U.S. Army Center for Health Promotion
and Preventive Medicine

TIER II ECOLOGICAL RISK ASSESSMENT REPORT

Twin Cities Army Ammunition Plant
Arden Hills, Minnesota

New Brighton/Arden Hills Superfund Site

Final Report
December 2004
Volume I: Main Report and Appendices A-J

Prepared by
Environmental Health Risk Assessment Program
Surface Water and Wastewater Program
Health Effects Research Program
Quality Systems Office
Laboratory Consultants Office

Prepared for
Twin Cities Army Ammunition Plant
U.S. Army Operations Support Command

USACHPPM Project No. 39-EJ-7067-00

Readiness Thru Health

Minnesota Pollution Control Agency

November 24, 2004

Mr. Michael R. Fix
Department Of Army
Twin Cities Army Ammunition Plant
4700 Highway 10, Suite A
Arden Hills, MN 55112

RE: Consistency Test for the Tier II Ecological Risk Assessment Report, Twin Cities Army Ammunition Plant, MN (December 2003) Volumes I and II

Dear Mr. Fix:

Staff at the Minnesota Pollution Control Agency (MPCA) and U.S. Environmental Protection Agency (U.S. EPA) have completed review of the New Red-Lined Changes to the Tier II Ecological Risk Assessment Report, Twin Cities Army Ammunition Plant (TCAAP), Red-Lined Final Draft Report, Volumes I and II, Based on MPCA Comments and dated October 14, 2004. MPCA staff and U.S. EPA review and approval of this submittal follows upon review and comments for a number of previous draft submittals of this document.

Based upon our review, MPCA staff and U.S. EPA have determined that, in accordance with Chapter XIV of the TCAAP Federal Facility Agreement, the Tier II Ecological Risk Assessment Report passes the Consistency Test.

If you have any questions, please contact Dagmar Romano at (651) 296-7776, or Tom Barounis at (312) 353-5577.

Sincerely,

Dagmar Romano

Dagmar Romano
Project Manager
Superfund and Emergency Response Section
Remediation Division

Dagmar Romano
for
Tom Barounis
Remedial Project Manager
U.S. EPA

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DEPARTMENT OF DEFENSE
UNITED STATES OF AMERICA

DEPARTMENT OF THE ARMY
US ARMY CENTER FOR HEALTH PROMOTION AND PREVENTIVE MEDICINE
5158 BLACKHAWK ROAD
ABERDEEN PROVING GROUND MD 21010-5403

REPLY TO
ATTENTION OF

MCHB-TS-REH

3 DEC 2004

MEMORANDUM FOR BRAC Office, Twin Cities Army Ammunition Plant (DAIM-BO-A-TW/Mr. Michael R. Fix), 4700 Highway 10 - Suite A, Arden Hills, MN 55112-3928

SUBJECT: Transmittal of the Tier II Ecological Risk Assessment Report, Twin Cities Army Ammunition Plant, MN (December 2004) Final Report, Volumes I and II

1. Two copies of the subject report are enclosed for your records. Also enclosed with each report are three CD-ROMs containing: the final report and the work plan (CD 1); files of the statistical analysis performed on the 1999-2000 sampling data (CD 2); and files regarding the investigative data (CD 3).

2. Our point of contact is Ms. Lia Gaizick. She can be reached at DSN 584-8148, commercial (410) 436-8148 or at lia.gaizick@us.army.mil. Please feel free to contact her if you have any questions or comments regarding this project.

FOR THE COMMANDER:

Jack M. Heller

Encl

JACK M. HELLER
Director
Health Risk Management

CF (w/encl):
USEPA Region V (Mr. Tom Barounis/SRF-5J) (3 cy)
MPCA (Ms. Dagmar Romano) (2 cy)
US Fish and Wildlife Service (Mr. Dave Warburton)
MN Department of Natural Resources (Mr. Tom Balcom)
✓ Tecumseh/Wenck Installation Support Services (Mr. Keith Benker)
Alliant Techsystems (Mr. Jon Bode)
USAEC (SFIM-AEC-CD/Mr. Pete Rissell)
Restoration Advisory Board (Mr. Matt Swanson)
Army National Guard (Mr. Dave Hamernick)

Tier II Ecological Risk Assessment Report
Twin Cities Army Ammunition Plant
Arden Hills, Minnesota

December 2004

Executive Summary

1. PURPOSE

The Tier II Ecological Risk Assessment (ERA) was designed to follow up on the recommended actions reported in the Tier I ERA in order to conclude the risk assessment process for five aquatic sites so that site managers can make decisions as to whether to consider the need for remedial action at each study site. The Tier I ERA is considered to be a screening-level assessment that "screened" for the potential for adverse ecological effects due to chemical exposure and identified missing information and data gaps. The primary data used in that assessment was collected as part of the Operable Unit 2 Feasibility Study. This report also provides the results of more recent field investigations and a refined risk analysis.

The following sections summarize the results for each site. Additionally, a table is provided at the end of the executive summary that summarizes the findings for each site.

2. Marsden Lake (North)

Marsden Lake (North) refers to that portion of Marsden Lake immediately surrounding Patrol Road. The north area of the lake was defined as a distinct area to focus the assessment in terms of characterizing any potential risk due to contaminants that may have been introduced as a result of former activities at the Grenade Range. It was originally suspected that the Grenade Range could be a specific source of COPCs in the northern portion of the lake. Contaminants in Grenade Range soils may have been transported via runoff into the lake, resulting in metals contamination in the surface water and sediment. Known soil contaminants at the Grenade Range were remediated to an industrial land use standard in 1999.

2.1 Investigations. Marsden Lake sites were evaluated in the 1997 Tier I screening-level ecological risk assessment, but the study areas were not sufficiently characterized for an adequate Tier I screening evaluation. This limitation was addressed within the Tier II risk assessment by performing additional site investigations, including water and sediment sampling and chemical analyses. The data obtained from these investigations were evaluated using the same Tier I screening process that was finalized in 1997, in order to complete that phase of the risk assessment for these sites and identify COPCs that were carried through the Tier II risk evaluation. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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 amphibians.

The following assessment endpoints were evaluated in the risk assessment:

- 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 & productivity of aquatic mammals, waterfowl, and wading birds

2.2 Conclusion. Surface water and sediment chemicals of concern (COCs) were identified that were found at concentrations greater than background and toxicological benchmark concentrations. Surface water COCs were aluminum, barium, lead, manganese, mercury, and zinc. Sediment COCs were antimony, arsenic, vanadium, and semivolatile organic compounds (SVOCs).

In summary, adverse effects were found to be either not apparent or possible, albeit unlikely for four of the five endpoints assessed. The risk assessment identified the potential for adverse effects due to aluminum, barium, and manganese for fish, aquatic invertebrates, and algae. However, important uncertainties remain related to the chemical speciation of aluminum and the validity of the toxicological benchmarks used for each of these COCs. Furthermore, it is not clear that these COCs are related to TCAAP-specific activities. These uncertainties question the validity of the identified risks for this endpoint.

2.3 Recommendations. The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

3. Marsden Lake (South)

Marsden Lake (South) refers to the portion of Marsden Lake directly adjacent to the Outdoor Firing Range. The south area was defined as a distinct area to focus the assessment in terms of characterizing any potential risks due to contaminants that could have come from activities at the Outdoor Firing Range. It was suspected that the Outdoor Firing Range could be a specific source of COPCs in Marsden Lake. Contaminants in the Outdoor Firing Range soils may have been transported via runoff into the lake, resulting in metals contamination in the surface water and sediment in the area defined as Marsden Lake (South). Known soil contaminants at the Outdoor Firing Range were remediated to an industrial land use standard in 1999.

3.1 Investigations. Marsden Lake sites were evaluated in the 1997 Tier I screening-level ecological risk assessment, but the study areas were not sufficiently characterized for an adequate Tier I screening evaluation. This limitation was addressed within the Tier II risk assessment by performing additional site investigations, including water and sediment sampling and chemical analyses. The data obtained from these investigations were evaluated using the same Tier I screening process that was finalized in 1997, in order to complete that phase of the risk assessment for these sites and identify COPCs that were carried through the Tier II risk evaluation. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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 amphibians.

The following assessment endpoints were evaluated in the risk assessment:

- 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 & productivity of aquatic mammals, waterfowl, and wading birds

3.2 Conclusions. Surface water and sediment chemicals of concern (COCs) were identified that were found at concentrations greater than background and toxicological benchmark concentrations. The surface water COC was manganese. Sediment COCs were aluminum, cadmium, lead, vanadium, and semivolatile organic compounds (SVOCs).

In summary, adverse effects were found to be either not apparent or possible, albeit unlikely in four of the five endpoints assessed. The risk assessment identified the potential for adverse effects in benthic organisms due to lead. However, sediment lead concentrations at the site are not statistically greater than nearby Snail Lake, which points to the uncertainty that the lead is related to TCAAP-specific activities. Lead was retained as a COC because it occurs at concentrations that appear greater than relatively pristine Minnesota Valley National Wildlife Refuge Lakes. Lead may be elevated regionally.

3.3 Recommendations. The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

4. Pond G

Pond G is a small pond (20 x 80 ft) that receives drainage from an area in the vicinity of Site G. The specific source(s) of the COCs in Pond G is suspected to be Site G (uncontrolled landfill), Site F (former burning area), or both. The source areas are no longer in operation; therefore the contamination is historical. The secondary sources for these COCs would be the sediment, and surface runoff from contaminated upland soil areas. Groundwater seeps have also been suggested as a possible source.

4.1 Investigations. Pond G was evaluated in the 1997 Tier I screening-level ecological risk assessment, but the study area was not sufficiently characterized for an adequate Tier I screening evaluation (USACHPPM 1997 and 1999). This limitation was addressed within the Tier II risk assessment by performing additional water and sediment sampling and chemical analyses (USACHPPM 1999). The data obtained from these investigations were evaluated using the same Tier I screening process that was finalized (USACHPPM 1997), in order to complete that phase of the risk assessment for these sites and identify COPCs that were carried through the Tier II risk evaluation.

The following assessment endpoint was evaluated in the risk assessment: water and sediment quality for wildlife using the pond.

4.2 Conclusions. Surface water and sediment chemicals of concern (COCs) were identified that were found at concentrations greater than toxicological benchmark concentrations. Since a comparable reference location for Pond G is not available and, therefore, was not used in the selection of COCs (USACHPPM 1999), it is difficult to determine whether many of the inorganics present in the Pond are actually contaminants. Thirteen inorganics were identified as surface water COCs. Six inorganics, as well as Aroclor 1248, SVOCs, and OC pesticides were identified as sediment COCs.

In summary, the risk assessment indicated that adverse effects in birds and mammals exposed to Pond G are not apparent.

4.3 Recommendations. The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

5. Rice Creek

Rice Creek is a tributary of the Mississippi River, which flows through the northwest corner of the installation. The TCAAP accounts for approximately 2 percent of Rice Creek's 474-square-kilometer basin and is near the downstream end of the creek. The State of Minnesota classifies Rice Creek as a "Class 1C, 2BD, 3B water".

Rice Creek potentially receives groundwater from the perched unit 1 aquifer under site K, and was once thought to potentially receive volatile organic compounds (VOCs) from this source (with discharge between OU2-FS sampling locations RCK04SE and RCK05SE). Site K consists primarily of building 103 and surrounding property and occupies approximately 21 acres. The site has been used mainly for munitions manufacturing and assembly operations.

5.1 Investigations. In order to address deficiencies in previous surface water sampling and analysis data used in the 1997 Tier I risk assessment, additional surface water sampling was performed at the creek between September 1999 and June 2000.

The following assessment endpoints were evaluated in the risk assessment:

- Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species

5.2 Conclusions. Surface water chemicals of concern (COCs) were not identified because on-site concentrations of each COPC were either found at levels less than those found upstream and/or were less than toxicological benchmark concentrations. Therefore, risks were not identified at this site.

5.3 Recommendations. The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

6. Sunfish Lake

Sunfish Lake is a shallow 5.7-hectare lake in the southeast corner of the installation. It was suspected that Site H-1, which contains an old unpermitted landfill and also included metal debris scattered on the bottom of a portion of the Sunfish Lake sediments along the northcentral shore, could be the specific source of COPCs detected in the lake during the OU-2 Feasability Study. Also, contaminated soils from the site may have been transported via runoff into the lake, and metal contaminated leachate from the landfill may have migrated into the lake, causing metals contamination in the sediment and surface water.

Site H-1 is no longer used and the upland soil areas of the site were remediated in 1999-2001 because they contained concentrations of metals (i.e., arsenic, antimony, copper, lead, and manganese) that posed excessive risks to human health (Stone & Webster 2002).

6.1 Investigations. The Tier I risk assessment (USACHPPM 1997) identified potential sediment-related risks, which required further investigtion. Therefore, sediment toxicity tests were conducted usign field-collected sediments. In order to address deficiencies in previous surface water sampling and analysis data used in the 1997 Tier I risk assessment, additional surface water sampling was performed at the creek between September 1999 and June 2000. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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 amphibians.

The following assessment endpoints were evaluated in the risk assessment:

- 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 & productivity of aquatic mammals and wading birds

6.2 Conclusions. Surface water and sediment chemicals of concern (COCs) were identified that were found at concentrations greater than background and toxicological benchmark concentrations. Surface water COCs were cadmium, mercury, and zinc. Sediment COCs were aluminum, chromium, lead, vanadium, and zinc.

In summary, adverse effects were found to be either not apparent or possible, albeit unlikely in four of the five endpoints assessed. The risk assessment identified that elevated sediment concentrations of chromium, lead, and zinc pose a potential risk for benthic organisms, but other data suggest that there may be sufficient sulfide and organic material to render lead and zinc biologically unavailable. Toxicity test results are consistent with a conclusion of little to no adverse effects, but limitations in the tests have been identified.

6.3 Recommendations. The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

7. Round Lake (Current Scenario)

Round Lake is approximately 125 acres and is located just southwest of the TCAAP. The lake is under the management of the U.S. Fish and Wildlife Service as a unit of the Minnesota Valley National Wildlife Refuge. 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 tenant of the building noticed that many of the floor drains were still connected to the storm sewer and has since remedied the situation.

7.1 Investigations. The Tier I risk assessment (USACHPPM 1997) identified potential sediment-related risks, which required further investigtion. Therefore, sediment toxicity tests were conducted using 1999 field-collected sediments. In addition, the sediment-metal bioavailability study and the toxicity tests from the southern portion of Round Lake (USACHPPM 1998a,b) were incorporated into the Tier II risk characterization. In order to address deficiencies in previous surface water sampling and analysis data used in the 1997 Tier I risk assessment, additional surface water sampling was performed at the lake between September 1999 and June 2000. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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 amphibians.

The following assessment endpoints were evaluated in the risk assessment:

- 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 & productivity of aquatic mammals, waterfowl, and wading birds

7.2 Conclusions. Surface water and sediment chemicals of concern (COCs) were identified that were found at concentrations greater than background and toxicological benchmark concentrations. Surface water COCs were barium, cadmium, and zinc. Sediment COCs were cadmium, chromium, copper, lead, silver, vanadium, zinc, and PCBs. Though PCBs were not detected in the sediment during the last sampling event in 1992, historical levels were high and PCBs have been detected in fish inhabiting the lake.

In summary, the risk assessment 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 related to these COCs remain. Adverse effects were not apparent for amphibians and are possible, albeit unlikely for waterfowl and wading birds. Additionally, there may be a potential for adverse toxicological effects due to exposure to PCBs in mink, wading birds, and belted kingfishers, but the possibility is low for both the littoral and profundal areas of Round Lake because the exposure estimates are not greater than toxicity benchmarks.

7.3 Recommendations. The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

8. Round Lake (Future Scenario)

The current USFWS management plan for Round Lake (USFWS 1982 and 1998) identifies two potential strategies for improving habitat quality to attract higher numbers of migrating wildlife, especially waterfowl and shorebirds. In summary, the preferred course calls for the implementation of actions to increase the ratio of emergent vegetation to open water from the existing 10:90 to a more productive ratio of about 50:50 by initiating a complete draw down of the lake. The USFWS refers to this plan as their *optimum wildlife alternative*. This completed draw down will de-water the lake, which will consolidate the sediments and allow for emergent plants to germinate and take root. Water levels would then be brought up with the emergent growth to normal pool levels. This pool level would be maintained for several years until emergent plants become sparse again. Additional drawdowns would not de-water the lake to the same extent.

8.1 Investigation. During the deliberations over the scope and design of the Tier II risk assessment and associated field activities, it became evident that additional clarity was needed regarding two issues: (1) the potential USFWS management alternatives for Round Lake needed to be addressed and (2) ecological risks associated with each of those alternatives needed better definition. The risk assessment describes chemical-risks associated with potential wildlife management actions at Round Lake based on expected changes in the sediment physicochemical characteristics.

The future physiochemical characteristics of the sediment in Round Lake are important factors that determine if there will be an increase in ecological risk from exposure to sediment COCs as a result of the proposed management practice. In the first year of the management plan, there will be a complete drawdown, exposing a large majority of sediment in the northern section of the lake to the ambient air. Even though the current status of the lake indicates that metals may be bound to some extent and unavailable to biota, drawdown of the lake could alter this situation. One of the results of drawdown is an acceleration of the oxidation (change in the oxidation-reduction state) and breakdown of the muck that accumulates on the bottom of the lake as the sediment becomes exposed to the ambient air. There are also environmental factors that may have an effect on sediment physiochemistry as a result of drawdowns. These factors (i.e., sediment drying, bioturbation, pH, and total organic carbon) may affect the bioavailability and spatiotemporal distribution of sediment COCs. The environmental factors affecting sediment drying and oxidation properties can include the ambient air velocity, amount of rainfall during drawdown, sediment organic carbon content, and algal and plant infestations.

There is a potential that surface sediments could become oxidized due to wind mixing in shallow water areas and wind movement over exposed sediments. This would render contaminants more bioavailable and would release surface sediment COCs to the surface water. However, based on the current information it seems unlikely that the management plan would inadvertently release deep sediment constituents to the surface. The high organic carbon content of the sediment may prevent drying and limit oxidation and 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.

The following assessment endpoints were evaluated in the risk assessment:

- 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 & productivity of aquatic mammals, waterfowl, and wading birds

8.2 Conclusions. The surface water COCs selected were the same as those selected under the Round Lake (current) scenario. However, the sediment COCs differ slightly due to the potential for mobilization and release of COCs from deep sediment. The sediment COCs selected were cadmium, chromium, copper, lead, manganese, nickel, silver, vanadium, zinc, and PCBs.

In summary, the risk assessment qualitatively compared the results of the current scenario risk estimates to what may occur if the USFWS management plan is implemented. Based on the information available, it is expected that if the management plan is put into effect, then all ecological entities will possibly experience increased exposure to COCs. Adverse toxic effects may occur in all assessment receptors because bioavailable concentrations of COCs would increase in each media (i.e., water, sediment, plants) therefore increasing receptor exposure. Due to the speculative nature of what may occur upon drawdown, no information could be provided on the magnitude of the potential increase in exposure. Therefore, it is impossible to predict the magnitude of the potential toxicological effects the drawdown would have on biota at Round Lake. Based on results for the current scenario, the primary concern under the future scenario would be adverse changes in the survival, growth, and reproduction of benthic organisms.

8.3 Recommendation. The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

Tier II Ecological Risk Assessment Executive Summary Table

Site and Management Objective	Risk Attributes†	Fish, Invertebrates, Algae	Benthic Organisms	Amphibians	Waterfowl Wading Birds	Mammals	Risk Summary
Rice Creek Prevent unacceptable toxic effects (in organisms living) in the creek	Presence: Magnitude: Uncertainty: 1° COCs:	Not Present -- Very Low --	Not Applicable -- -- --	Not Applicable -- -- --	Not Applicable -- -- --	Not Applicable -- -- --	No COCs were identified, therefore adverse effects were not identified.
Pond G Protect the health of wildlife using the pond for drinking water	Presence: Magnitude: Uncertainty: 1° COCs:	Not Applicable -- -- --	Not Applicable -- -- --	Not Applicable -- -- --	Not Apparent -- Low --	Not Apparent -- Low --	Adverse effects in birds and mammals exposed to Pond G are not apparent.
Marsden Lake (South) Prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake	Presence: Magnitude: Uncertainty: 1° COCs:	Possible Low Low Mn	Potential Moderate Moderate Pb	Not Apparent -- Moderate --	Possible Low Low Al, Pb (waterfowl)	Possible Low Moderate Al	Adverse effects were found to be either not apparent or possible in four of the five endpoints assessed. Potential adverse effects in benthic organisms due to lead were identified. However, sediment lead concentrations at the site are not statistically greater than nearby Snail Lake, which points to the uncertainty that the lead is related to TCAAP-specific activities. Lead was retained as a COC because it occurs at concentrations that appear greater than relatively pristine MVNWR lakes. Lead may be elevated regionally.
Marsden Lake (North) Prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake	Presence: Magnitude: Uncertainty: 1° COCs:	Potential Moderate High Al, Ba, Mn	Possible Low Low As, Sb	Not Apparent -- Moderate --	Possible Low Low Al, Pb (waterfowl)	Possible Low Moderate Al, Sb	Adverse effects were found to be either not apparent or possible for four of the five endpoints assessed. The potential for adverse effects due to Al, Ba, and Mn were identified for fish, aquatic invertebrates, and algae. However, important uncertainties remain related to the chemical speciation of Al and the validity of the toxicological benchmarks used for each of these COCs. Furthermore, it is not clear that these COCs are related to TCAAP-specific activities. These uncertainties question the validity of identified risks for this endpoint.
Sunfish Lake Prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake	Presence: Magnitude: Uncertainty: 1° COCs	Possible Low Low --	Potential Low Moderate Cr, Pb, Zn	Not Apparent -- Moderate --	Possible Low Low --	Possible Low Moderate Al, Cr	Adverse effects were found to be either not apparent or possible in four of the five endpoints assessed. Elevated sediment concentrations of Cr, Pb, and Zn pose a potential risk for benthic organisms, but other data suggest that there may be sufficient sulfide and organic material to render Pb and Zn biologically unavailable. Tox test results are consistent with a conclusion of little to no adverse effects, but limitations in the tests have been identified.
Round Lake Prevent unacceptable toxic effects to benthos, fish, wildlife, and effects that reduce habitat quality in the lake	**Current Scenario** Presence: Magnitude: Uncertainty: 1° COCs	Potential Low - Moderate High Ba	Confirmed High Moderate Cr, Cu, Ag, Zn	Not Apparent -- Moderate --	Possible Low Low PCBs	Potential Low Moderate Ag, Ba^*, Cr^*, Cu^*	Adverse effects were confirmed in benthic organisms in sediment toxicity tests. The potential for adverse effects due to Ba (for fish, aquatic invertebrates, and algae) and Ag (for mammals) were also identified, but important uncertainties related to these COCs remain. Adverse effects were not apparent for amphibians and are unlikely for waterfowl and wading birds. There may be a potential for adverse toxicological effects due to exposure to PCBs in mink, wading birds, and belted kingfishers for both the littoral and profundal areas of Round Lake but the potential is low because the exposure estimates are not greater than toxicity benchmarks. *Only NOAEL HQs were exceeded.
	Future Scenario Presence: Magnitude: Uncertainty: 1° COCs	Possible Increase Unknown High Inorganics and PCBs	Possible Increase High High Metals	Possible Increase Unknown High Inorganics and PCBs	Possible Increase Unknown High Inorganics and PCBs	Possible Increase Unknown High Inorganics and PCBs	Based on information available, if the USFWS Management Plan is put into effect, then all ecological entities will possibly experience increased exposure to COCs. Uncertainty is very high due to the speculative nature of what may occur in the future.

1° COCs: Primary Chemicals of Concern are those contributing to the potential or detected presence of adverse effects. Other COCs may exist but are not emphasized in this table because they are not risk drivers.

† Risk presence is scaled using the following categories in order of increasing confidence of the occurrence of an adverse effect: Not apparent, Possible, Possible Increase, and Potential (See Section 8.2 for an explanation of these categories)

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Appendices

Acronyms and Abbreviations

ANOVA	Analysis of Variance
AOCs	Areas of Concern
AQUIRE	Aquatic Toxicity Information Retrieval database
ARARs	Applicable or Relevant and Appropriate Requirements
ATP	Adenosine Triphosphate
ATSDR	Agency for Toxic Substances and Disease Registry
AUF	Area Use Factors
AVS	Acid Volatile Sulfides
AWQC	Ambient Water Quality Criteria
BAF	Bioaccumulation Factors
BEHP	Bis (2-ethylhexyl)phthalate
CCME	Canadian Council of Ministers of the Environment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COCs	Chemicals of Concern
COPCs	Chemicals of Potential Concern
DDT	p,p'-dichlorodiphenyl-trichloroethane
DLS	Directorate of Laboratory Sciences
DO	Dissolved Oxygen
DQOs	Data Quality Objectives
ECOTOX	U.S. EPA Ecotoxicology database
ERA	Ecological Risk Assessment
ERAGS	Ecological Risk Assessment Guidance for Superfund
FETAX	Frog Embryo Teratogenesis Assay - *Xenopus*
FFA	Federal Facilities Agreement
FS	Feasibility Study
GLWQI	Great Lakes Water Quality Initiative
GPS	Global Positioning System
HQ	Hazard Quotient
LOAEL	Lowest-Observed-Adverse-Effect level
MDNR	Minnesota Department of Natural Resources
MERLA	Minnesota Environmental Response and Liability Act
MPCA	Minnesota Pollution Control Agency
msl	Mean sea level
MWQS	Minnesota Water Quality Standards
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NOAEL	No-Observed-Adverse-Effect level
OC Pesticides	Organochlorine pesticides
ORNL	Oak Ridge National Laboratory
OU-2	Operable Unit-2
PAHs	Polycyclic Aromatic Hydrocarbons
PCBs	Polychlorinated Biphenyls
PNAs	Polynuclear Aromatics
QA	Quality Assurance
QAPP	Quality Assurance Project Plan
QC	Quality Control
RAB	Restoration Advisory Board
RBAL	Risk Based Action Level
RCRA	Resource Conservation and Recovery Act
ROD	Record of Decision
SEM	Simultaneously Extractable Metals
SQL	Sample Quantitation Limit
SVE	Soil Vapor Extraction
SVOCs	Semivolatile Organic Compounds
TCAAP	Twin Cities Army Ammunition Plant

TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TCE	Trichloroethene
TICs	Tentatively Identified Compounds
TKN	Total Kjeldahl Nitrogen
TOC	Total Organic Carbon
TRVs	Toxicity Reference Values
TSS	Total Suspended Solids
UCL	Upper Confidence Limit
USACE	United States Army Corps of Engineers
USACHPPM	United States Army Center for Health Promotion and Preventive Medicine
USAEHA	United States Army Environmental Hygiene Agency
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UST	Underground Storage Tank
VES	Visual Encounter Survey
VOC	Volatile Organic Compounds
WHO	World Health Organization

U.S. ARMY CENTER FOR HEALTH PROMOTION AND PREVENTIVE MEDICINE
Aberdeen Proving Ground, Maryland

Tier II Ecological Risk Assessment Report

Twin Cities Army Ammunition Plant
Arden Hills, Minnesota

December 2004

1. INTRODUCTION

The ecological risk assessment (ERA) investigations at the Twin Cities Army Ammunition Plant (TCAAP) are proceeding under the authority of the *Comprehensive Environmental Response, Compensation, and Liability Act* (CERCLA) as amended by the *Superfund Amendments Reauthorization Act of 1986* (SARA), the *Minnesota Environmental Response and Liability Act* (MERLA), and the EPA's *National Oil and Hazardous Substances Pollution Contingency Plan* (NCP). The ERA project is one part of the remedial investigation/feasibility study process at TCAAP, and as such it is designed pursuant to the Federal Facilities Agreement (FFA) established in 1987 between the United States Environmental Protection Agency (USEPA), the Minnesota Pollution Control Agency (MPCA), and the United States Army (Army). The United States Fish and Wildlife Service (USFWS) manages one of the sites evaluated (Round Lake) as a management unit of their Minnesota Valley National Wildlife Refuge.

A Tier I ERA of Aquatic Systems was completed in October 1997 by the U.S. Army Center for Health Promotion and Preventive Medicine (USACHPPM 1997). This Tier I ERA used available sediment and surface water data collected in 1993 and 1994 by Montgomery Watson, Inc. who conducted the Operable Unit 2 (OU-2) Feasibility Study (FS), limited sediment biological evaluations, and recent (1994-1995) annual monitoring report surface water data (Montgomery Watson 1994, Federal Cartridge Company 1995, and USACE 1996a). The Tier I ERA is considered to be a screening-level assessment that "screened" for the potential for adverse ecological effects due to chemical exposure and identified missing information and data gaps.

The Tier II ERA was designed to follow up on the recommended actions reported in the Tier I ERA (USACHPPM 1999). It was designed to provide the level of detail needed to complete the risk assessment process for the aquatic sites; that is, so that the risk managers can make a decision as to the need for remedial action at each study site.

The ERA was conducted following recent guidance provided by four sources (shown below). The ERA was designed specifically for the TCAAP site and draws upon the strengths of each of these guidelines.

- Guidelines for Ecological Risk Assessment (USEPA 1998)
- Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments (USEPA 1997a)
- Tri-Service Procedural Guidelines for Ecological Risk Assessments (Tri-Services 1996)
- Risk Assessment Handbook Volume II: Environmental Evaluation (USACE 1996b)

Ecological risk assessments are designed and conducted to provide information to risk managers about the potential adverse effects of different management decisions. This ERA report documents the process used to evaluate the likelihood of adverse ecological effects as a result of exposure to chemicals released into aquatic environments during past operations of the facility. Adverse responses can range from subtle chronic effects in an individual organism to a loss of an ecosystem function. A risk does not exist unless the particular chemical(s) acts on an ecological component long enough and at a sufficient intensity to elicit an adverse effect. The USEPA guidance (1998) for ERA recommends a four-phased process consisting of (1) planning, (2) problem formulation, (3) analysis, and (4) risk characterization. Figure B1 illustrates how these phases are related. These phases are defined in the bullets below.

- Planning: a dialogue between risk managers, risk assessors, and interested parties before the risk assessment begins. This phase establishes management objectives, defines the decisions the risk assessment will support, and outlines agreements on the scope, complexity, and focus of the risk assessment.

- Problem Formulation: a process for generating and evaluating hypotheses about why ecological effects may occur. The problem is defined, the plan for analyzing exposure and effects and other data is outlined, and the methods for characterizing risks are described. The outcome of this phase is three products: assessment endpoints, conceptual models, and the analysis plan.

- Analysis: the technical evaluation of the data to reach conclusions about ecological exposure and the relationships between the stress and effects. The products of this phase are the exposure profile and the stress-response profile.

- Risk Characterization: where the results of the analysis are used to estimate and interpret risk to the assessment endpoints.

The remainder of this report presents the outcome of each of these phases. Appendix A presents tables and Appendix B presents figures.

2. PLANNING SUMMARY

This phase was completed as part of the development of the project Work Plan (USACHPPM 1999). This ERA needs to satisfy the requirements of the Superfund program with regard to the protection of the environment from the release or potential release of hazardous substances, pollutants, or contaminants. The planning process was designed to ensure that the ERA would meet the risk managers' needs. The site risk managers are: Mr. Marty McCleery (Twin Cities Army Ammunition Plant), Mr. Thomas Barounis (USEPA Region 5), and Ms. Dagmar Romano (MPCA). The primary interested parties are the U.S. Fish and Wildlife Service (USFWS) and the local Restoration Advisory Board (RAB). The risk managers approved the products of this planning phase (USACHPPM 1999).

There are six sites: Round Lake, Rice Creek, Sunfish Lake, Pond G, Area B wetlands, and Marsden Lake. These sites are known collectively as the *aquatic sites* (or the *surface water and sediment sites*). The aquatic sites are within the New Brighton/Arden Hills Superfund Site, also known as TCAAP, Ramsey County, Minnesota (Figure B1-2). The aquatic sites were previously considered as a part of the Operable Unit 2 (OU-2); however, in 1995 the FFA parties administratively separated these sites from the OU-2. Today, they stand alone as the aquatic sites within the TCAAP Superfund site. No decision documents have yet been produced for five of the six sites regarding the need for risk management actions to protect these aquatic systems. The wetlands at Area B are the exception; specifics are discussed below.

