discussed in Section 3, to the west and east of the Upper Simmons Reservoir, several known or suspected hazardous waste sites are present which have the potential to discharge contaminated stormwater and/or groundwater to the Upper Simmons Reservoir.

Sedimentation from the landfill to the Upper Simmons Reservoir has been a problem in the past; however, construction of, and improvements to Sedimentation Pond 2 and extensive remediation and erosion control efforts on the part of RIRRC have generally mitigated the transport of suspended particulates to the Upper Simmons Reservoir. Landfill-derived sediments deposited in Upper Simmons Reservoir prior to these improvements were dredged in 1996. Within the main body of the reservoir (i.e., south of sample location SW/SED95-06), sediments were removed down to the original bottom sediment, which is composed mainly of mineral wetland soils and wetland peat which formed prior to construction of the reservoir dam (GZA, 1993a). Based on a post dredge survey, conducted by Coast Line Engineering, Inc. in September, October, and November of 1996 and March 1997 over-dredging (typically on the order of 6 inches) occurred over much of the reservoir, therefore, essentially all that is left is original bottom sediments. However, as discussed previously, measurable thicknesses of landfill-derived sediments are still present in the North Basin of the USR.

During the March 1993 sediment investigation, landfill-derived sediment thicknesses were estimated to be up to 3 feet near the mouth of Cedar Swamp Brook, but just 0.5 foot (or less) near the dam between the Upper Simmons Reservoir and the Lower Simmons Reservoir (GZA, 1993a). Therefore, it is unlikely that significant quantities of landfill-derived sediment flowed over the dam and into the Lower Simmons Reservoir.

As explained in Section 5.00, essentially all groundwater from below the OU1 Landfill is expected to discharge to the Upper Simmons Reservoir; direct groundwater transport from the landfill to the Lower Simmons Reservoir is not expected to occur. Since transport of particulates to the Lower Simmons via surface water flow is expected to be minimal, and discharge of groundwater from the OU1 Landfill to the Lower Simmons is not expected, the potential for the Lower Simmons to have received significant levels of contaminants from the landfill is limited to surface water flow.

The northern (upgradient) portions of both the Upper Simmons and Lower Simmons Reservoirs support large areas of emergent cattail marsh. The remainder of the Upper Simmons Reservoir is open water habitat, up to 9 feet deep. Based on conversations with fishermen observed at the Upper Simmons and Lower Simmons Reservoirs, and fish collection for tissue analyses performed by Environmental Science Services (see Appendix K and Section 2.22.3 in Appendix I), these waterbodies support chain pickerel (*Esox niger*), pumpkin seed (*Lepomis gibbosus*), yellow perch (*Perca flavescens*), and large mouth bass (*Micropterus salmoides*).

As reported in the Ecological Characterization report, many wildlife species have been observed in, or are expected to use the Upper and Lower Simmons Reservoirs for feeding and breeding. In particular, great egrets, great blue herons, and other wading predaceous birds have been commonly observed feeding in the reservoirs.

9.12.3 Almy Reservoir and Almy Watershed

No surface water flow, or evidence of surface water flow from the landfill site toward the Almy Reservoir has been observed. Only about 3 percent of the groundwater from beneath OU1 landfill is expected to flow toward the Almy, where it likely discharges to wetlands between the OU1 Landfill and the Almy Reservoir along Central Avenue.

Because only a small percentage of groundwater flow from the OU1 Landfill is expected to flow to the Almy or the Almy Watershed, the potential for the landfill to result in significant contamination in these areas is considered to be low. This finding was substantiated by groundwater quality data collected during the OU2 RI.

9.12.4 Cedar Swamp Brook Headwaters and Former Swimming Hole

The northwestern corner of the OU2 Study Area, west of the active landfill, is a forested area, most of which is occupied by the eastern portion of Cedar Swamp. Cedar Swamp Brook flows out of the forest and into the former Swimming Hole.

Cedar Swamp is a forested wetland which supports a red maple swamp in the lower, wetter areas, and a mix of red oak and red maple forest within the drier portions of the wetlands and in the transition areas between the wetlands and upland red oak forest. At the point of the confluence of these tributaries, just inside the western boundary of OU2 (see Figure 3-2) Cedar Swamp is a channel approximately 8 to 12 feet wide dominated by sand and gravel substrates, with shallow undercuts in the bank. During the summer of 1993, flow in Cedar Swamp Brook in the western portion of OU2 had slowed significantly, and it is possible that during very dry years, Cedar Swamp Brook dries up above the Swimming Hole (see Section 5.3 of this report for more details). A rip-rap dike is located at the outlet of the former Swimming Hole that impedes the movement of fish and other aquatic organisms from the western portion of Cedar Swamp Brook to the Upper Simmons Reservoir. However, several brook trout were observed in the Swimming Hole, and benthic macroinvertebrates are present in the western portion of Cedar Swamp Brook. Therefore, it is likely that during times of low flow, the Swimming Hole acts as a refuge for aquatic organisms.

No surface water flows from the landfill area to the Swimming Hole, or wetlands west of the Swimming Hole were observed during the wetland delineation field work conducted in the summer and fall of 1993. Based on the groundwater contour plans (see Figures 5-1 to 5-5), flow in this area is west to east, from the wetland west of the landfill toward the landfill area. Therefore, there is no migration of contaminants in groundwater from the landfill to the Swimming Hole or wetlands west of the former Swimming Hole.

Because the western portion of Cedar Swamp and the Swimming Hole have little potential to be impacted by contaminants from the site, background surface water and sediment samples were collected from these areas. However, as noted in Section 6.3 of this report, the M. E. Adams and Lot 66 CERCLIS sites discharge contaminated groundwater to this area. Based on the results of chemical analyses, sediment data from SED96-46 was eliminated from the background data set because of the potential for contamination from an off-site source.

9.13 Exposure Point Concentrations

Surface water, sediment and soil data collected during May 1993, December 1995, October, 1996, and May 1998 were used to represent existing conditions. Contaminant migration pathways evaluated for future conditions were: (1) transport of groundwater contaminants to surface water of the Upper Simmons Reservoir; and (2) transport of groundwater contaminants to surface water of the Almy Reservoir.

9.13.1 Existing Condition EPCs

Aquatic and wetland habitats within the OU2 Study Area were divided into six different exposure areas based on contaminant transport pathways, and physical habitat conditions. The six exposure areas identified were: (1) Sedimentation Ponds 2&3 and Channels (which includes existing portion of Cedar Swamp Brook and the Quarry Stream); (2) Sedimentation Pond 4; (3) Upper Simmons Reservoir; (4) Lower Simmons Reservoir; (5) Almy Reservoir; and (6) Almy Watershed. Surface water and sediment data from these areas were grouped, and the average and maximum concentration of each contaminant detected within the data group were considered to be EPCs for current environmental receptors. Table 9-1 presents a list of the surface water and sediment samples used to represent each OU2 exposure area.

Averages were calculated using one-half of the SQL for "not detected" results. In the event that one-half the SQL was greater than the maximum detected, that SQL result was not included in the calculation of the average.

Data collected from the same sampling stations, either as duplicate quality control split samples or from multiple rounds of sampling, were average together to derive "within station" averages. For those stations for which they where applicable, "within station" averages were subsequently used to represent those sampling stations when group averages were calculated and group maximums were identified.

Tables 9-2 through 9-7 present maximum and average concentrations for sediment samples from Sedimentation Pond 4, Sedimentation Ponds 2&3 and Channels, Upper Simmons Reservoir, Lower Simmons Reservoir, Almy Reservoir, and Almy Watershed. Note that, in order to compare site concentrations of organic contaminants in sediment to equilibrium partitioning-based benchmark concentrations organic contaminant concentrations presented on these tables were converted to a mg/kg TOC basis (see Section 9.14.1.1). Sediment organic contaminant concentrations based on sediment dry weight are presented in the tables in Appendix I.

Tables 9-8 through 9-13 present average and maximum detected concentrations for surface water samples from Sedimentation Pond 4, Sedimentation Ponds 2&3 and Channels, Upper Simmons Reservoir, Lower Simmons Reservoir, Almy Reservoir, and Almy Watershed.

All soil data was considered together as one group. Although contaminant deposition may be higher within areas of predominant wind direction, fugitive dust transport was assumed to distribute contaminants relatively uniformly within the affected area. Soil samples were concentrated in the area of the predominant wind direction. As with surface water and sediment, the average and maximum soil concentration were considered to be EPCs.

The usability of some surface water and sediment data for the ERC was affected by landscape changes that occurred between the time of sample collection and the time of this writing. The original intent of some of the samples collected also impacted their usability in the ERC. The following paragraphs discuss which sample locations were eliminated from EPC calculation for the ERC.

9.13.1.1 Data Usability Issues Related to Activities on the Landfill Property

Construction of the Phase IV Landfill, additional work to the Quarry Stream, and dredging of Sedimentation Basins 2 and 3 all have potential ramifications for usability of data collected prior to these changes. Table 9-1 presents a list of surface water and sediment samples used for the ERA. In evaluating which data to eliminate, we took the decided that data from areas still in existence should be used, and data from areas that have been eliminated should not be used. Therefore, we eliminated surface water and sediment data from the section of Cedar Swamp Brook which was moved for construction of the Phase IV Landfill (i.e., SW/SED95-16, -17, and 20), and data collected from Sedimentation Pond 1 which was eliminated for Phase IV construction (i.e., SW/SED95-25 and -26). Although changes have occurred to the Quarry Stream, and Sedimentation Ponds 2 and 3 were dredged, we assumed that existing and future contaminant levels in these areas will be comparable to, or less than, the data collected in 1995 and 1996, therefore these data were retained for use in the ERA.

9.13.1.2 Data Eliminated as Not Representative of Habitat Conditions

Table 9-1 presents a list of surface water and sediment samples used for the ERA. Sample SW96-47 was not included in the EPC calculations for the Quarry Stream because it was a sample of a groundwater seep co-mingled with surface water runoff from stockpiled alternate cover materials consisting of composted septage sludge and other off-site materials. This sample was not considered representative of conditions in the Quarry Stream. Runoff from these stockpiles likely impacted the Quarry Stream beginning upstream of location SW/SED95-22. Sample SW96-47 was collected during Round 2 to see if elevated levels of a few contaminants detected in the Quarry Stream during the first round of sampling might be related to these seeps from the stockpile areas. Surface water sample SW95-19 was not included in the calculation of EPCs for Cedar Swamp Brook because this surface water sample was collected from the discharge of the Phase II Groundwater Diversion System, which is apparently affected by landfill leachate, and therefore, it is not representative of surface water concentrations within Cedar Swamp Brook.

9.13.1.3 Data Eliminated Due to Dredging of Upper Simmons Reservoir

Table 9-1 presents a list of surface water and sediment samples used for the ERA. GZA conducted an extensive sediment sampling program in the Upper Simmons Reservoir in 1992 and 1993 (note, inorganic landfill-derived sediment samples from this effort were designated with an "I" after the sample number (e.g., SED93-21-I), and samples from the organic, or original sediment layers were designated with an "O" after the sample number (e.g., SED93-21-O)). In the spring of 1995, at the request of EPA, GZA performed a screening level risk assessment using the 1993 sediment data. This risk assessment report was submitted to EPA on June 30, 1995 (GZA 1995) and the results were discussed at a meeting on July 12, 1995, attended by EPA, RIDEM, RIRRC, GZA, Haliburton NUS (consultant to the EPA), and the United States Fish and Wildlife Service (USFW). At that meeting there was general agreement that the contaminant concentrations detected in both the landfill-derived sediments and the original bottom sediments did not appear to present a significant ecological concern. However, because of the small number of PCB analyses performed on landfill-derived sediments, USFW requested that additional landfill derived sediments be collected and analyzed for PCBs and pesticides. RIRRC complied with this request, and all samples collected in 1995 (samples SED95-04, -05 -06, and -08) were collected from the surficial landfill-derived sediments. However, because the landfill-derived sediments had been removed at the time the risk assessment was performed, these data were not used in the calculation of EPCs for the ERC.

Subsequent to the 1995 sample collection, Landfill-derived sediments were dredged from the Upper Simmons Reservoir. As discussed previously, virtually all of the landfill-derived sediments appear to have been excavated from the main body of the reservoir. However, measurable thicknesses of landfill-derived sediments are present in the North Basin of USR (i.e., north of sample location SW/SED95-06). Based on this perceived pattern, data from landfill-derived sediment samples collected from the main body of the reservoir (i.e., SED93-24-I, -26-I, -27-I, -30-I; SED95-04, -05 and -07) were eliminated. All sediment sample data (i.e., landfill-derived and "original" sediments) from the North Basin of USR (including the Cedar Swamp Brook delta) were retained for the ERC.

9.13.1.4 Background Data Eliminated Due to Influence of Other Known Sites

Since the use of background data was limited to comparing the maximum detected concentrations to maximum concentrations detected within OU2 (for the purposes of COPEC selection; see Section 9.14.2), the background data set was reviewed to see if samples may have been significantly affected by other known sites of chemical releases. In this case, "significance" was judge based on the number and type of maximum detections attributable to the sample being reviewed. Sediment sample SED96-46 contained the maximum detected concentrations (among background samples) of 3 VOCs, 5 SVOCs and 3 inorganics, and was downgradient of a know hazardous waste disposal site (M. Earl Adams Company located on Peck Hill Road). Therefore, it was eliminated from the background data set.

Surface water sample SW96-46, on the other hand, had the maximum detected concentration of only one contaminant (chloromethane) and was retained because it was comparable to other background samples, and did not affect the outcome of COC selection. Relative to surface water, sediments have a much higher capacity to bind chemical contaminants. It is common for historic releases of chemical contaminants to measurably affect sediments while repeated sampling of surface water from the same location yields "non-detect" results, or results comparable to background. This is particularly true once active disposal has ceased, as is the case with M.E. Adams.

Chemical data for SED96-46 are presented in Tables 6-15 through 6-18. Chemical data for SW96-46 are presented in Tables 6-20 through 6-25.

9.13.2 Future Condition EPCs

The primary point of discharge of OU1 groundwater is the Upper Simmons Reservoir, which is estimated to receive 97 percent of the groundwater which passes beneath the landfill. Almy Reservoir receives approximately 3 percent of the landfill groundwater. Based on the hydraulic properties of the OU1 Landfill and OU2 Study Area (see Section 5.00 of this report), VOC concentrations in groundwater discharging to these reservoirs are though to be generally at steady state, therefore, existing VOC data are also representative of future conditions. However, it is unknown whether metals and SVOCs have reached steady state conditions, therefore, the concentrations of these constituents in groundwater discharging to these waterbodies may increase in the future. Therefore, groundwater data and a dilution model were used to estimate surface water concentrations (considering the incremental contribution by the OU1 waste cell only) under future conditions.

Section 7.00 presents the methods used to estimate future concentrations in surface water due to discharge of contaminated groundwater.

9.14 Selection of Contaminants of Potential Ecological Concern

9.14.1 Contaminant Screening Procedure

Inorganic contaminants were screened by comparing EPCs to background concentrations and to toxicity "benchmarks". Toxicity benchmarks are concentrations in a particular environmental medium below which the risk of harm due to exposure to an individual compound is assumed to be negligible. For each exposure area, the maximum detected concentration of the inorganic contaminant was compared to the maximum detected concentration in the background data set. The maximum concentration detected within the exposure area was also compared to the appropriate toxicity benchmark concentration. Contaminants were considered to be COPECs if they had a maximum concentration greater than the maximum background concentration, and the maximum concentration was greater than the toxicity benchmark concentration. If the maximum concentration was less than the maximum background concentration, or the maximum concentration was below the benchmark, then the contaminant was eliminated from further consideration.

As per EPA policy, background data were not considered in the screening procedure for organic (i.e., manmade) contaminants; organic contaminants were screened from the ERA based on the comparison to benchmark concentrations only.

Toxicity benchmark concentrations are intended to be protective of organisms whose main route of exposure is via direct contact; they typically do not take into consideration potential impacts to higher trophic level organisms due to exposure via the food web. Therefore, some contaminants were retained because they have the potential to be highly bioaccumulative, and therefore, have a significant potential to adversely affect higher trophic levels.

9.14.1.1 ARARs and Toxicity Benchmark Concentrations

The term "benchmark" is used as a generalized term because ecological risk assessments rely upon a mixture of state and federal criteria, standards and guidelines. To facilitate discussion, ARARs are included in the term "benchmarks" along with other guideline values. The only ARARs identified specifically for the ERA were Water Quality Standards and Guidelines from RIDEM for the protection of freshwater aquatic life. For contaminants for which EPA Ambient Water Quality Criteria (AWQC) are available, RIDEM has adopted the EPA AWQC as State Water Quality Standards. For those contaminants for which they are available, AWQC were used as the toxicity benchmark concentrations. Many priority pollutants do not have AWQC, and for many of these, RIDEM developed Minimum Data Base Guidelines (MDBGs). These were also considered ARARs.

Surface Water Quality Benchmarks

As mentioned above, AWQCs for the protection of aquatic organisms were used as water quality benchmark concentrations for those contaminants for which they are available. AWQC for the protection of aquatic life are expressed as acute and chronic concentrations. Acute AWQC are intended to protect organisms from toxic effects during short-term exposures to contaminants in surface water. Chronic AWQCs are intended to protect organisms from exposure to contaminants over a longer period of time. Because the contaminant migration from the OU1 Landfill is likely to be an ongoing input of contaminants to surface water, rather than periodic or pulsed discharges, chronic benchmarks are more appropriate for this ecological screening.

