FINAL DRAFT

SUPPLEMENTAL ECOLOGICAL RISK ASSESSMENT AND REGULATORY FRAMEWORK FOR CERCLA ACTIVITIES AT ROUND LAKE, TWIN CITIES ARMY AMMUNITION PLANT

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This document was prepared by The Risk and Regulatory Analysis Team of the Environmental Sciences Division at the Oak Ridge National Laboratory (ORNL) for the U.S. Army Environmental Command in collaboration with the U.S. Army Environmental Command and the Twin Cities Army Ammunition Plant. Previous investigations by the U.S. Army Center for Health Promotion and Preventive Medicine (USACHPPM) and Wenck Associates, Inc. were used in this assessment.

The ORNL Environmental Sciences Division has been involved in the development of tools and methods for ecological risk assessment over the past two decades. ORNL assisted the USEPA in the development of the ecological risk assessment framework and EPA guidance (Framework for Ecological Risk Assessment, EPA/630/R-92/001 Feb 1992). In 1996, scientists from the Environmental Sciences Division developed toxicological benchmarks for the assessment of the effects of chemicals on aquatic species and mammalian and avian wildlife species for the U. S. Department of Energy. These toxicological benchmarks have been adopted nationally as comparative tools in screening assessments as well as lines of evidence to evaluate ecological effects at many contaminated sites. The Risk and Regulatory Analysis Team at ORNL is engaged in developing methods and providing analysis and assessments for the management of risks to human health and the environment that are associated with environmental contamination. Staff of the Risk and Regulatory Analysis Team developed and maintain the Risk Assessment Information System (RAIS) for DOE Oak Ridge Operations Office and provide risk-based decision support data and tools to the USEPA, Office of Superfund Remediation and Technology Innovation.

EXECUTIVE SUMMARY

This supplemental Ecological Risk Assessment (ERA) to the 2004 Tier II ERA reassesses potential risks from exposure to Round Lake sediments, as a part of CERCLA activities at the Twin Cities Army Ammunition Plant (TCAAP). This assessment is based on contaminant characterization information from 2011 sediment monitoring data, 2011 sediment toxicity testing results, 2012 fish tissue residue data and the draft 2012 U.S. Fish and Wildlife Service (USFWS) Management Plan for Round Lake. The analysis focuses on the effects of the final Chemicals of Concerns (COCs), as determined by Wenck in the 2012 Draft Feasibility Study, on populations and communities of organisms inhabiting Round Lake. The final COCs for Round Lake are cadmium, chromium, copper, lead, zinc and PCBs in sediment. In addition, a reassessment of the surface water COCs included in the Tier II ERA conducted by USACHPPM (2004) was performed.

Twin Cities Army Ammunition Plant (TCAAP) was placed on the National Priorities List as the New Brighton/Arden Hills Superfund Site in 1983. The source of TCAAP-related COCs in Round Lake was a storm sewer pipe that was connected to Building 502 at Site I. Part of the Site I facility had been used to produce artillery shell forgings. The water used to cool the production forges along with water from general cleanup flowed into the floor drains which discharged to the storm sewer which terminated at Round Lake. Due to the nature of the production process used at Building 502, PCBs and metals were expected to be the main chemicals of potential concern (COPCs). In 1969, it was discovered that many of the floor drains were still connected to the storm drain and this situation was remedied; effectively eliminating TCAAP as a source of contamination to Round Lake.

The 125 acre Round Lake is located in city of Arden Hills, Ramsey County MN which is to the south of the former TCAAP site. Round Lake represents an isolated wetland habitat surrounded by residential and industrial development which is bounded on all sides by major highways and roads. There are several lakes within a five mile radius of Round Lake located in more natural, less developed settings with larger water surface areas that serve migrating waterfowl.

A palustrine emergent wetland occurs at the edge of the lake; however, in the current 2004 Minnesota Valley Natural Wildlife Refuge (MVNWR) Conservation Plan, the USFWS describes Round Lake as a permanent wetland. The Minnesota Pollution Control Agency (MPCA) classifies the lake as a Class 2B/3B water of the state with the water uses of aquatic life and recreation/industrial consumption (Minn. R. ch 7050.0470, Subpart 1.B.). Round Lake is not classified for domestic consumption. The Trophic State Index (TSI) for Round Lake is 58, indicating that the MPCA classifies Round Lake as eutrophic. Round Lake is a depositional eutrophic lake with very high sedimentation rates (>1.5 cm/yr). The majority (95%) of the lake is a relatively flat shoal averaging approximately 4.5' in depth. No streams flow into Round Lake and surface water is recharged by precipitation and stormwater runoff. Based on the environmental setting for Round Lake, stormwater runoff from the surrounding highways, residences, commercial and industrial properties enter the lake. Deposition of the fine/organic sediments occurs in the deeper areas of the lake, creating higher sedimentation rates. The total organic content (TOC) of the sediments has been reported as 22%, suggesting an organic matter content of approximately 45% (Wenck, 2012). Anaerobic conditions exist at the surface of the substrate in the lake; with dissolved oxygen levels in the overlying water as low as 3.75 mg/L (USACHPPM, 2004). The total suspended solids in surface water are reported to average 8.00 mg/L, indicating high turbidity. Conductivity values for Round Lake have been reported as high as 669 µmhos/cm with an average value of 556 µmhos/cm (USACHPPM, 2004), which increases buffering capacity and reduces bioavailability.

The results of the 2004 Tier II ERA identified that adverse effects were confirmed in benthic organisms in sediment toxicity tests. The potential for adverse effects due to barium (for fish, aquatic invertebrates, and algae) and silver (for mammals) were also identified, but important uncertainties relating to these COCs remained and were eventually eliminated upon further evaluation and data collection. Adverse effects were not apparent for amphibians and were possible, but unlikely, for waterfowl and wading birds. Additionally, the potential for adverse effects due to PCBs exposure in mink, wading birds, and belted kingfishers was low for both the littoral and profundal areas of Round Lake because the exposure estimates were less than toxicity benchmarks (USACHPPM, 2004). The Tier II ERA (2004) concluded that the primary concern under the future scenario would be adverse changes in the survival, growth, and reproduction of benthic organisms.

In 2010, the USEPA and MPCA requested that the U.S. Army conduct an additional sediment investigation in Round Lake, which the Army conducted in 2011. Sediment samples were sectioned by depth and analyzed for metals, total PCBs, and TOC. The primary sampling effort to determine metal and PCB concentrations in sediment was conducted on a 200-ft by 200-ft sampling grid, with samples collected from the center of each grid. This resulted in 135 sampling grids which encompass Round Lake. During this supplemental ERA analysis, Round Lake COC sediment concentration data distribution analyses were performed with SAS Univariate Procedure (SAS 9.2, 3rd edition, 2013). The sediment data for each metal and total PCBs were analyzed at depth intervals of 0.0 to 0.5 feet, 0.0 to 1 feet, and 0.0 to 2 feet to determine statistical parameters. Goodness of fit analyses were explored with univariate protocols within SAS for the COC concentration and log transformed concentration data sets. Results indicate lognormal data distributions for each metal and PCBs.

The assessment endpoints evaluated in this supplemental ERA include: (1) survival, growth and reproduction of benthic organisms; (2) reproductive potential and productivity of aquatic mammals and waterfowl; and (3) survival, growth and reproduction of piscivorous species due to potential exposure to PCBs in fish tissue. Direct exposure to cadmium, chromium, copper, lead, zinc and PCBs in the sediments of Round Lake could occur to benthic invertebrates, aquatic mammals, waterfowl and aquatic vegetation by direct contact and ingestion/uptake. Subsequent indirect exposure of the contaminants to aquatic mammals and waterfowl could occur through the ingestion of benthic invertebrates and aquatic vegetation. Indirect exposure of PCBs to piscivorous birds and mammals could occur through the ingestion of fish that may have accumulated PCBs in their tissues. Two assessment endpoints, (1) survival, growth and reproduction of fish, aquatic invertebrates and algae and (2) survival, development and reproduction of amphibians, that were evaluated in the 2004 Tier II ERA were re-evaluated in this supplemental ERA and it was determined that the Round Lake COCs do not pose an unacceptable risk.

The benthic invertebrate community in Round Lake is representative of species that typically inhabit lentic littoral and profundal areas of a lake, predominantly comprised of amphipods, chironomids, mollusks and various insect larvae, predominantly Trichoptera. The species occur in depositional fine sediments mixed with organic matter or among vascular hydrophytes. For this supplemental ERA, the measures of effect used to evaluate the benthic invertebrate assessment endpoint are quantitative risk calculations and comparison to effects-based benchmarks as to-be-considered guidance values (TBC) for benthic organisms. There are no federal or Minnesota promulgated standards for sediments; consequently, comparison of concentration data to TBC guidance values is used as a measure of protectiveness. Threshold effect levels are only considered in ecological risk assessments if the protection of the individual ecological receptor is the goal for the presence of threatened and endangered species. No T&E benthic species occur at Round Lake, thus this supplemental ERA will focus on the benthic populations in the aquatic community in Round Lake. Additionally, supporting lines of evidence evaluated include results from studies addressing the bioavailability of the metals, benthic community surveys and sediment toxicity testing. The potential ecological risk to benthic invertebrates for the future use scenario was conducted in consideration of the draft 2012 USFWS Round Lake Conceptual Management Plan.

Results of the quantitative risk calculations show that the HQs for chromium, lead and PCBs are <1 at all depth intervals (0 to 2 feet), indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to chromium, lead and PCBs in sediments at depths of 0 to 2 feet. The HQs for cadmium and zinc slightly exceed 1 at the 0.0 to 0.5 foot depth interval, but are <1 at depths up to 1 to 2 feet. These results indicate a possible causal link of potential adverse effects to benthic organisms from exposure to cadmium and zinc at depths of 0.0 to 0.5 feet. The HQ for copper at these depths ranges from 1.3 to 3.2, indicating a possible causal link of potential adverse effects to benthic organisms from exposure to copper in sediments at Round Lake, especially at the 0.0 to 0.5 foot depth interval.

To perform a more in-depth analysis of the actual distribution of the trace metals and PCBs in the sediments of Round Lake and the subsequent potential for adverse effects to benthic organisms, the concentration data from each of the 200 x 200 foot grids at the 0.0 to 0.5 foot depth was compared to the MPCA SQT II benchmark. A relative ranking of the COC exceedances is as follows: copper > zinc > cadmium > lead, chromium and PCBs. Lead, chromium and PCBs exceed the benchmarks in ≤ 12% of the samples at all depths, indicating that these COCs are detected at lower concentrations and less frequently in the sediments of Round Lake. Sediments with contaminant concentrations that are higher than the recommended sediment quality guidelines only indicate that there is the potential for biological effects to occur, they do not necessarily indicate cause-effect relationships. The SQT II values do not consider the bioavailability of the metals in the sediments. Therefore, each COC was evaluated based on concentration data in each grid in relationship to its potential toxicity, bioavailability, ability to impact the biodiversity of the benthic population and co-location with other COCs. At Grid locations 6, 12, 17, 32 and 35 the detected concentrations of copper, cadmium and zinc are above the SQT II. However, results of studies with Round Lake sediments indicate that SEM/AVS ratios in the northern part of the lake ranged from 0.068 to 1.24, indicating that the divalent cationic metals are not likely to be bioavailable to benthic organisms. Numerous studies (Carlson et al, 1991; DiToro et al., 1992; Green et al., 1993; and Casas and Crecelius, 1994) using both freshwater and saltwater sediments have shown that acid-volatile sulfide (AVS) and interstitial water concentrations (IW) can be used to predict toxicity in sediments contaminated with divalent cationic metals (cadmium, copper, nickel, lead and zinc). In all of these studies, no toxicity was observed in amphipods, oligochaetes, snails, polychaetes and copepods when the simultaneously extracted metal (SEM) to AVS ratio was < 1.0. The relative affinity of metals for AVS is copper > lead > cadmium > zinc > nickel.

Results of TOC random sampling in 2011 show that the TOC concentrations in Round Lake sediments at depths up to 2 to 3 feet ranged from 63 to 330 g/kg (63,000 to 330,000 mg/kg) and the average total organic carbon in Round Lake sediments has been reported as 22%. Consequently, the TOC in the sediments of Round Lake influences the bioavailability of the chromium, copper, cadmium, lead and zinc detected at concentrations exceeding the SQT II at Grid locations 6, 9, 18, 19, 84, 86 and 98. The benthic community in the northern part of Round Lake (in proximity to grid locations 6, 11, 12, 17, 18, 19, 32, 35, 38, and 39) is healthy with diverse species composition. McGrath et al. (2002) states that toxicity is not expected when organic carbon is < 150 μmol/g; and Burton et al. (2005) reported no toxic effects to benthic community indices at 100 μmol/g (1.2 g/kg) of organic carbon.

The potential areas of concern for causing possible risk to benthic organisms are sediments in grids along the western edge of the lake (Grids 1, 3, 8, 10, 24, 26, 70, and 81), a cluster of grids in the deepest part of the lake (Grids 74, 85 and 97) and a few isolated grids in the southern tip of the lake (Grids 114, 120 and 129). Following an analysis of the measures of effect and the supporting lines of evidence (toxicity, bioavailability, ability to impact the biodiversity of the benthic population and co-location with other COCs), results indicate that copper and zinc in sediments may adversely impact the benthic organisms inhabiting Round Lake at Grid locations 1, 3, 8, 10, 24, 26, 70, 74, 81, 85, 97, 114, 120 and 129; cadmium in sediments may adversely affect the benthic organisms inhabiting Round Lake at Grid locations 3, 8, 10, 26, 85 and 97; and chromium at Grid 10 may adversely impact the benthic organisms inhabiting Round Lake. Benthic survey results indicate that the metal contaminants are not impacting the benthic community in the northern part of the lake; especially considering that the species inhabiting the lake are species that would typically be found in this type of environment. Based on the characteristics of the sediments at Round Lake, it is apparent that the toxicity of the metals is being strongly influenced by the bioavailability of the metals due to binding with particulate sulfide and organic carbon. The absence of SEM/AVS and TOC data in areas of the lake leads to some uncertainty concerning the actual potential risk of the grids listed above.

This supplemental ERA also considers potential ecological risk to benthic invertebrates for the future use scenario based on the draft USFWS Round Lake Conceptual Management Plan (USFWS, 2012) for water level management by periodically lowering the lake to a maximum drawdown elevation of 887.0 feet, which is only approximately 15% less than the normal lake level; thereby, exposing only a relatively small area of sediment. The drawdown elevation would only impact sediments around the edge of the lake and would be represented by the outer grids on the 200 x 200 ft grid map. During the proposed drawdown of the lake, possible oxidation of the exposed sediments and resuspension of deep sediment constituents to the surface might disrupt the metal sulfide binding in the sediments. However, results from several published studies with anoxic sediments indicate that the concentration of SEM cadmium, lead and zinc were not affected by resuspension. In addition, the SEM of copper was observed to increase with increasing resuspension and oxidation. The investigators found that during resuspension into oxic waters, iron and manganese monosulfide phases, which are usually present in large excess to other metal sulfides, buffered the initial oxidation of trace metal sulfide phases. The effects of bioirrigation and bioturbation from benthic organism activity were buffered and trace metal sulfide phases remained predominantly unoxidized for some time. It is important to note that benthic organisms actively maintain internal concentrations of essential metals, such as copper and zinc, through the use of homeostatic mechanisms and inorganic metals are not biomagnified or accumulated over two or more trophic levels (Chapman et al., 1996). Consequently, exposing the sediments in the outer grids through a periodic drawdown should not result in increased exposure or potential for adverse impacts to the benthic organisms inhabiting the sediments of Round Lake. In addition, the high organic carbon content of the sediment may prevent drying and limit oxidation and the subsequent release of bound metals. The accumulation of algae and plant material that will collapse on the sediment during drawdown may also limit drying and oxidation (USACHPPM, 2004). However, based on results for the current use scenario risk evaluation and conservative assumptions, concentrations of cadmium, copper, lead and zinc in outer Grids 1, 3, 70, 81 and 114 could possibly result in adverse impacts to benthic organisms during the proposed USFWS drawdown.

Mammals using Round Lake are red fox, muskrat (prominent use of shorelines) and mink. The muskrat was selected as the surrogate species for assessing the potential effects of the COCs on the reproductive potential and productivity of aquatic mammals. Muskrat lodges constructed of plant material (i.e., cattails) have been observed along the northeastern edge of Round Lake. Migrating waterfowl use Round Lake for stopover (resting and feeding) with some nesting. Species using the lake for nesting include Canada geese, mallard, blue-winged teal and wood ducks. Species using the lake for resting and feeding during spring and fall migrations include ringed-neck ducks, lesser scaup, black terns and common loons. The mallard was selected as the surrogate species for determining the potential adverse effects of the COCs to the reproductive potential and productivity of waterfowl. The bald eagle, belted kingfisher and great-blue heron were selected as the surrogate species for determining the potential adverse effects on the survival, growth and reproduction of piscivorous avian species feeding on fish from Round Lake that may contain PCB residues. Bald eagles have been observed nesting on the perimeter of the lake. Belted kingfishers have been observed on TCAAP and/or Round Lake and eat primarily fish. The great-blue heron have been reported in the TCAAP area from March through November but most leave the area for the winter. They do not nest at the site and are expected to forage there only occasionally. The mink was selected as the surrogate species for determining the potential adverse effects to piscivorous mammalian species feeding on fish in Round Lake that may contain PCB residues. Signs of mink activity at the lake have been observed (e.g., tracks in the winter, etc.). For this supplemental ERA, assessment endpoints for aquatic mammals, waterfowl and piscivorous species were not measured directly. The potential for adverse changes in the assessment endpoints were inferred by comparing estimates of exposure to estimates of health effects in the form of hazard quotients for metals and PCBs identified as final COCs. Exposure was estimated for an individual animal using a potential daily dose algorithm to predict estimates of doses averaged over a specified time frame. The likelihood for effects was estimated with the use of toxicity reference values (TRVs) of no-observable and lowest-observable effects from laboratory studies. For the piscivorous mammal (mink) and piscivorous birds (great blue heron, belted kingfisher and bald eagle), hazard quotients (HQ) were calculated based on the maximum concentration of measured PCBs in fish tissue as the ingested dose.

The assessment of potential impacts to aquatic mammals (muskrat as surrogate) indicates no exceedances of threshold or low effect levels based on exposure to sediment COC (metals and PCB) central tendency concentrations in the potential use areas for the muskrat (near shore area of Round Lake). The HQ's do not exceed unity indicating that on average exposures in the muskrat population are less than exposures known to be associated with adverse health effects. Likewise, the threshold effect and low effect levels based on the maximum measured concentration of cadmium, chromium, lead, zinc, and PCBs in sediment was not exceeded indicating the potential for adverse effects from these COC to muskrats is not likely. The threshold effect level and low effect level for copper was exceeded in a selected few sediment samples from areas along the northwestern/western shoreline based on the maximum concentrations indicating a potential for adverse effects. Since the northwestern/western shoreline is less favorable muskrat habitat and the potential for muskrat use in this area is less likely, exposure to sediments in this area is not likely. Although there are limited exceedances of effects levels for the muskrat based on the maximum copper sediment concentrations, there are no exceedances of effect levels based on measured sediment cadmium, chromium, lead, zinc and PCB concentrations suggesting the population exposure would be less than exposures known to be associated with adverse health effects. Furthermore, the potential for adverse effects to muskrats from copper would require repeated exposure to sediments along the northwestern/west shoreline which is not favorable muskrat habitat. Overall, the risk of adverse effects to aquatic mammals from exposure to COCs in sediments of Round Lake is not expected. Furthermore, the calculations of muskrat HQ's based on the maximum sediment COC concentrations and conservative exposure parameters provides an assessment of the risk of adverse effects that may result from exposures to sediments during a proposed future drawdown of the lake. Future risks to aquatic mammals from exposure to sediment COCs are not expected to exceed the HQ's based on the maximum sediment COC concentrations presented in this assessment.

The assessment of potential impacts to waterfowl (mallard as surrogate) indicates the hazard quotients for cadmium, chromium, copper, zinc, and PCB are less than 1.0 when the maximum detected concentration and the central tendency values are the basis for estimating potential exposure to metal residues in aquatic vegetation, invertebrates, and sediment. However, results indicate that exposure via the ingestion of aquatic vegetation, benthic invertebrates, and sediment exceeds the mallard threshold toxicity level (NOAEL) when the estimates of residues are based on the maximum concentration of lead in the sediment (HQ 1 > 1.0). The measured lead concentration in the north (Grids 11, 16, 18) and west (Grids 3, 8, 10, 26) adjacent to the shoreline and deeper sediments in the lake center region (Grids 85, 97, 98) are higher than other areas of Round Lake thus resulting in exceedance of the HQ 1, indicating that the potential for risk of adverse effects to waterfowl in these areas of the lake. It is noted that there is no exceedance of the HQ 2 (low effect level) for mallard when the maximum lead concentration is the basis for estimating exposure via sediments suggesting the population exposure would be less than exposures known to be associated with adverse health effects. The majority of the lead concentrations in the sediment (> 92% of grids) do not result in exceedance of the HQ 1 for the mallard based on estimates of exposure via ingestion of aquatic vegetation, benthic invertebrates, and sediment indicating that the risk of adverse effects to mallard population are not expected. Furthermore, the calculations of mallard HQ's based on the maximum sediment COC concentrations and conservative exposure parameters provides an assessment of the risk of adverse effects that may result from exposures to sediments during a proposed future drawdown of the lake. Future risks to waterfowl from exposure to sediment COCs may be expected in selected grids in the north and west adjacent to the shoreline (Grids 3, 8, 10, 11, 16, 18, 26) and selected grids in the deeper sediments in the lake center region (Grids 85, 97, 98).

For the piscivorous mammal (mink) and piscivorous birds (great blue heron, belted kingfisher, and bald eagle as surrogates), hazard quotients (HQ) calculated based on the maximum concentration of measured PCBs in fish tissue as the ingested dose and assuming 100% of the diet of each receptor species was fish from Round Lake were <1. Under this conservative scenario, the potential for risk of adverse effects to piscivorous avian and mammalian species is not expected from the consumption of fish at Round Lake. Furthermore, the calculations of HQ's for these species based on the maximum-exposed individual provides an assessment of the potential adverse effects that may result from exposure to PCBs in fish tissue during a proposed future drawdown of the lake. Future risks to piscivorous avian and mammalian species are not expected since fish tissue data indicates that bioaccumulation of PCBs up the food chain does not occur at levels that are toxic to receptor species.

In conclusion, a compilation of the areas within Round Lake that represent the potential for risk to ecological receptors is depicted in Figure 14. The highlighted grids on this map indicate the areas where detected concentrations of the final COCs may adversely impact the ecological receptors: benthic invertebrates and waterfowl (represented by the mallard).

- Benthic invertebrates – Grids 1, 3, 8, 10, 24, 26, 70, 74, 81, 85, 97, 114, 120 and 129 – potential exposure to cadmium, chromium, copper and zinc
- Waterfowl (mallard) – Grids 3, 8, 10, 11, 16, 18, 26, 85, 97 and 98 – potential exposure to lead

The Interstate Technology Regulatory Council (ITRC) Contaminated Sediments Team issued a guidance document (Incorporating Bioavailability Considerations into the Evaluation of Contaminated Sediment Sites, February 2011). The guidance notes the importance of bioavailability in risk assessments and in establishing technically defensible cleanup goals due to the low predictive value and conservative nature of SQGs when considered alone. The guidance states that the relationship between sediment contaminant concentrations and risk from exposure is not linear due to bioavailability considerations which may in some instances only result in a fraction of the contaminant being available to cause harm to ecological receptors (ITRC, 2011). In developing the guidance, ITRC identified several sites with similar sediment chemistry to Round Lake where bioavailability data was used in the remedial decision making process. Use of the bioavailability data (SEM/AVS ratios and TOC concentrations) was a key factor used in this supplemental ecological risk evaluation for Round Lake.

Based on the physical and chemical characteristics of Round Lake, two processes, lake succession (aging) and eutrophication, are determining the environmental conditions of the lake as well as potential exposure to ecological endpoints. Lake succession (aging) is the natural process by which a lake fills in over time with allogeneic erosional materials. Eutrophication is the process of increased nutrient input that can be accelerated by human activities, including stormwater runoff. Round Lake has no natural sources of water inflow to the lake; surface water is recharged by precipitation and stormwater runoff. The only outlet is a concrete structure with stoplogs to allow water level control during storm events. There is also limited connectedness to groundwater, an unconfined perched aquifer with glacial till below serves as an aquitard. Consequently, Round Lake is a depositional environment with sediment loading from stormwater runoff events. The sediment loading from stormwater events contributes to the natural recovery process by reducing the contaminants availability.

Round Lake exhibits several characteristics noted by EPA to be conducive to the natural recovery process. One consideration in the natural recovery process is the control of any significant sources of contaminants. EPA guidance notes that "MNR is likely to be effective most quickly in depositional environments after source control actions and active remediation of any high risk sediment have been completed" (Contaminated Sediment Remediation Guidance for Hazardous Waste Sites, page 4-11, OSWER 9355.0-85, EPA-540-R-05-012, December 2005). Id. at 4-11). Previously there have been releases of contaminants from production operations at TCAAP to Round Lake. Cessation of production operations at TCAPP eliminates the potential for any future releases. Other considerations that support the natural recovery process include:

- Anticipated land uses or new structures are not incompatible with natural recovery
- Natural recovery processes have a reasonable degree of certainty to continue at rates that will contain, destroy, or reduce the bioavailability or toxicity of contaminants within an acceptable time frame
- Sediment bed is reasonably stable and likely to remain so
- Sediment is resistant to resuspension (e.g., cohesive or well-armored sediment)
- Expected human exposure is low and/or can be reasonably controlled by institutional controls (*Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*, page 4-3, OSWER 9355.0-85, EPA-540-R-05-012, December 2005).

Land use at Round Lake is anticipated to remain as an USFWS wildlife refuge. No change in land use is anticipated which would be incompatible with MNR. Annual precipitation averages 29 inches/year resulting in an estimated average annual runoff to the lake of 200 to 300 acre-feet/year (excludes precipitation) [Wenck 2012]. Very high sedimentation rates (lake sediment dating 2003) of >1.5 cm/yr for the 20th century have been reported for the lake. Round Lake is overall a shallow depositional lake where the sediment is stable and resuspension is unlikely. The lake occupies approximately 125 acres with a maximum depth of 26' at the south-central end. However, less than 5% of the lake basin is more than 20' in depth; the majority is a relatively flat shoal averaging approximately 4.5' in depth (USFWS, 1992). Typically, shallow lakes <20' in depth do not exhibit mixing and turnover. USFWS does not currently allow fishing at Round Lake; and, a fish consumption advisory could be implemented by the USFWS for any future fishing activity if needed.

In some situations the natural recovery process may be occurring; however, the natural recovery process may be unable to reduce risks sufficiently within an acceptable time frame. In these situations, the natural recovery process can be accelerated or enhanced by applying a thin clean layer of material, usually as little as few inches. In most case natural material is recommended approximating common substrates found in the area. Such enhancement is distinguished from capping in that the purpose of the clean layer is to mix with the contaminated sediment. The addition of the thin clean layer of material is not designed to isolate the contaminants as in capping (where cap thickness can range up to several feet). Enhancement of degradation can also be facilitated by using additives to speed up the natural recovery (Contaminated Sediment Remediation Guidance for Hazardous Waste Sites, page 4-11, OSWER 9355.0-85, EPA-540-R-05-012, December 2005).

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1. INTRODUCTION

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Round Lake is located in city of Arden Hills, Ramsey County MN. Round Lake is located to the south of the former TCAAP site. Round Lake is described by USFWS as a 120-acre permanent wetland. Round Lake represents an isolated wetland habitat surrounded by residential and industrial development which is bounded on all sides by major highways and roads. No streams flow into Round Lake and surface water is recharged by precipitation and stormwater runoff. Nonpoint source runoff from the surrounding highways, residences, commercial and industrial properties may transport fertilizers, pesticides, particulates and petroleum byproducts into the lake. Round Lake is located off the installation but was part of TCAAP until 1974 when the U.S. Army transferred the lake to the U.S. Fish and Wildlife Service. Since under the purview of the USFWS, Round Lake has been managed by two different refuge systems the Sherburne National Wildlife Refuge System (SNWRS) and the Minnesota Valley National Wildlife Refuge (MVNWR). Round Lake is not a contiguous part of either refuge system. There are several lakes within a five mile radius of Round Lake located in more natural, less developed settings with larger water surface areas that serve migrating waterfowl. Round Lake was administered by the Sherburne National Wildlife Refuge from 1974 to 1979. The Sherburne Refuge is located approximately 40 miles NNW of the Twin Cities area (approximately 30 miles NW of Round Lake) and consists of 30,700 contiguous acres of wetlands, woodland, riverine and grassland habitats supporting over 230 species of birds (USFWS, 2004). Since 1979 Round Lake has been managed by USFWS as part of the MVNWR. The MVNWR primarily consists of eight units (not including Round Lake) along a 34-mile stretch of the Minnesota River and an additional unit located in the valley not immediately adjacent to the river. Figure 1 shows the northern-most extent of the eight unit MVNWR relative to the location of Round Lake approximately 10 miles to the NNW. The MVNWR was established as an unbroken corridor of floodplain and hillside forest, wetlands, oak savanna and native prairie along the Minnesota River. It is managed for the diverse and abundant native fish and wildlife populations that use the native plant communities of the Minnesota River and the Cannon River watersheds. Because of its wetland areas the Round Lake Unit was added as a remote part of the MVNWR. The current "USFWS Minnesota Valley National Wildlife Refuge and Wetland Management District – Comprehensive Conservation Plan and Environmental Assessment" (2004) states the objective for Round Lake is to maintain the lake at full basin water level to provide migration habitat for bald eagles and waterfowl, such as the canvasback and common loon. Specifically, the draft conceptual management plan (February 28, 2012) for Round Lake details plans to manage the lake for migratory habitat for diving ducks and other migratory waterfowl and water-dependent birds in the spring and fall. This would be accomplished by active water level management to fluctuate water levels 2 to 3 feet or less every 3 to 5 years. In addition, the USFWS is considering public use opportunities, such as, wildlife observation, environmental education and fishing. The USFWS draft conceptual plan for Round Lake will be considered in this supplemental ERA for the evaluation of potential risks for the future use scenario.

Figure 1. Map showing location of Round Lake in relationship to MVNWR.

2. REGULATORY FRAMEWORK

Under the CERCLA statute, two primary mandates establish the underlying legal and regulatory requirements for remedial activities at NPL sites. For a more detailed discussion of the legal/regulatory framework for CERCLA activities at Round Lake, see Appendix A. CERCLA §121(d)(1) requires remedial actions to attain a degree of cleanup that assures protection of human health and the environment [42 U.S.C. §9621(d)(1)]. This CERCLA requirement is implemented through means of a risk assessment. When such a risk is identified, remedial or removal action is required to address the unacceptable risk. CERCLA §121(d)(2)(A) requires that on-site remedial actions must meet the standards and criteria that are otherwise legally applicable to the substance, pollutant, or contaminant or that are relevant and appropriate under the circumstances [42 U.S.C. § 9621(d)(2)(A)]. The compliance with ARARs mandate arises under CERCLA 121(d)(2)(A) when an on-site remedial action is required. ARARs are only triggered when a remedial action is required because of unacceptable risk rather than the initiation of the CERCLA process.

Environmental Protection Agency guidance directs that all ecological risk assessments should generally be performed at every site following the eight step process described in Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, ERAGS, EPA 540-R-97-006, OSWER Directive # 9285.7-25, June 1997 (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 1, OSWER Directive 9285.7-28P, October 7, 1999). The goal of the Superfund is to reduce ecological risk to levels "that will result in the recovery and/or maintenance of healthy local populations/communities of ecological receptors that are or should be present at or near the site" (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 2, OSWER Directive 9285.7-28P, October 7, 1999). The Directive states that in evaluating ecological risks, the site should be characterized in terms of "1) magnitude; i.e., the degree of the observed or predicted responses of receptors to the range of contaminant levels, 2) severity; i.e., how many and to what extent the receptors may be affected, 3) distribution; i.e., areal extent and duration over which the effects may occur, and 4) the potential for recovery of the affected receptors."

3. SUMMARY OF TIER I AND II ECOLOGICAL RISK ASSESSMENTS

A Tier I screening-level risk assessment performed by USACHPPM in 1997 identified potential sediment-related risks indicating the need to perform a Tier II assessment. The Tier I assessment asserted that although Round Lake appears to be typical of a natural eutrophic pond environment, chemical impacts could be occurring. The Tier I assessment used sediment and surface water data collected from 1993 to 1995 by Montgomery Watson Inc., limited sediment biological evaluations from 1993 and a sediment-bioavailability study from the northern portion of the lake performed by the Minnesota Pollution Control Agency (MPCA) in 1994. The preliminary COCs identified in the Tier I screening assessment included barium and zinc in the surface waters; and aluminum, cadmium, chromium, copper, silver, vanadium, and zinc in sediments. Aquatic species, aquatic mammals, wading birds, benthic organisms, and amphibians were predicted to be potentially impacted by the surface water and sediment preliminary COCs. Results of the sediment biological evaluations were conflicting. Sediments in the southern portion of the lake were determined to be non-toxic to benthic organisms; however benthic diversity values were moderately low in the southern portion and moderate to moderately high in the northern portion where toxicity was expected to be greater. Results of the MPCA limited bioavailability investigation of sediment metals showed that there might be sufficient acid volatile sulfide in the sediments to bind cadmium, copper, mercury, and zinc.

Additional surface water sampling was performed between September 1999 and June 2000 for use in the Tier II ERA. USACHPPM also conducted studies to further evaluate the potential risks during the Tier II ERA. In 1995, sediment toxicity tests were conducted using field-collected sediments from the southern portion of the lake; and in 1999, sediment toxicity tests were conducted using field-collected sediments from the northern portion of the lake. USACHPPM also conducted a sediment-metal bioavailability study in 1998. Field surveys were conducted to measure ecosystem and receptor characteristics and controlled laboratory experiments were performed using field-collected water samples to measure adverse health effects in amphibians.

Based on results from the Tier I screening ERA, additional monitoring data and results from the additional studies; the chemicals of concern (COCs) selected for evaluation in the Tier II ERA by USACHPPM (2004) included barium, cadmium and zinc in surface water and cadmium, chromium, copper, lead, silver, vanadium, zinc and PCBs in sediments. The Tier II ERA evaluated five assessment endpoints for the current and future use scenario:

- Survival, growth and reproduction of fish, aquatic invertebrates and algal species
- Survival, growth and reproduction of benthic organisms
- Survival, development and reproduction of amphibians
- Reproductive potential and productivity of aquatic mammals (muskrat) and waterfowl
- Reproductive potential and productivity of aquatic mammals (mink), wading birds (Great Blue Heron) and piscivorous birds (Belted Kingfishers) exposed to PCBs.

In summary, the Tier II ERA for the current use scenario identified that adverse effects were confirmed in benthic organisms in sediment toxicity tests. The potential for adverse effects due to barium (for fish, aquatic invertebrates, and algae) and silver (for mammals) were also identified, but important uncertainties relating to these COCs were noted to remain, and were eventually eliminated upon further evaluation and data collection (Wenck 2012). Adverse effects were not apparent for amphibians and were possible, but unlikely, for waterfowl and wading birds. Additionally, the potential for adverse effects from exposure of PCBs in mink, wading birds, and belted kingfishers was low for both the littoral and profundal areas of Round Lake because the exposure estimates were less than toxicity benchmarks (USACHPPM, 2004). The future use scenario risk assessment qualitatively compared the results of the current scenario risk estimates to what may occur if the 1998 USFWS management plan for the optimum wildlife alternative (complete drawdown) was implemented. Based on the information available, the Tier II projected that if the 1998 management plan was implemented, then the aquatic species inhabiting Round Lake could possibly experience increased exposure to the COCs. The Tier II ERA (2004) concluded that the primary concern under the future scenario would be adverse changes in the survival, growth, and reproduction of benthic organisms.

4. SUPPLEMENTAL ECOLOGICAL RISK ASSESSMENT

4.1. Problem Formulation

4.1.1 Description of environmental conditions in Round Lake

Round Lake represents an isolated habitat surrounded by residential and industrial development which is bounded on all sides by major highways and roads (see Figures 2 and 3). A palustrine emergent wetland occurs at the edge of the lake; however, in the current 2004 MVNWR Conservation Plan, the USFWS describes Round Lake as a permanent wetland. MPCA classifies the lake as a Class 2B/3B water of the state with the water uses of aquatic life and recreation/industrial consumption (Minn. R. ch 7050.0470, Subpart 1.B.). Round Lake is not classified for domestic consumption. As shown in Figure 4 below, there are several lakes within a five mile radius of Round Lake that are located in less developed settings with larger water surface areas that likely serve as resting and feeding stopovers for migrating waterfowl.

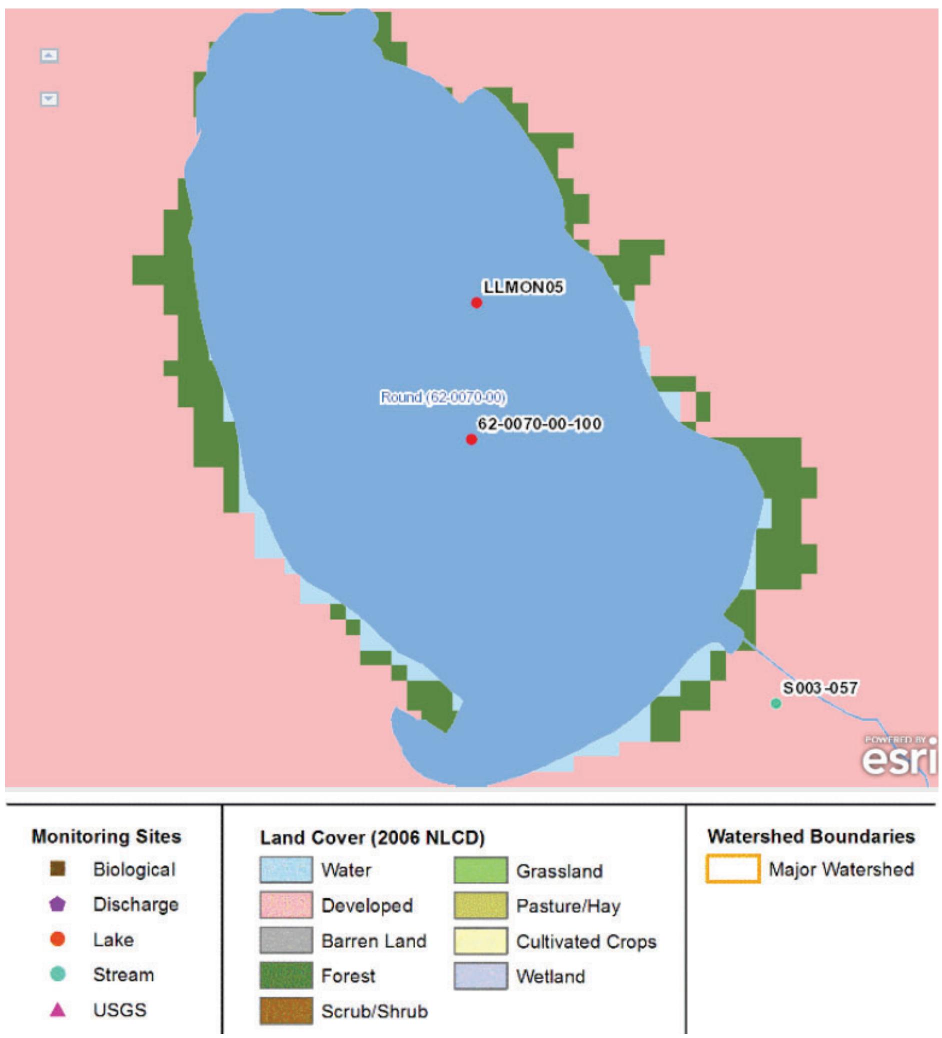


Figure 2. Land use map for Round Lake from MPCA Environmental Data Access System (July 2013)

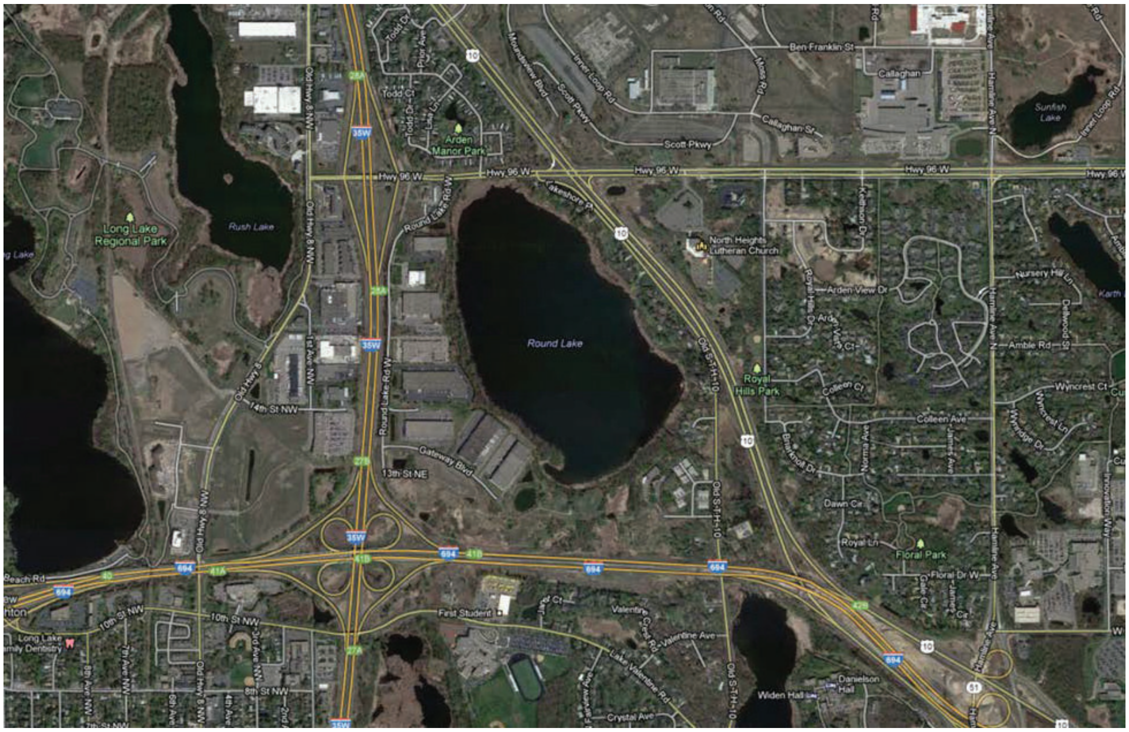


Figure 3. Map of Round Lake.
2013 Digital Globe, USGS Survey, USDA Farm Service Agency Data
[Google Maps (1 inch = approximately 1500 feet)]

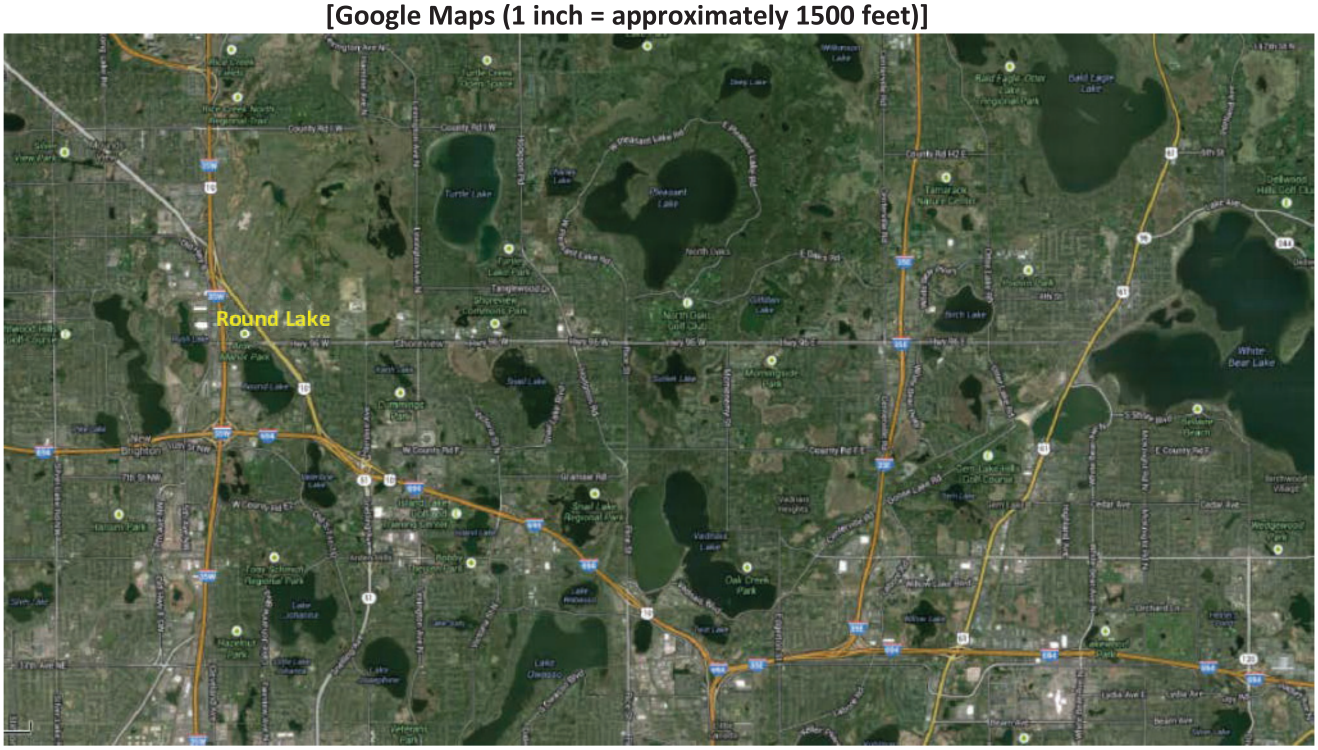
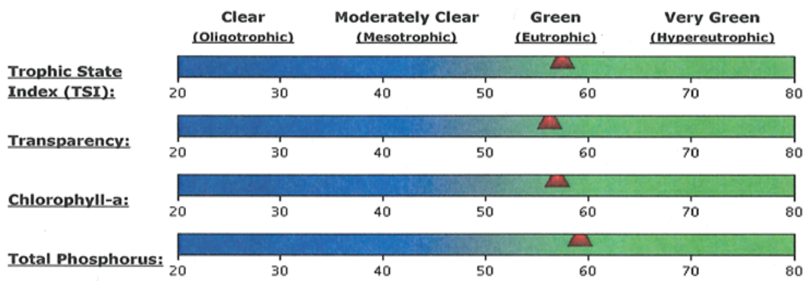


Figure 4. Round Lake and Proximity to Other Surface Waters.
2013 Digital Globe, USGS Survey, USDA Farm Service Agency Data
[Google Maps (1 inch = approximately 2 miles]

No natural streams flow into the lake; surface water is recharged by precipitation and stormwater runoff. Annual precipitation averages 29 inches/year resulting in an estimated average annual runoff to the lake of 200 to 300 acre-feet/year (excludes precipitation falling directly on the lake) (Wenck, 2012). There is also no natural outlet; the only outlet is a concrete structure with stoplogs to allow water level control. Excess water from Round Lake drains through the concrete outlet to Valentine Lake, which drains into a wetland before discharging to Long Lake and Rice Creek. Rice Creek eventually discharges to the Mississippi River. There is also limited connectedness to groundwater, an unconfined perched aquifer with glacial till below serves as an aquitard (Wenck, 2012). Based on the environmental setting for Round Lake, stormwater runoff from the surrounding highways, residences, commercial and industrial properties enter the lake which may transport fertilizers, pesticides, petroleum byproducts, and particulate matter. Consequently, Round Lake is a depositional environment with sediment loading from stormwater runoff events. The lake occupies approximately 125 acres with a maximum depth of 26' at the south-central end (maximum depth reported during 2011 sampling effort). However, less than 5% of the lake basin is more than 20' in depth; the majority is a relatively flat shoal averaging approximately 4.5' in depth (USACHPPM, 2004). Typically, shallow lakes <20' in depth do not exhibit mixing and turnover. Very high sedimentation rates (Engstrom, 2012) of >1.5 cm/year for the 20th century have been reported for the lake. High rates of sedimentation result in a decrease in water volume and lake storage capacity, an increase in evaporation rates, a decrease in light penetration, an increase in water temperature, a decrease in dissolved oxygen levels, smothering of bottom-dwelling life forms, inhibiting of recreational boating and fishing, impairing of the natural scenic beauty, and depreciation of property values (Helfrich et al., 2009).

Figure 5 presents a water quality data assessment from the Minnesota Pollution Control Agency's Environmental Data Access System for Round Lake. The Trophic State Index (TSI) for Round Lake is 58, indicating that MPCA classifies Round Lake as eutrophic. A eutrophic lake is typically shallow with a soft, mucky bottom and high level of nutrients (organic matter) with low dissolved oxygen on the bottom. The sediment composition of Round Lake is predominantly muck and peat and continues to 4 to 6 foot depths, with some cores near the shoreline encountering sand or clay at <4' depth (Wenck, 2012). The average depth is 4.5 feet. Deposition of the fine/organic sediments occurs in the deeper areas of the lake, creating higher sedimentation rates. The TOC content of the sediments has been reported as 22%, suggesting an organic matter content of approximately 45%. Anaerobic conditions exist at the surface of the substrate in the lake; with dissolved oxygen levels in the overlying water as low as 3.75 mg/L (USACHPPM, 2004). The total suspended solids average of 8.00 mg/L, indicating high turbidity. Conductivity values for Round Lake have been reported as high as 669 µmhos/cm with an average value of 556 µmhos/cm (USACHPPM, 2004), which increases buffering capacity and reduces bioavailability. High conductivity indicates a higher total dissolved mineral content. High turbidity and conductivity values are indicative of a eutrophic environment. Water turbidity is rarely directly lethal to fish and other aquatic life; however, excessive turbidity leads to low productivity and poor fish growth. Highly turbid lake waters decrease light penetration, limit photosynthesis by microscopic green plants (phytoplankton), and reduce the abundance of aquatic animals (zooplankton and insects) that feed on these tiny plants and, in turn, serve as important fish food organisms. Ultimately, growth and reproduction of sport fish are inhibited. In addition, turbid waters not only are much less aesthetically pleasing than clear lake waters, they frequently are unsuitable for domestic use (http://www.lakesuperiorstreams.org).