2.1 MANAGEMENT GOAL AND OBJECTIVES

Management goals for a risk assessment are established by risk managers and can be derived in a variety of ways. For the TCAAP site, the Superfund program primarily mandates the goal, which is to protect the environment from the release or potential release of hazardous substances, pollutants, or contaminants. This management goal is general and the risk assessors must develop ecological values relevant to this goal for TCAAP that can be measured or estimated and ensure that the managers agree with their interpretation. The regulatory interpretation of Superfund mandated goals is the National Contingency Plan (NCP) and *Ecological Risk Assessment Guidance for Superfund* (ERAGS; USEPA 1997a). These documents define environmental protection as the identification and mitigation of impacts such as bioaccumulation, death, reproductive impairment, growth impairment, and loss of critical habitat. In addition, numerous other federal and state laws and regulations concerning environmental protection can be designated under Superfund as "Applicable" or "Relevant and Appropriate" Requirements (ARARs).

The USEPA (1998) recommends that the risk management/risk assessment team interpret the goal into more specific objectives about what must occur in order for the management goal to be achieved. The first step in this interpretation is to identify valued ecological resources associated with each site that are susceptible to adverse effects and that can be measured or estimated. By their nature, such interpretations are relevant to the ecosystem being assessed and are done on a case-by-case basis. The following management objectives (which identify valued ecological resources) are statements that must be satisfied in order for the risk managers' goals to be met.

Management Objectives

Rice Creek. Prevent unacceptable toxic effects [in organisms living] in the creek.

Pond G. Protect the health of wildlife using the pond for drinking water.

Sunfish Lake. Prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake.

Marsden Lake. Prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake.

Area B Wetlands. Prevent unacceptable toxic effects in the wetlands and effects that reduce the health of wildlife associated with the wetlands.

Round Lake. Prevent unacceptable toxic effects to benthos, fish, wildlife, and effects that reduce habitat quality in the lake.

The expected future use of the lands associated with the aquatic environments on the TCAAP installation is not expected to change in the foreseeable future. The planned future use of TCAAP as a training area for Army Reserve organizations is not expected to result in uses that will be in direct conflict with the continued biological productivity and value associated with the aquatic systems on the installation.

The proposed USFWS management plan for Round Lake (USFWS 1982 and 1998) identifies two potential strategies for improving habitat quality to attract higher numbers of migrating wildlife, especially waterfowl and shorebirds. In summary, the preferred course calls for the implementation of actions to increase the ratio of emergent vegetation to open water from the existing 10:90 to a more productive ratio of about 50:50 by initiating a complete draw down of the lake. The USFWS refers to this plan as their *optimum wildlife alternative*. This completed draw down will de-water the lake, which will consolidate the sediments and allow for emergent plants to germinate and take root. Water levels would then be brought up with the emergent growth to normal pool levels. This pool level would be maintained for several years until emergent plants become sparse again. Additional drawdowns would not de-water the lake to the same extent. Such actions would be expected to provide a more attractive mix, of open water and cattail-brush, for breeding and migratory waterfowl. The improved habitat would also benefit numerous other marsh birds and animals including muskrats, mink, heron, and marsh wrens (USFWS 1982). The second potential action, though less desirable from the USFWS point of view, would be partial drawdowns that would increase the coverage of submerged vegetation; however, it would not allow as broad a consolidation of bottom sediments as the preferred actions would. Regardless of the active management strategy, the level of potential ecological risk associated with the characteristics and extent of the contaminants in the system may influence the future use of the site.

2.2 RISK MANAGEMENT DECISIONS

The risk assessment was shaped by the kind of decision it will support. For many risk assessments, including this ERA for the TCAAP sites, there may be a range of possible management decisions for managing risk. If different management options can be identified, then the risk assessment can be tailored to predict potential risk across the range of these options. The risk managers have established the management decisions shown below (USACHPPM 1999).

Management Decisions to be Supported by the Risk Assessment

Rice Creek. No action or define TCAAP source of contaminants.

Pond G. No action or consider a remedy.

Sunfish Lake. No action or consider a remedy.

Marsden Lake. No action or consider a remedy.

Area B Wetlands. No action; further ecological investigation is not warranted.

Round Lake. No action or consider a remedy (e.g., clean up/stabilize sediments).

During the development of the Work Plan it was decided that further ecological investigations at Area B were not warranted. Information regarding the site is presented in the planning and problem formulation phases of the assessment since at the time it was considered a study site.

3. PROBLEM FORMULATION

The problem formulation phase of this Tier II assessment was completed and approved in the Work Plan (USACHPPM 1999). This section is a restatement and refinement of the Problem Formulation that was presented in the Tier II Work Plan (USACHPPM 1999).

This first phase of the ERA is the formal process for generating and evaluating preliminary hypotheses about why ecological effects have occurred, or may occur, from human activities. During this phase, management goals developed during planning (Section 2) were evaluated to establish objectives for the risk assessment, the problem was defined, and the plan for analyzing data and characterizing risk was determined (USACHPPM 1999). The Problem Formulation for each of the study sites is provided in the following subsections. Unless otherwise stated, the information on the sources has been taken from the OU-2 Record of Decision (ROD) and TCAAP Fact Sheets (U.S. Army 1997 and TCAAP 1998).

In order to clarify the process used to develop the final list of chemicals of concern (COCs), a COC selection process was developed after the initial draft of the Tier II risk assessment (USACHPPM 2001). All prior references to COCs have been revised to be consistent with this characterization. Section 5.1 further describes this process.

3.1 MARSDEN LAKE

3.1.1 Available Information on Marsden Lake

Marsden Lake is a major feature of the TCAAP landscape. It is a large, permanently flooded palustrine emergent wetland marsh of about 220 acres (including open water) located along the eastern edge of the installation. Figure B1-3 is an orthophoto of the site.

3.1.1.1 Marsden Lake Contaminant Source(s)

Based upon historical information, there is no obvious source of chemical release, but the lake receives drainage from a large surface area of the installation. Several potential contaminant sources exist for this ecological system—thought to be associated with a potential for contaminant migration into Marsden Lake, whether by surface or groundwater flow, munitions impact, and/or past dumping practices. Marsden Lake is fed with water transient from Sunfish Lake; however the flow between the lakes is very low. Marsden Lake may receive water with elevated metal concentrations from Sunfish Lake; however, this scenario seems unlikely given the hydrology of these systems, the affinity of these metals to bind to organic matter, and the localized extent of contamination found in the Sunfish Lake sediments. Marsden Lake also receives runoff from Lexington Avenue, which runs along the entire length of marsh. This is a non-Army source and may complicate any investigations of military impacts on the marsh.

The Grenade Range (located in the central portion of the eastern shore of the lake) may have impacted Marsden Lake (see Figure D1). However based on the results of the Tier I assessment this could not be conclusively determined. The Range is approximately 19 acres and contains two proofhouses, three targets, three grenade catchers, a drainfield, a site of a former underground storage tank (UST), and several other concrete bunkers and woodsheds. Honeywell Defense Systems Division operated the range from the late 1960s into the mid-1970s to test rifle grenade fuses. Two phases of field investigations were conducted — Phase I in 1993 and Phase II in 1993/94. These investigations identified potential impacts of the soil and groundwater from activities conducted at the range. Based on the historical activities at the Grenade Range, the preliminary chemicals of potential concern (COPCs) were metals and include antimony, beryllium, cadmium, and lead.

The Outdoor Firing Range (located on the southwestern edge of the lake) is another potential source to Marsden Lake (see Figure E1). This range is approximately 150 acres and was built in 1943. It consists of the proofhouse (Bldg 145) and the earthen barricade 1200 yards away. Three bullet catchers at 600, 840, and 1900 yards were added in 1955. A grenade catcher and Bldg 170/173 were built by the mid-1960s. It was used primarily to test the outdoor accuracy and performance of small arms ammunition manufactured at TCAAP. Between 1961 and 1967, the site was used to test the arming sensitivity of grenades. The buildings of the range have been demolished. Three phases of field investigations were conducted at the Outdoor Firing Range — Phase I in 1994, Phase II in 1995, and Phase III in 1996. These investigations identified potential impacts of the soil and groundwater contamination from the past range activities. Based on the historical activities at the Outdoor Firing Range, the preliminary COPCs are metals and include antimony, cadmium, copper, and lead.

3.1.1.2 Marsden Lake Ecological System

The vegetation of Marsden Lake is dominated by cattail (*Typha sp.*), much of which are floating mats of emergent vegetation. Marsden Lake is used by a number of waterfowl species including Canada geese (*Branta canadensis*) and Mallard (*Anas platyrhynchos*). Common loons (*Gavia immer*) were heard but not seen during the field investigation. Numerous reptile and amphibian species use Marsden Lake including the snapping turtle (*Chelydra serpentina*), the painted turtle (*Chrysemis picta*), frogs (*Hyla versicolor*, *Pseudacris triseriata,* and *Rana pipiens*), and the toad (*Bufo americanus*). The most notable herptile using Marsden Lake is the Blandings turtle (*Emydoidea blandingi*), a species of concern since its population is declining nationwide. Numerous Blandings turtles were seen in Marsden Lake during a 1989 TCAAP survey (Linck 1989). A number of muskrat (*Ondatra zibethicus*) lodges are located in Marsden Lake, which probably helped curtail the encroachment of the cattail mat. Other mammals, such as the red fox and mink, are known to use this lake. Also, in the summer of 2001, a coyote was observed along Hamlin Avenue, southeast of the gravel pit. In previous years, the Minnesota DNR has released Trumpeter swans into Marsden Lake.

3.1.1.3 Previous Evidence of Potential or Real Ecological Effects Associated with Marsden Lake

The Tier I ERA (USACHPPM 1997) concluded that Marsden Lake had not been adequately characterized for chemical impact. In 1991, the *Ecological Assessment* (USAEHA 1991) reported the detection of thallium in the surface waters in 1987. Based upon the available data used in the Tier I ERA, the most significant potential for risks are due to the detection of pesticides and zinc in the sediments. Also, aluminum, barium, and zinc in surface water may have the potential to cause toxicity (see Appendix Q for the results of the original Tier I risk assessment). The assessment remained inconclusive with regard to potential risk from copper, mercury, and silver in the surface waters. This was due to method reporting limits that were too high for screening purposes.

3.1.2 Marsden Lake Risk Management Decisions to Date

The 1997 Tier I assessment concluded that sufficient information regarding the nature and extent of possible contamination was not available. Therefore, it was determined that further assessment implementing the recommendations of the Tier I risk assessment were warranted. Those recommendations were to better characterize the nature and extent of any contamination, if it exists, by performing additional sediment and surface water sampling at two areas near sites where possible releases were suspected to have potentially impacted the marsh. The two sites of concern were the Grenade Range (Marsden Lake North study area) and the Outdoor Firing Range (Marsden Lake South study area). See Figures D1 and E1 in the corresponding appendices.

3.1.3 Marsden Lake Assessment Endpoints

The Tier II Work Plan defined four assessment endpoints[1] for Marsden Lake.

- Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species
- Survival, growth, and reproduction of benthic organisms[2]
- Survival, development, and reproduction of amphibians
- Reproductive potential & productivity of aquatic mammals, waterfowl, and wading birds

No assessment endpoints evaluated during the Tier I risk assessment could be eliminated from further consideration because of the limitations of the available data for the site.

3.1.4 Marsden Lake Initial Conceptual Model

The conceptual model[3] for Marsden Lake consists of five risk hypotheses. Figure B1-4 illustrates the hypothesized links between the possible contaminant sources and potential adverse ecological effects.

1. Based upon the aquatic toxicity of COCs (suspected to be primarily metals), if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

2. Based upon the toxicity of COCs, if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment.

4. The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COCs in the surface waters and sediments.

5. The health of aquatic mammals, such as mink or muskrat, may be impaired if they are excessively exposed to COCs in the surface waters and sediments, or residues in food.

Marsden Lake Preliminary COPCs

- Undetermined at the start of the investigation
- Suspected to be heavy metals, if any

3.1.5 Marsden Lake Analysis Plan Summary

Specific measures were chosen for evaluation of the risk hypotheses. Measurements of chemical concentrations in the surface waters and surface sediments were made during the investigation at the two study areas: Marsden Lake North and South.

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[1] Assessment endpoints are explicit expressions of the actual environmental values that are to be protected. Unacceptable risks of adverse effects for those values, that drive decision-making, are those that may reduce populations of species or significantly disrupt community structure (USEPA 1998).

[2] Benthic organisms are those that inhabit the sediments.

[3] As defined by the USEPA (1998a), a conceptual model is a written description and visual representation of predicted responses by ecological entities (i.e., the selected assessment endpoints) to stressors. A conceptual model consists of two principal products: a risk hypothesis and a diagram that illustrates exposure and response relationships. In this case the model includes the preliminary chemicals of potential concern (preliminary COPCs).

1. The first hypothesis, which deals with the assessment endpoint of "survival, growth, and reproduction of fish, aquatic invertebrates and algal species," was evaluated with the following measures.

- Measures of exposure [4]
  Unfiltered water column concentrations of identified chemicals

- Measures of ecosystem characteristics [5]
  Water hardness, total suspended solids, total organic carbon, dissolved oxygen, and nitrogen & phosphorus levels

- Measures of effect [6]
  Minnesota Water Quality Standards or other effects-based benchmarks for fish, invertebrates, and algal species.

For the exposure measurement, unfiltered water column chemical concentrations are used from the site investigation. Characteristics of the water column were also measured in order to provide standard water chemistry information. These characteristics were used to calculate site-specific water quality benchmarks (i.e., water hardness and pH). Other characteristics were used in the risk characterization as lines of evidence in discussing the quality of the habitat (i.e., dissolved oxygen, phosphorus, and nitrogen). For the effects measurement, the Minnesota Water Quality Standards (MWQS) are used, when available for a chemical. When MWQS are not available, other sources were consulted to develop a toxicity benchmark.

2. The second hypothesis, which deals with the assessment endpoint of "survival, growth, and reproduction of benthic organisms," was evaluated with the following measures.

- Measures of exposure
  Concentrations of identified chemicals in surface sediments (1 ft depth)

- Measures of ecosystem and receptor characteristics
  Total organic carbon

- Measures of effect
  Effects-based benchmarks for benthic organisms

For the exposure and receptor measurements, investigation results were used. Total organic carbon and percent solids were also measured and were used in discussing the quality of the habitat and the relationship to bioavailability and toxicity of identified COPCs. Measures of effects consisted of an evaluation of effects-based benchmarks.

3. The third hypothesis, which deals with the assessment endpoint of "survival, development, and reproduction of amphibians," was evaluated with the following measures.

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[4] Measures of exposure are measures of how exposure may be occurring, including how a contaminant moves through the environment and how it may interact with the assessment endpoint.

[5] Measures of ecosystem and receptor characteristics are those that include ecological system characteristics that influence the behavior and location of assessment endpoints, the distribution of a contaminant, and life history characteristics of the assessment endpoint that may affect exposure or response.

[6] Measures of effect (often to as a measurement endpoints) are used to evaluate the response of the assessment endpoint when exposed to a chemical.

- Measures of exposure
  Unfiltered water column and surface sediment concentrations of chemicals

- Measures of ecosystem and receptor characteristics
  Species diversity of amphibians

- Measures of effect
  Modified FETAX assay, amphibian egg mass, hatching success, immunological assays, and mortality

No previous assessment investigated possible effects in the amphibian populations. For the exposure measurements, investigation results were used. Field surveys were conducted to measure ecosystem and receptor characteristics. Effects were assessed with bioassays, where field-collected water samples were used for controlled laboratory experiments on amphibian health.

4. The fourth and fifth hypotheses, which deal with the assessment endpoint of "reproductive potential and productivity of aquatic mammals, waterfowl, and wading birds," were evaluated with the following measures.

- Measures of exposure
  Exposure model using water and sediment concentrations of chemicals and estimates of residues in food

- Measures of ecosystem and receptor characteristics
  Life history habits and exposure factors for representative wildlife species

- Measures of effect
  Laboratory-derived chronic effect levels or other appropriate toxicity benchmarks

For the exposure measurements, current water and sediment data were used to develop exposure profiles for the selected receptors. This included bioaccumulation models, where appropriate. An evaluation of toxicity thresholds was performed to select effects measures. Where appropriate, distributions of exposure and effects are presented in comparison to site concentrations.

3.2 POND G

3.2.1 Available Information on Pond G

This is a small (20 x 80 ft) pond that receives drainage from Site G. It is immediately to the northeast of Site G and south of Site F. Figure B1-5 is an orthophoto of the site. Also see Figure F1 for location of Sites G and F.

3.2.1.1 Pond G Contaminant Source(s)

The contaminant source for the pond is not known. Potential source areas include Site G and Site F. Surface run-off from these sites and other contaminant pathways (i.e., groundwater seeps) have been suggested.

Site G is east of Snelling Avenue near the center of the installation. The site contains an irregularly shaped landfill area that was used for general refuse from World War II until late 1976. Some of its contents include material from demolished buildings, urethane foam, floor sweepings, scrap metal grindings, barrels, oil filters, ashes from scrap paper burning cages, and other miscellaneous refuse. Most of the site is now an artificial plateau (and covered with a clay cap) characterized by a fairly high and steep face with protruding debris. The site is approximately 4.6 acres, where VOCs extend from the surface down to the water table, a distance of 130-170 ft. Trichloroethene (TCE) is the only COC for the site. A Soil Vapor Extraction (SVE) System operated at the site from 1986 until 1998. Current investigations are being completed which suggest that the SVE system will not need to be turned on again. The remaining scenarios are to leave the existing cap in place, use no cap, or passive vent with the use of existing vents, focused on a small recalcitrant shallow soil layer. A study at Site G (Stone & Webster 1998), found that the only chemical of concern identified for Site G soils was TCE. It also reported that no SVOCs or pesticides were detected in groundwater at the site. In addition, there has been no evidence (Stone & Webster 2002) that the impacted soil is a source of groundwater contamination. It was also concluded that the clay cap over the Site G dump effectively reduces the dermal and oral exposure pathways to humans.

Site F is just north of Site G. It occupies close to 10 acres in area and is part of the old 326 Burning Area. During the 1950s, scrap primers, tracer butts, mercury crack cases, and cut incendiary bullets were burned using propellant powder or oil to initiate burning. Cyanide pots and containers were also disposed of at this area. The burying of mercury crack cases and the burning of primers containing lead styphnate continued in the 1960s. During the 1970s burning continued in burning kettles and in a burning cage. In the late 1970s and in 1980 the burning kettles remained and an incinerator was present at the site. In the early 1980s, Site F was to be used on an interim basis as a RCRA interim permitted site, however activities were stopped after 1983. The COCs for soil at the site were antimony, cadmium, chromium, copper, lead, mercury, nickel, and silver. The soil at Site F was remediated using a soil washing/soil leaching technology and off-site disposal. The last remedial activities occurred in 1997.

3.2.1.2 Pond G Ecological System

Just north and south of the fill areas of Site G are bands of trees; otherwise, the area is lightly vegetated with grasses and forbs. Pond G is used by wildlife in the vicinity. The installation's red fox population uses it as one source of drinking water and wading birds are also known to frequent the pond.

3.2.1.3 Previous Evidence of Potential or Real Ecological Effects Associated with Pond G

An earlier investigation, the *Ecological Assessment* (USAEHA 1991), concluded that copper in the surface water was present at a concentration acutely toxic to freshwater life. The Tier I risk assessment, using data collected during the OU-2 FS in 1992, concluded that aluminum, barium, manganese, and zinc in the surface water may be consistently toxic to most aquatic organisms (USACHPPM 1997). That assessment could not determine if there is risk from copper, mercury, and silver in the surface waters because the Operable Unit 2 FS data were based on analytical method reporting limits that were inadequate for screening purposes. The Tier I assessment also identified the sediment COPCs, copper, lead, zinc, polychlorinated biphenyls (i.e., the Aroclor 1254 mixture), and pesticides (i.e., p,p-dichlorodiphenyltrichloroethane) and it's metabolites, as being likely to cause toxic conditions for sediment-dwelling organisms.

3.2.2 Pond G Risk Management Decisions to Date

As only one sample was collected from the pond during the OU-2 FS, the Tier I assessment recommended performing additional sediment and surface water sampling at several locations in order to better characterize the nature and extent of any contamination. This recommendation was implemented and the risk assessment was refined.

3.2.3 Pond G Assessment Endpoint

The Tier II Work Plan defined one assessment endpoint for Pond G.

- Water and sediment quality for wildlife using the pond

There are three *potential* assessment endpoints for Pond G. In fact, the 1997 Tier I risk assessment screened for effects for each of the three endpoints. The other two could be (1) survival, growth, and reproduction of aquatic invertebrates and algal species and (2) survival, growth, and reproduction of benthic organisms. During the development of the Tier II Work Plan, the ecological relevance of the first two endpoints was determined to be very low. The Tier I risk assessment concluded that overall ecological impact is unlikely, due to the pond's small size. Also, because the pond is sufficiently segregated geographically and hydrologically from other aquatic systems, any contamination in the pond will not migrate to other surface water systems. Nonetheless, wildlife tend to use the pond as a drinking water source and the Tier I risk assessment identified contaminants in the pond which can be toxic for wildlife consuming the water. Based on this argument, the management objective for Pond G was limited to protecting the health of wildlife using the pond for drinking water (Section 2.2).

3.2.4 Pond G Initial Conceptual Model

Based upon the revised assessment endpoint for the pond, a re-evaluation of the Tier I ERA conclusions was made. The Tier I risk assessment showed that water ingestion did not show a potential for harm to wildlife drinking from the pond. The mammal exposure model did, however, show that aluminum and vanadium might pose an unacceptable risk for those animals that may ingest sediment from the pond. The modeled risks in the Tier I assessment were significant for the following biotic exposure pathways: wading bird consumption of fish and benthic invertebrates, waterfowl ingestion of benthic invertebrates, and mammal consumption of pond vegetation. Each of these biotic pathways does not represent actual conditions at the site. No fish exist in the pond, benthic invertebrates are not likely to occur in abundance, if at all; and mammals visiting the pond are not likely to consume the vegetation in the pond. Therefore, all of these pathways become irrelevant to the site based on actual conditions and the revised assessment endpoint. Regardless of these Tier I results, the uncertainties were sufficient enough to warrant further study — the extent of contamination of the pond is unknown (only one sample was taken) and a potential exists for the water ingestion model to have provided a false negative based upon one sample.

Pond G Preliminary COPCs

Undetermined at the start of the investigation

The revised conceptual model for Pond G is as follows:

- Based upon the toxicity of chemicals present in the water or sediment of the pond, if they exist at sufficient concentrations then adverse toxic health effects may occur in foxes and wading birds that use the pond as a drinking water source.

Figure B1-6 illustrates the links between the hypothesized contaminant source(s) and potential adverse ecological effects.

3.2.5 Pond G Analysis Plan Summary

Three types of specific measures were chosen for evaluation. The measures are shown in the bullets below.

- Measures of exposure
  Water consumption exposure models using unfiltered water column concentrations of chemicals

- Measures of receptor characteristics
  Water consumption and life history habits of Red fox and Great Blue heron

- Measures of effect
  Laboratory derived chronic effects-levels or other appropriate toxicity benchmarks

Exposures were estimated for wildlife that consume the water from Pond G by using a water exposure model. Life history characteristics and exposure factors for the Red fox and Great blue heron were used to develop the model. The chemical concentrations detected during the Tier II field investigation were used to calculate the exposure point concentrations. The estimated exposures were compared to laboratory-derived chronic effects levels or other appropriate toxicity benchmarks.

3.3 RICE CREEK

3.3.1 Available Information on Rice Creek

Rice Creek is a tributary of the Mississippi River, which flows through the northwest corner of the installation. The TCAAP accounts for approximately 2 percent of Rice Creek's 474-square-kilometer basin and is near the downstream end of the creek. The State of Minnesota classifies Rice Creek as a "Class 1C, 2Bd, 3B water." Figure B1-7 is an orthophoto of Rice Creek.

3.3.1.1 Rice Creek Contaminant Source

Rice Creek potentially receives groundwater from the perched unit 1 aquifer under site K, and was once thought to potentially receive volatile organic compounds (VOCs) from this source (with discharge between OU-2 FS sampling locations RCK04SE and RCK05SE). Site K consists primarily of Bldg. 103 and surrounding property and occupies approximately 21 acres. The site has been used mainly for munitions manufacturing and assembly operations.

As a shallow groundwater contamination site at TCAAP, the Site K plume exhibits varying degrees of VOC contamination with trichloroethene (TCE) and the breakdown products 1,2-dichloroethene and trans 1,2-dichloroethene being the primary contaminants of concern.. Most of the plume is found under Bldg. 103; however the source for the Site K TCE plume has not been identified. An interim remedial action to contain and treat the contaminated groundwater was initiated at the site in 1986. The action consists of an air-stripping operation to remove the VOCs. Treated water is then discharged through the storm sewer into Rice Creek. Per the OU-2 ROD, the selected cleanup method for the site is to continue using the existing groundwater recovery and treatment system with modifications to the system (U.S. Army 1997). In addition, characterization of the shallow unsaturated zone soil will also be conducted. The site cleanup is expected to take several decades to complete.

3.3.1.2 Rice Creek Ecological System

Rice Creek generally has a sandy bottom and grasses cover its banks. A broad-leaved deciduous scrub-shrub wetland is located along the western bank of the creek. This wetland is suspected to be permanently flooded, however, year-round observations on the hydrology of this wetland have not been made. The wetland contains a number of small oxbow ponds that are used by amphibians for breeding. Various waterfowl species, Belted kingfishers (*Megaceryle alcyon*), groundhogs (*Marmota monax)*, cottontail rabbits (*Sylvilagus floridanus*), and red fox (*Vulpes vulpes)*, have been observed using the area immediately adjacent to Rice Creek.

3.3.1.3 Previous Evidence of Potential or Real Ecological Effects Associated with Rice Creek

The *Ecological Assessment* remained inconclusive with regards to potential ecological effects from TCAAP activities (USAEHA 1991). It stated that Rice Creek occasionally suffers from low water levels that could complicate assessments of contaminant effects on populations of stream-dwelling organisms. The report recommended that the Creek be evaluated by an aquatic biologist to focus upon a benthic invertebrate survey, and that sediment and surface water sampling should be performed. Both of these recommendations were followed within the Tier I risk assessment (USACHPPM 1997).

The Tier I assessment concluded that the stream quality of Rice Creek is below average as it enters TCAAP; and that it remains so throughout it's course on post. Though some chemical impacts were potentially present according to the assessment, a benthic macroinvertebrate survey provided no evidence that the sediments in the creek are adversely affected by TCAAP operations. In the surface water, barium may have the potential for causing toxicity. Zinc was shown not to pose a hazard to aquatic organisms. Though levels of zinc which exceed water quality standards (as defined in the Tier I ERA) were detected during the initial sampling and analysis of the OU-2 FS, additional sampling during the OU-2 FS and the annual monitoring programs of 1994 and 1995 at five Rice Creek stations indicate that water concentrations of zinc do not consistently exceed the standard. The assessment remained inconclusive with regard to potential risk from copper, mercury, and silver in the surface water due to analytical method detection limits that were inadequate for screening purposes because they were higher than toxicological benchmarks.

3.3.2 Rice Creek Risk Management Decisions to Date

Investigation of the spatial extent of site characterization at Rice Creek was not completely adequate. The Tier I risk assessment recommended additional surface water monitoring to establish whether aluminum, barium, cadmium, copper, silver, and mercury concentrations are consistently high enough to reduce the survival, growth, and reproduction of fish, aquatic invertebrates, and algal species in the creek on and off TCAAP due to Army activities. Specifically, the Tier I risk assessment recommended that water samples be collected from three locations in the creek every quarter for one year. Specifically, including an upstream sample as a reference location because no reference location was previously sampled for Rice Creek. The locations were to include one upstream, one downstream, and one in between (near the Bldg. 103 discharge point). The goal would be to design the monitoring to determine if aluminum, barium, cadmium, copper, silver, and mercury concentrations consistently exceed their water quality benchmarks. The Tier II Work Plan included this recommendation. It was determined that no further action for the sediments of the creek would be warranted (USACHPPM 1999).

3.3.3 Rice Creek Assessment Endpoints

The Tier II Work Plan defined two assessment endpoints for Rice Creek.

- Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species
- Survival, growth, and reproduction of benthic organisms

Though the presence of some chemicals of concern in the sediments of the creek was identified, the Tier I risk assessment found that the potential for adverse changes in the ability of benthic organisms to survive, grow, and reproduce were negligible (USACHPPM 1997). This conclusion was based on a review of standard chemical-specific sediment quality benchmarks and a benthic macroinvertebrate survey that provided evidence that the benthic community was not adversely impacted.

Therefore, the Tier II risk assessment focused upon the uncertainties of the risk estimates for the chemicals in the surface waters of the creek.

3.3.4 Rice Creek Initial Conceptual Model

The conceptual model for Rice Creek is as follows:

- Based upon the aquatic toxicity of aluminum, barium, cadmium, copper, silver, and mercury, if they are present in the creek above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates and algal species.

If the concentrations of these chemicals are not consistently higher on-post than off-post and up gradient, then it will be assumed that TCAAP is not the source for these contaminants. Figure B1-8 illustrates the links between the hypothesized contaminant source and potential adverse ecological effects.

Rice Creek COPCs

Surface Water
Aluminum
Barium
Cadmium
Copper
Mercury
Silver

3.3.5 Rice Creek Analysis Plan Summary

Three types of specific measures were chosen for evaluation. The measures are shown in the bullets below.

- Measures of exposure
  Unfiltered water column concentrations of the COPCs

- Measures of ecosystem characteristics
  Water hardness, total suspended solids, total organic carbon, dissolved oxygen, and nitrogen & phosphorus levels

- Measures of effect
  Minnesota Water Quality Standards or effects-based benchmarks for fish, invertebrates, and algal species

For the exposure measurement, unfiltered water column COPC concentrations are used from the surface water monitoring. The Tier II Work Plan indicated that COPC concentrations would be monitored by collecting water samples from several locations in the creek every quarter for one year. The locations included two upstream, one downstream, and one in between (near the Bldg. 103 discharge point). Characteristics of the water column were also measured in order to provide standard water chemistry information.

For the effects measurement, the Minnesota Water Quality Standards (MWQS) were used, when available for a COPC. When MWQS were not available, other sources were consulted to develop a toxicity benchmark. Where appropriate, distributions of effects are presented in comparison to site concentrations.

3.4 SUNFISH LAKE

3.4.1 Available Information on Sunfish Lake

Sunfish Lake is a shallow 5.7-hectare lake in the southeast corner of the installation. Sunfish Lake drains through Marsden Lake, which in turn drains off-post in at least three different directions. Sunfish Lake is fed with water entering from a ditch on the west side of the lake. Figure B1-9 is an orthophoto of the site.

3.4.1.1 Sunfish Lake Contaminant Source

The most likely source of contamination for Sunfish Lake is Site H, including the metal debris scattered on the bottom of Sunfish Lake. Site H has been divided into two areas (H-1 and H-2) to better define past disposal practices (see Figure H1). Area H-1 (820 x 500 ft) is on the northern shore of the lake. Area H-2 (200 x 260 ft) is northeast of Area H-1. During the 1940s, a northern bay of Sunfish Lake was filled. Area H-1 now occupies this filled area. The eastern portion of Area H-1 was used as a burning area for class A combustible waste (paper, wood, cardboard, etc.) from 1940 to 1946. In the 1960s, solvent contaminated corncobs were burned in this area. The southwest corner of Area H-1 may have been used for burial and dumping of industrial sludge, paint residue, incineration ash, and solvents. Metal debris, including small caliber shells, is scattered on the shore of Sunfish Lake at Site H and on the bottom of Sunfish Lake.