For contaminants for which AWQC are not available, other sources of water quality benchmarks were consulted. Sources consulted are listed in order of preference, beginning with the preferred sources: RIDEM MDBGs, Great Lakes Tier II values (published by the Office of Solid Waste and Emergency Response [EPA 1996]), Lowest Observed Effect Levels (LOELs) cited by EPA in the Ambient Water Quality Criteria Documents. Note that many of the Tier II values cited in EPA, 1996 were derived by the Oak

Ridge National Laboratories (Suter and Mabrey, 1994). Subsequent to the publication of EPA 1996, Oak Ridge National Laboratories updated their Tier II values (Suter and Tsao, 1996), and when available these updated values were used in place of those published by EPA in 1996.

AWQC for seven metals (cadmium, chromium, copper, lead, nickel, silver, and zinc) are hardness dependent. For these metals the value used as the benchmark within a given exposure area was calculated based on the average hardness value measured among the samples collected from that exposure area. The only exceptions to this were when average hardness was less than 25 mg/l as $CaCO_3$ in which case, 25 mg/l was used as a hardness value in the calculations as per EPA guidance.

In their "Interim Final Rule" EPA recommends used of dissolved metals data (EPA 1995a) for comparison to AWQC for arsenic, cadmium, chromium, copper, lead, mercury (acute only), nickel and silver (acute only). For these comparisons, a total-to-dissolved conversion factor is applied to the published AWQC (which is based on total metals) to derive an AWQC for dissolved arsenic, cadmium, chromium, copper, lead, nickel, and selenium (EPA, 1998). As part of the OU2 RI, both total and dissolved inorganic analyses were performed. When possible, benchmark concentrations based on total metals were compared to total metals data, and dissolved metals benchmarks were compared to dissolved metals benchmarks. In cases where a benchmark was available for a dissolved metal but not for total concentrations, the dissolved benchmarks were used to conservatively compare to the maximum concentrations detected.

Water quality benchmarks used for this ecological screening are presented on Table 9-14.

Uncertainties Statement

Generally, water quality benchmarks used to evaluate contaminant levels in surface water are derived based on laboratory toxicity data. Characteristics of water used for these tests (i.e., basically pure water without suspended particulates or dissolved organic carbon) may significantly increase toxicity as compared to wetland pools, streams and ponds similar to those found in OU2. Species used to derive the benchmarks may be more sensitive than species found in the nearby water ways. Metal speciation and adsorption of VOCs and SVOCs to dissolved organics and to suspended particulates in natural surface water may significantly decrease bioavailability and toxicity. Use of AWQC, which are also derived from laboratory aquatic toxicity tests, is likely to be conservative (i.e., overly protective) because AWQC are intended to protect all but the most sensitive aquatic species and toxicity test solutions probably contain higher proportions of bioavailable contaminants.

Sediment Quality Benchmarks

The metallic contaminant section below describes the protective toxicity benchmarks used for inorganic contaminants in sediments and wetland soils, and the organic contaminant section describes how protective toxicity benchmarks were developed for organic contaminants in sediment.

Metallic Contaminants in Sediment

Toxic effects levels developed by the Ontario Ministry of the Environment (Persaud, et al., 1993) or U.S. National Oceanic and Atmospheric Administration (Long and Morgan, 1991) were used as toxicity benchmarks for metals in sediment and wetland soils. Screening levels presented by Persaud et al. were developed for freshwater bodies, and were used preferentially over Long and Morgan effects ranges because the latter values are based on data from a combination of marine, estuarine and freshwater systems. Long and Morgan values were used only when they present a value for a metal for which no guideline in available from Persaud.

Persaud et al. (1993) present guidelines as Lowest Effects Levels (LELs), which are defined as levels of sediment contamination that can be tolerated by the majority of benthic organisms. Long and Morgan present Effects-Range Low (ER-Ls) which are the lower tenth percentile of concentrations observed or expected to have an adverse effect on benthic organisms.

Table 9-15 presents sediment quality benchmarks used for metals in sediment.

Organic Compounds in Sediment

The equilibrium partitioning method (EqP; EPA, 1991a) was used to derive chronic benchmarks for non-polar organic compounds. This approach is based on the premise that the bioavailable portion of the organic compound is largely limited to the concentration in pore water (i.e., water held in the pore spaces between sediment or soil grains), and at equilibrium, the concentration in pore water is mainly determined by the chemical's organic carbon/water partitioning coefficient (Koc) and the Total Organic Carbon (TOC) content of the sediment. Therefore, given the chemical's partitioning coefficient and TOC content of the sediment, the benchmark is the total concentration of the compound in sediment that results in a pore water concentration equal to the chronic water quality benchmark.

EPA has derived Sediment Quality Criteria (SQC) for five non-polar organic compounds (acenaphthene, fluoranthene, phenanthrene, dieldrin and endrin) using the equilibrium partitioning approach. For contaminants for which EPA SQC are not available, other sources of EqP-based sediment benchmarks were consulted. Sources consulted are listed in order of preference, beginning with the preferred source: EPA Sediment Quality Benchmarks (SQBs) for organic contaminants presented as Ecotox Thresholds in EPA's EcoUpdate, Volume 3, Number 2 (EPA, 1996), EqP-based sediment benchmarks developed for the U.S. Department of Energy by Oak Ridge National Laboratories (ORNL; Hull and Suter, 1994), sediment benchmarks derived by GZA using water quality benchmarks (as identified in Section 3.12.1) and published Kow or Koc values. Note that many of the sediment benchmarks published in Hull and Suter (1994) were based on Tier II water quality benchmarks developed by ORNL in 1994. Subsequent to the publication of those sediment benchmarks, those Tier II water quality benchmarks were updated by Suter and Tsao (1996). GZA used the available updated water quality benchmarks from Suter and Tsao (1996) to calculate updated EqP-based benchmarks for the affected organic contaminants.

Sediment quality benchmarks used for organic contaminants are presented on Table 9-15.

TOC analyses were conducted for all sediment samples collected in support of this ERA. Prior to the calculation of EPCs as discussed above, concentrations of organic chemicals detected in all sediment samples were normalized to the TOC content of that sample, and were expressed as mg/kg-TOC using the flowing formula:

$$\frac{\text{Contaminant}_{\text{mg/kg dry wgt.}}}{(\text{TOC}_{\text{mg/kg dry wgt.}})\,(1 \times 10^{-6}\ _{\text{kg/mg}})}$$

EPCs were then calculated as discussed above, thus, organic contaminant EPCs presented in the ecological risk assessment tables are also presented as mg/kg-TOC. The benchmark concentrations for organic contaminants in sediment were also expressed as mg/kg-TOC (Table 9-15), and were compared directly to the converted average and maximum detected concentrations for each exposure area.

Uncertainties Statement

The Equilibrium Partitioning (EqP) method of characterizing sediment contamination is subject to a number of limitations and uncertainties. The method is recommended for use with sediments with TOC concentrations between 0.2 to 12 percent (EPA, 1992a). The EPA Science Advisory Board (EPA, 1992a) estimated an uncertainty factor of five for EqP benchmarks. That is, contaminant concentrations between one-fifth and five times the EqP benchmark are within a gray area within which observable impacts may or may not occur. For concentrations below one-fifth of the EqP benchmark, there is a high degree of certainty that impacts would not occur, and for concentrations above five times the benchmark there is a high degree of certainty that impact would occur. These uncertainties should be kept in mind when evaluating results of the comparison between Site data and the toxicity benchmarks.

LELs and SELs were derived using a method referred to as "The Screening Level Concentration Approach", which is based on the co-occurrence of benthic species and sediments with a range of concentrations of the contaminant for which they are being derived. These benchmarks are not based on any data that suggests that the contaminant has caused an adverse effect to the benthic community, rather the method relies on the use of a variety of species (presumably ranging from sensitive to tolerant species), and a wide variety of contaminant concentrations. A potential pit fall of this method is that the co-occurrence data available is skewed to the lower end of the range of concentrations. This could lead to overly conservative LELs and SELs. Persaud did not present the co-occurrence data used, or an analysis of the data. However, the LELs and SELs compare well with benchmarks developed by the U.S. National Oceanic and Atmospheric Administration (Long and Morgan, 1991) for benthic marine organisms which were based on data that did indicate adverse effects. Therefore, it is unlikely that the Persaud numbers are excessively conservative.

Persaud et al. (1993) notes that many of the LELs presented therein may be below background concentrations. Rojoko (1990) evaluated metals data for sediment throughout Massachusetts, a neighboring state, and developed sediment concentrations for several metals that are considered "normal" in the absence of obvious anthropogenic sources. Table 9-16 presents Rojoko's "normal" values for Massachusetts lakes. Comparisons of these "normal" sediment concentrations for Massachusetts to the sediment benchmarks from Persaud et al. (1993) and Long and Morgan (1991) suggests that for all of these metals, except manganese (for which the "normal" value is less than the LEL), the benchmark concentrations could frequently indicate a potential for risk even in waterbodies that have no identifiable source of metal contaminant other than those associated with common land uses (i.e., residential and farm land).

Sediment toxicity benchmarks have been developed primarily for bottom sediments of lakes, ponds, rivers and streams. Application of these benchmarks to sediments of the seasonally inundated wetlands within OU2 introduces uncertainty in three ways: The type of organisms found in seasonally saturated sediments, and their sensitivity to contaminants may differ from those found in bottom sediments. Bioavailability of contaminants in seasonally unsaturated (oxic) wetland soil may differ from continually saturated bottom sediments. Finally, the exposure regime of organisms in seasonally saturated/inundated wetland soils may differ from organisms in contact with bottom sediments. In general, it is unclear whether the use of these benchmarks will overestimate or underestimate toxicity to organisms in wetland soils.

9.14.1.2 Soil Benchmarks

No State or Federal benchmarks are available which are intended to protected ecological receptors from risk due to exposures to soil contaminants. Therefore, soil benchmarks developed by Oak Ridge National Laboratories for the protection of soil invertebrates (Efroymson, et al., 1997a) and plants (Efroymson, et al., 1997b) were used as benchmarks for this ERA. Table 9-17 presents the soil benchmark concentrations.

9.14.2 Background Concentrations

Section 6.00 presents the background analytical results of surficial soil, sediment, and surface water. For the ERA, the maximum detected concentration in soil, sediment and surface water, among all of the background samples within each medium were used in this evaluation. The maximum detected background concentration for each contaminant in each medium was compared to the corresponding maximum target sample concentration within each exposure area. If the exposure area maximum was less than the background maximum, then that contaminant was not considered to be elevated within that exposure area, and it was not considered further in the evaluation of that exposure area.

9.14.3 Surface Water Contaminant Screening

Sedimentation Pond 4 Surface Water

Table 9-8 presents maximum and average concentrations of contaminants detected in Sedimentation Pond 4 surface water, and indicates which contaminants were retained as COPECs.

Only one organic compound was detected in surface water (butylbenzylphthalate), and it was retained as a COPEC because the concentration detected was above the benchmark.

Thirteen metals were detected, only barium was retained as a COPEC because it exceeded its benchmark concentration.

Sedimentation Ponds 2&3 and Channels Surface Water

Table 9-9 presents maximum and average concentrations of contaminants detected in Sedimentation Ponds 2&3 and Channels surface water, and indicates which contaminants were retained as COPECs.

Several VOCs were detected, but all had maximum concentrations below benchmarks, therefore, none were retained as COCs.

Two SVOCs were detected in surface water, and one (phenol) was retained as a COC because the maximum detected concentration is slightly above the benchmark.

Aldrin was the only PCB or pesticide detected in surface water. The maximum concentration is well below the benchmark, however, aldrin was retained as a COC because it has a high potential for bioaccumulation.

Nine inorganics (aluminum, ammonia, barium, beryllium, cyanide, iron, mercury, selenium, and zinc) were retained as COCs because the maximum concentration of each exceeded the benchmark. Note that with two exceptions, all of these inorganics were each retained on the basis of total analyses (i.e., unfiltered samples). The exceptions were barium and selenium, for which the maximum dissolved results were higher than the maximum total results. Both of these metals were retained because the maximum detected concentration exceeded both background and the benchmark.

Upper Simmons Reservoir Surface Water

Table 9-10 presents maximum and average concentrations of contaminants detected in Upper Simmons Reservoir surface water, and indicates which contaminants were retained as COPECs.

Three VOCs (1,2 dichlorobenzene, 1,4-dichlorobenzene, and chlorobenzene) were retained as COPEC because they slightly exceed their benchmark concentrations.

One SVOC (butylbenzylphthalate) was retained as COPECs because it slightly exceeded its benchmark.

One pesticide (delta-BHC) was detected at a concentration well below its benchmark; however, it was retained because it has a high potential for bioaccumulation through the food web.

Eight inorganic contaminants (aluminum, ammonia, barium, copper, cyanide, manganese, selenium, and thallium) were retained as COPEC because they exceeded their benchmarks. With the exception of copper and manganese, all were retained on the basis of total analyses, although dissolved barium, and selenium also exceeded the benchmark (also, the maximum dissolved selenium result was greater then the maximum total selenium result). Copper was retained on the basis of the dissolved metals result: the maximum dissolved copper concentration was approximately twice as large as the maximum total copper result. Manganese was retained on the basis of both total and dissolved results.

Lower Simmons Reservoir Surface Water

Table 9-11 presents maximum and average concentrations of contaminants detected in Lower Simmons Reservoir surface water, and indicates which contaminants were retained as COPECs.

Two VOCs were detected in Lower Simmons Reservoir surface water samples but the maximum detected concentrations were below the benchmark values, and therefore they are not considered COPECs. No SVOCs were detected in Lower Simmons Reservoir surface water.

Three pesticides were detected in Lower Simmons Reservoir surface water. The maximum concentration of 4,4'-DDT was above the benchmark concentration, and was therefore retained as a COPEC. Maximum detected concentrations of aldrin and endosulfan I were below their benchmarks, however, these pesticides were retained as COPECs because of their potential for bioaccumulation.

Four metals (aluminum, barium, iron, and silver) were retained as COPECs because the maximum detected concentrations exceeded background and their respective benchmark values. All four were retained on the basis of total metals analyses. The dissolved concentration of barium also exceeded the benchmark concentration.

Almy Reservoir Surface Water

Table 9-12 presents maximum and average concentrations of contaminants detected in Almy Reservoir surface water, and indicates which contaminants were retained as COPECs.

No organic contaminants (VOCs, SVOCs, Pesticides or PCBs) were detected in surface water of the Almy Reservoir.

Copper was the only contaminant retained as a COPEC. It was retained on the basis of both total and dissolved metals analyses.

Almy Watershed Surface Water

Table 9-13 presents maximum and average concentrations of contaminants detected in surface water of the Almy Watershed, and indicates which contaminants were retained as COPECs.

Two VOCs were detected in surface water samples from the Almy Watershed; both were retained as COPECs because their maximum concentrations exceeded the benchmarks.

Three metals (copper, lead and silver) were retained as COPECs. Silver was retained on the basis of total metals analyses. Copper and lead were retained on the basis of dissolved metals analyses; lead was not detected in the total metals analyses, and the maximum dissolved concentration of copper was about three times greater than the maximum total concentration.

9.14.4 Sediment Contaminant Screening

Sedimentation Pond 4 Sediment

Table 9-2 presents maximum and average concentrations of contaminants detected in Sedimentation Pond 4 sediment, and indicates which contaminants were retained as COPECs.

No organic contaminants were detected in the sediment sample from Sedimentation Pond 4. Several metals were detected, however, only barium was retained as a COPEC. The detected concentration of barium was slightly higher than the maximum background concentration, and there is no benchmark for barium.

Sedimentation Ponds 2&3 and Channels Sediment

Table 9-3 presents maximum and average concentrations of contaminants detected in sediment samples from Sedimentation Ponds 2&3 and Channels, and indicates which contaminants were retained as COPECs.

Acetone was the only VOC retained as a COPEC. Seven SVOCs were retained as COPECs; one of which (carbozole) was retained because a benchmark value was not available.

Three pesticides were detected in Sedimentation Ponds 2&3 and Channels sediment samples; all three were retained as COPECs.

Ten metals (arsenic, barium, chromium, copper, iron, manganese, mercury, nickel, vanadium, and zinc) were retained as COPECs. Most were retained as COPEC because the maximum detected concentrations exceeded background and their benchmark concentrations; two metals (barium and vanadium) were retained because the maximum concentrations were above background, and no benchmark values were available.

Upper Simmons Reservoir Sediment

Table 9-4 presents maximum and average concentrations of contaminants detected in Upper Simmons Reservoir sediment, and indicates which contaminants were retained as COPECs.

Two VOCs (acetone and carbon disulfide) were retained as COPECs because the maximum concentrations slightly exceeded their benchmarks. In addition, styrene was retained because a benchmark was not available. Five SVOCs were retained as COPECs; three were retained because maximums exceeded the benchmarks; two were retained because no benchmarks were available.

The maximum concentrations of all pesticides and PCBs were well below their respective benchmarks. However, these contaminants have a high potential for bioaccumulation, and were thus retained as COPEC.

Seven metals (arsenic, cadmium, iron, manganese, mercury, nickel, and zinc) were retained as COPEC because the maximum concentrations exceeded background concentrations and their benchmarks. Four additional inorganics (barium, beryllium, cyanide, and vanadium) were retained because maximum concentrations were above background, and there were no benchmarks available.

Lower Simmons Reservoir Sediment

Table 9-5 presents maximum and average concentrations of contaminants detected in Lower Simmons Reservoir sediment samples, and indicates which contaminants were retained as COPEC.