The **Trophic State Index (TSI)** is a number that summarizes a lake's overall nutrient richness. Nutrient richness ranges from clear lakes, low in nutrients (oligotrophic), to green lakes, with very high nutrient levels (hypereutrophic). The chart below shows the overall TSI rating for this lake (top bar), followed by TSI ratings for the individual parameters that contribute to nutrient richness. The TSI calculations are based on data collected between June and September 2000 to 2009.



Overall Trophic State Index for This Lake: 58

Parameter	10-Year Average of All Summer Samples	Parameter TSI	Expected TSI Range for Lakes in Same Ecoregion	Number of Samples
Transparency (meters)	1	56	43 - 54	11
Chlorophyll-a (parts per billion)	15	57	46 - 61	11
Total Phosphorus (parts per billion)	46	59	49 - 61	11

Water transparency is an excellent indicator of water quality, and the majority of these data are collected by volunteers. Join the MPCA's Citizen Lake Monitoring Program and help collect this important information for your lake.

Figure 5. MPCA Trophic State Index for Round Lake (2013)

In summary, Round Lake is a small isolated habitat within a larger area of residential and industrial development. The USFWS categorizes it as a 120-acre permanent wetland. Round Lake is a depositional eutrophic lake with very high sedimentation rates (>1.5 cm/yr). The majority (95%) of the lake is a relatively flat shoal averaging approximately 4.5' in depth. There are no natural streams which flow into the lake; lake water is recharged by precipitation and stormwater runoff. Based on the current physical and chemical characteristics of Round Lake, two processes, lake succession (aging) and eutrophication, are determining the environmental conditions of the lake. Lake succession is the natural process by which a lake fills with allogeneic materials causing shallowing and the evolution from an aquatic to terrestrial habitat. Eutrophication is the process of increased nutrient input (productivity) that can be accelerated by human activities, including stormwater runoff.

4.1.2 Summary of 2011 Monitoring Data

In 2010, the USEPA and MPCA requested that the U.S. Army conduct additional sediment investigation in Round Lake. The USEPA stated that more and better sediment data were needed to formulate and evaluate remedial alternatives. The MPCA stated that the limited data set in the southern two-thirds of the lake had introduced significant uncertainty to the alternatives evaluation. In an August 2010 meeting among the Round Lake stakeholders, the U.S. Army agreed to conduct the additional investigation work. Following USEPA and MPCA approval of sediment sampling objectives and procedures for sample collection and laboratory analysis, sediment samples were collect by piston-core methods through the ice in late-January and February 2011. Sediment samples were sectioned by depth and analyzed for metals, PCBs, and total organic carbon (TOC). Collection of sediment samples for toxicity analysis was conducted through the ice on March 11, 2011.

The primary sampling effort to determine metal and PCB concentrations in sediment was conducted on 200-foot by 200-foot sampling grid, with samples collected from the center of each grid. This resulted in 135 sampling grids which encompass Round Lake (see Figure 6). In order to evaluate the organic content of the sediment, a subset of the sampling grids (15 grids) were sampled for TOC. Sediment core sampling was conducted using a piston interface core sampler using push rods. The piston core sampler was used to collect approximately six feet of sediment from each sample point (unless refusal prevented a full recovery). At one location, where not even a full 12 inches of sample could be recovered, no sample was collected (Location 113). The analytical results from the 2011 sediment sampling were initially provided to USEPA and MPCA in May 2011, and later as part of a partial draft of the Round Lake Feasibility Study (Wenck, 2012).

Figure 6. Round Lake sediment sample locations (2011 sampling event, Wenck 2012).

Statistical analyses of the 2011 sediment monitoring data for the final sediment COCs were conducted by ORNL to support this supplemental ERA. Round Lake COC sediment concentration data distribution analyses was performed with SAS Univariate Procedure (SAS 9.2, 3rd edition, 2013) and the results are summarized below in Table 1. (Details on statistical approaches and output are provided in Appendix B). The sediment data for each metal and total PCBs was analyzed at depths of 0.0-0.5 feet, 0.0-1 feet, and 0.0-2 feet to determine statistical parameters. Goodness of fit analyses were explored with univariate protocols within SAS for the COC concentration and log transformed concentration data sets. Results indicate the data distributions as lognormal for each metal and PCBs. Percentiles and the 95% UCL from the analyses are presented in Table 1. The statistical analyses provide concentrations of metals and PCB for comparison to TBC guidance values and for risk estimates for ecological endpoints in this supplemental ERA.

Table 1. Statistical analysis of 2011 sediment monitoring data for the final COCs

Statistics	Round Lake Sediment Concentrations (mg/kg)[a]					
	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
***0.0 - 0.5 ft*[b]**						
Distribution	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
25th percentile	0.5	22	47	17	95	0.03
50th percentile	1.0	30	88	26	152	0.04
75th percentile	2.4	60	311	56	398	0.08
Maximum	26.6	295	924	258	854	0.89
Central Tendency[c]	1.1	33	102	30	175	0.05
95% UCL[d]	9.8	103	685	152	642	0.48
***0 - 1 ft*[b]**						
25th percentile	0.3	16	21	8	60	0.02
50th percentile	0.6	22	48	19	100	0.03
75th percentile	1.3	40	164	39	235	0.05
Maximum	28.5	816	1500	258	1150	9.03
Central Tendency	0.7	25	59	18	118	0.04
95% UCL	7.2	100	685	105	618	0.28
***0 - 2 ft*[b]**						
25th percentile	0.2	14	18	7	53	0.02
50th percentile	0.5	19	25	10	74	0.03
75th percentile	1.0	29	91	26	149	0.04
Maximum	28.5	816	1540	258	1250	9.03
Central Tendency	0.5	22	41.3	14	92	0.03
95% UCL	5.5	95	614	80	595	0.17

[a] 2011 Sediment Monitoring data (Wenck, 2012) analyzed with SAS 9.2 (3rd edition, 2013)
[b] 0-0.5 ft – 134 samples; 0-1 ft – 268 samples; 0-2 ft – 397 samples
[c] Central Tendency = Geometric mean for lognormal data.
[d] 95% UCL: The 95% upper confidence limit on geometric mean for lognormal distribution.

4.1.3 Conceptual Model

The conceptual model for this supplemental ERA is presented in Figure 7. Direct exposure to cadmium, chromium, copper, lead, zinc and PCBs in the sediments of Round Lake could occur to benthic invertebrates, aquatic mammals and waterfowl and aquatic vegetation by direct contact and ingestion/uptake. Subsequent indirect exposure of the contaminants to aquatic mammals and waterfowl could occur through the ingestion of benthic invertebrates and aquatic vegetation. Piscivorous birds and mammals could be indirectly exposed to PCBs through the ingestion of fish that may have accumulated PCBs in their tissues. The assessment endpoints to be evaluated are:

- Survival, growth and reproduction of benthic organisms
- Reproductive potential and productivity of aquatic mammals and waterfowl
- Survival, growth and reproduction of piscivorous species due to potential exposure to PCBs in fish tissue

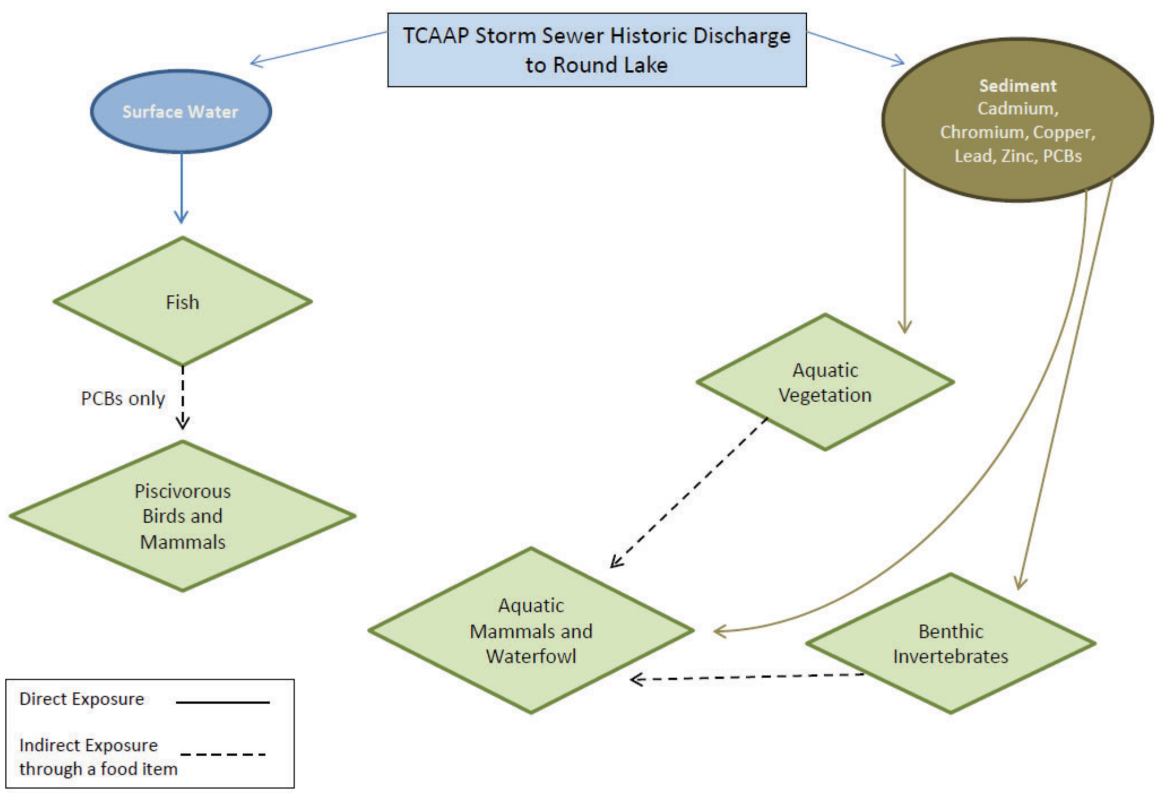


Figure 7. Conceptual model for Round Lake supplemental ERA

Two assessment endpoints evaluated in the 2004 Tier II ERA 1) survival, growth and reproduction of fish, aquatic invertebrates and algae and 2) survival, development and reproduction of amphibians, were re-evaluated by ORNL and it was determined that the Round Lake COCs do not pose an unacceptable risk. An explanation of this evaluation follows.

Potential effects to the survival, growth and reproduction of fish, aquatic invertebrates and algae were evaluated during the Tier II ERA to determine if the surface water COCs (barium, cadmium and zinc) were present above acceptable levels to produce adverse toxic effects to these aquatic species. The evaluation was conducted using quarterly surface water sampling in 1999 and 2000 to capture any seasonal changes in concentrations. The potential for adverse toxicological effects in these aquatic organisms was inferred by comparing site water concentration data to the Minnesota Chronic Water Quality Standards for Class 2B water (Minn. R. ch 7050.0222), which are the adopted USEPA National Water Quality Criteria for the protection of aquatic life, established under section 304(a)(1) of the Clean Water Act, United States Code, title 33, section 1314. Minn R. ch 7050.0218, Subpart 4 states that the USEPA criteria are applicable to Class 2 waters of the state. Under the Minnesota rules, a chronic standard (CS) is defined as the highest water concentration of a toxicant to which organisms can be exposed indefinitely without causing chronic toxicity. Chronic toxicity is defined as a stimulus that lingers or continues for a long period of time, often one-tenth the life span or more. A chronic effect can be mortality, reduced growth, reproduction impairment, harmful changes in behavior, and other nonlethal effects. None of the surface water samples collected for cadmium and zinc exceeded the chronic WQS for Class 2B waters (see Table 2), which are determined to be protective of at least 95% of a population of aquatic organisms; therefore, cadmium and zinc are excluded as COCs for Round Lake. In the absence of a MN chronic WQS for barium, a chronic barium benchmark was derived using USEPA Tier II criteria (based mostly on acute data and the absence of a complete data set used to derive a USEPA Ambient WQC for the protection of aquatic life). One hundred percent of the surface water samples collected for barium were greater than the derived Tier II criteria (3.8 µg/L); however, no samples exceeded the acute benchmark (110 µg/L). Consequently, the Tier II ERA determined that adverse toxicological effects are possible due to elevated barium concentrations in the surface water. In 2005, a revised benchmark value of 683 µg/L for barium was derived by MPCA and no samples exceeded this benchmark, resulting in barium also being excluded as a COC. The surface water COCs do not exceed the applicable requirements (ARARs) and pose no risk to fish, aquatic invertebrates and algae. Consequently, this assessment endpoint requires no further evaluation.

Table 2. Minnesota Water Quality Standards for surface water COCs.

COC	Minnesota Chronic WQS	Surface Water Concentration[a] (µg/L)
Barium	683[b]	76.45
Cadmium	1.1[c]	0.0186
Zinc	106[c]	1.37

[a]Concentrations are the central tendency (median) [Source: Tier II ERA, Tables 6-9, 6-10, 6-11]
[b]Benchmark value determined by MPCA
[c]Minnesota Administrative Rules Chapter 7050.0222 – Water Quality Standards for Class 2 Waters of the State; Aquatic Life and Recreation.

The 2004 Tier II ERA assessed the potential effects to the survival, development and reproduction of amphibians due to contaminants in Round Lake. The primary mechanism for the selected COCs to induce effects in amphibians is through contact as developing egg masses and then as juveniles. Contact was quantified as environmental concentrations in surface water, with the assumptions that the COC exposure is random within the site in that the populations of amphibians would likely lay eggs in a random spatial pattern. The FETAX (Frog Embryo Teratogenesis Assay-Xenopus) test was used to determine mortality and developmental toxicity from exposure to surface water at Round Lake. The FETAX tests, histopathological examinations, and immunological examinations are better indicators of site specific chemically-caused adverse effects in amphibians, because they are based on amphibian exposure to site water (see Table 3 below from USACHPPM, Appendix M, 2004). Laboratory data investigating embryo mortality (a sensitive life stage), developmental effects, and sensitive indicators of stress (immunocharacterization assays) suggest that surface water from Round Lake is not toxic to frogs. Additionally, data collected by Jannett (1997) and others (as discussed in USACHPPM 2004, Appendix M) corroborates a conclusion that adverse impacts of chemical exposures to amphibians are unlikely. In addition, as part of the amphibian study conducted at TCAAP (see USACHPPM, 2004, Tier II ERA Appendix M), Round Lake as a whole was examined to determine amphibian species richness and relative abundance. Visual surveys were difficult at Round Lake given the extensive cattail mats concentrating along the shores. However, five species of frogs were observed by aural surveys, visual encounter surveys (VES), and/or trapping: *Bufo americanus, Ranid ssp, Rana camitans, Pseudacris crucifer,* and *Rana pipiens.* Overall, relatively few amphibians were found at Round Lake. This assessment endpoint requires no further evaluation.

Table 3. Results of 1999 FETAX tests for Round Lake surface water.

Site	Petri Dish ID	Initial Number of Embryos	Mortality Count	Mortality Rate (%)	Number Malformed	Malformation Rate (%)
Round Lake	7	25	0	0	4	17
Round Lake	8	25	2	8	2	9
Snail Lake	5	25	5	20	1	6
Snail Lake	6	25	1	4	2	9
Control	1	25	1	4	1	4
Control	2	25	2	8	2	9
Risk Estimate		• The cumulative mortality rate was not greater statistically than at the reference lake or controls • The cumulative malformation rate was not greater statistically than at the reference lake. • The rates of specific malformations were not greater statistically than at the reference lake (data not shown, see text). • These estimates must be viewed with caution because the power to detect statistical differences was lower than expected (see text).				

†Information in this table was taken from Table 4 and Table 5 of the amphibian report found in Appendix M, *An Evaluation of the Effects From Potential Exposure to Military-Related Substances to Amphibians at Twin Cities Army Ammunition Plant*, USACHPPM 1999.

4.2. Exposure and Effects Analysis

4.2.1 Exposure Profiles for Ecological Receptors

The ecological receptors being evaluated in this supplemental ERA are benthic invertebrates, aquatic mammals, waterfowl and piscivorous birds and mammals.

Benthic survey studies were conducted by USACHPPM in 1993 (southern end of lake) and 1995 (northern end of lake). The benthic invertebrate community in the northern part of Round Lake is dominated by the following species: *Crangonyx gracilis* (crustacean – amphipod)*, Chaoborus punctipennis* (insect – dipteran -phantom midge), and *Chironomus decorus* (insect – dipteran – chironomid). Three mollusk species were relatively abundant: *Physella* gyrina (gastropod)*; Mentus dilatus* (gastropod); and *Sphaerium* sp. (bivalve) (see Table 26). The benthic invertebrate community in the southern part of Round Lake is dominated by *Crangonyx gracilis* (27.8%), *Chironomus decorus* (7.4%) and the mollusks *Mentus dilates* (5.4%) and *Sphaerium* sp. (see Table 27). The species collected typically inhabit lentic littoral and profundal areas of a lake. These species would be expected to inhabit Round Lake since the majority (95%) of the lake is a relatively flat shoal averaging approximately 4.5' in depth. The species occur in depositional fine sediments mixed with organic matter or among vascular hydrophytes. Sprawlers, such as *Chaoborus* and *Hesperophylax*, are known to inhabit the surface of floating leaves of vascular hydrophytes and burrowers, such as chironomids and mollusks, inhabit the fine sediments. The species are primarily herbivores (shredders feeding on vascular hydrophytes), detritivores (collectors/gatherers feeding on fine particulate organic matter), or predators (piercers and engulfers feeding on plant and animal tissue) (Merritt and Cummins, 1978).

Mammals using Round Lake are red fox, muskrat (prominent use of shorelines) and mink. The muskrat was selected as a surrogate species for determining the potential adverse effects to aquatic mammals using Round Lake from the exposure to the final COCs in sediments. Muskrat lodges constructed of plant material (i.e., cattails) have been observed along the northeastern edge of Round Lake (USACHPPM, 2004). The local populations may be exposed to contamination in riparian and wetland areas that occur on the outer edge of the lake. Muskrat will swim and feed on vegetation (primarily cattails) in the waters and use the sediments for building their dens. The home range of an individual muskrat is small, indicating that some individuals of the muskrat population may be continuously exposed (USACHPPM, 2004). The shoreline of Round Lake appears to have suitable habitat for muskrats (i.e. cattails) only in the northern, eastern and southeastern areas. The shoreline length of these areas is approximately 1000 meters. During field investigations, many dens were observed in the northern portion of the lake, but a formal count at the lake was not recorded.

Migrating waterfowl use Round Lake for stopover (resting and feeding) with some nesting. Species using the lake for nesting include Canada geese, mallard, blue-winged teal and wood ducks. Species using the lake for resting and feeding during spring and fall migrations include ringed-neck ducks, lesser scaup, black terns and common loons (USACHPPM, 2004). The mallard was selected as the surrogate species for determining potential adverse effects to migrating waterfowl from exposure to the final COCs in Round Lake sediments.

The bald eagle, belted kingfisher and great-blue heron are selected as the surrogate species for determining the potential adverse effects to piscivorous avian species feeding on fish in Round Lake that may contain PCB residues. Bald eagles have been observed nesting on the perimeter of the lake. Marsh birds also use the palustrine emergent wetland around the edge of the lake. Belted kingfishers have been observed on TCAAP and/or Round Lake and eat primarily fish. Belted kingfishers generally feed on fish that swim near the surface or shallow water. Kingfishers generally catch fish only in the upper 12 to 15 cm of the water column. Belted kingfishers capture fish by diving either from a perch overhanging the water or after hovering above the water. Fish are swallowed whole, head first, after being beaten on a perch. The average length of fish caught in a Michigan study was less than 7.6 cm but ranged from 2.5 to 17.8 cm; fish caught in Ohio streams range from 4 to 14 cm in length (USEPA Wildlife Exposure Factors Handbook, 1993). Because these birds prefer to perch in trees and avoid areas disturbed by human activity, the duration is expected to be very low during any given season spent in the TCAAP vicinity. This is indicated by high traffic volumes along the northern portion of the lake, a low density of trees surrounding the northern and eastern shorelines, and human activity in the area. The Great-blue heron have been reported in the TCAAP area from March through November but most leave the area for the winter (USGS 1995). They do not nest at the site and are expected to forage there only occasionally. Herons which feed primarily on fish are known to forage up to 24.4 km from their nesting colonies, with population densities of 2.3 birds/km along streams (USEPA 1993).

The mink was selected as the surrogate species for determining the potential adverse effects to piscivorous mammalian species feeding on fish in Round Lake that may contain PCB residues. Signs of mink activity at the lake have been observed (e.g., tracks in the winter, etc.). Shorelines and emergent vegetation are the mink's principal hunting areas. Mink are opportunistic feeders, taking whatever prey is abundant. Mammals are the mink's most important prey year-round in many parts of their range, but mink also hunt aquatic prey such as fish, amphibians, and crustaceans and other terrestrial prey such as birds, reptiles, and insects, depending upon the season (USEPA 1993). Mink will swim and feed on muskrat and other animals in the waters. Because they are expected to move in and out of areas of contamination Mink will not be continuously exposed at Round Lake. The Round Lake system is a small isolated habitat within a larger developed area with only one apparent migratory corridor in and out of the system (i.e., associated with the southern outfall into Valentine Lake). Such a small and isolated habitat may not be sufficient to support mink over the long term, though they have been observed (USACHPPM, 2004).

Results of a 1981 fish survey indicate that black crappie, black and brown bullheads and fathead minnow inhabit the lake. The Minnesota Department of Natural Resources collected fish samples in 2012 to determine the PCB tissue residue levels in the black bullhead, brown bullhead and green sunfish in filet and whole organism. The results are presented in Table 4.

Table 4. 2012 PCB tissue residue levels in fish inhabiting Round Lake

Round Lake Fish Tissue PCB Data (MN DNR, 2012)						
Year[a]	Species	Number of Fish	Length[b] (inches)	Weight[b] (kg)	Tissue Analyzed	PCB (ppm)
2012	Black Bullhead	4	6.4	0.06	Filet	0.025[c]
2012	Black Bullhead	4	7.0	0.085	Filet	0.04
2012	Black Bullhead	2	7.3	0.1	Whole Fish	0.262
2012	Brown Bullhead	5	7.5	0.094	Filet	0.025[c]
2012	Brown Bullhead	4	7.8	0.11	Filet	0.025[c]
2012	Brown Bullhead	2	8.1	0.11	Whole Fish	0.132
2012	Green Sunfish	4	4.6	0.04	Filet with skin	0.025[c]
2012	Green Sunfish	4	4.6	0.038	Filet with skin	0.025[c]
2012	Green Sunfish	2	5.1	0.055	Whole Fish	0.18

[a]Fish collected on 12/10/2012
[b]Average of fish collected
[c]PCB concentration was not greater than the detection limit (0.025 ppm)

4.2.2 Effects Analysis of Final COCs to Benthic Invertebrates

Copper. Copper is known to be toxic to aquatic organisms. Considerable research has been conducted on the levels of copper toxicity in aqueous solutions, but much less data is available on the toxicity of copper in sediments. Table 5 summarizes the available data for the toxicity of copper-containing sediments to invertebrates and benthic organisms. As illustrated in the table, the level of toxicity varies greatly among species. Among those tests that involved multiple species, it was generally observed that Daphnia magna and Gammarus pulex were some of the most sensitive species, and Tubifex tubifex was the most resistant species. Among the results in Table 5 for the freshwater species tested, the lowest observable effects concentration (LOEC) for copper in sediment ranged from 17 mg/g (dry weight) (for 28-day tests with Chironomus riparius) to 944 mg/kg (for 4-day tests with Chironomus crassiforceps). The median lethal concentration (LC50) for copper in sediment for the freshwater organisms ranged from 60 mg/kg (dry weight) in 28-day tests with Chironomus riparius to 4,522 mg/kg (dry weight) in 14-day tests with Chironomus tentans and from 128 mg/kg in 28-day tests to 1,078 mg/kg in 10-day tests with Hyalella azteca.

Table 5. Published data for toxicity of copper in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Source: M.A. Cairns et al. (1984)						
Chironomus tentans	Freshwater	LC50	10 day	2296 mg/dry kg	1690 to 3119	In spiked actual river sediment; 1.8% TOC
Daphnia magna	Freshwater	LC50	2 day	937 mg/dry kg	679 to 1291	In spiked actual river sediment; 1.8% TOC
Chironomus tentans	Freshwater	LC50	10 day	857 mg/dry kg	685 to 1073	In spiked actual pond sediment; 3.0% TOC
Daphnia magna	Freshwater	LC50	2 day	681 mg/dry kg	478 to 969	In spiked actual pond sediment; 3.0% TOC
Gammarus lacustris	Freshwater	LC50	10 day	964 mg/dry kg	777 to 1196	In spiked actual pond sediment; 3.0% TOC
Hyalella azteca	Freshwater	LC50	10 day	1078 mg/dry kg	922 to 1259	In spiked actual pond sediment; 3.0% TOC
Source: U. Borgmann and W.P. Norwood (1997)						
Hyalella azteca	Freshwater	LEC	10 week	114 µg/dry g	N/A	In spiked actual sediment; Data given as 1.8 µmol/dry g
Hyalella azteca	Freshwater	LC25	4 week	997 µg/dry g	362 to 1118	In spiked actual sediment; Data given as 15.7 (5.7 to 17.6) µmol/dry g
Source: Suedel (1995)						
Hyalella azteca	Freshwater	LC50	14 day	N/A	247 to 424 mg/dry kg	In spiked actual sediment

Table 5. Published data for toxicity of copper in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Chironomus tentans	Freshwater	LC50	14 day	N/A	1026 to 4522 mg/dry kg	In spiked actual sediment
Source: R.V. Hyne and D.A Everett (1998)						
Corophium sp.	Freshwater	LC50	10 day	999 mg/kg	848 to 1178	Spiked sediment
Source: T. Hagopian-Schlekat, G.T. Chandler, and T.J. Shaw (2001)						
Amphiascus tenuiremis	Freshwater	LC50	96 hour	281.9 µg/dry g	260.3 to 305.3	In spiked actual sediment
Source: M.R. Peck, D.A. Klessa, and D.J. Baird (2002)						
Chironomus crassiforceps	Freshwater	NOEC	4 day	644 mg/kg	N/A	Spiked sediment with pH = 4
C. crassiforceps	Freshwater	LOEC	4 day	944 mg/kg	N/A	Spiked sediment with pH = 4
C. crassiforceps	Freshwater	EC50	4 day	1292 mg/kg	1024 to 1763	Spiked sediment with pH = 4
C. crassiforceps	Freshwater	NOEC	4 day	22 mg/kg	N/A	Spiked sediment with pH = 6
C. crassiforceps	Freshwater	LOEC	4 day	403 mg/kg	N/A	Spiked sediment with pH = 6
C. crassiforceps	Freshwater	EC50	4 day	1287 mg/kg	1056 to 1518	Spiked sediment with pH = 6
Source: D. Milani et al. (2003)						
Hyalella azteca	Freshwater	IC25 (growth)	N/A	76 mg/kg	72 to 78	For spiked, bulk sediment
Chironomus riparius	Freshwater	IC25 (growth)	N/A	78 mg/kg	35 to 143	For spiked, bulk sediment
Hexagenia spp.	Freshwater	IC25 (growth)	N/A	38 mg/kg	31 to 49	For spiked, bulk sediment
Tubifex tubifex	Freshwater	IC25 (reprod.)	N/A	181 mg/kg	161 to 493	For spiked, bulk sediment; data shown are combined for endpoints: no. young per adult, no. cocoons per adult, and percent cocoons hatched
Hyalella azteca	Freshwater	LC25	28 day	81 mg/kg	57 to 106	For spiked, bulk sediment
Chironomus riparius	Freshwater	LC25	10 day	265 mg/kg	191 to 318	For spiked, bulk sediment

Table 5. Published data for toxicity of copper in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Hexagenia spp.	Freshwater	LC25	21 day	60 mg/kg	55-65	For spiked, bulk sediment
Tubifex tubifex	Freshwater	LC25	28 day	349 mg/kg	300 to 393	For spiked, bulk sediment
Hyalella azteca	Freshwater	LC50	28 day	128 mg/kg	110 to 158	For spiked, bulk sediment
Chironomus riparius	Freshwater	LC50	10 day	402 mg/kg	307 to 488	For spiked, bulk sediment
Hexagenia spp.	Freshwater	LC50	21 day	93 mg/kg	90 to 98	For spiked, bulk sediment
Tubifex tubifex	Freshwater	LC50	28 day	524 mg/kg	478 to 567	For spiked, bulk sediment
Source: Y.E. Roman, et al. (2007)						
Tubifex tubifex	Freshwater	NOEC	28 day	78.3 mg/dry kg	N/A	In laboratory-formulated sediment
Hyalella azteca	Freshwater	NOEC	28 day	53.2 mg/dry kg	N/A	In laboratory-formulated sediment
Chironomus riparius	Freshwater	NOEC	28 day	59.5 mg/dry kg	N/A	In laboratory-formulated sediment
Lumbriculus variegatus	Freshwater	NOEC	28 day	80.5 mg/dry kg	N/A	In laboratory-formulated sediment
Gammarus pulex	Freshwater	NOEC	35 day	94.7 mg/dry kg	N/A	In laboratory-formulated sediment
Tubifex tubifex	Freshwater	LOEC	28 day	102 mg/dry kg	N/A	In laboratory-formulated sediment
Hyalella azteca	Freshwater	LOEC	28 day	95.4 mg/dry kg	N/A	In laboratory-formulated sediment
Chironomus riparius	Freshwater	LOEC	28 day	89.2 mg/dry kg	N/A	In laboratory-formulated sediment
Lumbriculus variegatus	Freshwater	LOEC	28 day	103 mg/dry kg	N/A	In laboratory-formulated sediment
Gammarus pulex	Freshwater	LOEC	35 day	176 mg/dry kg	N/A	In laboratory-formulated sediment

Table 5. Published data for toxicity of copper in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Tubifex tubifex	Freshwater	LC50	28 day	327 mg/dry kg	302 to 354	In laboratory-formulated sediment
Hyalella azteca	Freshwater	LC50	28 day	316 mg/dry kg	281 to 355	In laboratory-formulated sediment
Chironomus riparius	Freshwater	LC50	28 day	320 mg/dry kg	279 to 366	In laboratory-formulated sediment
Lumbriculus variegatus	Freshwater	LC50	28 day	211 mg/dry kg	194 to 229	In laboratory-formulated sediment
Gammarus pulex	Freshwater	LC50	35 day	151 mg/dry kg	115 to 198	In laboratory-formulated sediment
Source: M. Marinkovic, et al. (2011)						
Chironomus riparius	Freshwater	LOEC	28 day	17.2 mg/dry kg	N/A	In spiked artificial sediment
Chironomus riparius	Freshwater	LC50	28 day	60.2 mg/dry kg	50.8 to 69.6	In spiked artificial sediment
Source: F.O. Costa, A.D. Correia, and M.H. Costa (1998)						
Gammarus lacustris	Estuarine	LC50	10 day	6.8 mg/dry kg	5.8 to 8.3	In spiked actual sediment
Source: I.D. Marsden and C.H.T. Wong (2001)						
Paracorophium excavatum	Estuarine	LC50	10 day	55 µg/dry g	N/A	Data obtained from abstract; paper not available on-line
Source: G.E. Batley, et al. (2004)						
Tellina deltoidalis	Estuarine	NOEC	10 day	650 µg/g	N/A	Data obtained from table cited by Simpson and King (2005); Particulate copper in estuarine sediments
Tellina deltoidalis	Estuarine	LC50	10 day	1020 µg/g	N/A	See above
Melita plumulosa	Estuarine	NOEC	10 day	520 µg/g	N/A	See above
Melita plumulosa	Estuarine	LC50	10 day	1300 µg/g	N/A	See above
Source: C.K. King, S.A. Gale, and J.L. Stauber (2006)						

Table 5. Published data for toxicity of copper in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Melita plumulosa	Estuarine	NOEC juvenile	10 day	460 mg/dry kg	N/A	Spiked, whole sediment tests; pH 7.8 to 8.2, temp 20.5 to 21.5 °C, salinity 30 to 32%
Melita plumulosa	Estuarine	NOEC adult	10 day	< 550 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LOEC juvenile	10 day	820 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LOEC adult	10 day	550 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 juvenile	10 day	790 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 adult	10 day	1520 mg/dry kg	N/A	See above

[1]**Organisms**

Amphiascus tenuiremis is a burrowing and sediment-ingesting copepod
Chironomus crassiforceps is a tropical chironomid (non-biting midge; a family of nematoceran flies resembling mosquitoes)
Chironomus riparius is a non-biting midge, also called the harlequin fly
Chironomus tentans is a non-biting midge
Corophium sp. is an euryhaline amphipod crustacean
Daphnia magna is a cladoceran freshwater water flea
Gammarus lacustris is an amphipod crustacean
Gammarus pulex is an amphipod crustacean
Hexagenia spp. is the burrowing mayfly
Hyalella Azteca is an amphipod crustacean
Lumbriculus variegatus is an oligochaete worm, sometimes called blackworm
Melita plumulosa is an estuarine amphipod crustacean
Paracorophium excavatum is a tube-dwelling corophioid amphipod
Tellina deltoidalis is a bivalve mollusk
Tubifex tubifex is an oligochaete worm

[2]**Acronyms for Parameters:**

EC50 = effective concentration resulting in an inhibition of growth and/or reproduction of test organisms by 50%
IC25 = concentration resulting in an inhibition of reproduction of test organisms by 25%
LEC = lowest effect concentration
LC25 = concentration estimated to be lethal to 25% of test organisms exposed
LC50 = median lethal concentration for test organisms exposed (95% confidence interval)
LOEC = lowest concentration at which adverse effects are observed in test organisms
NOEC = concentration at which no adverse effects are observed in test organisms

Zinc. Zinc is a common element, found throughout the earth's crust. In sufficiently high concentrations, it has been shown to be toxic to aquatic organisms. While considerable research has been devoted on the levels of zinc toxicity in aqueous solutions, the data are sparse on the toxicity of zinc in sediments. Table 6 summarizes the available numerical data for the toxicity of zinc-containing sediments to invertebrates and benthic organisms. The data in the table indicate that the level of toxicity varies greatly among species. Among the results in Table 6 for the freshwater species tested, the lowest observable effects concentration (LOEC) for zinc in sediment ranged from 83 µg/g (for 10-week tests with *Hyalella azteca*) to greater than 785 µg/g (for 10-day tests with *Hyalella azteca* and *Chironomus tentans*). The median lethal concentration (LC50) for zinc in sediment for the freshwater organisms ranged from 69 mg/kg (for 72-hour tests with *Limnodrillus hoffmeisteri*) to 759 mg/kg (for 72-hour tests with *Stagnicola attenuata*).

Table 6. Published data for toxicity of zinc in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Source: G. Dave (1992)						
Daphnia magna	Freshwater	EC50	24 hour	3040 mg/dry kg	2400 to 3860	In spiked actual sediment
Daphnia magna	Freshwater	EC50	48 hour	543 mg/dry kg	401 to 726	In spiked actual sediment
Source: K. Liber et al. (1996)						
Chironomus tentans	Freshwater	LOEC	10 day	> 785 µg/g	N/A	Paper states "no measureable sediment toxicity" at concentration of 785 µg/g
Hyalella Azteca	Freshwater	LOEC	10 day	> 785 µg/g	N/A	See above
Source: U. Borgmann and W.P. Norwood (1997)						
Hyalella Azteca	Freshwater	LOEC	10 week	83 µg/dry g	N/A	In spiked actual sediment; Data given as 1.27 µmol/dry g
Hyalella Azteca	Freshwater	LC25	4 week	3530 µg/dry g	2680 to 4580	In spiked actual sediment; Data given as 54 (41 to 70) µmol/dry g
Source: M. Galar-Martinez, et al (2008)						
Hyalella Azteca	Freshwater	LC50	72 hour	114 mg/kg	103 to 146	In spiked actual sediment; single species tests [units in paper are unclear, assumed to be mmol/kg; data given as 1.75 (1.58-2.23)]
Limnodrillus hoffmeisteri	Freshwater	LC50	72 hour	69 mg/kg	65 to 116	In spiked actual sediment; single species tests [units in paper are unclear, assumed to be mmol/kg; data given as 1.06 (0.99-1.78)]
Stagnicola attenuata	Freshwater	LC50	72 hour	592 mg/kg	537 to 688	In spiked actual sediment; single species tests [units in paper are unclear, assumed to be mmol/kg; data given as 9.05 (8.21-11.02)]
Hyalella	Freshwater	LC50	72 hour	501	456 to	In spiked actual sediment;

Table 6. Published data for toxicity of zinc in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
azteca				mg/kg	523	multiple species tests [units in paper are unclear, assumed to be mmol/kg; data given as 7.66 (6.98-8.0)]
Limnodrilus hoffmeisteri	Freshwater	LC50	72 hour	553 mg/kg	457 to 618	In spiked actual sediment; multiple species tests [units in paper are unclear, assumed to be mmol/kg; data given as 8.46 (6.99-9.45)]
Stagnicola attenuata	Freshwater	LC50	72 hour	759 mg/kg	718 to 817	In spiked actual sediment; multiple species tests [units in paper are unclear, assumed to be mmol/kg; data given as 11.61 (10.98-12.49)]
Source: T. Hagopian-Schlekat, G.T. Chandler, and T.J Shaw (2001)						
Amphiascus tenuiremis	Estuarine	LC50	96 hour	671.3 µg/dry g	593.5 to 759.4	In spiked actual sediment
Source: C.K. King, S.A. Gale, and J.L. Stauber (2006)						
Melita plumulosa	Estuarine	NOEC juvenile	10 day	< 2290 mg/dry kg	N/A	Spiked, whole sediment tests; pH 7.8 to 8.2, temp 20.5 to 21.5 °C, salinity 30 to 32%
Melita plumulosa	Estuarine	NOEC adult	10 day	2290 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LOEC juvenile	10 day	2290 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LOEC adult	10 day	4530 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 juvenile	10 day	1790 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 adult	10 day	> 9040 mg/dry kg	N/A	See above
Source: M.S. Adams and J. L. Stauber (2008)						
Melita	Estuarine	LC50	10 day	3420	3320	In spiked actual sediment;

Table 6. Published data for toxicity of zinc in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
plumulosa		juvenile		mg/kg	to 3510	Summary of data from cited references
Melita plumulosa	Estuarine	LC50 juvenile	42 day	> 1770 mg/kg	N/A	In spiked actual sediment; Summary of data from cited references
Melita plumulosa	Estuarine	NOEC juvenile	13 day	500 mg/kg	N/A	In spiked actual sediment; Summary of data from cited references
Mysella anomala	Estuarine	NOEC	10 day	3700 mg/kg	N/A	In spiked actual sediment; Summary of data from cited references
Soletellina alba	Estuarine	NOEC	10 day	3950 mg/kg	N/A	In spiked actual sediment; Summary of data from cited references
Australonereis ehlersi	Estuarine	NOEC	10 day	3100 mg/kg	N/A	In spiked actual sediment; Summary of data from cited references
Nephtys australiensis	Estuarine	NOEC	10 day	3900 mg/kg	N/A	In spiked actual sediment; Summary of data from cited references

[1]**Organisms**

Amphiascus tenuiremis is a meiobenthic copepod (crustacean)
Chironomus tentans is a non-biting midge
Hyalella Azteca is an amphipod crustacean
Limnodrillus hoffmeisteri is an oligochaete worm, also called sludge worms
Melita plumulosa is an estuarine amphipod crustacean
Stagnicola attenuata is an air-breathing snail

[2]**Acronyms for Parameters:**

LC25 = concentration estimated to be lethal to 25% of test organisms exposed
LC50 = median lethal concentration for test organisms exposed (95% confidence interval)
LOEC = lowest concentration at which adverse effects are observed in test organisms
NOEC = concentration at which no adverse effects are observed in test organisms

Cadmium. Cadmium is considered to be a trace element in the earth's crust in that no significant deposits of cadmium-containing ores are known. Nevertheless, cadmium has a variety of industrial uses. Dissolved and soluble forms of cadmium can be toxic to aquatic organisms at elevated concentrations. Table 7 summarizes the available numerical data for the toxicity of cadmium-containing sediments to invertebrates and benthic organisms. As can be seen in the table, the level of toxicity varies greatly among species. Among those tests that involved multiple species, it was generally observed that *Hyallela azteca* and *Chironomus riparius* are much more sensitive to cadmium than *Tubifex tubifex* and *Hexagenia* spp. Table 7 shows that, for the freshwater species tested, the lowest observable effects concentration (LOEC) for cadmium in sediment can be as low as 0.5 mg/kg (for 10-day tests with *Chironomus riparius*). The median lethal concentration (LC50) for cadmium in sediment for the freshwater organisms ranged from 6 mg/kg (for 28-day tests with *Chironomus riparius*) to 1375 mg/kg (for 10-day tests with *Ilyodrilus templetoni*).

Table 7. Published data for toxicity of cadmium in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Source: D. Milani et al. (2003)						
Hyalella azteca	Freshwater	IC25 (growth)	N/A	10 mg/kg	6 to 18	For spiked, bulk sediment
Chironomus riparius	Freshwater	IC25 (growth)	N/A	16 mg/kg	14 to 20	For spiked, bulk sediment
Hexagenia spp.	Freshwater	IC25 (growth)	N/A	14 mg/kg	8 to 25	For spiked, bulk sediment
Tubifex tubifex	Freshwater	IC25 (reprod.)	N/A	301 mg/kg	259 to 938	For spiked, bulk sediment; data shown are combined for endpoints: no. young per adult, no. cocoons per adult, and percent cocoons hatched
Hyalella azteca	Freshwater	LC25	28 day	21 mg/kg	16 to 32	For spiked, bulk sediment
Chironomus riparius	Freshwater	LC25	10 day	28 mg/kg	26 to 30	For spiked, bulk sediment
Hexagenia spp.	Freshwater	LC25	21 day	560 mg/kg	357 to 752	For spiked, bulk sediment
Tubifex tubifex	Freshwater	LC25	28 day	600 mg/kg	526 to 702	For spiked, bulk sediment
Hyalella azteca	Freshwater	LC50	28 day	33 mg/kg	28 to 44	For spiked, bulk sediment
Chironomus riparius	Freshwater	LC50	10 day	39 mg/kg	36 to 46	For spiked, bulk sediment
Hexagenia spp.	Freshwater	LC50	21 day	815 mg/kg	595 to 1024	For spiked, bulk sediment
Tubifex tubifex	Freshwater	LC50	28 day	787 mg/kg	715 to 931	For spiked, bulk sediment
Source: K.A. Gust (2006)						
Hyalella azteca	Freshwater	LC50	10 day	484 mg/kg	417 to 550	In spiked actual bayou sediment
Source: K.A. Gust and J.W. Fleeger (2006)						
Ilyodrilus templetoni	Freshwater	LC50	10 day	1375 mg/kg	1340 to 1412	In spiked actual bayou sediment
Source: M. Marinkovic, et al. (2011)						
Chironomus riparius	Freshwater	LOEC	10 day	0.5 mg/dry kg	N/A	In spiked artificial sediment

Table 7. Published data for toxicity of cadmium in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Chironomus riparius	Freshwater	LC50	28 day	6.1 mg/dry kg	4.9 to 7.3	In spiked artificial sediment
Source: A.S. Green, G.T. Chandler, and E.R. Blood (1993)						
Amphiascus tenuiremis	Estuarine	LC50	96 hour	37.9 µg/g	N/A	Data as cited in Hagopian-Schlekat, Chandler, and Shaw (2001)
Source: M.H. Fulton et al. (1999)						
Ampelisca verrilli	Estuarine	LC50	10 day	4.5 mg/kg	3.9 to 5.5	In spiked sediments
Ampelisca abdita	Estuarine	LC50	10 day	11.8 mg/kg	9.9 to 14.2	In spiked sediments
Palaemonetes pugio	Estuarine	LC50	10 day	17.9 mg/kg	16.2 to 19.9	In spiked sediments
Brachionus plicatilis	Estuarine	LC50	24 hour	41.9 mg/kg	35.8 to 49.1	In spiked sediments
Source: CCME (1999)						
Rhepoxynius abronius	Estuarine	LC50	10 day	9.2 mg/kg (avg. of range)	6.9 to 11.5	Data cited as results from numerous papers
Source: DeWitt, T.H, et al. (1999)						
Chaetocorophium cf. *lucasi*	Estuarine	NOEC juvenile	10 day	681 µg/g	N/A	In spiked actual sediment
Chaetocorophium cf. *lucasi*	Estuarine	LOEC juvenile	10 day	748 µg/g	N/A	In spiked actual sediment
Chaetocorophium cf. *lucasi*	Estuarine	LC50 juvenile	10 day	748 µg/g	727 to 770	In spiked actual sediment
Chaetocorophium cf. *lucasi*	Estuarine	NOEC adult	10 day	681 µg/g	N/A	In spiked actual sediment
Chaetocorophium cf. *lucasi*	Estuarine	LOEC adult	10 day	748 µg/g	N/A	In spiked actual sediment
Chaetocorophium cf. *lucasi*	Estuarine	LC50 adult	10 day	880 µg/g	846 to 914	In spiked actual sediment
Source: C.K. King, S.A. Gale, and J.L. Stauber (2006)						
Melita plumulosa	Estuarine	NOEC juvenile	10 day	620 mg/dry kg	N/A	Spiked, whole sediment tests; pH 7.8 to 8.2, temp 20.5 to 21.5 °C, salinity 30 to 32%
Melita plumulosa	Estuarine	NOEC adult	10 day	260 mg/dry kg	N/A	See above

Table 7. Published data for toxicity of cadmium in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Melita plumulosa	Estuarine	LOEC juvenile	10 day	820 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LOEC adult	10 day	>260 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 juvenile	10 day	1630 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 adult	10 day	>260 mg/dry kg	N/A	See above
Source: X. Jiang et al. (2007)						
Acartia pacifica	Estuarine	LC50 resting eggs	72 hour	3.44 mg/dry kg	2.64 to 4.47	In spiked actual sediment
Source: K. Chung, M.H. Fulton and G.I. Scott (2007)						
Mercenaria mercenaria	Saltwater	LC50 juvenile	10 day	1.66 mg/dry kg	N/A	In spiked actual sediment
Rhepoxynius abronius	Saltwater	LC50	10 day	9.8 mg/dry kg	N/A	In spiked actual sediment
Ampelisca abdita	Saltwater	LC50	10 day	2600 mg/dry kg	N/A	In spiked actual sediment

[1]**Organisms**

Acartia pacifica is a marine calanoid copepod
Ampelisca abdita is an infaunal, tube-dwelling amphipod
Ampelisca verrilli is an infaunal, tube-dwelling amphipod
Amphiascus tenuiremis is a burrowing and sediment-ingesting copepod
Brachionus plicatilis is a rotifer species that feeds on phytoplankton and bacteria
Chironomus riparius is a non-biting midge, also called the harlequin fly
Chaetocorophium cf. *lucasi* is a marine amphipod
Melita plumulosa is an estuarine amphipod crustacean
Mercenaria mercenaria is a saltwater hard-shell clam, also known as a quahog
Hexagenia spp. is the burrowing mayfly
Hyalella Azteca is an amphipod crustacean
Ilyodrilus templetoni is a bulk-deposit feeding tubificid oligochaete worm
Palaemonetes pugio is a common shrimp species
Rhepoxynius abronius is a marine amphipod
Tubifex tubifex is an oligochaete worm

[2]**Acronyms for Parameters:**

IC25 = concentration resulting in an inhibition of reproduction of test organisms by 25%
LC25 = concentration estimated to be lethal to 25% of test organisms exposed
LC50 = median lethal concentration for test organisms exposed (95% confidence interval)
LOEC = lowest concentration at which adverse effects are observed in test organisms

NOEC = concentration at which no adverse effects are observed in test organisms

Lead. Lead is toxic to aquatic organisms at sufficiently high concentrations. Data are scarce on the toxicity of lead in sediments. Table 8 summarizes the available numerical data for the toxicity of lead-containing sediments to invertebrates and benthic organisms. Among the results in Table 8 for the freshwater species tested, the two experimentally derived LC50 values are 3800 mg/kg (for 4-day tests with *Chironomus dilutus*) and 6840 mg/kg (for 4-week tests with *Hyalella azteca*).

Table 8. Published data for toxicity of lead in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Source: G. Dave (1992)						
Daphnia magna	Freshwater	EC50	24 hour	13,400 mg/dry kg	11,200 to 16,200	In spiked actual sediment
Daphnia magna	Freshwater	EC50	48 hour	7600 mg/dry kg	6300 to 8890	In spiked actual sediment
Source: U. Borgmann and W.P. Norwood (1999)						
Hyalella Azteca	Freshwater	EC25 (growth)	4 weeks	6220 µg/g	N/A	In spiked actual sediment; data given as 30 µmol/g
Hyalella Azteca	Freshwater	EC50 (growth)	4 weeks	18,900 µg/g	N/A	In spiked actual sediment; data given as 91 µmol/g
Hyalella Azteca	Freshwater	LC25	4 weeks	3730 µg/g	3520 to 4140	In spiked actual sediment; data given as 18 (17-20) µmol/g
Hyalella Azteca	Freshwater	LC50	4 weeks	6840 µg/g	6420 to 7460	In spiked actual sediment; data given as 33 (31-36) µmol/g
Source: W.T. Mehler et al. (2011)						
Chironomus dilutes	Freshwater	LC50	4 days	3800 mg/dry kg	N/A	In joint toxicity sediment tests with a pyrethroid insecticide; data given are for Pb alone
Source: T. Hagopian-Schlekat, G.T. Chandler, and T.J. Shaw (2001)						
Amphiascus tenuiremis	Estuarine	LC50	96 hour	2462 µg/dry g	2097 to 2891	In spiked actual sediment
Source: C.K. King, S.A. Gale, and J.L. Stauber (2006)						
Melita plumulosa	Estuarine	NOEC juvenile	10 day	580 mg/dry kg	N/A	Spiked, whole sediment tests; pH 7.8 to 8.2, temp 20.5 to 21.5 °C, salinity 30 to 32%
Melita plumulosa	Estuarine	NOEC adult	10 day	3560 mg/dry kg	N/A	See above

Table 8. Published data for toxicity of lead in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Melita plumulosa	Estuarine	LOEC juvenile	10 day	1020 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LOEC adult	10 day	> 3560 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 juvenile	10 day	1980 mg/dry kg	N/A	See above
Melita plumulosa	Estuarine	LC50 adult	10 day	> 3560 mg/dry kg	N/A	See above
Source: X. Jiang et al. (2007)						
Acartia pacifica	Estuarine	LC50 resting eggs	72 hour	358 mg/dry kg	321 to 400	In spiked actual sediment
Source: M.S. Adams and J. L. Stauber (2008)						
Melita plumulosa	Estuarine	NOEC juvenile	13 day	300 mg/kg	N/A	In spiked actual sediment; Summary of data from cited references

[1]**Organisms**

Acartia pacifica is a marine calanoid copepod
Amphiascus tenuiremis is a burrowing and sediment-ingesting copepod
Chironomus dilutus is a non-biting midge
Daphnia magna is a cladoceran freshwater water flea
Hyalella Azteca is an amphipod crustacean
Melita plumulosa is an estuarine amphipod crustacean

[2]**Acronyms for Parameters:**

EC25 = *effective* concentration resulting in an inhibition of growth and/or reproduction of test organisms by 25%
EC50 = effective concentration resulting in an inhibition of growth and/or reproduction of test organisms by 50%
LC25 = concentration estimated to be lethal to 25% of test organisms exposed
LC50 = median lethal concentration for test organisms exposed (95% confidence interval)
LOEC = lowest concentration at which adverse effects are observed in test organisms
NOEC = concentration at which no adverse effects are observed in test organisms
PEC = probable effects concentration above which harmful effects on sediment-dwelling organisms would be expected to occur

Chromium. Chromium can exist in sediments in two oxidation states, Cr(III) and Cr(VI). Chromium(III) is relatively insoluble and nontoxic, while Cr(VI) is much more soluble and toxic. In whole-sediment toxicity testing, benthic organisms showed great tolerance to Cr(III); hence, the vast majority of toxicity test data for chromium is focused upon the effects of Cr(VI). Considerable research has been conducted on the levels of chromium toxicity in aqueous solutions, but much less data is available on the toxicity of chromium in sediments. Table 9 summarizes the available numerical data for the toxicity of chromium-containing sediments to invertebrates and benthic organisms. As can be seen in the table, LC50 data are non-existent for the toxicity of chromium in sediments to freshwater organisms. Among the results in

Table 9 for the freshwater species tested, the EC50 data range from 167 mg/kg for 48-hour tests with *Daphnia magna* for Cr(VI) to 436 mg/kg for 24-hour tests with *Daphnia magna* for Cr(III).