The upper 15 ft of soil at Site H was contaminated by heavy metals and/or minor amounts of volatile organic compounds. Per the OU-2 ROD, the selected clean-up method for the site was to excavate all of the affected soil and waste materials, and to dispose of them in a permitted offsite landfill. Currently the following clean up actions have been completed at Site H: a characterization and bounding of contaminated soils at the areas of concern (AOCs), excavation of 1,602 cubic yards of the contaminated soils from AOCs, characterization of the Site H dump site, stabilization of contaminated soils in the dump, removal of metals contaminated soils and debris at Site H dump (6,998 cubic yards), design and construction of the soil cover at the Area H1-3 dump, and backfill with clean fill and regrade of site H (Stone & Webster 2001). The remediation of Site H is complete; however, minimal work will continue at Site H in order to remove erosion controls from the soil cover at Area H1-3.

3.4.1.2 Sunfish Lake Ecological System

Cattail (*Typha sp.*) and sedges (*Carex sp.*) dominate the outfall of the lake. Sunfish Lake is used by a number of waterfowl species including Canadian geese (*Branta canadensis*) and Mallard (*Anas platyrhynchos*). Pied-billed grebes (*Podilymbus podiceps*) and a Great Blue heron (*Ardea herodias*) were seen during field investigations. In the past, Sunfish Lake was used by the Minnesota Department of Natural Resources (DNR) to rear walleye (*Stizostedion vitreum vitreum*) and muskellunge (*Esox masquinongy*). Sunfish Lake received copper sulfate treatments in the past to control algal growth. It has been suspected that Sunfish Lake experiences "freeze-through" with resulting winter die-off of fish populations. However, some relatively large muskellunge and walleye have been captured from Sunfish Lake during fish fry harvesting, indicating that 100 percent winter kill was not occurring. The lake supports populations of frogs and turtles, and herons have been seen frequenting the site. Red fox, muskrat, and mink are known to use Sunfish Lake. Active mink dens are suspected to be present in the banks of the lake. Other mustelid dens are thought to be present, but have not been confirmed.

3.4.1.3 Previous Evidence of Potential or Real Ecological Effects Associated with Sunfish Lake

Sunfish Lake COPCs

Surface Water
Aluminum
Barium
Cadmium
Copper
Mercury
Silver
Zinc

Sediment
Aluminum
Chromium
Copper
Lead
Vanadium
Zinc

In the *Ecological Assessment* (USAEHA 1991), surface water and sediment data collected indicated that concentrations of metals were not sufficiently elevated to be threatening to aquatic organisms. In two individual samples, high levels of antimony and chromium were found along the face of the northern bank of the lake. The *Ecological Assessment* reported that cyanide was detected in the surface water entering the lake in 1987 and was at levels well above the chronic national ambient water quality criteria (AWQC).

The Tier I risk assessment (USACHPPM 1997) reported that Sunfish Lake sediments show signs of organic over-enrichment and might also be suffering from chemical impacts. Aluminum, barium, and zinc in the surface waters may be able to cause toxic effects in the water column. The assessment remained inconclusive with regard to potential risk from copper, mercury, and silver in the surface waters. This was due to method reporting limits that were too high for screening purposes because they were above the relevant water quality standards. The substances thought to be contributing the most to these risks in the sediment were: aluminum, chromium, copper, lead, vanadium, and zinc. Benthic macroinvertebrates, amphibians, wading birds, and waterfowl were also identified as being potentially at risk. Though numerous muskrat homes and other mammalian tracks and dens have been observed, aquatic mammals are predicted to be experiencing the highest chemical risks at Sunfish Lake, based on the screening evaluation.

The Tier I assessment recommended a sediment bioavailability evaluation and a concurrent benthic community diversity survey. It stated that such studies should be designed to evaluate the remaining COPCs in sediment that are specific for benthic organisms: chromium, copper, lead, and zinc. For surface waters, the recommendation was to collect water samples from several locations in the lake every quarter for one year. The purpose of such data collection would be to provide the necessary data to determine if aluminum, barium, cadmium, copper, mercury, silver, and zinc concentrations consistently exceed their water quality benchmarks.

A preliminary study (USACHPPM 1998) addressed some of these remaining sediment issues at Sunfish Lake. The study found that copper, lead, and zinc in Sunfish Lake sediments were not bioavailable at the time of the study, given the analytical technique (i.e., the simultaneous extracted metals/acid volatile sulfide test). However, the study recommended further bioassays to address the limitations of the data.

3.4.2 Sunfish Lake Risk Management Decisions to Date

Investigation of the spatial extent of contamination at Sunfish Lake was considered to be adequate. However, it was determined that further assessment implementing the surface water recommendations of the Tier I risk assessment were warranted. Also, it was decided that, because of the uncertainties in the sediment bioavailability test predictions, that benthic risks would also be further evaluated with sediment toxicity tests. Last, it was decided that the Tier I risk estimates for the wildlife associated with the lake needed to be refined. The Tier II Work Plan outlined these points.

3.4.3 Sunfish Lake Assessment Endpoints

The Tier II Work Plan defined four assessment endpoints for Sunfish Lake.

- 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 and wading birds

3.4.4 Sunfish Lake Initial Conceptual Model

The conceptual model for Sunfish Lake consists of five risk hypotheses. Figure B1-10 illustrates the hypothesized links between the contaminant source and potential adverse ecological effects.

1. Based upon the aquatic toxicity of aluminum, barium, cadmium, copper, silver, mercury, and zinc, if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

2. Based upon the toxicity of aluminum, chromium, copper, lead, zinc, and vanadium; if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment. (An adequate evaluation of potential problems for the amphibian community of the lake was a major data gap in the Tier I assessment.)

4. The survival, growth, and reproduction of wading birds may be at risk if they are excessively exposed to aluminum and chromium that may occur in the benthic invertebrates that they consume at the lake.

5. The health of aquatic mammals, such as muskrat, may be impaired if they are excessively exposed to sediments containing elevated levels of aluminum and vanadium.

3.4.5 Sunfish Lake Analysis Plan Summary

Specific measures were chosen for evaluation of the risk hypotheses.

1. The first hypothesis, which deals with the assessment endpoint of "survival, growth, and reproduction of fish, aquatic invertebrates and algal species," was evaluated with the following measures.

- Measures of exposure
  Unfiltered water column concentrations of aluminum, barium, cadmium, copper, silver, mercury, and zinc

- Measures of ecosystem characteristics
  Water hardness, total suspended solids, total organic carbon, dissolved oxygen, and nitrogen & phosphorus levels

- Measures of effect
  Minnesota Water Quality Standards or other effects-based benchmarks for fish, invertebrates, and algal species.

For the exposure measurement, unfiltered water column COPC concentrations are used from the surface water monitoring. The Tier II Work Plan indicated that COPC concentrations would be monitored by collecting water samples every quarter for one year from several locations in the lake and a reference lake (i.e., Snail Lake). Characteristics of the water column were also measured in order to provide standard water chemistry information. For the effects measurement, the Minnesota Water Quality Standards (MWQS) are used, when available for a COPC. When MWQS are not available, other sources were consulted to develop a toxicity benchmark.

2. The second hypothesis, which deals with the assessment endpoint of "survival, growth, and reproduction of benthic organisms," was evaluated with the following measures.

- Measures of exposure
  Concentrations of aluminum, chromium, copper, lead, zinc, and vanadium in surface sediments (1 ft depth) and SEM/AVS

- Measures of ecosystem and receptor characteristics
  Total organic carbon and the presence/absence of benthic indicator species

- Measures of effect
  Effects-based benchmarks for benthic organisms and sediment toxicity tests using benthic organisms

For the exposure measurements, historical surface sediment data and analyses were used. Historical data were also used for the measurements of total organic carbon and the presence/absence of benthic indicator species. Measures of effects consisted of a re-evaluation of effects-based benchmarks and new sediment toxicity testing.

3. The third hypothesis, which deals with the assessment endpoint of "survival, development, and reproduction of amphibians," was evaluated with the following measures.

- Measures of exposure
  Unfiltered water column and surface sediment concentrations of COPCs

- Measures of ecosystem and receptor characteristics
  Species diversity of amphibians

- Measures of effect
  Modified FETAX assay, amphibian egg mass, hatching success, immunological assays, and mortality

No previous assessment investigated possible effects in the amphibian populations. For the exposure measurements, historical surface data were used. Field surveys were conducted to measure ecosystem and receptor characteristics. Effects were assessed with bioassays, where field-collected water samples were used for controlled laboratory experiments on amphibian health.

4. The fourth and fifth hypotheses, which deal with the assessment endpoint of "reproductive potential and productivity of aquatic mammals and wading birds," were evaluated with the following measures.

- Measures of exposure
  Redesigned exposure model using water and sediment concentrations of COPCs and estimates of COC residues in food

- Measures of ecosystem and receptor characteristics
  Life history habits and exposure factors for representative wildlife species

- Measures of effect
  Laboratory-derived chronic effect levels or other appropriate toxicity benchmarks

For the exposure measurements, current and historical water and sediment data were used to develop exposure profiles for the selected receptors. This included bioaccumulation models, where appropriate. A re-evaluation of toxicity thresholds was performed to select effects measures.

3.5 ROUND LAKE

3.5.1 Available Information on Round Lake

The U.S. Fish and Wildlife Service (USFWS) manages Round Lake as a management unit of the Minnesota Valley National Wildlife Refuge. It is located just southwest of the Twin Cities Army Ammunition Plant, across Highway 10 and Highway 96. The lake itself is approximately 125 acres in size and has a maximum depth of 21 feet with less than 5% of the basin more than 20 feet deep. The balance is a relatively flat shoal area averaging approximately 4.5 feet in depth (USFWS 1982). Figure B1-11 is an orthophoto of the site.

3.5.1.1 Round Lake Contaminant Source

Historical sources of pollutants to Round Lake include activities at part of Site I of the TCAAP facility. Site I consists of building 502 and its associated structures, facilities, and surrounding property. The site occupies approximately 43 acres and is about 700 feet north of the southern boundary of TCAAP, near the intersection on Highways 10 and 96 (USAEHA 1991). Round Lake may also have received contamination by unknown sources, other than the Army, which are out of the Army's control. Non-TCAAP sources of contamination include runoff from Highways 10 and 96, past known and unknown spills from other industrial areas around the lake, and poor waste management practices in the watershed upgradient of the lake (see section 6.5.1 for further discussion). There is a storm sewer in the northwest corner that drains into Round Lake and is not connected to TCAAP. To the west of the Lake are/were three companies, Scholl's Inc., Minnesota Diversified Products Inc., and a hide and tallow company (USFWS 1982).

The source of entry of TCAAP waste products to Round Lake was through a storm sewer at the northern end of Round Lake. Storm-water runoff from Site I was collected in surface drainage ditches and entered Round Lake through a storm sewer outfall located at the southwestern boundary of TCAAP. The outfall discharged surface drainage from buildings 501, 502, and 515 and the southern portion of building 503. Part of the Site I facility was used to produce artillery shell forgings. The production forges were cooled by water that was discharged to floor drains. Water used in general cleanup operations was also discharged into these drains. The floor drains in the production areas of the buildings were connected to the storm sewer, rather than the sanitary sewer. The forging equipment used large quantities of hydraulic fluid containing high percentages of PCBs. Leakage of this equipment, lubricants used in the forging process, and other contaminants consequently had a pathway into the storm sewer. The storm sewer discharged into a ditch that feeds into Round Lake through a sewer outfall at the intersection of Highway 96 and 10. During 1942-1943, approximately 1.9 million liters (500,000 gallons) per day were discharged from building 502 and building 501 into the lake. In 1944, forges were added to both buildings and daily discharges increased and remained high until the end of the war. The storm sewer was altered in 1953 in response to complaints about grease and oil pollution in Round Lake, and water from some of the floor drains was subsequently sent to the sanitary sewer. For about 1 year before this conversion, Round Lake may have received about 3.8 million liters (1 million gallons) of waste water per day from building 502. Some cyanide wastes were still discharged through to the floor drains through 1967. In November 1967, the Federal Water Pollution Control Administration (FWPCA) investigated the disposal of industrial wastes at TCAAP in the context of controlling the potential for water pollution. In building 502, a variety of wastes were identified as being produced and discarded to the rinse water drains. These wastes contained alkali, muriatic acid, sodium-cadmium cyanide, cadmium cyanide, sodium cyanide, and sodium hydroxide. From the rinse water drains, these wastes were discharged to a holding pit, diluted, and then discharged to the combined sewer system. Other industrial wastes identified in the FWPCA report that were possibly being produced in building 502 and disposed of by discharge to the sewer system included nitric and sulfuric acids, cleansers, caustics, rust inhibitors (such as chromates and dichromates), and sodium aluminate (ANL 1991). In 1969, the tenant of the building noticed that many of the floor drains were still connected to the storm sewer. This situation has since been remedied. Additionally, a landfill was identified on the northern shore of Round Lake in 1991. The landfill was constructed by the brother of one of the residents and is reportedly composed of rubble fill.

3.5.1.2 Round Lake Ecological System

A palustrine emergent wetland has developed around the edge of the lake. This wetland is dominated by cattail (*Typha sp.*). A small stand of willow (*Salix sp.*) exists along the northern shore. Mammals known to use Round Lake are red fox, muskrat, and mink. The wetland areas are also used by a number of typical marsh birds, with Red-winged blackbirds (*Agelaius phoeniceus*) and the Yellow-headed Blackbird (*Xanthocephalus xanthocephalus*) being dominant. Waterfowl can potentially use Round Lake as a foraging, nesting, or resting area. However, large numbers of various species of waterfowl were not observed during the field investigation. A few Canada geese (*Branta Canadensis*) and one Common loon (*Gavia immer*) were seen. Waterfowl species rearing broods on the lake in recent years include Canada geese, Mallards, Blue-winged teal, and Wood ducks. Large concentrations of Ringed-neck ducks and Lesser scaup use Round Lake as a resting and feeding area during their spring and fall migrations. Round Lake also has confirmed use by Black terns and Common loons.

3.5.1.3 Previous Evidence of Potential or Real Ecological Effects Associated with Round Lake

The *Ecological Assessment* (USAEHA 1991) summarized several limited data sets for Round Lake. Three surface water samples had contaminant concentrations greater than the AWQC for cyanide and thallium. Fish tissue analyses showed that black bullheads were accumulating some contaminants from the lake; however, metal residues were similar to those in various bullheads from other Minneapolis metropolitan area lakes. The residues of total PCBs in the lake's bullheads are higher than those found in the samples from other area lakes. However, the levels were characterized as low and insignificant by the USFWS biologist who summarized the results of the 1981 sampling effort. He also described the lake system as apparently healthy, but he qualified that conclusion based upon limited sampling. The levels of sediment metals were "undoubtedly toxic" to some benthic organisms. The lack of correspondingly high levels of surface water metals is probably due to the tendency for metals to bind to sediments.

The Tier I risk assessment (USACHPPM 1997) reported that though Round Lake appears to be typical of a natural eutrophic pond environment, chemical impacts could be occurring. Barium and zinc in the surface waters may be able to cause toxic effects in the water column. The assessment remained inconclusive with regard to potential risk from copper, mercury, and silver in the surface waters. This was due to method detection limits which were inadequate for screening purposes. The substances thought to be contributing the most to this potential risk in sediment are limited to the metals: aluminum, cadmium, chromium, copper, silver, vanadium, and zinc. Upon further review of existing information, it was determined that lead should be re-evaluated as a contributor to the potential risk in sediments. Aquatic mammals, wading birds, benthic organisms, and amphibians of the system are predicted to be impacted by the elevated metals. The northern portion of the lake appeared to pose more risk to benthic organisms based on the screening risk model. However, the benthic evaluation conducted in 1993, and included in the 1997 Tier I risk assessment (USACHPPM 1997), indicated that this might not be the case. The benthic evaluation indicated that the southern end, near the lake's outfall, is more impacted in terms of biodiversity. A limited bioavailability investigation of sediment metals (MPCA 1994) showed that there might be sufficient acid volatile sulfides in the sediments to bind cadmium, copper, mercury, and zinc. Because acid volatile sulfide levels in sediments are dynamic, vary seasonally, and only a few samples were collected during one season, that data did not provide conclusive evidence that the metals are biologically unavailable.

The Tier I ERA recommended performing sediment toxicity tests at the southern end of the lake, a bioavailability evaluation at the northern end, and concurrent benthic community diversity surveys. It was suggested that the toxicity tests be designed to determine if the benthic impacts in the southern portion of the lake are related to toxicity. It was suggested that the bioavailability study be designed to evaluate the remaining COPCs in sediment. For surface waters, the recommendation was to collect water samples from several locations in the lake every quarter for one year. The purpose of that data collection would be to provide the necessary data to determine if barium, cadmium, copper, mercury, silver,, and zinc concentrations consistently exceed their water quality benchmarks.

Two preliminary studies have addressed some of these remaining sediment issues at Round Lake-*Bioavailability of Sediment Metals in Round and Sunfish Lakes* (USACHPPM 1998b) and *Sediment Toxicity Evaluation of Round Lake* (USACHPPM 1998a). The bioavailability report found that data from three of the four sample locations in the lake indicated that sufficient acid volatile sulfide existed in the northern sediments to bind cadmium, copper, nickel, lead, and zinc decreasing their bioavailability. However, the most vulnerable time for toxic effects in organisms would be expected to occur in the cold winter months, which were not evaluated. The sediment toxicity report found that the sediments in the southern portion of the lake were not toxic to laboratory cultured macroinvertebrates under current conditions.

The issue of polychlorinated biphenyls (PCBs) in Round Lake was raised during the Tier II ERA Planning Meeting (4 March 1998). At this meeting, it became apparent that, given the historical information on PCB data collected at Round Lake, the Tier I ERA may not have adequately addressed risk from PCBs. A detailed review and evaluation of all the available PCB data collected in the lake since the 1980's was developed as part of the Work Plan. A copy of this review is presented in Appendix P. The review recommended that *further investigation* into possible PCB-related biological effects at Round Lake not be performed beyond that already planned for macroinvertebrates and amphibians. The results of the risk evaluation contained in Appendix P are also addressed in Section 8 (Risk Characterization).

3.5.2 Round Lake Risk Management Decisions to Date

Investigation of the spatial extent of contamination at Round Lake was considered to be adequate. However, it was determined that further assessment, which implements the surface water recommendations of the Tier I risk assessment, was warranted. Because of the uncertainties in the SEM/AVS predictions of the bioavailability studies, benthic risks needed further assessment using sediment toxicity tests. Additionally, the risk estimates for the wildlife associated with the lake were to be refined.

Of the study sites in the risk assessment, Round Lake is the only site that has the potential for "active management" as an ecological system. The USFWS would like to physically improve the habitat quality of the lake to attract higher numbers of migrating wildlife, especially waterfowl and shorebirds (USFWS 1998, 1982). During the deliberations over the scope and design of the risk assessment and associated field activities, it became evident that additional clarity was needed regarding two issues: (1) the potential USFWS management alternatives for Round Lake needed to be addressed and (2) ecological risks associated with each of those alternatives needed better definition. Simply put, the risk assessment would need to describe chemical-risks associated with potential wildlife management actions at Round Lake. (Descriptions of the potential chemical risks associated with wildlife management are addressed in this report as the Round Lake Future scenario).

3.5.3 Round Lake Assessment Endpoint

The Tier II Work Plan defined four assessment endpoints for Round Lake.

- 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, waterfowl, and wading birds (Note: Per the Work Plan, waterfowl are an ecological entity of concern for future scenarios only because the current (or "no action") risks to waterfowl were found to be acceptable in the Tier I risk assessment.)

3.5.4 Round Lake Current Initial Conceptual Model

The general conceptual model for Round Lake consists of five risk hypotheses. Figure B12 illustrates the links between the hypothesized contaminant source and potential adverse ecological effects under the current (or "no action") scenario.

1. Based upon the aquatic toxicity of barium, cadmium, copper, mercury, silver, and zinc, if they are present in the water above acceptable levels for sufficient duration's, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

2. If sediment concentrations of aluminum, cadmium, chromium, copper, lead, silver, zinc, and vanadium are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may be at risk due to contaminants present in the lake.

4. The health of aquatic mammals, such as mink or muskrat, may be impaired if they are excessively exposed to sediments containing elevated levels of aluminum, cadmium, and vanadium.

5. Because the Tier I risk assessment failed to properly screen for mercury (i.e., the detection limits were too high), wading birds and mink that consume fish or macrobenthic organisms in the lake may be at risk.

Round Lake
COPCs

Surface Water
Barium
Cadmium
Copper
Mercury
Silver
Zinc

Sediment
Aluminum
Cadmium
Chromium
Copper
Lead
Silver
Zinc
PCBs

3.5.5 Round Lake Future Conceptual Model

The general conceptual model for Round Lake future consists of nearly the same six risk hypotheses and COPCs as for the current Round Lake scenario. Chemicals in surficial and deep sediments may become mobilized during active disturbances to the sediments during water level manipulations associated with the proposed USFWS management plan. Such mobilization or change in bioavailability of COPCs may result in adverse ecological responses greater than under the current ("no action") scenario. Figure B12 illustrates the deep sediment stressor link between the hypothesized contaminant source and potential adverse ecological effects under the future condition.

1. Based upon the aquatic toxicity of surface water COPCs, if they are present in the water above acceptable levels for sufficient duration's, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

2. If sediment concentrations of COPCs are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may be at risk due to contaminants present in the lake.

4. The health of aquatic mammals, such as mink and muskrat, may be impaired if they are excessively exposed to sediments containing elevated levels of COPCs.

5. The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COPCs that may occur in the benthic invertebrates and fish that they consume at the lake.

3.5.6 Round Lake Analysis Plan Summary

Specific measures were chosen for evaluation of the risk hypotheses for both current and future scenarios.

1. The first hypothesis, which deals with the assessment endpoint of "survival, growth, and reproduction of fish, aquatic invertebrates and algal species," was evaluated with the following measures.

- Measures of exposure
  Unfiltered water column concentrations of barium, cadmium, copper, mercury, silver, and zinc

- Measures of ecosystem characteristics
  Water hardness, total suspended solids, total organic carbon, and nitrogen & phosphorus levels

- Measures of effect
  Minnesota Water Quality Standards or other effects-based benchmarks for fish, invertebrates, and algal species.

For the exposure measurement, unfiltered water column COPC concentrations are used from the surface water monitoring. The Tier II Work Plan indicated that COPC concentrations would be monitored by collecting water samples every quarter for one year from several locations in the lake and a reference lake (i.e., Snail Lake). Characteristics of the water column were also measured in order to provide standard water chemistry information. For the effects measurement, the Minnesota Water Quality Standards (MWQS) are used, when available for a COPC. When MWQS are not available, other sources were consulted to develop a toxicity benchmark.

2. The second hypothesis, which deals with the assessment endpoint of "survival, growth, and reproduction of benthic organisms," was evaluated with the following measures.

- Measures of exposure
  Concentrations of identified aluminum, cadmium, chromium, copper, lead, silver, zinc, and vanadium in surface sediments (1 ft depth) (collected as part of the Tier I assessment)

- Measures of ecosystem and receptor characteristics
  Total organic carbon and the presence/absence of benthic indicator species

- Measures of effect
  Effects-based benchmarks for benthic organisms and sediment toxicity tests using benthic organisms

For the exposure measurements, historical surface sediment data and analyses were used. Historical data were also used for the measures total organic carbon and the presence/absence of benthic indicator species. Measures of effects consisted of a re-evaluation of effects-based benchmarks and new sediment toxicity testing.

3. The third hypothesis, which deals with the assessment endpoint of "survival, development, and reproduction of amphibians," was evaluated with the following measures.

- Measures of exposure
  Unfiltered water column and surface sediment concentrations of COPCs

- Measures of ecosystem and receptor characteristics
  Species diversity of amphibians

- Measures of effect
  Modified FETAX assay, amphibian egg mass, hatching success, immunological assays, and mortality

No previous assessment investigated possible effects in the amphibian populations. For the exposure measurements, investigation results were used. Field surveys were conducted to measure ecosystem and receptor characteristics. Effects were assessed with bioassasys, where field-collected water samples were used for controlled laboratory experiments on amphibian health.

4. The fourth and fifth hypotheses, which deal with the assessment endpoint of "reproductive potential and productivity of aquatic mammals, waterfowl, and wading birds," were evaluated with the following measures (refer to each scenario's conceptual site model for specifics on receptors to be evaluated).

- Measures of exposure
  Current scenario: Exposure model using water and sediment concentrations of COPCs and estimates of COPC residues in food
  Future scenario: Estimates of whether concentrations of chemicals in the surface sediments will increase or become more, or less bioavailable during or after full and partial water level draw down, based on predicted changes in the chemical-physical attributes of the lake.

- Measures of ecosystem and receptor characteristics
  Current: Life history habits and exposure factors for representative wildlife species
  Future: Estimates of changes in the chemical-physical attributes of the sediments during and after full and partial water level draw down, based on a review of available literature.

- Measures of effect
  Current: Laboratory-derived chronic effect levels or other appropriate toxicity benchmarks
  Future: No specific measures were used. The measures used for the current scenario were used indirectly.

For the exposure measurements, current water and sediment data were used to develop exposure profiles for the selected receptors. Under the current scenario, this included bioaccumulation models, where appropriate. An evaluation of toxicity thresholds was performed to select effects measures. Where appropriate, distributions of exposure and effects are presented in comparison to site concentrations.

3.6 SITE B WETLANDS

3.6.1 Available Information on Site B

The Site B Wetlands are a series of prairie potholes associated with Site B. The area is located in the northern most part of the installation between Site A and Marsden Lake.

3.6.1.1 Site B Contaminant Source

Site B consists of three separate subsites (areas B1, B2, and B3) totaling about 0.9 hectares. Several abandoned farmsteads are present in this area. There are numerous clumps of trees, especially around the abandoned farmsteads. There is no documentation of hazardous substance disposal at this site, although it is possible that small amounts of sewage sludge were dumped before 1966 (Montgomery Watson 1997). The southwestern corner of area B3 is part of a larger landfill area. During that field study, chunks of asphalt were visible in the fill face. Pursuant to the Operable Unit 2 Record of Decision (U.S. Army 1997), Site B was further characterized in the fall of 1998 (USACE 2001). Sewage sludge, suspected burn areas, or oily layers were not discovered during the characterization activities. The conclusion of the dump characterization is that its contents are not toxic, hazardous, or contaminated.

3.6.1.2 Site B Ecological System

The area is dominated with emergent vegetation, (*Typha* sp.). The most notable herpetile known to use the area is the Blandings turtle (*Emydoidea blandingi*), a species of concern since its population is declining nationwide. Blandings turtles were seen in an important habitat area adjacent to Site B (Linck 1989). One muskrat (*Ondatra zibethicus*) lodge was observed during a site visit in the winter of 1995. Red fox are also known to use this area.

3.6.1.3 Previous Evidence of Potential or Real Ecological Effects Associated with Site B

The Site B Wetlands were not evaluated in the *Ecological Assessment* (USAEHA 1991) and the Tier I ERA (USACHPPM 1997) was based upon the limited data for area B3 available at the time. The Tier I ERA reported the most significant potential for risks are due to detection of pesticides in the sediments. However, the source of the pesticides was unknown, though suspected to be due to routine applications, and not due to a spill or release. Some inorganics were identified as potential contaminants (aluminum, barium, manganese, and zinc in surface water). The assessment remained inconclusive with regard to potential contamination/risk from copper, mercury, and silver in the surface waters. This was due to method reporting limits that were inadequate for screening purposes. Aluminum and vanadium in the sediments were identified as potential contaminants, in addition to the pesticides. This site provides some of the best TCAAP habitat for amphibian species. However, insufficient toxicity information existed to screen the potential contaminants for their ability to be toxic to these species throughout their life-cycle without performing toxicity testing. The sediments and surface water at Areas B1 and B2 had not been characterized. The shallow and deep soils, and groundwater at Areas B1 and B2 were characterized during the OU-2 FS. The surrounding soils and groundwater were found to be unimpacted by chemical contamination.

3.6.2 Site B Risk Management Decisions to Date

It was determined that further ecological investigations were not warranted. Site B has been determined to be a clean site. Strong evidence exists to conclude that no contamination is actually present at the site and no contaminants have been identified with the dump near the three areas of Site B (Montgomery Watson 1997 and USACE 2001). The ecological risk assessment for this site ended during the development of the Tier II Work Plan.

4. TIER II FIELD AND ANALYTICAL CHEMISTRY INVESTIGATIONS

This section presents the results of the site investigations performed during 1999 and 2000 (USACHPPM 1999). More specifically, this section presents the results of the field measurements, analytical chemistry data, and their associated quality control (QC) outcomes. A summary of the investigation at each of the study sites is provided. Historical analytical chemistry investigations at each of the sites are also discussed. (Biological and toxicological data collected during the same timeframe are reported separately in Section 6.)

Appendix C presents photo-documentation of portions of the field sampling effort. Volume 2 (i.e., CD ROM No. 2) provides all of the project records for field and laboratory measurements, project quality assurance reports, and data review and validation.

4.1 MARSDEN LAKE SITE INVESTIGATION RESULTS

In order to address deficiencies in previous sampling and analysis data, site characterization activities were conducted at two areas of Marsden Lake: referred to as Marsden Lake (North), possibly impacted by historical operations at the Grenade Range, and Marsden Lake (South), possibly impacted by historical operation of the Outdoor Firing Range.

4.1.1 Analytical Chemistry Investigations of Marsden Lake: 1992-1993

The Tier I ERA report (USACHPPM 1997) used data from the OU2-FS (Montgomery Watson 1994 and 1997). In that investigation, five sediment and five surface water samples were collected from Marsden Lake in October 1992. One sediment and surface water sample were collected at the inlet to Marsden Lake from Sunfish Lake (ML01). One sediment and surface water sample were collected near the major source of surface water drainage, on the western shoreline of Marsden Lake. The remaining samples were collected from the western shore to evaluate the distribution of potential contamination around the perimeter of the lake. Samples were analyzed for metals, cyanide, and PCBs. Two additional surface water samples were collected from Marsden Lake in June 1993 to verify previous sampling results. These results are presented in Appendix Q (1997 Tier I Report).

4.1.2 Site Characterization of Marsden Lake (North)

During the 1999 sampling for the Tier II investigation, Marsden Lake was separated into northern and southern sections to define distict areas that were associated with potential source areas. Marsden Lake North was considered to be the shoreline area along the Grenade Range impact areas. Specifically, just on the northern and southern sides of Patrol Road. The intention of sampling in these areas was to characterize potential contaminant transport from the Grenade Range into the lake. Characterization of all of Marsden Lake was not in the scope of this project.

4.1.2.1 Marsden Lake (North) Results: 1999

Sampling locations and measurement results from the 1999 investigation are summarized in Appendix D. Sampling and analysis were conducted following the procedures identified in the Work Plan (USACHPPM 1999), and are also summarized in Appendix K of this report. Originally ten primary samples of surface water and surface sediment were to be collected from the area around the Grenade Range in the Northern area of Marsden Lake during September 1999. However, due to difficulties in sampling as discussed below in section 4.1.2.2, only eight samples of the ten collected were used to represent Marsden Lake North for chemical analysis of inorganics and metals, semivolatile organic compounds (SVOCs), and polychlorinated biphenyl (PCB) Aroclor mixtures. (Section 4.1.2.2 explains this decision.) In addition, total organic carbon (TOC) was also measured in the sediments. Water measurements included hardness, total suspended solids (TSS), TOC, total kjeldahl nitrogen (TKN), and total phosphorus. Water pH, dissolved oxygen (DO) content, conductivity, and temperature were measured in the field.

In the eight surface water samples, 16 of 18 analyzed inorganics and metals were detected. Beryllium and selenium were not detected in the water. No pesticides or PCBs were detected in surface water. One SVOC [Bis(2-ethylhexyl)phthalate] was detected in two samples. Average values for general water quality parameters were: 13.1° C, 6.7 pH, 7.5 mg/L DO, 187 mg/L $CaCO_3$, 2.7 mg/L TKN, 14.0 mg/L TOC, 0.28 mg/L total phosphorus, and 84.0 mg/L TSS.