Two VOCs were retained as COPECs; in the case of acetone it exceeded its benchmark value, and in the case of chloromethane, there was no benchmark available. 4-chloro-3-methyulphenol was the only SVOC retained; it was retained because a benchmark was not available.

Five pesticides were detected in Lower Simmons Reservoir sediment. All were well below their benchmarks but were retained because they have a high potential for bioaccumulation.

Nine metals (arsenic, cadmium, chromium, copper, iron, manganese, nickel, selenium, and zinc) were retained as COPEC because the maximum detected concentrations exceeded background and available benchmark concentrations. Five inorganics (barium, beryllium, cyanide, thallium, and vanadium) were retained because they exceeded background and no benchmark concentrations were available.

Almy Reservoir Sediment

Table 9-6 presents maximum and average concentrations of contaminants detected in the Almy Reservoir sediment samples, and indicates which contaminants were retained as COPEC.

The only class of organic contaminants detected in Almy Reservoir sediments were SVOCs, and none of them were retained as COPECs.

Nine metals (arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, and zinc) were retained as COPECs because the maximum detected concentrations exceeded background, and available benchmark values. Four additional metals (barium, beryllium, thallium, and vanadium) exceeded background and no benchmarks were available, thus, they were retained as COPECs.

Almy Watershed Sediment

Table 9-7 presents maximum and average concentrations of contaminants detected in the Almy Watershed sediment samples, and indicates which contaminants were retained as COPEC.

None of the VOCs or SVOCs detected were retained as COPECs because all were below benchmark concentrations.

Four pesticides were also detected, and all were well below their benchmark concentrations. However, all detected pesticides were retained as COPECs because they have the potential to be highly bioaccumulative.

Six metals (cadmium, copper, iron, manganese, selenium, and zinc) were retained as COPECs because the maximum detected concentrations were above background and their benchmark values. Three additional metals (barium, beryllium, and thallium) were retained as COPECs because they were above background, and benchmark values were not available.

9.14.5 Soil Contaminant Screening

Table 9-17 presents maximum and average concentrations of contaminants detected in upland, forested soils around the active landfill facility, and indicates which contaminants were retained as COPEC.

Benchmark concentrations were available for only three VOCs and one SVOC detected in soil samples from around the landfill. In all four of these cases, the contaminants were eliminated as COPECs because the maximum concentrations were below their benchmarks. The remainder of the VOCs and SVOCs (16 all together) were retained as COCs because of the lack of benchmark concentrations (background data were not used to screen out organic contaminants as COPECs).

Six pesticides were detected in upland soil samples. All were retained because there were no benchmarks available for them, and because they have a high potential for bioaccumulation.

Five metals (chromium, lead, manganese, selenium, vanadium, and zinc) were retained as COPEC because the maximum concentrations were greater than background and their benchmark concentrations.

9.14.6 Potential Future Conditions

9.14.6.1 Future Surface Water Conditions

Refer to Table 9-18 for a summary of estimated future discharge concentrations to the Upper Simmons Reservoir surface water. Table 9-19 presents TQs for estimated future surface water concentrations in Almy Reservoir.

Based on the groundwater transport model presented in Section 7.00, the only contaminants in groundwater from OU1 that may exceed their water quality benchmarks in the future are barium and thallium in the Upper Simmons Reservoir (Table 9-18). These two metals were retained as COPECs for the Upper Simmons Reservoir. None of the estimated future surface water contaminant concentrations in the Almy Reservoir exceed their benchmarks (Table 9-19).

Seven pesticides were detected in groundwater samples from wells within the watershed to Upper Simmons Reservoir. Although the estimated future concentrations are very small (mainly 10^{-6} mg/l to 10^{-7} mg/l) and are all well below their benchmarks, based on EPA policy these contaminants have been retained as Future COPECs because they have a relatively high potential for bioaccumulation. One pesticide was detected in samples from wells within the drainage area to Almy Reservoir. As with the Upper Simmons Reservoir, estimated future concentration of this pesticide is very small (5.2×10^{-7}) and was well below the benchmark, this pesticide was retained because it has a relatively high potential for bioaccumulation.

9.14.7 Contaminants of Concern

Table 9-20 presents a summary of COPECs selected from surface water within each of the exposure areas, and Table 9-21 presents a summary of COPECs selected for sediment within each of the exposure areas.

9.15 Conceptual Site Model

Site-specific, biological evaluations performed for the risk assessment were focused on those areas which have the greatest potential for adverse impact from the OU1 Landfill; namely Sedimentation Pond 4, Sedimentation Ponds 2 & 3 and Channels, the Upper Simmons Reservoir, and the Lower Simmons Reservoir. Relative to the potential impacts to the Upper Simmons Reservoir, the OU1 landfill has little potential to impact the Almy Reservoir. There are not surface water flow paths from the OU1 landfill toward the Almy Reservoir. Only a small percentage of groundwater from the OU1 landfill flows toward the Almy, and contaminant migration, if it occurs, is limited.

9.15.1 Exposure Assessment

9.15.1.1 Distribution of COCs

Sediment

Table 9-16 presents "normal" concentrations of several metallic COPECs in unimpacted Massachusetts lakes (Rojoko 1990), and compares them to the average and maximum detected concentrations within each of the six exposure areas identified for OU2. None of the metallic COPECs in Sedimentation Pond 4 exceed these "normal" levels. With few exceptions (mainly for manganese and zinc), average concentrations do not exceed the range of "normal" values. However, many of the maximum detected concentrations exceed the "normal" levels. This pattern indicates that many contaminants may be found in the waterbodies of the OU2 Study Area concentrations slightly higher than can be expected in unimpacted waterbodies. However, because so few average concentrations exceed the "normal" levels, these comparisons indicate that contamination within these areas is relatively minor.

Soil

Table 9-22 compares maximum and average metallic COPEC concentrations in soil to typical ranges and averages found in U.S. soils, and to the 90th percentile found in unimpacted soils as determined by the Massachusetts Department of Environmental Protection. With the exception of zinc, metals concentrations in upland soils are less than, or comparable to typical ranges in unimpacted areas. In addition, the high maximum and average for zinc are driven by one very high detection: if data from that sample is not included in calculations, the average concentration of zinc is 76 mg/kg, which is well within the normal concentration ranges. These comparisons suggest that metals concentrations in soils are just slightly higher than, or comparable to typical ranges from unimpacted areas in New England.

9.15.1.2 Exposure Pathways

Waterbodies within the CLF Drainage Area

Stormwater erosion and groundwater transport from the OU1 Landfill has resulted in elevated levels of contaminants in the surface water and sediment of the waterbodies within the CLF Drainage Area. For the purposes of this report the "CLF Drainage Area" Sedimentation Pond 4, Sedimentation Ponds 2 & 3 and Channels, the Upper Simmons Reservoir, and the Lower Simmons Reservoir. COPECs within the CLF Drainage Area include VOCs, SVOCs, PCBs, Pesticides, and inorganic contaminants. Contaminants are selected as COPECs by comparing site-related concentrations in surface water and sediment to conservative benchmark concentrations, which are intended to protect aquatic organisms from adverse effects due to direct exposure. Therefore, by definition, surface water COPECs present a concern for adverse effects to aquatic organisms exposed to contaminants in the water column, and sediment COPECs present a concern for adverse effects to benthic organisms that live in or on the sediment.

The Upper and Lower Simmons Reservoirs are relatively large waterbodies (approximately 50 acres each) and are surrounded by significant areas of open space including woodlands and agricultural fields. These waterbodies represent a significant aquatic habitat area, and support local populations of fish and wildlife which use the reservoirs for feeding and breeding. Fish, amphibians and reptiles may be impacted directly by exposure to OU1-related contaminants, and they may be impacted indirectly by the reduction or loss of plankton or benthic invertebrates which serve as prey organisms. Such reductions in the availability of prey species may also indirectly affect mammalian and avian wildlife that feed in the reservoirs. Therefore, aquatic organisms which live in the water column and in or on the sediment of the Upper and Lower Simmons Reservoirs are considered to be significant resources which support local fish and wildlife.

The sedimentation ponds within the active portion of the landfill are engineered structures, built with the intention of capturing eroded sediments before surface water leaves the landfill property and enters the Upper Simmons Reservoir. In addition, the Quarry Stream and the lower portion of Cedar Swamp Brook are channelized streams which are surrounded by earthmoving and land filling activities. These activities isolate these streams from other habitat types, and greatly reduces the value of these streams as habitat. In addition, the surrounding activities result in significant sediment loads to these streams. Consistent input of sediment to these basins and stream sections is likely to prevent the establishment of a healthy, diverse aquatic community. Given that the intended purpose of these structures prevents the establishment of a diverse community, and that the input of sediment is likely to present a greater risk of harm as compared to the input of chemical contaminants, exposure of the aquatic community to surface water and sediment COPECs is not a significant concern for the sedimentation ponds or the channelized portions of Quarry Stream and Cedar Swamp Brook.

In addition to the risk of harm to organisms that are exposed directly, COPECs may accumulate in organisms that serve as prey for higher trophic level species. Thus they present a potential risk of harm to wildlife species that have a relatively small potential for direct exposure, but which feed within the waterbodies of the CLF Drainage Area. Some contaminants have a greater potential for accumulating within prey organisms, and thus present a greater potential for risk to the higher trophic levels. Heavy chlorinated organic contaminants, such as organochlorine pesticides, have long been recognized as having a relatively high potential for bioaccumulation, and a recent article by Russel et al., (1999) indicated that biomagnification through the aquatic food web (i.e., greater exposure to, and uptake by higher trophic level species due to accumulated contaminants in prey) occurs for contaminants with log Kow values greater than 5.5. Several of the organic COPECs within the CLF Drainage Area have log Kow values greater than 5.5.

Certain inorganic contaminants are also recognized as having a relatively high potential for bioaccumulation, and thus risk to higher trophic-level species. COPECs, which have a higher potential for bioaccumulation include cadmium and mercury.

As mentioned above, the Upper and Lower Simmons Reservoirs are relatively large aquatic habitats that are well integrated with surrounding wetland and upland habitats, and are used by local fish and wildlife for feeding. Therefore, the presence of bioaccumulative contaminants in surface water and sediments of the reservoirs raises the concern that higher trophic-level organisms may be adversely affected by OU1-related COPECs.

Although the protection of aquatic life within the sedimentation ponds and the channelized portions of the Quarry Stream and Cedar Swamp Brook is not considered a significant concern, there is an aquatic community within the waterbodies, and predaceous wading birds and ducks have been seen feeding in these areas. For this reason, potential impacts to higher trophic-level species that feed in waterbodies within the active portion of the landfill property are considered to be a significant concern.

Future Conditions with Waterbodies of the CLF Drainage Area

As discussed in Sections 9.13.2 and 9.14.6, contaminants in groundwater may migrate to the Upper Simmons Reservoir and result in concentrations which are different than those represented by surface water data used to evaluate existing conditions. A groundwater transport model was used to estimate potential "worse case" future concentrations in the Upper Simmons Surface water, and these estimated values were screened based on background concentrations and/or toxicity benchmark concentrations. Barium, thallium, and several pesticides were retained as COPEC for future conditions. Because of the potential presence of bioaccumulative contaminants in surface water in the future, there is the potential for risk to receptors the live or feed in the Upper Simmons Reservoir due to direct exposure as well as through the food chain.

Terrestrial Exposures

Contaminants in soil may adversely affect organisms that are exposed directly, such as plants or soil invertebrates, or may accumulate in prey organisms and present a risk to higher trophic-level organisms. With few exceptions, contaminants detected in soil were retained as COPECs because there was not a benchmark available to screen for affects due to direct exposure, or because they are considered to have a high potential for bioaccumulation.

Given the lack of benchmark concentrations, and the conservative uncertainties surrounding the available benchmarks (see Section 9.14), the main concern for contaminants in upland soil of the OU2 is the potential for toxic effects to higher trophic level-organisms through the food web.

9.15.2 Selection of Assessment Endpoints and Measurement Endpoints

9.15.2.1 Ability of Affected Water bodies to Support Local Fish and Wildlife

The presence of COPECs in surface water and sediment of the waterbodies within the CLF Drainage Area have the potential to directly affect exposed receptors through direct exposure, or through the food web. They also have the potential to indirectly affect higher trophic levels organisms by eliminating or reducing the prey base made up of pelagic and benthic aquatic organisms. The following assessment and measurement endpoints were chosen to evaluate whether adverse effects via these routes may be occurring within the CLF Drainage Area.

As discussed above, the risks to aquatic organisms in the sedimentation ponds and channelized portions of Cedar Swamp Brook and Quarry Stream are not a significant concern, therefore, these evaluations were limited to surface water in the Upper and Lower Simmons Reservoirs.

Assessment Endpoint - Protection of Fish From Toxic Effects of COPECs

Fish populations within the Upper and Lower Simmons Reservoirs represent a recreational fishery which warrants protection. Contaminants within the water column of the reservoirs have the potential to cause toxic effects to exposed fish. In addition, fish may be exposed to sediment contaminants due to resuspension of sediments and the release of sediment contaminants into the water column.

Two measurement endpoints were used to evaluate this assessment endpoint:

1. Toxicity tests were performed on surface water samples from the Upper and Lower Simmons Reservoirs using the water flea *Ceriodaphnia dubia* as a surrogate species for fish. C. dubia are generally more sensitive to inorganic contaminants compared to fish, and therefore, this represents a conservative assessment.

2. Sediment elutriate toxicity tests were performed on fathead minnows (*Pimephales promelas*) as part and an intensive pre-dredging sediment sampling effort for the Upper Simmons Reservoir in 1993. These tests were intended to evaluate potential toxicity to pelagic organisms due to resuspension of sediments during dredging. However, they can also be used to evaluate exposure due to resuspension of sediments due to wind and wave action in this shallow waterbody.

Assessment Endpoint - Protection of Planktonic and Epiphytic Organisms as a Prey Base for Fish

Fish depend upon phytoplankton, zooplankton, and invertebrates that live on submerged and emergent plants as a prey base. Surface water contaminants may cause toxic effects, which may reduce the density and diversity of these species. In addition, sediment-bourn contaminants may be released during resuspension and cause adverse effects to organisms in the water column. Reductions in the density of these prey organisms may indirectly affect the ability of the water body to support fish.

Three measurement endpoints were used to evaluate this assessment endpoint:

1. The *C. dubia* toxicity tests on surface water samples from the Upper and Lower Simmons Reservoirs were used to evaluate this assessment endpoint (as well as to evaluate toxic effects to fish).
2. During the 1993 work at the Upper Simmons, sediment elutriate tests were also performed using *C. dubia*. As with the fathead minnow elutriate tests, these tests were intended to evaluate potential toxicity of resuspended sediments during dredging; but can also be used to evaluate exposure do to resuspension caused by wind and wave action in this shallow waterbody.
3. A qualitative survey of the plankton community was performed in conjunction with the surface water *C. dubia* toxicity tests.

<u>Assessment Endpoint - Protection of the Benthic Community as a Prey Base for Fish and Wildlife</u>

Sediment COPECs have the potential to cause toxic effects to benthic invertebrates and other organisms that live in, or in close contact with sediments (e.g., bullheads, amphibians) of the Upper and Lower Simmons Reservoirs. Benthic organisms serve as prey base for local populations of fish and wildlife, and a reduction in this prey base could result is indirect adverse effects to species that feed on them. Therefore, the maintenance of benthic species as a prey base for fish and wildlife was chosen as the assessment endpoint to evaluate direct exposure to sediment COPECs.

Three measurement endpoints were used to evaluate the potential for harm to benthic aquatic organisms:

1. Sediment samples from the Upper and Lower Simmons Reservoirs were collected for whole sediment toxicity tests using the amphipod *Hyalella azteca*.
2. A qualitative evaluation was performed of the benthic invertebrate community of the Upper and Lower Simmons Reservoirs.
3. The ratio between Acid Volatile Sulfides and Simultaneously Extracted Metals was measured.

Assessment Endpoint - Protection of Local Wildlife Species from Toxic Effects due to Exposure to Surface Water and Sediment COPECs Through the Food Web

Several COPECs in the waterbodies of the CLF Drainage Area have a high potential for bioaccumulation, including several pesticides (DDD, DDE, DDT, aldrin, alpha-chlordane, delta-BHC, and endosulfan), cadmium and mercury. These COPECs have the potential to accumulate in prey organisms, and cause adverse effects to higher trophic-level organisms that feed within the CLF Drainage Area. Protection of wildlife species that feed within waterbodies of the CLF Drainage Area was chosen as the assessment endpoint to evaluate potential risks due to exposure to surface water and sediment contaminants through the food web.

The measurement endpoint chosen to evaluate this assessment endpoint was an evaluation of the potential for toxic effects to the great blue heron. The great blue heron was chosen as the indicator species because herons and great egrets (which are in the same ecological guild as great blue herons) have been observed feeding within the waterbodies of the CLF Drainage Area, and it represents the top of the food web for shallow water bodies such as the sedimentation ponds, stream channels, and much of the Upper and Lower Simmons Reservoirs. Because of the potential for biomagnification, species at the top of the food web will have the greatest exposure to contaminants which have accumulated in aquatic prey organisms.

This evaluation uses a food web model to estimate the level of exposure due to consumption of prey organisms from the water bodies within the CLF Drainage Area, as well as consumption of water and incidental ingestion of sediment. The estimated exposure dose is then compared to toxic effects information from the literature to evaluate whether their is a significant risk of toxic effects to the heron.

9.15.2.2 Ability of Upland Soils to Support Local Wildlife

Elevated contaminant levels have been detected in surficial soil samples from the wooded area surrounding the active landfill. These contaminants may have been transported from the landfill in the past due to settlement of fugitive dust.