Table 9. Published data for toxicity of chromium in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Source: G. Dave (1992)						
Daphnia magna	Freshwater	EC50	24 hour	436 mg/dry kg	360 to 529	Data for Cr(III) in spiked actual sediment
Daphnia magna	Freshwater	EC50	48 hour	195 mg/dry kg	145 to 263	Data for Cr(III) in spiked actual sediment
Daphnia magna	Freshwater	EC50	24 hour	170 mg/dry kg	134 to 218	Data for Cr(VI) in spiked actual sediment
Daphnia magna	Freshwater	EC50	48 hour	167 mg/dry kg	141 to 200	Data for Cr(VI) in spiked actual sediment
Source: W.J. Berry et al. (2002)						
Ampelisca abdita	Estuarine	LC50	10 day	> 3000 mg/kg	N/A	Data for Cr(III) in spiked sediments as cited in Rifkin, Gwinn and Bouwer (2004)
Source: D.S. Becker et al. (2006)						
Ampelisca abdita	Estuarine	NOEC	10 day	1310 mg/kg	N/A	Data for total chromium in spiked actual sediment
Source: R.M. Burgess et al. (2007)						
Americamysis bahia	Estuarine	LC50	7 day	44.6 mg/wet kg	41.3 to 45.0	Data for total chromium in spiked actual sediment
Ampelisca abdita	Estuarine	LC50	7 day	48.8 mg/wet kg	45.0 to 55.0	Data for total chromium in spiked actual sediment

[1]**Organisms**

Ampelisca abdita is an infaunal, tube-dwelling amphipod
Americamysis bahia is a small shrimp-like crustacean
Daphnia magna is a cladoceran freshwater water flea

[2]**Acronyms for Parameters:**

EC50 = effective concentration resulting in an inhibition of growth and/or reproduction of test organisms by 50%
LC50 = median lethal concentration for test organisms exposed (95% confidence interval)
NOEC = concentration at which no adverse effects are observed in test organisms
PEC = probable effects concentration above which harmful effects on sediment-dwelling organisms would be expected to occur

PCBs. Polychlorinated biphenyls (PCBs) are organic compounds that do not occur naturally in the environment. The majority of PCBs were manufactured for use as dielectric and coolant fluids. PCBs are stable compounds and do not readily decompose; hence, they can be extremely persistent in the environment. Exposure to PCBs has been linked to adverse effects in aquatic organisms. In addition, PCBs can accumulate and magnify in food chains thereby affecting species that feed upon aquatic organisms. While a considerable amount of data exists on the toxicity of water-borne concentrations of PCBs, the data for the toxicity of PCBs in sediments are scarce. Table 10 summarizes the available numerical data for the toxicity of PCB-containing sediments to invertebrates and aquatic organisms. For total PCBs, a no-effect concentration of 1.07 mg/kg was reported in a 120-day study with a marine polychaete. For Aroclor-1254, 96-hour LC50 values ranged from > 3.4 mg/kg to > 60 mg/kg in saltwater shrimp to > 500 mg/kg in *Pimephales promelas*, a freshwater fish. For Aroclor-1242, a 96-hour LC50 of > 0.78 mg/kg was reported for a saltwater shrimp species.

Table 10. Published data for toxicity of PCBs in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Source: D.W. McLeese and C.D. Metcalfe (1980)						
Pimephales promelas	Freshwater	LC50	96 hour	> 500 mg/kg	N/A	Data for Aroclor-1254 in sediment as reported by Halter and Johnson (1977)
Source: P.C. Fuchsman, et al. (2006)						
Pontoporeia hoyi	Freshwater	NOEC	9 day	420 µg/g organic carbon	N/A	Data for a PCB mixture (seven congeners) in sediment as reported by Landrum et al. (1989)
Source: D.R. Nimmo et al. (1971)						
Uca minax and *Penaeus duorarum*	Saltwater	LC50	96 hour	> 60 mg/kg	N/A	Data for Aroclor-1254 in actual sediment
Source: D.W. McLeese and C.D. Metcalfe (1980)						
Crangon septemspinosa	Saltwater	LC50	96 hour	> 0.78 mg/kg	N/A	Data for Aroclor-1242 in spiked sediment
Crangon septemspinosa	Saltwater	LC50	96 hour	> 3.4 mg/kg	N/A	Data for Aroclor-1254 in spiked sediment
Source: M. Salizzato et al. (1998)						
Vibrio fischeri	Estuarine	EC50	N/A	40 µg/dry g	30 to 50	Data for total PCB concentration in spiked actual sediment
Source: P.M. Chapman (1996)						
Neanthes arenaceodentata	Saltwater	NOEC	120 day	1.07 mg/dry kg	N/A	Data for total PCB concentration in spiked actual sediment
Source: P.C. Fuchsman, et al. (2006)						
Rhepoxynius abronius	Saltwater	LC50	10 day	2900 µg/g organic carbon	N/A	Data for Aroclor-1254 in sediment as reported by Swartz et al. (1988)

Table 10. Published data for toxicity of PCBs in sediments to invertebrates and benthic organisms

Organism[1]	Habitat/ Media	Parameter[2]	Duration	Value	Range	Observations/ Comments
Microarthridion littorale	Estuarine	LC50 females	4 day	6400 µg/g organic carbon	N/A	Data for Aroclor-1254 in sediment as reported by DiPinto et al. (1993)
Microarthridion littorale	Estuarine	LC50 males	4 day	3000 µg/g organic carbon	N/A	Data for Aroclor-1254 in sediment as reported by DiPinto et al. (1993)
Macoma nasuta	Saltwater	NOEC	119 day	81 µg/g organic carbon	N/A	Data for PCB mixture (thirteen congeners) in sediment as reported by Boese, et al. (1995)
Neanthes arenaceodentata	Estuarine	NOEC	4 weeks	2560 µg/g organic carbon	N/A	Data for Aroclor mixture (seven congeners) in sediment as reported by Murdoch et al. (1997)

[1]**Organisms**

Crangon septemspinosa is a species of shrimp, also called the sand shrimp
Macoma nasuta is the bent-nose clam
Microarthridion littorale is an estuarine meiobenthic copepod
Neanthes arenaceodentata is a marine polychaete worm
Penaeus duorarum is the pink shrimp
Pimephales promelas is the fathead minnow
Pontoporeia hoyi is a freshwater amphipod
Rhepoxynius abronius is a marine infaunal ampiphod
Uca minax is the fiddler crab
Vibrio fischeri is a toxin-producing bacterium that is often found residing in the light-emitting organ of marine animals such as squid and fishes.

[2]**Acronyms for Parameters:**

EC50 = effective concentration resulting in an inhibition of growth and/or reproduction of test organisms by 50%
LC50 = median lethal concentration for test organisms exposed (95% confidence interval)
LOEC = lowest concentration at which adverse effects are observed in test organisms
NOEC = concentration at which no adverse effects are observed in test organisms
PEC = probable effects concentration above which harmful effects on sediment-dwelling organisms would be expected to occur
TEL = threshold effects level

4.2.3 Effects Analysis of Final COCs to Avian and Mammalian Receptors

The no-observed adverse effect and the low-observed adverse effect levels for the avian and mammalian receptor species evaluated in this assessment are presented in Tables 11 and 12 below. An explanation is provided of the toxicity study used to derive the TRVs for each COC.

Table 11. Toxicity Values for Avian Species

Final COC	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	Endpoint	Source
Cadmium	1.45	20	Reduced egg production	White and Finley, 1978
Chromium	1	5	Duckling survival	Haseltine et al., 1985
Copper	47	61.7	Reduced growth Observed mortality	Mehring et al., 1960
Lead	1.1	11.3	Reduced egg hatching success	Edens et al., 1976
Zinc	14.5	130.9	Reduced egg hatchability	Stahl et al., 1990
PCBs	0.176	0.88	Decreased egg production	Platanow and Reinhart, 1973

Cadmium. The study used to derive TRVs for avian species is an oral study conducted by White and Finley (1978) with mallard ducks exposed to cadmium chloride at doses of 0, 1.6, 15.2 and 210 ppm in the diet for 90 days. No adverse effects were observed in the mallard at doses ≤15.2 ppm (1.45 mg/kg/d); consequently, this value was determined to be the chronic NOAEL. At the 210 ppm dose level (20 mg/kg/d), reduced egg production was observed; consequently, this value was determined to be the chronic LOAEL. These values are also reported as acceptable for use in risk assessments by ORNL (Toxicological Benchmarks for Wildlife: 1996 Revision, Sample et al.) and USACHPPM (Standard Screening-level Measures of Ecotoxicological Effects, Revision 3.1, November 2001).

Chromium. The study used to derive TRVs for avian species is an oral study conducted by Haseltine et al. (1985) with black ducks exposed to trivalent chromium as $CrK(SO_4)$ at doses of 0, 10 and 50 ppm in the diet for 10 months. No adverse effects were observed at the 10 ppm dose (1 mg/kg/d); consequently, this value was determined to be the chronic NOAEL. Duckling survival was reduced at the 50 ppm dose (5 mg/kg/d); consequently, this value was determined to be the chronic LOAEL. These values are also reported as acceptable for use in risk assessments by ORNL (Toxicological Benchmarks for Wildlife: 1996 Revision, Sample et al.) and USACHPPM (Standard Screening-level Measures of Ecotoxicological Effects, Revision 3.1, November 2001).

Copper. The study used to derive TRVs for avian species is an oral study conducted by Mehring et al. (1960) with one-day old chicks exposed to copper oxide at doses of 0, 36.8, 52.0, 73.5, 104.0, 147.1, 208.0, 294.1, 403, 570, 749 and 1180 ppm in the diet for 10 weeks. No adverse effects were observed at doses ≤570 ppm (47 mg/kg/d); consequently, this value was determined to be the chronic NOAEL. Thirty percent reduced growth and 15% mortality was observed at the 749 ppm dose (61.7 mg/kg/d); consequently, this value was determined to be the chronic LOAEL. This study was determined to be the acceptable study by ORNL (1996). USACHPPM (2001) used a study conducted by Norvell et al (1975) with the only observed effect of reduced weight gain. The ORNL (1996) accepted value will be used to calculate risk to avian species in this Supplemental ERA.

Lead. The study used to derive TRVs for avian species is an oral study conducted by Edens et al. (1976) with Japanese quail exposed to lead acetate (100% bioavailable) at doses of 0, 1, 10, 100 and 1000 ppm in the diet for 12 weeks. No adverse effects were observed at the 10 ppm dose (1.13 mg/kg/d); consequently, this value was determined to be the chronic NOAEL. Reduced egg hatching success was observed at the 100 ppm dose (11.3 mg/kg/d); consequently, this value was determined to be the chronic LOAEL. Since the study considered exposure over 12 weeks and throughout the critical lifestage of reproduction, the ORNL (1996) accepted value will be used to calculate risk to avian species in this Supplemental ERA.

Zinc. The study used to derive TRVs for avian species is an oral study conducted by Stahl et al. (1990) with White Leghorn hens exposed to zinc sulfate at doses of 0, 20, 200 and 2000 supplemental Zn plus 28 ppm Zn in the diet for 44 weeks. No adverse effects were observed at doses ≤228 ppm (14.5 mg/kg/d); consequently, this value was determined to be the chronic NOAEL. Reduced egg hatchability (<20%) was observed at the 2028 ppm dose (131 mg/kg/d); consequently, this value was determined to be the chronic LOAEL. These values are also reported as acceptable for use in risk assessments by ORNL (Toxicological Benchmarks for Wildlife: 1996 Revision, Sample et al.) and USACHPPM (Standard Screening-level Measures of Ecotoxicological Effects, Revision 3.1, November 2001).

PCBs. The study used to derive TRVs for avian species is an oral study conducted by Platanow and Reinhart (1973) with chickens exposed to Aroclor-1254. The only observed adverse effect was decreased egg production at 0.88 mg/kg/d; consequently this value was established as the chronic LOAEL. The chronic NOAEL was determined by applying an uncertainty factor of 5 to the chronic LOAEL to estimate a chronic NOAEL of 0.176 mg/kg/d. ORNL (1996) reports an oral study with Ring-necked pheasant exposed to Aroclor-1254 at doses of 0, 12.5 and 50 mg/bird/week dosed orally via gelatin capsule for 17 weeks. Reduced egg hatchability was observed at both doses; consequently, the chronic LOAEL was determined to be 0.18 mg/kg/d. The chronic NOAEL was estimated by multiplying the chronic LOAEL by an uncertainty factor of 0.1. The USACHPPM accepted value will be used to calculate risk to avian species in this Supplemental ERA.

Table 12. Toxicity Values for Mammalian Species

Final COC	NOAEL (mg/kg/d)	Endpoint	Source	LOAEL (mg/kg/d)	Endpoint	Source
Cadmium	1.0	Reduced fetal implantations Reduced fetal survivorship Increased fetal resorptions	Sutuo et al., 1980	10	Reduced fetal implantations Reduced fetal survivorship Increased fetal resorptions	Sutuo et al., 1980
Chromium	3.28	Reduced body weight and food consumption	MacKenzie et al., 1958	13.14	Mortality	Steven et al., 1976
Copper	11.7	Kit survivorship	Aulerich et al. 1982	15.4	Kit survivorship	Aulerich et al., 1982
Lead	42	Pup mortality; birth weight	Ronis et al., 1998	126	Pup mortality; birth weight	Ronis et al., 1998
Zinc	160	Reduced fetal growth Increased fetal resorptions	Schlicker and Cox 1968	320	Reduced fetal growth Increased fetal resorptions	Schlicker and Cox 1968
PCBs	0.136	Decreased # of litters Decreased offspring weights Decreased offspring survival	McCoy et al., 1995	0.68	Decreased # of litters Decreased offspring weights Decreased offspring survival	McCoy et al., 1995

Cadmium. The study used to derive TRVs for mammals is an oral gavage study conducted by Sutuo et al. (1980) in rats exposed to cadmium chloride at doses of 0, 0.1, 1.0 and 10 mg/kg in the diet from six weeks through mating and gestation. No adverse effects were observed at the 1 mg/kg/d dose level. At the 10 mg/kg/d dose, fetal implantations were reduced by 28%, fetal survivorship was reduced by 50% and fetal resorptions increased by 400%. Consequently, the chronic NOAEL was determined to be 1 mg/kg/d and the chronic LOAEL was determined to be 10 mg/kg/d. These values are also reported as acceptable for use in risk assessments by ORNL (Toxicological Benchmarks for Wildlife: 1996 Revision, Sample et al.) and USACHPPM (Standard Screening-level Measures of Ecotoxicological Effects, Revision 3.1, November 2001).

Chromium. The study used to derive the no-effect TRV for mammals is an oral study conducted by MacKenzie et al. (1958) in rats exposed to hexavalent chromium as $K_2Cr_2O_4$ at doses of 0, 0.45, 2.2, 4.5, 7.7, 11.2 and 25 ppm in water for over a year. No adverse effects were observed at any dose level; consequently, the maximum dose (3.28 mg/kg/d) was determined to be the chronic NOAEL. ORNL and USACHPPM are in agreement with this value. The study used to derive the low-effect TRV for mammals is an oral study conducted by Steven et al. (1976) in rats exposed to hexavalent chromium at doses of 0, 134 and 1000 ppm in water for three months (subchronic). Increased mortality was observed at the 1000 ppm dose (131.4 mg/kg/d). This dose was considered the subchronic LOAEL. A chronic LOAEL was estimated by multiplying the subchronic LOAEL by an uncertainty factor of 10, resulting in a chronic LOAEL of 13.14 mg/kg/d. This ORNL (1996) accepted value will be used to calculate risk to aquatic mammals in this Supplemental ERA. The USACHPPM (2001) value will not be used as they used an uncertainty factor of 15 for extrapolation from subchronic to chronic exposures, which is an overly conservative assumption.

Copper. The study used to derive TRVs for mammals is an oral study conducted by Aulerich et al. (1982) with mink exposed to copper sulfate at doses of 0, 25, 50, 100 and 200 ppm in the diet for 357 days. At doses ≥50 ppm, the percentage of kit mortality increased; consequently, the chronic NOAEL was determined to be 11.7 mg/kg/d (25 ppm) and the chronic LOAEL was determined to be 15.14 mg/kg/d (50 ppm). The ORNL values will be used to calculate risk to aquatic mammals in this Tier II Supplement. ORNL (1996) based the NOAEL/LOAEL on total copper, indicating that the actual dosing during the study was as follows: 25, 50, 100 and 200 ppm Cu supplemental plus 60.5 ppm Cu in base feed making the total dosage higher than that reported by USACHPPM (2001). The ORNL (1996) accepted value will be used to calculate risk to aquatic mammals in this Supplemental ERA.

Lead. The study used to derive TRVs for mammals is an oral study conducted by Ronis et al. (1998) with rats exposed to lead acetate. Pup mortality and reduced birth weight was observed at 126 mg/kg/d and determined to be the chronic LOAEL. This value will be used to calculate risk to avian species in this Tier II Supplement. The chronic NOAEL was established at 42 mg/kg/d. This is the study determined to be acceptable for use by USACHPPM (2001) and will be used to calculate risk to aquatic mammals in this Supplemental ERA. This study was not evaluated by ORNL (1996).

Zinc. The study used to derive TRVs for mammals is an oral study conducted by Schlicker and Cox (1968) with rats exposed to zinc oxide at doses of 0, 2000 and 4000 pm Zn in the diet for days 1 – 16 of gestation (critical lifestage). No effects were observed at 2000 ppm; consequently the chronic NOAEL was determined to be 160 mg/kg/d. Reduced fetal growth rates and increased rates of fetal absorptions were observed in rats exposed to the 4000 ppm diet; consequently, the chronic LOAEL was determined to be 320 mg/kg/d. These values are also reported as acceptable for use in risk assessments by ORNL (Toxicological Benchmarks for Wildlife: 1996 Revision, Sample et al.) and USACHPPM (Standard Screening-level Measures of Ecotoxicological Effects, Revision 3.1, November 2001).

PCBs. The study used to derive TRVs for mammals is an oral study conducted by McCoy at al. (1995) with oldfield mice exposed to Aroclor-1254 at doses of 0 and 5 ppm in the diet for 12 months. Decreased offspring weights and survival and decreased number of litters were observed at the 5 ppm dose (0.68 mg/kg/d). USACHPPM (2001) divided the chronic LOAEL by an uncertainty factor of 5 to estimate a chronic NOAEL from the chronic LOAEL, resulting in the chronic NOAEL of 0.136 mg/kg/d. ORNL (1996) used an uncertainty factor of 10, which is an overly conservative assumption. The USACHPPM accepted value will be used to calculate risk to aquatic mammals in this Supplemental ERA.

4.3. Risk Estimation Procedures for the Benthic Organism Assessment Endpoint

For this supplemental ERA, the measures of effect used to evaluate the benthic invertebrate assessment endpoint are quantitative risk calculations and comparison to effects-based benchmarks for benthic organisms. Additionally, supporting lines of evidence evaluated include results from metal bioavailability tests, benthic community structure and sediment toxicity testing. The first measurement endpoint used as a link to determine causality of effects to these benthic invertebrate receptors is a hazard quotient (HQ) calculation. An HQ is the ratio between the estimated exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. As such the HQ ratio indicates whether or not an estimated exposure exceeds the selected toxicological criterion. Benthic invertebrate HQ's for each COC are calculated using the central tendency value from the 2011 sediment monitoring event and the threshold-effect and probable-effect sediment benchmarks (MPCA SQT I and SQT II benchmarks). The second risk estimate is a comparison of COC data from the 2011 sediment monitoring event (central tendency value) to potential TBC guidance values for sediments. Additional lines of evidence evaluated to support estimates of potential effects to benthic invertebrates include an assessment of the bioavailability of COCs in the sediment as determined by measurements of TOC and AVS/SEM in sediment samples. Information concerning the benthic invertebrate community inhabiting Round Lake was evaluated to determine potential impacts to the population dynamics and community structure from exposure to the metals and PCBs in the sediments. Also, results of sediment toxicity tests from Round Lake and data from the published literature were analyzed to assess biological effects of sediment contamination on survival and growth of benthic invertebrate test species.

4.4. Risk Estimation Procedures/Models for Aquatic Mammals, Waterfowl and Piscivorous Species Assessment Endpoints

For this supplemental ERA, assessment endpoints for aquatic mammals, waterfowl and piscivorous species were not measured directly. The potential for adverse changes in the assessment endpoint were inferred by comparing estimates of exposure to estimates of health effects in the form of hazard quotients for metals and PCBs identified as final COCs. We used single-point estimates of exposure and effect that highlight the variability in the collected site data. Exposure was estimated for an individual animal using a potential daily dose algorithm to predict estimates of doses averaged over a specified time frame. The likelihood for effects was estimated with the use of toxicity reference values (TRVs) of no-observable and lowest-observable effects from laboratory studies (see Tables 11 and 12 above).

For the COCs, risk estimates consisting of a hazard ratio matrix for each receptor-COC combination were calculated (similar to that shown below). In this matrix, each value is a hazard quotient (indicated by the letters A-D), where the calculated exposure dose to the receptor is based on either the maximum or central tendency detected concentration in Round Lake sediment and is divided by either the no-effects or low-effects TRV. Hazard quotients designated as HQ 1 are based on the No-effect TRV (NOAEL), and HQ 2 values are based on the low-effect TRV (LOAEL). HQs less than one indicate that the calculated exposure is less than a selected level of concern.

	2011 Sediment Data	HQ 1	HQ 2
COC	Maximum Concentration	A	B
	Central Tendency Value	C	D

Hazard quotient A indicates the likelihood for the maximally exposed individual animal in the population to experience an average daily dose greater than the highest level associated with no observable health effects in a laboratory population. If A is equal to 1 or less, then no excessive hazard exists for the exposed population. Hazard quotient B indicates whether or not the possibility exists for a maximally exposed individual animal in the population to experience an exposure greater than the lowest level associated with observable health effects in a laboratory population. Hazard quotient C indicates the likelihood that, on average, individual animals among the exposed site population will experience a daily dose greater than the highest level associated with no observable health effects in a laboratory population. Hazard quotient D indicates the likelihood that, on average, individual animals among the exposed population will experience a daily dose greater than the lowest level associated with observable health effects in a laboratory population.

For the piscivorous mammal (mink) and piscivorous birds (great blue heron, belted kingfisher and bald eagle), HQs were calculated based on the maximum concentration of measured PCBs in fish tissue as the ingested dose. Fish were collected from Round Lake in December 2012 and tissues (whole fish, filet, and filet with skin) were analyzed for PCBs (2012 MN DNR sampling). HQ 1's were estimated using the mammal or avian PCB threshold toxicity value (NOAEL) and HQ 2's were estimated using the mammal or avian PCB low effect toxicity value (LOAEL).

4.4.1 Exposure/Contact Modeling for Avian and Mammalian Ecological Receptors

Contact was quantified as a potential dose, which is the amount of the COC (metals and PCBs) ingested from sediment and aquatic food (vegetation and benthic invertebrates) per day with the use of an exposure algorithm (USEPA 1998a) for aquatic mammals and waterfowl. Also, the potential dose from PCBs in fish tissue was assessed for piscivorous mammals and birds. The following sections present the methods for dose estimation for each wildlife receptor. The methods for quantifying possible bioaccumulation of the COC into food tissues are described below. The exposure model was adapted from the USEPA Wildlife Exposure Factors Handbook (USEPA 1993) and Guidelines for Ecological Risk Assessment (USEPA 1998a). The following equation describes the general approach for calculating an estimated potential average daily dose.

$$DI = \sum_{k=1}^{m} (C_k \cdot F_k \cdot NIR)$$

where, DI is the average daily intake of a substance through oral exposure (mg/kg-d), C_k is the substance concentration in the k^{th} diet item of the animal (mg/kg), F_k is the portion of the animal's diet that is the k^{th} diet item for the season (unitless), NIR is the normalized ingestion rate (kg/kg-d), and m is the number of diet items. In this model, ingested sediment is considered a "diet item." It is recognized that the diet and feeding habits of animals can be quite variable over the course of a year, as seasons and available food resources change. Additionally, differences in feeding habits between years can also be different. This exposure model calculates a seasonal average daily dose of a chemical that an individual organism receives. The ingestion rates and exposure point concentrations are time-averaged over periods of time approximating a season (see Table 13). Most of the ingestion rates in the USEPA Wildlife Exposure Factors Handbook (1993) are associated with seasonal activities. The modeled "diet" of the animal is developed by determining the seasonal diet that would represent a reasonable maximum exposure for the particular feeding guild of the animal (e.g., herbivore, omnivore, carnivore, piscivore).

Table 13. Ingestion rates for ecological receptors.

Species	Body Weight (kg)	Diet	Ingestion Rate – Food (kg/kg-d)
Muskrat	1.55	90% emergent vegetation 10% sediment	0.34
Mallard	1.2	Emergent vegetation 30% spring, 60% other months Benthic invertebrates 60% spring, 30% other months 10% sediment for 12 months	0.0671
Mink	1	100% Trophic Level 3 fish	0.22
Great-blue Heron	2	100% Trophic Level 3 fish	0.18
Belted Kingfisher	0.15	100% Trophic Level 3 fish	0.0672
Bald Eagle	4.6	100% Trophic Level 3 fish	0.371

4.4.2 Estimating Metal and PCB Residues in Dietary Items and Sediment

To derive the average daily dose of metal residues in dietary items for this ecological risk assessment, the measured concentrations of metals and PCBs from the 2011 sediment monitoring event were analyzed. The 2011 sediment sampling event was predesigned on a 200’ x 200’ foot grid that encompassed all of Round Lake. The resulting grid consisted of the collection of 135 sample locations and 134 sediment samples which were analyzed for the presence of metals and PCBs . The primary area of Round Lake for feeding and use activities of the muskrat (species representing aquatic mammals) would be adjacent to the shoreline and consequently, encompass the outer grids of the lake. The primary use area for feeding and use activities for the waterfowl, piscivorous mammal, and piscivorous birds would be the entire lake; consequently, encompassing all 135 grids. The sediment COC data from the appropriate grid units (0-0.5 foot depth interval) were statistically analyzed to determine the maximum concentration and the central tendency value for each metal and PCBs. Both the maximum concentration and central tendency value are used to support dose estimates for the ecological endpoints for the defined use patterns.

4.4.3 Dose Algorithm for Aquatic Mammals and Waterfowl

Mammals (Muskrat). The muskrat was selected as a species of concern as muskrat lodges constructed of plant material (i.e., cattails) have been observed at Round Lake (USACHPPM, 2004). The local populations may be exposed to contamination in riparian and wetland areas. Potential muskrat exposure to Round Lake COC was evaluated for two pathways: incidental sediment ingestion and food ingestion. The primary food resource for muskrat is aquatic vegetation. Muskrats are primarily herbivorous but some populations are more omnivorous (USEPA 1993). They may consume crayfish, amphibians, turtles, and young birds. Muskrats generally feed at night on aquatic vegetation growing near their dens and have been known to dig for food on lake and pond bottoms. Roots and basal portions of aquatic plants make up most of their diet although shoots, bulbs, tubers, sterns and leaves are also eaten. There is also evidence that muskrats prefer cattails, bulrushes and other marsh grasses and sedges to other types of vegetation. An estimate for incidental sediment ingestion during feeding is included. The muskrat dietary proportions are assumed to be 90% for emergent vegetation and 10% incidental ingestion of sediment.

The estimated dose (mg/kg-d) was calculated for the muskrat assuming ingestion of contaminated aquatic vegetation and incidental ingestion of sediment. The dose model consists of the dose equation (modified from USEPA 2000) and dose parameters below.

$$\text{Dose} = [C_s \times NIR_s) + (C_v \times NIR_v)] \times AUF$$

Dose Model Parameters for Muskrat

Parameter	Definition	Value
C_s	Concentration of the metal in the surface sediment	mg/kg
C_v	Concentration of the metal in the vegetation consumed (wet weight)	mg/kg
NIR_s	Normalized ingestion rate of sediment (dry weight)	0.0044 kg/kg-d
NIR_v	Normalized ingestion rate of vegetation (wet weight)	0.31 kg/kg-d
AUF	Area use factor	1

The concentration terms (C_s, C_v) were developed from the 2011 sediment sampling data (as described above for estimating metal and PCB residues in dietary items and sediment). The maximum and central tendency concentrations of each COC from the 2011 sediment sampling (0-0.5 foot interval) were used as the concentration terms for dose modeling.

The normalized ingestion rate terms (NIR_s and NIR_v), normalized to body weight, were estimated based on regulatory guidance in the Wildlife Exposure Factors Handbook (USEPA 1993). The NIR_s and NIR_v terms were calculated based on the normalized ingestion rate of food multiplied by the portion of the entire diet as either sediment or vegetation. The average normalized ingestion rate of food was set at a value of 0.34 kg/kg-d based on the wet weight (ww) of consumed greens for an adult male muskrat (US EPA 1993, Svihla and Svihla 1931).

Normalized ingestion rate of aquatic vegetation (NIR_v): 0.31 kg ww/kg-d
This value was calculated as described above: (0.34 kg ww/kg-d)(0.90) = 0.31 kg ww/kg-d
Normalized ingestion rate of sediment (NIR_s): 0.0044 kg dw/kg-d
This value was calculated as described above: (0.34 kg ww/kg-d)(0.10)(0.13)=0.0044 kg dw/kg-d
Note: EPA (1993, p.4-22) recommends that this term be expressed on a dry weight basis. Therefore, a wet weight to dry weight conversion factor (0.13) was used. This factor was derived by assuming that

87% of the consumed greens are water and the remaining portion is the dry weight. EPA (1993, p. 4-14) presents this water content for aquatic macrophytes, the main staple for muskrats.

The Area Use Factor is determined by multiple factors that influence territory size or distance to foraging sites. These include habitat preference, prey abundance, and population density. The area use factor for the muskrat was set to 1 because the size of an individual muskrat's range (0.17 ha, USEPA 1993) is very small relative to size of Round Lake (50.61 ha).

Waterfowl (Mallard). Potential mallard exposure to site COCs was evaluated for three pathways: aquatic vegetation ingestion, benthic invertebrate ingestion, and incidental ingestion of sediment. At Round Lake, at least some of the COCs may be present in the sediment or food of these animals. As described above, the primary food resources for mallard ducks are aquatic vegetation and benthic invertebrates. In summer, fall, and winter, mallards feed primarily on seeds and aquatic vegetation and occasionally on invertebrates associated with leaf and litter wetlands. In late spring and early summer, females shift from an herbivorous diet to a diet containing more invertebrates in order to obtain more protein for reproduction. Juvenile mallards consume mainly invertebrates for the protein as well during the late spring and early summer. The annual diet of the mallard was used to estimate exposure to aquatic vegetation and benthic sediment organisms like invertebrates. Also, an estimate for incidental sediment ingestion is included in order to account for that taken up during feeding (USACHPPM, 2004).

The estimated dose (mg/kg-d) was calculated for the mallard assuming ingestion of contaminated aquatic vegetation, invertebrates, and incidental ingestion of sediment. The dose model consists of the dose equation below (modified from USEPA 2000) and the following description of the dose parameters.

$$\text{Dose} = [(C_v \times NIR_v)+(C_i \times NIR_i)+(C_s \times NIR_s)] \times AUF$$

Dose Model Parameters for Mallard

Parameter	Definition	Value
C_v	Concentration of the metal in the vegetation consumed (wet weight)	mg/kg
C_i	Concentration of the metal in the invertebrate consumed (wet weight)	mg/kg
C_s	Concentration of the metal in the surface sediment	mg/kg
NIR_{v3}	Normalized ingestion rate of vegetation (wet weight)	0.155 kg/kg-d
NIR_{i3}	Normalized ingestion rate of invertebrate (wet weight)	0.138 kg/kg-d
NIR_{v9}	Normalized ingestion rate of vegetation (wet weight)	0.309 kg/kg-d
NIR_{i9}	Normalized ingestion rate of invertebrate (wet weight)	0.069 kg/kg-d
NIR_s	Normalized ingestion rate of sediment (dry weight)	0.007 kg/kg-d
AUF	Area use factor	0.17

The concentration terms (Cs, Ci, Cv) were developed from the 2011 sediment sampling data (as described above for estimating metal and PCB residues in dietary items and sediment). The maximum and central tendency concentrations of each COC from the 2011 sediment sampling (0-0.5 foot interval) were used as concentration terms for dose modeling.

The proportion of the diet from rooted emergent aquatic vegetation/plants (wet weight) is estimated to be 30% during the 3 months of the year in late spring and early summer and 60% for the other 9 months in the year, based upon the data presented in the Wildlife Exposure Factors Handbook (USEPA 1993) for annual use pattern.

The proportion of the diet that is benthic invertebrates (wet weight) is 60% during the 3 months of the year in late spring and early summer and 30% for the other 9 months in the year, based upon data presented in the Wildlife Exposure Factors Handbook (USEPA 1993) for annual use patterns.

The proportion of the diet that is sediment (dry weight) is 10%, this value has been selected from the values presented in the Wildlife Exposure Factors Handbook (USEPA 1993) for waterfowl.

The normalized ingestion rates for aquatic vegetation, invertebrates, and sediment (NIR_v, NIR_i, and NIR_s) for the mallard were calculated based on the normalized ingestion rate of food multiplied by the portion of the entire diet that is vegetation, invertebrates, or sediments from above. The average normalized ingestion rate of food for the mallard is not provided in the Wildlife Exposure Factors Handbook (USEPA 1993). Therefore, an average normalized ingestion rate was estimated using the allometric equation for all birds; by using a body weight of 1.2 kg, the normalized ingestion rate is calculated to be 0.0655 kg/kg-d (as kg dry matter per day). A range of 1.043-1.246 kg has been reported in the Wildlife Exposure Factors Handbook (USEPA 1993) as a year round estimate for both sexes. Using the same allometric equation and the range of body weight reported by USEPA (1993), then a range of 0.0598-0.0671 kg/kg-d is obtained. The high end of the range is used for the mallard.

Normalized ingestion rate of vegetation for 3 months of the year (NIR_{v3}): 0.155 kg ww/kg-d
This value was calculated as described above: [(0.0671 kg dw /kg-d)/(0.13)]*(0.30) =0.155 kg ww/kg-d.
Note: This term must be expressed on a wet weight basis so a dry weight to wet weight conversion factor (0.13) was used. This factor was based on the 87% water content of aquatic macrophytes in Table 4-2 of the Wildlife Exposure Factors Handbook.

Normalized ingestion rate of invertebrates for 3 months of the year (NIR_{i3}): 0.138 kg ww/kg-d
This value was calculated as described above: [(0.0671 kg dw/kg-d)/(0.29)]*(0.60)=0.138 kg ww/kg-d.
This term must be expressed on a wet weight basis so a wet weight to dry weight conversion factor (0.29) was applied to the normalized ingestion rate. This factor was derived by assuming that 71% of the consumed benthic organisms are water and the remaining portion is the dry weight.

Normalized ingestion rate of vegetation for 9 months of the year (NIR_{v9}): 0.309 kg ww/kg-d
This value was calculated as described above: [(0.0671 kg dw /kg-d)/(0.13)]*(0.60) =0.155 kg ww/kg-d.
Note: This term must be expressed on a wet weight basis so a dry weight to wet weight conversion factor (0.13) was used. This factor was based on the 87% water content of aquatic macrophytes in Table 4-2 of the Wildlife Exposure Factors Handbook.

Normalized ingestion rate of invertebrates for 9 months of the year (NIR_{i9}): 0.069 kg ww/kg-d
This value was calculated as described above: [(0.0671 kg dw/kg-d)/(0.29)]*(0.30)=0.138 kg ww/kg-d. This term must be expressed on a wet weight basis so a wet weight to dry weight conversion factor

(0.29) was applied to the normalized ingestion rate. This factor was derived by assuming that 71% of the consumed benthic organisms are water and the remaining portion is the dry weight.

Normalized ingestion rate of sediment for 12 months (NIR_s): 0.007 kg dw /kg-d
This value was calculated as described above: (0. 0671 kg dw/kg-d)(0.1)= 0.00671 kg dw/kg-d.

The Area Use Factor is determined by multiple factors that influence home range size. These include the habitat preference and abundance, prey abundance, and population density. Each pair of mallards uses a home range, and the drake commonly establishes a territory that he defends against others. Range size is controlled by habitat, in particular the type and distribution of water features and population density (CH2M Hill 2001). In wetlands and rivers in Minnesota, Kirby et al. (1985, as cited in USEPA 1993) reported home ranges of 540 ha and 620 ha for females and males, respectively. Because Round Lake is located in a developed area with more natural aquatic habitat in close proximity (more than 13 water bodies suitable for waterfowl habitat within a 3 mile radius), an area use of 300 ha for the mallard was used to generate an AUF for Round Lake (50.61 ha). The Round Lake AUF for the mallard was 0.17.

4.4.4 Bioaccumulation Algorithms

The wildlife exposure model is designed to account for water, sediment, and food ingestion exposures, where the exposed animal receives the majority (if not all) of its total exposure from potential chemical residues in these media. The direct measurement of chemical concentrations in food items of concern is preferred to minimize uncertainty in exposure assessments. However, the potential accumulation into food items of concern must be modeled mathematically since sufficient site-specific data on the bioaccumulation of metals in food items of concern are not available at Round Lake. This section presents the bioaccumulation models used to predict the residue levels of the COCs that may accumulate in biological tissues.

Site-specific data or bioaccumulation models do not exist that are specific to each type of food item. To deal with this uncertainty, various food items can be grouped together into categories for which data or models may exist. The following food item groups were designated for evaluation of the bioaccumulation of COCs that may be consumed by the receptors assessed.

- Tissues of aquatic vegetation (e.g., shoots, stems, leaves of marsh grasses and cattails)
- Aquatic, epibenthic invertebrates (e.g., midges, snails)
- Fish (e.g., minnows, bullhead)

***Aquatic Vegetation*.** The uptake of metal COCs by aquatic vegetation is potentially important for animals that may consume vegetation growing in contaminated areas. Exposure to muskrats and waterfowl include possible contributions from dietary vegetation. Plants selectively accumulate inorganic chemicals from solution; and some chemicals are selectively excluded from seeds and fruits (Alloway et al. 1990). Accumulation of inorganic chemicals in leaves, stems, and roots of plants can be modeled using soil to plant concentration factors, as shown in the equation below.

$$Cv = Cs \times BAFv \times 0.12$$

where, Cv is the vegetation tissue wet-weight concentration of the chemical in units of mg chemical/kg tissue, Cs is the concentration of the chemical in the sediment in units of mg/kg, BAFv is the dry-weight based soil-to-plant bioaccumulation factor, which is unitless, and the constant (0.12) is a dry weight to wet weight conversion factor based on a water content of 88% (USEPA 1993). The BAF is the ratio of the concentration of the chemical in the organism to the concentration of the chemical in the sediment.

The roots and rhizomes of aquatic macrophytes are generally the primary areas of metal accumulation (Wang et al 1997). COC residues in rooted emergent aquatic vegetation (i.e., macrophyte) were quantitatively estimated for the metal and PCB COCs.

CH2M Hill (2001) summarized concentration factors for terrestrial plant tissues in support of the Army Risk Assessment Modeling System (ARAMS). When values for aquatic plants were not found, the median bioaccumulation factors from CH2M Hill (2001) are used assuming that the plants of interest for this assessment are vascular and that concentrations can be modeled using the equations developed for terrestrial plants. Central tendency values (e.g., medians) were used for this assessment, rather than maximum values or 95% upper prediction limits because high-end normalized ingestion rates for the receptors were selected. If high-end estimates are used for all parameters, then the resulting output is much higher than expected exposures in the field.

Literature indicates that macrophyte roots can accumulate larger concentrations of Cd, Cr, Cu, Pb, and Zn than other plant organs (USACHPPM, 2004). For aquatic plants, the accumulation of divalent cations strongly depends on levels of calcium both within the environment (water column and sediment) and within the plant tissue. With higher calcium levels in plant tissues, accumulation and deposit of other divalent cations may be blocked. In addition, TOC content of the sediment influences bioavailability of metals with high TOC rendering metals less bioavailable. Bioaccumulation in aquatic vegetation of COCs from sediment was estimated based on the sediment concentrations from the 2011 Round Lake sediment sampling. Sediment to aquatic plant BAFs for cadmium, chromium, and PCBs were obtained from EPA (1999) and BAFs for copper, lead, and zinc were obtained from CH2M Hill (2001) and are provided in Table 14.

Table 14. Bioaccumulation Factors for final COCs in Round Lake sediments.

COC	Sediment to aquatic plant (dw)	Sediment to benthic invertebrate
Cadmium	0.36 – USEPA, 1999	0.6 dw – Bechtel Jacobs, 1998
Chromium	0.01 – USEPA, 1999	0.1 dw – Bechtel Jacobs, 1998
Copper	0.54 - CH2MHILL, 2001	0.33 ww - USEPA, 1999
Lead	0.20 – CH2MHILL, 2001	0.63 ww – USEPA, 1999
Zinc	1.0 – CH2MHILL, 2001	0.57 ww – USEPA, 1999
PCB (Aroclor 1016, 1254)	0.01 – USEPA, 1999	0.53 ww – USEPA, 1999

Aquatic, Epibenthic Invertebrates. The benthic invertebrate fraction of the diet for waterfowl and muskrat can consist of aquatic insects, worms, bivalves, and a variety of other organisms. Bioaccumulation in benthic invertebrates was quantitatively estimated for each COC. Benthic invertebrates are exposed to metals and other inorganics through direct contact with sediment , ventilation of overlying or pore water, or ingestion of sediment and/or food particles (Rand 1995). These chemicals can be adsorbed to the dermal surface or assimilated into the body. Data for estimating tissue concentration of inorganics in aquatic (benthic) invertebrates are limited. Cadmium and chromium may accumulate to some degree and Bechtel Jacobs has developed a database for metals (Bechtel Jacobs 1998). Their report and database contains contaminant uptake data from published and unpublished literature; where they developed BAFs and regression equations for estimating metal concentrations in benthic invertebrates. In addition, EPA (1999) calculated BAFs for other metals including copper, lead, zinc, and PCBs. Table 14 presents the selected BAF values for these benthic invertebrates.

In the majority of cases, BAF values are provided in units of dry-weight. For these cases, the method for calculating the tissue residue is shown in the equation below;

$$Ci = Cs \times BAFi \times 0.29$$

where, Ci is the invertebrate tissue wet weight concentration of the chemical in units of mg chemical/kg tissue, Cs is the concentration of the chemical in the sediment in units of mg/kg, BAFi is the soil-to-invertebrate bioaccumulation factor, which is unitless, and the constant (0.29) is a dry weight to wet weight conversion factor, the inverse of the water content of 71% for invertebrates (USEPA 1993). The conversion factor is removed to calculate the wet-weight residue in cases where the BAF is provided in units of wet-weight.

4.4.5 Dose Algorithm for PCBs in Fish Consumed by Piscivorous Mammals and Birds

The potential risk to piscivorous mammals (mink) and birds (Great blue heron, Belted kingfisher, Bald eagle) from exposure to PCBs through bioaccumulation in the food chain, primarily by consumption of fish (brown and black bullhead), was evaluated. Sampling of fish from Round Lake by Minnesota DNR in December 2012 and subsequent tissue analyses provided whole fish and filet data for PCBs (see Table 4 above).

The estimated dose (mg/kg-d) of PCBs was calculated for piscivorous mammal (mink) and birds (great blue heron, belted kingfisher, and bald eagle) assuming ingestion of PCB contaminated fish.

A basic ingestion model and the dose parameters below are used to assess ingestion of fish by mink and birds.

$$Dose = (C_f \cdot NIR_f)$$

Dose Model Parameters for Piscivorous Mammal and Bird

Parameter	Definition	Value
C_f	Concentration of PCB in fish (whole and filet, wet weight)	mg/kg
NIR_m	Normalized mink ingestion rate of fish tissue (wet weight)	0.22 kg/kg-d
NIR_h	Normalized great blue heron ingestion rate of fish tissue (wet weight)	0.18 kg/kg-d
NIR_k	Normalized belted kingfisher ingestion rate of fish tissue (wet weight)	0.0672 kg/kg-d
NIR_e	Normalized bald eagle ingestion rate of fish tissue (wet weight)	0.371 kg/kg-d

The normalized ingestion rate (NIR) for mink was estimated based on regulatory guidance in the Wildlife Exposure Factors Handbook (USEPA 1993). The NIR_m term equals 0.22 kg/kg-d of aquatic food for adult male mink in the diet (USEPA 1993). It is assumed for this assessment that the aquatic food is 100% fillet of fish as mink are known to eat flesh only.

The NIR_h for the great blue heron of aquatic food was set at a value of 0.18 kg kg/kg-d in the Wildlife Exposure Factors Handbook (USEPA 1993). For the great blue heron, it is assumed that 100% of the aquatic food is whole fish.

The NIR_k for the belted kingfisher of aquatic food was set at a value of 0.0672 kg/kg-d in the Wildlife Exposure Factors Handbook (USEPA 1993). For the belted kingfisher, it is assumed that 100% of the aquatic food is whole fish.

The NIR_e for the bald eagle of aquatic food was set at a value of 0.371 kg/kg-d in the Wildlife Exposure Factors Handbook (USEPA 1993). For the bald eagle, it is assumed that 100% of the aquatic food is whole fish.

4.5. Risk Characterization for the Benthic Organism Assessment Endpoint for the Current Use Scenario

The goal of the Superfund is to reduce ecological risk to levels “that will result in the recovery and/or maintenance of healthy local populations/communities of ecological receptors that are or should be present at or near the site” (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 2, OSWER Directive 9285.7-28P, October 7, 1999). A principle in ecological risk assessment is that Superfund remedies should generally be designed to protect local populations and communities of biota and not to protect organisms on an individual basis (except in the instance of the presence of T&E species) (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999). It should be noted that no endangered, threatened or special concern benthic species are known to inhabit Round Lake. In characterizing ecological risks, lines of evidence are used to evaluate risk including toxicity tests, plant and animal residue data, bioavailability factors, assessment of existing impacts at site, media chemistry, reference site data, and risk calculation comparing exposures estimated for the site with toxicity values from literature. Using the lines of evidence approach, effects on individuals and group of individuals can be extrapolated to local populations and communities (The Role of Screening-Level Risk Assessments and Refining

Contaminants of Concern in Baseline Ecological Risk Assessments Eco Risk Assessment Guidance, page 3, EPA OSWER Guidance 9345.0-14, EPA 540/F-01/014, June 2001). EPA notes "[t]he performance of multi-year field studies at Superfund sites to try to quantify or predict long-term changes in local populations is not necessary for appropriate risk management decisions … Data from discrete field and laboratory studies, if properly planned and appropriately interpreted, can be used to estimate local population or community-level effects." EPA points out in guidance that "Typically, no one line of evidence can stand on its own" (Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, page 4-6, EPA OSWER Guidance 9285.7-25, EPA 540-R-97-006, June 1997). When ecological risk assessments involve more than one line of evidence, strength of evidence approach is used, using professional judgment, to integrate the information to support a conclusion. When some lines of evidence are conflicting, professional judgment is used to determine which data should be considered more reliable or relevant. EPA acknowledges that unlike the detailed guidelines and risk range established for characterizing human health risk, detailed guidelines for site-specific ecological risk assessment do not exist. Ecological risk is evaluated qualitatively and quantitatively, which inherently has some uncertainty. The potential risk to the benthic organisms inhabiting Round Lake from exposure to the final COCs in sediments are evaluated using USEPA approaches in OSWER directives and guidance established for CERCLA activities.

The measures of effect used to evaluate the assessment endpoint for benthic organisms are as follows:

- Quantitative risk calculations
- Comparison to effects-based benchmarks for benthic organisms

Supporting lines of evidence that will be evaluated are:

- Sediment toxicity testing results and published literature data
- Metal bioavailability tests and published literature data
- Benthic community structure surveys.

4.5.1 Measures of effect

The first measure of effect used as a link to determine causality of effects from the final COCs to these receptors is an HQ calculation. An HQ ratio indicates whether or not an estimated exposure is greater than the selected toxicological criterion. They are commonly used in risk assessments as a means of confidently identifying low risk situations. Table 15 provides a comparison of the HQ calculations from the Tier II ERA with the HQ calculations using the 2011 sediment concentration data and the no-observed adverse effect level (MPCA SQT I Benchmark). . It should be noted that the HQs for the Tier II ERA (using 1992 sediment concentration data) were calculated based on a low-effect benchmark . Based on the 2011 sediment sampling and MPCA SQT I Benchmark, the HQs for cadmium, chromium, lead, zinc, and PCBs are <1, indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to cadmium, chromium, lead, zinc and PCBs in sediments at depths of 0 – 2 feet.