In the eight sediment samples, 14 of 18 analyzed inorganics and metals were detected. Beryllium, cobalt, silver, and thallium were not detected in the sediment. No PCBs, pesticides, or SVOCs were detected in the sediments. Average sediment TOC was 9.87%.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Marsden Lake (North) met the field data quality goals for precision, accuracy, and comparability. Field completeness and representativeness goals were not met in Marsden Lake North due to problems in accessing the intended locations of ML-13 and ML-15. The field sampling crew noted that even though every effort was made to sample at the locations shown in the Work Plan (USACHPPM 1999), the sampling of the exact locations was not possible due to movement of cattail mats at the site. The following subsection discusses the implications of the change in sampling locations. Additionally, representativeness may also be questionable for surface water samples ML-16, ML-17, ML-19, and ML-20. At the time of sampling, these locations were covered with dense floating cattail mats, such that surface water was not accessable. In order to collect surface water samples as near to the intended locations as possible, the sampling team cut holes in the cattail mat with a machete, to remove the root systems and expose the underlying surface water. Surface water samples were then collected the following day to allow time for settling of soil and root tissue after the cutting. (See Appendix K, Section K1.1.4 for further detail).

The analytical laboratory portion of the investigation at Marsden Lake (North) met the laboratory data quality goals for precision, accuracy, completeness, representativeness, and comparability. The target detection limits for both pesticides and semivolatile organic compounds were generally not reached in the sediment samples collected at the site. Target levels listed in the QAPP represent dry-weight reporting levels based on a sample with negligible water content. Actual sediment reporting levels varied, depending on the percent water in the respective sediment sample. Additional discussion regarding this issue can be found in Appendix K (Section K1.2.6).

4.1.2.2 Marsden Lake (North) Summary

The 1999 sampling effort presented in Section 4.1.2.1 was based on the conclusions and recommendations of the Tier I ERA report (USACHPPM 1997). Data and information used in the 1997 Tier I assessment, and data collected during the 1999 site investigation activities were evaluated to determine their quality before any conclusions or management decisions could be made about the site. The 1992-1993 data examined the spatial aspect of metals, PCBs, and pesticides. The 1999 data examined the spatial aspect of metals, PCBs/pesticides, and SVOCs.

During the Montgomery Watson investigation (1992-1993), 5 samples were collected in Marsden Lake, but of those five only three were located in the northern section of Marsden Lake (ML05, ML04, and ML03) (See Appendix Q, Figure 7b). Sample ML05 was located near the culvert on the north side of Patrol Road, sample ML04 was located on the south side of the road, on the western shoreline, and sample ML03 was located farther away from the road, just south of ML04. The location of the confirmatory samples taken in 1993 was not identified in the OU2-FS.

Because of the differences in detection limits and collection methods between the 1992-1993 data and the 1999 data, it is best not to combine the two data sets for statistical evaluations. The 1999 investigation used low-level trace metals analysis (USEPA Method 1669) for surface water, so the detection limits were much lower than that in the 1992-1993 surface water data. For example, in the 1992-1993 data, antimony, cadmium, chromium, cobalt, copper, lead, and nickel were undetected, but using the trace metals analysis in 1999, these metals were detected in surface water samples. Sediment metals from the two sampling periods are slightly more comparable because the detection limits did not differ greatly between the two sampling events. However, the comparison should not be quantitative. Sampling in both investigations were single events, and thus, the temporal component of chemical concentrations at this site is not completely understood. The characterization of the spatial variability of chemical concentrations is discussed below.

Because the main source of TCAAP chemical input to Marsden Lake North was suspected to be the grenade range, the 1999 sampling focused on areas around the impact sites of the range. Water flows from the south to the north through culverts (when not obstructed by beaver activity) on both sides of the Sand Pad, which is located in the middle of Patrol Road. Samples were taken on both the north and south sides of the road. The intent of sampling in this area was to determine the potential movement of chemicals that may have been on the range, and through runoff and air dispersion processes been transported into the lake.

Due to problems with wind and moving cattail mats during the 1999 sampling, two sample points were not collected as near to the suspected source area as planned. This was documented in the monthly QA reports (see Quality Assurance Reports, CD2). Sample points ML-13 and ML-15 (see Appendix D, Figure D1) are outside of the sampling area proposed in Figure 8 of Appendix C in the Work Plan (USACHPPM 1999). In order to determine if these two points were characteristic of Marsden Lake North, several evaluations of the analytical data were made. First, a crude graphical comparison of the distance from the center of the closest impact area to the sample point, versus surface water analyte concentration at the sample point was made (See Appendix D, Figures D12-D19). This comparison showed that surface water analyte concentrations from ML-13 and ML-15 were similar to concentrations from ML-11 and ML-14. Samples ML-11 and ML-14 are each inside or right on the edge of an impact area. Next, the same types of graphical comparisons were made for the sediment analyte data (see Appendix D, Figures D20-D37). Because the sediment may be a contaminant sink, more weight was placed on the results of this comparison. The sediment comparison showed that ML-13 and ML-15 appeared to have concentrations less similar to the other Marsden Lake North sediment samples.

Since ML-13 and ML-15 were both located near the western shore of the lake, a decision was made to compare these sediment concentrations to the sediment concentrations collected from Marsden Lake South. The potential source area for Marsden Lake South was suspected to be the outdoor firing range located on the western shore of the lake. The purpose of this comparison was to determine if ML-13 and ML-15 are more characteristic of the potential source area for Marsden Lake South. Boxplots of the concentration data from Marsden Lake North and Marsden Lake South were evaluated to determine how the concentrations from ML-13 and ML-15 compared to the means and interquartile ranges from the Marsden Lake South samples (see Appendix B2). The results of this comparison showed that in most cases the concentrations of analytes detected at sites 13 and 15 fell closer to the mean concentrations detected at Marsden Lake South. Aluminum, barium, chromium, magnesium, and nickel concentrations for ML-13 and ML-15 were outside the interquartile range for Marsden Lake North, but were within the interquartile range for Marsden Lake South. Antimony and copper concentrations at ML-13 and ML-15 were within the interquartile ranges for both Marsden Lake North and Marsden Lake South locations.

However, cadmium and thallium had 7 and 5 undetected values in Marsden Lake North, respectively, and 6 and 10 nondetects in Marsden Lake South; respectively. Therefore, based on the boxplots for cadmium and thallium it is difficult to determine that the spread of the data is not skewed to one side or the other for these two analytes between North and South because of the large number of nondetects. Beryllium and selenium were also not detected in either section of the lake.

Based on professional judgment when looking at the available data and the graphical presentations, ML-13 and ML-15 are considered to be less characteristic of the potential source area in Marsden Lake North and more characteristic of the potential source area in Marsden Lake South. This is plausible given that the direction of water flow is from south to north, and if contaminants were introduced in the southern parts of the lake from the outdoor firing range, they could be carried northward and deposited along the shorelines near the southwestern side of Patrol Road.

Because ML-13 and ML-15 are not considered to be characteristic of the potential source area in Marsden Lake North, field completeness was, therefore, not met for metals and inorganic data. The QAPP (see summary table K1 in Appendix K) specified that field completeness should be 90%, and removing ML-13 and ML-15 from Marsden Lake North data would result in field completeness of 80%. This may indicate that the spatial variability of chemical concentrations associated with the grenade range area is more limited to the areas closest to Patrol Road and the small and large impact areas. This information does not imply that the area around the potential source to Marsden Lake North has not been characterized, such that a confident management decision cannot be made. The data collected from ML-13 and ML-15 will be combined with data collected from Marsden Lake South for statistical analysis. The data from Marsden Lake North was checked for outliers using the Dixon's Test. No values were found to be outliers; therefore, compiling ML-13 and ML-15 with the Marsden Lake South data does not mask any outliers. Further discussion on the statistical analysis of the data will be presented in Section 5.

4.1.3 Site Characterization of Marsden Lake (South)

During the 1999 sampling for the Tier II investigation, Marsden Lake was considered as two distinct sections to correspond with the potential source areas of the grenade range in the north and the outdoor firing range in the south. The area being characterized in Marsden Lake South was the area along the western shoreline, adjacent to the outdoor firing range.

4.1.3.1 Marsden Lake (South) Results: 1999

Sampling locations and measurement results from the 1999 investigation are summarized in Appendix E. Sampling and analysis were conducted following the procedures identified in the Work Plan (USACHPPM 1999), and are also summarized in Appendix K of this report. Originally, ten primary samples of surface water and surface sediment were collected from the near the Outdoor Firing Range along the western shore of Marsden Lake during September 1999 for chemical analysis of inorganics and metals, semivolatile organic compounds (SVOCs), and polychlorinated biphenyl (PCB) Aroclor mixtures. Total organic carbon (TOC) was also measured in the sediments. Water measurements included hardness, total suspended solids (TSS), TOC, total kjeldahl nitrogen (TKN), and total phosphorus. Water pH, dissolved oxygen (DO) content, conductivity, and temperature were measured in the field. After examination of other Marsden Lake data, it was determined that two samples collected to characterize the northern section of the lake around the grenade range were more characteristic of the western shore area associated with the outdoor firing range (see section 4.1.2.2). The analytical data obtained from those sampling locations (ML-13 and ML-15) were combined with the Marsden Lake South data and are summarized below.

In the twelve surface water samples, 16 of 18 analyzed inorganics and metals were detected. Beryllium and selenium were not detected in the water. No pesticides, PCBs, or SVOCs were detected in surface water. Average values for general water quality parameters were: 16.2° C, 7.2 pH, 5.9 mg/L DO, 108 mg/L $CaCO_3$, 0.6 mg/L TKN, 9.1 mg/L TOC, 0.02 mg/L total phosphorus, and 5.6 mg/L TSS.

In sediments, 15 of 18 analyzed inorganics and metals were detected. Beryllium, silver, and thallium were not detected in the sediment. No PCBs were detected in the sediments; however, 4,4'-dichlorodiphenyl-dichloroethane (4,4'-DDD) was detected in one field duplicate sample (ML-13). One SVOC was [Butylbenzylphthalate] detected in one sediment sample (ML-03). Average sediment TOC was 19.5%.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Marsden Lake (South) met the field data quality goals for precision, accuracy, completeness, and representativeness, and comparability. The field sampling crew noted that even though every effort was made to sample at the locations shown in the Work Plan (USACHPPM 1999), the sampling of the exact locations was not possible due to movement of cattail mats at the site. The following subsection discusses the implications of the change in sampling locations.

The analytical laboratory portion of the investigation at Marsden Lake (South) met the laboratory data quality goals for precision, accuracy, completeness, and representativeness, and comparability. The target detection limits for both pesticides and semivolatile organic compounds were generally not reached in the sediment samples collected at the site. Target levels listed in the QAPP represent dry-weight reporting levels based on a sample with negligible water content. Actual sediment reporting levels varied, depending on the percent water in the respective sediment sample. Additional discussion regarding this issue can be found in Appendix K (Section K1.2.6).

4.1.3.2 Marsden Lake (South) Summary

The 1999 sampling effort presented in section 4.1.2.1 was based on the conclusions and recommendations of the Tier I ERA report (USACHPPM 1997). Data and information used in the 1997 Tier I assessment, and data collected during the 1999 site investigation activities were evaluated to determine their quality before any conclusions or management decisions could be made about the site. The 1992-1993 data examined the spatial aspect of metals, PCBs, and pesticides. The 1999 data examined the spatial aspect of metals, PCBs/pesticides, and SVOCs.

During the Montgomery Watson investigation (1992-1993), 5 samples were collected in Marsden Lake, but of those five only two were located in the southern section of Marsden Lake (ML01 and ML02) (See Appendix Q, Figure 7a). Sample ML01 was located at the inlet to Marsden Lake from Sunfish, and sample ML02 was located just off the western shoreline, near the center of the outdoor firing range. The location of the confirmatory samples taken in 1993 was not identified in the OU2-FS.

Because of the differences in detection limits and collection methods between the 1992-1993 data and the 1999 data, it is best not to combine the two data sets for statistical evaluations. The 1999 investigation used low-level trace metals analysis (USEPA Method 1669) for surface water, so the detection limits were much lower than that in the 1992-1993 surface water data. For example, in the 1992-1993 data, antimony, cadmium, chromium, cobalt, copper, lead, and nickel were undetected, but using the trace metals analysis in 1999, these metals were detected in surface water samples. Sediment metals from the two sampling periods are slightly more comparable because the detection limits did not differ greatly between the two sampling events. However, the comparison should not be quantitative. Sampling in both investigations were single events, and thus, the temporal component of chemical concentrations at this site is not completely understood. The characterization of the spatial variability of chemical concentrations is discussed below.

Because the main source of TCAAP chemical input to Marsden Lake South was suspected to be the outdoor firing range, the 1999 sampling focused on the western shoreline of the lake, which is directly adjacent to the outdoor firing range. The intent of sampling in this area was to determine the potential movement of chemicals that may have been on the range, and through runoff and air dispersion processes been transported into the lake.

Due to problems with wind and moving cattail mats during the 1999 sampling, three sample points were not collected as near to the suspected source area as planned. This was documented in the monthly QA reports (see Quality Assurance Reports, CD2). Sample points ML-01, ML-02, and ML-03 (see Appendix D, Figure D1) are outside of the sampling area proposed in Figure 8 of Appendix C in the Work Plan (USACHPPM 1999). The samples were collected farther away from the shoreline than intended. In order to determine if these points were characteristic of the other samples collected in Marsden Lake South, several evaluations of the analytical data were made. The concentration data were visually compared to determine if concentrations from the three sampling locations were much different than the other Marsden Lake South concentrations. For both surface water and sediment there were no major differences in concentrations. Then, Dixon's tests were run on both surface water and sediment data to determine if there were outlying points (see Appendix E, Table E9 and E10). The Dixon's test results showed that ML-01, ML-02, and ML-03 were not different than the other Marsden Lake South samples and therefore, are considered to provide sufficient information on the spatial variability of chemical concentrations in the southern portion of Marsden Lake for purposes of making confident management decisions concerning the potential source area.

4.2 POND G SITE INVESTIGATION RESULTS

In order to address deficiencies in previous sampling and analysis data, additional sampling was performed at the pond.

4.2.1 Analytical Chemistry Investigations of Pond G: 1992-1993

The Tier I ERA report (USACHPPM 1997) used data from the OU2-FS (Montgomery Watson 1994 and 1997). In that investigation, one surface water and one sediment sample were collected slightly north of the center of Pond G. Samples were analyzed for metals, cyanide, and PCBs.

4.2.2 Pond G Results: 1999

Sampling locations and measurement results from the 1999 investigation are summarized in Appendix F. Sampling and analysis were conducted following the procedures identified in the Work Plan (USACHPPM 1999), and are also summarized in Appendix K of this report. Three primary samples of surface water and surface sediment were collected from Pond G in September 1999 for chemical analysis of inorganics and metals, semivolatile organic compounds (SVOCs), and polychlorinated biphenyl (PCB) Aroclor mixtures. Water pH, dissolved oxygen (DO) content, conductivity, and temperature were measured in the field, while water hardness was measured in the laboratory.

In surface water, 18 of 18 analyzed inorganics and metals were detected. No pesticides or PCBs were detected in surface water. Two SVOCs were detected in the water. Bis(2-ethylhexyl)phthalate was detected in two of the primary water samples, the field duplicate, and the field blank sample.

Butylbenzylphthalate was detected in the field duplicate sample. Average values for general water quality parameters were: 12.3°C, 6.5 pH, 4.1 mg/L DO, 46.7 mg/L $CaCO_3$.

In sediments, 13 of 18 analyzed inorganics and metals were detected. Beryllium, cobalt, selenium, silver, and thallium were not detected in the sediment. The pesticides detected in the sediment were 4,4'-DDT, 4,4'-DDD, 4,4'-DDE, and dieldrin. One PCB mixture (Aroclor-1248) was detected in the field duplicate. There was one estimated detection of 4-Methylphenol in a field duplicate sample. However, the target detection limits for SVOCs were not reached, so there is uncertainty in the results. All other SVOCs were reported as non detects. It is not possible to determine if SVOCs are not present, or if they are present but below the achieved detection limit.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Pond G met the field data quality goals for precision, accuracy, completeness, and representativeness, and comparability. The analytical laboratory portion of the investigation at Pond G met the laboratory data quality goals for precision, accuracy, completeness, and representativeness, and comparability. The target detection limits for both pesticides and semivolatile organic compounds were generally not reached in the sediment samples collected at the site. Target levels listed in the QAPP represent dry-weight reporting levels based on a sample with negligible water content. Actual sediment reporting levels varied, depending on the percent water in the respective sediment sample. Additional discussion regarding this issue can be found in Appendix K (Section K1.2.6).

4.2.3 Pond G Summary

Information collected during the 1999 site investigation was used to characterize exposures and ecological effects. However, the data from all supporting site investigations should be evaluated before final conclusions are made about the site. This information is necessary in order to determine if the variability of chemical concentrations has been adequately characterized, so that decisions about the site can be made with confidence. Pond G is a very small pond, about 20 by 80 feet in size so it does not require many samples to characterize the nature and extent of chemical concentrations. The three samples taken during 1999 were located across the pond and are adequate to characterize this small site. The data from the 1992-1993 site investigation only provided one sample location nearer to the northern end.

It is not appropriate to compare the chemical concentrations detected during this earlier site investigation to the 1999 data because the detection limits are quite different. The 1999 investigation used low-level trace metals analysis (USEPA Method 1669) for surface water, so the detection limits were much lower than that in the 1992-1993 surface water data. For example, in the 1992-1993 data, antimony, beryllium, cadmium, chromium, cobalt, copper, lead, nickel, and selenium were undetected, but using the trace metals analysis in 1999, these metals were detected in surface water samples. Sediment metals from the two sampling periods are slightly more comparable because the detection limits did not differ greatly between the two sampling events. However, the comparison should not be quantitative. Sampling in both investigations were single events, and thus, the temporal component of chemical concentrations at this site is not completely understood; however, the three samples taken in 1999 have been validated and are considered to be adequate for characterizing spatial variability of chemical concentrations.

4.3 RICE CREEK WATER MONITORING RESULTS

In order to address deficiencies in previous surface water sampling and analysis data, additional surface water sampling was performed at the creek.

4.3.1 Analytical Chemistry Investigations of Rice Creek Water: 1992-1995

The Tier I ERA report (USACHPPM 1997) evaluated data from the OU2-FS (Montgomery Watson 1994 and 1997), and from the 1994 and 1995 annual monitoring reports. In the Montgomery Watson investigation, seven surface water and seven sediment samples were taken from Rice Creek, on the TCAAP property in 1992. Samples were analyzed for metals, cyanide, and PCBs. Two additional samples were collected in 1993, to verify the 1992 sampling results. The annual monitoring data analyzed surface water from five locations in Rice Creek for metals, TCE (including its breakdown products), and PCBs. The annual monitoring data was used to evaluate concerns over some elevated COPC concentrations in surface water, including lead, mercury, and zinc. Zinc was detected at high levels during the 1992 sampling, but during the 1993 confirmatory sampling this was not the case. The annual monitoring data (1994-1995) was also used to verify zinc concentrations, which may have been erroneous in the 1992 sampling. In addition, in October 1992, Minnesota Pollution Control Agency took two samples each of sediment and surface water in Rice Creek. These samples were analyzed for metals and cyanide.

4.3.2 Rice Creek Water Results: 1999-2000

Sampling locations and measurement results from the 1999 – 2000 water monitoring are summarized in Appendix G. Sampling and analysis were conducted quarterly, following the procedures identified in the Work Plan (USACHPPM 1999), and are also summarized in Appendix K of this report. Four surface water samples were collected each quarter for one year, two of the four samples were collected upstream of the building 103 drainage pipe and two samples were collected downstream of the building 103 drainage pipe. (The upstream samples served as the reference site). Sampling began in September 1999, with an additional 4 samples collected (8 total) in the 4th quarter (June 2000).

Surface water samples were analyzed for aluminum, barium, cadmium, copper, mercury and silver. Water quality parameters analyzed included hardness, total suspended solids (TSS), TOC, total keljhal nitrogen (TKN), and total phosphorus. Water pH, dissolved oxygen (DO) content, conductivity, and temperature were measured in the field.

In upstream surface water, 6 of 6 analyzed metals were detected. Average water quality parameters for upstream samples were as follows: 15.84° C, 8.05 pH, 8.35 mg/L DO, 180 mg/L $CaCO_3$, 2.59 mg/L TKN, 18.2 mg/L TOC, 0.174 mg/L total phosphorus, and 54.7 mg/L TSS.

In downstream surface water, 6 of 6 analyzed metals were detected. Average water quality parameters for downstream samples were as follows: 16.0° C, 8.08 pH, 9.43 mg/L DO, 177.8 mg/L $CaCO_3$, 2.65 mg/L TKN, 17.9 mg/L TOC, 0.176 mg/L total phosphorus, and 52.0 mg/L TSS.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Rice Creek met the field data quality goals for precision, accuracy, completeness, and representativeness, and comparability. During the second quarter sampling event, the 4th sampling location was moved for the remainder of the study in order to locate the sampling site closer to the Bldg. 103 discharge point.

The analytical laboratory portion of the investigation at Rice Creek met the laboratory data quality goals for target detection limits, precision, accuracy, completeness, and representativeness, and comparability.

4.3.3 Summary of Rice Creek Water Data

The 1999-2000 sampling effort presented in Section 4.3.2 was based on the conclusions and recommendations of the Tier I ERA report (USACHPPM 1997). Data and information used in the 1997

Tier I assessment, and data collected during the 1999-2000 water monitoring activities were evaluated to determine their quality and utility in order to support confident conclusions about the site. The 1992-1995 data examined all preliminary COPCs (metals, PCBs), in a spatial context. However, the 1999-2000 water monitoring data looked at the selected COPCs (aluminum, barium, cadmium, copper, mercury, and silver), as was determined by the conclusions of the Tier I assessment and presented in the Work Plan (USACHPPM 1999). The 1999-2000 sampling focused on the temporal variability of COPC concentrations.

Seven surface water and sediment samples were collected during the 1992 Montgomery Watson investigation (used in the OU2-FS, Montgomery Watson 1994). Two of these samples were collected where Rice Creek enters TCAAP, and the other five were collected in the creek corridor as it travels through TCAAP (see Figure 4a in Appendix Q). Two samples of co-located sediment and surface water were taken during the October 1992 Minnesota Pollution Control Agency sampling, in which one sample was taken at about 30 feet downstream of the Site K outfall and one was taken about 100 feet upstream of the upper Site K outfall. Also, five samples per year that were collected in 1994 and 1995 as part of the TCAAP annual monitoring program were evaluated in the Tier I ERA. One of these samples was collected annually in the creek as it enters TCAAP and one was collected annually from the creek as it leaves TCAAP. The other three samples were collected from various points in the creek as it runs through TCAAP. During the 1999-2000 water quality investigations by USACHPPM (see Figure G1 in Appendix G), surface water samples were collected quarterly from four locations in the creek. The 1999-2000 water samples (N=10) were the only data collected quarterly.

Generally, one would want to compare the chemical concentrations from all of these sampling events in order to provide a comprehensive evaluation of the spatial and temporal variability of chemical presence. However, it is not appropriate to compare the chemical concentrations detected during the 1992 Montgomery Watson sampling and 1994-1995 annual monitoring, to the 1999-2000 data because the detection limits are quite different. The 1999-2000 investigation used low-level trace metals analysis (USEPA Method 1669) for surface water, so the detection limits were much lower than that in the 1992-1993 surface water data, and the annual monitoring data. As an example, for the metals analyzed in the 1992-1993 Rice Creek investigation that correspond with those metals evaluated in the 1999-2000 study, cadmium, copper, and mercury were undetected. (In the Tier II these three metals were re-evaluated, despite their previous non-detection, because the previous detection limits were not below the screening benchmarks). However, during the 1994, 1995, and 1999-2000 investigations, cadmium and copper were detected. Also, using the trace metals analysis method in 1999-2000, mercury was detected in surface water samples. In addition, the Tier I assessment did not evaluate a comparison between upstream and downstream sites in Rice Creek. However, the 1999-2000 sampling was designed to look at these reaches for comparative purposes.

The Site K outfall is the suspected source of chemical input to Rice Creek, and therefore, the sampling events were focused on comparing reaches downstream to reaches upstream of this area. The 1999-2000 data reported concentrations of chemicals that were achieved by using an analytical method that could yield data quality sufficient for risk managers to make confident decisions (i.e., ensuring low detection limits). Sampling was performed as agreed to in the Work Plan (USACHPPM 1999), with minor modifications that were documented in monthly QA reports. Sampling locations and the number of samples were chosen based on what was determined would be statistically sufficient information to determine the variability of chemical concentrations. These data (1999-2000) were validated and are considered to adequately characterize the temporal variability of COPC (aluminum, barium, cadmium, copper, mercury, and silver) concentrations at Rice Creek.

4.4 SUNFISH LAKE WATER MONITORING RESULTS

In order to address deficiencies in previous surface water sampling and analysis data, additional surface water sampling was performed at the lake.

4.4.1 Analytical Chemistry Investigations of Sunfish Lake Water: 1992-1993

The Tier I ERA report (USACHPPM 1997) evaluated the data that was presented in the OU2-FS (Montgomery Watson 1994 and 1997). That data consisted of ten samples collected in 1992 and two confirmatory samples collected in 1993. The samples were analyzed for metals, cyanide, PCBs, and VOCs. There are no annual monitoring sampling locations at Sunfish Lake, so the only surface water data that was used in the Tier I report was the 1992-1993 Montgomery Watson data.

4.4.2 Sunfish Lake Water Results: 1999-2000

Sampling locations and measurement results from the 1999 – 2000 water monitoring are summarized in Appendix H. Sampling and analysis were conducted quarterly following the procedures identified in the Work Plan (USACHPPM 1999), and summarized in Appendix K of this report. Three surface water samples were collected each quarter for one year, beginning in September 1999.

Surface water samples were analyzed for aluminum, barium, cadmium, copper, mercury, silver and zinc. Water quality parameters analyzed included hardness, total suspended solids (TSS), TOC, total kejeldahl nitrogen (TKN), and total phosphorus. Water pH, dissolved oxygen (DO) content, conductivity, and temperature were measured in the field.

In surface water 7 of 7 analyzed metals were detected. Average water quality parameters were: 14.5°C, 8.27 pH, 7.88 mg/L DO, 67.50 mg/L $CaCO_3$, 0.728 mg/L TKN, 6.77 mg/L TOC, 0.0650 mg/L total phosphorus, and 2.67 mg/L TSS.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Sunfish Lake met the field data quality goals for precision and completeness. Our best professional judgment was that field accuracy was not compromised during the first quarter sampling event, when it was discovered that the field blank sample contained zinc at a concentration higher that detected in the other samples. This is further discussed in Appendix K (Section K1.1.2). Field representativeness and comparability may be slightly compromised since the fourth quarter sampling event (June 2000) occurred during a rapid warming of the lake. The temperature profile conducted at Sunfish Lake revealed that the entire water column, from the surface to the maximum depth of 5 ft, was a thermocline as defined in the QAPP, since the temperature varied more than 1°C per meter. However, there was no discernable top layer or bottom layer, and the lake was very shallow. Therefore the thermal structure fit neither the definition of well mixed nor stratified. Since there was no evidence of a distinct top layer or bottom layer, only surface water was collected.

The analytical laboratory portion of the investigation at Sunfish Lake met the laboratory data quality goals for target detection limits, precision, accuracy, completeness, and representativeness, and comparability.

Appendix H also contains the sediment data from the Tier I ERA.

4.4.3 Summary of Sunfish Lake Water Data

The 1999-2000 sampling effort presented in Section 4.4.2 was based on the conclusions and recommendations of the Tier I ERA report (USACHPPM 1997). Data and information used in the 1997 Tier I assessment, and data collected during the 1999-2000 water monitoring activities were evaluated to determine their quality and utility in order to support confident conclusions about the site. The 1992-1993 data examined all preliminary COPCs (metals, PCBs, pesticides, and VOCs), in a spatial context. However, the 1999-2000 water monitoring data looked at the selected COPCs (aluminum, barium, cadmium, copper, mercury, silver, and zinc), as was determined by the conclusions of the Tier I assessment and presented in the Work Plan (USACHPPM 1999). The 1999-2000 sampling focused on characterizing the temporal variability of COPC concentrations.

During the 1992 Montgomery Watson investigation (used in the OU2-FS, Montgomery Watson 1994) ten co-located surface water and sediment samples were collected. Seven samples were collected in Sunfish Lake. Five of those samples were located offshore from the potential source area (Area H-1 and the unpermitted landfill, which are near the center of the northwestern shoreline). The other two samples were located in the southern end of the lake. Outside of the lake, three addition samples were collected from the outlet ditch at the northern tip of Sunfish Lake. This ditch leads to Marsden Lake. In 1993, two confirmatory surface water samples were collected, one in the southern portion of the lake and one offshore from the unpermitted landfill area. (See Appendix Q, Figure 3). Surface water samples were collected quarterly from three locations in the lake during the 1999-2000 water quality investigation by USACHPPM (see Figure H1 in Appendix H). The 1999-2000 water samples were the only data collected quarterly, resulting in a total of 12 samples.

Quantitative comparisons of chemical concentrations from the sampling events discussed above should not be done. It is not appropriate to compare the chemical concentrations detected during the 1992 Montgomery Watson sampling to the 1999-2000 data because the detection limits are quite different. The 1999-2000 investigation used low-level trace metals analysis (USEPA Method 1669) for surface water, so the detection limits were much lower than that in the 1992-1993 surface water data, and the annual monitoring data. As an example, for the metals analyzed in the 1992-1993 Sunfish Lake investigation that correspond with those metals evaluated in the 1999-2000 study, cadmium, copper, mercury, and silver were undetected. (In the Tier II these metals were re-evaluated, despite their previous non detection, because the previous detection limits were not below the screening benchmarks). However, during the 1999-2000 investigations, the detection limits were below screening benchmarks.

Because the main suspected source of chemical input to Sunfish Lake was Area H-1 and the unpermitted landfill contained within its boundary, the majority of surface water samples collected during the 1992-1993 investigation were taken near that area. Up-gradient samples (in the southern area of the lake) were taken, as well as down gradient samples (in the northern portion of the lake). Twelve samples were collected in all in 1992-1993, with the two 1993 confirmatory samples used to verify concentrations of zinc and aluminum that appeared to be erroneous in the 1992 analysis. That investigation was used to characterize the spatial variability of chemical concentrations in Sunfish Lake. The 1999-2000 data reported concentrations of chemicals that were achieved by using an analytical method that could yield data quality sufficient for risk managers to make confident decisions (i.e., ensuring low detection limits). Sampling was performed as agreed to in the Work Plan (USACHPPM 1999), with minor modifications that were documented in monthly QA reports. Sampling locations and the number of samples were chosen based on what was determined would be statistically sufficient information to determine the variability of chemical concentrations. These data (1999-2000) were validated and are considered to adequately characterize the temporal variability of COPC (aluminum, barium, cadmium, copper, mercury, silver, and zinc) concentrations at Sunfish Lake.

4.5 ROUND LAKE WATER MONITORING RESULTS

In order to address deficiencies in previous surface water sampling and analysis data, additional surface water sampling was performed at the lake.

4.5.1 Analytical Chemistry Investigations of Round Lake Water: 1992-1995

The Tier I ERA report (USACHPPM 1997) evaluated data from the OU2-FS (Montgomery Watson 1994 and 1997), and from the 1994 and 1995 annual monitoring reports. In the Montgomery Watson investigation, twenty co-located surface water and sediment samples were collected from Round Lake, in 1992. Samples were analyzed for metals, cyanide, and PCBs. Two additional samples were collected in 1993, to verify the 1992 sampling results. The 1994 and 1995 annual monitoring data analyzed for metals, PCBs, and TCE (and its breakdown products) at two locations in the lake. The data were used to evaluate concerns over some elevated COPC concentrations in surface water, including zinc. Zinc was detected at high levels during the 1992 sampling, but during the 1993 confirmatory sampling this was not the case. The annual monitoring data (1994-1995) was also used to verify the zinc concentrations, which may have been erroneous in the 1992 sampling.

4.5.2 Round Lake Water Results: 1999-2000

Sampling locations and measurement results from the 1999 – 2000 water monitoring are summarized in Appendix I. Sampling and analysis were conducted quarterly, following the procedures identified in the Work Plan (USACHPPM 1999), and are also summarized in Appendix K of this report. Five surface water samples were collected each quarter for one year, beginning in September 1999.