Assessment Endpoint - Protection of Local Wildlife Species from Toxic Effects due to Exposure to Soil COPECs Through the Food Web

Several COPECs in surficial soil samples from around the active portion of the Landfill have a high potential for bioaccumulation, including several pesticides (DDD, DDE, DDT, aldrin, alpha-chlordane, delta-BHC, and endosulfan), cadmium and mercury. These COPECs have the potential to accumulate in prey organisms, and cause adverse effects to higher trophic-level organisms that feed within the OU2 Study Area. Protection of wildlife species that forage within wooded areas around the active portion of the landfill property was chosen as the assessment endpoint to evaluate potential risks due to exposure to surficial soil contaminants through the food web.

Food chain evaluations for two common wildlife species likely to use the wooded area around the active portion of the landfill were used as measurement endpoints to evaluate this assessment endpoint:

1. The potential for toxic effects to the America robin. The America robin was chosen as an indicator species to represent wildlife that feed predominately on earthworms and other invertebrates that live in, or in close association with the soil.
2. The potential for toxic effects to the meadow vole. The meadow vole was chosen as an indicator species to represent wildlife that feed predominately on vegetation within the area surrounding the landfill.

As with the great blue heron, these assessments involved estimating the degree of exposure of receptors to soil contaminants through the food web, and comparing those estimated exposures to toxicological information.

9.20 ANALYSIS

This section presents the methods and results for each of the measurement endpoints introduced above.

9.21 Measurement Endpoint - Surface Water Toxicity Tests on *Ceriodaphnia dubia*

Chronic toxicity tests were run on surface water samples from the Upper and Lower Simmons Reservoirs. Chemical analyses were also run on co-located surface water samples.

9.21.1 Methods

Surface water samples were collected for *C. dubia* toxicity tests on May 27 (from the Upper Simmons Reservoir) and May 28 (from the Lower Simmons Reservoir), 1998. Co-located samples were also collected for chemical analyses. Samples from the Upper Simmons were designated SW98-50, -51, and -52; samples from the Lower Simmons were designated SW98-53 and -54. Sample locations are shown on Figure 3-1.

Water samples were collected from the mid-water column. Metallic contaminant analyses were performed on filtered samples (i.e., to yield dissolved metals results). All metals analysis samples were collected with a peristaltic pump with an in-line 0.45 micron cartridge filter; a new filter and tubing were used for each sample. Samples for the remaining analyses were collected either with a horizontal Beta bottle sampler, or as a grab sample, depending on water depth. The Beta bottle was decontaminated using standard procedures in between each sample.

The water depth at SW98-50 and -52 was 10 and 7 feet deep, respectively. Water depth at SW98-51 and -53 was approximately 2 feet. Water depth at SW98-54 was approximately 1 foot.

Samples were stored on ice in a cooler for shipment to the laboratories. Toxicity test samples were sent to New England Bioassay, Inc. of Manchester, Connecticut (NEB). Samples for Chemical analyses were sent to Mitken Corp. of Warwick, Rhode Island; chemical analyses were performed using CLP methods. Samples were shipped to these laboratories on the day they were collected.

Toxicity test methods are described in detail in the NEB report which is presented in Appendix G. Chronic (7-day) toxicity tests were performed on the water samples using *C. dubia* as a the test species. Test water consisted of undiluted sample water from the site, with ten replicate test chambers per sample; the samples collected on May 27 or 28, 1998 were used throughout the test period. Two laboratory control tests were run; one for the samples collected on May 27, 1998 from the Upper Simmons Reservoir, and the other for the samples collected May 28, 1998 from the Lower Simmons. Laboratory controls were run with laboratory fresh water prepared in accordance with EPA guidelines. Tests were run as static-renewal tests, with renewals occurring every 24 hours. Endpoints measured during the toxicity test were survival and reproduction.

9.21.2 Results

Toxicity Tests

The NEB report attached in Appendix G presents detailed results of the chronic *C. dubia* toxicity tests, including survival, reproduction, and water quality measurements. Survival of *C. dubia* was 100 percent in all five surface water samples collected from the Upper and Lower Simmons Reservoirs. *C. dubia* reproduction in the sample water from the Upper and Lower Simmons Reservoirs was not statistically different from the laboratory control water.

Chemical Analyses

Toxicity tests were warranted based on data collected during sampling conducted in 1995 and 1996. Chemical analyses were performed on surface water samples co-located with the toxicity test samples to demonstrate that toxicity test samples contained COPECs at concentrations comparable to the earlier results. Tables 9-23 and 9-24 compare the chemical data from the toxicity test samples to data from the earlier rounds of sampling from the Upper Simmons Reservoir, and the Lower Simmons Reservoir, respectively. Comparisons are based on the frequency of detection, the maximum detected concentration, and the average concentrations.

As can be seen from Table 9-23, with few exceptions, the maximum and average concentrations of dissolved metals and ammonia in the Upper Simmons Reservoir toxicity test samples were comparable to, or higher than those detected in the previous samples. The exceptions were the maximum concentrations of iron and manganese, and the maximum and average concentrations of selenium, which were lower than the earlier results, or were not detected in the toxicity test samples. Few VOCs, SVOCs, or PCB/Pesticides were detected in the toxicity test surface water samples. For the most part, however, organic contaminants were also detected at low frequencies and low concentrations in the earlier samples. Since most organics detected in the previous samples were found in just one of ten samples, perceived differences between the two sampling rounds are attributable to the difference in sample sizes. Therefore, the small number of organic contaminants detected in the toxicity test samples is not a significant consideration in terms of their representativeness of conditions in the Upper Simmons.

Results were similar for the Lower Simmons Reservoir. As can be seen in Table 9-24, three PCB/Pesticides were detected in one of the toxicity test samples, while no PCB/Pesticides were detected in the earlier sample set. Eight dissolved metals which were detected at low frequencies (mostly in one of five samples) in the earlier sample set were not detected in the two toxicity test samples. Otherwise, concentrations of dissolved metals and ammonia in the toxicity test samples were comparable to those detected in the earlier sample set.

9.22 Measurement Endpoint - Qualitative Survey of Plankton Community in Upper and Lower Simmons Reservoirs

Plankton were sampled with a 35-micron plankton net on May 20, 1998. Separate plankton tows were performed in the delta area of the Upper Simmons, the North Basin of USR, the main body (or South Basin) of the Upper Simmons, and in the North End of the Lower Simmons Reservoir. Plankton were preserved in 70 percent ethanol, and returned to GZA in Newton Massachusetts for identification by a GZA limnologist.

Results of the qualitative plankton survey are presented in Table 9-25. Twenty-four taxa were identified in total. Taxa richness at each station steadily increased with distance from the inflow of Cedar Swamp Brook into the Upper Simmons, ranging from 11 taxa at the Upper Simmons delta area, to 18 taxa in the north end of the Lower Simmons Reservoir. It is the opinion of GZA's limnologist that this assemblage represents a reasonably diverse community for the reservoirs.

GZA

9.23 Measurement Endpoint - Sediment Elutriate Toxicity Test on *Ceriodaphnia dubia*

During an intensive sampling effort in 1993, performed to evaluate potential risks from the subsequent dredging of landfill-derived sediments, three landfill derived sediment samples were submitted to NEB for acute sediment elutriate toxicity test using *C. dubia*. These tests were used to evaluate the potential toxicity to pelagic organisms due to resuspension of sediments into the water column of the reservoirs.

9.23.1 Methods

Samples for the elutriate toxicity tests were co-located with chemical analyses for sediment samples SED93-21-I, 24-I, and 30-I (Note, data from SED93-24-I and 30-I were not included in the EPCs for the contaminant screening because sediments represented by these samples were dredged in 1996. Samples were collected using a petite ponar dredge.

A detailed description of the elutriate test methodology is presented in the NEB report, which is included as Appendix H. The sediment elutriate was prepared by mixing the sample with laboratory control water in a 1:4 ratio by volume, turning the mixture overnight, then removing the sediment from the mixture via settling and filtering. *C. dubia* was then subjected to different dilutions of the elutriate during an acute (48-hour) exposure period to gage its toxicity. Additionally, to gage the effect of suspended particulates which passed through the filter, a second test was run using elutriate that had been centrifuged to remove most suspended particulates.

9.23.2 Results

Bioassay results indicated that suspended particulates that passed through the filter and into the sediment elutriate were significantly toxic to *C. dubia*. Forty-eight hour lethal concentrations for 50 percent of the population (48-hour LC50) for samples SED93-21-I, -24-I, and -30-I were 40.2, 58.9, and greater than 100 percent, respectively. Acute No Observed Effects levels (NOELs) of 12.5, 12.5, and 6.25 were determined for these sediment samples, respectively. However, based on additional tests with centrifuged elutriate, it appears that most of the toxicity of the filtered elutriate was caused by suspended particulates rather than chemical contaminants.

Initially, elutriate preparations were filtered through a 1 μm filter; however, after filtering the elutriate preparations still contained between 14.5 and 32.5 parts per thousand of suspended solids. *C. dubia* are filter feeders which are very sensitive to suspended particulates. To gage whether elutriate toxicity may have been caused by suspended solids, NEB ran a second set of 100 percent elutriate solution tests using elutriate that had been centrifuged to remove most of the suspended material. Removal of suspended solids increased survivability of *C. dubia* significantly in the 100 percent solution for SED93-21-I, and –30-I (survival increased from 7 to 83 percent, and 70 to 100 percent, respectively, and slightly increased survivability in the SED93-21-I sample (survival increased from 33 to 40 percent). Because suspended solids were not measured in the centrifuged elutriate solutions, it is possible the significant levels of suspended solids remained in the SED93-21-I sample.

Because of the confounding factor of the high suspended solids remaining in the sediment elutriate, these tests are inconclusive. However, the results of the screening tests with the centrifuged elutriate preparation suggests that the toxicity was caused by the suspended solids, not chemical contaminants.

9.24 Measurement Endpoint - Sediment Elutriate Toxicity Test on Fathead Minnows (*Pimephales promelas*)

Samples collected in 1993 for elutriate toxicity testing with *C. dubia* were also tested for acute toxicity to fathead minnows.

9.24.1 Methods

Section 9.23.1 above describes sample collection methods.

Samples were sent to NEB for acute, 96-hour toxicity tests using fathead minnows. The preparation of the elutriate and elutriate dilutions were the same as for the acute *C. dubia* tests (Section 9.23.1).

9.24.2 Results

No significant acute toxicity to fathead minnows was observed for elutriates from samples SED93-21-I, or -24-I; therefore, the NOEL for these samples was 100 percent elutriate solution. For SED93-30-I, survival in the 100 percent elutriate solution was 53 percent; survival in the 50 percent solution was 97 percent. These survival levels resulted in a 96-hour LC50 of greater than 100 percent, and a NOEL of 50 percent elutriate solution.

9.25 Measurement Endpoint - Whole Sediment Toxicity Test on *Hyalella azteca*

Chronic toxicity tests were run on sediment samples from the Upper and Lower Simmons Reservoirs. Chemical analyses were also run on co-located sediment samples.

9.25.1 Methods

Sediment samples were collected for *H. azteca* toxicity tests on May 27 (from the Upper Simmons Reservoir) and May 28 (from the Lower Simmons Reservoir), 1998. Co-located samples were also collected for chemical analyses. Samples from the Upper Simmons were designated SED98-50, -51, and -52; samples from the Lower Simmons were designated SED98-53 and -54. Sample locations are shown on Figure 3-1.

Sediment samples were collected with a petite ponar grab sampler. This apparatus collects sediment from the surface to a depth of about 4 to 5 inches. Sediments collected at each location are described as follows:

SED98-50 - (from the north end of the main body of the Upper Simmons) consisted of very loose, black organic sediment, overlain by a "dusting" (estimated to be 1 to 2 mm thick) of gray silt, with clods of organic peat in the bottom of the ponar grab. The gray dusting was presumably landfill-derived sediments although other significant sources of sedimentation for this area have been observed (note, landfill-derived sediment in this area prior to dredging was about 2 feet thick). The remainder of the sample was naturally developed organic sediment.

SED98-51 - (from the mouth of the Upper Simmons Reservoir Delta Area) consisted of layers or veins of gray and black silt. This material was residual or redeposited landfill-derived sediment. At least one ponar grab sample retrieved had dark brown, mucky peat in the bottom of the dredge, indicating that there was about 3 to 5 inches of landfill-derived sediment overlying the naturally developed sediments. Note that prior to dredging, landfill-derived sediment in this area was about 2 feet thick.

SED98-52 - (from the Upper Simmons Reservoir delta area) consisted of coarse sand, intermingled with layers of gray silts, overlain by a layer of red and gray silty flocculant. This material is residual or redeposited landfill-derived sediments; the predominance of sand is due to the location in a relatively high energy area due to the inflow of Cedar Swamp Brook.

SED98-53 - (From the Lower Simmons, approximately 900 feet south of the Upper Simmons dam) consisted of very soft, brown to dark brown organic muck.

SED98-54 - (from the Lower Simmons, approximately 300 feet south of the Upper Simmons dam) collected in an area dominated by a carpet of aquatic, submerged grass with slow flowing water. The grass was underlain by silty, medium to coarse sand. The sample collected was soft gray and black silty sediment which was embedded among the stems and in the root mat of the aquatic grass. The thickness of the silty sediment ranged from about 2 to 6 inches, overlying sandy material.

Samples were stored on ice in a cooler for shipment to the laboratories. Toxicity test samples were sent to New England Bioassay, Inc. of Manchester, Connecticut (NEB). Samples for chemical analyses were sent to Mitken Corp. of Warwick, Rhode Island; chemical analyses were performed using CLP methods. Samples were shipped to laboratories on the day they were collected.

Toxicity test methods are described in detail in the NEB report which is presented as Appendix G. Chronic (14-day), whole sediment toxicity tests were performed on each of the sediment samples using the amphipod *Hyalella azteca* as a the test species. Note that the standard exposure period for the whole sediment toxicity tests performed is 10-days, however, the exposure period was increased to 14-days at the request of EPA. One laboratory control was run, which consisted of artificial sediment prepared in accordance with EPA guidance. Water overlying the sediment in the test chambers consisted on laboratory fresh water prepared in accordance with EPA guidance. Overlying water was replaced using a flow-through system, which replaced the water at a rate of about 2 volumes per day. Overlying water was aerated by gentle bubbling to prevent anaerobic conditions in the water. Endpoints evaluated were survival and growth in weight.

9.25.2 Results

Toxicity Tests

Amphipod survival in the samples from the Upper and Lower Simmons Reservoir ranged from 86 to 94 percent, which is well above the EPA control acceptability criterion of 80 percent. At the end of the 14-day exposures, average amphipods growth in the site sediment test chambers ranged 0.252 to 0.303 mg dry weight per individual, which was an increase of about 3.3 to 4.0 times compared to the average initial weight of 0.076 mg per individual. Survival in the control sediments was poor at only 35 percent. Surviving amphipods in the control tests had low weight gain (weighing about one-half the weight of the amphipods in the OU2 sediment sample runs). This indicates that the amphipods in the control tests may have starved because of a lack of natural food sources in the artificial sediment, the flow through conditions (which would reduce the potential for development of a microbial community to act as a food source), and the longer, 14-day exposure period, which was requested by EPA. Because of the poor control performance, the site sample results were compared to each other to evaluate whether there were any statistically significant differences.

Amphipod survival in all the sediments samples from the Upper and Lower Simmons Reservoirs exceeded the EPA criteria of 80 percent for acceptable the control samples (i.e., "clean sediment"), and there were no significant differences between the site samples. Growth of amphipods in the samples from the Upper and Lower Simmons Reservoirs was high, with amphipods increasing in weight by an average of 3.3 to 4.0 times among the different samples. It is clear from these data that sediments from the Upper and Lower Simmons Reservoir did not cause toxic responses in the test organisms.

Chemical Analyses

Most of the landfill-derived sediment samples in the Upper Simmons Reservoir were dredged in 1996; however, residual landfill-derived sediments apparently remained in the Upper Simmons after dredging, and there is the possibility of ongoing sedimentation from the landfill or other area sources such as New England Ecological Development (NEED). Previous sampling efforts indicated that the landfill-derived sediment samples consistently contained higher contaminant concentrations compared to the underlying naturally derived "original" sediments. Based on the timing of the dredging and the sampling, and the possibility of ongoing input of landfill-derived sediments, there was a concern that the toxicity test samples would be chemically "cleaner" than the landfill-derived sediments, and therefore would not be representative of worse case conditions. However, comparing the toxicity test samples to data for the previously collected samples showed that the toxicity test samples contained contaminant concentrations comparable to, or higher than the previous sample results.

Table 9-26 summarizes chemical data for the toxicity test samples from the Upper Simmons Reservoir, and compares those data to summary statistics from all previous samples collected from the Upper Simmons (including landfill-derived sediment samples from the main body of the reservoir). With few exceptions, average and maximum concentrations of contaminants detected in the toxicity test samples are comparable to, (and usually higher than) the corresponding statistic for all previous samples. As described above, toxicity test samples SED98-51 and -52 were comprised mainly, or entirely of landfill-derived sediment, and this is reflected in the chemical data.

The few exceptions where previous chemical data were higher than toxicity test data included VOCs, phthalate esters, mercury, selenium and vanadium. These contaminants were generally detected at low frequencies in the previous samples, and often the higher summary statistics were driven by one very high concentration. Given the variability inherent in sampling and analyses of sediments, and the large differences in sample sizes between the two data sets, the three toxicity test samples were considered adequately representative of "worse case" conditions in the Upper Simmons Reservoir.