Table 15. A comparison of the Hazard Quotients for benthic organisms from the Tier II ERA and using the 2011 sediment data.

COC	Tier II ERA (2004)[a]	2011 Sediment Sampling[b]
Cadmium	<1	<1
Chromium	2	<1
Copper	7	1.3
Lead	<1	<1
Zinc	3	<1
PCBs (1992)	<1	<1[c]
PCBs (1987 & 1990)	70	<1[c]

[a]Hazard Quotient calculated using central tendency concentrations of the 1992 sediment data and low-effect sediment benchmarks (as presented in Appendix O of the Tier II ERA).
[b]Hazard Quotient calculated using central tendency concentrations of the 2011 sediment data (0-2 ft, 397 samples) and MPCA SQT I benchmarks (no-observed-adverse effect level).
[c]For PCBs, the hazard quotient is calculated based on the sum of Aroclors that were detected.

Table 16 presents the HQ calculations for the 2011 sediment sampling results at three depth intervals of 0.0-0.5', 0.0-1.0' and 0.0-2.0' using central tendency concentrations and the threshold-effect benchmark (MPCA SQT I). The HQs for chromium, lead and PCBs are <1 at all depth intervals, indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to these COCs in sediments at Round Lake. The hazard quotients for cadmium and zinc slightly exceed 1 for the 0.0 - 0.5' depth interval, but are <1 at depths up to 1 – 2 feet. These results indicate a possible causal link of potential adverse effects to benthic organisms from exposure to cadmium and zinc at depths of 0.0 – 0.5 feet. The HQ for copper at these depths ranges from 1.3 – 3.2, indicating a possible causal link of potential adverse effects to benthic organisms from exposure to copper in sediments at Round Lake, especially at the 0.0 – 0.5 foot depth interval.

Table 16. Hazard Quotients for benthic organisms using 2011 sediment data at three depth intervals

Sediment Depth Interval	COC					
	Cadmium	Chromium	Copper	Lead	Zinc	PCBs[b]
0.0 - 0.5 ft	1.1	0.8	3.2	0.8	1.5	0.8
0.0 - 1 ft	0.7	0.6	1.8	0.5	0.9	0.6
0.0 - 2 ft	0.5	0.5	1.3	0.4	0.8	0.5

[a]Hazard Quotient calculated using central tendency concentrations of the 2011 sediment data (0.0-0.5 ft, 134 samples; 0.0-1 ft, 268 samples; 0-2 ft, 397 samples) and MPCA SQT I benchmarks.
[b]For PCBs, the hazard quotient is calculated based on the sum of Aroclors that were detected.

The second measure of effect evaluated is a comparison of the 2011 concentration data of the final COCs to effects-based benchmarks. There are no Minnesota or federal promulgated standards for sediment. The NCP recognizes that in the absence of an ARAR for a media, non-promulgated advisories, criteria, or guidance may be useful in determining what is protective in developing CERCLA remedies. The NCP classifies such advisories, criteria, and guidance as to-be-considered guidance or TBC. The identification and use of TBCs are not mandatory under CERCLA and are only to be used on an "as appropriate" basis (NCP Final Rule preamble, 55 FR 8744, March 8, 1990). TBCs can be and often are preliminary remediation goals (PRGs) based on readily available information; however, such goals are modified throughout the RI/FS process. Final remedial goals are determined when the remedy is selected considering factors in the NCP, including environmental evaluations (40 CFR 300.430(e)(2)(i); NCP Final Rule preamble, 55 FR 8712-8713, March 8, 1990).

The potential TBC guidance values for sediments in Round Lake are presented in Table 17. However, it should be noted that exceedance of criteria or standards does not necessarily indicate causation, because the regulatory values are intended to be safe levels and not a threshold signifying absolute occurrence of adverse effect (Suter et al. 2002). Benchmarks found in published literature may be biased in that they are generally based on laboratory studies in which the forms of the chemical used in the tests are likely to be more toxic than that found at contaminated sites (in the field), combined toxic effects are not observed, the test species and test media may not be representative of the study site, and lab test conditions may not be representative of field conditions. When laboratory data is extrapolated to the field, environmental and ecological processes that may affect the sensitivity of the organism to the chemicals are not considered.

For sediment, TBCs originate from several sources; the following sources provide TBCs for Round Lake:

- *Guidance For The Use And Application Of Sediment Quality Targets For The Protection Of Sediment-Dwelling Organisms in Minnesota*, MPCA Document Number: tdr-gl-04, February 2007.
- *A Preliminary Evaluation of Sediment Quality Assessment Values for Freshwater Ecosystems*, Smith et. al. 1996
- *Approach to the Assessment of Sediment Quality in Florida Coastal Waters*, Florida Department of Environmental Protection, Office of Water Policy, MacDonald 1994USEPA 2000
- *Incidence of Adverse Biological Effects within Ranges of Chemical Concentrations in Marine and Estuarine Sediments*, Long et.al. 1995
- *Guidance on Remedial Actions for Superfund Sites with PCB Contamination*, EPA OSWER Directive 9355.4-01, EPA/540/G-90/007, August 1990

Table 17. Sediment TBC Values for final COCs at Round Lake.

Final COC (Tier II ERA)	TBC Guidance Value (in mg/kg)							
	Threshold Effect Level				Low Effect Level			
	SQT I[a]	TEL-FW[b]	TEL-ME[c]	ER-L[d]	SQT II[a]	PEL-FW[b]	PEL-ME[c]	ER-M[d]
Cadmium	0.99	1.0 (PQL)	1.0 (PQL)	1.2	5.0	3.53	4.21	9.6
Chromium	43	37.3	52.3	81	110	90	160	370
Copper	32	35.7	18.7	34	150	197	108	270
Lead	36	35	30.2	47	130	91.3	112	218
Zinc	120	123	124	150	460	315	271	410
Total PCBs	0.06	0.033	0.022	0.023	0.68	0.277	0.189	0.18

[a]MPCA Sediment Quality Targets, February 2007. SQT I values represent contaminant concentrations below which harmful effects on benthic invertebrates are unlikely. SQT II values represent contaminant concentrations above which harmful effects on benthic organisms are likely.
[b]Smith et al. 1996. TEL-FW are freshwater threshold effect levels; PEL-FW are freshwater probable effect levels.
[c]MacDonald 1994. TEL-ME are marine/estuarine threshold effect levels; PEL-ME are marine/estuarine probable effect levels.
[d]Long et al. 1995. ER-L are low effects range values; ER-M are median effects range values.

The potential TBC guidance values presented in Table 17 were established by various federal and state agencies using an approach first developed for the National Oceanic and Atmospheric Administration (NOAA) linking biological effects data with contaminant concentrations in field-collected sediments from about 200 marine/estuarine and freshwater environments in the United States. This U.S. National Status and Trends Program (NSTP) approach was originally used by Long and Morgan (1991) to develop informal sediment quality guidelines (SQGs) for a set of trace metals, PCBs, pesticides and polynuclear aromatic hydrocarbons (PAHs). NSTP sediment assessment values are computed from observed correlations between contaminants and effects in sediments containing various mixtures of contaminants. The biological effects data includes data from equilibrium partitioning models, laboratory-spiked sediment toxicity tests, field studies for toxicity and/or benthic community composition. The NSTP approach has been adopted and modified by the Florida Department of Environmental Protection and the Canadian Council of Ministries to develop sediment guideline values, as provided in Table 17. It is noted that sediments with contaminant concentrations that are higher than the recommended sediment quality guidelines only indicate that there is the potential for biological effects to occur. The Sediment Quality Guidelines developed by these agencies do not infer cause-effect relationships. “Exceeding an assessment value may indicate an increased likelihood of toxic effects, but correlation is not proof of cause, and it cannot be assumed that the contaminant present in excess of the assessment value is necessarily responsible for the observed effects.” (Smith et al., 1996; Borgmann, 2003). The guidelines are used to predict the absence or presence of toxicity in field-collected sediments.

Using the NSTP approach, data are segregated into two classes – effect and no effect – based on whether biological effects were observed at the concentrations measured in the sediment sample. Under this approach, Long and Morgan (1991) established that the effects range-low (ER-L) concentration is equivalent to the lower 10^{th} percentile or the low end of the range of concentrations in which effects were observed or predicted. The ER-L represents concentrations above which adverse effects may begin or are predicted among sensitive life stages and/or species or as determined in sublethal tests. The effects range-median (ER-M) concentration is equivalent to the 50th percentile point and represents the concentration above which effects were frequently observed or predicted. Long and Morgan (1991) state that the ER-L and ER-M values are to be used as guidance in evaluating sediment contamination data; however, there is “no intent expressed or implied that these values represent official NOAA standards.”

MacDonald (1994) provides numerical sediment quality assessment guidelines (SQAGs) for Florida coastal waters for the Florida Department of Environmental Protection (see Table 17). The approach used followed that by Long and Morgan in 1991 and derives two guidelines: a threshold effect level (TEL), a concentration below which sediment-associated contaminants are not considered to represent significant hazards to aquatic organisms (minimal effects range) and a probable effect level (PEL), a concentration above which adverse biological effects are usually or always associated (probable effects range). In the range of concentrations between the TEL and the PEL adverse biological effects are possible (possible effects range); however, it is difficult to predict the occurrence, nature, or severity of the effects. Values are to be used to monitor trends in environmental contamination and not intended to be used as sediment quality criteria. The report cautions that bioavailability should be considered along with the SQAGs to prevent the potential for either under- or over-protection of aquatic resources. The preliminary guidelines are broadly applicable in the southeast; however, MacDonald indicated that care should be exercised in applying SQAGs elsewhere in North America.

Long et al (1995) updated and expanded the NOAA database and modified the guideline values from the Long and Morgan study in 1991 by quantifying the percent incidence of adverse biological effects, comparing the guidelines with other data or methods and eliminating freshwater sediment data. This study identified values for the effects range-low (ERL) and effects range-medium (ERM). These values delineated three concentration ranges: concentrations below the ERL value (minimal-effects range) where effects are rarely observed; concentrations equal and above the ERL but below ERM (possible effects range) where effects would occasionally occur; and concentrations equal to and above the ERM (probable-effects range) where effects would frequently occur. ERL and ERM values were derived for nine trace metals, total PCBs, 13 PAHs and two pesticides (see Table 17). The incidence of biological effects was quantified for each of these ranges. The incidences of biological effects for the trace metals in marine and estuarine sediments increased with increasing concentration; however, this pattern was not observed for PCBs. Table 18 provides the percent incidence of effects for the three concentration ranges for trace metals in marine and estuarine sediments. The authors noted that the ERL and ERM values were not normalized to account for the presence of acid volatile sulfide (AVS) or total organic carbon (TOC) concentrations. The bioavailability of the trace metals in sediments are controlled by physical-chemical properties of the sediments, with high AVS concentrations and TOC concentrations in the sediments reducing the bioavailability of the metals. Significant differences in toxicity can occur at similar metal concentrations over relatively small ranges in TOC and/or AVS concentrations. The bioavailability of the metal COCs in Round Lake will be discussed in greater detail below.

Table 18. Percent incidence of effects for ERL and ERM values as reported by Long et al, 1995

Chemical	% < ERL	% ERL - ERM	%> ERM
Cadmium	6.6	36.6	65.7
Chromium	2.9	21.1	95.0
Copper	9.4	29.1	83.7
Lead	8.0	35.8	90.2
Zinc	6.1	47.0	69.8
PCBs	18.5	40.8	51.0

Smith et al. (1996) derived freshwater sediment quality assessment values for recommended sediment quality guidelines for freshwater sediments in Canada using the NSTP approach of Long and Morgan (1991) and Long et al (1995). The authors expanded the original NOAA database to incorporate additional information on the toxicity of chemicals in freshwater sediments from sites throughout North America to form the freshwater Biological Effects Database (BEDs). A threshold effect level (TEL) and a probable effect level (PEL) were derived for eight trace metals, total PCBs, six PAHs and eight pesticides (see Table 17 above). The TEL is calculated as the geometric mean of the lower 15th percentile concentration of the effect data set and the 50th percentile concentration of the no-effect data set. Below this level, the frequency of effects is expected to be less than 25%. The PEL is calculated as the geometric mean of the 50th percentile of the effect data set and the 85th percentile of the no-effect data set. The PEL is the level above which adverse biological effects are expected to occur at a frequency of greater than 50%. The values defined three ranges of chemical concentrations: those that were (1) rarely, (2) occasionally and (3) frequently associated with adverse biological effects. The incidence of adverse biological effects within the three ranges of chemical concentrations defined by the TEL and PEL were determined by Smith et al. (1996) and are presented in Table 19. The incidence of adverse biological effects below the TEL was very low, indicating that the reliability of the TELs was high. The incidence of adverse biological effects above the PELs for all trace metals was less than 50%, indicating that the PELs did not adequately describe the concentration above which adverse biological effects frequently occurred. Measured concentrations higher than PELs only indicate potential for adverse biological effects to occur. As with the sediment quality guideline values derived by Long et al. (1995), the TELs and PELs were not normalized to account for the presence of AVS and TOC in sediments; consequently, the bioavailability of the metals was not considered.

Table 19. Percent incidence of adverse biological effects for TEL and PEL values as reported by Smith et al, 1996

Chemical	% ≤ TEL	% >TEL - <PEL	% ≥ PEL
Cadmium	11	12	47
Chromium	2	19	49
Copper	4	38	44
Lead	5	23	42
Zinc	5	32	36
PCBs	4	40	50

MacDonald et al. (2000) developed consensus-based sediment quality guidelines (SQGs) for contaminants in freshwater sediments using the published SQGs. The SQGs were grouped into two categories. The threshold effect concentration (TEC) identifies contaminant concentrations below which harmful effects on sediment-dwelling organisms are not expected. The consensus-based TECs were derived using values developed by other investigators for federal and state agencies in the United States and Canada, as described above; including the freshwater TELs (Smith et al., 1996) and ER-Ls [from Long and Morgan (1991) that included both marine/estuarine and freshwater sediment data], as well as lowest effect levels (Persaud et al., 1993), minimal effect thresholds (EC and MEN-VIQ, 1992) and sediment quality advisory levels (USEPA, 1997). The consensus-based TEC was calculated by determining the geometric mean of these values. The probable effect concentration (PEC) identifies contaminant concentrations above which adverse effects are expected to occur more often than not. The consensus-based PECs were also derived using values developed by other investigators for federal and state agencies in the United States and Canada; including the freshwater PELs (Smith et al., 1996) and ER-Ms [from Long and Morgan (1991) that included both marine/estuarine and freshwater sediment data], as well as severe effect levels (Persaud et al., 1993) and toxic effect thresholds (EC and MEN-VIQ, 1992). The consensus-based PEC was calculated by determining the geometric mean of these values. SQGs were developed for eight trace metals, ten PAHs, total PCBs and nine pesticides. The reported incidence of effects provided an accurate basis for predicting toxicity above the PEC and the absence of toxicity below the TEC (see Table 20). However, the authors indicate that the consensus-based SQGs are directly relevant for assessing freshwater sediments that are influenced by multiple sources of contaminants and reflect the toxicity of mixtures of sediment-associated contaminants (MacDonald et al., 2000).

Table 20. Percent incidence of toxicity for TEC and PEC values as reported by McDonald et al, 2000

Chemical	% ≤ TEC	% >TEC - <PEC	% ≥ PEC
Cadmium	19.6	44.6	93.7
Chromium	28	64.4	91.7
Copper	17.7	64	91.8
Lead	18.4	53.6	89.6
Zinc	18.4	60.9	90
PCBs	11.1	31	82.3

In 2007, The Minnesota Pollution Control Agency (MPCA) adopted the consensus-based SQGs (TECs and PECs) developed by MacDonald et al. (2000) as recommended sediment quality targets (SQTs) (MPCA, 2007) to be used throughout the state (see Table 17). The TEC was used as the MPCA Level I SQT defined to identify contaminant concentrations below which harmful effects on sediment dwelling organisms are unlikely. The PEC was used as the MPCA Level II SQT defined to identify contaminant concentrations above which harmful effects on sediment dwelling organisms are likely to be observed. The SQTs are based on values developed for the St. Louis River in northeastern Minnesota to protect benthic invertebrates (Crane *et al*. 2000, 2002; Crane and MacDonald 2003), but can be used throughout the state as benchmark values for making comparisons to surficial sediment chemistry measurements. The SQTs provide useful tools for making sediment management decisions, especially when considered as part of a weight-of-evidence approach that includes other sediment quality indicators; such as, geochemical characteristics, sediment toxicity, benthic invertebrate community structure and tissue residue chemistry (Crane *et al*. 2000, 2002a; Crane and MacDonald 2003). As discussed in the supporting paper, the range of concentration between the SQT I (TEC) and the SQT II (PEC) is not intended to indicate if the concentration is toxic or nontoxic. Since concentrations below the SQT I are unlikely to result in harmful effects on sediment dwelling organism and values up to the SQT II are not indicative of toxicity, the SQT II would be the appropriate value to use to predict harmful effects on ecological receptors. It should be noted that the SQT values do not consider the potential bioavailability of the contaminants to aquatic organisms.

MPCA guidance specifically notes several considerations in applying the SQTs:

- Applicability of the SQTs in sediment assessments is increased when used in conjunction with other sediment assessment tools such as sediment chemistry, sediment toxicity testing, bioaccumulation studies, and effects on *in situ* benthic invertebrates.
- Variations in physical, chemical, biological factors in the sediment environment, such as highly modified depositional systems will result in higher uncertainty in applying the SQTs;
- Where additional assessment phases are conducted (i.e., sediment toxicity tests, benthological surveys, and bioaccumulation assessments) SQTs are used in conjunction with these other tools to make decisions about the spatial and temporal extent of contamination and the need for remediation;
- Numerical SQTs should not be regarded as blanket values of regional sediment quality; rather variations in environmental conditions among sites may necessitate the need for modifications of the SQTs to reflect local conditions;
- Substances that occur at concentrations above the Level I SQT but below the Level II SQT should be considered moderate concern; and,
- Chemicals not positively correlated to the results of toxicity tests should be considered a relatively lower priority.

The threshold effect levels presented in Table 17 represent a level below which toxicity is rarely observed from chronic exposure. This level may be appropriate if protection of the individual ecological receptor is the goal; however, a principle in ecological risk assessment is that Superfund remedies should generally be designed to protect local populations and communities of biota and not to protect organisms on an individual basis, except in the instance of the presence of T&E species (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999). Even in the situation of the land use as a wildlife refuge, other Superfund Sites that have been designated as a national wildlife refuge have based unacceptable ecological risk on the population-level effects (See *Corrective Action Decision/Record of Decision for Rocky Flats Plant (USDOE) Peripheral Operable Unit and Central Operable Unit*, September 2006). It should be noted that no endangered, threatened or special concern benthic species inhabit Round Lake. Due to this principle in ecological risk assessment, the low-effect levels would be more appropriate for use as TBC at this site; consequently, the low-effect level values (SQTII, PEL-FW, PEL-ME and ER-M) will be used to evaluate potential risks to benthic organisms from exposure to the contaminants in sediments at Round Lake.

Table 21 and Figures 8 – 10 present the results of the comparison of the 2011 sediment concentration of the metals and PCBs (geometric mean of the lognormal data) to the low-effect TBC guidance values in Table 17 above, showing the percentage of samples that exceed the benchmark at the depths of 0.0 to 0.5 feet, 0.0 to 1.0 feet and 0.0 to 2.0 feet. This comparison shows that the percentage of sediment samples exceeding the low-effect benchmarks decreases with depth for all COCs. A relative ranking of the COC exceedances is as follows: copper > zinc > cadmium > PCBs , lead and chromium. Chromium, lead and PCBs exceed the benchmarks in ≤ 12% of the samples at all depths, indicating that these COCs are detected at lower concentrations and less frequently in the sediments of Round Lake. This further substantiates the results of the hazard quotient calculations of <1 at all depths for chromium, lead and PCBs (see Tables 15 and 16), indicating that exposure of benthic invertebrates to chromium, lead and PCBs in the sediments of Round Lake would be limited.

To perform a more in-depth analysis of the actual distribution of the trace metals and PCBs in the sediments of Round Lake and the subsequent potential for adverse effects to benthic organisms, the concentration data from each of the 200 x 200 foot grids at the 0.0 to 0.5 foot depth is compared to the MPCA SQT II benchmark. This value was selected from the available TBC guidance values as the most relevant and appropriate value for comparison in Round Lake; since, the SQTs were developed specifically for use in the state as opposed to the other TBCs that are intended for use in other geographical areas [i.e., Canada (PEL-FW) or the Southeast (PEL-ME)] or focus on marine or estuarine environments (ER-M). As discussed above, the SQT s were developed using the consensus-based SQGs derived by McDonald *et. al.* (2000). These values were supplemented by other published effects-based freshwater SQGs. The SQTs were developed by MPCA to be used as indicators to assess sediment quality throughout the state of Minnesota. Table 22 presents the grid by grid concentration that exceeded the SQT II for each of the final COCs at the 0.0 to 0.5 foot depth. This depth profile was selected because the results of the quantitative risk calculations show that the HQs for chromium, lead and PCBs are <1 at all depth intervals (0 to 2 feet), the HQs for cadmium and zinc slightly exceed 1 at the 0.0 to 0.5 foot depth interval, but are <1 at depths up to 1 to 2 feet, and the HQ for copper is 3.2 at the 0.0 to 0.5 foot depth, but only slightly exceeds 1 at depths up to 1 to 2 feet. In addition, the percentage of sediment samples exceeding the low-effect benchmarks decreases with depth for all COCs (see Table 21). Wenning et al. (2005) indicate that most of the benthic macroinvertebrates (insect larvae, crustaceans, oligochaetes and mollusks) inhabiting freshwater sediments live in the upper 10 cm (3.9 inches) of the sediment, where they may construct burrows. The USEPA in its Contaminated Sediment Remediation Guidance for Hazardous Waste Sites (2005), states “Typically, the population of benthic organisms is greatest in the top few centimeters of sediment. In fresh waters, the decline in population density with depth is such that the mixed layer is commonly five to 10 cm [2 to 4 inches] deep.” The Interstate Technology and Regulatory Council’s (ITRC) Contaminated Sediments Team indicate that the depth of the bioactive zone is typically defined as 0-6 inches for freshwater sediment (ITRC, 2011).

Table 21. Percentages of COCs exceeding low-effect TBC benchmarks

	COC Sediment Screening					
	Cadmium	**Chromium**	**Copper**	**Lead**	**Zinc**	**PCBs**
0 - 0.5 ft	1.1[a]	33[a]	102[a]	30[a]	175[a]	0.05[a]
% of samples >						
SQT 2	12	4	37	7	18	1
PEL-FW	17	10	31	10	29	10
PEL-ME	13	<1	47	7	32	11
ER-M	5	0	27	1	29	12
0 - 1 ft	0.7[a]	25[a]	59[a]	18[a]	118[a]	0.04[a]
% of samples >						
SQT 2	7	4	26	4	13	1
PEL-FW	10	9	22	6	20	6
PEL-ME	8	1	32	4	22	7
ER-M	3	<1	20	<1	17	7
0 - 2 ft	0.5[a]	22[a]	42[a]	14[a]	92[a]	0.03[a]
% of samples >						
SQT 2	5	3	20	3	10	1
PEL-FW	7	6	17	4	14	4
PEL-M	6	1	24	3	16	5
ER-M	2	<1	15	<1	12	5

[a]Geometric mean of lognormal data (mg/kg)

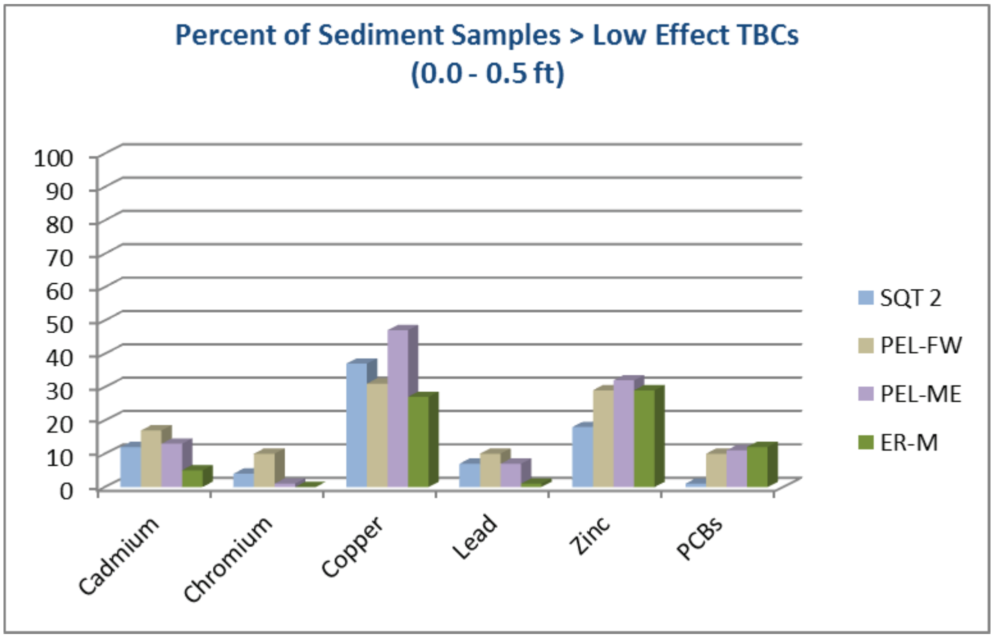


Figure 8. Comparison of 0.0 to 0.5 ft sediment COC data to low effect TBCs.

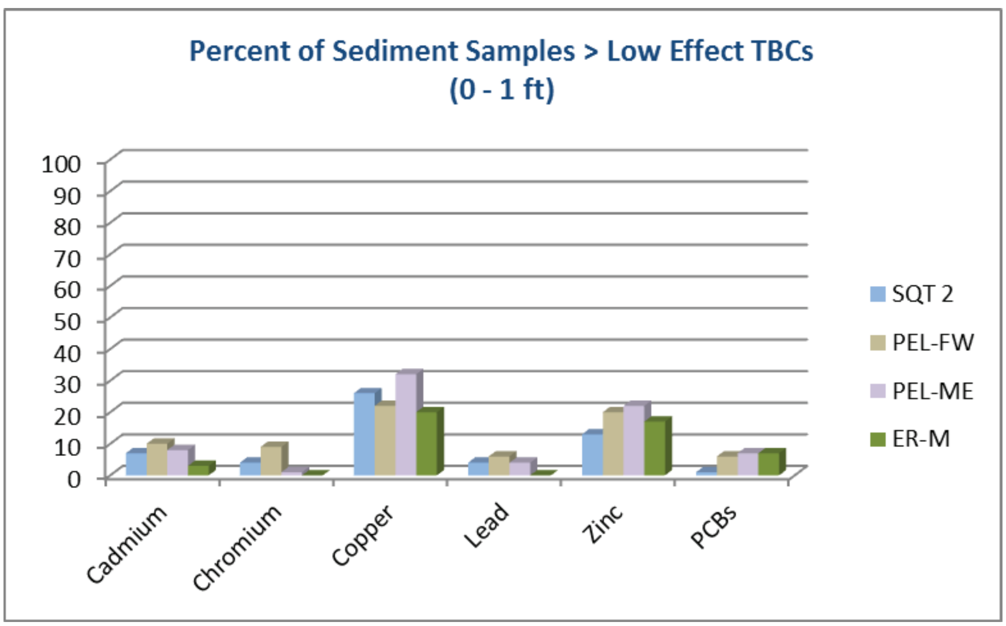


Figure 9. Comparison of 0 to 1 ft sediment COC data to low effect TBCs.

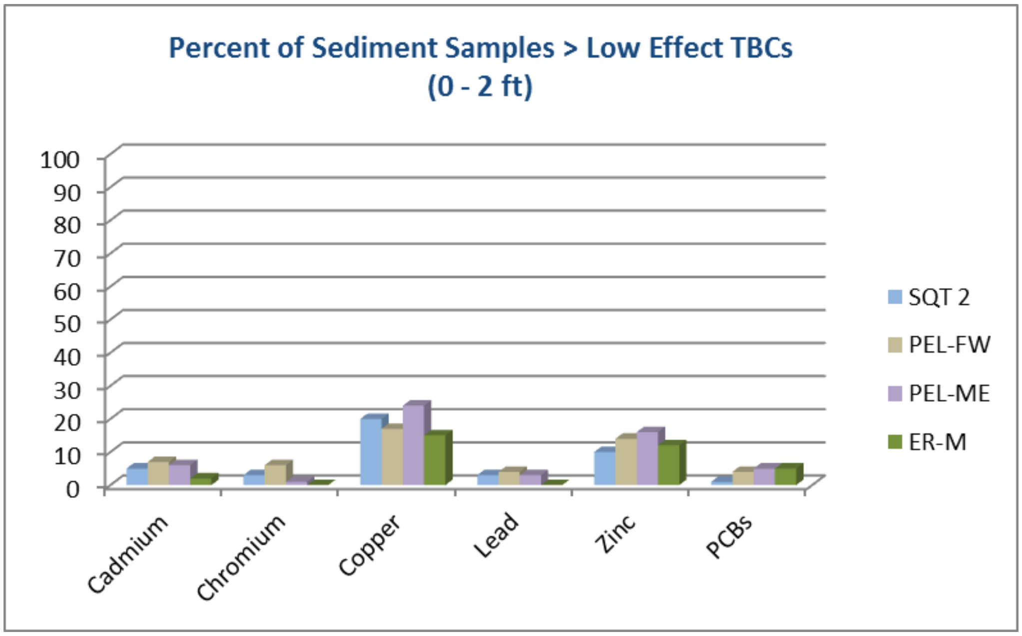


Figure 10. Comparison of 0 to 2 ft sediment COC data to low effect TBCs.

Table 22. Concentration of COCs at the 0.0 to 0.5 foot depth exceeding SQT II by grid location (mg/kg).

Grid	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
1			902		609	
2			473			
3	10.5		582	133	595	
4	6.2		313			
5			179			
6			378			
7	6.2		530			
8	26.6		359	152	614	
9	7.2		621		582	
10	12.7	295	408	195	664	
11	5.5		326	131	593	
12	9.8		685		577	
14			338			
15			250			
16				258		
18	8.6		267	175	841	
19			741		551	0.676
23	6.3		191			
24			717		576	
25			192			
26	8.6	112	317	168	628	
28			459			
32	8.5		568		662	0.887
34			163			
35			154			
38	9.1		317			
39			226			
42			311			
43			158			
46			306		477	
70		118	615		606	
73			346			
74			739		742	
76			241			
81			650		572	
84			453		488	
85	15.3		496	156	642	
86			538		554	0.826
90			154			
94			212			
96			327			
97	14.5	129	686	143	848	

Table 22. Concentration of COCs at the 0.0 to 0.5 foot depth exceeding SQT II by grid location (mg/kg).

Grid	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
98	15.2	115	321	167	591	
104			185			
106			287			
114			466		497	
116			378			
118			378			
120			510		496	
129		133	924		854	
130			210			

Copper concentrations exceed the SQT II value (150 mg/kg) in 50 of 135 grids sampled with concentrations ranging from 154 - 741 mg/kg in 48 grids. The highest copper concentrations were detected in Grids 1 and 129 at 902 and 924 mg/kg, respectively. Copper concentrations in 8 of these grids (Grids 5, 23, 25, 34, 35, 43, 90 and 104) were below the freshwater PEL developed for Canada by Smith et al. (1996; see Table 18). Copper concentrations in an additional 6 grids (15, 18, 39, 76, 94 and 130) were below the ER-M value of 270 mg/kg reported by Long et al. (1995; see Table 18). Smith et al. (1996) reported that the incidence of biological effects at concentrations exceeding the PEL-FW of 197 mg/kg is 44% (see Table 19). Long et al. (1995) reported that the incidence of biological effects is 29% for concentrations equal and above the ERL but below ERM (possible effects range, see Table 18). As provided in Table 5 above, the median lethal concentration (LC50) for copper in sediment for freshwater organisms ranged from 60 mg/kg (dry weight) for 28-day tests with *Chironomus riparius* to 4522 mg/kg (dry weight) for 14-day tests with *Chironomus tentans* and from 128 mg/kg in 28-day tests to 1,078 mg/kg in 10-day tests with *Hyalella azteca*. The most cited study concerning the toxicity of copper to benthic invertebrates in freshwater sediments is a static toxicity test conducted by Cairns et al. (1984). Results in pond sediments indicated a dose-response relationship with LC_{50} values (95% confidence intervals), in mg/kg based on dry-weight sediment copper concentrations, of 1078 (922 – 1,259) for *Hyalella*; 857 (685 – 1,073) for *Chironomus*; and 964 (777 – 1,196) for *Gammarus*. The 95% CI LC_{50} values from this study ranged from 685 to 1,259 mg/kg. Copper concentrations at only two sample locations (Grid 1 at 902 mg/kg and Grid 129 at 924 mg/kg) exhibited concentrations higher than the lowest LC_{50} of 857 mg/kg for *Chironomus,* a predominant species that has been collected in proximity to both locations at Round Lake. The sediment grid locations with the maximum copper concentrations exceeding 685 mg/kg (the low end of the 95% CI) are Grid 12 (685 mg/kg)[high diversity], Grid 97 (686 mg/kg), Grid 24 (717 mg/kg), Grid 74 (739 mg/kg), Grid 19 (741 mg/kg), Grid 1 (902 mg/kg) and Grid 129 (924 mg/kg). Copper appears to exhibit the most potential for adverse effects to benthic invertebrates inhabiting the sediments of Round Lake since the HQ for copper ranges from 1.3 – 3.2, indicating a possible causal link of potential adverse effects to benthic organisms from exposure to copper in sediments at Round Lake, especially at the 0.0 – 0.5 foot depth interval in the grid locations 1, 12, 19, 24, 74, 97 and 129.

Zinc concentrations exceed the SQT II value (460 mg/kg) in 24 of 135 grids sampled with concentrations ranging from 477 – 854 mg/kg. The highest zinc concentrations were detected in Grid 129 (854 mg/kg), Grid 97 (848 mg/kg) and Grid 18 (841 mg/kg). As reported in a study conducted by Borgmann and Norwood (1997) (Table 6 above), the lowest observable effect concentration (LOEC) for zinc in sediment in a 10-week test with *Hyalella azteca* was 83 mg/kg; whereas the concentration estimated to be lethal to 25% of the population (LC25) for the same species in a 4-week test was 3,530 mg/kg, a value well above the SQT II. The median lethal concentration (LC50; concentration lethal to 50% of the population) for zinc in sediment for various freshwater organisms ranged from 69 mg/kg for 72-hour tests with *Limnodrillus hoffmeisteri* to 759 mg/kg for 72-hour tests with *Stagnicola attenuata*. Consequently, the toxicity of zinc varies greatly among species in laboratory spiked sediment toxicity tests. The lowest observable effect concentration (LOEC) reported for *Chironomus* and *Hyalella azteca* in a 10-day study conducted by Liber et al. (1996) was > 785 mg/kg (see Table 6). *Hyalella azteca* is an amphipod, as is *Crangonyx gracilis,* which was the most dominant species collected in benthic surveys for both the northern and southern part of Round Lake (see Tables 26 and 27). Zinc concentrations in Round Lake only exceeded this LOEC in three grids (Grid 18, 97 and 129). The hazard quotient for zinc (1.5) slightly exceeded 1 at the 0.0 to 0.5 foot depth interval, but was <1 at depths up to 1 – 2 feet, indicating a slightly possible causal link of potential adverse effects to benthic organisms from exposure to zinc in sediments at Round Lake, especially at the 0.0 – 0.5 foot depth interval at grid locations 18, 97 and 129.

Cadmium concentrations exceed the SQT II value (5.0 mg/kg) in 16 of 135 grids sampled with concentrations ranging from 5.5 to 15.3 mg/kg in 14 of the grids. The highest detected concentration of cadmium was 26.6 mg/kg in Grid 8. In a study conducted by Milani et al. (2003), cadmium inhibited growth to 25% of a population of *Chironomus riparius* at 16 mg/kg and to *Hyalella azteca* at 10 mg/kg. In the same study, lethality to 50% of the population (LC50; 10-day test) for *Chironomus riparius* was 39 mg/kg and for *Hyalella azteca* (28-day study) was 33 mg/kg (see Table 7 above). Cadmium concentrations in Round Lake exceeded the lowest effect level reported in the literature (10 mg/kg) at six grid locations (Grid 3, 8, 10, 85, 97 and 98). The hazard quotient for cadmium (1.1) slightly exceeded 1 at the 0.0 to 0.5 foot depth interval, but was <1 at depths up to 1 – 2 feet, indicating a slightly possible causal link of potential adverse effects to benthic organisms from exposure to cadmium in sediments at Round Lake, especially at the 0.0 – 0.5 foot depth interval at grid locations 3, 8, 10, 85, 97 and 98.

Lead concentrations exceed the SQT II value (130 mg/kg) in 10 of 135 grids sampled with concentrations ranging from 131 to 195 mg/kg in 9 of the grids (3, 8, 10, 11, 18, 26, 85, 97 and 98). The highest concentration of lead was detected in Grid 16 (258 mg/kg). As reported in Table 8 above, the two experimentally derived LC50 values are 3800 mg/kg for 4-day tests with *Chironomus dilutus* in a study conducted by Mehler et al. (2011) and 6840 mg/kg for 4-week tests with *Hyalella azteca* conducted by Borgmann and Norwood (1999). This data would indicate that lead is only toxic to aquatic invertebrates in sediments at very high concentrations, an order of magnitude higher than any detected concentration in the sediments of Round Lake. It should be noted that the HQ for lead was <1 at all depth intervals (see Table 16), indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to lead in sediments at Round Lake. Chromium concentrations exceed the SQT II value (110 mg/kg) in only 6 of 135 grids, with values ranging from 112-133 in five grids (Grids 26, 70, 97, 98, and 129), slightly above the SQT II. The highest detected concentration of chromium was in Grid 10 at 295 mg/kg, where chromium is co-located with cadmium, copper, lead and zinc. As reported in Table 9 above, the effective concentration resulting in an inhibition of growth and/or reproduction of test organisms by 50% (EC50) ranges from 167 mg/kg for 48-hour tests with *Daphnia magna* for Cr(VI) to 436 mg/kg for 24-hour tests with *Daphnia magna* for Cr(III). Chromium III is relatively insoluble and will sorb to organic matter. The average TOC concentrations at Round Lake from the 2011 data are reported as 22% with approximately 45% organic matter content; indicating that the bioavailability of chromium will be limited. In addition, it should be noted that the HQ for chromium was <1 at all depth intervals (see Table 16), indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to chromium in sediments at Round Lake. The only potential harmful effects to benthic invertebrates inhabiting the sediments in Round Lake from exposure to chromium would occur in Grid 10 with the highest detected chromium concentration of 295 mg/kg.

PCB concentrations exceed the SQT II value of 0.68 mg/kg in only 2 grids; Grid 86 at 0.826 mg/kg and Grid 32 at 0.887 mg/kg. As reported in Table 10, a no-effect concentration of 1.07 mg/kg was reported for total PCBs in a 120-day study with a marine polychaete, a value that is higher than the concentration of PCBs detected in these grids. For Aroclor-1254, 96-hour LC50 values ranged from > 3.4 mg/kg to > 60 mg/kg in saltwater shrimp to > 500 mg/kg in *Pimephales promelas*, a freshwater fish. For Aroclor-1242, a 96-hour LC50 of > 0.78 mg/kg was reported for a saltwater shrimp species. It should be noted that the HQ for total PCBs was <1 at all depth intervals (see Table 16), indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to PCBs in sediments at Round Lake.

4.5.2 Supporting Lines of Evidence

The SQT II exceedances for the COCs will be evaluated using the following supporting lines of evidence: sediment toxicity test results from Round Lake sediment samples, bioavailability of metals and PCBs to benthic organisms and benthic community structure. This approach is supported by the ITRC's Contaminated Sediments Team as the Sediment Quality Triad (SQT) procedure for evaluating the effect of COCs on benthic organisms. The SQT is a weight-of-evidence approach that integrates sediment chemistry (SEM/AVS, TOC), aquatic toxicity testing and benthic community analysis (ITRC, 2011). Figure 11 below shows the locations of data collected for the supporting lines of evidence for bioavailability studies, TOC sampling, benthic survey results and sediment toxicity test results.

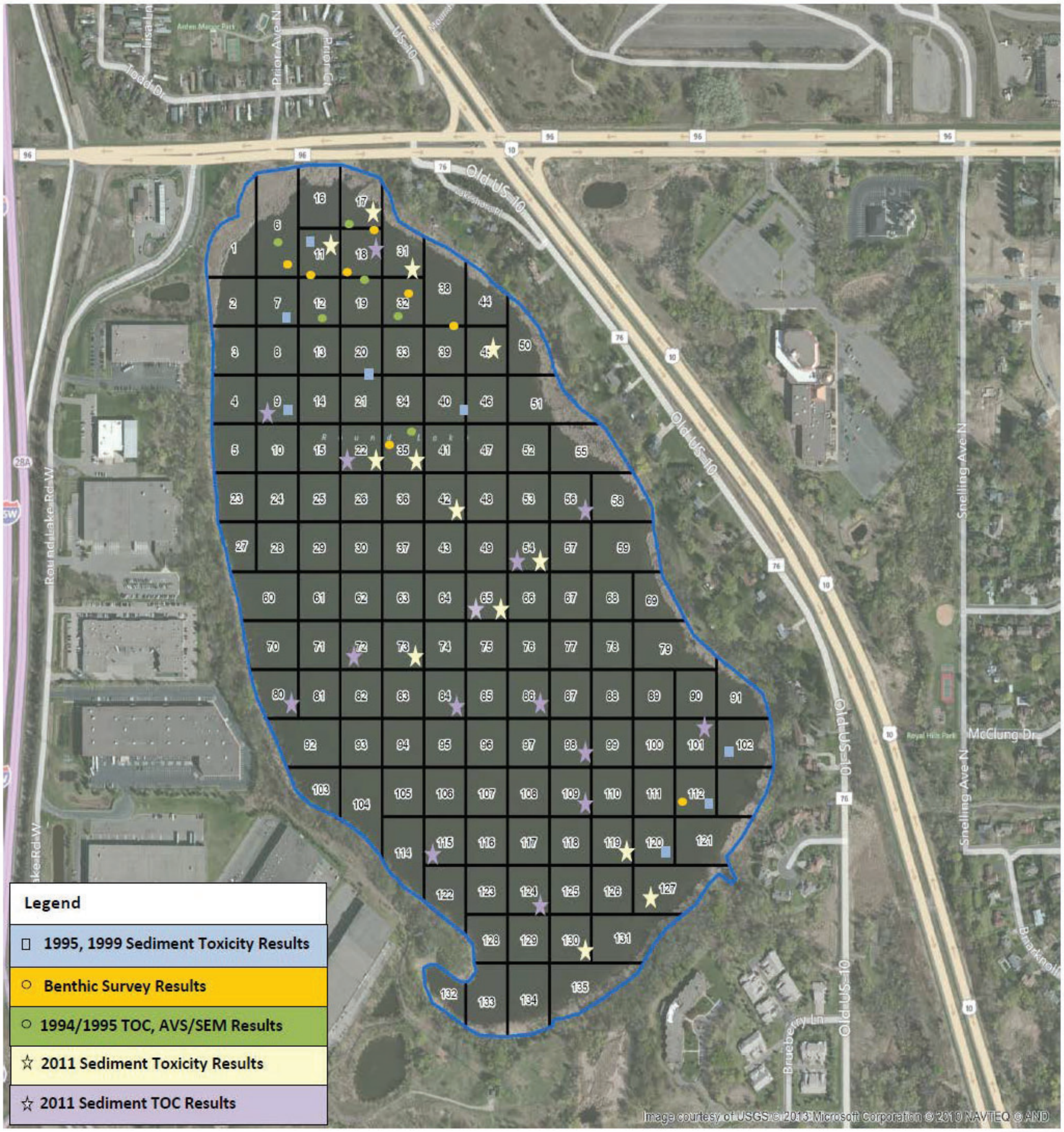


Figure 11. Sample locations for all supporting lines of evidence data

The first supporting line of evidence used to assess the potential for adverse effects to benthic organism from exposure to the COCs is an analysis of the toxicity studies performed with Round Lake sediments at locations indicated on Figure 11. Sediment toxicity tests provide direct quantifiable evidence of the biological effects of sediment contamination on survival and growth that can only be inferred from chemical or benthic community analyses due to water quality fluctuations, physical parameters, and biotic interactions (EPA/600/R-94/024, June 1994, Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates). Tables 23 and 24 present the acute and chronic toxicity test results for studies conducted in 1995, 1999 and 2011 (USCHPPM 1998; 2000 and Wenck 2011) using lake sediments (top 6 inches) with the test species *Chironomus riparius*, *Chironomus tentans*, and *Hyalella azteca* and following standard ASTM/USEPA testing methods.

Table 23. Acute sediment toxicity test results

Location (Date)	Chironomid species		*Hyalella azteca*	
	Survival (%)	Growth Ash Weight (mg)	Survival (%)	Growth Dry Weight (mg)
Grid 7 (1999)	**41**	na	na	na
Grid 9 (1999)	83	na	na	na
Grid 11 (1999)	**44**	na	na	na
Grid 11 (2011)	90	1.26	**66.25**	0.06
Grid 17 (2011)	95	1.14	**40**	**0.04**
Grid 20 (1999)	86	na	na	na
Grid 22 (2011)	93.75	1.25	86.25	**0.04**
Grid 31 (2011)	95	1.54	88.75	0.11
Grid 35 (2011)	96.25	1.06	86.25	0.10
Grid 40/46 (1999)	84	na	na	na
Grid 42 (2011)	96.25	**0.59**	83.75	0.06
Grid 54 (2011)	97.25	**0.68**	91.25	0.06
Grid 65 (2011)	95	0.83	78.75	0.05
Grid 73 (2011)	97.5	**0.56**	**25**	0.08
Grid 102, 112, 120 (1995)	**62.5**	**0.325**	90	0.16
Grid 119 (2011)	98.75	0.77	75	0.07
Grid 127 (2011)	98.75	0.87	90	0.08
Grid 130 (2011)	92.5	0.97	86.25	0.05

1995 – *Chironomus riparius* and *Hyalella azteca*, 14-day test; 1999 – *Chironomus riparius*, 14-day test; 2011 – *Chironomus tentans* and *Hyalella azteca*,10-day test

Table 24. Chronic sediment toxicity test results

Location (Date)[a]	Chironomid species				*Hyalella azteca*	
	Survival (%)	Growth Ash Weight (mg)	Emergence Rate (%)	Mean Time to Emergence (d)	Survival (%)	Growth Dry Weight (mg)
Grid 7 (1999)	**40**	na	**39**	15.0	na	na
Grid 9 (1999)	**13**	na	**11**	17.2	na	na
Grid 11 (1999)	**60**	na	**58**	15.6	na	na
Grid 11 (2011)	**28.10**	1.53	na	na	87.5	0.34
Grid 17 (2011)	na	na	na	na	**58.75**	0.30
Grid 20 (1999)	**13**	na	**13**	15.7	na	na
Grid 35 (2011)	93.8	1.22	na	na	97.5	0.59
Grid 40/46 (1999)	79	na	78	18.1	na	na
Grid 54 (2011)	80.20	1.50	na	na	na	na
Grid 65 (2011)	66.70	1.67	na	na	93.75	0.31
Grid 102, 112, 120 (1995)	100	na	na	**27.5**	na	na
Grid 119 (2011)	65.60	1.36	na	na	97.5	0.35
Grid 130 (2011)	89.60	1.29	na	na	93.75	0.34

[a]1995 - *Chironomus riparius* and *Hyalella Azteca*, 30-day test; 1999 – *Chironomus riparius*, 30-day test; 2011 – *Chironomus tentans* - 20-day test; *Hyalella Azteca* – 28 day test

The growth of *Hyalella azteca* was slightly reduced following acute exposure at Grids 17 and 22; no adverse effects to growth of this species following chronic exposure were observed. Survival of *Hyalella azteca* was reduced following acute exposure in 3 samples at the following locations: Grid 11 (66.25%), Grid 17 (40%) and Grid 73 (25%); whereas, survival was only reduced to 58.75% in one sample (Grid 17) following chronic exposure. These results would indicate that exposures to the COCs in sediments across the lake are not significantly impacting the growth and survival of this species.

Adverse effects to the growth of chironomid species following acute exposure were only observed in four samples (Grids 42, 54, 73 and combined 102, 112 and 120); no adverse effects to the growth of chironomids following chronic exposure were observed. However, survival and emergence rate of chironomids was reduced following acute and chronic exposure (Grids 7, 9, 11 and 20) in sediments in the northwestern edge of the lake where benthic survey data indicates a benthic population with taxa richness and moderately high diversity values (See Table 26; DI = 2.76). Survival in chironomids following acute exposure was slightly reduced (62.5%) in the combined Grids 102, 112 and 120, located in the southwestern part of the lake.

A summary of the toxicity test results indicates that exposure to sediments in grid locations 9, 20, 31, 35, 40/46, 65, 119, 127 and 130 did not produce short-term toxic effects to either benthic test species.

Likewise, exposure to sediments in grid locations 35, 40/46, 54, 65, 119 and 130 did not produce long-term toxic effects to either benthic test species. Adverse effects to the test species (chironomid and amphipod) were observed sporadically across the lake. Growth was slightly impacted in both species following acute exposure; however, it was not impacted following chronic exposure. Survival of *Hyalella azteca* was only reduced in Grid 17 following both acute and chronic exposure. The most sensitive endpoint was survival and emergence rate in chironomids following chronic exposure at two locations in the northern part of the lake (survival rate 13% in Grids 9 and 20; emergence rates 11% in Grid 9 and 13% in Grid 20). With the exception of grids 9 and 20, these results correlate well with toxicity effect levels from the literature since the metal concentrations in these grids either did not exceed the SQT II values or were below reference toxicity levels reported from the literature. None of the COCs were detected above the SQT II levels in Grid 20. The lowest observable effect concentration (LOEC) for zinc reported for *Chironomus* and *Hyalella azteca* in a 10-day study conducted by Liber et al. (1996) was > 785 mg/kg (see Table 6) and the concentration of zinc in Grid 9 was 582 and <460 in Grid 20. Cadmium and copper were the only other COCs detected in Grid 9 above SQT II levels. In a study conducted by Milani et al. (2003), cadmium inhibited growth to 25% of a population of *Chironomus riparius* at 16 mg/kg and to *Hyalella azteca* at 10 mg/kg, levels higher than the detected concentration of cadmium of 7.2 mg/kg in Grid 9. The accepted LC_{50} values for copper for *Chironomus tentans* and *Hyalella azteca* in a static toxicity test conducted by Cairns et al. (1984) ranged from 685 to 1,259 mg/kg (see Table 5) and the concentration of copper in Grid 9 was 621 mg/kg.