Surface water samples were analyzed for barium, cadmium, copper, mercury, silver and zinc. Water quality parameters analyzed included hardness, total suspended solids (TSS), TOC, total kejeldahl nitrogen (TKN), and total phosphorus. Water pH, dissolved oxygen (DO) content, conductivity, and temperature were measured in the field.

In surface water 5 of 6 analyzed metals were detected. Silver was not detected in any primary samples from Round Lake. Average water quality parameters were: 13.9°C, 8.29 pH, 9.60 mg/L DO, 156 mg/L $CaCO_3$, 1.13 mg/L TKN, 12.5 mg/L TOC, 0.0540 mg/L total phosphorus, and 10.79 mg/L TSS.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Round Lake met the field data quality goals for precision, accuracy, completeness, and representativeness, and comparability. The analytical laboratory portion of the investigation at Round Lake met the laboratory data quality goals for target detection limits, precision, accuracy, completeness, and representativeness, and comparability.

Appendix I also contains the surface sediment data from the Tier I ERA. The appendix contains sediment data for various sediment depth intervals, which were obtained during the OU-2 Feasibility Study (Montgomery Watson 1997).

4.5.3 Summary of Round Lake Water Data

The 1999-2000 sampling effort presented in Section 4.5.2 was based on the conclusions and recommendations of the Tier I ERA report (USACHPPM 1997). Data and information used in the 1997 Tier I assessment, and data collected during the 1999-2000 water monitoring activities were evaluated to determine their quality and utility in order to support confident conclusions about the site. The 1992-1995 data examined preliminary COPCs (metals, pesticides, and PCBs,) in a spatial context. However, the 1999-2000 water monitoring data looked at the temporal variability of the selected COPCs (barium, cadmium, copper, mercury, silver, and zinc), as was determined by the conclusions of the Tier I assessment and presented in the Work Plan (USACHPPM 1999). Preliminary studies were performed at Round Lake in 1995 (USACHPPM 1998a; USACHPPM 1998b), and in 1999 (see Appendix M); however, surface water chemical concentrations were not analyzed during these studies.

In the 1992 Montgomery Watson investigation, twenty samples were collected from the lake. Sixteen of the samples were collected from the northern end of the lake. Three samples were collected across the center of the lake and two samples were collected in the southern portion of the lake. In 1993, two confirmatory samples were collected, but the sampling locations were not identified in the OU2-FS. The 1994 and 1995 annual monitoring data was collected from one sampling location in the northern portion of Round Lake, near the Line E outfall. The location was an NPDES permit monitoring station where quarterly data were collected. Studies conducted in the early 1990's showed differences in some biological data (macroinvertebrate community indices) so it was determined that more sampling was necessary in the northern part of Round Lake because that area is associated with the main point of entry of potential TCAAP contaminants (the line E outfall, as well as a second outfall). A total of 20 samples were collected there during 1999-2000 (see Appendix I, Figure I1).

Quantitative comparisons of chemical concentrations from the sampling events discussed above should not be done. It is not appropriate to compare the chemical concentrations detected during the 1992-1993 Montgomery Watson sampling and the 1994-1995 annual monitoring data, to the 1999-2000 data because the detection limits are quite different. The 1999-2000 investigation used low-level trace metals analysis (USEPA Method 1669) for surface water, so the detection limits were much lower than that in the 1992-1993 surface water data, and the annual monitoring data. As an example, for the metals analyzed in the 1992-1993 Round Lake investigation that correspond with those metals evaluated in the 1999-2000 study, cadmium, copper, and silver were undetected. (In the Tier II these metals were re-evaluated, despite their previous non-detection, because the previous detection limits were not below the screening benchmarks). However, during the 1999-2000 investigations, the detection limits were below screening benchmarks, and cadmium and copper were detected.

Because the main suspected source of chemical input to Round Lake was through the storm sewer outfall from Line E in the northern section of the lake, the majority of surface water samples collected during the 1992-1993, 1994-1995, and 1999-2000 investigation were taken near that area. Other samples were collected from the central and southern portions of the lake to determine the spatial variability of chemical concentrations in surface water and sediment. Twenty-two samples were collected in all in 1992-1993, with the two 1993 confirmatory samples used to verify concentrations of zinc and aluminum that appeared to be erroneous in the 1992 analysis. This investigation was used to characterize the spatial variability of chemical concentrations in Round Lake. The 1999-2000 data provided concentrations of chemicals that were achieved by using an analytical method that could yield data quality sufficient for risk managers to make confident decisions. Sampling was performed as agreed to in the Work Plan (USACHPPM 1999), with minor modifications that were documented in monthly QA reports. Sampling locations and the number of samples were chosen based on what was determined would be statistically sufficient information to determine the temporal variability of chemical concentrations. These data (1999-2000) were validated and are considered to adequately characterize the temporal variability of COPC (barium, cadmium, copper, mercury, silver, and zinc) concentrations at Round Lake.

4.6 SNAIL LAKE SITE INVESTIGATION AND WATER MONITORING RESULTS

Snail Lake was used as the reference site for site characterization at the Marsden Lake sites and water monitoring at Sunfish and Round Lakes (USACHPPM 1999). Sampling locations and measurement results from the 1999 – 2000 sampling events are summarized in Appendix J. Sampling and analysis were conducted following the procedures identified in the Work Plan (USACHPPM 1999), and are also summarized in Appendix K of this report.

4.6.1 Reference Site Selection

Based on location and land use, Snail Lake is assumed to be free from installation-related contamination. It has been used as a reference/comparison site for the TCAAP lakes for some time, beginning in 1993. The lake was used as a comparison lake for Round and Sunfish Lakes during the 1993 benthic macroinvertebrate evaluation included in the Tier I ecological risk assessment (USACHPPM 1997). The same investigators used Snail Lake again in June 1995, as a comparison site for the sediment toxicity tests conducted in the southern portion of Round Lake (USACHPPM 1998a). Snail Lake was again used as a comparison site in September 1995, for the sediment bioavailability study at Round and Sunfish Lakes (USACHPPM 1998b).

Snail Lake was selected as the reference site for planned Tier II risk assessment evaluations for Round Lake, Sunfish Lake, and Marsden Lake (USACHPPM 1999). During the development of the work plan (at the 5 May 1998 comments resolution meeting for the amphibian study) the issue of reference site selection was discussed. On that day, after a site visit to Snail Lake and Turtle Lake (an alternative comparison site), it was determined, by consensus, that Snail Lake could be used as the reference site for the amphibian study evaluating Round, Sunfish, and Marsden Lakes.

In the spring of 1998, the land surrounding Snail Lake included residential, mowed lawn and some forested areas primarily on the eastern edge. A potential problem discussed at the May meeting was the possibility of effects of road construction to Snail Lake. It was determined that construction along Route 96 near the northern end of Snail Lake did not appear to have caused any erosion or other disturbances to the site. Adequate grassy and cattail habitat existed along the north and south end of the Lake. Amphibian survey fieldwork at Snail Lake was conducted at these locations. The east and most of the west bank of Snail Lake are residential with rocky beaches and mowed lawns. These habitats were not expected to be conducive for amphibians. Snail Lake is visually similar to the study sites in vegetation, and habitat structure. Snail Lake has a cattail portion that is visually similar to cattail portions in the study sites at Marsden Lake, Sunfish Lake, and Round Lake. Based on location, the reference site is assumed free of military-related contamination. Anglers, boaters, and swimmers use Snail Lake for recreation which may disturb wildlife and amphibian species. However, we used the north and south ends of the Lake for our reference areas when performing the amphibian study. The north end is dominated by cattails with some forested area. At the south end of the lake there is a recreational park with a parking lot, swimming area, boat dock, some mowed grass, a small wetland area with cattails. We also surveyed the southeast side where it is mostly forested and relatively undisturbed.

The technical uncertainties of using Snail Lake as a reference site for comparison of chemical concentration data are discussed in the report where such comparisons are made (see Section 5.3).

4.6.2 Analytical Chemistry Investigations of Snail Lake: 1995-1999

The Tier I ERA Report did not provide a comparison of surface water samples from TCAAP sites to a reference area. Therefore, no analytical chemistry data on surface water in Snail Lake prior to the first investigations performed in support of Tier II work are available (i.e., prior to 1999). In 1995, two studies were conducted using Snail Lake as a reference site. Three water and sediment samples were collected from the southern portion of Snail Lake for the sediment toxicity evaluation of Round Lake (USACHPPM 1998a). The water samples in this study only evaluated water quality parameters (e.g., hardness, dissolved oxygen) and not analytical chemistry. Two sediment samples were also collected at the southern end of the lake in support of the bioavailability study (USACHPPM 1998b). Chemical analysis was not performed on any media samples collected for the amphibian study.

In 1999 and 2000 sampling was conducted at Snail Lake to provide a reference site for comparison in the Tier II ERA. Two different sampling activities were performed. In September of 1999, ten samples were collected throughout Snail Lake with the intent to use the sample data for comparisons with samples collected in September 1999 from Marsden Lake. In 2000, three samples were collected from the north, south, and central portions of Snail Lake (locations SL-01, SL-06, and SL-09) during January, April, and June. In total, nine samples in 2000 and 3 samples in September 1999 were collected and used to compare the temporal variability of Round Lake and Sunfish Lake to this chosen reference site.

4.6.3 Snail Lake Site Characterization Water Results: 1999

Ten primary samples of surface water and surface sediment were collected from Snail Lake in September 1999 for chemical analysis of inorganics and metals, semivolatile organic compounds (SVOCs), and polychlorinated biphenyl (PCB) Aroclor mixtures. Total organic carbon (TOC) was also measured in the sediments. Water measurements included hardness, total suspended solids (TSS), TOC, total kejeldahl nitrogen (TKN), and total phosphorus. Water pH, dissolved oxygen (DO) content, conductivity, and temperature were measured in the field.

In surface water, 15 of 18 analyzed metals were detected. Beryllium, cadmium, and thallium were not detected in the water. No pesticides, PCBs, or SVOCs were detected in surface water. Average values for general water quality parameters were: 17.4° C, 8.7 pH, 9.7 mg/L DO, 101.2 mg/L $CaCO_3$, 0.80 mg/L TKN, 9 mg/L TOC, 0.02 mg/L total phosphorus, and <1 mg/L TSS.

In sediments, 15 of 18 analyzed metals were detected. Beryllium, silver, and thallium were not detected in the sediment. No PCBs or SVOCs were detected in the sediments. However, four pesticides were detected: dichlorodiphenyltrichloroethane (4,4'-DDT), delta-BHC, heptachlor, and lindane. Out of ten samples, 4,4'-DDT, delta-BHC, and heptachlor were detected at one location each (note that heptachlor was also detected in the field duplicate). Lindane was detected in 2 samples out of ten. Average sediment TOC was 2.35%.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Snail Lake met the field data quality goals for precision, accuracy, completeness, and representativeness, and comparability. The analytical laboratory portion of the investigation at Snail Lake met the laboratory data quality goals for target detection limits, precision, accuracy, completeness, and representativeness, and comparability.

Some sample holding times were missed due to various reasons and did cause several samples to be rejected. The Snail Lake mercury analyses (performed on dried samples at 60°C) were done after realizing that the requested reporting levels were not achieved using wet sample weights. The holding time for mercury had been met on the initial wet weight analyses. For consistency purposes, the two mercury analyses sets (dry and wet weights) were performed for the remaining (quarterly) data. As more wet sample weights were measured, and the resulting reporting levels closely matched each other, the results indicated that negligible or no mercury was lost during the 60°C drying step. Refer to Appendix K (Section K1.2.4) for additional discussion of this topic.

There was one sample (SC-SL-FB-01) from the site that had rejected data. In surface water from sample SC-SL-FB-01, the PCBs as well as the following pesticides were rejected due to extraction holding times being grossly exceeded: Aldrin, Alpha-BHC, Beta-BHC, Delta-BHC, Lindane, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, Dieldrin, Endosulfan I, Endosulfan II, Endosulfan Sulfate, Endrin, Endrin Aldehyde, Heptachlor, Heptachlor Epoxide, Methoxychlor, Chlordane, and Toxaphene. In sediment from SC-SL-FB-01, copper was rejected because the blank had a negative value greater than two times the instrument detection limit (IDL) and the matrix spike recovery was above the upper control limit. Since the sample (SC-SL-FB-01) was a field blank, the rejected data does not significantly affect the results because blank data was not used during statistical analysis.

4.6.4 Snail Lake Water Monitoring Results: 1999-2000

Three primary samples of surface water were collected quarterly for one year, beginning in September 1999. Additionally, during the 3rd and 4th quarters, 3 bottom water samples were collected in addition to the surface water samples because the temperature profile of the water during these quarters showed the possibility of stratification within the lake. All samples were analyzed for the following metals: aluminum, barium, cadmium, copper, mercury, silver, and zinc. Water quality measurements also included hardness, TSS, TOC, TKN, and phosphate. Water pH, DO, conductivity and temperature were measured in the field.

In surface water 7 of 7 analyzed metals were detected, with the exception of cadmium not being detected in the first quarter. The average water quality parameters were: 12.7°C, 8.4 pH, 10.35 mg/L DO, 122.5 mg/L $CaCO_3$, 0.685 mg/L TKN, 8.98 mg/L TOC, 0.0233 mg/L PO_4, and 1.33 mg/L TSS.

Section 4.7 presents the data quality assessment. The field portion of the investigation at Snail Lake met the field data quality goals for precision, accuracy, completeness, and representativeness, and comparability. The analytical laboratory portion of the investigation at Snail Lake met the laboratory data quality goals for target detection limits, precision, accuracy, completeness, and representativeness, and comparability.

4.7 SUMMARY OF THE DATA QUALITY ASSESSMENT

Data quality was determined through the assessment of the project data quality objectives (DQOs) and the validation of field and laboratory data. Precision, accuracy, completeness, representativeness and comparability for the field sampling and the laboratory analyses were reviewed, calculated (where applicable) and compared to the QAPP target values. There were several QAPP modifications identified in the monthly Project QA Reports. These modifications were reported to the Project QA Manager and documented. For the most part, the minor deviations and modifications to the QAPP did not affect the outcome of the reported data. The only deviation that resulted in a change in the way data was reported was a problem regarding field representativeness. Surface water and sediment samples ML-13 and ML-15 were unable to be collected at the intended locations in Marsden Lake North due to problems with wind direction and moving cattail mats. After an evaluation of the analytical data it was decided to combine the data from those two samples with the data from Marsden Lake South (see sections K1.1.3 and K1.1.4). There were also several minor deviations to the QAPP that were identified in the monthly Project QA Reports.

The Project Quality Assurance (QA) Manager conducted a 100 % field data validation. The field data for all sampling events are considered to be technically sound and defensible. The Science Applications International Corporation (SAIC) performed a 100 % external laboratory data validation on the September 1999 and January 2000 sampling events. The data sets for these sampling events were considered valid by SAIC. External laboratory data validation on the April and June 2000 sampling events was not conducted. In summary, external data validation was performed on 100 % of the analytical chemistry data for the September 1999 site characterization study areas (i.e., Marsden Lake, Pond G, and Snail Lake), nearly 25 % of the analytical chemistry data for the water monitoring study areas (i.e., Rice Creek, Round Lake, Sunfish Lake, and Snail Lake), and 100 % of the field-collected data.

The external analytical chemistry data validation rate exceeded the QAPP required rate of 10 % and the external field data validation rate met the QAPP required rate of 100 %.

There were some aspects of QA that were not met. As shown in various QA reports, the sub-contracting laboratories did not receive a full copy of the QAPP, which is the document that defines the requirements for the project. The laboratories were however provided with lists of target compounds and required reporting levels for the project. Initially this caused some confusion with detection levels for pesticides and PCBs, but Trimatrix Laboratories was able to amend their final analytical report showing that the detection levels had been lowered considerably. Since they did not receive a full copy of the QAPP, sub-contracting laboratories were not aware of the requirements to submit monthly QA reports to the Project QA manager. The QAPP required that method audits for each type of analysis performed in the USACHPPM Directorate of Laboratory Sciences (DLS) be conducted at least once during the project. This was not done. This requirement was not identified in the QAAP for the sub-contracting laboratories. Some sample holding times were missed due to various reasons and did cause several samples to be rejected; however, the laboratory completeness was still well over the QAPP target of 95 %.

5. SELECTION OF CHEMICALS OF CONCERN

5.1 COC SELECTION PROCESS

In order to clarify the process used to develop the final list of chemicals of concern (COCs), the following diagram (Exhibit 5-1) was developed after the initial draft of the Tier II risk assessment report (USACHPPM 2001). All references to COCs have been revised to be consistent with this characterization.

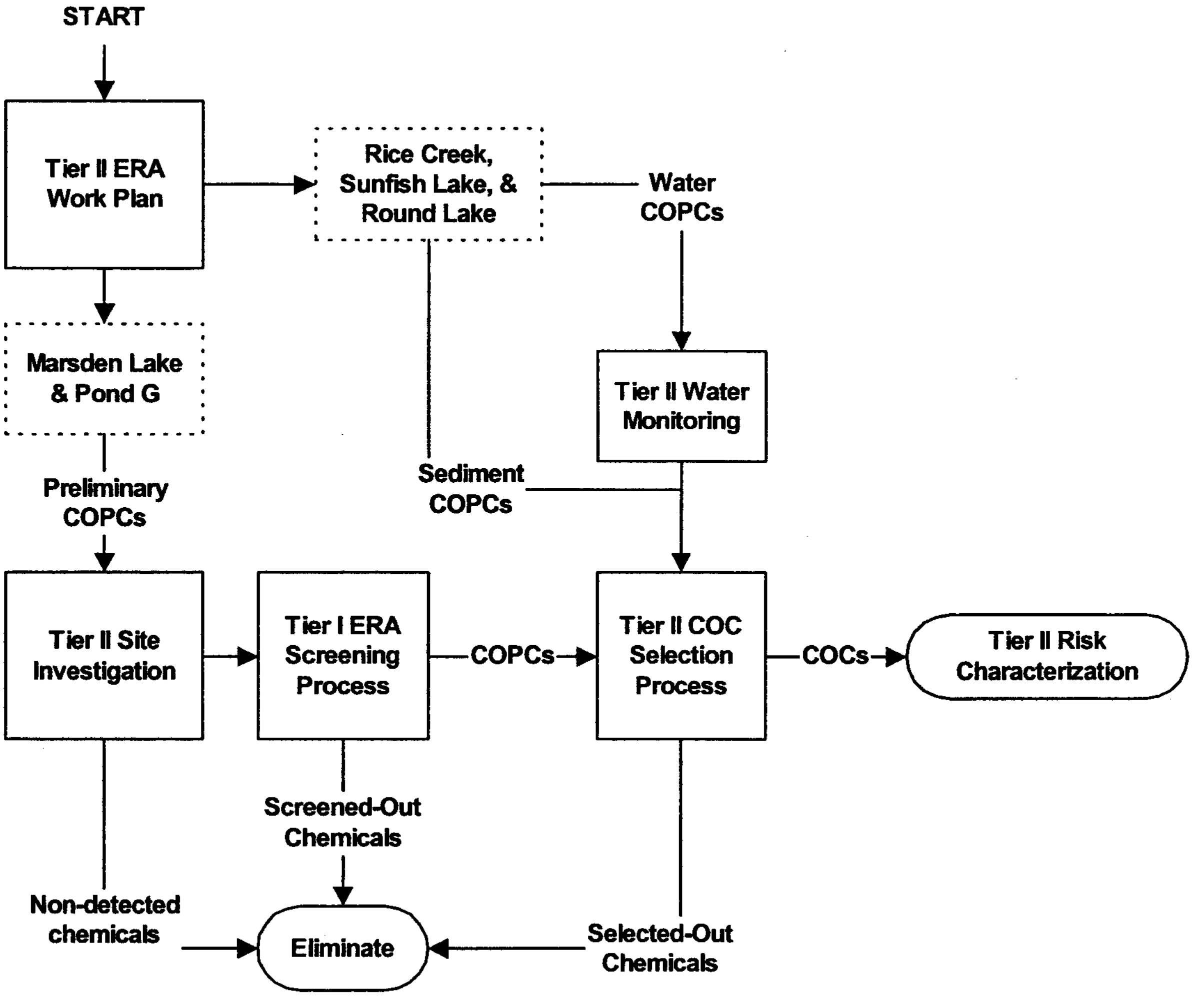
Under this scheme, the chemicals analyzed for during site investigation sampling prior to the application of the Tier I ERA "screening" process are considered *preliminary* chemicals of potential concern (preliminary COPCs). Those chemicals detected during this early stage and identified by the Tier I process as needing further investigation, because under conservative "screening" assumptions they may contribute to unacceptable ecological risks, are considered to be COPCs. Therefore, COPCs are outputs of the Tier I process.

The Tier I screening process was completed for Round Lake, Sunfish Lake, and Rice Creek in 1997. The nature and extent of contamination at those study areas was sufficiently characterized, therefore; the remaining Tier II investigations could be focused upon specific questions related to the selected COPCs (USACHPPM 1997). A summary of the Tier I findings for these three study sites was presented in Section 3 (Problem Formulation).

The Tier I screening process was not completed for the Marsden Lake sites and Pond G prior to the initiation of the Tier II phase of the risk assessment because the study areas were not sufficiently characterized (i.e., too few sampling data) for an adequate Tier I screening evaluation (USACHPPM 1997 and 1999). To address this limitation additional site investigations (i.e., water and sediment sampling and chemical analyses) were conducted at Marsden Lake (North), Marsden Lake (South) and Pond G (USACHPPM 1999). The data obtained from these investigations proceeded through the same Tier I screening process that was finalized in 1997 (USACHPPM 1997), in order to complete that phase of the risk assessment for these sites and identify COPCs for further Tier II risk evaluation. The Tier I reassessment for Marsden Lake (North), Marsden Lake (South) and Pond G is presented in this report in Section 5.2.

Once COPCs are identified for each of the six sites from the Tier I screening process, chemicals of concern (COCs) were selected using evaluation methods described in Section 5.3. Chemicals of concern (COCs) are those chemicals that are a subset of the COPCs that may not be naturally occurring or a result of non-site sources and are present at sufficient frequency, concentration, and location to pose potential ecological risks. The selected COCs were evaluated in the remaining phases of the risk assessment (Sections 6, 7, and 8).

Exhibit 5-1. Tier II COC Selection Process



<u>COPCs</u> (Chemicals of Potential Concern). Chemicals that may contribute to an excess risk.

<u>Preliminary COPCs</u>. Chemicals that are originally evaluated during site investigations.

<u>COCs</u> (Chemicals of Concern). TCAAP-released chemicals that may pose an excess risk.

5.2 TIER I RE-ASSESSMENT FOR MARSDEN LAKE AND POND G

As stated in the Problem Formulation Section for these sites (Sections 3.1 and 3.2), a previous Tier I Ecological Risk Assessment was conducted. The following two paragraphs summarize the previous report's findings for Marsden Lake and Pond G. (See Appendix Q for the full report.)

The 1997 Tier I assessment for Marsden Lake was based on five samples (co-located surface water and sediment) and concluded that the most significant potential for risks were due to detections of pesticides and zinc in the sediments. The report also concluded that aluminum, barium, and zinc in surface water would have the potential to cause toxicity. The surface water data for zinc were not as robust as they could be — for data collected from 1993 to 1995 did not support significant risks from zinc originally identified by the 1992 data set. The assessment remained inconclusive with regards to potential risk from copper, mercury, and silver in the surface waters. This was due to method detection limits that were inadequate for screening purposes.

The 1997 Tier I assessment for Pond G was based upon only one sample and concluded that all ecological receptors would be expected to exhibit unacceptable risks when exposed to Pond G. In surface water, barium, and zinc would be consistently toxic. In the sediments, copper, lead, zinc, PCB 1254, and p,p-DDT metabolites would likely cause toxic conditions for sediment organisms. The assessment remained inconclusive with regards to potential risk from copper, mercury, and silver in the surface waters. This is due to method detection limits that were inadequate for screening purposes.

Due to limited sampling data at Marsden Lake and Pond G, a re-assessment was recommended using additional sampling data designed to better characterize the nature and extent of possible contamination and to identify chemicals of potential concern (COPCs). See sections 4.1.2.1, 4.1.3.3, and 4.2.2 for details on the chemicals that were analyzed for. The Tier I re-assessment included the following steps.

1. Chemicals not detected were eliminated from further consideration
2. Data quality review and comparison with laboratory blanks
3. Comparison to background sediment data
4. Toxicity screening and development of conservative hazard quotients

The implementation of each of these steps is described below. A summary of COPC selection process for each site is provided in Section 5.2.5 and Tables A1 – A3 (in Appendix A).

5.2.1 Step 1 — Chemicals Not Detected

Chemicals analyzed for, but not detected in any sample of a site medium are not considered to be COPC candidates. To be selected, a chemical must be found in at least one sample of the environmental medium at a reported concentration. To be included, a chemical must have concentrations above the sample quantitation limit (SQL), which is the lowest level that a chemical may be accurately and reproducibly quantified, or have concentrations that are quantified but estimated (i.e., less than the SQL and labeled with a "J" qualifier). Where samples have an associated duplicate analysis, the higher value of the two results was used.

Care must be taken when evaluating analytical results in which very high detection limits are attained, since a nondetection may mask the presence of a chemical at a concentration less than the quantitation limit. In such cases, chemicals may need to be assessed qualitatively if they are present in other site media. Detection limits also need to be evaluated with respect to toxicity benchmarks.

Other than the comparability of the detection limits to appropriate toxicity benchmarks, this criterion is self-evident. The Quality Assurance Project Plan (QAPP) defined toxicity benchmarks (termed risk-based action levels in the QAPP) and project method reporting levels (i.e., the target detection limits) (USACHPPM 1999). The risk-based action levels were developed for use as benchmarks for the development of target detection limits and selection of the most appropriate analytical methods for the site investigations. The QAPP identified that in some cases the target detection limit was higher than the risk-based level (e.g., for some pesticides, polychlorinated biphenyls (PCBs), and semivolatile compounds). In these cases, the implications for uncertainty in estimating presence of contamination, and therefore risk, were considered minor since the analytical methods selected were considered to be consistent with professional practice and therefore adequate for the purposes of the study. The goal of the analyses then, was to meet the target detection limits defined in the QAPP, and not necessarily the toxicity benchmarks.

5.2.1.1 Step 1: Marsden Lake (North)

Surface water and sediment samples were analyzed for metals, organochlorine (OC) pesticides, polychlorinated biphenyls (PCBs), and semivolatile organic compounds (SVOCs) according to methods defined in the QAPP (USACHPPM 1999).

Based on no detection in surface water samples, beryllium, selenium, OC pesticides, PCBs, and SVOCs (except for bis(2-ethylhexyl)phthalate detected in 2 samples) were eliminated from consideration as surface water-COPCs. Based on no detection in sediment samples, beryllium, cobalt, silver, thallium, and PCBs were eliminated from consideration as sediment-COPCs. See Table A1.

The achieved detection limits for SVOCs in sediment were very high relative to the target detection limits due to sample matrix problems. Also, the achieved detection limit for most OC pesticides in sediment (0.05 mg/kg) was 5 times greater than the target detection limit (0.01 mg/kg). Therefore, even though SVOCs and OC pesticides were not detected in any sediment sample, they could not be eliminated from further consideration as sediment-COPCs based on this criterion alone.

5.2.1.2 Step 1: Marsden Lake (South)

Surface water and sediment samples were analyzed for metals, organochlorine (OC) pesticides, polychlorinated biphenyls (PCBs), and semivolatile organic compounds (SVOCs) according to methods defined in the QAPP (USACHPPM 1999).

Based on no detection in surface water samples, beryllium, selenium, OC pesticides, PCBs, and SVOCs were eliminated from consideration as surface water-COPCs. Based on no detection in sediment samples, beryllium, silver, thallium, and PCBs were eliminated for consideration as sediment-COPCs. See Table A2.

The achieved detection limits for SVOCs in sediment were very high relative to the target detection limits due to sample matrix problems. Also, the detection limit for most OC pesticides in sediment (0.08 mg/kg) was 8 times greater than the target detection limit (0.01 mg/kg). There was one detection of 4,4'-DDD in a sediment duplicate sample (ML-13). The value (0.095 mg/kg) was less than five times the achieved detection limit (0.08 mg/kg), but because the target detection limit was not met (0.01 mg/kg), 4,4'-DDD cannot be eliminated as a sediment COPC based on this criterion alone.

5.2.1.3 Step 1: Pond G

Surface water and sediment samples were analyzed for metals, organochlorine (OC) pesticides, polychlorinated biphenyls (PCBs), and semivolatile organic compounds (SVOCs) according to methods defined in the QAPP (USACHPPM 1999).

Based on no detection in surface water samples, mercury, silver, OC pesticides, PCBs, and all SVOCs with one exception (bis(2-Ethylhexyl)phthalate) were eliminated from consideration as surface water-COPCs. Based on no detection in sediment samples, beryllium, cobalt, selenium, silver, thallium, and all PCBs except Aroclor-1248 were eliminated from consideration as sediment-COPCs. See Table A3.

The achieved detection limits for SVOCs in sediment were very high relative to the target detection limits due to sample matrix problems. Also, the detection limit for most OC pesticides in sediment (0.05 mg/kg) was 5 times greater than the target detection limit (0.01 mg/kg). Therefore, even though no SVOCs and a majority of OC pesticides were not detected in any sediment sample, the compounds could not be eliminated from further consideration as sediment-COPCs based on this criterion alone.

5.2.2 Step 2 — Data quality review and comparison with laboratory blanks

In response to variations or failures in certain data quality measures, individual results may be assigned a data qualifier. In general, chemicals with data qualifiers that indicate an estimated concentration on a positively detected chemical ("J" flags) can be included in the risk assessment. Data qualified with "R" flags are rejected for use in risk assessment because performance requirements in the sample or in the associated quality control analysis were not met. Chemicals that occur in samples as a result of sample contamination should not be included in the risk assessment ("UB" flagged) if they meet the five times rule (see appendix K, attachment 2).

Certain chemicals may be introduced into samples inadvertently during sampling or laboratory analysis. Examination of sampling blanks (bottle, equipment, field, and trip) and laboratory blanks (method, instrument) helps to identify any contamination that may have occurred. During data validation, the amounts of the common laboratory contaminants (see Exhibit 5-2) detected in site samples are compared with the amounts detected in associated blanks.

Exhibit 5-2. Common Laboratory Contaminants

Acetone	Hexanes
2-Butanone (methylethylketone)	Methylene chloride
Carbon dioxide	Phthalates
Diethylether	Solvent preservatives (e.g., cyclohexanone)
Freons	Toluene
Aldol reaction products of acetone (e.g., 4-hydroxy-4-methyl-2-pentanone)	

Source: USACE 1996b, USEPA 1988a and b.

If the common laboratory contaminant amount in the sample is less than or equal to ten times the amount in any blank sample associated with the site sample (the "ten times" rule), the chemical is generally reported as undetected in the site sample, with the detection limit raised to the amount in the sample (USEPA 1988a and b). If multiple blank samples are associated with a given sample, the blank containing the highest concentrations of the chemical is used in the comparison.

For organic and inorganic chemicals other than those identified as common laboratory contaminants, if the amount in the sample is less than or equal to five times the amount of the maximum blank value associated with the sample (the "five times" rule), the chemical is reported as undetected in the site sample (noted with a "UB" flag) (USEPA 1989).

If the chemical concentration in all site samples can be attributable to sampling or laboratory contamination, it should not be evaluated further in the risk assessment. This criterion is intended as a guide, as professional judgment should play an important role in its application. There are times when some detected compounds do not meet the criteria, yet the contamination could only be attributed to laboratory or sampling artifacts.

A data set may contain compounds reported as tentatively identified compounds (TICs). Compounds so reported are considered tentative because the analytical method did not specifically analyze for the compound. The concentration of a TIC can only be estimated, because corresponding standards have not been analyzed to calibrate the instrument. TICs may be reported as specific compounds; however, TICs are often reported in a generic manner, such as "petroleum hydrocarbons." In many cases the exact nature of a TIC is not known. A large number of TICs in a sample suggests that additional chemicals not specifically analyzed for are present, indicating that the scope of the chemical analysis may not have been broad enough.