Table 9-27 presents a summary of chemical data for Lower Simmons Reservoir toxicity tests, and compares the summary statistics to data from all previous samples collected from the Lower Simmons. With few exceptions, maximum and average concentrations detected in the toxicity test sample set are comparable to, and in many cases higher than, concentrations detected previously in sediment samples. Therefore, the toxicity test samples were representative of Lower Simmons sediment samples with the highest observed contaminant concentrations.

9.26 Measurement Endpoint - Qualitative Survey of Benthic Community

Sediment samples at toxicity test locations SED98-51 (Upper Simmons Reservoir at the Cedar Swamp Brook delta) and SED98-54 (Lower Simmons Reservoir) were screened through a 500 micron sieve to look for benthic macroinvertebrates. Invertebrates observed in the vicinity of SED98-51 were limited to tubificid worms, which were relatively abundant, and small blood-red chironamids (*Chironimus* sp.) which were sparse. The vicinity of sediment sample SED98-54 supported a dense carpet of submerged aquatic grass, with dark gray silty sediments embedded among the grass stems (this material was sampled for the toxicity test). The grass and silty sediment supported an abundant population of *Hyalella azteca* (perhaps ten to twenty individuals were recovered in each dredge).

As expected, the benthic infauna of these areas was depauperate. This was likely due physical disturbance caused by siltation and, in the Upper Simmons, recent dredging.

9.27 Measurement Endpoint - Food Web Evaluation for Great Blue Heron

A food web assessment was performed for the great blue heron to evaluate whether contaminants that may accumulated in aquatic organisms within the CLF Drainage Area may cause toxic effects to higher trophic-level species. This section presents a summary of the methods and results of the food web model for the great blue heron. Appendix I presents a detailed report for this assessment.

9.27.1 Methods

This food web assessment considers exposure to sediment and surface water contaminants within Upper and Lower Simmons Reservoirs, and the stream channels and sedimentation ponds within the active portion of the landfill property. The risk estimate for wildlife exposure is based on assessment of risk to the great blue heron (*Ardea herodias*), a largely piscivorous wading bird known to utilize these exposure points as foraging habitat. An exposure model which incorporates the feeding and foraging habits of the heron was used to estimate the heron's exposure to contaminants in sediment, surface water and in representative prey organisms. Concentrations of organic contaminants in prey organisms were estimated using a widely accepted model (Gobas, 1993) which predicts the bioaccumulation of organic contaminants through an aquatic food-web. Assumptions used in this model were intended to conservatively represent the trophic relationships of aquatic species within the CLF Drainage Area. Inorganic contaminant concentrations in the prey of the heron were estimated based on bioconcentration or bioaccumulation factors recently published in regulatory and scientific literature. The exposure levels (doses) calculated for the heron were compared to toxicological reference doses obtained from current literature to assess the potential for adverse health effects.

Exposure assumptions used to estimate daily doses of contaminants to the heron were conservative. For example, we assumed that the heron would be exposed to site contaminants year round, although herons are likely to be present for only about seven or eight months each year; we assumed that incidentally ingested sediment equals 3.9 percent of the herons diet, which is likely to be high. The exposure model was run several different times, with each run corresponding to a different exposure point within the CLF Drainage Area (i.e., Sed Pond 4, Sed Ponds 2&3 and Channels, Upper Simmons Reservoir, Lower Simmons Reservoir, and the North Basin of the Upper Simmons Reservoir). We also evaluated the entire CLF Drainage Area as one exposure point. For each run of the model, the exposure area being evaluated (i.e., the individual exposure points, or the entire CLF Drainage Area) was assumed to comprise 100 percent of the heron's foraging area. In the case of the evaluation of the entire CLF Drainage Area, EPCs from each separate exposure area were weighted according to the size of the available habitat within the exposure area, and estimated doses from each area were summed to get a weighted dose from the entire CLF Drainage Area.

Because contaminant levels in the landfill-derived sediments are generally higher than in the naturally derived "original" sediments, and landfill derived sediments are still present in the North Basin of USR, we evaluated the data set from the North Basin of USR as a separate exposure point, as well as part of the entire Upper Simmons Reservoir exposure point.

Within each of the different exposure points identified within the CLF Drainage Area, average concentrations within surface water and sediment were used as the basis for estimating the exposure of herons to site contaminants. Maximum concentrations were not used to estimate heron exposure to COPECs because this would have produced an overly conservative assessment. The herons feeding range (approximately 0.129 to 0.98 kilometers of shoreline) is large relative to the habitat size provided by the different exposure points. Also, a fairly large numbers of surface water and sediment samples (27 surface water sampling locations, and 36 sediment samples) were used to represent the CLF Drainage Area. Therefore, it is highly unlikely that any individual heron would be exposed exclusively to an area with concentrations of COPECs comparable to the maximum concentrations. In addition, because the CLF Drainage Area was broken up into several smaller areas, if there were significant areas with COPEC concentrations consistently higher than the rest of the CLF Drainage Area, these conditions would be adequately represented by the average concentration for that exposure point.

However, in order to get an idea of the magnitude of difference between risk estimates calculated using the averages, and risk estimates using the maximum COPEC concentrations, the food web model for the Upper Simmons Reservoir was rerun using the maximum concentrations. The results of the average-based and maximum-based risk estimates for the Upper Simmons Reservoir are compared below.

Estimated doses of each contaminant were compared to toxicity values (referred to as reference doses, or RfDs) from the literature to evaluate whether herons within the CLF Drainage Area may be being exposed to levels that have a significant potential to cause adverse effects. Depending upon the availability of the different RfDs, estimated doses were compared to Lowest Observed Adverse Effects Levels (LOAELs), and/or No Observed Adverse Effects Levels (NOAELs). Comparisons for the individual contaminants were expressed as Toxicity Quotients (TQs), which are the estimated dose divided by the LOAEL or the NOAEL. In addition, Hazard Quotients (HQs) were calculated by summing the TQs within each exposure area in order to gage the potential total risk to the heron.

9.27.2 Results

In evaluating exceedances of RfDs, emphasis was given to exceedances of the LOAELs rather than NOAELs. LOAELs are doses which have been shown to cause adverse effects to test organisms, whereas NOAELs did not result in any adverse effect, and there is a significant probability that a higher dose would be required to cause a significant adverse effect to test organisms. Therefore, an exceedance of a NOAEL indicates an exceedance of a "safe" dose, but does not indicate whether a toxic dose has been reached or exceeded. This concept is discussed further in Section 4.00 of Appendix J.

Table 9-28 summarizes exceedances of reference doses by exposure area for the great blue heron. Considering the CLF Drainage Area as a whole, only DDT and the sum of its degradation products (DDTR) and thallium resulted in estimated doses above their respective LOAELs. LOAEL TQs for these COPECS in the CLF Drainage Area were relatively low, ranging from 3.4 to 7.4. When the different portions of the CLF Drainage Area are considered as individual exposure areas; none of the COPEC in the Upper Simmons Reservoir or the North Basin of USR exceeded their LOAELs. Comparing the different exposure areas, the Lower Simmons Reservoir had the the greatest number of persistent COPECs which exceeded LOAELs (DDT, DDTR, and thallium), and the highest LOAEL TQs (ranging from 6.5 to 12.8). When the Sedimentation Ponds 2 & 3 and Channels exposure area, and Sedimentation Pond 4 are considered as separate exposure areas, four additional COPECs exceed their LOAELs (mercury, benzo(a)pyrene and benzo(a)anthracene in Sedimentation Ponds 2 & 3 and Channels, and mercury and butylbenzylphthalate in Sedimentation Pond 4), with LOAEL TQs ranging from 1.3 to 4.0 .

The exceedances of LOAELs by DDT and DDTR in the Lower Simmons were driven by the detection of DDT in one of the surface water samples collected for the toxicity tests. This detection also drove exceedances of the LOAELs by DDT and DDTR in the entire CLF Drainage Area.

The estimated doses of butylbenzylphthalate exceeded its LOAEL only in Sedimentation Pond 4, with a LOAEL TQ of just 1.4. The estimated doses of benzo(a)anthracene and benzo(a)pyrene exceeded their LOAELs only in Sedimentation Ponds 2&3 and Stream Channels with LOAEL TQs of just 1.3 and 3.1. It should be noted that concentrations of benzo(a)anthracene and benzo(a)pyrene in the Sedimentation Ponds 2&3 and Stream Channels exposure area (which were detected only in sediment, not surface water, throughout the CLF Drainage Area) were less than concentrations detected in the Upper Simmons Reservoir, although benzo(a)anthracene and benzo(a)pyrene did not exceed their reference doses in the Upper Simmons. Potential risks to the heron indicated in the Sedimentation Ponds 2&3 and Stream Channels exposure area were driven by the relatively low TOC content of sediments in that area (e.g. an average of 1.5 percent in the sedimentation ponds versus 11 percent in the Upper Simmons). Organic contaminants are more available to be taken up by the biota in low TOC systems compared to higher TOC systems, and this was reflected in the results of the Gobas model.

Mercury exceeded its LOAEL only in Sedimentation Ponds 2 & 3 and Channels. Note that total mercury was detected in the surface water of the sedimentation ponds, and that dissolved mercury, which is more bioavailable than total mercury, was not detected in any of the exposure areas. The total mercury results in surface water were used to estimate body burdens in prey organisms. These detections drove the exceedances of LOAELs in the sedimentation ponds.

LOAEL-based total Hazard Quotients (HQs), which are the sum of TQs for each COPEC, where greater than 1 in all exposure areas except the Upper Simmons Reservoir; the North Basin of USR had a LOAEL-based total HQ of just 1.6, although none of the estimated doses for individual COPECs exceeded their LOAEL. Aside from the Upper Simmons Reservoir and the North Basin of USR, LOAEL-based total HQs ranged from 1.5 to 19.8, with the highest LOAEL-based TQ being in the Lower Simmons Reservoir.

Aside from DDT, DDTR, benzo(a)anthracene, benzo(a)pyrene, butylbenzylphthalate, mercury, and thallium, (which had exceedances of LOAELs and, therefore, exceeded NOAELs in one or more exposure area), NOAELs were also exceeded by DDE, and beryllium in one or more exposure areas. Exceedances by these two COPECs were very small, with NOAEL TQs ranging from 1.2 to 1.6. With the exception of the Upper Simmons Reservoir, the North Basin, and Sedimentation Pond 4 (which had NOAEL-based total HQs of 8.6, 16.4, and 14.5, respectively), NOAEL-based total HQs were high, ranging from 90 to 193. However, these high NOAEL-based HQs were all driven by thallium or mercury. Due to conservative uncertainties in the modeling of potential risks through the food web, and because the detected concentrations of these contaminants are not likely to have resulted primarily from the OU1 Landfill, thallium and mercury are not considered to pose significant OU1 Landfill related risks. These uncertainties are discussed in detail in Section 5.00 of Appendix I.

To gauge the magnitude of difference the use of maximum sediment and surface water concentrations would make to the risk estimates for the heron, the heron food web model for the Upper Simmons Reservoir was re-run using maximum concentrations. In general, maximum-based TQs for individual COPECs were on the order of 0.5 (or 50%) to 4 times higher than the average-based TQs; the LOAEL-based and NOAEL-based total HIs calculated using maximums were about 2 times greater than those based on average COPEC concentrations. Use of maximum concentrations did not result in LOAEL exceedances by any of the individual COPEC, however the LOAEL-based total HI increased from 0.7 based on average concentrations to 1.8 based on maximum concentrations. Use of the maximum concentrations resulted in additional exceedances of the NOAEL by individual contaminants; these exceedances were by DDT (and DDTR), and thallium, with NOAEL-based TQs of 1.1 (2.0), and 5.8, respectively. The NOAEL-based total HQ increased from 8.6 based on average concentrations, to 21 based on maximum concentrations.

9.28 Measurement Endpoint - Food Web Evaluation for American Robin

A food web assessment was performed for the American robin to evaluate whether contaminants that may accumulated in soil invertebrates and plant tissues within the wooded areas surrounding the landfill may cause toxic effects to higher trophic-level species. This section presents and summary of the methods and results of the food web model for the American robin. Appendix J presents a detailed report for this assessment.

9.28.1 Methods

For this food web assessment, we estimated the exposure of the American robin to surficial soil contaminants which have accumulated in prey species of the robin, and from incidental ingestion of contaminated soil. These estimated exposure doses are then compared to toxicological reference values from the literature to gage whether there may be a significant risk of harm to the robin.

Chemical data used in the assessment were from 15 surficial soil samples collected from the wooded areas around the active portion of the landfill property. The spacing between these samples was large relative to the potential foraging area of the American robin, raising the possibility that some individuals within the local population may be being exposed to concentrations that are higher than the average concentrations. Therefore, we did this assessment based on both the average concentrations in surficial soils, and the maximum detected concentrations.

The EPA's Wildlife Exposure Factors Handbook (EPA 1993b) presents three daily ingestion rates (0.75, 0.89, and 1.52 kg/kg/day) for the American robin. All three of these ingestion rates were estimate during periods in which the robins consumed only fruits. Robin ingestion rates vary with the energy content in food, thus they consume less food when consuming high-energy prey compared to periods when they are primarily consuming low energy food. For this risk characterization we assumed that 40 percent of the robin's diet consists of invertebrates, which generally have a significantly higher energy content than do fruits. Therefore, it was considered reasonably conservative to use the average concentration of daily food ingestion from these studies, rather than the maximum ingestion rate.

A bioaccumulation model comparable to the Gobas model used for the aquatic food web in the great blue heron assessment (see Appendix I), is not available for terrestrial systems. Therefore, with the exception of the VOCs and pesticides, we estimated body burden concentrations in soil invertebrates by multiplying the average- or maximum-based EPCs by a bioconcentration factor (BCF). Uptake of VOCs and pesticides by earthworms was estimated using the mechanistic model of Jager (1998).

As with the heron food web, we compared estimated doses to LOAELs and NOAELs when available. To facilitate comparisons between the estimated doses and the NOAELs and LOAELs we calculated TQs (see Section 9.27.1) and total HQs.

9.28.2 Results

In evaluating exceedances of RfDs, emphasis was given to exceedances of the LOAELs, because LOAELs are doses which have been shown to cause adverse effects to test organisms, whereas NOAELs did not result in any adverse effect, and there is a significant probability that a higher dose would be required to cause a significant adverse effect to test organisms. This concept is discussed further in Section 4.00 of Appendix J.

Table 9-29 presents a summary of contaminants which resulted in estimated doses greater than their respective LOAELs and NOAELs. None of the individual contaminants resulted in doses that exceeded a LOAEL based on average concentrations in soil. The sum of average-based LOEL TQs resulted in a total LOAEL HQ of 4.4. When estimated exposures are based on maximum detected concentrations in soil, DDT, DDTR, and two metals (lead and zinc) result in LOAEL-TQs greater than 1, however, with the exception of zinc (which had a LOAEL TQ of 24), these LOAEL TQs are small (i.e., 2.2 or less). As discussed in detail in Section 5.20 of Appendix J, these results for zinc were driven by one very high, anomalous detection of zinc in soil (the anomalous maximum was 6,702 mg/kg versus a penultimate concentration of 178 mg/kg). The total LOAEL HQ based on maximum concentrations was 30.8.

Exposure estimates based on average concentrations did not result in exceedances of LOAELs, but did result in exceedances of the NOAEL by DDT, DDTR, chromium, lead and zinc, with NOAEL TQs ranging from 1.8 to 19.3. Aside from those contaminants that exceeded LOAELs based on the maximum detected concentrations (as discussed above), exposure estimates based on maximum detected concentrations also resulted in the exceedance of the NOAEL by bis(2-ethylhexl)phthalate and DDE both with NOAEL TQs of 1.9, and chromium with a NOAEL TQ of 3.5. The average based, total NOAEL HQ was 39.3; the maximum based total NOAEL HQ was 279. The high values for both the average-based and maximum-based total NOAEL HQs were driven by zinc, which has an average-based NOAEL TQ of 19.3, and a maximum-based NOAEL TQ of 220. As discussed above and in Section 5.20 of Appendix J, results for zinc were driven by one anomalously high detection in soil.

9.29 Measurement Endpoint - Food Web Evaluation for Meadow Vole

A food web assessment was performed for the meadow vole to evaluate the potential for risks to higher trophic level organisms, which have a diet dominated by plant material. This section presents a summary of the methods and results of the food web model for the meadow vole. Appendix J presents a detailed report for this assessment.

GZA

9.29.1 Methods

For this food web model, we estimated the exposure of meadow voles due to ingestion of plants and soil invertebrates which have accumulated COPECs from the soil surrounding the landfill, and from incidental ingestion of soil. These estimated exposure doses are then compared to toxicological reference doses from the literature to gage whether there may be a significant risk of harm to the vole.

The soil data used, and the methods in which the data were used were identical to methods used for the robin food web model (Section 9.28 above).

The EPA Wildlife Exposure Factors Handbook (EPA, 1993b) presents daily food ingestion data (on a weight to weight basis) from one study. This study found that daily ingestion ranged from 0.30 to 0.35 kg/kg/day. Because of the narrowness of this range, and the relatively uniform diet of the meadow vole, we used the mid-point of this range (i.e., 0.325 kg/kg/day) to represent the total daily food ingestion rate.

9.29.2 Results

In evaluating exceedances of RfDs, emphasis was given to exceedances of the LOAELs, because LOAELs are doses which have been shown to cause adverse effects to test organisms, whereas NOAELs did not result in any adverse effect, and there is a significant probability that a higher dose would be required to cause a significant adverse effect to test organisms. This concept is discussed further in Section 4.00 of Appendix J.