The prediction of the toxicity of metal contaminants in sediments to benthic organisms is challenging due to the strong influence of the properties of the sediments (Campana et al, 2012). The bulk sediment metal concentration is a poor predictor of the potential for effects to benthic organisms; similar dry weight metal concentrations can exhibit a wide range of effects on benthic organisms (McGrath et al., 2002). The ITRC Contaminated Sediment Team concluded that "the relationship between contaminant concentration in sediments and the risk from exposure is not linear … that only the bioavailable fraction of an environmental contaminant may be taken up and subsequently result in an effect on an organism" (ITRC, 2011). Several investigators have determined that the prediction of metal toxicity in sediments is not possible using sediment quality guidelines alone (i.e. such as the TBC values presented in Table 11) because they do not establish a cause-effect relationship and they do not consider bioavailability; consequently, resulting in a conservative prediction of when toxic effects will not occur. Borgmann (2003) indicates that exceedance of an assessment value indicates an increased likelihood of toxic effects, but it is not proof of cause and it cannot be assumed that the contaminant present in excess is responsible for the observed effects. McGrath et al. (2002) states that it is critically important to consider bioavailability when developing sediment quality guidelines. The toxicity of metal contaminants in sediments is dependent on the bioavailability of the metals in both the sediment and water (porewater, burrow water, overlying water) phases and on the sensitivity of the organism to the metal exposure (Campana et al., 2012). The bioavailability of the metals in sediments is controlled by (1) the metal binding with particulate sulfide, organic carbon and iron hydroxide phases; (2) the sediment-water partitioning relationships; (3) organism physiology (uptake rates and assimilation efficiencies); and (4) organism behavior (feeding selectivity and burrow irrigation) (Simpson and Batley, 2007).

The second supporting line of evidence used to assess the potential for adverse effects to benthic organisms in Round Lake is the bioavailability of cadmium, chromium, copper, lead and zinc in the sediments. This evidence will be presented as results of studies conducted with Round Lake sediments at the locations indicated on Figure 11 and results from a review of the literature. The ITRC Contaminated Sediments Team guidance (Incorporating Bioavailability Considerations into the Evaluation of Contaminated Sediment Sites, February 2011) notes the importance of bioavailability in risk assessments and in establishing technically defensible cleanup goals due to the low predictive value and conservative nature of SQGs when considered alone. The guidance states that the relationship between sediment contaminant concentrations and risk from exposure is not linear due to bioavailability considerations which may in some instances only result in a fraction of the contaminant being available to cause harm to ecological receptors (ITRC, 2011).

Numerous studies (Carlson et al, 1991; DiToro et al., 1992; Green et al., 1993; and Casas and Crecelius, 1994) using both freshwater and saltwater sediments have shown than acid-volatile sulfide (AVS) and interstitial water concentrations (IW) can be used to predict toxicity in sediments contaminated with divalent cationic metals (cadmium, copper, nickel, lead and zinc). In all of these studies, no toxicity was observed in amphipods, oligochaetes, snails, polychaetes and copepods when the simultaneously extracted metal (SEM) to AVS ratio was < 1.0. The AVS in the sediment reacts with the simultaneously extracted metal (SEM) and forms an insoluble metal sulfide that is relatively non-available for uptake by benthic organisms. Consequently, the metal will exist in the metal sulfide form in the sediment if the AVS is present at a concentration in excess of the reactive form of the metal, leaving low free metal concentrations in the pore water to cause toxicity to the organisms (McGrath et al., 2002). Berry et al. (1996) conducted ten-day toxicity tests with the amphipod *Ampelisca abdita* in sediments with varying AVS concentrations and spiked with cadmium, copper, lead, nickel or zinc. Ninety-seven point seven percent of the 43 sediments with SEM/AVS ratios < 1.0 were not toxic (2.3% caused >24% mortality); whereas, sediments with SEM/AVs ratios > 1.0 were frequently toxic (80% caused >24% mortality). In sediments with SEM/AVS < 1.0, there was no detectable metal in the interstitial water. The relative affinity of metals for AVS is copper > lead > cadmium > zinc > nickel. The authors indicated that the presence of additional binding factors may account for the fact that not all sediments with SEM/AVS ratios > 1.0 caused increased mortality.

Long-term studies have also been conducted with similar results. DeWitt et al. (1996) observed no significant effects on survival, growth or reproductions to a marine amphipod following 28-day exposure to sediments containing more AVS than cadmium, with cadmium concentrations up to 363 mg/kg on a dry weight basis. The highest cadmium concentration detected in the sediments of Round Lake was 26.6 mg/kg (see Table 21), an order of magnitude below the NOEC reported in this study. Sibley et al. (1996) observed the same results with *Chironomus tentans* exposed to zinc concentrations as high as 270 mg/kg. Liber et al (1996) conducted a colonization experiment for 16 months with sediments from a freshwater mesotrophic pond and observed no difference in benthic community structure with zinc concentrations below 700 mg/kg dry weight. There are only 4 grid locations in Round Lake with zinc concentrations exceeding 700 mg/kg, Grids 18, 74, 97 and 129 (see Table 21). Burton et al. (2005) conducted a 6 to 37 week field study to validate concentrations of zinc in freshwater sediments that would be tolerated by benthic macroinvertebrate communities. No adverse effects to benthic macroinvertebrates were observed in a freshwater eutrophic lake with very organic sediments, zinc concentrations as high as 913 mg/kg, and SEM/AVS ratios of 0.2 and 0.7. These environmental conditions are very similar to Round Lake and maximum detected zinc concentrations were around 850 mg/kg (see Table 21). At SEM/AVS ratios > 8.0, community indices of abundance, diversity and evenness were significantly lower; ratios > 2.0 resulted in occasional effects; and ratios < 2.0 resulted in no effects to community indices. Hansen et al. (1996) summarized data from sediment toxicity tests with lab-spiked sediments and field sediments and observed that with all tests combined, 98.1% of the sediments were non-toxic with an SEM/AVS ratio ≤ 1.0 and 58% were toxic with an SEM/AVS ratio > 1.0.

In anaerobic sediments, organic carbon is an additional binding phase controlling metal partitioning, particularly for cadmium, copper and lead (Ankley et al., 1996). In aerobic sediments, binding factors other than AVS control bioavailability, including particulate organic matter and iron and manganese oxyhydroxides (Batley et al., 2005 in Wenning). Organic carbon is an important partitioning phase for metals in sediment because of the tendency of the positive charged metal ions to bind to the negatively charged organic matter. McGrath et al. (2002) states that toxicity is not expected when organic carbon is < 150 µmol/g; and Burton et al. (2005) reported no toxic effects to benthic community indices at 100 µmol/g (1.2 g/kg) of organic carbon. The average total organic carbon (TOC values range from 63 to 330 g/kg) in Round Lake is 22% with organic matter comprising approximately 45% of the sediment. Besser et al. (2003) showed that increasing the amount of organic matter in sediments increased the partitioning of cadmium and copper to sediments and lowered the toxicity of the sediments to the amphipod, *Hyallela azteca*.

Two preliminary studies (MPCA, 1994; USACHPPM, 1998) were conducted using the SEM/AVS methodology and Round Lake sediments to assess the potential bioavailability of the metal COCs. The results are presented in Table 25. MPCA (1994) sampled 4 locations in the northern part of the lake and found that SEM/AVS ratios ranged from 0.16 to 0.34, with the exception of one location with a ratio of 1.35 (Sample RL 2, Grid 6). MPCA concluded that AVS content in northern Round Lake is relatively high, indicating that sufficient acid volatile sulfide is present to bind the metals to the sediment; thus, decreasing their bioavailability. USACHPPM (1998) sampled four locations in 1995. Three of the four sample locations in the lake indicated that sufficient acid volatile sulfide (2.1 – 28 µmol/g) existed in the northern sediments to bind the divalent transition metals cadmium, copper, nickel, lead, and zinc, decreasing their bioavailability. The SEM/AVS ratios in this study ranged from 0.068 to 0.093, with the exception of one location with a ratio of 0.89 to 1.24 (RL0501, Grid 18/19). USACHPPM determined that the sample duplicate (1.24; RL0501) was due to zinc. This sample location had a high organic carbon content (230,000 mg/kg) reported in the study, which, as indicated in Burton et al. (2005), no toxic effects to benthic community indices were reported at 100 µmole/g (1,200 mg/kg) of organic carbon. USACHPPM's analysis of sediments in Grid 6 resulted in an SEM/AVS ratio of 0.068, considerably lower than the value of 1.35 determined by MPCA. In addition, Wenck performed sediment sampling in 2011 at 15 sampling locations at Round Lake to determine the total organic carbon (TOC) content of the sediments (see Table 25 below). Average TOC concentrations at Round Lake from the 2011 data are reported as 22% with approximately 45% organic matter content. TOC concentrations at 0 – 1' depth at grid locations 9, 18, 22, 54, 56, 65, 72, 80, 84, 86, 98, 101, 109, 115 and 124 range from 63 – 330 g/kg (63,000 to 330,000 mg/kg).

In summary, results of studies with Round Lake sediments indicate that SEM/AVS ratios in the northern part of the lake ranged from 0.068 to 1.24, indicating that the divalent cationic metals are not likely to be bioavailable to benthic organisms in the sediments at Grid locations 6, 12, 17, 32 and 35, where detected concentrations of copper, cadmium and zinc are reported above the SQT II. However, the cadmium, copper, lead and zinc detected in sediments at Grid 18/19 (SEM/AVS 0.89 to 1.24) may have limited bioavailability to the benthic organisms inhabiting the sediments at this location; however, reported TOC concentrations for sediment in this grid are high, ranging from 110 to 230 g/kg . As explained above, study results reported in the literature by several investigators indicated that toxicity to benthic invertebrates is not observed at SEM/AVS ratios < 1.0 in short and long-term studies conducted with both laboratory spiked and field samples. In addition, results of TOC random sampling in 2011 show that the TOC concentrations in Round Lake sediments at 0 – 1' depths ranged from 63 to 330 g/kg (63,000 to 330,000 mg/kg) and the average total organic carbon in Round Lake sediments has been reported as 22%. Consequently, the elevated TOC in the sediments of Round Lake influence the bioavailability of the chromium, copper, cadmium, lead and zinc detected at concentrations exceeding the SQT II at Grid locations 6, 9, 18, 84, 86 and 98. Burton et al. (2005) reported no toxic effects to benthic community indices at 100 µmol/g (1.2 g/kg) of organic carbon and Besser et al. (2003) showed that increasing the amount of organic matter in sediments increased the partitioning of cadmium and copper to sediments and lowered the toxicity of the sediments to the amphipod, *Hyallela azteca*. Based on the characteristics of the sediments at Round Lake, it is apparent that the toxicity of cadmium, chromium, copper, lead and zinc is being strongly influenced by the bioavailability of the metals due to binding with particulate sulfide and organic carbon.

Table 25. Bioavailability of Metals in Sediments

Grid (Sample Location)	SEM/AVS Ratio	Total Organic Carbon (g/kg – 0-1')
Grid 6 (RL 2 - 1994)	1.35	77
Grid 6 (RL0701 - 1995)	0.068	
Grid 9		170 - 190
Grid 12 (RL 3 - 1994)	0.34	
Grid 17 (RL 1 - 1994)	0.17	
Grid 18/19 (RL0501 - 1995)	0.89 – 1.24	230
Grid 18		150 - 110
Grid 22		190 - 330
Grid 32 (RL 4 – 1994)	0.16	
Grid 32 (RL0901 – 1995)	0.093	
Grid 35 (RLXX01 – 1995)	0.076	
Grid 54		300 – 230
Grid 56		190 - 280
Grid 65		280 – 290
Grid 72		180 - 240
Grid 80		63 – 170
Grid 84		210
Grid 86		190 – 220
Grid 98		190 – 140
Grid 101		140 – 170
Grid 109		310 – 300
Grid 115		140 – 180
Grid 124		180

SEM/AVS ratios: MPCA 1994; USACHPPM 1995 Round Lake – *Bioavailability of Sediment Metals in Round and Sunfish Lakes* (USACHPPM 1998b);
TOC Data: USACHPPM 1995/Wenck 2011

The third supporting line of evidence used to assess the potential for adverse effects to benthic organisms is the benthic invertebrate community structure in Round Lake. The goal of the Superfund is to reduce ecological risk to levels "that will result in the recovery and/or maintenance of healthy local populations/communities of ecological receptors that are or should be present at or near the site" (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 2, OSWER Directive 9285.7-28P, October 7, 1999). A principle in ecological risk assessment is that Superfund remedies should generally be designed to protect local populations and communities of biota and not to protect organisms on an individual basis (except in the instance of the presence of T&E species) ((Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999). It should be noted that no endangered, threatened or special concern benthic invertebrate species inhabit Round Lake.

Species richness, abundance, and diversity are good indicators of the health of the benthic populations being supported by an aquatic habitat and of the structure of the benthic community. Benthic survey studies were conducted by USCHPPM in 1993 (northern and southern end of lake) and 1995 (north to north-central part of the lake). The sample locations are indicated in Figure 11 and results of the surveys are presented in Tables 26 and 27.

The benthic invertebrate community in the northern part of Round Lake is relatively healthy with diversity indices ranging from 2.5 to 3.3 (see Table 26). The species composition in the northern part of the lake is dominated by *Crangonyx gracilis* (27.7%; crustacean – amphipod)*, Chaoborus punctipennis* (17.2%; insect – dipteran -phantom midge), *Chironomus decorus* (7.2 %; insect – dipteran – chironomid). Three mollusk species were relatively abundant: *Physella* gyrina (6.6%; gastropod)*; Mentus dilatus* (5.4%; gastropod); and *Sphaerium* sp. (5.2%; bivalve). The benthic community in the southern part of the lake is more stressed with diversity values ranging from 1.78 to 1.86. The species composition in the southern part of the lake is dominated by *Crangonyx gracilis* (27.8%), *Chironomus decorus* (7.4%) and the mollusks *Mentus dilates* (5.4%) and *Sphaerium* sp. (5.2%). As with the northern part of the lake, the benthic community inhabiting Round Lake is predominantly comprised of amphipods, chironomids, mollusks and various insect larvae, predominantly Trichoptera .

The benthic species inhabiting Round Lake are typical of a lentic depositional environment (Merritt and Cummins, 1978). The species collected typically inhabit lentic littoral and profundal areas of a lake. These species would be expected to inhabit Round Lake since the majority (95%) of the lake is a relatively flat shoal averaging approximately 4.5' in depth. The sediment composition of Round Lake is predominantly muck and peat and continues to 4 to 6 foot depths, with some cores near the shoreline encountering sand or clay at <4' depth (Wenck, 2012). The species occur in depositional fine sediments mixed with organic matter or among vascular hydrophytes. Sprawlers, such as *Chaoborus* and *Hesperophylax*, are known to inhabit the surface of floating leaves of vascular hydrophytes and burrowers, such as chironomids and mollusks, inhabit the fine sediments. The species are primarily herbivores (shredders feeding on vascular hydrophytes), detritivores (collectors/gatherers feeding on fine particulate organic matter), or predators (piercers and engulfers feeding on plant and animal tissue) (Merritt and Cummins, 1978).

The benthic populations in the northern part of Round Lake (around grid locations 6, 11, 12, 17/18, 18/19, 32, 35 and 38/39; see Table 26 and Figure 11) are healthy with diverse species composition. Consequently, this would indicate that the cadmium, copper, lead and zinc at detected concentrations exceeding the SQT II are not impacting the benthic community in this part of the lake; especially considering that the species inhabiting the lake are species that would typically be found in this type of environment. There is not enough information to characterize the benthic population in the southern part of the lake, since the results are composite results from 3 replicate samples from one location. However at Grid 112 along the southeastern edge of the lake, the benthic community appears to be stressed even though sediment toxicity testing performed in this area of the lake indicated that the sediments did not adversely affect the growth and survival of *Chironomus tentans* and *Hyalella azteca*.

Table 26. Species composition and abundance of benthic macroinvertebrates communities in Northern Round Lake

Species	1993 sampling event			1995 sampling event				Species composition
	Round 1	Round 2	Round 3	RL0501	RL0701	RL0901	RLXX01	
Phylum Annelida Class Hirudinea *Erpobdella punctate*	2							0.3
Glossiphonia complanata	1							0.2
Helobdella papillata					1	2		0.5
Class Oligochaeta *Amphichaeta americanus*				4				0.7
Aulodrilus americanus	8	8	11					4.7
Aulodrilus pluriseta				3	5		4	2.1
Phylum Mollusca Class Bivalvia *Sphaerium sp.*	2	9	19					5.2
Sphaerium striatum					1	1	1	0.5
Class Gastropoda *Gyraulus deflatus*						1	2	0.5
Mentus dilatus	11	13	7					5.4
Physella gyrina	4	4			4	19	7	6.6
Phylum Arthropoda Class Crustacea *Crangonyx gracilis*	40	5	16	67	3	25	3	27.7
Class Insecta Order Ephermeroptera *Caenis amica*	2		2					0.7
Order Trichoptera *Agrypnia vestita*	1	3						0.7
Hesperophylax designatus	16		7					4.0
Trichoptera sp.				1				0.2
Order Coleoptera *Haliplus* sp.	1							0.2
Order Hemiptera								

Table 26. Species composition and abundance of benthic macroinvertebrates communities in Northern Round Lake

Species	1993 sampling event			1995 sampling event				Species composition
	Round 1	Round 2	Round 3	RL0501	RL0701	RL0901	RLXX01	
Corixid sp.							1	0.2
Plea striolata						1		0.2
Order Diptera *Ablabesmyia mallochi*	11		4					2.6
Chaoborus punctipennis	1	3	7	12	27	1	47	17.2
Chironomus decorus	11	24	7					7.4
Cryptochironomus fulvus	1							0.2
Culicoides sp.	1							0.2
Glyptotendipes lobiferus				24	4			4.9
Paratendipes sp.				3	4			1.2
Polypedilus fallax				3	2		19	4.2
Procladius subletti		8			1			1.6
Taxa Richness	**16**	**9**	**9**	**8**	**10**	**8**	**8**	
Abundance	**113**	**77**	**80**	**117**	**52**	**51**	**84**	
Diversity (H)	**2.76**	**2.58**	**2.67**	**2.6**	**3.3**	**2.5**	**2.7**	

Table 27. Species composition and abundance of benthic macroinvertebrate communities in southern Round Lake

Species	Round 4 - 1993	Round 4 - 1995	Species Composition
Phylum Annelida Class Hirudinea *Glossiphonia complanata*	1		0.3
Phylum Mollusca Class Bivalvia *Sphaerium sp.*	10		5.2
Sphaerium striatum		9	0.5
Class Gastropoda *Gyraulus deflatus*		2	0.5
Mentus dilatus	6		5.4
Phylum Arthropoda Class Crustacea *Crangonyx gracilis*	1	9	27.8
Class Insecta Order Trichoptera *Hesperophylax designatus*	20		0.7
Hydroptila consimilis	1		4.0
Order Diptera *Chaoborus punctipennis*	5		2.6
Chironomus decorus		10	7.4
Procladius subletti		2	1.6
Taxa Richness	**7**	**5**	
Abundance	**44**	**32**	
Diversity (H)	**1.86**	**1.78**	

4.5.3 Summary of Potential Risks to the Benthic Organism Endpoint for the Current Use Scenario

The HQ for PCBs was <1 at all depth intervals (see Table 16) and total PCBs exceeded SQT II values in Grids 32 and 86 (see Table 22) at concentrations lower than the no-effect concentration of 1.07 mg/kg reported for total PCBs in the literature (see Table 10); consequently, it was concluded that there is not a potential risk of adverse effects to benthic invertebrates from exposure to PCBs in sediments at Round Lake.

Figure 12 illustrates the grids with metal concentrations that may cause adverse effects to the benthic organisms inhabiting the sediments of Round Lake. The potential areas of concern for causing possible risk to benthic organisms are sediments in grids along the western edge of the lake, a cluster of grids in the deepest part of the lake and a few isolated grids in the southern tip of the lake. Each COC was evaluated based on concentration data in each grid in relationship to its potential toxicity, bioavailability, ability to impact the biodiversity of the benthic population and co-location with other COCs. Co-located metals at concentrations that exceeded the low effect benchmark (SQT II) were evaluated based on these lines of evidence in consideration of possible additive exposure to the metals in the sediments at these locations. An evaluation of the measures of effect and the supporting lines of evidence for each COC is described below.

The HQ for chromium was <1 at all depth intervals (see Table 16) indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to chromium in sediments at Round Lake. However, chromium was detected in Grid 10 at 295 mg/kg, above the lowest reported EC_{50} of 167 mg/kg for 48-hour tests with *Daphnia magna* for Cr(VI). In the other grids where chromium was co-located with other metals, the concentrations were below the EC_{50}. Consequently, exposure to chromium in Grid 10 may adversely impact the growth, survival and reproduction of benthic species inhabiting the sediment at this location in Round Lake.

The HQ for lead was <1 at all depth intervals (see Table 16), indicating that there is not a direct link of causality of potential adverse effects to benthic invertebrates from exposure to lead in sediments at Round Lake. Lead was detected above the SQT II at concentrations ranging from 131 to 258 mg/kg. Published toxicity data indicates that lead is only toxic to aquatic invertebrates in sediments at very high concentrations (3800 mg/kg to 6840 mg/kg in *Chironomus tentans* and *Hyalella azteca*, respectively; see Table 8), an order of magnitude higher than the detected concentration in the sediments. Consequently, exposure to lead would not be expected to adversely impact the growth, survival and reproduction of benthic species inhabiting the sediment in Round Lake.

The HQ for cadmium (1.1) slightly exceeded 1 at the 0.0 to 0.5 foot depth interval, but was <1 at depths up to 1 – 2 feet, indicating a possible causal link of potential adverse effects to benthic organisms from exposure to cadmium in sediments at Round Lake, especially at the 0.0 – 0.5 foot depth interval. In a study conducted by Milani et al. (2003; see Table 7), cadmium inhibited growth to 25% of a population of *Chironomus riparius* at 16 mg/kg and to *Hyalella azteca* at 10 mg/kg. Cadmium concentrations in Round Lake exceeded the lowest effect level reported in the literature (10 mg/kg) at six grid locations (Grid 3, 8, 10, 85, 97 and 98). Cadmium is also co-located with copper and zinc at grid locations 9, 11, 12, 18, 26 and 32; however, reported SEM/AVS ratios (see Table 25) for Round Lake sediments at Grids 12 (0.34) and 32 (0.093 to 0.16) indicate that the cadmium, copper and zinc would not be bioavailable for uptake by the benthic organisms and consequently, would not adversely impact them. Numerous studies (Carlson et al, 1991; DiToro et al., 1992; Green et al., 1993; and Casas and Crecelius, 1994) reported no toxicity was observed in amphipods, oligochaetes, snails, polychaetes and copepods when the simultaneously extracted metal (SEM) to AVS ratio was < 1.0. DeWitt et al. (1996) observed no significant effects on survival, growth or reproductions to a marine amphipod following 28-day exposure to sediments containing more AVS than cadmium, with cadmium concentrations up to 363 mg/kg on a dry weight basis, an order of magnitude higher than the highest cadmium concentration detected in the sediments of Round Lake of 26.6 mg/kg (see Table 21). In addition, organic carbon is an important partitioning phase for metals in sediment because of the tendency of the positive charged metal ions to bind to the negatively charged organic matter. The reported TOC concentration for sediments in Grid 9 (170 to 190 g/kg), Grid 18 (110 to 230 g/kg) and Grid 98 (140 – 190 g/kg) indicate that the cadmium, copper and zinc could not be bioavailable for uptake by the benthic organisms. McGrath et al. (2002)

states that toxicity is not expected when organic carbon is < 150 µmol/g; and Burton et al. (2005) reported no toxic effects to benthic community indices at 100 µmol/g (1.2 g/kg) of organic carbon. TOC values range from 63 to 330 g/kg in sediments from 0 – 1', and the average total organic carbon in Round Lake is 22% with organic matter comprising approximately 45% of the sediment. Besser et al. (2003) showed that increasing the amount of organic matter in sediments increased the partitioning of cadmium and copper to sediments and lowered the toxicity of the sediments to the amphipod, *Hyallela azteca*. Based on the characteristics of the sediments at Round Lake, it is apparent that the toxicity of cadmium, copper, and zinc is being strongly influenced by the bioavailability of the metals due to binding with particulate sulfide and organic carbon.

Benthic surveys conducted with Round Lake sediments show that the sediments at locations in the northern part of the lake around Grids 6, 11, 12, 17/18, 18/19, 32, 35 and 38/39 (see Table 26 and Figure 11) were not impacting the growth, survival and reproduction of benthic organisms. The benthic invertebrate community in the northern part of Round Lake is relatively healthy with diversity indices ranging from 2.5 to 3.3 (see Table 26). The species composition in the northern part of the lake is dominated by *Crangonyx gracilis* (27.7%; crustacean – amphipod), *Chaoborus punctipennis* (17.2%; insect – dipteran -phantom midge), *Chironomus decorus* (7.2 %; insect – dipteran – chironomid). Three mollusk species were relatively abundant: *Physella* gyrina (6.6%; gastropod); *Mentus dilatus* (5.4%; gastropod); and *Sphaerium* sp. (5.2%; bivalve). These species occur in depositional fine sediments mixed with organic matter or among vascular hydrophytes. They typically inhabit the lentic littoral and profundal areas of a lake.(Merritt and Cummins, 1978) and would be expected to inhabit Round Lake since the majority (95%) of the lake is a relatively flat shoal averaging approximately 4.5' in depth (Wenck, 2012). This would indicate that the cadmium, copper, and zinc at detected concentrations exceeding the SQT II at these grid locations are not impacting the benthic community in this part of the lake; especially considering that the species inhabiting the lake are species that would typically be found in this type of environment. Consequently, the risk evaluation of all lines of evidence indicate that exposure to cadmium in Grids 3, 8, 10, 26, 85 and 97 may adversely impact the growth, survival and reproduction of the benthic organisms inhabiting Round Lake.

The HQ for zinc (1.5) slightly exceeded 1 at the 0.0 to 0.5 foot depth interval, but was <1 at depths up to 1 – 2 feet, indicating a possible causal link of potential adverse effects to benthic organisms from exposure to zinc in sediments at Round Lake, especially at the 0.0 – 0.5 foot depth interval. The lowest observable effect concentration (LOEC) reported for *Chironomus* and *Hyalella azteca* in a 10-day study conducted by Liber et al. (1996) was > 785 mg/kg (see Table 6). *Hyalella azteca* is an amphipod, as is *Crangonyx gracilis,* which was the most dominant species collected in benthic surveys for both the northern and southern part of Round Lake (see Tables 26 and 27). Zinc concentrations in Round Lake exceeded this LOEC in three grids (Grid 18, 97 and 129). Zinc is also co-located with copper in Grids 1, 3, 8, 9, 10, 11, 12, 19, 24, 26, 32, 46, 70, 74, 81, 84, 85, 86, 97, 98, 114 and 120 at concentrations that exceed the low-effect benchmark (460 mg/kg). As described above, levels of SEM/AVS and TOC and results of benthic surveys indicate that exposure to zinc and copper in the sediments at grid locations 9, 11, 12, 18, 19, 32, 84, 86 (TOC values of 190 to 220 g/kg were reported for Grid 86) and 98 would not be expected to adversely impact the growth, survival and reproduction of benthic organisms in sediments at these locations. In addition, Burton et al. (2005) conducted a 6 to 37 week field study to validate concentrations of zinc in freshwater sediments that would be tolerated by benthic macroinvertebrate communities. No adverse effects to benthic macroinvertebrates were observed in a freshwater eutrophic lake with very organic sediments, zinc concentrations as high as 913 mg/kg, and SEM/AVS ratios of 0.2 and 0.7. These environmental conditions are very similar to Round Lake and maximum detected zinc concentrations were around 850 mg/kg (see Table 21). Results from toxicity studies performed with Round Lake sediments indicated that no adverse effects to growth, survival and reproduction were observed in benthic test species from sediments at Grid 46. In addition, zinc and copper concentrations were below reference toxicity levels reported from the literature, indicating that the toxicity study results correlate well with published studies at this location. Consequently, the risk evaluation of all lines of evidence indicate that exposure to zinc in sediments in Grids 1, 3, 8, 10, 24, 26, 70, 74, 81, 85, 97, 114, 120 and 129 may adversely impact the growth, survival and reproduction of benthic organisms inhabiting Round Lake.

Copper is the most pervasive metal detected in the sediments of Round Lake and appears to exhibit the most potential for adverse effects to benthic invertebrates inhabiting the sediments of Round Lake since the HQs for copper range from 1.3 – 3.2, indicating a possible causal link of potential adverse effects to benthic organisms from exposure to copper in sediments at Round Lake, especially at the 0.0 – 0.5 foot depth interval. Copper concentrations in Grids 5, 23, 25, 34, 35, 43, 90 and 104 were below the freshwater PEL (197 mg/kg) developed for Canada by Smith et al. (1996; see Table 18). Smith et al. (1996) reported that the incidence of biological effects at concentrations exceeding the PEL-FW of 197 mg/kg is 44% (see Table 19). Copper concentrations in Grids 15, 18, 39, 76, 94 and 130 were below the ER-M value of 270 mg/kg reported by Long et al. (1995; see Table 18). Long et al. (1995) reported that the incidence of biological effects is 29% for concentrations equal and above the ERL but below ERM (possible effects range, see Table 18).   Published data concerning the toxicity of copper to benthic organisms indicate that concentrations >685 mg/kg could potentially be lethal to 50% of the population of *Chironomus* (95% CI LC_{50} 685 to 1,259 mg/kg, see Table 5), a predominant species that has been collected in the northern and southern areas of Round Lake. Copper concentrations in Grids 2, 4, 7, 14, 28, 42, 73, 96, 106, 116 and 118 ranged from 287 to 530 mg/kg, concentrations that are below the 95% CI for the LC_{50} reported for copper. The sediment grid locations with the maximum copper concentrations exceeding 685 mg/kg (the low end of the 95% CI) are Grid 12 (685 mg/kg)[high diversity], Grid 97 (686 mg/kg), Grid 24 (717 mg/kg), Grid 74 (739 mg/kg), Grid 19 (741 mg/kg), Grid 1 (902 mg/kg) and Grid 129 (924 mg/kg). However, as described above, results of bioavailability studies, toxicity studies and benthic surveys for Round Lake indicate that exposure to copper in the sediments at grid locations 6, 9, 11, 12, 18, 19, 32, 35, 38, 39, 46, 84, 86, 98 and 130 would not be expected to adversely impact the growth, survival and reproduction of benthic organisms in sediments at these locations. Consequently, the risk evaluation of all lines of evidence indicate that exposure to copper in sediments in Grids 1, 3, 8, 10, 24, 26, 70, 74, 81, 85, 97, 114, 120 and 129 may adversely impact the growth, survival and reproduction of benthic organisms inhabiting Round Lake.

In conclusion, the potential areas of concern for causing possible risk to benthic organisms are sediments in grids along the western edge of the lake (Grids 1, 3, 8, 10, 24, 26, 70, and 81), a cluster of grids in the deepest part of the lake (Grids 74, 85 and 97) and a few isolated grids in the southern tip of the lake (Grids 114, 120 and 129). Following an analysis of the measures of effect and the supporting lines of evidence (toxicity, bioavailability, ability to impact the biodiversity of the benthic population and co-location with other COCs), results indicate that copper and zinc in sediments may adversely impact the benthic organisms inhabiting Round Lake at Grid locations 1, 3, 8, 10, 24, 26, 70, 74, 81, 85, 97, 114, 120 and 129; cadmium in sediments may adversely affect the benthic organisms inhabiting Round Lake at Grid locations 3, 8, 10, 26, 85 and 97; and chromium at Grid 10 may adversely impact the benthic organisms inhabiting Round Lake. Benthic survey results indicate that the metal contaminants are not impacting the benthic community in the northern part of the lake; especially considering that the species inhabiting the lake are species that would typically be found in this type of environment. Based on the characteristics of the sediments at Round Lake, it is apparent that the toxicity of the metals is being strongly influenced by the bioavailability of the metals due to binding with particulate sulfide and organic carbon. The absence of SEM/AVS and TOC data in areas of the lake lead to some uncertainty concerning the actual potential risk of the grids listed above.

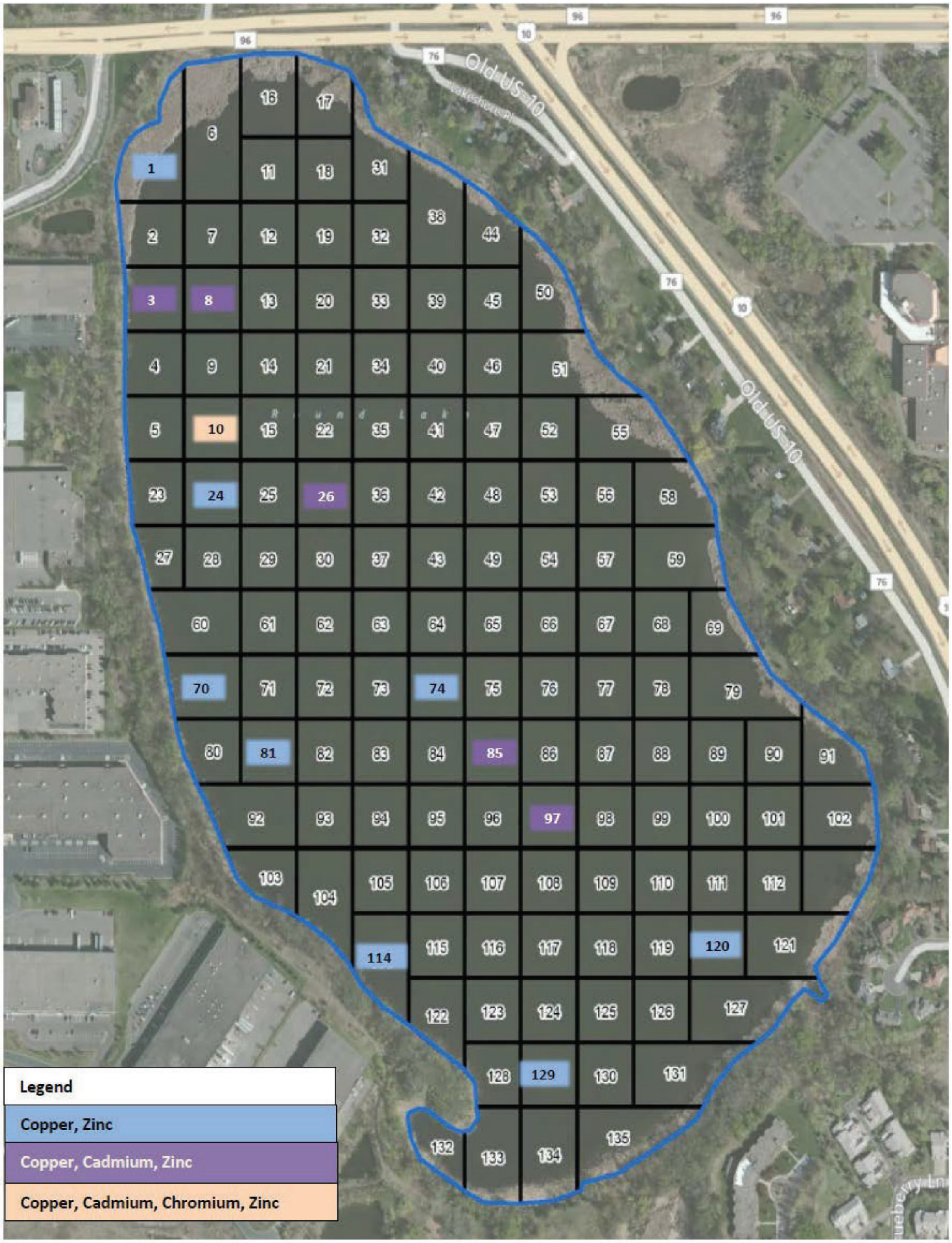


Figure 12. Map of Round Lake showing the grid locations for potential effects to benthic organisms

4.6. Risk Characterization for the Benthic Organism Endpoint for the Future Use Scenario

This supplemental ERA also evaluates the potential ecological risk to benthic invertebrates for the future use scenario based on the draft USFWS Round Lake Conceptual Management Plan (USFWS, 2012). The Tier II ERA evaluated ecological risk for the future use scenario using the USFWS optimum wildlife alternative, as proposed in their 1982 and 1998 management plans for Round Lake. This alternative calls for the implementation of actions to increase the ratio of emergent vegetation to open water from the existing 10:90 to 50:50 by a complete drawdown of the lake. The 2012 draft USFWS plan calls for water level management by periodically lowering the lake to a maximum drawdown elevation of 887.0 feet, which is only approximately 15% less than the normal lake level; thereby, exposing only a relatively small area of sediment. Figure 13 is a contour map showing the drawdown elevation which would only impact sediments around the edge of the lake represented by the outer grids on the 200 x 200 foot grid map in Figure 14. The results for the current use scenario, as described in Section 4.5, will be used as a basis for the consideration of potential adverse effects to the benthic community for the future use scenario since potential exposure was evaluated using the actual concentration detected in each grid at the 0.0 to 0.5 foot depth and would represent the maximum concentration of the COC that the benthic organisms would be exposed to during a drawdown scenario. Each COC was evaluated based on concentration data in each grid in relationship to its potential toxicity, bioavailability, ability to impact the biodiversity of the benthic population and co-location with other COCs. Exposure to the outer grids will be used to evaluate the potential adverse effects to benthic organisms in a future use scenario.

During the proposed drawdown of the lake, possible oxidation of exposed sediments or resuspension of deeper sediment to the surface might disrupt the metal sulfide binding in sediments. Prause et al. (1985) observed that in anoxic sediments, lead was not released over 50 days after resuspension. Calmano et al. (1994) conducted a long-term mobilization study in anoxic sediments and found that total releases of metals was small: cadmium (5%) > zinc > copper > lead (0.7%). In a study conducted by Simpson et al. (1998) the concentration of SEM cadmium, lead and zinc were not affected by resuspension and SEM concentrations of copper were observed to increase with increasing resuspension time. The investigators found that during resuspension into oxic waters, iron and manganese monosulfide phases, which are usually present in large excess to other metal sulfides, buffered the initial oxidation of trace metal sulfide phases. The effects of bioirrigation and bioturbation from benthic organism activity were buffered and trace metal sulfide phases remained predominantly unoxidized for some time. It is important to note that benthic organisms actively maintain internal concentrations of essential metals, such as copper and zinc, through the use of homeostatic mechanisms and inorganic metals are not biomagnified or accumulated over two or more trophic levels (Chapman et al., 1996). Consequently, exposing the sediments in the outer grids through a periodic drawdown should not result in increased exposure or potential for adverse impacts to the benthic organisms inhabiting the sediments of Round Lake. In addition, the high organic carbon content of the sediment may prevent drying and limit oxidation and the subsequent release of bound metals. The accumulation of algae and plant material that will collapse on the sediment during drawdown may also limit drying and oxidation (USACHPPM, 2004). However, based on results for the current use scenario risk evaluation and conservative assumptions, concentrations of cadmium, copper and zinc in outer

Grids 1, 3, 70, 81 and 114 could possibly result in adverse impacts to benthic organisms during the proposed USFW drawdown.

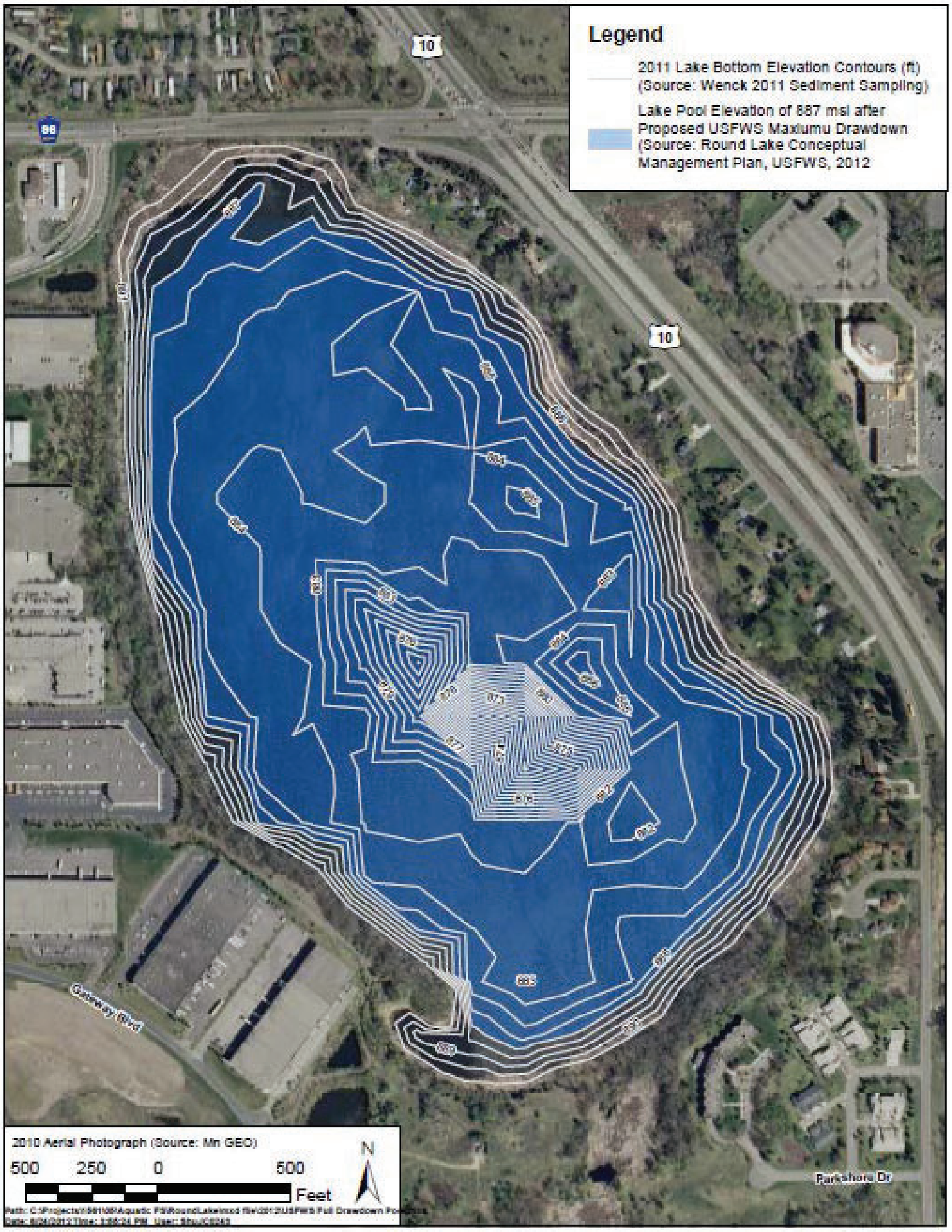


Figure 13. USFWS Proposed Maximum Drawdown Pool Elevation (887 ft) for Round Lake

4.7. Risk Characterization for the Aquatic Mammal and Waterfowl Assessment Endpoint for the Current and Future Use Scenarios

For the COCs, risk estimates consisting of a hazard ratio matrix for each receptor-COC combination were calculated (similar to that shown below). In this matrix, each value is a hazard quotient (indicated by the letters A-D), where the calculated exposure dose to the receptor is based on either the maximum or central tendency detected concentration in Round Lake sediment and is divided by either the no-effects or low-effects TRV. Hazard quotients designated as HQ 1 are based on the No-effect TRV (NOAEL), and HQ 2 values are based on the low-effect TRV (LOAEL). HQs less than one indicate that the calculated exposure is less than a selected level of concern.

	2011 Sediment Data	HQ 1	HQ 2
COC	Maximum Concentration	A	B
	Central Tendency Value	C	D

Hazard quotient A indicates the likelihood for the maximally exposed individual animal in the population to experience an average daily dose greater than the highest level associated with no observable health effects in a laboratory population. If A is equal to 1 or less, then no excessive hazard exists for the exposed population. Hazard quotient B indicates whether or not the possibility exists for a maximally exposed individual animal in the population to experience an exposure greater than the lowest level associated with observable health effects in a laboratory population. Hazard quotient C indicates the likelihood that, on average, individual animals among the exposed site population will experience a daily dose greater than the highest level associated with no observable health effects in a laboratory population. Hazard quotient D indicates the likelihood that, on average, individual animals among the exposed population will experience a daily dose greater than the lowest level associated with observable health effects in a laboratory population. The potential for effects in aquatic mammals [surrogate species muskrat] at Round Lake were evaluated based upon the possible contact and ingestion of sediments and vegetation containing the COCs by the animals during foraging and den building activities. A healthy number of muskrat dens have recently been present in the northern portion of the lake. These dens were surveyed by USACHPPM (Keith Williams and Matt McAtee) during the winter of 1994. Likewise, the potential for effects in waterfowl [surrogate species mallard] at Round Lake were evaluated based upon the possible contact and ingestion of vegetation, invertebrates, and sediments containing the COCs by waterfowl during feeding activities. The TRVs, based on NOAELs and LOAELs, are linked to the assessment endpoints because it can be inferred that an endpoint based on reproduction, or other health endpoint when reproductive data are missing, will indicate the effect of the chemical on the reproductive potential and productivity of the exposed animals. The line of evidence used to assess causality to the receptors was the calculation of an HQ derived by modeling the dietary dose and dividing that value by a TRV. Dietary doses were calculated based on the maximum measured COC concentration and the central tendency value in conjunction with dietary intake parameters that encompass the upper bound of potential exposures. The potential for adverse toxicological effects in aquatic mammals and waterfowl was estimated using comparisons of single point estimates of exposure and effect that highlight the variability in the collected site data. The potential for adverse changes in the assessment endpoint is inferred by comparing estimates of exposure to estimates of health effects in the form of hazard quotients. A ratio that exceeds unity triggers further consideration of the underlying scientific basis of the prediction.

Aquatic Mammal

Hazard quotients were calculated for the aquatic mammal (muskrat as surrogate) based on the exposure parameters described above (Section 4.4) and central tendency values for the COC sediment data from locations adjacent to the shore (outer grids 2011 sediment sampling) to correlate to potential exposure patterns based on den-building activities. HQ 1's were estimated using the mammal threshold toxicity value (NOAEL) and HQ 2's were estimated using the mammal low effect toxicity value (LOAEL) (see Table 12). Results indicate that exposure via the ingestion of aquatic vegetation and sediment does not exceed the threshold toxicity level when the estimates of metal and PCB residues are based on the central tendency concentration of COC in the sediment (HQ 1 > 1.0). Likewise, the low effect toxicity level is not exceeded when the central tendency concentration in the sediment is the basis for calculating the HQ 2. All muskrat HQ's where the central tendency value is the basis for estimating metal and PCB residues in aquatic vegetation and sediment are below 1.0 indicating that on average exposures in the muskrat population are less than exposures known to be associated with adverse health effects.

Table 28. Hazard Quotients for Potential Exposure of Muskrats to COC via Ingestion of Aquatic Vegetation and Sediment

COC	Sediment Concentration[a] (mg/kg)	Dose (mg/kg-d)	HQ 1[b]	HQ 2[c]
Cadmium	0.9	0.02	0.0	0.0
Chromium	23.8	0.11	0.0	0.0
Copper	67.5	1.65	0.1	0.1
Lead	29.9	0.35	0.0	0.0
Zinc	132.9	5.53	0.0	0.0
PCB	0.04	0.00	0.0	0.0

[a]Central tendency value (0.0 - 0.5 ft interval)
[b]HQ 1 was calculated using the NOAEL
[c]HQ 2 was calculated using the LOAEL

In addition, HQs were calculated for the muskrat based on the exposure parameters described above and the maximum concentration for the metal and PCB sediment data from locations adjacent to the shore (2011 sediment sampling). The muskrat HQ 1 and HQ 2 for cadmium, chromium, lead, zinc and PCBs are less than 1.0 when the maximum detected concentration is the basis for estimating potential exposure to metal residues in aquatic vegetation and sediment. However, results indicate that exposure via the ingestion of aquatic vegetation and sediment exceeds the threshold toxicity level (NOAEL) when the estimates of metal residues are based on the maximum concentration of copper in the sediment (HQ 1 > 1.0). Likewise, the copper HQ 2 is exceeded when the maximum measured copper concentration in the sediment and the LOAEL is the basis for calculating the HQ 2. Sediment samples in a selected few grids (5) along the northwestern/western shoreline indicate higher copper concentrations that could result in HQ values >1 compared to concentrations along the northeastern/eastern/southeastern shoreline. However, previous ecological risk assessments at Round Lake identified the northeast, east, and southeastern shoreline as favorable muskrat habitat based on physical and ecological conditions of the shoreline (water access and aquatic vegetation, cattails) and the observation of muskrat dens in these areas (USACHPPM, 2004). Consequently, potential adverse impacts to muskrat populations inhabiting Round Lake from exposure to copper in the sediments are not expected.