5.2.2.1 Step 2: Marsden Lake (North)

No data were rejected from site samples, no tentatively identified compounds were reported, and no known blank contamination was identified for organic analyses. Bis(2-ethylhexyl)phthalate was detected in two surface water samples (sample locations 19 and 20). Phthalates (plasticizers) are very common contaminants in the laboratory. Sources of phthalates include tygon tubing, soft plastic items, and plastic sheeting. The contamination could have occurred in the laboratory. Even though it was not detected in the associated laboratory blanks, the detected concentrations (8.3 and 44 µg/L) are both within five times the detection limit (10 µg/L). This suggests that the detection in the site sample may be due to laboratory contamination. Given this information and the fact that phthalates were not detected in the other sampling media at this study site, bis(2-ethylhexyl)phthalate was eliminated as a surface water COPC.

Aluminum, chromium, cobalt, magnesium, and silver were detected in surface water laboratory method blank samples using trace metals analysis; however, these analytes were not eliminated from further consideration because their concentrations were more than 5 times the highest associated blank. Therefore, the only chemical eliminated from the COPC list based on criterion was bis(2-ethylhexyl)phthalte from surface water.

5.2.2.2 Step 2: Marsden Lake (South)

No data were rejected from site samples, no tentatively identified compounds were reported, and no known blank contamination was identified for organic analyses. Aluminum, chromium, cobalt, magnesium, and silver were detected in surface water laboratory method blank samples using trace metals analysis; however, these analytes were not eliminated from further consideration because their concentrations were more than 5 times the highest associated blank.

Butylbenzylphthalate was detected in one sediment sample (sample location SC-ML-SD-03). Phthalates (plasticizers) are very common contaminants in the laboratory. The contamination could have occurred in the laboratory. It was not detected in the associated laboratory blanks, and the estimated detected concentration (870 µg/kg) is within five times the target detection limit and 5 times the achieved detection limit. This suggests that the detection in the site sample may be due to laboratory contamination. Given this information and the fact that phthalates were not detected in the other sampling media at this study site, Butylbenzylphthalate was eliminated as a sediment COPC.

5.2.2.3 Step 2: Pond G

No data were rejected from site samples, no tentatively identified compounds were reported, and for sediment analysis no known blank contamination was identified in organic analysis. One chemical, bis(2-ethylhexyl)phthalate (BEHP), was eliminated from the potential surface water-COPC list based on this criterion. The reason for eliminating this chemical is that it is a common laboratory contaminant (found in many plastics) and was found in the field blank, as well as in 2 of the 3 water samples taken, indicating that its presence is probably due to contamination that occurred during sampling or analysis. In addition dieldrin, 4,4'-DDT, 4,4'-DDD, and 4,4'-DDE were detected in one sediment sample (PG-02). Two of the values were below the achieved detection limit (0.06 mg/kg), 0.03 mg DDT/kg and 0.05 mg DDE/kg. However, because the target detection limit was not reached these pesticides could not be eliminated as sediment COPCs.

Aluminum, chromium, cobalt, magnesium, and silver were detected in surface water laboratory method blank samples using trace metals analysis; however, these analytes were not eliminated from further consideration because their concentrations were more than 5 times the highest associated blank. No chemicals were eliminated from the COPC list based on this aspect of the criterion.

5.2.3 Step 3 — Comparison to Background Sediment Data

A sediment COPC was defined by a set evaluation criteria. First, if a substance was not detected at the detection limit it was not considered a COPC. Second, if any of the site sediment concentrations was greater than the background screening value (based on data provided by USFWS; see Appendix Q for additional detail), then the substance was selected as a COPC. If all site concentrations are below the screening value, then it was not considered a COPC. Lastly, if no background screening value was available, and concentrations were detected above the detection limit, then the substance was considered a COPC. No background sediment values were provided for antimony, cobalt, mercury, or OC pesticides (including DDT).

The selection process for determining COPCs in surface waters was less restrictive than for sediments, primarily because background-screening values were not available in 1997. In order to be considered a surface water COPC, the constituent must be detected above the detection limit.

5.2.3.1 Step 3: Marsden Lake (North)

Aluminum, barium, chromium, lead, magnesium, manganese, nickel, selenium, and zinc were eliminated as sediment COPCs because all site concentrations were below the Tier I background screening value (see Exhibit 5-3). Background screening values were not available for antimony and mercury; therefore, these two analytes were retained as COPCs.

Exhibit 5-3.Marsden Lake (North)* Sediment Comparison to Tier I Background Screening Values

Analyte	Maximum Sediment Concentration (ppm)	Tier I Screening Value (ppm)	Eliminate or Retain for Further Evaluation?
Al	6700	10674	Eliminate
Sb	48	Na	Retain
As	8.9	5	Retain
Ba	100	238	Eliminate

Cd	1.4	1	Retain
Cr	14	24	Eliminate
Cu	43	30	Retain
Pb	26	32	Eliminate
Mg	3300	13734	Eliminate
Mn	330	2046	Eliminate
Hg	0.14	Na	Retain
Ni	14	26	Eliminate
Se	1.2	10	Eliminate
Zn	86	101	Eliminate

Na: No Tier I screening value was provided for antimony and mercury.
*Note: Does not include samples ML-13 and ML-15

5.2.3.2 Step 3: Marsden Lake (South)

Barium, magnesium, manganese, nickel, and selenium were eliminated as COPCs because all site concentrations were below the background screening value (see Exhibit 5-4). Background screening values were not available for antimony, cobalt, and mercury; therefore, these three analytes were retained as COPCs.

Exhibit 5-4. Marsden Lake (South)* Sediment Comparison to Tier I Background Screening-Values

Analyte	Maximum Sediment Concentration (ppm)	Tier I Screening Value (ppm)	Eliminate or Retain for Further Evaluation?
Al	15000	10674	Retain
Sb	1.0	Na	Retain
As	12	5	Retain
Ba	190	238	Eliminate
Cd	1.6	1	Retain
Cr	30	24	Retain
Cu	52	30	Retain
Pb	63	32	Retain
Mg	4300	13734	Eliminate
Mn	680	2046	Eliminate
Hg	0.18	Na	Retain
Ni	23	26	Eliminate
Se	1.4	10	Eliminate
Zn	140	101	Retain

Na: No Tier I screening value was provided for antimony and mercury.
*Note: Includes samples ML-13 and ML-15

5.2.3.3 Step 3: Pond G

Beryllium, cobalt, selenium, silver, and thallium were eliminated as sediment COPCs because all concentrations were below the detection limits at all sample locations. Barium, cadmium, magnesium, manganese, and nickel were eliminated as COPCs because all site concentrations were below the background screening value. Background screening values were not available for mercury, dieldrin, DDT, DDD, and DDE; therefore, they were retained as COPCs.

Exhibit 5-5. Pond G Sediment Comparison to Tier I Background Screening-Values

Analyte	Maximum Sediment Concentration (ppm)	Tier I Screening Value (ppm)	Eliminate or Retain for Further Evaluation?
Al	18000	10674	Retain
Sb	0.63	Na	Retain
As	7.6	5	Retain
Ba	190	238	Eliminate
Cd	0.81	1	Eliminate
Cr	36	24	Retain
Cu	39	30	Retain
Pb	39	32	Retain
Mg	3600	13734	Eliminate
Mn	390	2046	Eliminate
Hg	0.14	Na	Retain
Ni	22	26	Eliminate
Zn	110	101	Retain
Aroclor-1248	0.39	0.05	Retain
Dieldrin	0.071	Na	Retain
DDT	0.03	Na	Retain
DDD	0.16	Na	Retain
DDE	0.053	Na	Retain

Na: No Tier I screening value was provided for antimony and mercury.

5.2.4 Step 4 — Toxicity screening and development of conservative hazard quotients

Assessment endpoints were selected for all sites during the development of the Work Plan (USACHPPM 1999), and are discussed in Section 3. For this part of the Tier I re-assessment, hazard quotients (HQs) were calculated for each assessment endpoint using the methods approved in the original 1997 Tier I risk assessment report (USACHPPM 1997). Toxicity values used in the 1997 Tier I report were the values used in this Tier I assessment. The Tier I water and sediment toxicity reference values used to calculate HQs for aquatic and benthic organisms are presented in Appendix Q (Tables 7 & 8). If any location had an analyte HQ exceeding unity, that analyte was carried through to be evaluated in the Tier II process.

DETERMINATION OF HAZARDS FOR AQUATIC AND BENTHIC ORGANISMS

Hazard quotients were calculated only for those preliminary COPCs not eliminated prior to this step. Tables A1-A3 indicate which analytes were eliminated prior to this step. As per the request of the regulatory agencies, hazards or HQs were calculated on a sample-by-sample basis (See Appendix Q).

There was no Tier I selenium water quality benchmark available for comparison with the site concentrations. No benthic organism HQs were calculated for SVOCs and OC Pesticides due to detection limit problems, so these chemical groups should be carried through the risk evaluation because it cannot be said with absolute certainty that they are not present.

DETERMINATION OF HAZARDS FOR WILDLIFE

Hazard quotients were calculated only for those preliminary COPCs not eliminated prior to this step. Tables A1-A3 indicate which analytes were eliminated prior to this step. Total average daily dietary doses were calculated for all receptors and the resulting modeled dose was compared to the corresponding toxicity benchmark value to calculate the hazard quotient. See Appendix Q for the dose algorithms and toxicity benchmarks that were used.

For all wildlife receptors, hazard quotients could not be calculated for antimony, manganese, and thallium because toxicity values were not available from the 1997 Tier I assessment. No HQs were calculated for SVOCs and most of the OC Pesticides due to detection limit problems. Only DDT, DDD, and DDE were modeled because these chemicals were qualified as detections and toxicity data were available. These preliminary COPCs should be carried through the risk evaluation because it cannot be said with absolute certainty that they are not present or that they pose no hazard.

5.2.4.1 Step 4: Marsden Lake (North)

Hazard quotients are presented in Tables A4 – A9. The overall potentials for risk are not uniform across the lake, such that for some analytes (i.e., mercury), there are several orders of magnitude difference in the HQs across sampling locations, per receptor. Also, among receptors there are varied HQ results for each analyte. Please note that sampling locations ML-13 and ML-15 were combined with the Marsden Lake South data set (see section 4.1.2.2)

For aquatic organisms (Table A4), barium was the primary contaminant of potential concern, with HQs ranging from 10.5 to 52.6 over all eight sample locations. For all other surface water analytes, HQs were less than or equal to 4.3. Of the eight sampling locations, aluminum and manganese had HQs greater than one at 5 locations. Lead and mercury had exceedances at 2 locations each. Antimony, arsenic, cadmium, chromium, cobalt, copper, magnesium, nickel, silver, thallium, and zinc all had HQ values less than one.

For benthic organisms (Table A5), copper is most likely the primary COPC. Copper HQs exceeded unity at 3 of the 8 sampling locations. Antimony had the largest HQ value (HQ=4) at location ML-17, which was the only location with an antimony HQ exceeding unity. For all other sediment analytes (arsenic, cadmium, and mercury), HQs were less than or equal to 2.3. Of the eight sampling locations, arsenic had HQs greater than one in only 3 locations and cadmium had an exceedance in only 1 of the sampling locations. No mercury HQs were greater than one.

For amphibians, (Table A6), there were only two toxicity values available for use in calculating HQs. Therefore, due to the numerous data gaps, and for ease of reporting, amphibians were excluded from the summary table presentation (Table A1). Toxicity reference values were available for cadmium and cobalt. Cadmium HQs were greater than one at 2 locations, with values ranging from 1.0 to 1.6. No cobalt HQs exceeded one.

For wading birds (Table A7), mercury was the primary COPC, with the most significant exposure pathway being fish ingestion. Mercury HQs for the wading bird ranged from 624.2 to 13,747.8. Arsenic, barium, and chromium had HQs exceeding one at all locations. Zinc had HQ exceedances at 7 of the 8 locations. Aluminum, copper, lead, and zinc had HQ exceedances at 7 of the 8 sample locations, and HQs for cadmium were exceeded at three locations. No HQs were calculated for SVOCs and OC Pesticides due to detection limit problems, so these chemical groups should be carried through the risk evaluation because it cannot be said with absolute certainty that they are not present.

For waterfowl (Table A8), mercury was the only analyte with HQs greater than one, at two locations. The most significant exposure pathway is through benthos ingestion (i.e., invertebrates). The mercury HQs were 1.3 and 1.1. Aluminum, arsenic, barium, cadmium, chromium, copper, nickel, and zinc do not exceed unity at any location. No HQs were calculated for SVOCs and OC Pesticides due to detection limit problems, so these chemical groups should be carried through the risk evaluation because it cannot be said with absolute certainty that they are not present.

For aquatic mammals (Table A9), aluminum is the primary COPCs with the most significant exposure pathway being sediment ingestion. Aluminum HQs exceeded unity at all of the sampling locations, with values ranging from 7.9 to 701.6. Antimony, arsenic, barium, and cadmium had HQs exceeding unity at all locations, and mercury had exceedances at two locations. Chromium, cobalt, copper, nickel, silver, and zinc had no HQs greater than one.

5.2.4.2 Step 4: Marsden Lake (South)

Hazard quotients are presented in Tables A10 – A15. The overall potentials for risk are not uniform across the lake, such that for some analytes (i.e., mercury), there are several orders of magnitude difference in the HQs across sampling locations, per receptor. Also, among receptors there are varied HQ results for each analyte. Note that samples ML-13 and ML-15 were combined with this data set.

For aquatic organisms (Table A10), barium is the primary COPC, with HQs ranging from 9.2 to 23.4 over the 12 sampling locations. Aluminum, lead, and mercury had HQs exceeding one at one location only (ML-08), with values of 8.8, 2.1, and 2.4 respectively. Antimony, arsenic, cadmium, chromium, cobalt, copper, manganese, nickel, silver, thallium, and zinc did not have HQs exceeding unity.

For benthic organisms (Table A11), no one specific COPC was identified as being the primary risk driver; however, arsenic exceeded the HQ value of one at all 12 sampling locations. Hazard quotient values for arsenic ranged from 1.1 to 2. In addition, the largest sediment HQ calculated from any analyte was 3.3 (copper at ML-07). Copper and lead each had HQs exceeding unity at 10 locations, there were five exceedances for cadmium, and three exceedances for chromium. Zinc had one HQ exceeding unity at location ML-15 (HQ = 1.2). Aluminum, antimony, cobalt, and mercury did not have any locations with HQ exceedances. As discussed previously 4,4'-DDD could not be eliminated as a sediment COPC. The HQ calculated for the single detection of the pesticide in the ML-13 duplicate sample was 11.9.

For amphibians, (Table A12), there were only two toxicity values available for use in calculating HQs. Therefore, due to the numerous data gaps, and for ease of reporting, amphibians were excluded from the summary table presentation (Table A1). Toxicity reference values were available for cadmium and cobalt. Cadmium HQs were greater than one at 5 locations, with values ranging from 1.0 to 1.5. No cobalt HQs exceeded one.

For wading birds (Table A13), mercury is the primary COPC, with HQs ranging from 433.9 to 17,978 over the 12 sampling locations. The most significant pathway for mercury risk to the wading bird is through fish ingestion. Aluminum, arsenic, barium, chromium, mercury, and zinc had HQs greater than one at all sampling locations. Cadmium and copper had HQ exceedances at only one of the sampling locations (ML-08), and lead had 4 locations with exceedances.

For waterfowl (Table A14), aluminum is the primary COPC, with HQs ranging from 1.8 to 8.3 over the 12 sampling locations. The most significant pathway for aluminum exposure is through benthos ingestion (i.e., invertebrates). Mercury also had HQ exceedances at all 12 sampling locations with values ranging from 1.0 to 1.7, and chromium had HQ exceedances at 8 of the 12 locations with values ranging from 1.0 to 1.8.

For aquatic mammals (Table A15), aluminum is the primary COPC, with HQs ranging from 42.5 to 1561.3 over the 12 sampling locations. Arsenic, barium, and cadmium, also had HQ exceedances at all 12 sampling locations, and chromium, and mercury had one HQ exceedance each at sampling location ML-08.

5.2.4.3 Step 4: Pond G

Hazard quotients are presented in Tables A16 – A19. The overall potentials for risk are mostly uniform across the site. This site was only evaluated as a drinking water source and water and sediment ingestion will be the main factor determining any exposure to receptors.

For aquatic organisms (Table A16), barium is the primary COPC, with HQs ranging from 13.4 to 26.3 over the three sampling locations. Aluminum also had HQs exceeding one at all three sampling locations with aluminum HQs ranging from 1.9 to 12.8. Lead had HQ exceedances at two of the three sampling locations, ranging from 1.0 to 2.5, and Mercury had one exceedance (HQ=2.5). Antimony, arsenic, cadmium, chromium, cobalt, copper, magnesium, nickel, selenium, silver, thallium, and zinc had no HQs exceeding one. Manganese had an HQ of 1.1 in the field duplicate sample. There was no Tier I water quality benchmark provided for beryllium or selenium so they were carried through the risk assessment.

For benthic organisms (Table A17), copper is the primary COPC, with HQs ranging form 1.1 to 2.4 over the three sampling locations. Arsenic, chromium, and lead had only one HQ exceedance each, 1.3, 1.4, and 1.3, respectively. Aluminum, mercury, and zinc all had HQs less than one. Aroclor 1248 was detected in the field duplicate sample and had an HQ of 6.5. Some organochlorine pesticides were detected in the sediment at location PG-02 and HQs were calculated. For 4,4'-DDT the resulting HQ was 4.3. For 4,4'-DDD which was also detected in the field duplicate, resulting HQs were 20.0, and 12.1, respectively, and the HQ for 4,4'-DDE was 10.6. Dieldrin was also detected at PG-02 but had an HQ less than unity.

For wading birds (Table A18), aluminum was the only COPC with an HQ greater than one. In sample PG-03 and in the field duplicate, the aluminum HQs calculated were just above one, 1.7 and 1.4, respectively. The main exposure pathway for this analyte is through sediment ingestion while drinking water from the pond. No toxicity screening value was available for beryllium (which was detected in the surface water). A toxicity value was also not available for dieldrin; however, toxicity values were available for DDT and its metabolites. Hazard Quotients for DDT and its metabolites were below one.

For aquatic mammals (Table A19), aluminum and arsenic are the main COPCs, with HQs ranging from 456.5 to 2200.7 for aluminum, to 74.1 to 102.7 for arsenic over the three sampling locations. Antimony, barium, cadmium, manganese, selenium, and thallium all had HQ exceedances at all three sampling locations. Mercury had two HQs exceeding unity, 1.1 and 1.7. Lead and chromium only had one exceedance each, at sites PG-03. Beryllium did not exceed unity at any location. A toxicity value was not available for dieldrin so an HQ could not be calculated. However, toxicity values were available for DDT and its metabolites. Hazard Quotients for DDT and its metabolites were below one.

5.2.5 Summary of Retained COPCs

For each of the three sites (Marsden Lake North, Marsden Lake South, and Pond G), four steps were followed in this Tier I assessment to select the final list of COPCs to be evaluated in the follow-on Tier II assessment. Tables A1-A3 summarize the process for each site. Step 1 eliminated chemicals that were not detected. Step 2 was a data quality review and a comparison with laboratory blanks. Laboratory contaminants were eliminated in this step. Step 3 was the elimination of some sediment analytes based on a comparison to the Tier I background sediment screening values. Lastly, step 4 was a comparison to toxicity benchmarks using media concentrations and dose models, otherwise known as hazard quotient calculations. If all receptors evaluated for a specific analyte had HQs less than one at all locations, the analyte was eliminated as a COPC.

Exhibit 5-6 presents the final selected COPCs. The source of potential unacceptable risks to receptors at these sites is based on exposure to COPCs through water, sediment, or through the food chain. Based on the fate and transport of chemicals, the primary media for exposure at Marsden Lake North, Marsden Lake South, and Pond G are water and sediment. It is from these media that contaminants are transferred through the food chain. To better illustrate which COPCs are associated with a particular medium, codes (s, w, or b) were added to the left of the selected COPCs for each site within Exhibit 5-6. Analytes with an "s" code are associated with site sediment, those with a "w" code are associated with site surface water, and those with a "b" are associated with both media.

It is also important to understand that when a COPC is selected (as shown in Exhibit 5-6) it does not necessarily mean that it may pose an unacceptable hazard for *all* ecological receptors at the site. Each endpoint receptor may have a different set of relevant COPCs. For example, a given COPC may not pose any hazard for wildlife, but may for aquatic organisms. On the contrary, some aquatic organisms may not have HQ exceedances, but wildlife receptors may. The final list does not discriminate among the receptors.

The final selected COPCs are those listed in the 2002 columns of Exhibit 5-6. The previous Tier I results from 1997 are presented for summary purposes only. A direct comparison should not be made between the 1997 COPC results and the 2002 COPC results. In order to make a direct comparison between assessments it would have been necessary to have the same sampling methods and analytical methods to quantify the data. Detection limits used during the 1997 and 2002 analyses are different, and values should not be compared quantitatively. In the 1997 Tier I evaluation, only five locations were sampled in Marsden Lake, and no differentiation was made around the two potential source areas. For Pond G, only one sample was taken in the 1997 assessment. Vanadium was the only COPC from the 1997 Tier I assessment that was not evaluated during the 2002 Tier I reassessment because it was not analyzed for. In the 1997 Tier I assessment, vanadium was estimated to pose a potential risk to aquatic mammals at Round Lake, Sunfish Lake, Pond G, and Marsden Lake, but no HQ value was greater than four (see Appendix Q). During the 1997 assessment, no sediment HQ was calculated for vanadium because no toxicity benchmark was selected. The potential for risk to receptors from vanadium is an uncertainty and will be treated as such during the remainder of this assessment. No HQs were calculated for wildlife for Sb, Mn, and Tl because no toxicity values were presented for these analytes in the original 1997 Tier I assessment. Toxicological reference values (TRVs) only for aquatic mammals were presented for cobalt and silver in the 1997 Tier I report. For beryllium in Pond G no water, amphibian, wading bird, or waterfowl HQs were calculated because no Tier I screening values were provided in the Tier I ERA report (USACHPPM 1997). Beryllium was retained as a surface water COPC due to lack of a screening value. These analytes were selected as COPCs due to the uncertainty of the potential for risk. Tier I dieldrin screening values were also not available for wading birds or mammals so dieldrin was retained as a sediment COPC at Pond G.

Exhibit 5-6. Summary of Selected COPCs and Comparison of Previous and Current Tier I Results*

Prelimary COPCs	Chemicals of Potential Concern (COPCs)									
	Marsden Lake						Pond G			
	1997		North (2002)		South (2002)		1997		2002	
Aluminum	b	Al	w	Al	b	Al	b	Al	b	Al
Antimony		X	b	Sb	b	Sb		X	b	Sb
Arsenic		X	b	As	b	As		X	b	As
Barium	b	Ba	w	Ba	w	Ba	w	Ba	w	Ba
Beryllium		X		X		X		X	w	Be
Cadmium		X	b	Cd	b	Cd		X	w	Cd
Chromium		X	w	Cr	b	Cr		X	b	Cr
Cobalt		X	w	Co	b	Co		X	w	Co
Copper	(w)	X inconclusive	b	Cu	b	Cu	s (w)	Cu inconclusive	b	Cu
Lead		X	w	Pb	b	Pb	s	Pb	b	Pb
Magnesium		X		X		X		X		X
Manganese	w	Mn	w	Mn	w	Mn	w	Mn	w	Mn
Mercury	s (w)	X inconclusive	b	Hg	b	Hg	w	Hg		X
Nickel		X		X		X		X		X
Selenium		X		X		X		X	w	Se
Silver	(w)	X inconclusive	w	Ag	w	Ag	(w)	X Inconclusive		X
Thallium		X	w	Tl	w	Tl		X	w	Tl
Vanadium	s	V		†		†	s	V		†
Zinc	b	Zn	b	Zn	b	Zn	b	Zn		X
SVOCs		X	s	SVOCs	s	SVOCs		X	s	SVOCs
OC Pesticides	s	DDT metabolites	s	OC Pesticides	s	OC Pesticides	s	DDT Metabolites	s	OC Pesticides
PCBs		X		X		X	s	Aroclor 1254	s	Aroclor 1248

*: The "X" indicates that the preliminary COPC was not selected as a COPC. The codes indicate whether the COPC is specific to the water column (w), sediment (s), or whether it is a COPC for both media (b).

†: Samples from the 1999 field investigation were not analyzed for vanadium, but it was carried through as a COPC due to uncertainty about the potential for risk to wildlife.

5.3 TIER II COC SELECTION PROCESS

Chemicals that are known, or suspected, to be contaminants and that also may pose an ecological risk are defined as chemicals of concern (COCs). The analysis phase of the Tier II risk assessment focuses upon identified COCs, not COPCs. The objectives of the COC selection process is to identify a subset of the detected chemicals that are not naturally occurring or a result of non-site sources and are present at sufficient frequency, concentration, and location to pose potential ecological risks. The following process was used to identify those COPCs that are not appropriate to retain or select as COCs for each study site (USACE 1996b). For a list of the COPCs being evaluated in this selection process see the text boxes for Rice Creek, Sunfish Lake, and Round Lake in section 3, and for Marsden Lake and Pond G see Exhibit 5-6.

For Round Lake, Sunfish Lake, and Rice Creek

1. Chemicals not detected.
2. Data quality review and comparison with laboratory blanks.

For Round Lake, Sunfish Lake, Rice Creek, Pond G, and Marsden Lake Sites

3. Comparability with toxicological benchmarks and bioaccumulation potential through the foodchain.
4. Chemical distribution and detection frequency in sampled media.
5. Site-relatedness and comparability with background concentrations.

These criteria were applied sequentially to the available data. Each study site was treated independently. Once a COPC was eliminated based upon a criterion, it was not considered further for that study site. Each of the criteria is discussed below. All of the selected COCs for each study site and environmental medium (i.e., surface water, sediment) are reviewed in Section 5.10.

5.3.1 Criterion 1 — Chemicals Not Detected

This criterion was described above in the Tier I re-assessment (Section 5.2.1). This evaluation is not repeated for Marsden Lake and Pond G sites; however, this evaluation was conducted for the 1999-2000 COPC water monitoring data collected for Rice Creek, Sunfish Lake, and Round Lake.

5.3.1.1 Criterion 1: Rice Creek

During 1999-2000, surface water was analyzed for previously selected water-COPCs (aluminum, barium, cadmium, copper, mercury, and silver) according to methods defined in the QAPP (USACHPPM 1999). All analytes, with the exception of silver, were detected in the surface water. See section 5.3.2.1 for an explanation of why silver was considered undetected.

No sediment-COPCs were identified during the Tier I assessment.

The following chemicals were retained for further consideration as COCs:

Surface Water	Sediment
Aluminum	None
Barium	
Cadmium	
Copper	
Mercury	

5.3.1.2 Criterion 1: Sunfish Lake

During 1999-2000, surface water was analyzed for previously selected water-COPCs (aluminum, barium, cadmium, copper, mercury, silver, and zinc) according to methods defined in the QAPP (USACHPPM 1999). Silver was the only analyte not detected in the surface water (see section 5.3.2.2 for an explanation of silver being qualified as undetected), and was therefore eliminated as a COC.

Additional sediment chemistry analyses were not performed during 1999-2000; therefore, no sediment-COPCs were eliminated based on this criterion. See the text box in section 3.4.3 for the list of sediment COPCs to be evaluated.

After this point in the process, the following chemicals were retained for further consideration as COCs

Surface Water	Sediment
Aluminum	Aluminum
Barium	Chromium
Cadmium	Copper
Copper	Lead
Mercury	Vanadium
Zinc	Zinc

5.3.1.3 Criterion 1: Round Lake

Surface water was analyzed for previously selected water-COPCs (barium, cadmium, copper, mercury, silver, and zinc) according to methods defined in the QAPP (USACHPPM 1999). Silver was not detected in any surface water sample during 1999-2000. Silver was, therefore, eliminated from further consideration.

Additional sediment chemistry analyses were not performed during 1999-2000; therefore, no sediment-COPCs were eliminated based on this criterion. See the text box in section 3.5.4 for the list of sediment COPCs to be evaluated.

After this point in the process, the following chemicals were retained for further consideration as COCs

Surface Water	Sediment
Barium	Aluminum
Cadmium	Cadmium
Copper	Chromium
Mercury	Copper
Zinc	Lead
	Silver
	Vanadium
	Zinc
	PCBs

5.3.2 Criterion 2 — Data quality review and comparison with laboratory blanks

This criterion was described above in the Tier I re-assessment (Section 5.2.2). This evaluation is not repeated for Marsden Lake and Pond G sites; however, this evaluation was conducted for the 1999-2000 COPC water monitoring data collected for Rice Creek, Sunfish Lake, and Round Lake.

5.3.2.1 Criterion 2: Rice Creek

No data were rejected from site samples and no tentatively identified compounds were reported. Over the four quarters sampled, trace metals analysis results for surface water samples showed several metals detections in laboratory method blanks. For Rice Creek, aluminum, barium, cadmium, mercury, and silver were detected in blanks during at least one sampling quarter (see Appendix K Attachment 2 for the specific metals that were detected in the blanks and how these were handled).

Following the procedure outlined in Appendix K Attachment 2, even though silver was detected, it was eliminated from further consideration as a surface water COPC under criterion one because during all quarters it was detected at less than 5 times the maximum method blank.

Based on this criterion the following chemicals were retained for further consideration as COCs:

Surface Water	Sediment
Aluminum	None
Barium	
Cadmium	
Copper	
Mercury	

5.3.2.2 Criterion 2: Sunfish Lake

No data were rejected from site samples and no tentatively identified compounds were reported. Over the four quarters sampled, trace metals analysis results of surface water samples showed several metals detections in laboratory method blanks. For Sunfish Lake, aluminum, barium, cadmium, copper, mercury, silver, and zinc were detected in method blanks during at least one sampling quarter (see Appendix K Attachment 2 for the specific metals that were detected in the blanks and how these were handled).

Following the procedure outlined in Appendix K Attachment 2, even though silver was detected in one sample, it was eliminated from further consideration as a surface water COPC under criterion one because during all quarters it was detected at less than 5 times the maximum method blank.

After this point in the process, the following chemicals were retained for further consideration as COCs

Surface Water	Sediment
Aluminum	Aluminum
Barium	Chromium
Cadmium	Copper
Copper	Lead
Mercury	Vanadium
Zinc	Zinc

5.3.2.3 Criterion 2: Round Lake

No data was rejected from site samples and no tentatively identified compounds were reported. Over the four quarters sampled, trace metals analysis results of surface water samples showed several metals detections in laboratory method blanks. For Round Lake, barium, cadmium, mercury, silver, and zinc were detected in method blanks during at least one sampling quarter (see Appendix K Attachment 2 for the specific metals that were detected in the blanks and how these were handled).

Following the procedure outlined in Appendix K Attachment 2, even though silver was detected, it was eliminated from further consideration as a surface water COPC under criterion one because during all quarters it was detected at less than 5 times the maximum method blank

After this point in the process, the following chemicals were retained for further consideration as COCs

Surface Water	Sediment
Barium	Aluminum
Cadmium	Cadmium
Copper	Chromium
Mercury	Copper
Zinc	Lead
	Silver
	Vanadium
	Zinc
	PCBs

5.3.3 Criterion 3 — Comparability with toxicological benchmarks and bioaccumulation potential through the food chain

A toxicity screen to determine which chemical concentrations exceed applicable toxicity benchmarks is performed as the next step in COC selection. A version of this criterion is provided above in the Tier I re-assessment (Section 5.2.4). However, the Tier II QAPP (USACHPPM 1999) provided risk based action levels (RBALs) that were not provided in the Tier I report. In some instances values that were used in the Tier I were different than those selected as RBALs in the Tier II QAPP. Therefore, this evaluation was conducted for all sites except Pond G (see section 5.3.3.3). The maximum concentration of each COPC that has been selected up to this point was compared to the RBAL for that analyte. See the QAPP (USACHPPM 1999) for a complete list of the RBALs and the literature source of each value.