Table 9-30 presents a summary of contaminants which resulted in estimated doses greater than their respective LOAELs and NOAELs. None of the COPECs in soil resulted in dose estimates above the LOAEL; this is true whether dose estimates are based on average or maximum concentrations. In addition, none of the dose estimates based on average soil concentrations exceed the NOAEL. Vanadium exceeded its NOAEL based on maximum concentrations with a NOAEL TQ of 1.01.

NOAEL-based total HQs were 0.88 and 2.2 based on average and maximum detected soil concentrations, respectively.

9.210 Measurement Endpoint - Food Web Evaluation for Short-tailed Shrew

A food web assessment was performed for the short-tailed shrew to evaluate the potential for risks to higher trophic level organisms which feed primarily on soil invertebrates which have taken up soil contaminants. This section presents a summary of the methods and results of the food web model for the shrew. Appendix J presents a detailed report for this assessment.

9.210.1 Methods

For this food web model, we estimated the exposure of short-tailed shrews due to ingestion of soil invertebrates which have accumulated COPECs from the soil surrounding the landfill, and from incidental ingestion of soil. These estimated exposure doses are then compared to toxicological reference doses from the literature to gage whether there may be a significant risk of harm to the shrew.

The soil data used, and the methods in which the data were used were identical to methods used for the robin food web model (Section 9.28 above).

The maximum food ingestion rate cited in EPA, 1993a was used for the shrew in this food web assessment. This was done because the shrew was intended to represent a terrestrial receptor with "high-end" exposure potential, and because the diet of the shrew is relatively uniform, consisting predominately of invertebrates and other animal prey organisms, with relatively little lower energy plant material.

9.210.2 Results

In evaluating exceedances of RfDs, emphasis was given to exceedances of the LOAELs, because LOAELs are doses which have been shown to cause adverse effects to test organisms, whereas NOAELs did not result in any adverse effect, and there is a significant probability that a higher dose would be required to cause a significant adverse effect to test organisms. This concept is discussed further in Section 4.00 of Appendix J.

Table 9-31 presents a summary of contaminants which resulted in estimated doses greater than their respective LOAELs and NOAELs. Zinc was the only COPEC which resulted in an estimated dose above the LOAEL; this is true for both average- and maximum-based dose estimates. The average-based LOAEL TQ was 1.1, and the maximum based LOAEL TQ was 12.6.

Aside from zinc, NOAELs were exceeded by average-based doses of lead and vanadium, and maximum-based doses of chromium, lead, selenium and vanadium. Average-based NOAEL TQs for these COPECS ranged from 1.5 to 5.4, and maximum-based NOAEL TQs ranged from 1.1 to 9.

LOAEL-based total HQs were 2.3 and 15 for average- and maximum-based dose estimates, respectively. NOAEL-based total HQs were 10.5 and 41 based on average and maximum detected soil concentrations, respectively. The high NOAEL-based total HQ was driven by a anomolously high detection of zinc, as discussed earlier for the robin, and in detail in Section 5.20 of Appendix J.

9.220 Acid Volatile Sulfide and Simultaneously Extracted Metals

9.220.1 Methods

A subset of sediment samples from each of the exposure points were analyzed for acid volatile sulfides (AVS) and simultaneously extracted metals (SEM). The suit of metals analyzed were cadmium, copper, lead, mercury, nickel, and zinc. These are cationic metals which form sulfides with solubilities lower than iron and magnesium sulfides. Under anaerobic conditions, and in the presence of sulfides, cationic metals form metal sulfides with low solubility and low bioavailability. Since the SEM suite of metals form sulfides of lower solubility than the dominant natural cations (i.e., iron and magnesium) the sulfides will tend to be associated with these metals. The theory behind the use of this technique for risk assessment is that if the molar concentration of AVS exceeds the sum of the molar concentrations of each SEM then these metals are likely to be bound with the sulfides, and not bioavailable.

In order to evaluate whether SEM metals are likely to be bound to sulfides, for each sample, the sum of the molar concentrations of the SEM metals is divided by the AVS concentration. An SEM/AVS ratio of greater than one indicates that the AVS content is not sufficient to bind all of the SEM metals, therefore, a portion of those metals may be bioavailable. If the ratio is 1 or less, it can be concluded that all SEMs are bound to sulfides and that they are not bioavailable and, therefore, they do not pose a risk to benthic organisms which live in, or in association with the sediments. Note, however, that SEM/AVS rations greater than one do not necessarily indicate that a significant portion of the SEMs are bioavailable, since other binding phases (e.g., organic carbon)may act to reduce bioavailability (Ankley et al., 1996).

9.220.2 Results

Data summaries contained in Appendix I (Tables I-1, I-3, I-5, I-7, and I-9) present results of the AVS and SEM analyses performed on sediment samples, as well as the SEM/AVS ratios. Raw data for the AVS and SEM results are presented in Table 6-19. All samples analyzed from Sed Pond 4 and Sed Ponds 2&3 and Channels resulted in SEM/AVS less than 1. Eight of the twelve samples analyzed from the Upper Simmons Reservoir resulted in SEM/AVS ratios of 1 or less. Of the four samples with SEM/AVS ratios greater than 1, three of the four samples had ratios less than 2, (all SEM/AVS in samples from the USR North Basin were less than 1). Of the three samples analyzed from the Lower Simmons Reservoir, only one had detectable levels of AVS; SEM/AVS was below one for that sample. Based on the AVS detection limits reported for the other two samples, SEM/AVS was clearly greater than 1.

With the potential exception of the Lower Simmons Reservoir, SEM/AVS ratios strongly indicate that the suite of metals measure in the SEM procedure (cadmium, copper, mercury, nickel, and zinc) are bound to sulfides, are therefore not bioavailable, and do not pose a significant risk to benthic invertebrates. As discussed by Ankley et al. (1996), an SEM/AVS ratio of less than 1.0 is considered a no effect level, however a ratio greater than 1 does not necessarily indicate that toxicity is occurring. There are many examples where sediment SEM/AVS ratios were greater than 1, yet no toxicity occurred. Considering that most of the samples analyzed has SEM/AVS ratios less than 1, and most of the others were less than 2, these data indicate that these metals are not bioavailable.

9.30 RISK CHARACTERIZATION

The following sections interpret the results of the measurement endpoint analyses in terms of the assessment endpoints identified for the OU2 Study Area exposures.

9.31 Assessment Endpoint - Protection of Fish From Toxic Effects of COPECs

Two measurement endpoints were used to evaluate this assessment endpoint: Chronic toxicity tests on surface water samples using *C. dubia* (Section 9.21), and acute sediment elutriate toxicity using fathead minnows (Section 9.23).

None of the five surface water samples collected from the Upper Simmons Reservoir and Lower Simmons Reservoir were toxic to *C. dubia*. These results indicate that there is not a significant risk of toxic effects to organisms that live in the water column due to ambient water quality conditions.

Two of three elutriate solutions tested for the Upper Simmons Reservoir were not toxic to fathead minnows; the third test resulted in a 96-hour LC50 of greater than 100 percent elutriate solution. These results suggest that mortality of fish due to resuspension of sediments into the water column (e.g., during storm events) is not a significant consideration; mortality, if it were to occur, would likely be limited to a small number of individuals at sensitive life stages.

The two measurement endpoints evaluated strongly indicate that COPECs in ambient surface water, or in resuspended sediments do not cause significant toxicity. Therefore, there is no significant risk of harm to the fish populations of the Upper Simmons and Lower Simmons Reservoirs under current conditions.

9.32 Assessment Endpoint - Protection of Planktonic and Epiphytic Organisms as a Prey Base for Fish

Three measurement endpoints were used to evaluate this assessment endpoint: Chronic toxicity tests on surface water samples from the Upper and Lower Simmons Reservoirs (Section 9.21), a qualitative survey of the plankton community in the Upper and Lower Simmons Reservoirs (Section 9.22), and acute toxicity tests on sediment elutriate samples from the Upper Simmons Reservoir using *C. dubia* (Section 9.24).

None of the five surface water samples collected from the Upper and Lower Simmons caused chronic toxicity to *C. dubia*. These results suggest that ambient water quality conditions in the reservoirs do not present a risk of harm.

Based on observations of the plankton community from a single sampling event, the community of the Upper and Lower Simmons Reservoir was found to be reasonably diverse. Based on this limited assessment, it does not appear that the plankton community has been severely impacted by ambient water quality conditions within the reservoirs.

Results of the acute sediment elutriate toxicity test with *C. dubia* were inconclusive. The standard elutriate preparations (i.e., filtered) for all three samples caused significant mortality to *C. dubia*. However, when the elutriate was centrifuged, only one of the three samples cause significant mortality. These data suggest that suspended particulates were causing the observed toxicity in the filtered samples, and at a minimum, two of the elutriate samples were not acutely toxic to *C. dubia* due to COPECs. COPEC concentrations in the third sample may have been toxic, or there may have still been too much suspended solid in the centrifuged elutriate solution for these filter feeders.

With the possible exception of one of three sediment elutriate tests, all the data collected to evaluate this assessment endpoint indicate that COPECs within surface water and sediment do not pose a significant risk of harm to the plankton community. In the worse case, sediment COPECs may cause toxicity to a small percentage of zooplankton upon resuspension during storm events. However, this situation is not considered a significant risk to the plankton community. (Note that the sediments used in the elutriate toxicity tests have been removed by the 1996 dredging, however, more recent chemical analyses indicate that residual or redeposited landfill-derived sediments have comparable contaminant concentrations.) Therefore, this assessment clearly indicates that there is not a significant risk of harm to the plankton community of the Upper and Lower Simmons Reservoir, and there is no potential for significant indirect impacts on local fish.

9.33 Assessment Endpoint - Protection of the Benthic Community as a Prey Base for Fish and Wildlife

Three measurement endpoints were used to evaluate this assessment endpoint: chronic, whole sediment toxicity tests performed on samples from the Upper Simmons Reservoir and Lower Simmons Reservoir using the amphipod *H. azteca*, AVS and SEM analyses for five metals (cadmium, copper, mercury, nickel, and zinc), and a qualitative survey of benthic invertebrates in the Upper and Lower Simmons Reservoirs.

Sediment samples from the Upper and Lower Simmons Reservoir were not toxic to amphipods, indicating that COPECs do not present a significant risk of harm to the benthic community. Most of the SEM/AVS ratios for sediment samples from the CLF Drainage Area were 1 or less; all but three SEM/AVS ratios were less than 2. These data indicate that, for the great majority of sediments within the CLF Drainage Area, these five cations are not bioavailable, therefore, they do not present a significant risk of harm to benthic organisms. Observations made of the benthic community of the reservoir during one sampling round indicated that the sediment infaunal community is depauperate. This is likely due to the physical disturbance caused by siltation into the reservoirs rather than the COPECs. However, it is important to note that *H. azteca* was found at high densities in northern end of the Lower Simmons Reservoir, confirming the results of the toxicity test.

Based on the whole sediment toxicity test data, and AVS and SEM data, COPECs in sediment of the Upper and Lower Simmons Reservoirs do not pose a significant risk to the benthic community. Therefore, there is not a significant risk of indirect impacts to the fish and wildlife that depend on benthic invertebrates for food.

9.34 Assessment Endpoint - Protection of Local Wildlife Species from Toxic Effects Due to Exposure to Surface Water and Sediment COPECs Through the Food Chain

COPECs within waterbodies of the CLF Drainage Area have the potential to accumulate in exposed biota, and result in exposure of higher trophic-level organisms via consumption of the affected prey species. A food web model was used to estimate the degree of exposure (in terms of daily doses of COPECs) to the great blue herons via foraging in the CLF Drainage Area. Estimated daily doses were compared to toxicological data (LOAELs and NOAELs) from the literature to evaluate the potential for risk of harm to herons.

The food web model effectively ruled out the potential for significant risk to herons due to surface water and sediment COPECs migrating from the OU1 Landfill: none of the COPECs detected within the Upper Simmons Reservoir (which is the habitat area with the greatest potential to be impacted by the OU1 Landfill) resulted in exceedances of LOAEL reference doses. Within the CLF Drainage Area as a whole, exceedances of LOAELs and

NOAELs were limited to DDT (and its degradation products), butylbenzylphthalate, benzo(a) anthracene, benzo(a)pyrene, beryllium, mercury, and thallium. In addition, total HQs exceeded a value of one for NOAELs and LOAELs in the CLF Drainage Area as a whole, and in most of the separate exposure areas. However, the area with the greatest potential for risk to herons, as suggested by the food web model, was the Lower Simmons Reservoir. Among the different CLF Drainage Area exposure areas, the Lower Simmons Reservoir has the lowest potential for being impacted by the OU1 landfill and the highest potential for being impacted by agricultural activities and other waste sites in the area. This observation alone indicates that the OU1 landfill has relatively low potential for ecological risks within receiving water bodies.

Uncertainties related to this food web assessment are presented in Section 9.37 of this text, and Section 5.00 and 6.00 in Appendix I. Based on the relatively low TQs in combination with the distribution of contaminants relative to potential migration pathways from the OU1 Landfill, and conservative uncertainty inherent in the food web assessment, it is our opinion that contaminants which migrated from the OU1 landfill do not pose a significant risk of harm to herons or other piscivorous birds.

Exposure of herons was estimated based on average sediment and surface water concentrations, but not maximum concentrations. As discussed in Section 9.27, the use of maximum concentrations was considered to be overly conservative. However, in order to get an idea as to the magnitude of difference use of maximum concentrations would have, we ran the food web assessment for Upper Simmons using maximum concentrations. This resulted in increases in the TQs and total HQs on the order of 2 times greater than those produced using average concentrations. Therefore, for the Upper Simmons Reservoir data set, the affect of using maximum contaminant concentrations was relatively minor.

9.35 Assessment Endpoint - Protection of Local Wildlife Species from Toxic Effects Due to Exposure to Soil COPECs Through the Food Web

COPECs within soils surrounding the active portion of the landfill have the potential to accumulate in exposed biota, and result in exposure of higher trophic-level organisms via consumption of the affected prey species. A food web model was used to estimate the degree of exposure (in terms of daily doses of COPECs) to the American robin, the meadow vole, and the short-tailed shrew via foraging in wooded areas around the active portion of the landfill property. Estimated daily doses were compared to toxicological data (LOAELs and NOAELs) from the literature to evaluate the potential for risk of harm to birds and small mammals.

None of the estimated doses of soil COPECs to the meadow vole exceeded their LOAELs, either when calculated on the basis of average concentrations or maximum detected concentrations. The only COPEC to exceed a NOAEL was vanadium when doses were calculated from maximum detected concentrations (but not average concentrations);

the estimated dose resulted in a NOAEL TQ of 1.01. Taking the conservative uncertainties into consideration (see Section 9.37.4 and Sections 5.00 and 6.00 of Appendix J), it is our opinion that this food web effectively rules out the potential for significant risk to meadow voles and other small mammals which feed primarily on plant material.

With the exception of zinc, none of the estimated doses of soil COPECs to the American robin or the short-tailed shrew exceeded their LOAELs when calculated on the basis of average concentrations. (The average soil concentration of zinc resulted in an estimated dose to the robin which exceeded the LOAEL with a LOAEL TQ of 1.1.) Several contaminants resulted in estimated doses to the robin and shrew which exceeded LOAELs based on maximum concentrations, and exceeded NOAELs; these included bis(2-ethylhexyl)phthalate, DDE and DDT (and DDTR), chromium, lead, selenium and vanadium. Based on the relatively low TQs in combination with the distribution of contaminants relative to potential migration pathways from the OU1 Landfill, COPEC concentration which are comparable to typical concentrations for soil in the northeast and the U.S., and conservative uncertainty inherent in the food web assessment (see Section 9.37.4 and Sections 5.00 and 6.00 of Appendix J), it is our opinion that contaminants which may have migrated to surficial soils from the OU1 landfill to surrounding wooded areas do not pose a significant risk of harm to robins or shrew, or species with similar feeding habits.

9.36 Potential Future Risks in the Upper Simmons Reservoir

Two metals (barium and thallium) and several pesticides were retained as COPECs for surface water in the Upper Simmons Reservoir under future conditions (note, all estimated pesticide concentrations were well below surface water benchmark concentrations, however, they were retained because they are bioaccumulative). These COPECs present a potential risk to exposed biota due to direct exposure; in addition the pesticides present a concern for potential risks to higher trophic level organisms via the food web. The estimated future conditions could not be directly evaluated, however, results of the measurement endpoints used to evaluate existing conditions can be extrapolated to the future conditions.

Barium, thallium, and four of the pesticides retained as future conditions COPECs for the Upper Simmons Reservoir were detected in surface water samples from the Upper and/or Lower Simmons Reservoirs at concentrations comparable to, or greater than the predicted future concentrations. The existing conditions risk assessment found that there are no significant risks from these contaminants based on toxicity tests and qualitative surveys of the aquatic biota. Therefore, these COPECs do not present a potential risk of harm due to direct exposure under future conditions.