Table 29. Hazard Quotients for Potential Exposure of Muskrats to COC via Ingestion of Aquatic Vegetation and Sediment (maximum COC sediment concentration)

COC	Sediment Concentration[a] (mg/kg)	Dose (mg/kg-d)	HQ 1[b]	HQ 2[c]
Cadmium	10.5	0.19	0.2	0.0
Chromium	118.0	0.56	0.2	0.0
Copper	902.0	22.09	1.9	1.4
Lead	258.0	3.05	0.1	0.0
Zinc	841.0	34.98	0.2	0.1
PCB	0.5	0.0	0.0	0.0

[a]Maximum measured concentration (0.0 – 0.5 ft interval)
[b]HQ 1 was calculated using the NOAEL
[c]HQ 2 was calculated using the LOAEL

In conclusion, the assessment of potential impacts to aquatic mammals (muskrat as surrogate) indicates no exceedances of threshold or low effect levels based on exposure to sediment COC (metals and PCB) central tendency concentrations in the potential use areas for the muskrat (near shore area of Round Lake). The HQ's do not exceed unity indicating that on average exposures in the muskrat population are less than exposures known to be associated with adverse health effects. Likewise, the threshold effect and low effect levels based on the maximum measured concentration of cadmium, chromium, lead, zinc, and PCBs in sediment was not exceeded indicating the potential for adverse effects from these COC to muskrats is not likely. The threshold effect level and low effect level for copper was exceeded in a selected few sediment samples from areas along the northwestern/western shoreline based on the maximum concentrations indicating a potential for adverse effects. Since the northwestern/western shoreline is less favorable muskrat habitat and the potential for muskrat use in this area is less likely, exposure to sediments in this area is not likely. Although there are limited exceedances of effects levels for the muskrat based on the maximum copper sediment concentrations, there are no exceedances of effect levels based on measured sediment cadmium, chromium, lead, zinc and PCB concentrations suggesting the population exposure would be less than exposures known to be associated with adverse health effects. Furthermore, the potential for adverse effects to muskrats from copper would require repeated exposure to sediments along the northwestern/west shoreline which is not favorable muskrat habitat. Overall, the risk of adverse effects to aquatic mammals from exposure to COCs in sediments of Round Lake is not expected. Furthermore, the calculations of muskrat HQ's based on the maximum sediment COC concentrations and conservative exposure parameters provides an assessment of the risk of adverse effects that may result from exposures to sediments during a proposed future drawdown of the lake. Future risks to aquatic mammals from exposure to sediment COCs are not expected to exceed the HQ's based on the maximum sediment COC concentrations presented in this assessment.

Waterfowl

The potential for adverse effects in waterfowl (i.e., mallard) at Round Lake were evaluated based upon the possible contact and ingestion of sediments and vegetation containing the COCs by the waterfowl during feeding, resting, migration and/or breeding. Hazard Quotients (HQ) were calculated for the mallard based on the exposure parameters described above and the central tendency value for the COC sediment data from all grid locations (2011 sediment sampling). HQ 1's were estimated using the waterfowl threshold toxicity value (NOAEL) and HQ 2's were estimated using the waterfowl low effect toxicity value (LOAEL). Results indicate that exposure via the ingestion of aquatic vegetation, benthic invertebrates, and sediment does not exceed the mallard threshold toxicity level when the estimates of COC residues are based on the central tendency concentration in the sediment (HQ 1 > 1.0). Furthermore, the lead HQ 2 is not exceeded when the central tendency concentration in the sediment is the basis for calculating the HQ 2. All mallard HQ's where the central tendency value is the basis for estimating metal residues in aquatic vegetation, invertebrates, and sediment are below 1.0.

The assessment of potential impacts to waterfowl (i.e., mallard) indicates no exceedances of threshold or low effect levels based on exposure to sediment COC (metals and PCB) central tendency concentrations in Round Lake. The HQ's do not exceed unity indicating that on average exposures in the mallard population are less than exposures known to be associated with adverse health effects. Likewise, there are no exceedances of the low effect level based on exposure to the maximum concentrations of COC measured in sediment in Round Lake. The threshold effect level based on the maximum measured concentration of cadmium, chromium, copper, zinc, and PCBs in sediment was not exceeded indicating the potential for adverse effects to mallards is not likely. The threshold effect level for lead was exceeded in sediment samples from grids along the north/northwest region and a few deeper areas in the middle of Round Lake. It is noted that there are no exceedance of the HQ 2 (low effect level) for mallard when the maximum lead concentration is the basis for estimating exposure via sediments suggesting the population exposure would be less than exposures known to be associated with adverse health effects. Furthermore, the majority of the lead concentrations in the sediment (> 92% of grids) do not result in exceedance of the threshold effect level for the mallard based on estimates of exposure via ingestion of aquatic vegetation, benthic invertebrates, and sediment. Overall, potential adverse effects to the mallard from exposure to COC in sediment are not likely.

Table 30. Hazard Quotients for Potential Exposure of Mallards to COC via Ingestion of Aquatic Vegetation, Benthic Invertebrates, and Sediment

COC	Sediment Concentration[a] (mg/kg)	Dose (mg/kg-d)	HQ 1[b]	HQ 2[c]
Cadmium	1.1	0.00	0.0	0.0
Chromium	32.8	0.05	0.1	0.0
Copper	101.6	0.92	0.0	0.0
Lead	29.5	0.34	0.3	0.0
Zinc	174.9	2.64	0.2	0.0
PCB	0.05	0.00	0.0	0.0

[a]Central tendency value (0.0 – 0.5 ft interval)
[b]HQ 1 was calculated using the NOAEL
[c]HQ 2 was calculated using the LOAEL

Also, Hazard Quotients (HQ) were calculated for the mallard based on the exposure parameters described above and the maximum concentration of the COC sediment data from all grids (2011 sediment sampling). HQ 1's were estimated using the waterfowl threshold toxicity value (NOAEL) and HQ 2's were estimated using the waterfowl low effect toxicity value (LOAEL). The mallard HQ 1 and HQ 2 for cadmium, chromium, copper, zinc, and PCB are less than 1.0 when the maximum detected concentration is the basis for estimating potential exposure to metal residues in aquatic vegetation, invertebrates, and sediment. Likewise, the lead low-effect level (LOAEL) for the mallard is not exceeded when the maximum measured lead concentration in the sediment is the basis for calculating the HQ 2. However, results indicate that exposure via the ingestion of aquatic vegetation, benthic invertebrates, and sediment exceeds the mallard threshold toxicity level (NOAEL) when the estimates of residues are based on the maximum concentration of lead in the sediment (HQ 1 > 1.0). Results from the 2011 sediment sampling indicate the lead sediment concentrations are considerably less for the majority of the lake when compared to other areas of lake (in particular the north/northwestern region and a few deeper areas in the middle region). The measured lead concentration in the north (Grids 11, 16, 18) and west (Grids 3, 8, 10, 26) adjacent to the shoreline and deeper sediments in the lake center region (Grids 85, 97, 98) are higher than other areas of Round Lake thus resulting in exceedance of the HQ 1. It is noted that there are no exceedance of the HQ 2 (low effect level) for mallard when the maximum lead concentration is the basis for estimating exposure via sediments suggesting the population exposure would be less than exposures known to be associated with adverse health effects. Furthermore, the majority of the lead concentrations in the sediment (> 92% of grids) do not result in exceedance of the HQ 1 for the mallard based on estimates of exposure via ingestion of aquatic vegetation, benthic invertebrates, and sediment.

Table 31. Hazard Quotients for Potential Exposure of Mallards to COC via Ingestion of Aquatic Vegetation, Benthic Invertebrates, and Sediment (maximum COC sediment concentration)

COC	Sediment Concentration[a] (mg/kg)	Dose (mg/kg-d)	HQ 1[b]	HQ 2[c]
Cadmium	26.6	0.15	0.1	0.0
Chromium	118	0.49	0.5	0.1
Copper	924	8.32	0.2	0.1
Lead	258	2.97	2.6	0.3
Zinc	854	12.87	0.9	0.1
PCB	0.89	0.01	0.0	0.0

[a]Maximum measured concentration (0.0 – 0.5 ft interval)
[b]HQ 1 was calculated using the NOAEL
[c]HQ 2 was calculated using the LOAEL

In conclusion, the mallard hazard quotients for cadmium, chromium, copper, zinc, and PCB are less than 1.0 when the maximum detected concentration and the central tendency values are the basis for estimating potential exposure to metal residues in aquatic vegetation, invertebrates, and sediment. However, results indicate that exposure via the ingestion of aquatic vegetation, benthic invertebrates, and sediment exceeds the mallard threshold toxicity level (NOAEL) when the estimates of residues are based on the maximum concentration of lead in the sediment (HQ 1 > 1.0). The measured lead concentration in the north (Grids 11, 16, 18) and west (Grids 3, 8, 10, 26) adjacent to the shoreline and deeper sediments in the lake center region (Grids 85, 97, 98) are higher than other areas of Round Lake thus resulting in exceedance of the HQ 1, indicating that the potential for risk of adverse effects to waterfowl in these areas of the lake. It is noted that there is no exceedance of the HQ 2 (low effect level) for mallard when the maximum lead concentration is the basis for estimating exposure via sediments suggesting the population exposure would be less than exposures known to be associated with adverse health effects. The majority of the lead concentrations in the sediment (> 92% of grids) do not result in exceedance of the HQ 1 for the mallard based on estimates of exposure via ingestion of aquatic vegetation, benthic invertebrates, and sediment indicating that the risk of adverse effects to mallard population are not expected. Furthermore, the calculations of mallard HQ's based on the maximum sediment COC concentrations and conservative exposure parameters provides an assessment of the risk of adverse effects that may result from exposures to sediments during a proposed future drawdown of the lake. Future risks to waterfowl from exposure to sediment COCs may be expected in selected grids in the north and west adjacent to the shoreline (Grids 3, 8, 10, 11, 16, 18, 26) and selected grids in the deeper sediments in the lake center region (Grids 85, 97, 98).

4.8. Risk Characterization for the Piscivorous Species Assessment Endpoint

For the piscivorous mammal (mink) and piscivorous birds (great blue heron, belted kingfisher, and bald eagle), hazard quotients (HQ) were calculated based on the maximum concentration of measured PCBs in fish tissue as the ingested dose and assuming 100% of the diet of each receptor species was the brown and black bullhead and green sunfish (filet for mink and whole fish for great blue heron, bald eagle and belted kingfisher). Fish were collected from Round Lake in December 2012 and tissues (whole fish, filet, and filet with skin) were analyzed for PCBs. HQ 1's were estimated using the mammal or avian PCB threshold toxicity value (NOAEL) and HQ 2's were estimated using the mammal or avian PCB low effect toxicity value (LOAEL). No HQ 1 or 2 values were >1 for any of the receptor species at the maximum measured concentration and assuming a 100% fish diet, indicating that under this conservative scenario, the potential for adverse effects to piscivorous avian and mammalian species is not likely from the consumption of fish at Round Lake.

Table 32. Hazard Quotients for Potential Exposure of Piscivorous Mammals and Piscivorous Birds to PCBs via Ingestion of Fish from Round Lake

Modeled Receptor	Fish Tissue Diet	PCB Concentration[a]	HQ 1[b]	HQ 2[c]
Mink	Filet	0.04	0.06	0.01
Great Blue Heron	Whole Fish	0.26	0.27	0.05
Belted Kingfisher	Whole Fish	0.26	0.10	0.02
Bald Eagle	Whole Fish	0.26	0.55	0.11

[a]Maximum PCB concentration (mg/kg) in fish tissue (2012 MN DNR fish sampling)
[b]HQ 1 was calculated using the NOAEL
[c]HQ 2 was calculated using the LOAEL

In conclusion, hazard quotients (HQ) calculated for piscivorous species based on the maximum concentration of measured PCBs in fish tissue as the ingested dose and assuming 100% of the diet of each receptor species was fish from Round Lake were <1. Under this conservative scenario, the potential for risk of adverse effects to piscivorous avian and mammalian species is not expected from the consumption of fish at Round Lake. Furthermore, the calculations of HQ's for these species based on the maximum-exposed individual provides an assessment of the potential adverse effects that may result from exposure to PCBs in fish tissue during a proposed future drawdown of the lake. Future risks to piscivorous avian and mammalian species are not expected since fish tissue data indicates that bioaccumulation of PCBs up the food chain does not occur at levels that are toxic to receptor species.

4.9. Overall Conclusions

A compilation of the areas within Round Lake that represent the potential for risk to ecological receptors is depicted in Figure 14. The highlighted grids on this map indicate the areas where detected concentrations of the final COCs may adversely impact the ecological receptors: benthic invertebrates and waterfowl (represented by the mallard).

- Benthic invertebrates – Grids 1, 3, 8, 10, 24, 26, 70, 74, 81, 85, 97, 114, 120 and 129 – potential exposure to cadmium, chromium, copper and zinc
- Waterfowl (mallard) – Grids 3, 8, 10, 11, 16, 18, 26, 85, 97 and 98 – potential exposure to lead

Based on the physical and chemical characteristics of Round Lake, two processes, lake succession (aging) and eutrophication, determine the environmental conditions of the lake as well as potential exposure to ecological endpoints. Lake succession (aging) is the natural process by which a lake fills with allogeneic erosional materials. Eutrophication is the process of increased nutrient input (productivity) that can be accelerated by human activities, including stormwater runoff. There are no natural streams discharging to Round Lake; water level is recharged by precipitation and stormwater runoff. There is also no natural outlet; the only outlet is a concrete structure with stoplogs to allow water level control during storm events. There is also limited connectedness to groundwater, an unconfined perched aquifer with glacial till below serves as an aquitard. Consequently, Round Lake is a depositional environment with sediment loading from stormwater runoff events. The sediment loading from stormwater events contributes to the natural recovery process by reducing the contaminants availability. The ITRC Contaminated Sediments Team emphasizes the importance of bioavailability in risk assessments and in the remedial decision making process. The ITRC guidance notes the relationship between sediment contaminant concentrations and risk from exposure is not linear due to bioavailability considerations which may in some instances only result in a fraction of the contaminant being available to cause harm to ecological receptors (ITRC, 2011). The ITRC team analyzed several sites with metal contaminated sediments and found that bioavailability (SEM/AVS and TOC), toxicity testing, and benthic community surveys played a significant role in determining risk, the need for action, and the use of MNR.

Round Lake exhibits several characteristics noted by EPA to be conducive to the natural recovery process. One consideration in the natural recovery process is the control of any significant sources of contaminants. EPA guidance notes that "MNR is likely to be effective most quickly in depositional environments after source control actions and active remediation of any high risk sediment have been completed" (Contaminated Sediment Remediation Guidance for Hazardous Waste Sites, page 4-11, OSWER 9355.0-85, EPA-540-R-05-012, December 2005). Id. at 4-11). Although previously there have been releases of contaminants from production operations at TCAAP to Round Lake via storm water runoff in the storm sewer, the sources of the releases has been corrected and the cessation of production operations at TCAPP further eliminates the potential for any future releases. Other considerations that support the natural recovery process include:

- Anticipated land uses or new structures are not incompatible with natural recovery
- Natural recovery processes have a reasonable degree of certainty to continue at rates that will contain, destroy, or reduce the bioavailability or toxicity of contaminants within an acceptable time frame
- Sediment bed is reasonably stable and likely to remain so
- Sediment is resistant to resuspension (e.g., cohesive or well-armored sediment)
- Expected human exposure is low and/or can be reasonably controlled by institutional controls (*Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*, page 4-3, OSWER 9355.0-85, EPA-540-R-05-012, December 2005).

Land use at Round Lake is anticipated to remain as an USFWS wildlife refuge. No change in land use is anticipated which would be incompatible with MNR. Annual precipitation averages 29 inches/year resulting in an estimated average annual runoff to the lake of 200 to 300 acre-feet/year (excludes precipitation) [Wenck 2012]. Very high sedimentation rates of >1.5 cm/yr for the 20th century have been reported for the lake (Engstrom, 2012). Round Lake is overall a shallow depositional lake where the sediment is stable and resuspension is unlikely. The lake occupies approximately 125 acres with a maximum depth of 26' at the south-central end. However, less than 5% of the lake basin is more than 20' in depth; the majority is a relatively flat shoal averaging approximately 4.5' in depth (USFWS, 1992).

Typically, shallow lakes <20 in depth do not exhibit mixing and turnover. USFWS does not currently allow fishing at Round Lake; and, a fish consumption advisory could be implemented by the USFWS for any future fishing activity if needed.

In some situations the natural recovery process may be occurring; however, the natural recovery process may be unable to reduce risks sufficiently within an acceptable time frame. In these situations, the natural recovery process can be accelerated or enhanced by applying a thin clean layer of material, usually as little as few inches. In most case natural material is recommended approximating common substrates found in the area. Such enhancement is distinguished from capping in that the purpose of the clean layer is to mix with the contaminated sediment. The addition of the thin clean layer of material is not designed to isolate the contaminants as in capping (where cap thickness can range up to several feet). Enhancement of degradation can also be facilitated by using additives to speed up the natural recovery (Contaminated Sediment Remediation Guidance for Hazardous Waste Sites, page 4-11, OSWER 9355.0-85, EPA-540-R-05-012, December 2005).

Figure 14. Map of Round Lake showing the grid locations of potential risks to all ecological receptors

5. UNCERTAINTY ANALYSIS

For this supplemental ERA of Round Lake, a qualitative description of the uncertainty associated with estimates of potential adverse effects to ecological endpoints is provided. One area of uncertainty is attributed to the natural heterogeneity of the biota of the eutrophic natural ecosystem of Round Lake. This shallow lake system contains diverse populations of species interacting in complex ways and these interactions are known to greatly influence the distribution and abundance of species. As such, indices of ecosystem function may be better indicators of change in a lake ecosystem in response to adverse conditions than data for individual species. This supplemental ERA provides a combination of ecosystem indices and assessments of individual ecological receptors (representative species) to assess potential hazards to selected ecological populations as well as individuals. The combined assessment approach was selected to provide a range of output while evaluating both ecological populations and representative individual receptors.

Exposure Assessment

For Round Lake, sediment data COC concentrations were measured at 135 discrete grid locations and at depth intervals of up to six feet to determine the spatial distribution of metals and PCB COC. The systematic sampling and analysis of Round Lake sediment in 2011 provides adequate coverage of the entire lake and the resulting data are of sufficient quality (Wenck 2012) to support the ecological risk assessment. Descriptive statistical analyses of the 2011 data provide the central tendency value and the maximum measured value for each COC. Both the central tendency and maximum values were used in the exposure assessment to represent a range of potential COC concentrations for ecological endpoints. The assessment endpoints of the aquatic mammal, waterfowl, piscivorous mammal, and piscivorous birds were not directly measured. Estimates of daily intakes which are the amount of the COC (metals and PCBs) ingested from sediment and aquatic food (vegetation and benthic invertebrates) was modeled for these endpoints. Model parameters selected represent a seasonal range of conservative estimates for ingestion of aquatic vegetation and invertebrates as well as incidental exposure to sediment for the aquatic mammal (mink) and waterfowl (mallard). Likewise, the exposure assessment for the piscivorous mammal and bird assumed that ingestion of aquatic food was 100% fillet or whole fish from Round Lake; thus, providing a conservative estimate for the potential ingestion of PCBs via fish tissue.

Toxicity Assessment

For the ecological endpoints assessed, the ability to extrapolate toxicity at the individual level to that at the ecosystem level is limited as organisms responses to stressors in the environment can differ significantly from what would be expected based upon single-species laboratory studies. The fitness of organisms in natural ecosystems is dependent upon their interactions with other indigenous species and is critical to meaningful predictions regarding responses to stressors. The organisms also experience the multiple mechanisms driving the ecosystem's response to stressors. Likewise, extrapolation from high-dose short term toxicity tests to lower, long term exposures typically encountered in the field is a continued research area of ecotoxicologist. As such, there is bias in the use of acute toxicity databases which should be considered in predicting the consequences of sublethal exposures over extended time scales. Also, inherent uncertainty exists in extrapolation of observed adverse effects in a homogenous exposed test population to those in more variable population in an environmental setting.

For the aquatic mammal, waterfowl, piscivorous mammal, and piscivorous fish endpoints, the likelihood for effects was estimated with the use of toxicity reference values (TRVs) of no-observable and lowest-observable effects concentration from laboratory studies. Uncertainty (typically a factor of 10) is included in the derivation of these published TRVs and the use of both the no-observable and lowest-observable effects concentrations encompasses an assessment of the upper bound of potential adverse effects for these endpoints.

For benthic organism endpoint, each COC was evaluated based on sediment concentration data in relationship to sediment quality criteria, its potential toxicity, bioavailability, ability to impact the biodiversity of the benthic population, and co-location with other COC. The inclusion of additional lines of evidence in this assessment are essential as the prediction of the toxicity of metal contaminants in sediments to benthic organisms is challenging due to the strong influence of the properties of the sediments. The level of toxicity for each COC varies greatly among benthic species in the published literature. Thus, the toxicity of metal contaminants in sediments is dependent on the bioavailability of the metals in both the sediment and water (porewater, burrow water, overlying water) phases and on the sensitivity of the organism to the metal exposure. The bioavailability of the metals in sediments is controlled by (1) the metal binding with particulate sulfide, organic carbon and iron hydroxide phases; (2) the sediment-water partitioning relationships; (3) organism physiology (uptake rates and assimilation efficiencies); and (4) organism behavior (feeding selectivity and burrow irrigation). In addition, sediment quality guideline values are not normalized to account for the presence of AVS and TOC in sediments; consequently, they do not consider the potential bioavailability of the contaminants to aquatic organisms. Sediment quality guidelines alone do not establish a cause-effect relationship and they do not consider bioavailability; consequently, resulting in a conservative prediction of when toxic effects may occur. Exceedance of a sediment guideline indicates an increased likelihood of toxic effects, but it is not proof of cause and it cannot be assumed that the contaminant present in excess is solely responsible for the observed effects.

Finally, exceedance of criteria or standards does not necessarily imply causation, because the regulatory values are intended to be safe levels and not a number which signifies absolute occurrence of adverse effect. Benchmarks found in published literature may be biased in that they are generally based on laboratory studies in which the forms of the chemical used in the tests are likely to be more toxic than that found at contaminated sites (in the field), combined toxic effects are not observed, the test species and test media may not be representative of the study site, and lab test conditions may not be representative of environmental and ecological processes of field conditions.

Hazard Quotients

The potential for adverse changes in ecological assessment endpoints was inferred by comparing estimates of exposure to estimates of health effects in the form of hazard quotients. This assessment compares point estimates of exposure with a toxicity metric that is considered likely not to result in an adverse effect (i.e., the TRV). Both the maximum COC concentration and central tendency value represent the sediment concentrations to support the range of dose estimates for the ecological endpoints. The calculated exposure dose to the receptor is divided by either the no-effects or low-effects TRV. Hazard quotients designated as HQ 1 are based on the No-effect TRV (NOAEL), and HQ 2 values are based on the low-effect TRV (LOAEL). Interpretation is based on whether exposure is above, at, or below the TRV (i.e., whether the hazard quotient is above, at, or below 1) and describes the likelihood that the exposed individual animal in the population may experience an average daily dose greater than the highest or lowest level associated with no observable effect or lowest level health effects in a laboratory population. In this supplemental ERA, HQs were calculated for the maximally exposed animal and the central tendency dose to provide a range of HQs while including the upper bound estimate (maximum measured COC concentration). The potential for uncertainty in the estimation of the exposure and toxicity metrics must be considered by decision makers during interpretation of hazard quotients especially when the HQ is close to a value of 1.

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APPENDIX A

LEGAL/REGULATORY FRAMEWORK FOR CERCLA ECOLOGICAL RISK ASSESSMENTS

Appendix A: Legal/Regulatory Framework For CERCLA Ecological Risk Assessments

A.1 Site Background

TCAAP was placed on the National Priorities List as the New Brighton/Arden Hills Superfund Site in 1983. Round Lake is located off the installation but was part of TCAAP until 1974 when the U.S. Army transferred the lake to the U.S. Fish and Wildlife Service. It is currently under the control of the USFWS as part of the Minnesota Valley National Wildlife Refuge System with the use at transfer for waterfowl management. The specific source of TCAAP-related COCs in Round Lake was a storm sewer pipe that was connected to Building 502 at Site I. Part of the Site I facility had been used to produce artillery shell forgings. The production forges were cooled by water that was discharged to floor drains, along with water used in general cleanup operations. The floor drains were connected to the storm sewer that emptied into Round Lake. Due to the nature of the production process used at Building 502, PCBs and metals were expected to be the main COPCs. In 1969, the floor drains were disconnected to the storm sewer; consequently, no TCAAP-related source of contamination to Round Lake remains. This appendix will outline the various legal and regulatory drivers for any remedial action at Round Lake.

A.2 CERCLA Process

The Comprehensive Environmental Response, Compensation and Liability Act ("CERCLA") provides the President authority to respond to releases of hazardous substances [42 U.S.C. §9604(a)]. Executive Order 12580 delegates the President's authority under various CERCLA sections, including §9604(a), to the Secretary of the Department of Defense ("DOD"). See E.O. 12580, Sec. 2(d). DOD is considered the "lead agency" to plan and implement response actions under the NCP (40 CFR §300.5). The Defense Environmental Restoration Program ("DERP") states that the Secretary of DOD "shall carry out (in accordance with . . . CERCLA) all response actions with respect to releases of hazardous substances from . . . [e]ach facility or site owned by . . . the United States and under the jurisdiction of the Secretary" [10 U.S.C. §2701(c)]. DERP also provides that DERP activities are to be carried out subject to, and in a manner consistent with, section 120 of CERCLA [10 U.S.C. §2701(a)].

Under the CERCLA statute, two primary mandates establish the underlying legal and regulatory requirements identified and discussed in this document.

First, CERCLA §121(d)(1) requires remedial actions to attain a degree of cleanup that assures protection of human health and the environment [42 U.S.C. §9621(d)(1)]. This CERCLA requirement is implemented through means of a risk assessment, which based on site specific exposures, identifies contaminant exposure pathways that present either a current or potential future unacceptable risk. When such a risk is identified, remedial or removal action is required to address the unacceptable risk.

Protection of human health and the environment is demonstrated through the human health risk assessment and the ecological risk assessment. A myriad of provisions in the National Contingency Plan (NCP) regulations outline a very detailed process to access the risk to human health including a protective range for human health. In addition, a number of guidance documents provide a prescriptive approach in the development of the human health risk assessment and its use in establishing remedial goals for the site. For ecological risk assessment very little guidance is provided in the NCP as to how ecological risk assessment should be conducted. Several guidance documents outline the process to follow when conducting an ecological risk assessment. This process and the considerations involved in ecological risk assessment are discussed below.

Second, CERCLA §121(d)(2)(A) requires that on-site remedial actions must meet the standards and criteria that are otherwise legally applicable to the substance, pollutant, or contaminant or that are relevant and appropriate under the circumstances [42 U.S.C. § 9621(d)(2)(A)]. The compliance with ARARs mandate arises under CERCLA 121(d)(2)(A) when an on-site remedial action is required. ARARs are only triggered when a remedial action is required because of unacceptable risk rather than the initiation of the CERCLA process[1].

In the Remedial Investigation/Feasibility Study (RI/FS) if an unacceptable risk does exist, the mandate for ARAR compliance will be triggered by the remedial action. CERCLA provides that on-site remedial actions must meet the federal or more stringent state standards, requirements, and criteria that are otherwise legally applicable to the hazardous substance, pollutant or contaminant remaining onsite at the completion of the remedial action or that are relevant and appropriate under the circumstances of the release or threatened release to the substance, pollutant, or contaminant remaining onsite at the completion of the remedial action. [42 U.S.C. § 9621(d)(2)(A)].

For sediment, there are no promulgated standards, at either the state or federal level. In the absence of federal- or state-promulgated regulations, there are many criteria, advisories, guidance values, and proposed standards that are not legally binding but may serve as useful guidance for setting protective cleanup levels. These are not potential ARARs but are to-be-considered (TBC) guidance [40 CFR 300.400(g)(3)].

Potential ARARs and TBCs for Round Lake are discussed below.

[1] When no action is required to reduce, control or mitigate exposure, because the site conditions as they exist are already protective of human health and the environment, compliance with ARARs is not required (*ARAR's Q's & A's: General Policy, RCRA, CWA, SDWA, Post-ROD Information, and Contingent Waivers*, U.S. EPA OSWER Quick Reference Fact Sheet, Publication 9234.2-01/FS-A, June 1991). This EPA interpretation originates directly from the statutory language of the CERCLA statute which states "… the remedial action selected under section 9604 of this title or secured under section 9606 of this title shall require, at the completion of the remedial action, a level or standard of control for such hazardous substance, or pollutant, or contaminant which at least attains such legally applicable or relevant and appropriate standard, requirement, criteria, or limitation" [42 U.S.C. §9621(d)(2)(A)]. This statutory mandate arises in the remedy selection portion of the statute. In order to trigger the ARARs requirement, an unacceptable risk must be posed at the site which requires a response for which a remedial action must be selected. In EPA's guidance on the preparation of records of decisions, EPA states that no CERCLA §121 determinations are necessary when no remedial action is being selected (*A Guide to Preparing Superfund Proposed Plans, Records of Decision, and other Remedy Selection Decision Documents*, U.S. EPA OSWER Guidance, EPA 540-R-98-031, OSWER 9200-1-23P, page 8-6, July 1999).

A.3 Ecological Risk Assessment Process under CERCLA

In promulgating the NCP, EPA specifically noted that a uniform process should be used in the CERCLA process to develop risk assessments, including ecological risk. In addition, in order to promote efficiency and consistency, EPA provided extensive guidance for characterizing site specific risks and identifying preliminary goals to protect human health and the environment (*NCP Final Rule preamble,* 55 FR 8709-8710, March 8, 1990). EPA guidance directs that all ecological risk assessments should generally be performed at every site following the eight step process described in Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, ERAGS, EPA 540-R-97-006, OSWER Directive # 9285.7-25, June 1997 (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 1, OSWER Directive 9285.7-28P, October 7, 1999).

The eight steps of ecological risk assessment are as follows:

1. The screening-level problem formulation process and ecological effects evaluation
2. Estimating exposure levels and screening for ecological risks (the last two phases of the screening-level ecological risk assessment)
3. Problem formulation establishing the goals, breadth, and focus of the baseline ecological risk assessment. (It also establishes the assessment endpoints, or specific ecological values to be protected)
4. Establish the measurement endpoints (completing the conceptual model), the study design, and data quality objectives based on statistical considerations.
5. Field verification of the sampling plan
6. Site investigation and analysis phase
7. Risk characterization (including risk estimation and risk description)
8. Risk management

Steps 1 and 2 are often referred to as the screening level risk assessment. The screening level risk assessment provides a conservative estimate of ecological risk using a comparison of conservative toxicological based numbers with site media concentrations (The Role of Screening-Level Risk Assessments and Refining Contaminants of Concern in Baseline Ecological Risk Assessments Eco Risk Assessment Guidance, page 3, EPA OSWER Guidance 9345.0-14, EPA 540/F-01/014, June 2001). Screening level risk assessment are conducted to (1) estimate the likelihood that a particular ecological risk exists; (2) identify the need for site specific data collection, or (3) focus the site specific ecological risk assessment (*Id.* at page 2).

Steps 3 through 7 outline the process for a baseline ecological risk assessment. This process involves definition of potentially complete ecological pathways, the toxic mechanisms of the contaminants, and potential receptors. Using the conceptual model, ecological effects are measured in defined ecological assessment endpoints. Endpoints and measures selected should be "ecologically relevant to the site; i.e., important to sustaining the ecological structure and function of the local populations, communities and habitats present at or near the site", and" include species that are exposed to and sensitive to site-related contaminants" (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999). In characterizing ecological risks, lines of evidence are used to evaluate risk including toxicity tests, plant and animal residue data, bioavailability factors, assessment of existing impacts at site, media chemistry, reference site data, and risk calculation comparing exposures estimated for the site with toxicity values from literature. Using the lines of evidence approach, effects on individuals and group of individuals can be extrapolated to local populations and communities (*Id.* at page 3). EPA notes "[t]he performance of multi-year field studies at Superfund sites to try to quantify or predict long-term changes in local populations is not necessary for appropriate risk management decisions … Data from discrete field and laboratory studies, if properly planned and appropriately interpreted, can be used to estimate local population or community-level effects." (*Id.*). EPA points out in guidance that "Typically, no one line of evidence can stand on its own" (Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, page 4-6, EPA OSWER Guidance 9285.7-25, EPA 540-R-97-006, June 1997). When ecological risk assessments involve more than one line of evidence, strength of evidence approach is used, using professional judgment, to integrate the information to support a conclusion (*Id.* at page 7-2). When some lines of evidence are conflicting, professional judgment is used to determine which data should be considered more reliable or relevant (*Id.* at page 4-3). EPA acknowledges that unlike the detailed guidelines and risk range established for characterizing human health risk, detailed guidelines for site-specific ecological risk assessment do not exist (*Id.* at page I-2). Ecological risk is evaluated qualitatively and quantitatively, which inherently has some uncertainty (*Id.* at page I-3 and 7-5).

The goal of the Superfund is to reduce ecological risk to levels "that will result in the recovery and/or maintenance of healthy local populations/communities of ecological receptors that are or should be present at or near the site" (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 2, OSWER Directive 9285.7-28P, October 7, 1999). A further ecological risk management principle states "Contaminated media that are expected to constrain the ability of local populations and/or communities of plants and animals to recover and maintain themselves in a healthy state at or near the site (e.g., contamination that significantly reduces diversity, increases mortality, or diminishes reproductive capacity) should be remediated to acceptable levels." (*Id.*at page 4). In evaluating ecological risks, the site should be characterized in terms of "1) magnitude; i.e., the degree of the observed or predicted responses of receptors to the range of contaminant levels, 2) severity; i.e., how many and to what extent the receptors may be affected), 3) distribution; i.e., areal extent and duration over which the effects may occur, and 4) the potential for recovery of the affected receptors." (*Id.*).

Step 8, the final step of ecological risk assessment, is risk management. Although part of the ecological risk assessment, risk management is distinct from risk assessment. In risk management the results of the risk assessment are combined with other considerations of the NCP to make decisions. EPA notes the difficulty in establishing remediation goals for ecological receptors and notes they are best established on a site-specific basis:

> "Establishing remediation goals for ecological receptors is considerably more difficult than establishing such goals for the protection of human health due to the paucity of broadly applicable and quantifiable toxicological data. Further, owing to the large

variation in the kinds and numbers of receptor species present at sites, to their differences in their susceptibility to contaminants, to their recuperative potential following exposure, and to the tremendous variation in environmental bioavailability of many contaminants in different media, protective exposure levels are best established on a site-specific basis" (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 2, OSWER Directive 9285.7-28P, October 7, 1999).

Due to this site specific nature, EPA cautions on the use of a value alone in making remedial decisions:

> "In short, differences in assumptions and uncertainties, coupled with non-scientific considerations called for in various environmental statutes, can clearly lead to different risk management decisions in cases with ostensibly identical quantitative risks; i.e, the "number" alone does not determine the decision." (Guidance on Risk Characterization for Risk Managers and Risk Assessors, Memo from Henry Habicht II, Deputy Administrator to Assistant and Regional Administrators, attachment page 18, February 26, 1992)

EPA guidance indicates the following factors should be considered in risk management in the context of the NCP threshold and balancing criteria:

- "the magnitude of the observed or expected effects of site releases and the level of biological organization affected (e.g., individual, local population or community),
- the likelihood that these effects will occur or continue,
- the ecological relationship of the affected area to the surrounding habitat,
- whether or not the affected area is a highly sensitive or ecologically unique environment,
- the recovery potential of the affected ecological receptors and expected persistence of the chemicals of concern under present site conditions, and
- short- and long-term effects of the remedial alternatives on the site habitats and the surrounding ecosystem" (*Id.*at page 7).

In documenting the ecological risk assessment, EPA guidance states

> "For decision-makers, a complete characterization (key descriptive elements along with numerical estimates) should be retained in all discussions and papers relating to an assessment used in decision-making. Fully visible information assures that important features of the assessment are immediately available at each level of decision-making whether risks are for evaluating acceptable or unreasonable." (Guidance on Risk Characterization for Risk Managers and Risk Assessors, Memo from Henry Habicht II, Deputy Administrator to Assistant and Regional Administrators, attachment page 18, February 26, 1992).

> "[I]nformed EPA risk assessors and managers need to be completely candid about confidence and uncertainties in describing risks and in explaining regulatory decisions.

Specifically, the Agency's risk assessment guidelines call for full and open discussion of uncertainties in the body of each EPA risk assessment, including prominent display of critical uncertainties in the risk characterization." (*Id.*at page 2).

A.4 ARAR and Potential TBC Guidance

For Round Lake, the only potential associated promulgated standards for cleanup levels would be the Minnesota Water Quality Standards. Round Lake is classified in the regulations as Class 2B and 3B (Minn. R. ch 7050.0470, Subpart 1.B.). Class 2B waters are classified as to permit the propagation and maintenance of a healthy community of cool or warm water sport or commercial fish and associated aquatic life, and their habitats (Minn. R. ch 7050.0222, Subpart 4). Class 2B waters are also classified for aquatic recreation of all kinds, including bathing, for which the waters may be usable (Minn. R. ch 7050.0222, Subpart 4). This class of surface water is not protected as a source of drinking water. Class 3B waters are classified as to permit their use for general industrial purposes, except for food processing, with only a moderate degree of treatment (Minn. R. ch 7050.0223, Subpart 3). Based on monitoring at Round Lake, the standards associated with these surface water classifications are currently being met and no surface water COCs have been identified; thus, the Minnesota Water Quality Standards would not provide ARAR for any remedial action addressing the sediment.

As discussed above there are no state or federal promulgated standards for sediment. The NCP recognizes that in absence of ARAR for a media, non-promulgated advisories, criteria, or guidance may be useful in determining what is protective in developing CERCLA remedies. The NCP classifies such advisories, criteria, and guidance as to-be-considered guidance or TBC. The identification and use of TBCs are not mandatory under CERCLA and are only to be used on an "as appropriate" basis (*NCP Final Rule preamble*, 55 FR 8744, March 8, 1990). TBCs can be and often are preliminary remediation goals (PRGs) based on readily available information; however, such goals are modified throughout the RI/FS process. Final remedial goals are determined when the remedy is selected considering factors in the NCP, including environmental evaluations (40 CFR 300.430(e)(2)(i); *NCP Final Rule preamble*, 55 FR 8712-8713, March 8, 1990).

For sediment, TBCs originate from several sources. The following could be sources of TBCs for Round Lake:

- *Guidance For The Use And Application Of Sediment Quality Targets For The Protection Of Sediment-Dwelling Organisms In Minnesota*, MPCA Document Number: tdr-gl-04, February 2007.
- *A Preliminary Evaluation of Sediment Quality Assessment Values for Freshwater Ecosystems*, Smith et. al. 1996
- *Approach to the Assessment of Sediment Quality in Florida Coastal Waters*, Florida Department of Environmental Protection, Office of Water Policy, MacDonald 1994USEPA 2000
- *Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments*, Long et.al. 1995

- *Guidance on Remedial Actions for Superfund Sites with PCB Contamination*, EPA OSWER Directive 9355.4-01, EPA/540/G-90/007, August 1990

TBC values for COCs at Round Lake are included in the below table and discussed below.

Final COC (Tier II ERA)	TBC Guidance Value (in mg/kg)							
	Threshold Effect Level				Low Effect Level			
	SQT I[a]	TEL-FW[b]	TEL-ME[c]	ER-L[d]	SQT II[a]	PEL-FW[b]	PEL-ME[c]	ER-M[d]
Cadmium	0.99	1.0 (PQL)	1.0 (PQL)	1.2	5.0	3.53	4.21	9.6
Chromium	43	37.3	52.3	81	110	90	160	370
Copper	32	35.7	18.7	34	150	197	108	270
Lead	36	35	30.2	47	130	91.3	112	218
Zinc	120	123	124	150	460	315	271	410
Total PCBs	0.06	0.033	0.022	0.023	0.68	0.277	0.189	0.18

[a]MPCA Sediment Quality Targets, February 2007. SQT I values represent contaminant concentrations below which harmful effects on benthic invertebrates are unlikely. SQT II values represent contaminant concentrations above which harmful effects on benthic organisms are likely.
[b]Smith et al. 1996. TEL-FW are freshwater threshold effect levels; PEL-FW are freshwater probable effect levels.
[c]MacDonald 1994. TEL-ME are marine/estuarine threshold effect levels; PEL-ME are marine/estuarine probable effect levels.
[d]Long et al. 1995. ER-L are low effects range values; ER-M are median effects range values.

Guidance For The Use And Application Of Sediment Quality Targets For The Protection Of Sediment-Dwelling Organisms In Minnesota, Minnesota Pollution Control Agency, February 2007

This guidance document was issued by the Minnesota Pollution Control Agency (MPCA) to provide recommended sediment quality targets (SQTs) to be used throughout the state. The SQTs are based on values developed for the St. Louis River in northeastern Minnesota. The underlying paper the SQTs are based on developed a process to synthesize existing numerical sediment quality guidelines (SQGs) using a geometric mean of the SQGs that met specific selection criteria (MacDonald and Ingersoll et. al., 2000). The SQGs were developed for a threshold effect concentration (TEC), and a probable effect concentration (PEC). The TEC is intended to identify sediment concentrations below which adverse effects on sediment dwelling organisms are not expected to occur. The PEC is intended to define sediment concentrations above which adverse effects on sediment dwelling organisms are likely to be observed. The contaminant concentrations between the TEC and the PEC were not predicted to be neither toxic nor nontoxic and the SQGs are not intended to provide guidance in this range of concentrations. This paper points out that the application of the SQGs is strengthened when SQGs are used in combination with other sediment quality assessment tools (i.e., toxicity tests, bioaccumulation assessments, and benthic invertebrate community assessments). The SQGs developed by this paper do not consider the potential for bioaccumulation in aquatic organisms. This underlying paper also adopted an approach to evaluate mixture of contaminants using a mean PEC quotient. The mean PEC-Q is a unitless index that provides an assessment tool. This value is determined by dividing the concentration of each substance by its PEC, and then adding these PEC quotients together. The summation PEC-Qs for each individual contaminant in each class (i.e., metals, PAHs, and PCBs) is divided by the number of contaminants in the class. The mean PEC-Q for class are then added together and then divided by the number of classes in the mixture to obtain a mean PEQ for the mixture. The predictive ability of the mean PEC-Q to predict sediment toxicity was 85% for a mixture with seven trace metals, PAHs, PCBs, and DDE where the mean PEC quotient was greater than 0.5 and was identified in the paper as a useful threshold between toxic and non-toxic.

The MPCA adopted the use of the TEC, PEC, and mean PEC quotient in their guidance document. The TEC was used as the MPCA Level I SQT defined to identify contaminant concentrations below which harmful effects on sediment dwelling organism are unlikely. The PEC was used as the MPCA Level II SQT defined to identify contaminant concentrations above which harmful effects on sediment dwelling organism are likely to be observed. The MPCA guidance document also adopted the use of the mean PEC-Q for mixtures. MPCA selected a mean PEC-Q of ≤ 0.1, which has a low probability of observing sediment toxicity (< 10%), as goal to provide a high level of protection in sediment dwelling organisms. Alternatively MPCA uses a mean PEC-Q of 0.6 if the goal for the site is to reduce the potential for acute toxicity and permit natural recovery processes to further reduce concentrations.

MPCA guidance specifically notes several considerations in applying the SQTs:

- Applicability of the SQTs in sediment assessments is increased when used in conjunction with other sediment assessment tools such as sediment chemistry, sediment toxicity testing, bioaccumulation studies, and effects on *in situ* benthic invertebrates.
- Variations in physical, chemical, biological factors in the sediment environment, such as highly modified depositional systems will result in higher uncertainty in applying the SQTs;
- Where additional assessment phases are conducted (i.e., sediment toxicity tests, benthological surveys, and bioaccumulation assessments) SQTs are used in conjunction with these other tools to make decisions about the spatial and temporal extent of contamination and the need for remediation;
- Numerical SQTs should not be regarded as blanket values of regional sediment quality; rather variations in environmental conditions among sites may necessitate the need for modifications of the SQTs to reflect local conditions;
- Substances that occur at concentration above the Level I SQT but below the Level II SQT should be considered moderate concern; and,
- Chemicals not positively correlated to the results of toxicity tests should be considered a relatively lower priority.

The SQTs provided by this guidance would be a potential source for TBCs for Round Lake as discussed above on an as appropriate basis. The SQT I would not be appropriate for use as TBC based on the following:

- As discussed in the supporting paper, the range of concentration between the SQT I (TEC) and the SQT II (PEC) is not intended to indicate if the concentration are toxic or nontoxic. Since concentrations below the SQT I are unlikely to result in harmful effects on sediment dwelling organism and values up to the SQT II are not indicative of toxicity, the SQT II would be the appropriate value to use to predict harmful effects on ecological receptors.

- The SQT I level may be appropriate if protection of the individual ecological receptor is the goal; however, a principal in ecological risk assessment is that Superfund remedies should generally be designed to protect local populations and communities of biota and not to protect organisms on an individual basis (except in the instance of the presence of T&E species) (Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999).

The SQT II identifies the levels above which harmful effects are likely to be observed and would be considered appropriate for use as TBC.

A Preliminary Evaluation of Sediment Quality Assessment Values for Freshwater Ecosystems, Smith et. al. 1996

This paper describes the process for deriving freshwater sediment quality assessment values for recommended sediment quality guidelines for freshwater in Canada. Two assessment endpoints are derived and reported in this paper. A threshold effect level (TEL), a concentration below which toxicity is rarely observed, and a probable effect level (PEL), a concentration above which toxicity is observed. The TEL and PEL delineate three ranges: (1) a chemical concentration below or equal to the TEL which is rarely associated with adverse biological effects; (2) chemical concentration between the TEL and PEL which is occasionally associated with adverse biological effects; and (3) a chemical concentration equal to or above the PEL which is frequently associated with adverse biological effects. It is noted that sediments that are higher than the recommended sediment quality guidelines only indicate that there is the potential for biological effects to occur. These guidelines are intended to be used in coordination with other site specific information (i.e., background or further biological assessment) in making sediment management decisions. The TELs/PELs provided by this paper would be a potential source for TBCs for Round Lake as discussed above on an as appropriate basis. It should be noted that exceedance of criteria or standards does not necessarily imply causation, because the regulatory values are intended to be safe levels and not a number which signifies absolute occurrence of adverse effect (Suter et al. 2002). The TEL represents a level below which toxicity is rarely observed. This level may be appropriate if protection of the individual ecological receptor is the goal; however, a principal in ecological risk assessment is that Superfund remedies should generally be designed to protect local populations and communities of biota and not to protect organisms on an individual basis (except in the instance of the presence of T&E species) ((Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999). Due to this principal in ecological risk assessment, the PEL would be more appropriate for use as TBC at this site.

Approach to the Assessment of Sediment Quality in Florida Coastal Waters, Florida Department of Environmental Protection, Office of Water Policy, MacDonald 1994

This report was prepared to provide Florida Department of Environmental Protection numerical sediment quality assessment guidelines (SQAGs) for Florida coastal waters. The approach used followed that by Long and Morgan in 1990. Using existing databases, numerical sediment quality assessment guidelines were developed. Two guidelines are derived and reported in this guidance document. A threshold effect level (TEL), a concentration below which sediment-associated contaminants are not considered to represent significant hazards to aquatic organisms (minimal effects range), and a probable effect level (PEL), a concentration above which adverse biological effects are usually or always associated (probable effects range). In the range of concentrations between the TEL and the PEL adverse biological effects are possible (possible effects range); however, it is difficult to predict the occurrence, nature, or severity of the effects. When concentrations are within this probable effects range further investigation is recommended to determine if significant hazards to aquatic organisms exist. The SQAGs are intended to provide effective screening tools for sediment quality to identify priorities for further action. It is noted in the report that the SQAGs should not be used alone as sediment quality criteria. The reports cautions bioavailability should be considered along with the SQAGs to prevent the potential for either under- or over-protection of aquatic resources. The TELs/PELs provided by this report would be a potential source for TBCs for Round Lake as discussed above on an as appropriate basis. It should be noted that exceedance of criteria or standards does not necessarily imply causation, because the regulatory values are intended to be safe levels and not a number which signifies absolute occurrence of adverse effect (Suter et al. 2002). The TEL represents a level below which sediment-associated contaminants are not considered to represent significant hazards to aquatic organisms. This level may be appropriate if protection of the individual ecological receptor is the goal; however, a principal in ecological risk assessment is that Superfund remedies should generally be designed to protect local populations and communities of biota and not to protect organisms on an individual basis (except in the instance of the presence of T&E species) ((Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999). Due to this principal in ecological risk assessment, the PEL would be more appropriate for use as TBC at this site.

Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments, Long et.al. 1995

This paper describes the study to update and expand the guideline values from the Long and Morgan study in 1990; quantify the percent incidence of adverse biological effects; and compare the guidelines with other data or methods. This study identified values for the effects range-low (ERL) and effects range-medium (ERM). These values delineated three concentration ranges: concentrations below the ERL value (minimal-effects range) where effects are rarely observed; concentrations equal and above the ERL but below ERM (possible effects range) where effects would occasionally occur; and concentrations equal to and above the ERM (probable-effects range) where effects would frequently occur. The incidence of biological effects was quantified for each of these ranges. The incidences of biological effects for each of the ranges were as follows: 18% for concentrations below the ERL (minimal effect range); 46% for concentrations equal and above the ERL but below ERM (possible effects range); and, 90% for concentrations equal to and above the ERM (probable-effects range). The paper indicates that the guidelines provided should be used as informal screening tools and are not intended to preclude the use of toxicity tests or other measure of biological effects. The ERLs/ERMs provided by this paper would be a potential source for TBCs for Round Lake as discussed above on an as appropriate basis. It should be noted that exceedance of criteria or standards does not necessarily imply causation, because the regulatory values are intended to be safe levels and not a number which signifies absolute occurrence of adverse effect (Suter et al. 2002). At concentrations below the ERL value effects are rarely observed. This level may be appropriate if protection of the individual ecological receptor is the goal; however, a principal in ecological risk assessment is that Superfund remedies should generally be designed to protect local populations and communities of biota and not to protect organisms on an individual basis (except in the instance of the presence of T&E species) ((Ecological Risk Assessment and Risk Management Principles for Superfund Sites, page 3, OSWER Directive 9285.7-28P, October 7, 1999). Due to this principal in ecological risk assessment, the ERM would be more appropriate for use as TBC at this site.

Guidance on Remedial Actions for Superfund Sites with PCB Contamination, USEPA, EPA/540/G-90/007, August 1990

This guidance document provides EPA's general framework for the management of PCBs in Superfund remedial actions including determining cleanup levels. Sediment quality criteria (SQC) were developed for PCBs which could be considered in establishing remedial goals. SQCs were developed using the equilibrium partitioning approach using the federal ambient chronic aquatic life water quality criteria at two organic carbon concentrations. The SQC values for 10% and 1% organic carbon concentrations are 1.9 µg/g and 0.19 µg/g respectively. These SQCs would be a potential source for TBCs for Round Lake as discussed above on an as appropriate basis.