The toxicity reference values chosen as the RBALs were derived from various sources. The majority of the surface water benchmarks chosen as RBALs are Minnesota Water Quality Standards (MWQS) for Class 2B/2C/2D waters. When MWQS values were not available for a particular chemical, other literature sources were used (USEPA 1986, 1993, 1994, 1995a, 1995b, 1996 and ORNL 1994 and others). There are also various sources that provide toxicity reference values for sediment. The main source of sediment RBALs was "Low Effect Levels" from the Ministry of Environment and Energy, Ontario, Canada.

Standardized values for screening terrestrial wildlife exposures to surface water are currently under development by EPA and other organizations. The Great Lakes Water Quality Initiative (GLWQI) Final Rule (USEPA 1995a) published four water quality criteria for the protection of wildlife (birds and mammals) which feed on aquatic organisms: mercury, p,p'-dichlorodiphenyl-trichloroethane (DDT), 2,3,7,8-tetrachlordibenzo-p-dioxin (TCDD), and polychlorinated biphenyls (PCBs). In a few cases, chronic Federal Ambient Water Quality Criteria (AWQC) for chemicals that bioaccumulate are based on final residue values and the protection of sensitive mammals (PCBs and mink) or birds (DDT and brown pelican). Where such exposure pathways are relevant to the risk assessment, the GLWQI criteria and Federal and state AWQC can be used in screening water concentrations for COC selection. However, when the RBALs were developed during the Work Plan (USACHPPM 1999) the GLWQI values were not considered.

The maximum detected surface water concentration for each analyte on site was compared to the toxicity benchmark (i.e., RBAL) for that analyte, as identified in the QAPP.

Before a chemical could be eliminated based on the comparison with the toxicity benchmarks, the potential for the chemical to bioaccumulate in the food chain was examined. Chemicals that have the potential to significantly bioconcentrate or bioaccumulate in organisms, or biomagnify through the food web, should be retained for consideration as COCs, even where distribution is limited. Chemicals that bioaccumulate include those that are taken up by organisms by both direct exposure to a contaminated medium and by consumption of food containing the chemicals. By definition, biomagnification is that circumstance where bioaccumulation causes an increase in total body burden as one proceeds up the trophic ladder from primary producer to top carnivore (Schnoor 1996). Chemicals that bioconcentrate are taken up by organisms due to direct contact with a contaminated medium (e.g., water).

Organic chemicals with a relatively low affinity for lipids over water (i.e., those with octanol-water partition coefficients (K_{ow}) less than 1000 (or a log K_{ow} less than 3) are not expected to bioaccumulate or biomagnify. Organic chemicals with a relatively high affinity for soil or sediment (i.e., those with a soil-water partition coefficient (K_{oc}) of less than 500 (or a log K_{oc} less than 2.7) are also not expected to accumulate to appreciable levels within organisms (USACE 1996b). The chlorinated pesticides are the most well known of the chemical groups that tend to bioaccumulate and bio-magnify. PCBs and dioxins/furans are also strong bioaccumulators and biomagnifiers. Volatile organic compounds (VOCs) such as tetrachloroethene, toluene, trichloroethene, 1,1,1-trichloroethane, and xylenes are unlikely to bioaccumulate and biomagnify (Van Leeuwen et al. 1992, USEPA 1982). Semivolatiles, including PAHs, tend not to bioaccumulate and show little tendency to biomagnify because they are readily metabolized (Eisler 1987, Beyer and Stafford 1993). Metals do not have log K_{ow} values so this aspect was not useful in evaluating the potential for a metal to biomagnify.

The target detection limits for the project were based, in part, on toxicity benchmarks that were not developed for food-chain bioaccumulation exposures. Therefore, this criterion was used to identify the chemicals, from among those not yet eliminated by this point in the process, which would be biomagnification threats. If a chemical was so identified, then the comparability of the site concentrations to its selected toxicity benchmark only served to focus potential risk analysis and was not used to eliminate the chemical from further consideration.

Therefore, for this criterion, chemicals with concentrations that were not greater than the benchmark, and that were not determined to bioaccumulate were eliminated from further consideration. If a chemical was below the toxicity benchmark but was determined to be a bioaccumulator, it was retained for further evaluation. The remaining chemicals were retained as possible COCs for the purpose of the remaining criteria.

5.3.3.1 Criterion 3: Marsden Lake North

Of the surface water potential COCs, only mercury has strong potential for becoming a bioaccumulation/biomagnification threat to the ecological resources at the study site. Several of the SVOCs in the sediments have a limited potential for a bioaccumulation threat due to their biochemical properties. Therefore, mercury was flagged as a bioaccumulator as were SVOCs, so that even if the mercury or SVOC concentration was less than the toxicity benchmark, the chemical would be retained as a potential COC.

Aluminum, barium, lead, manganese, and mercury were retained as surface water COCs for further consideration at Marsden Lake North because their concentrations were greater than the RBALs. In addition, mercury was flagged as a bioaccumulator, as were cadmium, lead, and zinc. There is evidence for cadmium transfer through various trophic levels but it suggests that only the lower trophic levels exhibit biomagnificiation (Eisler 2000). However, there is potential for cadmium to be a bioaccumulation threat to the ecological resources at the study site. Cadmium is known to accumulate in the liver and kidney of aquatic organisms. However, there is considerable variation in the ability of fish tissue to accumulate cadmium from the ambient medium (Eisler 2000). Also, the significance of relatively high cadmium residues to animal and plant health is difficult to interpret (Eisler 2000). Due to the uncertain nature of cadmium biomagnification, cadmium was flagged and retained at this point in the process to conservatively evaluate its potential to cause potential risk to fish, aquatic invertebrates, and algae. Lead is known to bioaccumulate, especially in bone (Eisler 2000), and was flagged as a bioaccumulator. Zinc is ubiquitous in plant and animal tissues (Eisler 2000) and is known to accumulate in fish and other aquatic vertebrates, so zinc was also flagged as a bioaccumulator even though its concentration was less than the RBAL. Exhibit 5-7 shows the comparisons used to determine whether analytes would be retained or eliminated.

Exhibit 5-7. Marsden Lake North Comparison to Water Toxicity Values and Bioaccumulation Screen

Analyte	Maximum surface water concentration (μg/L)	RBAL	Bioaccumulator (Yes or No)	Outcome
Aluminum	540.25	125	N	Retain
Antimony	0.21	31	N	Eliminate
Arsenic	4.50	190	N	Eliminate
Barium	200.0	3.8	N	Retain
Cadmium	0.13	0.66	Y	Retain
Chromium	2.349	117 (total)	N	Eliminate
Cobalt	1.20	5.0	N	Eliminate
Copper	4.10	6.4	N	Eliminate
Lead	7.0	1.3	Y	Retain
Manganese	1300	80.3	N	Retain
Mercury	0.01	0.003 (total)	Y	Retain
Silver	0.0322	1.0	N	Eliminate
Thallium	0.02	18.0	N	Eliminate
Zinc	15.11	59	Y	Retain

Antimony, arsenic, cadmium, copper, and thallium, were retained as sediment COCs for further consideration at Marsden Lake North because their concentrations were greater than the RBALs. In addition, cadmium, mercury, vanadium, zinc, SVOCs, and OC Pesticides were flagged as bioaccumlators and retained as COCs. SVOCs, such as phthalate esters have a log K_{ow} of about 8, and the pesticide DDT has a log K_{ow} of about 6 (Schnoor 1996). These values indicate that these chemicals have a higher affinity for lipid than for water and are more likely to accumulate in fatty tissue of animals. Vanadium was also retained as a COC. Vanadium has been shown to accumulate in the bone, liver, and lungs of rats (WHO 2000), but it is not certain if it biomagnifies. Exhibit 5-8 shows the comparisons used to determine whether analytes would be retained or eliminated.

Exhibit 5-8. Marsden Lake North Comparison to Sediment Toxicity Values and Bioaccumulation Screen

Analyte	Maximum sediment concentration (mg/kg)	RBAL	Bioaccumulator (Yes or No)	Outcome
Antimony	48.0	None	N	Retain
Arsenic	8.90	6	N	Retain
Cadmium	1.40	0.6	Y	Retain
Copper	43.0	16	N	Retain
Mercury	0.14	0.2 (total)	Y	Retain
Vanadium	Na*	Na	Y	Retain
Zinc	86	120	Y	Retain
SVOCs	Na	Various	Y	Retain
OC Pesticides	Na	Various	Y	Retain

Na : Not available because detection limits were not met
*: Vanadium was not analyzed for during the 1999 sampling event. See section 5.2.5

5.3.3.2 Criterion 3: Marsden Lake South

Aluminum, barium, copper, lead, manganese, and mercury were retained as surface water COCs for further consideration at Marsden Lake South because their concentrations were greater than the RBALs. In addition cadmium was retained because there is evidence for cadmium transfer through various trophic levels but it suggests that only the lower trophic levels exhibit biomagnificiation (Eisler 2000). However, there is potential for cadmium to be a bioaccumulation threat to the ecological resources at the study site. Cadmium is known to accumulate in the liver and kidney of aquatic organisms. However, there is considerable variation in the ability of fish tissue to accumulate cadmium from the ambient medium (Eisler 2000). Also, the significance of relatively high cadmium residues to animal and plant health is difficult to interpret (Eisler 2000). Due to the uncertain nature of cadmium bioaccumulation, cadmium was flagged and retained at this point in the process to conservatively evaluate its potential to cause potential risk to fish, aquatic invertebrates, and algae. Zinc is ubiquitous in plant and animal tissues (Eisler 2000) and is known to accumulate in fish and other aquatic vertebrates, so zinc was also retained. Exhibit 5-9 shows the comparisons used to determine whether analytes would be retained or eliminated.

Exhibit 5-9 Marsden Lake South Comparison to Water Toxicity Values and Bioaccumulation Screen

Analyte	Maximum surface water concentration (μg/L)	RBAL	Bioaccumlator (Yes or No)	Outcome
Aluminum	1100.25	125	N	Retain
Antimony	0.10	31	N	Eliminate
Arsenic	3.60	190	N	Eliminate
Barium	89.0	3.8	N	Retain
Cadmium	0.19	0.66	Y	Retain
Chromium	3.549	117 (total)	N	Eliminate
Cobalt	2.007	5.0	N	Eliminate
Copper	9.00	6.4	N	Retain
Lead	6.80	1.3	Y	Retain
Manganese	370.0	80.3	N	Retain
Mercury	0.02	0.003 (total)	Y	Retain
Silver	0.0272	1.0	N	Eliminate
Thallium	0.04	18.0	N	Eliminate
Zinc	23.11	59	Y	Retain

Aluminum, antimony, arsenic, cadmium, cobalt, copper, lead, and zinc were retained as sediment COCs for further consideration at Marsden Lake South because their concentrations were greater than the sediment RBALs, or did not have an RBAL for comparison. In addition, cadmium, lead, mercury, zinc, SVOCs, and OC Pesticides were flagged as bioaccumaltors and retained as COCs. SVOCs, such as phthalate esters have a log K_{ow} of about 8, and the pesticide DDT has a log K_{ow} of about 6 (Schnoor 1996). These log K_{ow} values indicate that these chemicals have a higher affinity for lipid than for water and are more likely to accumulate in fatty tissue of animals. Cadmium and lead are known to bioaccumulate in some receptors. Vanadium was also flagged as a bioaccumulator because it has been shown that vanadium accumulates in the bone, liver, and lungs of rats (WHO 2000). Also, zinc is ubiquitous in plant and animal tissues (Eisler 2000) and is known to accumulate in fish and other aquatic vertebrates. Exhibit 5-10 shows the comparisons used to determine whether analytes would be retained or eliminated

Exhibit 5-10. Marsden Lake South Comparison to Sediment Toxicity Values and Bioaccumulation Screen

Analyte	Maximum sediment concentration (mg/kg)	RBAL	Bioaccumulator (Yes or No)	Outcome
Aluminum	15000	None	N	Retain
Antimony	1.0	None	N	Retain
Arsenic	12.0	6	N	Retain
Cadmium	1.00	0.6	Y	Retain
Chromium	30.0	26	N	Retain
Cobalt	11.00	None	N	Retain
Copper	52.00	16	N	Retain
Lead	63.00	31	Y	Retain
Mercury	0.18	0.2 (total)	Y	Retain
Vanadium	Na	Na	Y	Retain
Zinc	140.0	120	Y	Retain
SVOCs	Na	Various	Y	Retain
OC Pesticides	Na	Various	Y	Retain

Na : Not available because the detection limit was not met

5.3.3.3 Criterion 3: Pond G

The QAPP (USACHPPM 1999) explains that the COCs from Pond G are to be assessed for risk from ingestion of drinking water and incidental sediment ingestion during drinking. This criterion is not applicable to Pond G because the benchmarks defined in the Work Plan do not specifically apply to the red fox and great blue heron. Therefore, the COCs retained for Pond G at this point are as follows:

Surface Water	Sediment
Aluminum	Aluminum
Antimony	Antimony
Arsenic	Arsenic
Barium	Chromium
Beryllium	Copper
Cadmium	Lead
Chromium	SVOCs
Cobalt	OC Pesticides (DDT & metabolites and Dieldrin)
Copper	Aroclor-1248
Lead	
Manganese	
Selenium	
Thallium	

5.3.3.4 Criterion 3: Rice Creek

Aluminum, barium, and mercury were retained as surface water COCs for further consideration at Rice Creek because their concentrations were greater than the RBALs. In addition, mercury was flagged as having the potential to biomagnify in the food chain. Of the remaining potential surface water COCs, there is evidence for cadmium transfer through various trophic levels but it suggests that only the lower trophic levels exhibit biomagnificiation (Eisler 2000). However, there is potential for cadmium to be a bioaccumulation threat to the ecological resources at the study site. Cadmium is know to accumulate in the liver and kidney of aquatic organisms. However, there is considerable variation in the ability of fish tissue to accumulate cadmium from the ambient medium (Eisler 2000). Also, the significance of relatively high cadmium residues to animal and plant health is difficult to interpret (Eisler 2000). Due to the uncertain nature of cadmium biomagnification, cadmium was flagged and retained at this point in the process to conservatively evaluate its potential to cause potential risk to fish, aquatic invertebrates, and algae. Exhibit 5-11 shows the comparisons used to determine whether analytes would be retained or eliminated.

Exhibit 5-11. Rice Creek Comparison to Water Toxicity Values and Bioaccumulation Screen

Analyte	Maximum Detected Surface Water Concentration (µg/L)						
	1st Quarter	2nd Quarter	3rd Quarter	4th Quarter	Water Quality Benchmark	Bioaccumulator (Yes or No)	Outcome
Aluminum	78.25	5.22	240	200.49	125	N	Retain
Barium	72 J	86.01	96 J	74	3.8	N	Retain
Cadmium	0.025	0.015 U	0.0470	0.035	0.66	Y	Retain
Copper	1.5	0.90	1.7	1.6	6.4	N	Eliminate
Mercury	0.042	0.0009	0.0053	0.005	0.003	Y	Retain

No sediment COC were identified for Rice Creek during the Tier I assessment.

5.3.3.5 Criterion 3: Sunfish Lake

Barium was retained as a surface water COC for further consideration at Sunfish Lake because concentrations were greater than the RBALs. In addition, even though the mercury concentration was not greater than the benchmark, it was flagged as a potential biomagnifier and was retained for further consideration. Of the remaining potential surface water COCs, no information was found on the potential for aluminum bioaccumulation or biomagnification. Evidence for cadmium transfer through various trophic levels suggests that only the lower trophic levels exhibit biomagnificiation (Eisler 2000). However, there is potential for cadmium to be a bioaccumulation threat to the ecological resources at the study site. Cadmium is known to accumulate in the liver and kidney of aquatic organisms. However, there is considerable variation in the ability of fish tissue to accumulate cadmium from the ambient medium (Eisler 2000). Also, the significance of relatively high cadmium residues to animal and plant health is difficult to interpret (Eisler 2000). Due to the uncertain nature of cadmium biomagnification, cadmium was flagged and retained at this point in the process to conservatively evaluate its potential to cause risk to fish, aquatic invertebrates, and algae. Zinc is ubiquitous in plant and animal tissues (Eisler 2000) and is known to accumulate in fish and other aquatic vertebrates. Exhibit 5-12 shows the comparison used to determine whether analytes would be retained or eliminated.

Exhibit 5-12. Sunfish Lake Comparison to Water Toxicity Values and Bioaccumulation Screen

Analyte	Maximum Detected Surface Water Concentration (µg/L)						
	1st Quarter	2nd Quarter	3rd Quarter	4th Quarter	Water Quality Benchmark	Bioaccumulator (Yes or No)	Outcome
Aluminum	17	12.72	71	35.49	125	N	Eliminate
Barium	33.016	53.01	39 J	17	3.8	N	Retain
Cadmium	0.015 U	0.020	0.0180	0.014	0.66	Y	Retain
Copper	0.847	1.5	1.8	1.4	6.4	N	Eliminate
Mercury	0.0067 U	0.0012	0.0016	0.002	0.003	Y	Retain
Zinc	2.8	2.34	1.92	0.65	59	Y	Retain

U: qualified as undetected

Aluminum, chromium, copper, lead, vanadium, and zinc were retained as sediment COCs because their maximum concentrations were greater than the sediment RBALs, or because no RBAL was available for comparison. In addition to comparing to benchmarks, the bioaccumulation potential was also examined. Of the sediment COCs, the bioaccumulation potential of chromium is difficult to predict because it is dependent on the form of chromium. It has been shown that most plant and invertebrates species die before accumulating amounts of chromium that are toxic to predators (Eisler 2000); however, fish and terrestrial vertebrates accumulate chromium in their tissues, primarily liver tissue (WHO 2000). Lead is known to bioaccumulate, especially in bone, but has not shown evidence to biomagnify up the food chain (Eisler 2000). Vanadium has been shown to accumulate in the bone, liver, and lungs of rats (WHO 2000), but it is not certain if vanadium biomagnifies. Zinc is ubiquitous in plant and animal tissues (Eisler 2000) and is known to accumulate in fish and other aquatic vertebrates. Due uncertainty surrounding the potential for bioaccumulation, lead, vanadium, and zinc were flagged as bioaccumulators.

Exhibit 5-13. Sunfish Lake Comparison to Sediment Toxicity Values and Bioaccumulation Screen

Potential COC	Maximum Tier I Sediment Concentration (µg/g)	Toxicological Benchmark (µg/g)	Bioaccumulator (Yes or No)	Outcome
Aluminum	18000	Na	N	Retain
Chromium (total)	264	26	N	Retain
Copper	126	16	N	Retain
Lead	202	31	Y	Retain
Vanadium	61.5	NA	Y	Retain
Zinc	501	120	Y	Retain

NA: Not available

5.3.3.6 Criterion 3: Round Lake

Barium was retained as a surface water COC for further consideration at Round Lake because concentrations were greater than the surface water RBALs. In addition, even though the mercury, cadmium, and zinc concentrations were not greater than the benchmarks, they were flagged as potential bioaccumulators and were retained for further consideration. There is evidence for cadmium transfer through various trophic levels but it suggests that only the lower trophic levels exhibit biomagnificiation (Eisler 2000). However, there is potential for cadmium to be a bioaccumulation threat to the ecological resources at the study site. Cadmium is known to accumulate in the liver and kidney of aquatic organisms. However, there is considerable variation in the ability of fish tissue to accumulate cadmium from the ambient medium (Eisler 2000). Also, the significance of relatively high cadmium residues to animal and plant health is difficult to interpret (Eisler 2000). Due to the uncertain nature of cadmium bioaccumulation, cadmium was flagged and retained at this point in the process to conservatively evaluate its potential to cause potential risk to fish, aquatic invertebrates, and algae. There is conflicting data on the bioaccumulation potential of barium. One study indicated that there is a clear potential for accumulation by benthic animals (Morton 1987). On the other hand, in a Proposed Rule, Denial of Petition regarding barium compounds (USEPA 1997b), there is no evidence mentioned for the bioaccumulation potential of barium. Several textbooks describe the toxicity of the barium ion but no mention is made of bioaccumulation or biomagnification; therefore, barium was not flagged as a bioaccumulator (Rand 1995; Schnoor 1996). Zinc is ubiquitous in plant and animal tissues (Eisler 2000) and is known to accumulate in fish and other aquatic vertebrates, and was flagged as a bioaccumulator. Exhibit 5-14 shows the comparison used to determine whether analytes would be retained or eliminated.

Exhibit 5-14. Round Lake Comparison to Water Toxicity Values and Bioaccumulation Screen

Analyte	Maximum Detected Surface Water Concentration (µg/L)					Bioaccumulator (Yes or No)	Outcome
	1st Quarter	2nd Quarter	3rd Quarter	4th Quarter	Water Quality Benchmark		
Barium	63.016	91.011	87 J	71	3.8	N	Retain
Cadmium	0.015 U	0.054	0.026	0.013	0.66	Y	Retain
Copper	1.0	1.6	1.4	0.86	6.4	N	Eliminate
Mercury	0.00099	0.00085	0.0010	0.0011	0.003	Y	Retain
Zinc	0.035 U	4.037	1.82	2.85	59	Y	Retain

Aluminum, cadmium, chromium, copper, lead, silver, vanadium, zinc, and PCBs were retained as sediment COCs because the maximum concentrations exceeded the RBALs, or no RBAL was available for comparison. Cadmium, lead, vanadium, and zinc were also flagged as a potential bioaccumulators. See the explanation provided above for flagging these chemicals as bioaccumulators. PCBs were retained because the maximum sediment concentration reported in Appendix P was greater than the RBAL. Exhibit 5-15 shows the results of the toxicity and bioaccumulation screen.

Exhibit 5-15 Round Lake Comparison to Sediment Toxicity Values and Bioaccumulation Screen

Potential COC	Maximum sediment Concentration (µg/g)	RBAL (µg/g)	Bioaccumulator (Yes or No)	Outcome
Aluminum	15400	NA	N	Retain
Cadmium	11.2	0.6	Y	Retain
Chromium	172	26	N	Retain
Copper	1250	16	N	Retain
Lead	299.0	31	Y	Retain
Silver	86.8	1.0	N	Retain
Vanadium	63	NA	Y	Retain
Zinc	860	120	Y	Retain
PCBs	13*	0.007	Y	Retain

NA: Not available
*See appendix P of this report (Review of Existing PCB Data From Round Lake).

5.3.4 Criterion 4 — Chemical distribution and detection frequency in sampled media

In the Tier I assessment of Marsden Lake and Pond G, this information was not evaluated. However, in this more advanced step it will be discussed. Rice Creek, Sunfish Lake, and Round Lake will also be evaluated.

The physical distribution and frequency of detection of a chemical in a site medium or exposure area can be used to remove a chemical from consideration as a COC. The premise behind this criterion is that a chemical with limited presence in a medium or exposure area is unlikely to be contacted frequently and, therefore, does not pose as great a potential ecological risk as do more frequently detected chemicals (USACE 1996b). The distribution of the chemicals present in a site or exposure area should be examined by identifying where the chemicals were and were not detected and their frequency of detection. If this evaluation indicates that the distribution of a chemical is low (i.e., it is detected in only one or a few locations) it may be reasonable to exclude it as a COC (assuming an appropriate sampling design was used), or to select the chemical as a COC for a smaller exposure area of the site. Within the smaller exposure areas, chemicals detected in five percent or fewer samples (the "5 percent" rule) may also be considered for elimination (USACE 1996b). The following factors should be considered when applying this criterion:

1. Sample size. In a small data set, a limited frequency of detection of a chemical may be more an artifact of a limited sampling design rather that the infrequent presence of the chemical.

2. Quantitation limits. If the quantitation limit achieved in one or more of the analyses are high relative to other detected concentrations, the high quantitation limit may mask the presence of chemicals.

3. Sampling design. Biased sampling plans, intended to identify "hot spots," may over-represent the occurrence of chemicals (however, see the next point).

4. Magnitude of detected concentrations. Presence of a chemical at relatively high concentrations, even at a low frequency, may indicate the occurrence of a localized area of contamination that may need to be examined as a discrete exposure area (i.e., a hot spot), and may require further sampling. What constitutes a "high" or a "low" concentration depends upon the toxicity and other properties of the chemical, the medium in which it was detected, and the site history (whether the chemical was used at the site), and requires some degree of professional judgment to identify. In addition, what constitutes a discrete exposure area is a matter of professional judgment in light of management objectives or selected assessment endpoints.

Sample sizes at the study sites were small (not more than 12 samples, except for Round Lake with 20 total samples); therefore this criterion is not a large factor in the project. Nonetheless, when a chemical was detected in only 1 of 10 samples, then this criterion was evaluated. Decisions were made on a case-by-case basis. The approach was to retain a chemical for consideration as a COC even if the detection frequency was no higher than 10%, unless other information supported a case for elimination (e.g., probable laboratory contaminant).

5.3.4.1 Criterion 4: Marsden Lake (North)

The COPCs were examined to determine if there were chemicals with relatively low detection frequencies. Silver was only detected at one location; however, one detection out of eight samples (12.5%) was not considered a relatively low detection frequency for this purpose. Due to removing samples ML-13 and ML-15 from the Marsden North data set and combining them with the Marsden South data set, the total number of samples in the Marsden North data set was 8. With this small sample size, the detection frequency criterion is not useful at this site, and therefore, no chemicals were eliminated under this criterion.

5.3.4.2 Criterion 4: Marsden Lake (South)

The COPCs were examined to determine if there were chemicals with relatively low detection frequencies. One detection out of twelve samples (8.3%) was considered a relatively low detection frequency for this purpose. Two metals in surface water were identified with a detection frequency of 8.3% (cadmium and thallium). Based on the fact that the sample size was small (12 samples), unless other supporting information was at hand regarding the detection of the chemical, it was considered unreasonable to exclude a COPC at a 8.3% detection frequency, since the traditional point of departure is a 5% frequency. Therefore, cadmium and thallium were not eliminated.

5.3.4.3 Criterion 4: Pond G

This criterion was not applied to Pond G due to the very small sample size (N=3).

5.3.4.4 Criterion 4: Rice Creek

The COPCs were examined to determine if there were chemicals with relatively low detection frequencies. One detection out of ten samples (10%) was considered a relatively low detection frequency for this purpose. No chemicals in Rice Creek were eliminated from the list based on this criterion.

5.3.4.5 Criterion 4: Sunfish Lake

The COPCs were examined to determine if there were chemicals with relatively low detection frequencies. One detection out of twelve samples (8.3%) was considered a relatively low detection frequency for this purpose. However, based on the fact that the sample size was small (12 samples), unless other supporting information was at hand regarding the detection of the chemical, it was considered unreasonable to exclude a COPC. No chemicals were eliminated based on this criterion.

5.3.4.6 Criterion 4: Round Lake

The COPCs were examined to determine if there were chemicals with relatively low detection frequencies. One detection out of twenty samples (5%) was considered a relatively low detection frequency for this purpose. No chemicals were eliminated based on this criterion.

5.3.5 Criterion 5 — Site-relatedness and comparability with background concentrations

During most risk assessments, it is important to distinguish site contamination from background levels due to anthropogenic pollution or naturally occurring substances in order to determine the presence or absence of contamination and to compare with background contributions to ecological risk (USEPA 1992a). Some chemicals detected in site media may be naturally occurring or present as a result of ubiquitous or offsite chemical use. Therefore, it is appropriate to exclude them from the risk assessment or treat them separately within the risk assessment. Background sampling is conducted to distinguish site-related contamination from naturally occurring or other non-site-related levels of chemicals. In some instances, comparison with background is insufficient to identify chemicals that are derived from other sources, despite appropriate planning of background sample locations. If such chemicals are not site-related, however, they generally should not be included in the ERA, although this decision requires professional judgment for reasons noted below. If adequate and confirmable information is available that identifies a different site as the source of a chemical, even in the absence of background information, it may be appropriate to exclude that chemical as a COC.

Some chemicals (enumerated below) should be examined for presence in background samples (USACE 1996b).

1. Metals. Certain metals are naturally occurring in most environments and should be examined for presence in background locations (e.g., arsenic and zinc). If present in site media at concentrations comparable to background concentrations, they generally can be removed from consideration as a COC.

2. Polycyclic Aromatic Hydrocarbons. This class of compounds (referred to as PAHs, or sometimes as Polynuclear Aromatics(PNAs)) are often found in the environment from a number of ubiquitous sources (including burning of wood, combustion of gasoline, natural fires, and volcanoes. It may, in some instances, be appropriate to remove PAHs from consideration as COCs, provided that it can be demonstrated that PAHs are present in types and concentrations comparable to background and that PAHs were not disposed of, used, or generated at the site.

3. Pesticides. In areas where agricultural activities are commonplace, use of pesticides (insecticides, herbicides, fungicides, etc.) and other agricultural chemicals may result in commonplace presence in the local environment. If it can be demonstrated that pesticide concentrations are comparable to background concentrations (i.e., were applied according to their intended use) and that pesticides were not disposed of at the site, they can generally be removed from consideration as a COC.

4. Phthalate Esters. When not site-related, phthalate esters are most often found in samples as a result of sampling or laboratory contamination. Phthalate esters are also ubiquitous in the environment as a result of widespread use as plasticizers.

5. Other Organic Chemicals. Most industrial organic chemicals have no natural or ubiquitous presence in the environment; however, some organic chemicals may have natural sources, such as carbon disulfide, which is formed by microbial activity. Other organic chemicals may be present as a result of upgradient, offsite sources that are not being assessed. If background locations contain organic chemicals that were not used at the site and information is available to attribute the chemical to a specific offsite source, it usually is not appropriate to consider the chemical as a COC.

Background samples are kept discrete from the site data for the purposes of assessing exposures, and are used exclusively to identify non-site-related chemicals. The most appropriate measure of background quality is obtained by the collection of background data from unaffected onsite areas or nearby, offsite areas, or reference areas. The risk assessor should be involved in the selection of background sample numbers, types, and locations as part of the risk assessment minimum data requirements, to ensure that adequate data are collected. When selecting COCs, the background data collected should be reviewed to identify whether minimum requirements have been met, or in the case of historical data, whether background measurements are adequate. The following factors should be considered.

1. Are the locations of the background samples appropriate? Appropriate background sampling locations vary with the media being examined, but should generally be offsite; hydrologically upgradient for surface water and sediments: upwind of the site at the time of measurement and under usual climate conditions for air; and in areas remote from surface water drainage for soil. Background samples should also be located away from other potential offsite sources of contamination that would not impact the site, such as other sites, roadways, etc.

   If offsite areas have the potential to contribute chemicals to the site being assessed (for example, upgradient industrial facilities), part of the goal of identifying appropriate background sample locations should be to obtain sufficient background samples to identify potential chemical contributions from offsite sources.

2. Are the background samples comparable in type to the media being examined? Background samples should be as similar as possible to the site samples being evaluated. Background sampling locations should have similar habitat and soil conditions to the onsite locations. Soil and sediment depths and stream characteristics should be comparable. The type of analyses performed on site and background samples (such as filtered versus unfiltered water, soluble versus total metals) should also be comparable.

3. Are the numbers of background measurements sufficient? Erroneous conclusions may be drawn if the number of background samples collected is insufficient to adequately describe background. The number of background samples should be specified as a minimum requirement during the project planning stage. The actual number of samples with data available should be examined to determine if the minimum requirements have been met. For historical data, professional judgment must be used to determine whether adequate background samples are available, or if additional samples are required.

   Note: Sampling data from Superfund sites have shown that data sets with fewer than 10 samples per exposure area provide poor estimates of the mean concentration (i.e., there is a large difference between sample mean and the 95% upper confidence limit (UCL) of the mean), while data sets with 10 to 20 samples per exposure area provide somewhat better estimates of the mean, and data sets with 20 to 30 samples provide fairly consistent estimates of the mean (i.e., the 95%

UCL of the mean is close to the sample mean) (USEPA 1992b). In general, the UCL approaches the true mean as more samples are included in the calculation.

Acquisition of site-specific background information is always preferable to regional or national values when examining site-relatedness and comparability to background concentrations. Literature values describing regional or national background ranges for chemicals in soil, groundwater, surface water, and sediments may be used, but only if site-specific background is unavailable. Regional or national ranges are relatively insensitive and can lead to the erroneous exclusion or inclusion of chemicals.