Estimated future concentrations of pesticides in surface water of the Upper Simmons Reservoir were on the order of $1x10^{-6}$ to $1x10^{-7}$ mg/l. When running the Gobas model to estimate body burdens (see Appendix I), the value of $1x10^{-7}$ mg/l is used as a "zero concentration" when the contaminant was not a surface water COPEC. This was done because the model requires and input value greater than zero, however, $1x10^{-7}$ mg/l is low enough that it does not have an effect on the estimated body burdens. This indicates that these levels of pesticides in water are not likely to have an adverse effect through the food web. In addition, this input value is intended to represent dissolved or "free" contaminant levels in surface water; because of the strongly hydrophobic nature of pesticides, it is unlikely that these contaminants would be present in surface water in a "free" state. Therefore, assuming $1x10^{-7}$ mg/l of pesticides in water was a conservative assumption for the bioaccumulation model. Based on these considerations, it is our opinion that there are no significant risks in the future due migration of groundwater contaminants to the Upper Simmons Reservoir.

9.37 Uncertainties

The following sections summarize uncertainties inherent in the data collected and the evaluations performed for this ERA. Whether the uncertainties are likely to contribute to a conservative assessment (i.e., increase the chance of indicating a significant risk when risk is actually low; this is also referred to as a false positive finding), or to false negative findings is also discussed.

9.37.1 Chemical Data

The number of surface water and sediment samples collected within OU2 in support of (and used in) the ERA was relatively high (i.e., approximately 27 surface water sampling locations (with 1 or 2 rounds of samples for each location) and 36 sediment samples were used for the CLF Drainage Area). Considering these sample numbers relative to the size of the exposure areas being evaluated, the chemical data is generally expected to be representative of the range of site conditions. Background samples, on the other hand, were limited to eight surface water sampling locations and seven sediment samples. There is some concern that the full range of background levels were not represented by these data. This would contribute to a conservative assessment since some chemicals may have been included as COPECs when they might have been eliminated based on additional background values.

The discrepancy in the number of OU2 surficial soil samples versus background surficial soil samples was quite large: 26 surficial soils samples were collected within OU2, but only 2 background samples were collected. Again, this low number of background samples may not have been representative of the full range of background levels, and this may have resulted in the retention of some chemicals (based on comparisons of maximum detected concentrations) which might have otherwise been eliminated. For example, OU2 concentrations for lead, chromium, and vanadium (which resulted in slight exceedances of reference doses for terrestrial receptors) were comparable to, or below background levels published in the literature, but were retained because the OU2 maximum was above the site-specific background maximum. Additional background sampling may have allowed these metals to be eliminated as COPECs.

Between the time that the OU2 sampling began, and the ERA was prepared, large scale earthwork was performed (and is on-going), including the relocation of a portion of Cedar Swamp Brook, and the dredging of sedimentation ponds and the Upper Simmons Reservoir. For some areas, this called into question how well the chemical data set represented existing conditions. For the Upper Simmons Reservoir, data from post-dredging sampling (i.e., the chemical data co-located with the 1998 sediment and surface water toxicity tests) was compared to pre-dredging data, and these two data sets were found to be comparable. Therefore, for the Upper Simmons Reservoir, this did not appear to impart a significant degree of uncertainty.

Pre- and post construction or dredging data was not available for Cedar Swamp Brook or the sedimentation ponds. However, based on the results of the evaluation for the Upper Simmons Reservoir, it may be inferred that this work did not significantly affect chemical concentrations in surface water or sediments. If any change occurred, or is likely to occur in the future, it is likely to be toward lower concentrations. This is because the unlined portion of the landfill is being capped, and more recent landfill cells are lined and will be capped upon completion. In that case, this uncertainty would tend toward a conservative assessment.

In many samples, from several of the different exposure areas, dissolved selenium was detected but total selenium was not, though method detection limits were comparable. This produces some uncertainty as to the true presence and state of selenium. The effect of this uncertainty is unknown.

9.37.2 Toxicity Tests

There are a number of sources of uncertainty, which are common to all laboratory toxicity tests when used to evaluate potential risks due to in-place contaminants. A large source of uncertainty comes from extrapolating results for one life stage of a single species to an entire pelagic or benthic community. This extrapolation may impart conservative uncertainty if the test organism is more sensitive to the suit of contaminants than most of the receptor organisms. Conversely, if the test organisms is significantly less sensitive to the suit of contaminants compared to the most sensitive receptor organisms, toxicity test may lead to a false negative conclusion with regard to risk.

Another source of uncertainty stems from the representativeness of toxicity test samples. Often, the number of toxicity tests run is relatively small, and it is usually not feasible to confirm before the toxicity test is initiated that contaminant concentrations in those samples are representative of the site as a whole. For this ERA, it was determined that contaminant concentrations in surface water and sediment toxicity test samples were adequately representative of overall concentrations in the Upper Simmons Reservoir. Therefore, this is an insignificant source of uncertainty for this evaluation.

GZA

For whole sediment toxicity tests, the preparation of the sample (sieving and homogenization) may significantly alter the physiochemical condition, and thus potentially the bioavailability of contaminants in the sample. For example, anaerobic conditions and the formation of metal sulfides has been shown to be an important factor in reducing the bioavailability of cationic metals in sediment. Laboratory prepatory manipulations for toxicity tests are likely to aerate the sediments, which may liberate cationic metals from otherwise non-bioavailable metallic sulfides. This factor likely contributes to conservative uncertainty in whole sediment toxicity tests.

For this assessment, sediment elutriate toxicity test results were used to evaluate potential toxicity due to resuspension of in-place contaminants within sediments of Upper Simmons Reservoir. Sediment elutriate toxicity tests were developed to evaluate ocean disposal of dredge sediments. For most waterbodies and most situations, sediment elutriate toxicity tests may not be appropriate to evaluate toxicity of in-place sediment contaminants. However, Upper Simmons Reservoir is a shallow, relatively open waterbody. This situation promotes resuspension of sediments. In addition, much of the contaminant load enters the reservoir associated with suspended solids. Therefore, the sediment elutriate toxicity test is relatively well suited to the evaluation of the Upper Simmons Reservoir.

However, it is unclear as to whether the inconsistent toxicity observed in certain sediment elutriate toxicity tests was due to chemical contaminants or to physical conditions of the test, namely too high suspended solid concentrations. This uncertainty would contribute to a conservative evaluation, since mortality was observed, but it may not have been attributable to COPECs in the sediment..

9.37.3 Qualitative Biological Surveys

Conceptually, quantitative biological surveys which compare the community of a potentially affected area to the community of an unaffected reference site are very useful for evaluating potential effects of contaminants. However, the CLF Drainage Area was not amenable to such quantitative, comparative surveys. The potentially affected aquatic habitats are significantly physically altered, and it was not possible to identify a reference site with similar habitat conditions in order to isolate the potential effects of OU1-related COPECs.

Thus, a qualitative survey, with results evaluated based on professional judgement was the only option for this site. The main source of uncertainty for the qualitative biological surveys performed for OU2 stems from the complete reliance on professional judgement. However, the main intent of the surveys was simply to evaluate whether reasonably abundant and diverse communities exist in the water column and benthic habitats. For these purposes, this uncertainty was considered to be acceptable.

A relatively low level of effort was performed for these surveys, and under most circumstances, this would impart a significant degree of uncertainty. However, for the habitat being evaluated, and the intent of the surveys, this level of effort was adequate and is not considered to be a significant source of uncertainty.

9.37.4 Food Web Evaluations

There are significant uncertainties surrounding the applicability and adequacy of the RfDs. These include potential differences in the bioavailabilty of biologically incorporated contaminants relative the contaminants applied in the lab test (likely contributes to conservative uncertainty; see Appendix J, Section 6.00), and extrapolations from one species to another (the effect of this uncertainty is not known). The assessment performed for heron exposure to mercury is a significant example of such uncertainty. First, dissolved mercury, which is more bioavailable than total mercury, was not detected in any of the surface water samples of the CLF Drainage Area, so the use of total mercury data to estimate heron exposure likely significantly overestimated exposure. Second, this conservative estimate is greatly compounded by the use of a BCF for methylmercury, since methylmercury is significantly more bioavailable and bioaccumulative than inorganic forms of mercury. The RfD we used for mercury was based on methylmercury, which is also the most toxic form of mercury. The food web assessment assumed that all mercury in heron prey organisms was methylmercury. Aside from the likely over estimate of total mercury in prey organisms, as discussed above, this is a reasonable assumption for fish (95 to 100 percent of the mercury in fish can be expected to be methylmercury). However, it is likely to be a conservative assumption for other prey organisms (e.g., aquatic invertebrates and amphibians) which were assumed to comprise about 13 percent of the heron's diet.

The food web models performed required the use of many assumptions regarding habitat usage and feeding habits of the indicator species. Whenever possible we used reasonable conservative assumptions in order to reduce the likelihood of underestimating risks to receptors, without producing such a conservative assessment that it would not be useful in trying to evaluate risk. However, because the conservative influences of these assumptions were compounded, the exposure model likely overestimated the risks. Detailed discussions of uncertainties for each food web assessment are presented in Appendix I, Section 6.00 and Appendix J, Section 6.00. Highlights of these uncertainties are presented below.

Risks to Great Blue Heron due to Surface Water and Sediment COPECs

Because the Upper Simmons Reservoir receives groundwater and storm water (and associated sediments) directly from Central Landfill, the Upper Simmons has a higher potential to be impacted by the landfill as compared to the Lower Simmons Reservoir. Likewise, sedimentation ponds and stream channels within the active portion of the landfill property can be expected to be impacted more by landfill contaminants as compared to the Lower Simmons Reservoir. However, the pattern of RfD exceedances, as shown on Table 9-28, indicates that the Lower Simmons Reservoir poses a greater potential risk to piscivorous birds than do the Upper Simmons or the streams and sedimentation basins within the active portion of the landfill property. This pattern alone suggests that potential risks from the landfill are comparable to, or lower than risks related to past and present land use of the area surrounding the landfill.

TQs calculated for the great blue heron suggest that DDT (and DDTR) presented the greatest potential for risk within the CLF Drainage Area as a whole, and the Lower Simmons Reservoir specifically. These risks are driven by a single detection of DDT in a surface water sample from the Lower Simmons Reservoir (0.0001 mg/l in SW98-54); due to the hydrophobic nature of DDT this detection was likely due entirely to particulate matter in the water sample. However, this concentration was used to represent a dissolved (i.e., freely available) surface water concentration for the Gobas model, which would significantly overestimated the predicted concentration of DDT in prey items of the heron. Therefore, it is likely that DDT (and DDTR) are not present in the Lower Simmons or the CLF Drainage Area at concentrations that could result in a risk to piscivorous birds.

In addition to the conservative uncertainties surrounding potential risks from DDT, DDT has never been considered a significant landfill-related contaminant, and this is supported by the pattern of detections in environmental media of OU2:

- Neither DDT, nor its derivatives were detected within sediments of the active portion of the landfill;

- Soil sample detections have been limited to wooded areas surrounding the active portion of the landfill, some of which were formerly used for agricultural purposes, and;

- The greatest risk appears to be in the Lower Simmons which has the lowest potential (of areas within the CLF Drainage Area) to have been impacted by OU1 landfill derived contaminants, and the greatest potential to have been impacted by surrounding agricultural property and other hazardous waste sites.

Benzylbutylphthalate, benzo(a)anthracene, benzo(a)pyrene, thallium, and mercury were the only other COPECs which resulted in estimated doses above LOAELs. Significant uncertainty exist for each of these COPECs regarding how representative the EPCs are with respect to contaminants migrating from the OU1 landfill, and/or the RfDs used to evaluate the potential for risk.

The LOAEL exceedance by thallium in the Lower Simmons Reservoir was driven by the detection of thallium in three of the five samples tested. Toxicological reference doses for toxicity of thallium to birds were not available. TQs for thallium are based on a LOAEL for the rat divided by an uncertainty factor of 10; this incorporates a significant degree of uncertainty into the assessment of potential risk from this metal. It should also be noted that the detection of thallium in sediments of the Lower Simmons was anomalous with respect to other data for surface water and sediments potentially impacted by the landfill. Aside from the Lower Simmons Reservoir sediments, the only other detections of thallium within media of the CLF Drainage Area were low concentrations detected (on a total basis) in two of six surface water samples from the Upper Simmons Reservoir. Thallium was not detected in any of the surface water or sediment samples collected from the landfill property (i.e., the Sedimentation Pond 4, and Sedimentation Ponds 2&3 and Channels data sets), or in Upper Simmons Reservoir sediment samples, and was not detected in Lower Simmons Reservoir surface water. Based on these considerations, it is unlikely that the OU1 landfill has resulted in significant risk to herons due to thallium.

No RfD was identified for butylbenzylphthalate, therefore we used the LOAEL-based RfD for di-n-butylphthalate, which was the lowest RfD for a phthalate compound presented in Sample et al., 1996. Therefore, there is considerable uncertainty related to the significance of the exceedances, which were mainly limited to low-level exceedances of the NOAEL (Table 9-28). The one LOAEL exceedance was based on one sediment sample from Sedimentation Pond 4, and the TQ was just 1.4. Based on these considerations, it is unlikely that butylbenzylphthalate presents a significant risk to herons.

Exceedances of LOAEL reference doses by mercury were limited to the sedimentation ponds, and, as discussed above, were driven by the detections of total mercury in surface water samples; dissolved mercury was not detected in any samples from the CLF Drainage Area. Particulate-associated (total) mercury in surface water is likely to have low bioavailability, thus the estimated EPC for fish and amphibian tissues in the sedimentation ponds (which were 1 to 3 orders of magnitude higher than in the Upper and Lower Simmons Reservoirs despite comparable sediment concentrations) are likely to be greatly overestimated. As discussed in more detail in Appendix I, there are several other considerations which suggest that the food web assessment for mercury was very conservative. These include: all detected mercury was assumed to be methylmercury (which is much more bioavailable and more toxic compared to inorganic mercury), the reference doses were based on tests with mallards which may have a lower capacity to detoxify mercury compared to piscivorous birds, and the fact that fish tissue results performed for the Upper and Lower Simmons Reservoirs by ESS (see Appendix K) indicated that fish tissue mercury concentrations are comparable to, or lower than averages for Rhode Island, and low-impact waters in Massachusetts. In addition, the estimated tissue concentrations of mercury in fish were slightly greater than the detection limits achieved by ESS for fish samples, suggesting that the body burden estimates were conservative. Based on these considerations it is unlikely that mercury within the CLF Drainage area poses a risk to herons that is greater than anthropogenic background risks.

The RfDs used for both benzo(a)anthracene and benzo(a)pyrene, were based on a LOAEL for rats of 10 mg/kg/day divided by an uncertainty factor of 10 to account for the extrapolation from mammals to birds. Estimated daily doses for these PAHs resulted in LOAEL TQs of just 1.3 and 3.1 in the Sedimentation Ponds 2&3 and Stream Channels exposure area. Since the LOAEL TQs are less than 10, the use of mammal RfDs for birds presents a high degree of uncertainty, and the Sedimentation Ponds 2&3 and Stream Channels exposure area consists of engineered waterbodies which are used to manage migration of sediments from the landfill facility, it is our opinion that these PAHs do not present a significant risk of harm to herons or similar birds which may feed within Sedimentation Ponds 2&3 and Stream Channels.

Aside from the COPECs which exceeded LOAELs, estimated doses of DDE, benzo(a)pyrene and beryllium exceeded their NOAELs at one or more exposure points. However the NOAEL-based TQs for these COPECs were all very low (less than 1.9), and given the nature of NOAELs (i.e., they represent the highest non-toxic values among the organisms tested), these exceedances are not considered to be significant. NOAEL-based total HQs are large for many of the exposure points identified within the CLF Drainage area, however these high values are driven by the HIs for thallium and mercury which, as discussed above, are not likely to represent risks from the OU1 Landfill.

Finally, the TOC content of suspended solids was assumed to be equal to the TOC content of sediments, which may have underestimated suspended solid TOC by as much as an order of magnitude. This was a conservative assumption, and likely resulted in significant overestimates of organic contaminants within the prey of great blue herons.

Risk estimates for heron exposure to COPECs in the Upper Simmons Reservoir were recalculated using maximum surface water and sediment concentrations. This did not result in any exceedances of LOAELs by individual COPECs, although he LOAEL-based total HI did increase from 0.9 (based on averages) to 2.0 (based on maximums). A few additional COPECs exceeded their NOAELs (DDT, DDTR and thallium), and the NOAEL-based total HI increased by about 2 times. The additional exceedances are few, and the magnitude of the risk estimate increases were small. Based on this evaluation of the Upper Simmons Reservoir, use of the maximum concentrations to calculate risks estimates for the heron would not have changed the conclusions drawn from the food web assessment.

Risks to Terrestrial Receptors due to Soil COPECS

As mentioned above, lead, chromium, and vanadium concentrations within OU2 soils were comparable to, or less than, natural background levels measured in the eastern United States (Shacklette and Boerngen, 1984), and Massachusetts DEP's background value for Massachusetts. In addition, the average OU2 concentrations of these metals are comparable to the site-specific averages. These metals were retained as soil COPECs because the maximum OU2 concentration exceeded the maximum among the two background soil samples collected. Additional background sampling would have better represented the range of background concentration in the area, and these three metals likely would have been eliminated as COPECs. In any case, because the OU2 concentrations of these metals are comparable to background, the fact that the estimated doses exceeded RfDs demonstrates the conservative nature of the food web assessments.

Exceedances of RfDs by estimated doses of zinc, selenium, and bis(2-ethylhexyl)phthalate were driven by anomolously high detections of each of these COPECs. If the anomolously high detections are removed from the calculation of estimated doses, bis(2-ethylhexyl)phthalate would not have exceeded any of its reference doses, and only the maximum concentration of zinc would result in a small exceedance of a LOAEL, with a LOAEL TQ of 1.3. In addition, without the anomolously high detection of zinc, the average OU2 concentration would be below the site-specific background average. Selenium was detected in just one of the 15 OU2 soil samples; aside from this detection, selenium would not have been included as a COPEC. Furthermore, the maximum-based NOAEL TQ of 1.2 was the only RfD exceedance by the estimated dose of vanadium.