A.5 Use of Monitored Natural Recovery

EPA guidance has stated that due to the limited number of alternative options for sediment, all three of the major remedial approaches (MNR, in-situ capping, and removal) should be evaluated at every sediment remedial site (*Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*, page 7-3, OSWER 9355.0-85, EPA-540-R-05-012, December 2005). MNR relies on a wide range of naturally occurring processes including physical, biological, and chemical mechanisms to reduce the risk to human health and/or ecological receptors (*Id.* at 4-2). Most sites rely on natural sedimentation to isolate contaminated sediment (*Id.* at 4-1). Based on the physical and chemical characteristics of Round Lake, two processes, lake succession (aging) and eutrophication, are determining the environmental conditions of the lake. Lake succession (aging) is the natural process by which a lake fills in over geologic time with allochthonous erosional materials deposited from outside of the lake. Eutrophication is the process of increased nutrient input (productivity) that can be accelerated by human activities, including stormwater runoff. Round Lake has no natural sources of water inflow to the lake; surface water is recharged by precipitation and stormwater runoff through culverts and sewers. There is also no natural outlet; the only outlet is a concrete structure with stoplogs to allow water level control. There is also limited connectedness to groundwater, an unconfined perched aquifer with glacial till below serves as an aquitard. Consequently, Round Lake is a depositional environment with sediment loading from stormwater runoff events.

- To demonstrate the use of MNR multiple lines of evidence are usually used; however, not all lines of evidence or a particular type of information may be appropriate for every site (*Id.* at 4-1). Lines of evidence may include:
- Long-term decreasing trend of contaminant levels in higher trophic level biota (e.g., piscivorous fish)
- Long-term decreasing trend of water column contaminant concentrations averaged over a typical low-flow period of high biological activity (e.g., trend of summer low flow concentrations)
- Sediment core data demonstrating a decreasing trend in historical surface contaminant concentrations through time
- Long-term decreasing trends of surface sediment contaminant concentration, sediment toxicity, or contaminant mass within the sediment (*Id.* at 4-9).

EPA Guidance indicates MNR is should be considered when it will meet remedial objectives in a time frame that is reasonable as compared to active remedies. However, it is recognized MNR may take longer to reach all remedial objectives. The recovery period for all alternatives (MNR or recovery of ecological resources after dredging or capping) should also be considered (*Id.* at 4-12). Factors used to determine reasonableness include:

- The extent and likelihood of human exposure to contaminants during the recovery period, and if controlled by institutional controls, the effectiveness of those controls;
- The value of ecological resources that may continue to be impacted during the recovery period;
- The time frame in which affected portions of the site may be needed for future uses which will be available after MNR has achieved cleanup levels; and
- The uncertainty associated with the time frame prediction (*Id.* at pages 4-12 to 4-13).

One additional consideration in any MNR remedy is the control of any significant sources of contaminants. EPA guidance notes that “MNR is likely to be effective most quickly in depositional environments after source control actions and active remediation of any high risk sediment have been completed” (*Id.* at 4-11). Although previously there have been releases of contaminants from production operations at TCAAP to Round Lake via the storm sewer, the sources of the releases has been corrected and the cessation of production operations at TCAPP further eliminates the potential for any future releases.

EPA list sites conditions that are specifically conducive to Monitored Natural Recovery, many of which are present at Round Lake:

- Anticipated land uses or new structures are not incompatible with natural recovery
- Natural recovery processes have a reasonable degree of certainty to continue at rates that will contain, destroy, or reduce the bioavailability or toxicity of contaminants within an acceptable time frame
- Expected human exposure is low and/or can be reasonably controlled by institutional controls
- Sediment bed is reasonably stable and likely to remain so
- Sediment is resistant to resuspension (e.g., cohesive or well-armored sediment)
- Contaminant concentrations in biota and in the biologically active zone of sediment are moving towards risk-based goals on their own
- Contaminants already readily biodegrade or transform to lower toxicity forms
- Contaminant concentrations are low and cover diffuse areas
- Contaminants have low ability to bioaccumulate (*Id.* at 4-3).

Land use at Round Lake is anticipated to remain as an USFWS wildlife refuge. No change in land use is anticipated which would be incompatible with MNR. Round Lake is overall a shallow depositional lake where the sediment is stable and resuspension is unlikely. Annual precipitation averages 29 inches/year resulting in an estimated average annual runoff to the lake of 200 to 300 acre-feet/year (excludes precipitation) [2012 FS]. Very high sedimentation rates of >1.5 cm/yr for the 20th century have been reported for the lake (Engstrom, 2012). The lake occupies approximately 125 acres with a maximum depth of 26' at the south-central end. However, less than 5% of the lake basin is more than 20' in depth; the majority is a relatively flat shoal averaging approximately 4.5' in depth (USFWS, 1992). Typically, shallow lakes <20 in depth do not exhibit mixing and turnover. USFWS does not currently allow fishing at Round Lake; and, a fish consumption advisory could be implemented by the USFWS for any future fishing activity if needed.

A.6 Use of Enhanced Monitored Natural Recovery

In some situations MNR may be the most appropriate remedy; however, MNR may be unable to reduce risks sufficiently within an acceptable time frame. In these situations MNR can be accelerated or enhanced by applying a thin clean layer of material, usually as little as few inches. In most cases natural material is recommended approximating common substrates found in the area. Enhanced MNR is distinguished from capping in that the purpose of the clean layer is to mix with the contaminated sediment. Enhanced MNR is not designed to isolate the contaminants as in capping (where cap thickness can range up to several feet). Enhancement of degradation can also be facilitated by using additives to speed up the natural recovery (*Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*, page 4-11, OSWER 9355.0-85, EPA-540-R-05-012, December 2005).

A.7 ITRC - Incorporating Bioavailability Considerations into the Evaluation of Contaminated Sediment Sites

The Interstate Technology Regulatory Council (ITRC) Contaminated Sediments Team issued a guidance document (Incorporating Bioavailability Considerations into the Evaluation of Contaminated Sediment Sites, February 2011) to compile concepts, tools, and measures for assessing bioavailability. The guidance notes the importance of bioavailability in risk assessments and in establishing technically defensible cleanup goals due to the low predictive value and conservative nature of SQGs when considered alone. The guidance notes the relationship between sediment contaminant concentrations and risk from exposure is not linear due to bioavailability considerations which may in some instances only result in a fraction of the contaminant being available to cause harm to ecological receptors. The ITRC Contaminated Sediment Team in developing the guidance collected data on sites where bioavailability data was used in the remedial decision making process. Several of these sites addressed COCs and bioavailability factors that have also been considered at Round Lake (ITRC, 2011):

- Hackensack River, Study Area 7, Jersey City, New Jersey. Located on the eastern shore of the Hackensack River, the site addressed chromium concentrations in sediment and used multiple lines of evidence including SEM/AVS levels and benthic community survey results to demonstrate a very low bioavailability of chromium. This supported a sediment remedy of capping and MNR.
- Former General Motors North Tarrytown Assembly Plant, New York – Sediment concentrations (chromium, copper, lead, and zinc), exceeding New York screening levels were further evaluated using bioavailability including SEM/AVS, toxicity testing, and benthic community surveys. As of this reporting, a remedy was not yet selected pending results of testing.
- Tectronix Beaverton Creek, Washington County, Oregon – Beaverton Creek is a channelized fourth-order stream that flows from a residential area onto the site and then through a commercial area and into a nature park ½ mile downstream. Investigation revealed elevated levels of metals in the sediment (cadmium, copper, chromium, lead, mercury, nickel, silver, and zinc). AVS and SEM were evaluated to predict the potential toxicity of the sediment. Toxicity tests (10-day mortality sediment toxicity test with the amphipod *Hyalella azteca* and the 10-day growth and mortality sediment toxicity test with the midge *Chironomus dilutes*) were used to access bioavailability based on 13 surface sediment samples collected at a subset of 11 locations within the site and at the two upstream locations. Although sediment concentrations exceeded screening criteria, the evaluation of the bioavailability factors showed a low potential for toxicity. Sediment remediation was deemed unnecessary.
- Tri-State Mining District, Kansas – Pore water chemistry and toxicity testing is anticipated to be evaluated in establishing the cleanup level for cadmium, zinc, and lead based on sediment concentrations that correspond to a 10%–20% nonsurvival rate for benthic organisms (clams and mussels).
- Vandenberg Air Force Base (AFB), Site 5 Cluster (Bear Creek Pond) - Bear Creek Pond, generally undisturbed, is located downgradient of and received surface water from formerly active launch pads. Several metals were predicted initially to cause potential risk to benthic invertebrates.

Eight sediment samples were collected for toxicity testing (ten-day survival and growth bioassays using the amphipod *Hyallela azteca*). With comparison to reference locations, any adverse effect was negligible. In addition, AVS analysis indicated limited bioavailability. It was concluded that negligible risk existed at the site.

APPENDIX B

ROUND LAKE SEDIMENT MONITORING DATA FROM 2011 SAMPLING EVENT AND

STATISTICAL ANALYSES OF SEDIMENT MONITORING DATA

Round Lake sediment monitoring data from 2011 sampling event (Wenck 2012).

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC001	2/1/11	0.5-1	0.28	11.6	21.8	5.8	84.2	0.009
RLSC002	2/1/11	0.5-1	0.59	43.2	289.0	17.8	190.0	0.016
RLSC003	2/1/11	0.5-1	1.30	97.0	1040.0	38.4	505.0	0.033
RLSC004	2/1/11	0.5-1	6.60	101.0	732.0	100.0	619.0	0.258
RLSC005	2/1/11	0.5-1	2.60	816.0	342.0	53.8	404.0	0.071
RLSC006	2/1/11	0.5-1	0.54	30.3	158.0	19.8	141.0	0.034
RLSC007	2/1/11	0.5-1	0.32	15.6	27.8	11.7	59.2	0.019
RLSC008	2/1/11	0.5-1	1.60	40.6	273.0	42.8	272.0	1.184
RLSC009	2/1/11	0.5-1	1.00	106.0	754.0	40.3	615.0	0.030
RLSC010	2/1/11	0.5-1	7.00	116.0	853.0	105.0	752.0	0.036
RLSC011	2/1/11	0.5-1	9.30	57.6	383.0	107.0	472.0	0.061
RLSC012	2/2/11	0.5-1	0.48	16.7	29.8	14.4	72.1	0.019
RLSC013	2/2/11	0.5-1	0.15	14.5	16.1	7.1	43.6	0.017
RLSC014	2/3/11	0.5-1	0.49	19.7	22.2	12.6	62.1	0.024
RLSC015	2/3/11	0.5-1	0.66	17.2	32.2	12.3	72.4	0.053
RLSC016	2/3/11	0.5-1	0.11	37.6	516.0	12.7	173.0	0.059
RLSC017	2/3/11	0.5-1	0.05	5.3	4.0	2.6	12.2	0.005
RLSC018	2/3/11	0.5-1	28.50	87.5	468.0	164.0	755.0	9.035
RLSC019	2/4/11	0.5-1	0.50	20.2	41.3	45.8	85.2	0.017
RLSC020	2/4/11	0.5-1	0.34	20.9	23.4	8.8	65.6	0.022
RLSC021	2/4/11	0.5-1	0.44	17.1	26.2	7.7	55.4	0.029
RLSC022	2/7/11	0.5-1	0.32	8.7	15.6	3.8	36.7	0.035
RLSC023	2/7/11	0.5-1	0.31	21.4	160.0	12.8	211.0	0.018
RLSC024	2/8/11	0.5-1	0.65	14.5	19.2	22.9	64.3	0.022
RLSC025	2/8/11	0.5-1	1.30	32.7	105.0	32.1	181.0	0.024
RLSC026	2/8/11	0.5-1	0.82	21.3	45.2	25.3	99.1	0.022
RLSC027	2/8/11	0.5-1	0.16	13.4	82.5	9.0	79.6	0.009
RLSC028	2/8/11	0.5-1	0.44	15.4	24.7	8.6	51.3	0.026
RLSC029	2/8/11	0.5-1	0.41	14.0	22.4	7.0	43.4	0.031
RLSC030	2/9/11	0.5-1	0.62	26.3	64.9	14.4	118.0	0.032
RLSC031	2/9/11	0.5-1	3.10	30.0	96.8	47.1	195.0	0.108
RLSC032	2/9/11	0.5-1	1.00	59.3	432.0	25.4	316.0	0.032
RLSC033	2/9/11	0.5-1	0.24	9.5	6.6	4.3	20.7	0.009
RLSC034	2/9/11	0.5-1	0.47	15.0	21.2	7.2	65.0	0.023
RLSC035	2/9/11	0.5-1	0.43	12.7	22.4	5.9	57.1	0.036
RLSC036	2/9/11	0.5-1	0.65	17.9	28.9	6.9	69.4	0.035
RLSC037	2/10/11	0.5-1	0.65	13.8	24.1	4.9	72.2	0.045
RLSC038	2/10/11	0.5-1	1.90	39.6	296.0	30.3	266.0	0.036

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC039	2/10/11	0.5-1	0.57	15.5	20.7	6.6	65.1	0.020
RLSC040	2/10/11	0.5-1	0.23	9.4	7.1	3.3	21.0	0.008
RLSC041	2/10/11	0.5-1	0.65	15.5	23.8	6.3	57.7	0.034
RLSC042	2/10/11	0.5-1	1.10	40.9	155.0	20.1	217.0	0.028
RLSC043	2/10/11	0.5-1	0.60	17.9	22.6	5.1	57.3	0.038
RLSC044	2/10/11	0.5-1	0.35	8.2	19.2	10.8	45.3	0.023
RLSC045	2/10/11	0.5-1	0.03	6.5	3.4	3.3	18.8	0.007
RLSC046	2/11/11	0.5-1	0.14	13.9	18.2	6.6	68.8	0.030
RLSC047	2/11/11	0.5-1	0.13	13.5	18.8	5.7	63.8	0.027
RLSC048	2/11/11	0.5-1	0.16	19.3	20.1	8.4	80.4	0.036
RLSC049	2/11/11	0.5-1	0.13	16.4	17.7	6.6	69.3	0.028
RLSC050	2/11/11	0.5-1	0.07	6.1	7.4	4.4	22.7	0.007
RLSC051	2/11/11	0.5-1	1.70	27.3	114.0	33.5	140.0	0.069
RLSC052	2/11/11	0.5-1	0.15	10.6	13.6	6.6	41.4	0.028
RLSC053	2/11/11	0.5-1	0.77	17.9	22.9	17.6	63.5	0.015
RLSC054	2/14/11	0.5-1	0.12	18.7	17.8	8.3	66.9	0.026
RLSC055	2/14/11	0.5-1	0.02	2.9	0.7	1.6	7.1	0.005
RLSC056	2/14/11	0.5-1	0.15	17.8	20.6	9.2	77.5	0.033
RLSC057	2/14/11	0.5-1	0.16	17.3	23.9	8.2	77.5	0.035
RLSC058	2/14/11	0.5-1	0.03	5.4	4.2	2.6	13.2	0.006
RLSC059	2/14/11	0.5-1	0.09	14.9	16.8	7.2	36.3	0.019
RLSC060	2/14/11	0.5-1	0.14	11.4	15.7	5.5	42.1	0.029
RLSC061	2/14/11	0.5-1	0.16	17.8	21.3	8.4	50.3	0.036
RLSC062	2/14/11	0.5-1	0.11	15.8	24.9	8.0	62.6	0.025
RLSC063	2/15/11	0.5-1	1.20	21.5	29.1	36.1	101.0	0.038
RLSC064	2/15/11	0.5-1	0.39	13.6	19.8	6.4	84.0	0.032
RLSC065	2/15/11	0.5-1	0.42	15.2	19.4	6.4	64.1	0.038
RLSC066	2/15/11	0.5-1	0.81	26.4	51.0	15.2	110.0	0.030
RLSC067	2/15/11	0.5-1	0.30	17.4	19.6	7.5	65.0	0.025
RLSC068	2/15/11	0.5-1	0.39	18.9	23.8	8.4	63.7	0.027
RLSC069	2/15/11	0.5-1	0.04	6.1	5.0	3.3	16.5	0.005
RLSC070	2/15/11	0.5-1	0.46	18.9	36.5	12.5	65.2	0.020
RLSC071	2/16/11	0.5-1	0.35	15.7	21.4	6.8	52.7	0.029
RLSC072	2/16/11	0.5-1	0.50	17.7	24.7	8.3	54.5	0.033
RLSC073	2/16/11	0.5-1	1.30	22.7	41.4	35.5	116.0	0.040
RLSC074	2/16/11	0.5-1	0.88	81.1	528.0	31.0	481.0	0.023
RLSC075	2/16/11	0.5-1	0.39	15.1	27.1	7.0	78.4	0.061
RLSC076	2/16/11	0.5-1	0.44	13.2	16.9	5.2	48.4	0.035
RLSC077	2/16/11	0.5-1	0.28	9.5	7.9	4.6	41.5	0.005
RLSC078	2/16/11	0.5-1	0.47	21.8	23.2	8.1	76.4	0.029
RLSC079	2/16/11	0.5-1	0.02	3.2	0.9	1.3	8.0	0.005

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC080	2/16/11	0.5-1	0.39	20.3	37.6	13.5	75.3	0.017
RLSC081	2/17/11	0.5-1	0.55	13.7	45.1	9.4	69.9	0.038
RLSC082	2/17/11	0.5-1	0.63	13.7	21.1	5.5	68.4	0.056
RLSC083	2/17/11	0.5-1	0.70	23.1	28.8	11.4	79.9	0.051
RLSC084	2/17/11	0.5-1	1.90	20.2	43.8	43.5	136.0	0.044
RLSC085	2/17/11	0.5-1	1.10	197.0	1500.0	53.3	1150.0	0.053
RLSC086	2/17/11	0.5-1	1.10	24.7	38.2	34.7	110.0	0.043
RLSC087	2/17/11	0.5-1	0.34	14.0	17.7	6.3	40.5	0.016
RLSC088	2/17/11	0.5-1	0.05	6.8	6.0	4.6	18.6	0.005
RLSC089	2/17/11	0.5-1	0.58	19.1	24.9	9.9	82.6	0.028
RLSC090	2/18/11	0.5-1	0.40	19.0	17.2	8.7	57.7	0.015
RLSC091	2/18/11	0.5-1	0.20	8.6	9.3	8.6	29.0	0.007
RLSC092	2/18/11	0.5-1	0.39	11.6	15.5	5.7	46.1	0.028
RLSC093	2/18/11	0.5-1	0.41	17.5	22.4	9.3	58.9	0.022
RLSC094	2/18/11	0.5-1	0.46	13.5	20.5	5.7	45.6	0.043
RLSC095	2/18/11	0.5-1	0.42	12.2	19.7	5.6	54.6	0.049
RLSC096	2/18/11	0.5-1	0.56	19.9	25.7	7.9	83.8	0.043
RLSC097	2/19/11	0.5-1	1.10	131.0	1420.0	54.9	1090.0	0.024
RLSC098	2/21/11	0.5-1	0.50	103.0	738.0	32.8	660.0	0.025
RLSC099	2/21/11	0.5-1	0.23	14.6	13.2	7.0	53.7	0.032
RLSC100	2/21/11	0.5-1	0.18	18.5	15.9	8.5	71.7	0.038
RLSC101	2/21/11	0.5-1	0.18	18.1	13.3	8.2	51.8	0.020
RLSC102	2/21/11	0.5-1	0.07	13.5	8.3	5.7	30.8	0.010
RLSC103	2/21/11	0.5-1	0.11	10.3	12.7	6.0	33.4	0.008
RLSC104	2/21/11	0.5-1	0.24	16.4	16.6	6.4	54.4	0.029
RLSC105	2/21/11	0.5-1	0.28	26.0	18.4	8.8	53.3	0.045
RLSC106	2/22/11	0.5-1	0.40	22.2	23.4	9.7	85.5	0.037
RLSC107	2/22/11	0.5-1	0.56	15.1	20.2	6.8	73.6	0.044
RLSC108	2/22/11	0.5-1	0.47	13.7	21.4	6.8	62.0	0.050
RLSC109	2/22/11	0.5-1	0.36	12.3	19.2	6.1	54.1	0.047
RLSC110	2/22/11	0.5-1	0.48	19.9	24.2	9.8	99.9	0.050
RLSC111	2/22/11	0.5-1	0.46	18.0	19.4	7.9	52.8	0.033
RLSC112	2/23/11	0.5-1	0.33	19.4	19.4	9.5	57.0	0.017
RLSC114	2/23/11	0.5-1	1.30	85.7	500.0	47.1	519.0	0.035
RLSC115	2/23/11	0.5-1	0.42	13.1	22.7	6.9	41.7	0.038
RLSC116	2/23/11	0.5-1	0.57	20.4	65.8	18.8	108.0	0.035
RLSC117	2/23/11	0.5-1	0.30	22.5	25.3	9.7	69.7	0.025
RLSC118	2/23/11	0.5-1	0.48	32.6	121.0	20.3	159.0	0.025
RLSC119	2/23/11	0.5-1	0.33	25.9	21.6	8.0	64.4	0.027
RLSC120	2/23/11	0.5-1	0.24	13.1	14.6	6.3	48.1	0.020
RLSC121	2/23/11	0.5-1	0.49	26.8	62.6	21.1	115.0	0.023

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC122	2/23/11	0.5-1	0.49	12.5	14.3	19.2	54.3	0.012
RLSC123	2/23/11	0.5-1	0.15	19.1	22.1	8.2	62.8	0.033
RLSC124	2/24/11	0.5-1	0.42	20.7	22.6	9.8	51.9	0.024
RLSC125	2/24/11	0.5-1	0.24	16.6	18.2	8.2	59.5	0.019
RLSC126	2/24/11	0.5-1	0.54	24.7	50.6	21.1	96.6	0.023
RLSC127	2/24/11	0.5-1	0.22	20.1	23.8	9.1	54.4	0.014
RLSC128	2/24/11	0.5-1	0.39	22.3	100.0	23.1	137.0	0.068
RLSC129	2/24/11	0.5-1	0.97	95.0	553.0	44.9	565.0	0.022
RLSC130	2/24/11	0.5-1	0.90	18.0	40.0	30.1	98.1	0.021
RLSC131	2/24/11	0.5-1	0.29	19.4	14.7	13.5	49.0	0.012
RLSC132	2/24/11	0.5-1	0.27	17.4	14.9	7.5	47.9	0.011
RLSC133	2/24/11	0.5-1	0.88	20.6	55.0	25.3	100.0	0.024
RLSC134	2/24/11	0.5-1	0.71	28.5	109.0	30.4	169.0	0.037
RLSC135	2/24/11	0.5-1	0.04	7.2	7.7	5.9	23.2	0.015
RLSC001	2/1/11	0-0.5	2.30	68.7	902.0	45.5	609.0	0.027
RLSC002	2/1/11	0-0.5	3.50	66.4	473.0	60.0	360.0	0.036
RLSC003	2/1/11	0-0.5	10.50	103.0	582.0	133.0	595.0	0.066
RLSC004	2/1/11	0-0.5	6.20	70.8	313.0	101.0	440.0	0.295
RLSC005	2/1/11	0-0.5	3.90	46.3	179.0	74.7	305.0	0.157
RLSC006	2/1/11	0-0.5	3.10	46.3	378.0	54.4	318.0	0.159
RLSC007	2/1/11	0-0.5	6.20	68.6	530.0	67.1	454.0	0.286
RLSC008	2/1/11	0-0.5	26.60	94.8	359.0	152.0	614.0	0.577
RLSC009	2/1/11	0-0.5	7.20	78.8	621.0	101.0	582.0	0.248
RLSC010	2/1/11	0-0.5	12.70	295.0	408.0	195.0	664.0	0.375
RLSC011	2/1/11	0-0.5	5.50	63.1	326.0	131.0	593.0	0.283
RLSC012	2/2/11	0-0.5	9.80	90.1	685.0	75.9	577.0	0.159
RLSC013	2/2/11	0-0.5	0.68	24.0	76.1	23.7	125.0	0.031
RLSC014	2/3/11	0-0.5	4.30	67.1	338.0	79.9	438.0	0.044
RLSC015	2/3/11	0-0.5	4.70	56.0	250.0	56.5	389.0	0.041
RLSC016	2/3/11	0-0.5	1.60	34.2	136.0	258.0	243.0	0.504
RLSC017	2/3/11	0-0.5	0.56	10.6	41.8	21.5	71.4	0.094
RLSC018	2/3/11	0-0.5	8.60	94.3	267.0	175.0	841.0	0.351
RLSC019	2/4/11	0-0.5	2.20	75.4	741.0	56.9	551.0	0.676
RLSC020	2/4/11	0-0.5	0.97	38.4	126.0	23.1	192.0	0.034
RLSC021	2/4/11	0-0.5	0.64	29.0	143.0	22.2	211.0	0.041
RLSC022	2/7/11	0-0.5	0.39	21.4	46.6	9.6	83.1	0.026
RLSC023	2/7/11	0-0.5	6.30	60.4	191.0	75.4	329.0	0.217
RLSC024	2/8/11	0-0.5	3.70	95.1	717.0	49.2	576.0	0.081
RLSC025	2/8/11	0-0.5	1.80	39.6	192.0	32.1	266.0	0.141
RLSC026	2/8/11	0-0.5	8.60	112.0	317.0	168.0	628.0	0.282
RLSC027	2/8/11	0-0.5	0.20	6.7	17.1	6.4	29.7	0.011

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC028	2/8/11	0-0.5	1.90	72.3	459.0	42.6	398.0	0.079
RLSC029	2/8/11	0-0.5	1.30	25.5	87.6	26.3	132.0	0.029
RLSC030	2/9/11	0-0.5	0.53	17.4	23.2	8.4	52.9	0.034
RLSC031	2/9/11	0-0.5	2.90	24.0	65.2	47.3	179.0	0.077
RLSC032	2/9/11	0-0.5	8.50	97.0	568.0	109.0	662.0	0.887
RLSC033	2/9/11	0-0.5	1.70	23.7	40.5	38.1	116.0	0.065
RLSC034	2/9/11	0-0.5	1.50	40.7	163.0	26.1	241.0	0.038
RLSC035	2/9/11	0-0.5	1.70	41.3	154.0	29.3	295.0	0.032
RLSC036	2/9/11	0-0.5	0.76	20.4	78.4	14.7	124.0	0.037
RLSC037	2/10/11	0-0.5	1.00	32.7	139.0	22.9	225.0	0.039
RLSC038	2/10/11	0-0.5	9.10	83.2	317.0	62.3	424.0	0.052
RLSC039	2/10/11	0-0.5	0.97	40.2	226.0	22.4	230.0	0.040
RLSC040	2/10/11	0-0.5	2.40	44.5	141.0	52.5	264.0	0.027
RLSC041	2/10/11	0-0.5	1.20	28.4	84.8	23.5	152.0	0.028
RLSC042	2/10/11	0-0.5	2.40	80.0	311.0	48.8	460.0	0.185
RLSC043	2/10/11	0-0.5	3.20	60.2	158.0	55.2	294.0	0.028
RLSC044	2/10/11	0-0.5	0.89	17.2	42.3	28.4	95.2	0.072
RLSC045	2/10/11	0-0.5	1.70	34.7	67.2	49.5	202.0	0.033
RLSC046	2/11/11	0-0.5	0.50	49.9	306.0	33.5	477.0	0.026
RLSC047	2/11/11	0-0.5	0.46	24.9	50.7	26.2	121.0	0.046
RLSC048	2/11/11	0-0.5	0.26	45.9	41.0	13.8	91.7	0.026
RLSC049	2/11/11	0-0.5	0.33	20.9	52.8	13.0	110.0	0.038
RLSC050	2/11/11	0-0.5	0.39	9.2	18.9	20.1	45.5	0.018
RLSC051	2/11/11	0-0.5	0.26	7.0	20.5	10.4	36.2	0.024
RLSC052	2/11/11	0-0.5	0.71	25.6	64.3	14.6	121.0	0.023
RLSC053	2/11/11	0-0.5	1.70	23.5	60.6	44.0	149.0	0.027
RLSC054	2/14/11	0-0.5	0.46	13.7	20.2	5.8	66.8	0.037
RLSC055	2/14/11	0-0.5	0.03	4.0	3.2	6.5	20.3	0.011
RLSC056	2/14/11	0-0.5	0.20	20.9	31.2	19.3	84.2	0.025
RLSC057	2/14/11	0-0.5	0.14	16.3	28.9	8.6	71.9	0.030
RLSC058	2/14/11	0-0.5	0.21	22.7	102.0	20.4	141.0	0.037
RLSC059	2/14/11	0-0.5	0.11	17.0	23.3	9.0	53.6	0.025
RLSC060	2/14/11	0-0.5	0.10	13.5	16.8	6.5	48.8	0.023
RLSC061	2/14/11	0-0.5	0.16	22.0	48.0	12.2	64.9	0.022
RLSC062	2/14/11	0-0.5	0.15	14.7	32.0	9.8	75.1	0.026
RLSC063	2/15/11	0-0.5	1.30	21.6	45.5	30.0	111.0	0.056
RLSC064	2/15/11	0-0.5	0.80	26.5	65.7	18.3	143.0	0.029
RLSC065	2/15/11	0-0.5	0.72	31.3	91.1	15.8	153.0	0.033
RLSC066	2/15/11	0-0.5	1.10	31.4	112.0	25.7	194.0	0.032
RLSC067	2/15/11	0-0.5	0.48	18.6	25.4	7.3	73.5	0.023
RLSC068	2/15/11	0-0.5	0.35	21.1	25.7	9.4	76.1	0.029

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC069	2/15/11	0-0.5	0.08	6.0	18.5	4.5	25.6	0.008
RLSC070	2/15/11	0-0.5	4.10	118.0	615.0	85.7	606.0	0.073
RLSC071	2/16/11	0-0.5	0.58	29.5	104.0	17.2	129.0	0.019
RLSC072	2/16/11	0-0.5	0.66	19.9	51.0	19.9	92.9	0.028
RLSC073	2/16/11	0-0.5	2.80	63.4	346.0	70.3	442.0	0.090
RLSC074	2/16/11	0-0.5	4.20	96.9	739.0	88.9	742.0	0.119
RLSC075	2/16/11	0-0.5	0.69	22.0	56.7	11.4	134.0	0.062
RLSC076	2/16/11	0-0.5	0.87	48.2	241.0	22.1	255.0	0.047
RLSC077	2/16/11	0-0.5	0.20	8.6	7.3	3.8	23.2	0.005
RLSC078	2/16/11	0-0.5	0.48	21.7	24.5	9.0	75.0	0.029
RLSC079	2/16/11	0-0.5	0.04	3.1	4.1	1.8	59.7	0.008
RLSC080	2/16/11	0-0.5	1.30	26.6	89.0	34.4	147.0	0.088
RLSC081	2/17/11	0-0.5	1.70	66.5	650.0	60.3	572.0	0.107
RLSC082	2/17/11	0-0.5	0.34	20.5	22.6	7.0	43.9	0.023
RLSC083	2/17/11	0-0.5	0.80	25.7	60.9	15.9	112.0	0.034
RLSC084	2/17/11	0-0.5	3.00	66.6	453.0	56.9	488.0	0.083
RLSC085	2/17/11	0-0.5	15.30	94.4	496.0	156.0	642.0	0.561
RLSC086	2/17/11	0-0.5	3.80	77.6	538.0	61.3	554.0	0.826
RLSC087	2/17/11	0-0.5	0.61	17.5	31.1	23.6	89.3	0.030
RLSC088	2/17/11	0-0.5	0.45	9.5	21.3	14.1	48.1	0.050
RLSC089	2/17/11	0-0.5	0.53	25.0	63.2	21.2	121.0	0.035
RLSC090	2/18/11	0-0.5	1.20	38.1	154.0	30.7	187.0	0.019
RLSC091	2/18/11	0-0.5	0.46	10.7	16.2	25.6	49.6	0.028
RLSC092	2/18/11	0-0.5	1.40	32.9	133.0	28.4	167.0	0.023
RLSC093	2/18/11	0-0.5	0.98	26.1	67.0	25.2	112.0	0.025
RLSC094	2/18/11	0-0.5	0.88	44.2	212.0	22.8	254.0	0.051
RLSC095	2/18/11	0-0.5	0.79	29.6	86.2	16.3	130.0	0.029
RLSC096	2/18/11	0-0.5	2.10	66.1	327.0	46.7	439.0	0.086
RLSC097	2/19/11	0-0.5	14.50	129.0	686.0	143.0	848.0	0.479
RLSC098	2/21/11	0-0.5	15.20	115.0	321.0	167.0	591.0	0.049
RLSC099	2/21/11	0-0.5	0.27	15.5	18.4	9.2	76.3	0.042
RLSC100	2/21/11	0-0.5	0.31	26.0	46.9	14.2	109.0	0.031
RLSC101	2/21/11	0-0.5	0.38	25.8	47.9	18.9	107.0	0.022
RLSC102	2/21/11	0-0.5	0.35	21.9	60.3	17.5	98.7	0.009
RLSC103	2/21/11	0-0.5	0.43	10.7	23.2	10.8	49.3	0.016
RLSC104	2/21/11	0-0.5	1.40	47.2	185.0	38.7	243.0	0.029
RLSC105	2/21/11	0-0.5	0.83	27.8	74.9	29.4	140.0	0.032
RLSC106	2/22/11	0-0.5	0.90	88.2	287.0	36.3	352.0	0.046
RLSC107	2/22/11	0-0.5	0.31	18.5	42.4	9.1	82.5	0.041
RLSC108	2/22/11	0-0.5	1.10	24.6	54.1	21.5	120.0	0.077
RLSC109	2/22/11	0-0.5	0.84	25.2	64.2	16.5	138.0	0.058

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC110	2/22/11	0-0.5	0.69	21.1	53.2	17.1	117.0	0.037
RLSC111	2/22/11	0-0.5	0.77	25.3	54.6	21.1	104.0	0.025
RLSC112	2/23/11	0-0.5	0.65	27.3	68.3	21.3	117.0	0.024
RLSC114	2/23/11	0-0.5	2.40	75.5	466.0	63.1	497.0	0.053
RLSC115	2/23/11	0-0.5	0.96	36.1	149.0	23.6	178.0	0.030
RLSC116	2/23/11	0-0.5	1.90	59.1	378.0	59.9	428.0	0.122
RLSC117	2/23/11	0-0.5	0.66	30.2	116.0	23.9	157.0	0.075
RLSC118	2/23/11	0-0.5	2.00	59.3	378.0	43.1	420.0	0.144
RLSC119	2/23/11	0-0.5	0.55	24.8	71.0	21.4	114.0	0.039
RLSC120	2/23/11	0-0.5	1.00	87.0	510.0	48.7	496.0	0.059
RLSC121	2/23/11	0-0.5	1.10	31.9	77.8	43.6	156.0	0.032
RLSC122	2/23/11	0-0.5	1.20	27.1	72.4	31.3	145.0	0.046
RLSC123	2/23/11	0-0.5	0.92	31.9	78.8	21.8	130.0	0.050
RLSC124	2/24/11	0-0.5	1.10	31.5	124.0	38.5	185.0	0.042
RLSC125	2/24/11	0-0.5	0.44	24.9	48.6	66.1	92.5	0.017
RLSC126	2/24/11	0-0.5	2.20	45.9	141.0	75.1	243.0	0.168
RLSC127	2/24/11	0-0.5	1.00	38.3	133.0	47.2	213.0	0.041
RLSC128	2/24/11	0-0.5	1.60	25.8	69.0	54.3	135.0	0.062
RLSC129	2/24/11	0-0.5	2.40	133.0	924.0	73.0	854.0	0.127
RLSC130	2/24/11	0-0.5	3.40	54.0	210.0	75.1	305.0	0.051
RLSC131	2/24/11	0-0.5	0.74	21.0	32.2	25.2	82.5	0.021
RLSC132	2/24/11	0-0.5	0.43	14.5	20.1	21.5	73.5	0.011
RLSC133	2/24/11	0-0.5	1.10	23.9	54.4	38.5	126.0	0.029
RLSC134	2/24/11	0-0.5	1.30	35.1	109.0	45.8	199.0	0.053
RLSC135	2/24/11	0-0.5	0.15	5.7	7.9	9.6	28.7	0.020
RLSC001	2/1/11	1-2	0.20	10.4	8.6	4.7	24.6	0.008
RLSC002	2/1/11	1-2	0.50	21.6	17.8	8.9	53.1	0.022
RLSC003	2/1/11	1-2	0.25	14.5	18.7	7.5	43.5	0.027
RLSC004	2/1/11	1-2	1.20	35.9	189.0	22.1	171.0	0.023
RLSC005	2/1/11	1-2	0.27	44.7	214.0	13.2	207.0	0.024
RLSC006	2/1/11	1-2	0.23	14.0	12.3	6.2	35.8	0.014
RLSC007	2/1/11	1-2	0.22	17.2	14.8	7.6	49.7	0.019
RLSC008	2/1/11	1-2	0.43	17.5	48.2	11.4	70.7	0.023
RLSC009	2/1/11	1-2	0.18	14.4	16.9	7.4	54.0	0.030
RLSC010	2/1/11	1-2	0.97	81.2	543.0	28.2	413.0	0.028
RLSC011	2/1/11	1-2	0.16	66.8	795.0	18.8	306.0	0.014
RLSC012	2/2/11	1-2	0.12	23.2	21.6	8.6	70.0	0.023
RLSC013	2/2/11	1-2	0.14	15.6	15.6	5.9	56.5	0.031
RLSC014	2/3/11	1-2	0.31	12.5	17.0	5.4	52.3	0.033
RLSC015	2/3/11	1-2	0.58	16.8	27.7	8.9	64.1	0.038
RLSC016	2/3/11	1-2	0.08	22.3	39.4	11.2	82.8	0.006

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC017	2/3/11	1-2	0.10	9.3	10.3	6.5	34.6	0.007
RLSC018	2/3/11	1-2	0.14	72.3	864.0	21.6	494.0	0.025
RLSC019	2/4/11	1-2	0.35	16.1	17.6	6.2	75.9	0.029
RLSC020	2/4/11	1-2	0.23	12.4	13.5	4.8	48.9	0.025
RLSC021	2/4/11	1-2	0.29	14.6	18.1	5.7	60.7	0.029
RLSC022	2/7/11	1-2	0.32	11.1	17.2	5.2	44.2	0.034
RLSC023	2/7/11	1-2	0.18	8.5	12.9	4.3	22.6	0.020
RLSC024	2/8/11	1-2	0.23	8.7	15.2	5.0	42.2	0.030
RLSC025	2/8/11	1-2	0.30	13.8	19.0	5.6	55.3	0.029
RLSC026	2/8/11	1-2	0.29	10.8	17.5	5.3	41.8	0.027
RLSC027	2/8/11	1-2	0.07	6.4	7.5	6.3	20.8	0.008
RLSC028	2/8/11	1-2	0.69	18.6	26.1	8.0	55.0	0.035
RLSC029	2/8/11	1-2	0.52	13.9	22.2	5.5	55.1	0.039
RLSC030	2/9/11	1-2	0.46	16.3	22.9	5.9	61.2	0.041
RLSC032	2/9/11	1-2	0.76	19.1	30.6	9.3	71.5	0.029
RLSC033	2/9/11	1-2	0.33	10.4	8.4	3.8	20.9	0.005
RLSC034	2/9/11	1-2	0.54	15.2	21.7	7.1	51.9	0.025
RLSC035	2/9/11	1-2	0.70	16.2	22.1	6.2	69.7	0.033
RLSC036	2/9/11	1-2	0.66	14.5	20.3	5.9	71.9	0.057
RLSC037	2/10/11	1-2	0.59	13.5	19.5	5.1	51.2	0.039
RLSC038	2/10/11	1-2	0.67	16.4	16.7	7.0	47.8	0.021
RLSC039	2/10/11	1-2	0.58	13.1	17.0	6.8	43.8	0.022
RLSC040	2/10/11	1-2	0.54	9.9	9.7	4.7	32.3	0.005
RLSC041	2/10/11	1-2	0.56	14.6	19.2	6.3	63.1	0.028
RLSC042	2/10/11	1-2	0.93	17.4	28.0	7.6	74.2	0.035
RLSC043	2/10/11	1-2	0.54	8.9	15.0	3.8	44.0	0.029
RLSC044	2/10/11	1-2	0.05	3.6	1.8	1.5	8.2	0.005
RLSC046	2/11/11	1-2	0.10	19.6	21.6	10.0	65.7	0.023
RLSC047	2/11/11	1-2	0.13	15.3	19.4	7.5	56.1	0.029
RLSC048	2/11/11	1-2	0.12	15.8	21.5	7.6	61.0	0.026
RLSC049	2/11/11	1-2	0.40	16.4	19.9	7.7	76.7	0.034
RLSC050	2/11/11	1-2	0.03	5.4	1.9	2.4	9.6	0.005
RLSC051	2/11/11	1-2	0.98	25.1	90.8	15.6	105.0	0.017
RLSC052	2/11/11	1-2	0.32	14.7	18.0	7.4	54.5	0.019
RLSC053	2/11/11	1-2	0.05	9.1	6.7	4.1	20.1	0.005
RLSC054	2/14/11	1-2	0.10	9.3	16.6	5.0	44.6	0.022
RLSC055	2/14/11	1-2	0.02	4.1	0.6	1.6	6.6	0.005
RLSC056	2/14/11	1-2	0.11	16.4	20.1	9.2	57.2	0.024
RLSC057	2/14/11	1-2	0.13	19.4	29.1	10.0	80.9	0.028
RLSC058	2/14/11	1-2	0.02	4.4	1.9	1.7	7.7	0.005
RLSC059	2/14/11	1-2	0.02	5.1	3.6	2.6	16.3	0.005

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC060	2/14/11	1-2	0.13	14.3	17.6	6.3	42.7	0.028
RLSC061	2/14/11	1-2	0.14	12.1	16.3	5.1	44.5	0.030
RLSC062	2/14/11	1-2	0.14	14.2	18.3	6.9	53.7	0.032
RLSC063	2/15/11	1-2	0.40	12.4	16.7	13.2	66.2	0.034
RLSC064	2/15/11	1-2	0.35	14.6	23.6	7.2	82.6	0.036
RLSC065	2/15/11	1-2	0.51	15.1	19.8	5.2	71.5	0.035
RLSC066	2/15/11	1-2	0.44	17.5	23.0	9.6	68.3	0.020
RLSC067	2/15/11	1-2	0.49	18.6	21.9	8.4	67.5	0.021
RLSC068	2/15/11	1-2	0.20	13.0	12.9	6.5	41.6	0.016
RLSC070	2/15/11	1-2	0.30	13.4	17.4	6.1	41.5	0.033
RLSC071	2/16/11	1-2	0.48	16.4	23.6	7.7	63.7	0.049
RLSC072	2/16/11	1-2	0.39	12.8	19.4	5.8	59.9	0.031
RLSC073	2/16/11	1-2	0.71	13.1	21.6	25.4	80.8	0.039
RLSC074	2/16/11	1-2	1.70	20.1	53.6	44.4	148.0	0.028
RLSC075	2/16/11	1-2	0.65	18.5	30.1	8.3	98.0	0.059
RLSC076	2/16/11	1-2	0.43	17.1	21.5	7.6	58.2	0.034
RLSC078	2/16/11	1-2	0.42	15.8	18.1	6.7	59.1	0.028
RLSC079	2/16/11	1-2	0.02	3.5	1.0	1.4	6.8	0.005
RLSC080	2/16/11	1-2	0.18	12.4	11.0	5.4	31.5	0.014
RLSC081	2/17/11	1-2	0.69	16.5	22.9	5.9	60.0	0.045
RLSC082	2/17/11	1-2	0.38	11.8	18.8	6.5	42.4	0.028
RLSC083	2/17/11	1-2	0.64	14.8	19.9	6.8	68.2	0.045
RLSC084	2/17/11	1-2	1.20	16.4	23.0	32.7	93.0	0.052
RLSC085	2/17/11	1-2	0.86	18.7	55.9	33.9	123.0	0.041
RLSC086	2/17/11	1-2	0.72	17.6	22.9	25.9	81.9	0.041
RLSC087	2/17/11	1-2	0.02	4.6	2.6	2.3	10.6	0.005
RLSC088	2/17/11	1-2	0.09	6.6	7.0	3.8	18.4	0.005
RLSC089	2/17/11	1-2	0.64	18.0	20.7	8.9	62.6	0.031
RLSC090	2/18/11	1-2	0.40	12.9	17.6	7.5	45.1	0.016
RLSC091	2/18/11	1-2	0.22	13.9	10.5	6.5	35.8	0.009
RLSC092	2/18/11	1-2	0.52	14.3	18.1	6.7	41.9	0.031
RLSC093	2/18/11	1-2	0.23	10.8	11.1	5.6	34.0	0.006
RLSC094	2/18/11	1-2	0.50	15.4	18.3	6.3	51.8	0.035
RLSC095	2/18/11	1-2	0.55	12.6	15.3	4.8	70.4	0.047
RLSC096	2/18/11	1-2	0.55	16.0	20.9	6.9	70.5	0.043
RLSC097	2/19/11	1-2	1.30	227.0	1540.0	56.6	1250.0	0.034
RLSC098	2/21/11	1-2	1.10	17.3	29.3	37.7	117.0	0.036
RLSC099	2/21/11	1-2	0.18	19.9	21.3	9.8	83.3	0.040
RLSC100	2/21/11	1-2	0.19	15.8	17.8	8.0	55.3	0.031
RLSC101	2/21/11	1-2	0.14	17.9	18.6	8.9	62.0	0.019
RLSC102	2/21/11	1-2	0.17	20.5	17.0	9.9	61.0	0.010

Location	Date	Depth	Cadmium	Chromium	Copper	Lead	Zinc	PCBs
		(ft)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
RLSC103	2/21/11	1-2	0.06	16.7	11.4	7.1	41.5	0.014
RLSC104	2/21/11	1-2	0.17	15.7	14.8	6.4	53.9	0.038
RLSC105	2/21/11	1-2	0.22	21.4	22.0	9.5	77.8	0.041
RLSC106	2/22/11	1-2	0.28	22.4	23.9	10.1	91.9	0.046
RLSC107	2/22/11	1-2	0.68	23.4	28.0	12.1	85.6	0.041
RLSC108	2/22/11	1-2	0.61	19.8	27.1	9.7	74.7	0.031
RLSC109	2/22/11	1-2	0.59	17.4	24.6	8.5	85.1	0.050
RLSC110	2/22/11	1-2	0.52	19.7	21.9	8.8	87.8	0.043
RLSC111	2/22/11	1-2	0.63	15.9	23.0	7.2	79.5	0.045
RLSC112	2/23/11	1-2	0.36	19.2	15.6	8.3	45.5	0.016
RLSC114	2/23/11	1-2	0.33	14.6	18.1	5.9	50.2	0.019
RLSC115	2/23/11	1-2	0.51	18.4	22.5	6.3	59.3	0.036
RLSC116	2/23/11	1-2	0.24	11.6	16.8	5.8	56.1	0.031
RLSC117	2/23/11	1-2	0.40	21.2	31.6	10.7	80.4	0.030
RLSC118	2/23/11	1-2	0.24	17.4	31.9	7.9	75.2	0.024
RLSC119	2/23/11	1-2	0.28	17.5	17.2	6.8	61.5	0.028
RLSC120	2/23/11	1-2	0.27	20.9	40.6	11.0	73.3	0.017
RLSC121	2/23/11	1-2	0.18	19.6	15.1	8.8	49.7	0.015
RLSC122	2/23/11	1-2	0.39	15.2	15.8	12.1	50.9	0.012
RLSC123	2/23/11	1-2	0.30	14.0	15.0	6.4	53.4	0.032
RLSC124	2/24/11	1-2	0.41	15.3	18.5	7.0	60.8	0.029
RLSC125	2/24/11	1-2	0.26	23.8	23.4	10.7	65.1	0.016
RLSC126	2/24/11	1-2	0.31	25.7	29.4	14.3	77.6	0.025
RLSC127	2/24/11	1-2	0.20	17.6	15.7	6.9	41.2	0.016
RLSC128	2/24/11	1-2	0.39	36.0	218.0	28.1	260.0	0.030
RLSC129	2/24/11	1-2	0.33	15.1	27.3	7.2	54.8	0.022
RLSC130	2/24/11	1-2	0.33	16.5	17.1	7.8	55.1	0.018
RLSC131	2/24/11	1-2	0.20	12.2	11.1	5.2	32.1	0.015
RLSC133	2/24/11	1-2	0.33	15.3	23.5	10.9	52.0	0.013
RLSC134	2/24/11	1-2	0.08	15.4	17.4	7.2	44.0	0.012
RLSC135	2/24/11	1-2	0.09	11.1	8.4	5.3	25.4	0.007

Data analyses performed with SAS Univariate Procedure (SAS 9.2, 3rd edition, 2013).

Round Lake Univariate stats for Cadmium Each Depth - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-0.5

Moments			
N	134	Sum Weights	134
Mean	0.08582704	Sum Observations	11.500823
Std Deviation	1.24872279	Variance	1.55930861
Skewness	0.01489094	Kurtosis	0.2598918
Uncorrected SS	208.375127	Corrected SS	207.388046
Coeff Variation	1454.9294	Std Error Mean	0.10787322

Basic Statistical Measures			
Location		Variability	
Mean	0.08583	Std Deviation	1.24872
Median	-0.01010	Variance	1.55931
Mode	0.09531	Range	6.82137
		Interquartile Range	1.60944

Note: The mode displayed is the smallest of 2 modes with a count of 5.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	0.795629	Pr > |t|	0.4277
Sign	M	-1.5	Pr >= |M|	0.8614
Signed Rank	S	270.5	Pr >= |S|	0.5364

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.991478	Pr < W	0.5934
Kolmogorov-Smirnov	D	0.056672	Pr > D	>0.1500
Cramer-von Mises	W-Sq	0.067484	Pr > W-Sq	>0.2500
Anderson-Darling	A-Sq	0.399433	Pr > A-Sq	>0.2500

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	3.2809112
99%	2.7278528
95%	2.2823824
90%	1.8245493
75% Q3	0.8754687
50% Median	-0.0101014
25% Q1	-0.7339692
10%	-1.3470736

Depth=0-0.5

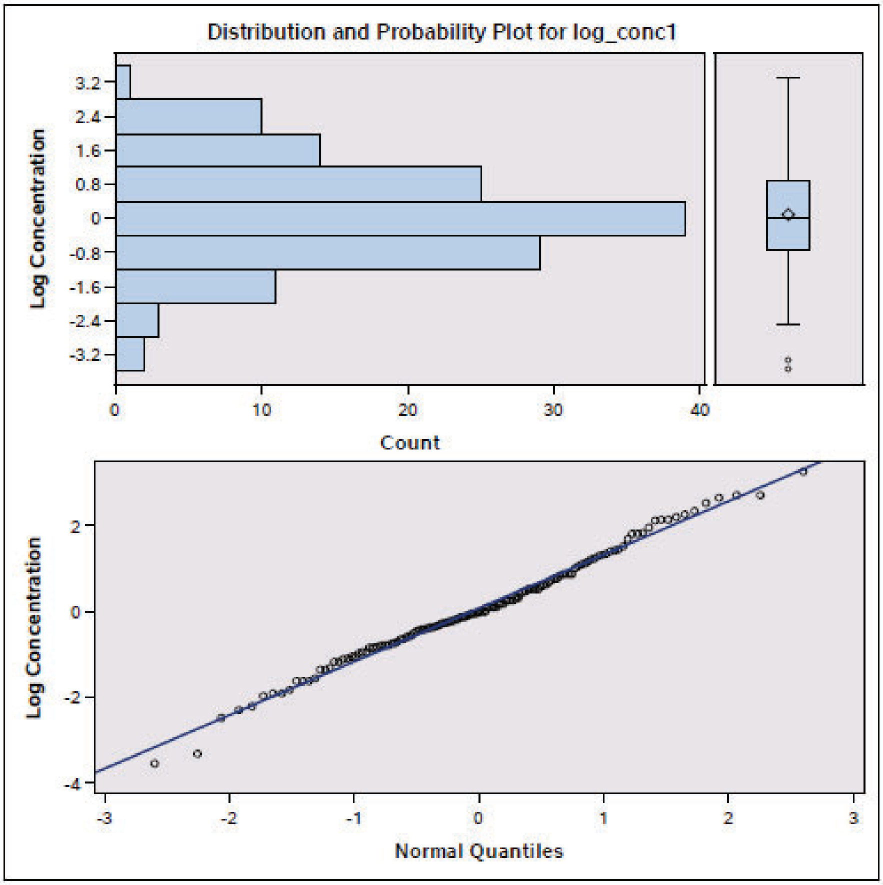
Quantiles (Definition 5)	
Quantile	Estimate
5%	-1.8971200
1%	-3.3242363
0% Min	-3.5404594

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-3.54046	55	2.54160	10
-3.32424	79	2.67415	97
-2.47694	69	2.72130	98
-2.30259	60	2.72785	85
-2.20727	59	3.28091	8

Round Lake Univariate stats for Cadmium Each Depth - Log-Concentration

The UNIVARIATE Procedure

Depth=0-0.5



Round Lake Univariate stats for Cadmium 0-1 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-1

Moments			
N	268	Sum Weights	268
Mean	-0.4188353	Sum Observations	-112.24786
Std Deviation	1.28214453	Variance	1.64389461
Skewness	0.19539538	Kurtosis	0.58613703
Uncorrected SS	485.933224	Corrected SS	438.91986
Coeff Variation	-306.12142	Std Error Mean	0.07831944

Basic Statistical Measures			
Location		Variability	
Mean	-0.41884	Std Deviation	1.28214
Median	-0.54473	Variance	1.64389
Mode	0.09531	Range	7.16662
		Interquartile Range	1.37103

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	-5.34778	Pr > |t|	<.0001
Sign	M	-48.5	Pr >= |M|	<.0001
Signed Rank	S	-6941	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.982906	Pr < W	0.0027
Kolmogorov-Smirnov	D	0.068089	Pr > D	<0.0100
Cramer-von Mises	W-Sq	0.310214	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	1.733509	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	3.349904
99%	2.727853
95%	1.974081
90%	1.308333
75% Q3	0.262364
50% Median	-0.544727
25% Q1	-1.108663
10%	-1.897120

Depth=0-1

Quantiles (Definition 5)	
Quantile	Estimate
5%	-2.441847
1%	-3.688879
0% Min	-3.816713

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-3.81671	213	2.67415	97
-3.81671	189	2.72130	98
-3.68888	192	2.72785	85
-3.54046	55	3.28091	8
-3.44202	179	3.34990	152

Round Lake Univariate stats for Cadmium 0-2 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-2

Moments			
N	397	Sum Weights	397
Mean	-0.6986178	Sum Observations	-277.35127
Std Deviation	1.24028034	Variance	1.53829533
Skewness	0.24919952	Kurtosis	0.92889537
Uncorrected SS	802.927485	Corrected SS	609.16495
Coeff Variation	-177.53346	Std Error Mean	0.06224789

Basic Statistical Measures			
Location		Variability	
Mean	-0.69862	Std Deviation	1.24028
Median	-0.73397	Variance	1.53830
Mode	-0.94161	Range	7.21314
		Interquartile Range	1.38629

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	-11.2232	Pr > |t|	<.0001
Sign	M	-108	Pr >= |M|	<.0001
Signed Rank	S	-23616	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.977055	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.07539	Pr > D	<0.0100
Cramer-von Mises	W-Sq	0.530721	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	3.061748	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	3.349904
99%	2.721295
95%	1.704748
90%	0.875469
75% Q3	-0.040822
50% Median	-0.733969
25% Q1	-1.427116
10%	-2.120264

Depth=0-2

Quantiles (Definition 5)	
Quantile	Estimate
5%	-2.703063
1%	-3.816713
0% Min	-3.863233

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-3.86323	351	2.67415	97
-3.81671	343	2.72130	98
-3.81671	324	2.72785	85
-3.81671	321	3.28091	8
-3.81671	213	3.34990	152

Round Lake Univariate stats for Chromium Each Depth - Log-Concentratior

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-0.5

Moments			
N	134	Sum Weights	134
Mean	3.49260118	Sum Observations	468.008558
Std Deviation	0.76765172	Variance	0.58928916
Skewness	-0.2463308	Kurtosis	0.44983998
Uncorrected SS	1712.9427	Corrected SS	78.3754587
Coeff Variation	21.9793696	Std Error Mean	0.06631501

Basic Statistical Measures			
Location		Variability	
Mean	3.492601	Std Deviation	0.76765
Median	3.397808	Variance	0.58929
Mode	2.370244	Range	4.55557
		Interquartile Range	1.02830

Note: The mode displayed is the smallest of 10 modes with a count of 2.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	52.66683	Pr > |t|	<.0001
Sign	M	67	Pr >= |M|	<.0001
Signed Rank	S	4522.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.980424	Pr < W	0.0507
Kolmogorov-Smirnov	D	0.080581	Pr > D	0.0321
Cramer-von Mises	W-Sq	0.159726	Pr > W-Sq	0.0188
Anderson-Darling	A-Sq	0.955115	Pr > A-Sq	0.0169

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	5.68698
99%	4.89035
95%	4.63473
90%	4.50092
75% Q3	4.10099
50% Median	3.39781
25% Q1	3.07269
10%	2.61740

Depth=0-0.5

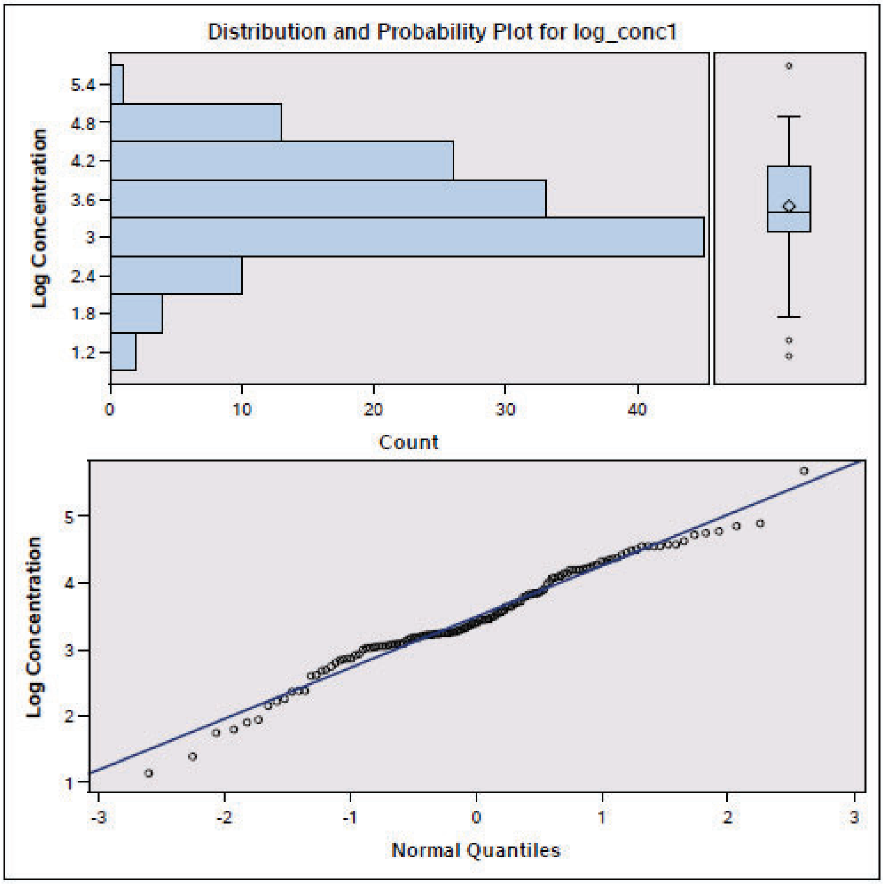
Quantiles (Definition 5)	
Quantile	Estimate
5%	2.15176
1%	1.38629
0% Min	1.13140

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
1.13140	79	4.74493	98
1.38629	55	4.77068	70
1.74047	134	4.85981	97
1.79176	69	4.89035	128
1.90211	27	5.68698	10

Round Lake Univariate stats for Chromium Each Depth - Log-Concentration

The UNIVARIATE Procedure

Depth=0-0.5



Round Lake Univariate stats for Chromium 0-1 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-1

Moments			
N	268	Sum Weights	268
Mean	3.23853735	Sum Observations	867.928011
Std Deviation	0.80738851	Variance	0.65187621
Skewness	0.48064454	Kurtosis	1.07445536
Uncorrected SS	2984.86823	Corrected SS	174.050948
Coeff Variation	24.930653	Std Error Mean	0.0493191

Basic Statistical Measures			
Location		Variability	
Mean	3.238537	Std Deviation	0.80739
Median	3.091042	Variance	0.65188
Mode	2.602690	Range	5.63970
		Interquartile Range	0.95476

Note: The mode displayed is the smallest of 2 modes with a count of 4.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	65.66497	Pr > |t|	<.0001
Sign	M	134	Pr >= |M|	<.0001
Signed Rank	S	18023	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.969197	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.108059	Pr > D	<0.0100
Cramer-von Mises	W-Sq	0.644378	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	3.238949	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	6.70441
99%	5.28320
95%	4.61512
90%	4.45085
75% Q3	3.69882
50% Median	3.09104
25% Q1	2.74406
10%	2.36085

Depth=0-1

Quantiles (Definition 5)	
Quantile	Estimate
5%	1.94591
1%	1.16315
0% Min	1.06471

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
1.06471	189	4.87520	231
1.13140	79	4.89035	128
1.16315	213	5.28320	219
1.38629	55	5.68698	10
1.66771	151	6.70441	139

Round Lake Univariate stats for Chromium 0-2 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-2

Moments			
N	397	Sum Weights	397
Mean	3.07338168	Sum Observations	1220.13253
Std Deviation	0.7668089	Variance	0.58799588
Skewness	0.72616813	Kurtosis	1.63053212
Uncorrected SS	3982.77933	Corrected SS	232.84637
Coeff Variation	24.9500054	Std Error Mean	0.03848504

Basic Statistical Measures			
Location		Variability	
Mean	3.073382	Std Deviation	0.76681
Median	2.928524	Variance	0.58800
Mode	2.797281	Range	5.63970
		Interquartile Range	0.70007

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	79.85914	Pr > |t|	<.0001
Sign	M	198.5	Pr >= |M|	<.0001
Signed Rank	S	39501.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.945584	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.123812	Pr > D	<0.0100
Cramer-von Mises	W-Sq	1.744905	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	8.842216	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	6.70441
99%	5.28320
95%	4.55388
90%	4.22975
75% Q3	3.36730
50% Median	2.92852
25% Q1	2.66723
10%	2.29253

Depth=0-2

Quantiles (Definition 5)	
Quantile	Estimate
5%	1.88707
1%	1.25276
0% Min	1.06471

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
1.06471	189	4.89035	128
1.13140	79	5.28320	219
1.16315	213	5.42495	361
1.25276	343	5.68698	10
1.28093	311	6.70441	139

Round Lake Univariate stats for Copper Each Depth - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-0.5

Moments			
N	134	Sum Weights	134
Mean	4.62106427	Sum Observations	619.222613
Std Deviation	1.21082999	Variance	1.46610927
Skewness	-0.1988124	Kurtosis	-0.4045082
Uncorrected SS	3056.46003	Corrected SS	194.992532
Coeff Variation	26.2024053	Std Error Mean	0.10459978

Basic Statistical Measures			
Location		Variability	
Mean	4.621064	Std Deviation	1.21083
Median	4.480709	Variance	1.46611
Mode	5.934894	Range	5.66556
		Interquartile Range	1.89178

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	44.17853	Pr > |t|	<.0001
Sign	M	67	Pr >= |M|	<.0001
Signed Rank	S	4522.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.976709	Pr < W	0.0212
Kolmogorov-Smirnov	D	0.079928	Pr > D	0.0353
Cramer-von Mises	W-Sq	0.111042	Pr > W-Sq	0.0831
Anderson-Darling	A-Sq	0.789272	Pr > A-Sq	0.0414

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	6.82871
99%	6.80461
95%	6.52942
90%	6.27288
75% Q3	5.73979
50% Median	4.48071
25% Q1	3.84802
10%	3.05871

Depth=0-0.5

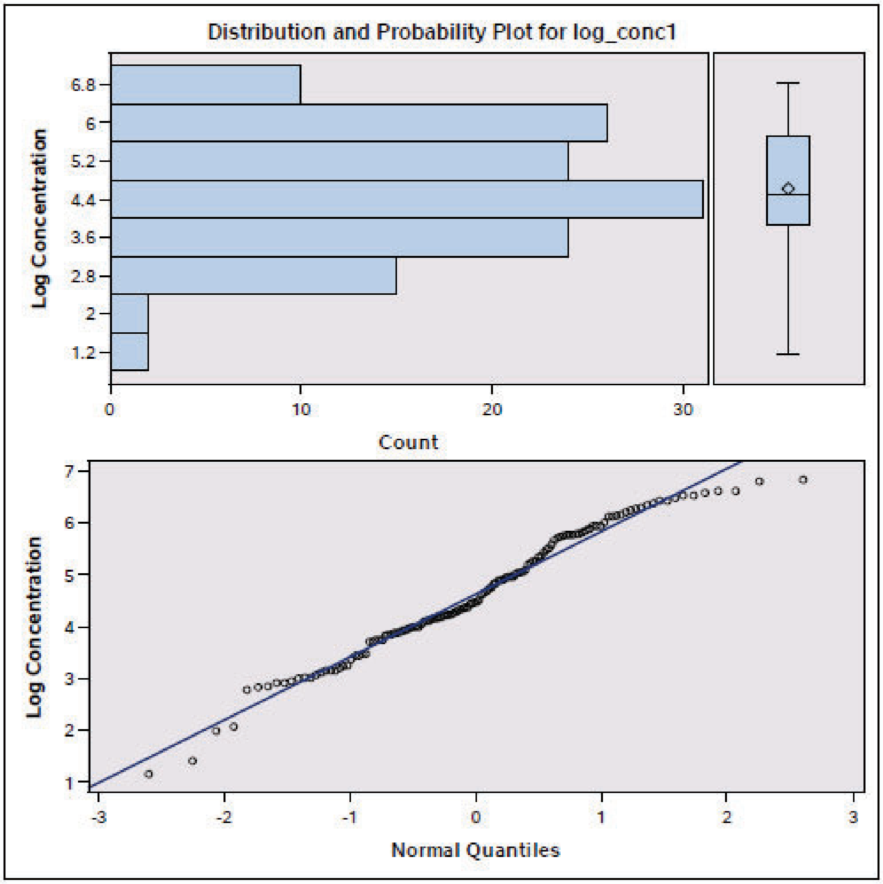
Quantiles (Definition 5)	
Quantile	Estimate
5%	2.83908
1%	1.41099
0% Min	1.16315

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
1.16315	55	6.57508	24
1.41099	79	6.60530	74
1.98787	77	6.60800	19
2.06686	134	6.80461	1
2.78501	91	6.82871	128

Round Lake Univariate stats for Copper Each Depth - Log-Concentration

The UNIVARIATE Procedure

Depth=0-0.5



Round Lake Univariate stats for Copper 0-1 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-1

Moments			
N	268	Sum Weights	268
Mean	4.07599075	Sum Observations	1092.36552
Std Deviation	1.40536918	Variance	1.97506254
Skewness	0.17550186	Kurtosis	-0.3101329
Uncorrected SS	4979.81347	Corrected SS	527.341699
Coeff Variation	34.4792044	Std Error Mean	0.08584659

Basic Statistical Measures			
Location		Variability	
Mean	4.075991	Std Deviation	1.40537
Median	3.858567	Variance	1.97506
Mode	2.954910	Range	7.69888
		Interquartile Range	2.00624

Note: The mode displayed is the smallest of 7 modes with a count of 3.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	47.47994	Pr > |t|	<.0001
Sign	M	132	Pr >= |M|	<.0001
Signed Rank	S	18020	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.963658	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.110544	Pr > D	<0.0100
Cramer-von Mises	W-Sq	0.71412	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	4.150963	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	7.313220
99%	6.946976
95%	6.529419
90%	6.206576
75% Q3	5.062595
50% Median	3.858567
25% Q1	3.056354
10%	2.740840

Depth=0-1

Quantiles (Definition 5)	
Quantile	Estimate
5%	2.041220
1%	1.163151
0% Min	-0.385662

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-0.3856625	189	6.80461	1
-0.0618754	213	6.82871	128
1.1631508	55	6.94698	137
1.2237754	179	7.25841	231
1.3862944	151	7.31322	219

Round Lake Univariate stats for Copper 0-2 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-2

Moments			
N	397	Sum Weights	397
Mean	3.72333751	Sum Observations	1478.16499
Std Deviation	1.39224599	Variance	1.93834889
Skewness	0.45524981	Kurtosis	0.21454524
Uncorrected SS	6271.29333	Corrected SS	767.586159
Coeff Variation	37.3924196	Std Error Mean	0.06987482

Basic Statistical Measures			
Location		Variability	
Mean	3.723338	Std Deviation	1.39225
Median	3.202746	Variance	1.93835
Mode	2.965273	Range	7.80157
		Interquartile Range	1.61605

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	53.28582	Pr > |t|	<.0001
Sign	M	194.5	Pr >= |M|	<.0001
Signed Rank	S	39491.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.932724	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.15883	Pr > D	<0.0100
Cramer-von Mises	W-Sq	2.464025	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	12.78206	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	7.339538
99%	6.946976
95%	6.421622
90%	5.934894
75% Q3	4.511958
50% Median	3.202746
25% Q1	2.895912
10%	2.541602

Depth=0-2

Quantiles (Definition 5)	
Quantile	Estimate
5%	1.960095
1%	-0.040822
0% Min	-0.462035

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-0.4620355	321	6.82871	128
-0.3856625	189	6.94698	137
-0.0618754	213	7.25841	231
-0.0408220	343	7.31322	219
0.5877867	311	7.33954	361

Round Lake Univariate stats for Lead Each Depth - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-0.5

Moments			
N	134	Sum Weights	134
Mean	3.38419891	Sum Observations	453.482654
Std Deviation	0.90605711	Variance	0.82093949
Skewness	-0.0803225	Kurtosis	-0.0030988
Uncorrected SS	1643.86045	Corrected SS	109.184952
Coeff Variation	26.7731636	Std Error Mean	0.07827141

Basic Statistical Measures			
Location		Variability	
Mean	3.384199	Std Deviation	0.90606
Median	3.267664	Variance	0.82094
Mode	3.068053	Range	4.96517
		Interquartile Range	1.19516

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	43.23672	Pr > |t|	<.0001
Sign	M	67	Pr >= |M|	<.0001
Signed Rank	S	4522.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.992792	Pr < W	0.7303
Kolmogorov-Smirnov	D	0.057801	Pr > D	>0.1500
Cramer-von Mises	W-Sq	0.058522	Pr > W-Sq	>0.2500
Anderson-Darling	A-Sq	0.327051	Pr > A-Sq	>0.2500

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	5.552960
99%	5.273000
95%	5.023881
90%	4.487512
75% Q3	4.034241
50% Median	3.267664
25% Q1	2.839078
10%	2.208274

Depth=0-0.5

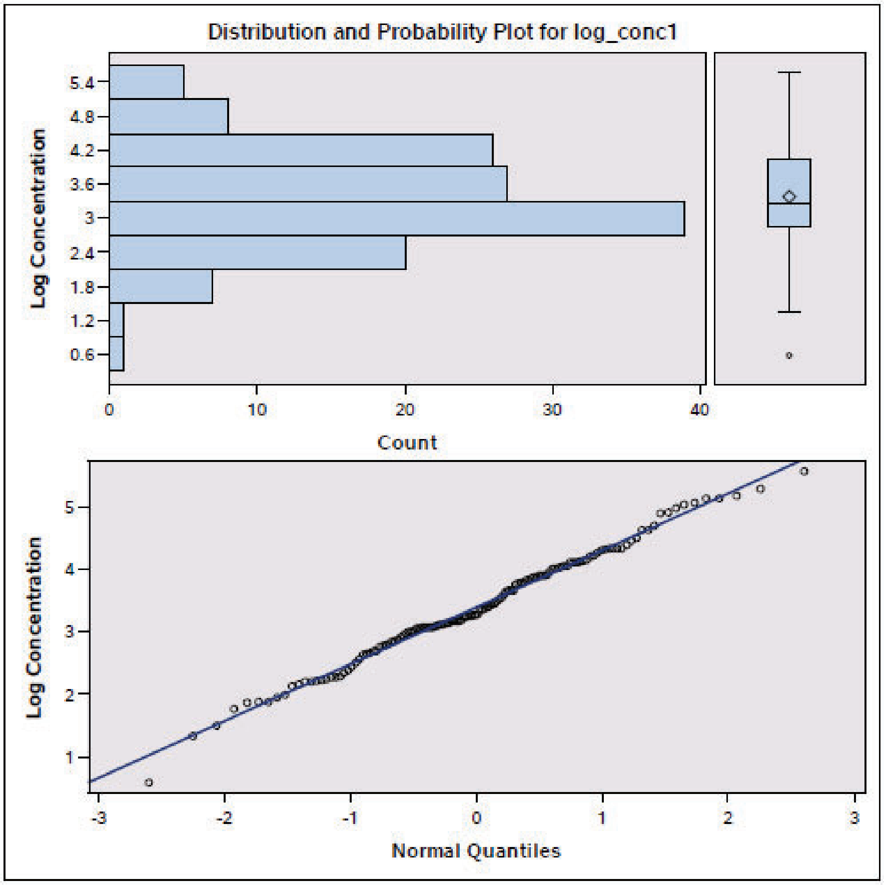
Quantiles (Definition 5)	
Quantile	Estimate
5%	1.871802
1%	1.335001
0% Min	0.587787

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
0.587787	79	5.11799	98
1.335001	77	5.12396	26
1.504077	69	5.16479	18
1.757858	54	5.27300	10
1.856298	27	5.55296	16

Round Lake Univariate stats for Lead Each Depth - Log-Concentration

The UNIVARIATE Procedure

Depth=0-0.5



Round Lake Univariate stats for Lead 0-1 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-1

Moments			
N	268	Sum Weights	268
Mean	2.90310635	Sum Observations	778.032501
Std Deviation	1.00602971	Variance	1.01209578
Skewness	0.23622737	Kurtosis	-0.4555478
Uncorrected SS	2528.94066	Corrected SS	270.229574
Coeff Variation	34.6535604	Std Error Mean	0.06145305

Basic Statistical Measures			
Location		Variability	
Mean	2.903106	Std Deviation	1.00603
Median	2.920379	Variance	1.01210
Mode	1.740466	Range	5.29060
		Interquartile Range	1.54652

Note: The mode displayed is the smallest of 6 modes with a count of 4.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	47.24105	Pr > |t|	<.0001
Sign	M	134	Pr >= |M|	<.0001
Signed Rank	S	18023	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.981353	Pr < W	0.0014
Kolmogorov-Smirnov	D	0.104911	Pr > D	<0.0100
Cramer-von Mises	W-Sq	0.337192	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	1.993387	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	5.552960
99%	5.164786
95%	4.653960
90%	4.252772
75% Q3	3.650658
50% Median	2.920379
25% Q1	2.104134
10%	1.757858

Depth=0-1

Quantiles (Definition 5)	
Quantile	Estimate
5%	1.526056
1%	0.587787
0% Min	0.262364

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
0.262364	213	5.11799	98
0.470004	189	5.12396	26
0.587787	79	5.16479	18
0.955511	192	5.27300	10
0.955511	151	5.55296	16

Round Lake Univariate stats for Lead 0-2 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-2

Moments			
N	397	Sum Weights	397
Mean	2.61804507	Sum Observations	1039.36389
Std Deviation	0.98541026	Variance	0.97103338
Skewness	0.54123668	Kurtosis	-0.1202144
Uncorrected SS	3105.63073	Corrected SS	384.529217
Coeff Variation	37.6391633	Std Error Mean	0.04945632

Basic Statistical Measures			
Location		Variability	
Mean	2.618045	Std Deviation	0.98541
Median	2.282382	Variance	0.97103
Mode	1.840550	Range	5.29060
		Interquartile Range	1.34884

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	52.93651	Pr > |t|	<.0001
Sign	M	198.5	Pr >= |M|	<.0001
Signed Rank	S	39501.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.956384	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.141938	Pr > D	<0.0100
Cramer-von Mises	W-Sq	1.4653	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	7.633838	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	5.552960
99%	5.123964
95%	4.380776
90%	4.010963
75% Q3	3.265759
50% Median	2.282382
25% Q1	1.916923
10%	1.648659

Depth=0-2

Quantiles (Definition 5)	
Quantile	Estimate
5%	1.335001
1%	0.470004
0% Min	0.262364

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
0.262364	213	5.11799	98
0.336472	343	5.12396	26
0.405465	311	5.16479	18
0.470004	321	5.27300	10
0.470004	189	5.55296	16

Round Lake Univariate stats for Zinc Each Depth - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-0.5

Moments			
N	134	Sum Weights	134
Mean	5.16403601	Sum Observations	691.980826
Std Deviation	0.88085224	Variance	0.77590066
Skewness	-0.1182887	Kurtosis	-0.6676651
Uncorrected SS	3676.60869	Corrected SS	103.194788
Coeff Variation	17.0574379	Std Error Mean	0.07609404

Basic Statistical Measures			
Location		Variability	
Mean	5.164036	Std Deviation	0.88085
Median	5.027159	Variance	0.77590
Mode	4.795791	Range	3.73931
		Interquartile Range	1.43047

Note: The mode displayed is the smallest of 2 modes with a count of 3.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	67.86387	Pr > |t|	<.0001
Sign	M	67	Pr >= |M|	<.0001
Signed Rank	S	4522.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.973159	Pr < W	0.0094
Kolmogorov-Smirnov	D	0.086337	Pr > D	0.0156
Cramer-von Mises	W-Sq	0.154623	Pr > W-Sq	0.0214
Anderson-Darling	A-Sq	1.037246	Pr > A-Sq	0.0097

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	6.74993
99%	6.74288
95%	6.46459
90%	6.38182
75% Q3	5.98645
50% Median	5.02716
25% Q1	4.55598
10%	3.98155

Depth=0-0.5

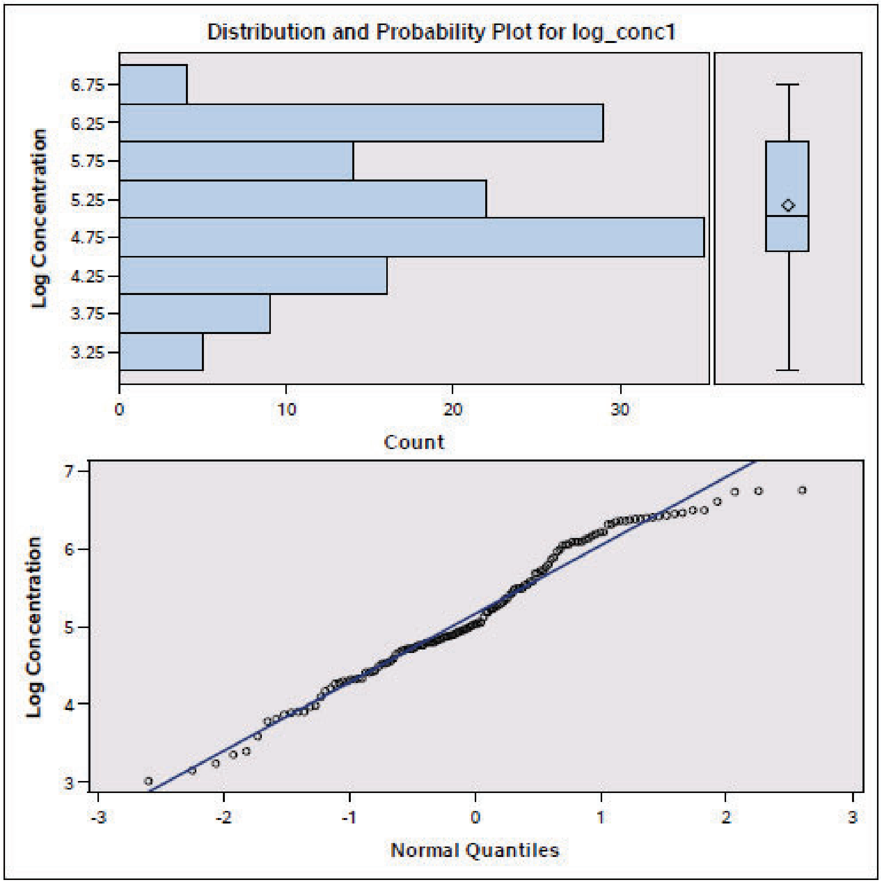
Quantiles (Definition 5)	
Quantile	Estimate
5%	3.78191
1%	3.14415
0% Min	3.01062

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
3.01062	55	6.49828	10
3.14415	77	6.60935	74
3.24259	69	6.73459	18
3.35690	134	6.74288	97
3.39115	27	6.74993	128

Round Lake Univariate stats for Zinc Each Depth - Log-Concentration

The UNIVARIATE Procedure

Depth=0-0.5



Round Lake Univariate stats for Zinc 0-1 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-1

Moments			
N	268	Sum Weights	268
Mean	4.76836713	Sum Observations	1277.92239
Std Deviation	0.98477653	Variance	0.9697848
Skewness	0.19004454	Kurtosis	-0.3238608
Uncorrected SS	6352.53565	Corrected SS	258.932543
Coeff Variation	20.6522799	Std Error Mean	0.0601548

Basic Statistical Measures			
Location		Variability	
Mean	4.768367	Std Deviation	0.98478
Median	4.610145	Variance	0.96978
Mode	4.700480	Range	5.08742
		Interquartile Range	1.37378

Note: The mode displayed is the smallest of 3 modes with a count of 3.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	79.26827	Pr > |t|	<.0001
Sign	M	134	Pr >= |M|	<.0001
Signed Rank	S	18023	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.972138	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.086287	Pr > D	<0.0100
Cramer-von Mises	W-Sq	0.520456	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	3.098549	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	7.04752
99%	6.74993
95%	6.42811
90%	6.31716
75% Q3	5.46144
50% Median	4.61015
25% Q1	4.08765
10%	3.74005

Depth=0-1

Quantiles (Definition 5)	
Quantile	Estimate
5%	3.24259
1%	2.50144
0% Min	1.96009

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
1.96009	189	6.73459	18
2.07944	213	6.74288	97
2.50144	151	6.74993	128
2.58022	192	6.99393	231
2.80336	203	7.04752	219

Round Lake Univariate stats for Zinc 0-2 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-2

Moments			
N	397	Sum Weights	397
Mean	4.52568635	Sum Observations	1796.69748
Std Deviation	0.97208808	Variance	0.94495523
Skewness	0.37080138	Kurtosis	0.251535
Uncorrected SS	8505.49152	Corrected SS	374.20227
Coeff Variation	21.4793514	Std Error Mean	0.0487877

Basic Statistical Measures			
Location		Variability	
Mean	4.525686	Std Deviation	0.97209
Median	4.306764	Variance	0.94496
Mode	3.725693	Range	5.24383
		Interquartile Range	1.03177

Note: The mode displayed is the smallest of 5 modes with a count of 3.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	92.76285	Pr > |t|	<.0001
Sign	M	198.5	Pr >= |M|	<.0001
Signed Rank	S	39501.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.955792	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.128136	Pr > D	<0.0100
Cramer-von Mises	W-Sq	1.48294	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	7.967326	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	7.13090
99%	6.74993
95%	6.38856
90%	6.11810
75% Q3	5.00395
50% Median	4.30676
25% Q1	3.97218
10%	3.57795

Depth=0-2

Quantiles (Definition 5)	
Quantile	Estimate
5%	3.03975
1%	2.04122
0% Min	1.88707

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
1.88707	321	6.74288	97
1.91692	343	6.74993	128
1.96009	189	6.99393	231
2.04122	324	7.04752	219
2.07944	213	7.13090	361

Round Lake Univariate stats for PCBs Each Depth - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-0.5

Moments			
N	134	Sum Weights	134
Mean	-2.9952328	Sum Observations	-401.36119
Std Deviation	0.99527397	Variance	0.99057027
Skewness	0.91219932	Kurtosis	0.80587522
Uncorrected SS	1333.91606	Corrected SS	131.745846
Coeff Variation	-33.228602	Std Error Mean	0.08597858

Basic Statistical Measures			
Location		Variability	
Mean	-2.99523	Std Deviation	0.99527
Median	-3.25645	Variance	0.99057
Mode	-4.46977	Range	5.16789
		Interquartile Range	1.05010

Note: The mode displayed is the smallest of 2 modes with a count of 3.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	-34.837	Pr > |t|	<.0001
Sign	M	-67	Pr >= |M|	<.0001
Signed Rank	S	-4522.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.923842	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.138028	Pr > D	<0.0100
Cramer-von Mises	W-Sq	0.750883	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	4.086332	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	-0.120474
99%	-0.191766
95%	-0.736055
90%	-1.394327
75% Q3	-2.558123
50% Median	-3.256450
25% Q1	-3.608222
10%	-3.865617

Depth=0-0.5

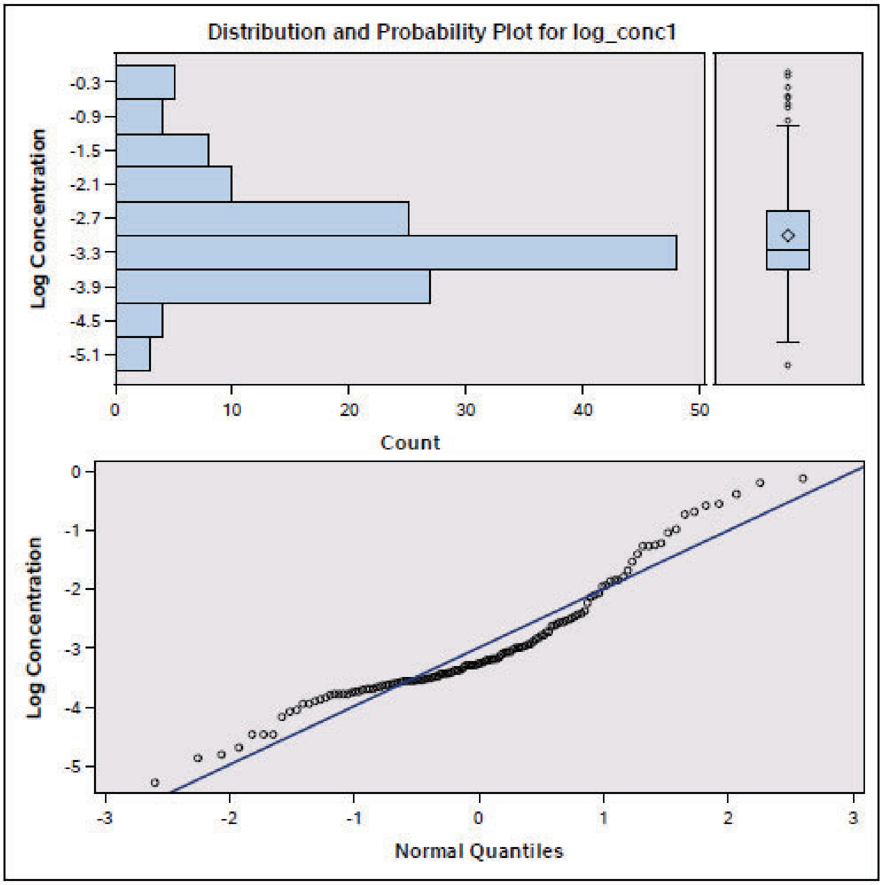
Quantiles (Definition 5)	
Quantile	Estimate
5%	-4.469766
1%	-4.866535
0% Min	-5.288367

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-5.28837	77	-0.578926	85
-4.86653	79	-0.549307	8
-4.81589	69	-0.390971	19
-4.68313	102	-0.191766	86
-4.46977	131	-0.120474	32

Round Lake Univariate stats for PCBs Each Depth - Log-Concentration

The UNIVARIATE Procedure

Depth=0-0.5



Round Lake Univariate stats for PCBs 0-1 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-1

Moments			
N	268	Sum Weights	268
Mean	-3.30584	Sum Observations	-885.96511
Std Deviation	0.99524024	Variance	0.99050314
Skewness	1.38102144	Kurtosis	4.62771766
Uncorrected SS	3193.32322	Corrected SS	264.464338
Coeff Variation	-30.105518	Std Error Mean	0.06079397

Basic Statistical Measures			
Location		Variability	
Mean	-3.30584	Std Deviation	0.99524
Median	-3.42805	Variance	0.99050
Mode	-3.69289	Range	7.56130
		Interquartile Range	0.77004

Note: The mode displayed is the smallest of 5 modes with a count of 4.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	-54.3778	Pr > |t|	<.0001
Sign	M	-132	Pr >= |M|	<.0001
Signed Rank	S	-17993	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.890402	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.143874	Pr > D	<0.0100
Cramer-von Mises	W-Sq	1.726574	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	9.025682	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	2.201106
99%	-0.120474
95%	-1.262308
90%	-2.103734
75% Q3	-2.995732
50% Median	-3.428055
25% Q1	-3.765770
10%	-4.286716

Depth=0-1

Quantiles (Definition 5)	
Quantile	Estimate
5%	-4.815891
1%	-5.328777
0% Min	-5.360193

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-5.36019	213	-0.390971	19
-5.32878	189	-0.191766	86
-5.32878	151	-0.120474	32
-5.29832	211	0.168476	142
-5.28837	203	2.201106	152

Round Lake Univariate stats for PCBs 0-2 ft - Log-Concentration

The UNIVARIATE Procedure
Variable: log_conc1 (Log Concentration)

Depth=0-2

Moments			
N	397	Sum Weights	397
Mean	-3.4719617	Sum Observations	-1378.3688
Std Deviation	0.92871792	Variance	0.86251698
Skewness	1.24751816	Kurtosis	5.30241333
Uncorrected SS	5127.20042	Corrected SS	341.556723
Coeff Variation	-26.749083	Std Error Mean	0.04661102

Basic Statistical Measures			
Location		Variability	
Mean	-3.47196	Std Deviation	0.92872
Median	-3.51493	Variance	0.86252
Mode	-3.53702	Range	7.56130
		Interquartile Range	0.65291

Note: The mode displayed is the smallest of 2 modes with a count of 7.

Tests for Location: Mu0=0				
Test	Statistic		p Value	
Student's t	t	-74.488	Pr > |t|	<.0001
Sign	M	-196.5	Pr >= |M|	<.0001
Signed Rank	S	-39471.5	Pr >= |S|	<.0001

Tests for Normality				
Test	Statistic		p Value	
Shapiro-Wilk	W	0.887928	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.138825	Pr > D	<0.0100
Cramer-von Mises	W-Sq	2.384546	Pr > W-Sq	<0.0050
Anderson-Darling	A-Sq	12.74879	Pr > A-Sq	<0.0050

Quantiles (Definition 5)	
Quantile	Estimate
100% Max	2.201106
99%	-0.191766
95%	-1.784685
90%	-2.584948
75% Q3	-3.184474
50% Median	-3.514926
25% Q1	-3.837379
10%	-4.537512

Depth=0-2

Quantiles (Definition 5)	
Quantile	Estimate
5%	-5.067206
1%	-5.360193
0% Min	-5.360193

Extreme Observations			
Lowest		Highest	
Value	Obs	Value	Obs
-5.36019	352	-0.390971	19
-5.36019	351	-0.191766	86
-5.36019	343	-0.120474	32
-5.36019	324	0.168476	142
-5.36019	300	2.201106	152

APPENDIX C

ECOLOGICAL RECPTOR EXPOSURE AND DOSE CALCULATIONS

Piscivorous Mammal and Bird Exposure, Dose, and HQ Calculations

PCB Fish Tissue Results (MNDNR 2013 sampling)

Year	DATECOL	SPEC	ANAT	NOFISH	LGTHIN	WTKG	PCBPPM	PCBCODE
2012	121012	BKB	FILET	4	6.4	0.06	0.025	K
2012	121012	BKB	FILET	4	7	0.085	0.04	
2012	121012	BKB	WHORG	2	7.3	0.1	0.262	
2012	121012	BRB	FILET	5	7.5	0.094	0.025	K
2012	121012	BRB	FILET	4	7.8	0.11	0.025	K
2012	121012	BRB	WHORG	2	8.1	0.11	0.132	
2012	121012	GSF	FILSK	4	4.6	0.04	0.025	K
2012	121012	GSF	FILSK	4	4.6	0.038	0.025	K
2012	121012	GSF	WHORG	2	5.1	0.055	0.18	

Exposure, Dose, and HQ Calculations

Dose= ($C_f \cdot NIR_f$)		Cf is max	PCB tissue conc						
	Cf	NIR	Dose		NOAEL	LOAEL		HQ	HQ
	mg/kg	kg/kg-d	mg/kg-d		mg/kg-d	mg/kg-d		NOAEL	LOAEL
Mink	0.0400	0.2200	0.0088		0.1360	0.6800		0.0647	0.0129
Blue heron	0.2620	0.1800	0.0472		0.1760	0.8800		0.2680	0.0536
Kingfisher	0.2620	0.0672	0.0176		0.1760	0.8800		0.1000	0.0200
Bald eagle	0.2620	0.3710	0.0972		0.1760	0.8800		0.5523	0.1105

Aquatic Mammal (Muskrat) Exposure, Dose and HQ Calculations

0-0.5 ft, from grid locations > 887 msl (40 grid locations)

	depth	Cd	Cr	Cu	Pb	Zn	PCBs
	ft	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
Mean		2.11	34.55	137.95	48.71	207.62	0.08
Geomean		0.88	23.83	67.53	29.91	132.98	0.04
Median		1.10	25.80	72.40	31.30	145.00	0.04
Max		10.5	118	902	258	841	0.5

Vegetation

Cv = Cs x BAFv x 0.12 — 0.12=dw to ww conversion factor

Sediment to plant dw (BAFv)	
Cd	0.36
Cr	0.01
Cu	0.54
Pb	0.2
Zn	1
PCB	0.01

COC	Max	BAFv	Cv
	mg/kg	unitless	mg/kg, ww
Cd	10.4	0.36	0.45
Cr	118	0.01	0.14
Cu	902	0.54	58.45
Pb	258	0.2	6.19
Zn	841	1	100.92
PCB	0.5	0.01	0.001

COC	Cen Tend	BAFv	Cv
	mg/kg	unitless	mg/kg, ww
Cd	0.88	0.36	0.04
Cr	23.82	0.01	0.03
Cu	67.53	0.54	4.38
Pb	29.91	0.2	0.72
Zn	133	1	15.96
PCB	0.04	0.01	0.000

Invertebrates

Ci = Cs x BAFi x 0.29 — 0.29=dw to ww conversion factor

Sediment to invertebrate (BAFi)		
Cd	0.6	dw
Cr	0.1	dw
Cu	0.33	ww
Pb	0.63	ww
Zn	0.57	ww
PCB	0.53	ww

COC	Max	BAFi	Ci
	mg/kg	unitless	mg/kg, ww
Cd	10.4	0.6	1.81
Cr	118	0.1	3.42
Cu	902	0.33	297.66
Pb	258	0.63	162.54
Zn	841	0.57	479.37
PCB	0.5	0.53	0.27

COC	Cen Tend	BAFi	Ci
	mg/kg	unitless	mg/kg, ww
Cd	0.88	0.6	0.15
Cr	23.82	0.1	0.69
Cu	67.53	0.33	22.28
Pb	29.91	0.63	18.84
Zn	133	0.57	75.81
PCB	0.04	0.53	0.0212

		Muskrat					
			kg/kg-d				
		NIRv	0.31	wet			
		NIRs	0.0044	dry			
		AUF	1				
	Muskrat			Muskrat		Muskrat	
			Dose (max)	NOAEL	HQ	LOAEL	HQ
COC	Cs x NIRs	Cv x NIRv	mg/kg-d	mg/kg-d		mg/kg-d	
Max Conc							
Cd	0.05	0.14	0.19	1.0	0.2	10.0	0.0
Cr	0.52	0.04	0.56	3.3	0.2	13.1	0.0
Cu	3.97	18.12	22.09	11.7	1.9	15.4	1.4
Pb	1.14	1.92	3.05	42.0	0.1	126.0	0.0
Zn	3.70	31.29	34.99	160.0	0.2	320.0	0.1
PCB	0.00	0.00	0.00	0.1	0.0	0.7	0.0
COC		Dose (cen tend)		NOAEL	HQ	LOAEL	HQ
Cen Tend			mg/kg-d	mg/kg-d		mg/kg-d	
Cd	0.00	0.01	0.02	1.0	0.0	10.0	0.0
Cr	0.10	0.01	0.11	3.3	0.0	13.1	0.0
Cu	0.30	1.36	1.65	11.7	0.1	15.4	0.1
Pb	0.13	0.22	0.35	42.0	0.0	126.0	0.0
Zn	0.59	4.95	5.53	160.0	0.0	320.0	0.0
PCB	0.00	0.00	0.00	0.1	0.0	0.7	0.0

Waterfowl (Mallard) Exposure, Dose and HQ Calculations

All grids 0 - 0.5 ft		Cd	Cr	Cu	Pb	Zn	PCBs	
Mean		2.35	43.45	191.43	43.72	249.63	0.09	
Geomean		1.09	32.87	101.60	29.49	174.87	0.05	
Median		0.99	29.90	88.30	26.25	152.50	0.04	
Max		26.60	295.00	924.00	258.00	854.00	0.89	
Vegetation								
Cv = Cs x BAFv x 0.12			0.12=dw to ww conversion factor					
					COC	Max	BAFv	Cv
	Sediment to plant dw (BAFv)					mg/kg	unitless	mg/kg, ww
	Cd	0.36			Cd	26	0.36	1.12
	Cr	0.01			Cr	295	0.01	0.35
	Cu	0.54			Cu	924	0.54	59.88
	Pb	0.2			Pb	258	0.2	6.19
	Zn	1			Zn	854	1	102.48
	PCB	0.01			PCB	0.88	0.01	0.00
					COC	Cen Tend	BAFv	Cv
						mg/kg	unitless	mg/kg, ww
					Cd	1.1	0.36	0.05
					Cr	32.8	0.01	0.04
					Cu	102	0.54	6.61
					Pb	29.5	0.2	0.71
					Zn	175	1	21.00
					PCB	0.05	0.01	0.00
Invertebrates								
Ci = Cs x BAFi x 0.29			0.29=dw to ww conversion factor					
	Sediment to invertebrate (BAFi)				COC	Max	BAFi	Ci
	Cd	0.6	dw			mg/kg	unitless	mg/kg, ww
	Cr	0.1	dw		Cd	26	0.6	4.52
	Cu	0.33	ww		Cr	295	0.1	8.56
	Pb	0.63	ww		Cu	924	0.33	304.92
	Zn	0.57	ww		Pb	258	0.63	162.54
	PCB	0.53	ww		Zn	854	0.57	486.78
					PCB	0.88	0.53	0.47
					COC	Cen Tend	BAFi	Ci
						mg/kg	unitless	mg/kg, ww
					Cd	1.1	0.6	0.19
					Cr	32.8	0.1	0.95
					Cu	102	0.33	33.66
					Pb	29.5	0.63	18.59
					Zn	175	0.57	99.75
					PCB	0.05	0.53	0.03

Mallard						
	3 months of the year				9 months of the year	
	kg/kg-d		90% diet		kg/kg-d	90% diet
NIRv3	0.155	wet	60i30v	NIRv9	0.309	30i60v
NIRs	0.007	dry				
NIRi3	0.138	wet	60i30v	NIRi9	0.069	30i60v
AUF	0.17					

	Mallard						Mallard		Mallard	
						Dose (max)	NOAEL	HQ	LOAEL	HQ
COC	Cs x NIRs	Cv x NIRv3	Ci x NIRi3	Cv x NIRv9	Ci x NIRi9	mg/kg-d	mg/kg-d		mg/kg-d	
Max Conc										
Cd	0.182	0.044	0.156	0.260	0.234	0.149	1.45	0.1	20	0.0
Cr	2.065	0.014	0.295	0.082	0.443	0.493	1	0.5	5	0.1
Cu	6.468	2.320	10.520	13.876	15.780	8.324	47	0.2	61.7	0.1
Pb	1.806	0.240	5.608	1.435	8.411	2.975	1.13	2.6	11.3	0.3
Zn	5.978	3.971	16.794	23.750	25.191	12.866	14.5	0.9	130.9	0.1
PCB	0.006	0.000	0.016	0.000	0.024	0.008	0.18	0.0	0.88	0.0
COC						Dose (cen ten	NOAEL	HQ	LOAEL	HQ
Cen Tend						mg/kg-d	mg/kg-d		mg/kg-d	
Cd	0.008	0.002	0.007	0.011	0.010	0.006	1.45	0.0	20	0.0
Cr	0.230	0.002	0.033	0.009	0.049	0.055	1	0.1	5	0.0
Cu	0.714	0.256	1.161	1.532	1.742	0.919	47	0.0	61.7	0.0
Pb	0.207	0.027	0.641	0.164	0.962	0.340	1.13	0.3	11.3	0.0
Zn	1.225	0.814	3.441	4.867	5.162	2.637	14.5	0.2	130.9	0.0
PCB	0.000	0.000	0.001	0.000	0.020	0.004	0.18	0.0	0.88	0.0