DDT and its derivatives slightly exceeded LOAEL and NOAEL-based RfDs for the American robin. As discussed in detail above, based on the distribution of DDT detections, DDT is not considered to be a contaminant which is related to the OU1 landfill. Therefore, risks to robins suggested by the food web assessment are not related to the OU1 landfill, but are likely related to historic agricultural activities in the areas where soil samples were collected (i.e., within the wooded areas surrounding the active landfill property).

9.37.5 AVS and SEM Analyses

The AVS and SEM analyses performed pertain only to those metals included in the suite of SEM analyzed; namely cadmium, copper, mercury, nickel and zinc. Several other metals were included as COPEC in sediment, and these are not addressed by SEM/AVS ratios. For those samples with SEM/AVS ratios less than 1, the inclusion of other sulfide-forming cations in the sum of SEM may have raised the ratio above 1. This is a significant uncertainty with respect to AVS and SEM evaluations, and increases the chance of a false negative finding.

On the other hand, SEM/AVS analyses are conservative because they do not account for all of the binding phases which could make metals less bioavailable. For example, many cations adsorb readily with organic particulates and organic ligands. In addition, there is evidence that the use of a SEM/AVS ratio of 1 to evaluate potential availability of copper and nickel may be very conservative (Simpson et al., 1997; Ankley et al., 1993). Copper and nickel sulfides are not soluble in HCL (which is used for the extraction procedure for AVS and SEM), copper sulfides may be found in valence states aside from CuS (e.g., CuS_2 (therefore, the 1:1 assumption does not apply), and finally, during extraction, released Fe may catalyze the oxidation of CuS, releasing copper but no sulfides. Again the 1:1 ratio would not apply.

9.40 ECOLOGICAL RISK ASSESSMENT SUMMARY AND CONCLUSIONS

Based on toxicity tests performed on surface water and sediment samples from the Upper and Lower Simmons Reservoirs, on AVS and SEM analyses performed for sediment, and on qualitative surveys of the biota in the reservoirs, this ERA clearly demonstrates that there are no significant risks to aquatic biota, and therefore, there are no significant risks of direct toxic effects to fish, or indirect impacts to fish and wildlife which depend on aquatic species for food.

Based on the relatively low TQs in combination with the distribution of the contaminants relative to potential migration pathways from the OU1 Landfill, and the conservative uncertainty inherent in the food web assessment, it is our opinion that contaminants which migrated from the OU1 landfill do not pose a significant risk of harm to herons or to other wildlife that may be exposed to surface water and sediment contaminants through the food web.

Food web assessments for the American robin, meadow vole, and short-tailed shrew effectively ruled out the potential for significant risks to wildlife that feed within the wooded areas surrounding the active portions of the landfill property. Based on the relatively low TQs in combination with the distribution of contaminants relative to potential migration pathways from the OU1 Landfill, COPEC at concentrations which are comparable to typical concentrations for soil in Massachusetts and the eastern United States, and the conservative uncertainty inherent in the food web assessment, it is our opinion that contaminants which may have migrated from the OU1 Landfill to surrounding wooded areas do not pose a significant risk of harm to meadow voles, robins, shrews, or species with similar feeding habits.

Because existing condition EPCs are comparable to, or greater than estimated future condition concentrations of COPECs in the Upper Simmons Reservoir, the results of the measurement endpoints evaluated for existing conditions can be extrapolated to future conditions. Based on these extrapolations, there is not a significant risk of harm under future conditions due to migration of groundwater contaminants to the Upper Simmons Reservoir.

Because the potential for adverse effects from OU1 Landfill-derived contaminants is much greater in water bodies of the CLF Drainage Area as compared to the Almy Reservoir due to the relatively minor contribution of landfill derived groundwater, risk assessment results which indicate that there are no significant risks in the CLF Drainage Area can be extrapolated to the Almy Reservoir. Therefore, there is not a significant risk of harm to receptors in the Almy Reservoir under current or future conditions due to the migration of OU1 Landfill Contaminants.

9.50 REFERENCES

Ankley, G.T. Mattson, V.R., Leonard, E.N., West, C.W., and Bennett, J.L., 1993. Predicting the Acute Toxicity of Copper in Freshwater Sediments: Evaluation of the Role of Acid-Volatile Sulfide. *Environmental Toxicology and Chemistry*, 12: 315-320.

Ankley, G. T., D. M. Di Toro, D. J. Hansen, and W. J. Berry, 1996. Technical Basis and Proposal for Deriving Sediment Quality Criteria for Metals. Env. Tox. Chem., vol. 15, no. 12, pp. 2056 - 2066.

Efroymson, R.A., M.E. Will, and G.W. Suter II, 1997a. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision. Oak Ridge National Laboratories, ES/ER/TM-126/R2.

Efroymson, R.A., M.E. Will, G.W. Suter II, and A.C. Wooten, 1997b. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Terrestrial Plants: 1997 Revision. Oak Ridge National Laboratories, ES/ER/TM-85/R3.

EPA, 1998. National Recommended Water Quality Criteria; Notice; Republication. Federal Register, Thursday, December 26, 1998. Vol 63, No. 237, pp. 68354 - 68364.

EPA, 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conduction Ecological Risk Assessments. Interim Final. US EPA Environmental Response Team. EPA 540-R-97-006

EPA, 1996. Ecotox Thresholds, ECO Update, Volume 3, Number 2. US Environmental Protection Agency, Office of Solid Waste and Emergency Response. EPA 540/F-95/038. PB95-963324.

EPA, 1995a. Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants; States' Compliance - Revision of Metals Criteria. U.S. Environmental Protection Agency. Federal Register. Vol. 60, No. 86, Thursday May 4, 1995, pp 22229 - 22237.

EPA, 1995. Supplemental Guidance to RAGS: US Environmental Protection Agency, Region 4 Bulletins, Ecological Risk Assessment. Bulletin from EPA Region 4. November.

EPA, 1993a. Wildlife Exposure Factors Handbook Volume I. US Environmental Protection Agency, Office of Research and Development. EPA/600/R-93/187a. NTIS No. PB94-174778.

EPA, 1993b. Wildlife Exposure Factors Handbook Volume II. US Environmental Protection Agency, Office of Research and Development. EPA/600/R-93/187b. NTIS No. PB94-177789.

EPA 1992a. An SAB Report: Review of Sediment Criteria Development Methodology for Non-ionic Organic Contaminants. US Environmental Protection Agency, Science Advisory Board. EPA-SAB-EPEC-93-002. November.

EPA, 1992b. ECO Update Developing a Work Scope for Ecological Assessments. Vol. 1, No. 4. US Environmental Protection Agency. Publication 9345.0-05I.

EPA, 1992c. Framework for Ecological Risk Assessment. US Environmental Protection Agency. EPA/630/R-92/001.

EPA, 1992d. ECO Update Briefing the BTAG: Initial Description of Setting, History, and Ecology of a Site. US Environmental Protection Agency, Vol. 1, No. 5. Publication 9345.0-05I.

EPA, 1991a. Proposed Technical Basis for Establishing Sediment Quality Criteria for Nonionic Organic Chemicals Using Equilibrium Partitioning. US Environmental Protection Agency, Office of Science and Technology, Health and Ecological Criteria Division, Washington, D.C. 20460.

EPA, 1991b. ECO Update Ecological Assessment of Superfund Sites: An Overview. Vol. 1, No. 2. US Environmental Protection Agency. Publication 9345.0-05I.

Gobas, F.A.P.C. 1993. A Model for Predicting the Bioaccumulation of Hydrophobic Organic Chemicals in Aquatic Food-Webs: Application to Lake Ontario. Ecological Modeling, Vol. 69, pp 1-17.

Gobas, F.A.P.C. and X. Zhang. 1994. A Bioaccumulation and Trophic Transfer Model for Contaminants in Aquatic Food Webs. Version 1.00, Copyright May 1994.

GZA, 1993a. Upper Simmons Reservoir Phase II Report.

GZA, 1995. Upper Simmons Reservoir Screening Level Risk Assessment for Sediments, Operable Unit 2 Remedial Investigation - Task 1. June 1995.

GZA, 1993d. Upper Simmons Reservoir Sediment Study Phase I Report/Phase II Work Plan. February 1993.

GZA, 1993e. Upper Simmons Reservoir Sediment Study Phase II Report. February 1993.

Hull, R.N. and G.W. Suter II, 1994. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1994 Revision. Prepared by Oak Ridge National Laboratory for U.S. Dept. of Energy, Office of Environmental Restoration and Waste Management. ES/ER/TM-95/R1.

Jager, T., 1998. Mechanistic Approach for Estimating Bioconcentration of Organic Chemicals in Earthworms (Oligochaeta). Environmental Toxicology and Chemistry, 17(10): 2080-2090.

Jones, D.S., G.W. Suter II, and R.N. Hull, 1997. Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision. Oak Ridge National Laboratory, ES/ER/TM-95/R4.

Long, E.R. Morgan, L.G., 1991. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. National Oceanic and Atmospheric Administration Technical Memorandum NOS OMA 52. 175 pp.

Persaud, D., Jaagumagi, and A. Hayton, 1993. Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario. Ontario Ministry of the Environment, Queen's Printer for Ontario.

Rojko, A.M., 1990. Heavy Metals in the Sediments of Massachusetts Lakes and Ponds. Master Thesis, Dept. of Civil Engineering, Northeastern University, Boston, Massachusetts.

Russell, R.W., K.A. Gillan, and G.D. Haffner, 1997. Polychlorinated Biphenyls and Chlorinated Pesticides in Southern Ontario, Canada, Green Frogs. *Environmental Toxicology and Chemistry*, 16(11): 2258-2263.

Sample, B.E., D.M. Opresco, G.W. Suter II, 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Oak Ridge National Laboratory for U.S. Department of Energy, Office of Environmental Management, ES/ER/TM-86/R3. June 1996.

Simpson, S.L., S.C. Apte, G.E. Batley, 1997. AVS/SEM Fractionation of Sediments: A Comment on the Interpretation of Simultaneously Extracted Cu and Ni.. SETAC News vol. 17, no. 4, pp 18-19.

Shacklette, H.T., and J.G. Boerngen, 1984. Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States. . U.S. Geological Survey Professional Paper 1270. United States Government Printing Office, Washington.

Suter, G.W. II and J.B. Mabrey, 1994. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1994 Revision. Prepared by Oak

Ridge National Laboratory for U.S. Dept. of Energy, Office of Environmental Restoration and Waste Management. ES/ER/TM-96/R1.

Suter, G.W. II and C.L. Tsao, 1996. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision. Prepared by OakRidge National Laboratory for U.S. Dept. of Energy, Office of Environmental Restoration and Waste Management. ES/ER/TM-96/R2.

10.00 CONCLUSIONS

The remedial studies at the Central Landfill were separated, at the direction of the EPA, into two parts, designated as Operable Unit 1 (OU1) and Operable Unit 2 (OU2). Operable Unit 1 addresses source control issues for the 154-acre Phase I, II, and III Landfill area. The Remedial Investigation (RI) and Feasibility Study (FS) for that portion of the study were completed and accepted by the EPA, the Record of Decision (ROD) for OU1 has been issued, and remedial actions commenced in July 1997.

The OU2 Study was developed to address issues associated with contaminant migration from the OU1 Landfill. This document is the third and final draft of the RI report for OU2. It relies and builds upon information developed during the OU1 efforts and includes our responses to three rounds of written comments from the EPA and RIDEM. EPA also provided comments on the Human Health Risk Assessment and Ecological Risk Assessment that have been incorporated into Sections 8 and 9 of this report. For the most part, OU1 studies are described only briefly in this report, and the reader is to refer to the March 1993 OU1 RI report for details.

Our conclusions are based on factual information, scientific principals, and professional judgment. In order to understand how we reached these conclusions, the report must be read in its entirety. An understanding of the findings of the OU1 study (summarized in Appendix B of this report) is also necessary to appreciate our recommendations regarding groundwater. Note that our conclusions are subject to the Limitations presented in Section 11.00 of this report.

The overall purpose of the OU2 Study is to provide data to: 1) evaluate the extent of migration of contaminants which originate in the OU1 Landfill; 2) support a Baseline Human Health and Ecological Risk Assessment; and, 3) provide the data necessary to complete a feasibility study. It is our opinion that, these objectives were met.

- The boundaries of the OU2 Study Area, and the proposed testing program, were developed based on information generated during the OU1 Remedial Investigation. Testing performed during the OU2 RI support these boundaries. Resulting data indicate that: there is little on-going migration of OU1 Landfill-derived contaminants in groundwater towards the Almy Reservoir; contaminated groundwater discharges to, and does not migrate beneath, the Upper Simmons Reservoir; and there is no groundwater flow from the OU1 Landfill towards the Scituate Reservoir.

- Groundwater elevation data, groundwater quality data, and an analytical model were used to delineate the area that may be underlain by groundwater contaminated by the OU1 Landfill. This area and the real properties that make up the area are delineated and shown on Figures 7-1 and 7-2.

- A review of regulatory files identified nine sites (See Section 3.10), other than the OU1 Landfill, which could also be contributing to groundwater contamination in the OU2 Study Area. Data developed during the OU2 RI substantiate that some groundwater outside the buffer shown on Figures 7-1 and 7-2 has been effected by these sources. Consequently, groundwater outside the buffer zone may not be suitable for consumption due to other areas contaminant sources.

- Testing conducted during the OU2 RI found the types and concentrations of contaminants in groundwater to be consistent with that found during the OU1 RI. Similarly, the results of the OU2 hydraulic testing were in good agreement with OU1 testing. These data indicate that VOC contamination emanating from the OU1 Landfill Hot Spot is at steady state, and consequently VOC concentrations in the Almy and Simmons Reservoir will not increase with time.

- A baseline Public Health Risk Assessment was completed. It evaluated the potential risks to: current users of groundwater potentially impacted by the OU1 Landfill (only one identified), residents who use the upper Simmons and Almy Reservoirs for recreational purposes, juveniles who trespass on the landfill, and landfill workers. That study found that a condition of No Significant Risk exists for these receptors under current and reasonably foreseeable future conditions. Note, that the OU1 Risk Assessment established that the consumption of groundwater, in a now defined area between the OU1 Landfill and the Upper Simmons and Almy Reservoirs (See Figures 7-1 and 7-2), may pose unacceptable risks to human health.

- An Ecological Risk Assessment was performed for the OU2 Study Area. Groundwater migration from Central Landfill to Cedar Swamp Brook and the Upper and Lower Simmons Reservoirs was found to be the primary pathway resulting in exposure of ecological receptors to contaminants at levels above protective benchmarks. Several contaminants were also identified in surficial soils in the wooded areas surrounding the landfill at levels in excess of protective benchmarks. Toxicity tests and food web assessments were conducted to assess what impact contaminants may have on infauna and higher trophic level organisms. Results of these efforts showed no toxic effects and indicated that adverse effects through the food web are not likely. Thus, the Ecological Risk Assessment concluded that contaminants detected in the OU2 Study Area do not present a significant risk of harm to ecological receptors.

- The OU2 RI was conducted to support a Feasibility Study. The RI demonstrated, however, that the only unacceptable risk to human health or the environment is associated with the consumption of groundwater within a limited and defined portion of the OU2 Study Area. RIRRC has already made public water available to that area. Consequently, provided appropriate institutional controls are put in place to eliminate that exposure pathway, the

OU2 Feasibility Study would conclude that the No Action Alternative is the preferred remedial approach. This is true because the record of decision for OU1 provides for groundwater, surface water, and air monitoring within the OU2 Study Area which will identify changes in groundwater quality associated with the Central Landfill site, should they occur.

11.00 LIMITATIONS

This report was prepared for the exclusive use of the Rhode Island Resource Recovery Corporation (RIRRC) in partial fulfillment of the requirements of an EPA-mandated Remedial Investigation and Risk Assessment conducted for the Operable Unit 2 Study Area at the Central Landfill in Johnston, Rhode Island. We completed the studies and this report for that explicit purpose in accordance with an EPA and RIDEM approved Work Plan, and practices being used by other professionals in Rhode Island at the time the efforts were undertaken. No other warranty, expressed or implied, is made.

The analyses and opinions presented in this report are based in part upon the data obtained from subsurface explorations. Should additional information become available, it may be necessary to re-evaluate the conclusions presented in this report.

The generalized soil profile described in the text is intended to convey trends in subsurface conditions. The boundaries between strata are approximate and idealized and have been developed by interpretations of widely spaced explorations and samples; actual soil transitions are probably more gradual. For specific information, refer to the exploration logs.

Water level readings have been made in the drill holes and observation wells at times and under conditions stated. These data have been reviewed and interpretations have been made in the text of this report. However, it must be noted that fluctuations in groundwater table elevations will occur due to variations in rainfall, temperature, and other factors acting or not acting at the time measurements were made.

Air, water, and solids were sampled at specific locations using the techniques described (when known) in this report. These methods varied between studies and may have affected laboratory results. When comparing data presented in this report, sampling protocols should also be reviewed.

The interpretations and conclusions provided in this report are based in part upon chemical testing conducted by others and are contingent upon the accuracy and validity of their results. These data have been reviewed, and interpretations made, in the text and on the figures included with this report. It should also be noted that fluctuations in the types and concentrations of contaminants and variations in their migration pathways may occur due to seasonal effects, disposal practices, and other factors.