Determination of comparability with background can be accomplished in several ways, depending on the amount of data available. Two methods that are available are statistical evaluation and numerical comparison. A statistical evaluation is best when enough site and background samples are available to test the null hypothesis that there is no difference between the site and background mean chemical concentration at a defined level of confidence. This approach can be used when the risk assessor has defined the minimum requirements for background and site sample numbers and sampling design. Several statistical tests are available with which to determine whether the two data groups, background and site, are comparable. Guidance on statistical methodologies (USN 1998 and 1999, Gilbert 1987, Ludwig and Reynolds 1988) should be consulted for tests applicable for use in specific site conditions. Test selection depends upon data distribution (normal, non-normal), whether nondetected values are included, if appropriate proxy values are used, number of samples, and other factors. This is the most rigorous method of determining comparability.

Numerical comparisons can be made when background data are more limited in number, making a statistical comparison less meaningful. This approach may be useful when historical data with limited background samples are being used, or when minimum requirements for data collection have not been met and less than optimal numbers of background sample results are available. The following numerical comparisons can be made:

1. Compare site and background arithmetic mean concentrations.
2. Compare site and background 95% UCL of the mean concentrations.
3. Compare the range of detected concentrations in both data sets.

For the most thorough numerical comparison, all three of these factors should be examined. In a numerical comparison, the definition of "comparability" is qualitative and should be based on well-reasoned judgment. Selecting a multiple of the background mean, such as a multiple of two, has often provided a benchmark against which to define comparability. As an example of this approach, site samples could be defined as comparable if the mean concentration were less than or equal to two times the mean background concentration.

The objective of the background analysis for the TCAAP project was to focus on chemicals associated with suspected site releases rather than background chemicals or regional pollutants within subsequent phases of the risk assessment process. Surface water data were assessed with a statistical evaluation, while sediment data was assessed with both statistical evaluations (Marsden Lake only) and numerical comparisons.

Snail Lake was chosen as the primary background site for statistical comparisons with surface water for Marsden Lake, Sunfish Lake and Round Lake. For comparisons of Snail Lake to Marsden Lake North and South, only the ten samples collected from Snail Lake during September 1999 were used in the analysis. For comparisons of Snail to Sunfish and Round Lakes, quarterly samples from second, third, and fourth quarters were combined with samples SL-03, SL-06, and SL-09 from the September 1999 sampling event.

Sediment data from Marsden Lake, Sunfish Lake, and Round Lake were numerically compared to Snail Lake sediment data as well as to the Tier I sediment data provided by the USFWS.. Due to questions about the appropriateness of the reference site as compared to Marsden Lake characteristics, a numerical comparison of the confidence interval was made among Marsden Lake, Snail Lake and the USFWS Tier I sediment data. The screening values used for comparison were calculated following the method recommended by the MPCA as described in the Tier I ERA (USACHPPM 1997). This USFWS dataset used a calculate the Tier I screening values (Table A20) represents sediments collected from lakes within other parts of the Minnesota Valley National Wildlife Refuge, which were selected because they were thought to be free of moderate amounts of urban/highway impacts (USACHPPM 1997). The following algorithm was used to generate the screening value:

$$\text{Value} = \text{mean} + \left(t_{95\%} \times \text{SD}\right)$$

where, $t_{95\%}$ is the t-value based upon the degrees of freedom and the 95% confidence interval, and SD is the standard deviation. This algorithm was also used with the Tier II Snail Lake data to generate a similar screening value. When comparing the two differing screening values (see Table A44) it is apparent that the Snail Lake sediments, in general, contain higher concentrations of inorganics than the USFWS sites – which is indicative of moderately impacted lakes characteristic of a suburban/urban environments. Because the TCAAP sites are impacted by regional urban/suburban inorganic pollution in addition to possible impacts due to military hazardous waste management, the final COC selection should put more weight on the differences between the sites and Snail Lake, rather than on the relatively pristine lakes of the Minnesota Valley National Wildlife Refuge.

The following logic was used to determine if a previously selected sediment COPC (from Tier I) should be eliminated from further consideration.

1. If the upper confidence limit for the site was less than the screening value from both the Tier I and the Tier II dataset, then the chemical was eliminated as a COC.
2. If the upper confidence limit for the site was greater than the screening value from both the Tier I and the Tier II dataset, then the chemical was retained as a COC.
3. If the upper confidence limit for the site was greater than the screening value from the Tier I dataset but less than or about equal to the screening value from the Tier II dataset, then boxplots of the data were generated to visually review the differences in the distributions of the data.
    a. If the distribution of the site data (especially the interquartile range) appeared to be elevated related to the Tier II Snail Lake data, then the chemical was retained as a possible COC.
    b. If the distribution of the site data (especially the interquartile range) appeared to be equal to or less than the Tier II Snail Lake data, then the chemical was eliminated as a possible COC.

In addition to numerical comparisons of sediment data, statistical comparisons were also made with data from Snail Lake to Marsden Lake North and to Marsden Lake South. Both techniques were used for Marsden Lake sediment due to the uncertainty of Snail Lake sediment being an adequate reference site. Snail Lake was assumed to be a relatively unpolluted lake and any constituents present there should represent naturally occurring levels or regional anthropogenic pollution not associated with military site releases.

Appendix A contains tables of summary statistics for each site. Not all tables in the appendix are referred to in this section.

5.3.5.1 Criterion 5: Marsden Lake (North) and Marsden Lake (South)

Site surface water concentrations at Marsden Lake were statistically compared to background surface water concentrations at Snail Lake. Marsden Lake (North) and Marsden Lake (South) were treated independently because the hypothesis is that those areas would reflect different source areas. Prior to analysis, the data were reviewed for transcription errors and possible outliers. The number of non-detects was counted for each measured analyte. During statistical analysis any analyte concentration less than the detection limit was substituted with the method detection limit value if fewer than 50 % of the values were non-detects (USEPA 2000).

In order to determine what type of statistical test to use in comparison of site surface water_data to background data, the distribution of the data needed to be determined. As recommended for data sets with less than 25 sample points, the Shapiro-Wilk test was used for each analyte at each site to determine the distribution of the data (e.g., normal, lognormal or unknown) (Naval Facilities Engineering West 1998, Gilbert 1987). Data distributions were not determined for analytes with greater than 50% non detects and the data distribution was considered unknown. Concentrations that were believed to be outliers were tested using the Dixon test (USEPA 2000; Naval Facilities Engineering West 1998).

Once the distribution was determined, summary statistics were calculated for each analyte. For normally distributed data, means and the 95th confidence interval were calculated, for lognormally distributed data geometric means and the 95th confidence interval on the geometric mean were calculated, and for unknown data distributions the median and 5th and 95th percentiles were calculated (Gilbert 1987). The following formulas as found in Gilbert (1987), were used to calculate the confidence interval for lognormally distributed data:

$$LL_a = \exp(\bar{y} + 0.5s_y^2 + \frac{s_y H_a}{\sqrt{n-1}})$$

$$UL_{1-a} = \exp(\bar{y} + 0.5s_y^2 + \frac{s_y H_{1-a}}{\sqrt{n-1}})$$

where: $\bar{y} = \frac{1}{n}\sum_{i=1}^{n} y_i$ and $s_y^2 = \frac{1}{n-1}\sum_{i=1}^{n}(y_i - \bar{y})^2$

and $y_i = \ln(x_i)$

and H_a and H_{1-a} are obtained from the subset of Land's tables provided in Gilbert (1987).

A t-test was performed on the normal data and on the log-transformed data to compare the site to background. A Wilcoxon Rank Sum test (equivalent to a Mann-Whitney test) (Gilbert 1987) was performed on data with an unknown distribution. Data associated with less than 50% detection frequency (i.e., greater than 50% non-detects) were compared to background with a test of proportions (Gilbert 1987; USEPA 2000). If distributions from each lake were different (e.g., one normal and one lognormal), then the data from both lakes were log-transformed and comparisons were made on the log-transformed data. If the distribution from one lake was unknown and the other lake was known, then a Wilcoxon Rank Sum test was used for comparisons. If one lake had greater than 50% non detects, and the other lake did not, then a test of proportions was used.

The QAPP requires a confidence level (i.e., alpha) of 0.10. To control the overall experiment-wise error rate at 0.10, one-half the alpha level (or 0.05) was used for the comparisons of (a) Marsden Lake (North) to Snail Lake and (b) Marsden lake (South) to Snail Lake (Zar 1996). Since the objective was also to determine if the sites were strictly greater than Snail Lake, one-tailed tests were used.

Marsden Lake (North)
The remaining potential COCs in surface water (7 metals) were considered under this criterion. Table A22 provides the summary statistics for surface water analytes in Marsden Lake North and Snail Lake. Table A23 details the statistical tests used, and provides the results of the statistical comparison of Marsden Lake North surface water analytes to those in Snail Lake. No data were determined to be outliers using the Dixon Test. Marsden Lake North was found to be significantly different than Snail Lake ($p<0.01$) for the following analytes selected as COCs up to this point: aluminum, barium, lead, manganese, mercury, and zinc. Therefore based on the background comparison criterion, cadmium was the only COC eliminated from further consideration in surface water.

The remaining sediment COCs (7 metals, SVOCs, and OC Pesticides), were evaluate under this criterion. Based on the sediment numerical comparisons described in Section 5.3.6, copper, mercury, and zinc were eliminated from further consideration as sediment COCs because their confidence intervals were less than the Tier I and Tier II screening values. Antimony was retained because its confidence interval was greater than the Tier II value. The confidence intervals for arsenic and cadmium exceeded the Tier I screening value but not the Tier II value, so boxplots were created (see Table A25). Based on the information in the boxplots (see Figures D38 and D39), Marsden Lake North sediment concentrations of cadmium are within the interquartile range of the Tier I data, so cadmium was eliminated as a sediment COC. The boxplot of data for arsenic shows that the majority of Tier I data was based on non detects and that no interquartile range was available. Therefore, arsenic was retained as a sediment COC. Vanadium was retained even though it was not analyzed for in the 1999 sampling event due to uncertainties about its potential toxicity.

The sediment data were also evaluated statistically. Table A24 provides the summary statistics for sediment analytes in Marsden Lake North and Snail Lake. Table A26 details the statistical tests used, and provides the results of the statistical comparison of Marsden Lake North sediment analytes to those in Snail Lake. No data were determined to be outliers using the Dixon Test. No analytes selected as COCs up to this point were significantly different those in Snail Lake. To be conservative, even though statistical analysis showed no significant differences between lakes, antimony was retained as COC because its confidence interval was greater than the Tier II value.

Even though they were not detected in the sediments, OC pesticides remained on the list up to this point due to a failure of the chemical analysis to achieve the required target detection limit (the achieved limit was 0.05 mg/kg, while the target was 0.01 mg/kg). However, OC pesticides were detected in the background site. Heptachlor, 4,4'-DDT, and delta-BHC were each detected at one sample location (0.03, 0.083, and 0.03 mg/kg, respectively) and those that were not suffered from the same detection limit problem mentioned above for the study site. This evidence, combined with the fact that there is no positive evidence that pesticides may be site-related chemical releases, suggests that pesticides at the study site are not likely to be present, if at all, at concentrations significantly greater than in nearby lakes not impacted by military activities. Therefore, the OC pesticides were eliminated as potential sediment COCs for this study site.

The SVOCs, like the OC pesticides, remained on the sediment list up to this point due to a failure of the chemical analysis to achieve most of the required target detection limits. SVOCs were also not detected in the reference site sediments. Since the detection limits achieved were very high relative to the target detection limits, unlike the OC pesticides, the SVOCs could not be eliminated from consideration based on a comparison to background.

Therefore, based on this criterion the final COCs selected for Marsden Lake North are:

Surface Water	Sediment
Aluminum	Antimony
Barium	Arsenic
Lead	Vanadium
Manganese	SVOCs
Mercury	
Zinc	

Marsden Lake (South)

The remaining surface water COCs (9 metals) were evaluated under this criterion. Table A28 provides the summary statistics for surface water analytes in Marsden Lake South and Snail Lake. Table A29 details the statistical tests used, and provides the results of the statistical comparison of Marsden Lake South surface water analytes to those in Snail Lake. Based on the Dixon test, several outliers were found in the data set (see Appendix E, Table E9). All values determined to be outliers were found at location ML-08. The outlying values were determined to be accurate. Statistical analyses were run including and excluding ML-08, and the results were similar. Therefore, ML-08 was not excluded from the analysis presented in this section. Marsden Lake South was found to be significantly greater than Snail Lake ($p<0.01$) only for manganese. Therefore based on the background comparison criterion, aluminum, barium, cadmium, copper, lead, mercury, and zinc were eliminated from further consideration as surface water COCs.

The remaining sediment COCs (11 metals, SVOCs, and OC Pesticides) were considered under this criterion. Based on the sediment numerical comparisons described in section 5.3.6, antimony, chromium, cobalt, mercury, and zinc were eliminated from further consideration as sediment COCs because their confidence intervals were less than the Tier I and Tier II screening values. Boxplots were prepared for aluminum, arsenic, cadmium, copper, and lead because their confidence intervals were greater than the Tier I screening value but less than the Tier II Snail Lake Screening value. (see Table A31). Based on the information displayed in the boxplots, arsenic and copper were eliminated as sediment COCs because their ranges were very similar to the FWS data ranges for these chemicals (see Appendix E, Figures E1 through E5). Aluminum, cadmium, and lead were retained as sediment COCs because their interquartile ranges were not within those of the Tier I values. Vanadium was also retained because no data were available to do a comparison.

The sediment data were also evaluated statistically. Table A30 provides the summary statistics for sediment analytes in Marsden Lake North and Snail Lake. Table A32 details the statistical tests used, and provides the results of the statistical comparison of Marsden Lake South sediment analytes to those in

Snail Lake. Several data points were determined to be outliers using the Dixon Test. Barium at ML-01, copper at ML-07, and manganese at ML-05 were determined to be outliers. When ML-01 was excluded for barium, the distribution of the data changed from lognormal to normal. The analysis was performed including and excluding ML-01 and the results were similar. All of these data points were found to be accurate and therefore, all values were used in the statistical analysis. No analytes selected as COCs up to this point were significantly different those in Snail Lake. To be conservative, even though statistical analysis showed no significant differences between lakes, aluminum, cadmium, and lead were retained as sediment COCs, based on the numerical comparisons. Vanadium was retained as a COC even though it was not analyzed for during the 1999 sampling event due to the uncertainty about its potential for toxicity effects in Marsden Lake South sediments.

Even though they were not detected in the sediments, OC pesticides remained on the list up to this point due to a failure of the chemical analysis to achieve the required target detection limit (the achieved limit was 0.05 mg/kg, while the target was 0.01 mg/kg). However, OC pesticides were detected in the background site. Those that were not, suffered from the same detection limit problem mentioned above for the study site. This evidence, combined with the fact that there is on positive evidence that pesticides may be site-related chemical releases, suggests that pesticides at the study site are not likely to be present, if at all, at concentrations significantly greater than in nearby lakes not impacted by military activities. Therefore, the OC pesticides were eliminated as potential sediment COCs for this study site.

The SVOCs, similarly to the OC pesticides, remained on the sediment list up to this point due to a failure of the chemical analysis to achieve most of the required target detection limits. SVOCs were also not detected in the reference site sediments. Since the detection limits achieved were very high relative to the target detection limits, unlike the OC pesticides, the SVOCs could not be eliminated from consideration based on a comparison to background.

Therefore, based on this criterion the final COCs selected for Marsden Lake South are:

Surface Water	Sediment
Manganese	Aluminum
	Cadmium
	Lead
	Vanadium
	SVOCs

5.3.5.2 Criterion 5: Pond G

A background comparison for Pond G was not performed because a suitable comparison site was not identified. As outlined in the QAPP (USACHPPM 1999), there is no comparable reference site for Pond G. The data were instead to be used to calculate health risks to wildlife that may use the pond as a drinking water source. In order to be considered a COC, the constituent had to be detected above the detection limit and not be a nutritive substance. See Table A33 for summary statistics of detected substances in Pond G. At this point in the process the following chemicals were retained as potential COCs:

Surface Water	Sediment
Aluminum	Aluminum
Antimony	Antimony

Arsenic
Barium
Beryllium
Cadmium
Chromium
Cobalt
Copper
Lead
Manganese
Selenium
Thallium

Arsenic
Chromium
Copper
Lead
Vanadium
SVOCs
OC Pesticides
Aroclor-1248

5.3.5.3 Criterion 5: Rice Creek

The remaining surface water COCs (4 metals) were evaluated under this criterion. Quarterly site concentrations downstream of the building 103 drainage pipe were compared to site concentrations upstream of the building 103 drainage pipe where the stream enters into the installation (reference site). Prior to analysis the data was visually reviewed for transcription errors and possible outliers. The number of non-detects for each metal and water quality parameter was counted. Silver was not detected so it was excluded from the analysis.

Rice Creek presented a special case as compared to the other sites. In the fourth quarter, the upstream and downstream site had a second measurement taken at each sampling location. Visual and statistical comparisons were made between the two sets of fourth quarter data. While the two sets were not drastically different, there were some statistically significant differences between the first set and second set of fourth quarter data. The second set of data could have been averaged with the fourth quarter data, excluded, or included as a fifth set of measurements. It was decided to retain the second set of fourth quarter measurements and use them as a "fifth" quarter. This increased the sample size for each location and thereby the ability to detect differences between locations.

Values that were believed to be outliers were tested using the Dixon test (USEPA 2000; Naval Facilities Engineering West 1998). One data point was determined to be an outlier, mercury from one site during first quarter (W1-RC-SW-01). Statistical analysis was run including and excluding this value and the results were the same.

During statistical analysis, any analyte concentration less than the detection limit was substituted with the method detection limit value (USEPA 2000). Several guidelines state that the detection limit, one half the detection limit, or zero may be used in statistical analysis for values that are below the detection limit (USEPA 2000; USEPA 1989; Naval Facilities Engineering West 1998). In cases where there are many non detects, the number chosen will not affect statistical comparisons, as long as the value is consistent, because the comparisons are based on the proportion of values above and below the detection limit in each site. In this report the detection limit value was chosen for ease of use (i.e., does not have to be divided) and to be conservative in estimating the parameter mean concentrations. Different detection limits were used for the last two monitoring quarters for several metals. However, the differences did not affect the analysis because all observed values were above the detection limits.

In order to determine what type of statistical test to use in comparison of site data to background data, the distribution of the data needed to be determined. As recommended for data sets with less than 25 sample points, the Shapiro-Wilk test was used for each analyte at each site to determine the distribution of the data (e.g., normal, lognormal) (Naval Facilities Engineering West 1998, Gilbert 1987). Because there were only 2 samples per quarter for each site, all quarters within one site were combined and tested for normality. Graphs were also used to determine the distribution.

Based on the results of the number of non-detects, the distribution of the data, and the objective to compare Rice Creek downstream to Rice Creek upstream, while accounting for quarter differences and location differences, the following analyses were performed. Two-factor repeated measures analysis of variance (ANOVA) with the repeated measures being the location within the creek was generally used to assess difference between locations within the site, differences between sites and quarters, and the site by quarter interaction. If significant site-by-quarter interaction was observed, a t-test or Mann-Whitney test to compare the sites at each quarter was performed. If more than 20% non-detects were observed, a Chi-square analysis or a Fisher's Exact test was used to compare the sites regardless of quarter. The QAPP alpha level was set at 0.10 for water monitoring. Because the objective was also to determine if the downstream site was strictly greater than Rice Creek upstream, a one-tailed test was used.

Table A34 shows the summary statistics for analytes in upstream versus downstream Rice Creek. Table A36 details the statistical tests used and the results. Of the remaining analytes selected as COCs up to this point, none were found to be significantly greater in downstream Rice Creek versus upstream Rice Creek. Aluminum, barium, cadmium, and mercury were eliminated as surface water COC.

As a result of following these 5 criteria, no COCs were selected for Rice Creek.

5.3.5.4 Criterion 5: Sunfish Lake and Round Lake

Quarterly surface water concentrations at Sunfish Lake and Round Lake were each compared to quarterly background concentrations at Snail Lake. Prior to analysis the data were visually reviewed for transcription errors and possible outliers. The number of non-detects for each metal was counted. Values that were believed to be outliers were tested using the Dixon test (USEPA 2000; Naval Facilities Engineering West 1998).

During statistical analysis, any analyte concentration less than the detection limit was substituted with the method detection limit value if fewer than 50 % of the values were non-detects. Different detection limits were used for different quarters for cadmium and silver. This did no affect the analysis of these metals because all the silver values were non-detects and the majority of non-detects for cadmium were in the first quarter. All other metals were measured with the same detection limit. If a count type analysis was performed, then the higher detection limit was used as the cutoff for non-detect/detect. Seventy-five percent of the non-detects for cadmium were in the first quarter. Therefore, an analysis was performed on the counts for all quarters and one was performed for only quarters 2, 3, and 4.

In order to determine what type of statistical test to use in comparison of site data to background data, the distribution of the data needed to be determined. As recommended for data sets with less than 25 sample points, the Shapiro-Wilk test was used for each analyte at each site to determine the distribution of the data (e.g., normal, lognormal) (Naval Facilities Engineering West 1998, Gilbert 1987). Because there were only 3–5 samples per quarter for each lake, all quarters within one lake were combined and tested for normality. Graphs were also used to examine the distribution.

Based on the results of the number of non-detects, the distribution of the data, and the objective to compare Round Lake and Sunfish Lake to Snail Lake while accounting for quarter differences and location differences, the following analyses were performed. Two-factor repeated measures analysis of variance (ANOVA) with repeated measures on location within lake was used to assess difference between locations within each lake, differences between lakes and quarters, and the lake by quarter interaction. If significant lake by quarter interaction was observed, a one-way ANOVA was used to compare the lakes at each quarter. Multiple comparisons of the lakes were made using a Dunnett's test or a Mann-Whitney test, to compare Round to Snail and Sunfish to Snail. If more than 15% non-detects were observed, a Chi-square analysis or a Fisher's Exact test was used to compare the lakes regardless of quarter or location. The QAPP alpha level was set at 0.10 for water monitoring. Therefore, to control the overall experiment-wise error rate at 0.10, one half the alpha level or 0.05 was used for the comparisons of (a) Round Lake to Snail Lake and (b) Sunfish Lake to Snail Lake (Zar 1996). Because the objective was also to determine if the sites were strictly greater than Snail Lake, a one-tailed test was used.

The sediment COCs from the Tier I ERA were evaluated to determine if they should be removed as contaminants based on data collected in Snail Lake. The sediment COCs evaluated at Sunfish Lake were: aluminum, chromium, copper, lead, vanadium, and zinc; and at Round Lake they were: aluminum, cadmium, chromium, copper, lead, silver, vanadium, and zinc.

Round and Sunfish Lake sediment data were not compared statistically to the Tier II Snail Lake sediment data and the Tier I background dataset from the USFWS. This decision was based on the fact that the datasets were generated with different sampling and analytical methods (e.g., detection limit differences) and because each set of data was collected seven years apart from each other. The USFWS data were collected in 1985, the Sunfish and Round Lake data were collected in 1992, and the Snail Lake data were collected in 1999. Nonetheless, a less robust numerical evaluation comparison of these data was performed. For an explanation of how this numerical evaluation was performed see section 5.3.5.

Vanadium and silver in Round and/or Sunfish Lake were not compared to Snail Lake because these metals were not analyzed for or were undetected in Round and Sunfish Lakes during the Tier II sediment sampling.

Sunfish Lake
The remaining surface water COCs (3 metals) were considered under this criterion. Table A39 shows summary statistics for the quarterly surface water data Sunfish Lake in comparison to Snail Lake. Table A41 details the statistical tests used and shows the results. During statistical analysis, cadmium was assessed for only quarters 2, 3, and 4 due to all non-detects in the first quarter. Because lake by quarter interactions were observed, a 2 factor repeated measures ANOVA was used. The results showed cadmium to be significantly greater ($p<0.01$) during quarters 3 and 4. Therefore cadmium was retained as a surface water COC. Mercury was found to be significantly greater than Snail Lake during all quarters and was retained as a COC. Zinc was found to be significantly greater ($p<0.01$) in the third quarter and was retained as a COC. Barium was not found to be significantly different than Snail Lake during any quarter and was eliminated as a COC.

The remaining Tier I sediment COCs (6 metals) were evaluated under this criterion using numerical comparisons described in section 5.3.6. The confidence intervals for aluminum, chromium, vanadium, and zinc were all greater than both screening values, so they were retained as possible COCs (see Tables A43 & A44). The confidence interval for lead was greater than the Tier I screening value but was about equal to the Snail Lake screening value, indicating that a quick decision could not be made without also reviewing boxplots of the data. The confidence interval for copper was greater than the Tier I value but less than the Tier II value. Boxplots for copper and lead are presented in Figure H-3 (Appendix H). Lead in Sunfish Lake was retained as a possible COC because it's interquartile range was elevated relative to Snail Lake. Copper in Sunfish Lake was eliminated as a potential COC because its distribution relative to copper in Snail Lake suggests that the concentrations of copper in Sunfish Lake are no different than concentrations expected from other regional sources, indicating a lack of evidence that the copper is due to a chemical release as a result of TCAAP activity.

The final COCs selected for Sunfish Lake are as follows:

Surface Water	Sediment
Cadmium	Aluminum
Mercury	Chromium
Zinc	Lead
	Vanadium
	Zinc

Round Lake

The remaining surface water COCs (4 metals) were evaluated under this criterion. Table A46 shows summary statistics of the quarterly surface water data in comparison to Snail Lake data. Table A48 details the statistical tests used and provides the results of the analysis. During statistical analysis, cadmium was assessed for only quarters 2, 3, and 4 due to all non-detects in the first quarter. Detection limits for cadmium were different in 3rd and 4th quarter analyses as compared to the first two quarters, but all of the values were above the detection limit A Chi-square analysis comparing the number of non-detects/detects of cadmium at each lake regardless of quarter and location was performed. Snail Lake had 33% (4/12) non-detects and Round Lake had 30% (6/20) non-detects at a detection limit of 0.015 μg/L. A two-factor repeated measures ANOVA was performed on the log transformed data for Round and Snail Lakes, which showed a significant lake by quarter interaction for cadmium. Cadmium was found to be significantly greater in Round Lake than Snail Lake ($p<0.01$) in quarters 2, 3, and 4, and was therefore retained as a surface water COC. Barium in Round Lake was found to be significantly greater ($p<0.01$) than in Snail during all quarters. Zinc was significantly greater ($p<0.01$) during quarters 2 and 3 and was retained as a COC. Mercury was not significantly different,so it was eliminated as a surface water COC.

The remaining sediment COCs (8 metals and PCBs) were evaluated under this criterion using numerical comparisons described in section 5.3.6. Tables A50 & A51 show summary statistics used to perform the comparisons. Seven metals as well as PCBs had confidence intervals greater than both the Tier I and Tier II screening values and were retained as COCs. Only aluminum was less than both the Tier I and Tier II screening values and was eliminated as a sediment COC.

The final COCs selected for Round Lake are as follows:

Surface Water	Sediment
Barium	Cadmium
Cadmium	Chromium
Zinc	Copper
	Lead
	Silver
	Vanadium
	Zinc
	PCBs

5.4 SUMMARY OF STUDY SITE COCS

Table A52 lists the selected COCs for each study site based on the above analysis.

5.4.1 Marsden Lake (North)

Surface water COCs for Marsden Lake (North) are as follows: aluminum, barium, lead,_manganese, mercury, and zinc. Antimony, arsenic, vanadium, and semivolatile organic compounds (SVOCs) were retained as possible sediment COCs. Antimony was retained because its confidence interval was greater than the Tier II value, and arsenic were retained because the boxplot showed that the majority of Tier I data was based on non detects and that no interquartile range was available. Because the achieved detection limits for the SVOCs in sediment were excessively high relative to the target detection limits, it must be assumed that several of these chemicals may be present at concentrations greater than the toxicological benchmarks identified in the QAPP. Nonetheless, there remains no evidence of SVOC contamination of Marsden Lake (North) sediments. See Table A53.

5.4.2 Marsden Lake (South)

The surface water COC for Marsden Lake (South) is manganese. Aluminum, cadmium, lead, vanadium, and semivolatile organic compounds were retained as possible sediment COCs. Based on the numerical comparisons, the interquartile ranges for aluminum, cadmium, and lead were outside the range of the Tier I values. Vanadium was retained even though it was not analyzed for during the 1999 sampling event due to uncertainties about its potential to cause toxicity, that were expressed in the 1997 Tier I report. Because the achieved detection limits for the SVOCs in sediment were excessively high relative to the target detection limits, it must be assumed that several of these chemicals may be present at concentrations greater than the toxicological benchmarks identified in the QAPP. Nonetheless, there remains no evidence of SVOC contamination of Marsden Lake (North) sediments. See Table A54.

5.4.3 Pond G

Surface water COCs in Pond G are as follows: aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, copper, cobalt, lead, manganese, selenium, and thallium. Aluminum, antimony, arsenic, chromium, copper, lead, OC pesticides, SVOCs and Aroclor-1248 are the sediment COCs. See Table A55.

5.4.4 Rice Creek

No surface water COCs have been retained in Rice Creek, and the Tier I assessment recommended that no further action was necessary on Rice Creek sediment. See Table A56.

5.4.5 Sunfish Lake

Cadmium, mercury, and zinc were retained as surface water COCs for Sunfish Lake because they were statistically different than Snail Lake. In addition, the Tier I assessment recommended that aluminum, chromium, copper, lead, vanadium, and zinc be examined as sediment COCs during this Tier II assessment. A bioavailability study (USACHPPM 1998b) was conducted and the results were inconclusive, so it was suggested that a bioassay using the lake sediment be performed (see Appendix L for results). To better determine sediment COCs to use in the risk evaluation, it was decided that the Tier I sediment concentrations should be numerically evaluated against Tier II reference area concentrations. After re-evaluating the Tier I sediment COCs for Sunfish Lake, aluminum, chromium, and zinc were retained as a sediment COC because there concentrations were greater than both the Tier I and Tier II screening values. Lead was retained because its interquartile range was elevated compared to the Tier I value. Vanadium was retained as a sediment COC even though it was not analyzed for during the 1999 sampling event, due to uncertainties about it's potential to cause toxic effects that were expressed in the 1997 Tier I report. Copper was eliminated as a COC based on the boxplot of the sediment data showing that it was within the interquartile range of the Tier I screening value. See Table A57.

5.4.6 Round Lake

The final surface water COC in Round Lake are barium, cadmium, and zinc. For sediment, the Tier I assessment recommended that aluminum, cadmium, chromium, copper, silver, vanadium, and zinc be examined as sediment COCs during this Tier II assessment. In addition it was recognized that lead should have been on the Tier I list. A bioavailability study (USACHPPM 1998b) was conducted and the results were inconclusive, so it was suggested that a bioassay using the lake sediment be performed (see Appendix L for results). To better determine sediment COCs to use in the risk evaluation, it was decided that the Tier I sediment concentrations should be numerically evaluated against Tier II reference area concentrations. After re-evaluating the Tier I sediment COCs for Round Lake, the following metals were retained as sediment COCs: cadmium, chromium, copper, lead, silver, vanadium, and zinc. PCBs were also retained as sediment COCs at this point. See Table A58.

6. EXPOSURE ANALYSIS

This exposure assessment describes potential or actual co-occurrence and contact of COCs with the ecological entities of concern. The assessment is based on the selected measures of exposure and ecosystem or receptor characteristics. For each study site, the COC sources are reviewed, the magnitude and distribution of the COCs is described, and exposure profiles for each ecological entity of concern are presented. Exposure profiles are the "products" of the exposure analysis.

Exposure profiles describe the co-occurrence and contact between COCs and the ecological entities of concern. Attributes of intensity, space, and time are important for characterizing exposure (USEPA 1998). Spatial and temporal co-occurrence is useful for evaluating stressors that can cause effects without physically contacting the entities of concern or for describing the spatio-temporal "footprint" of the stressors in relation to the ecological entities of concern. However, most stressors (particularly chemicals) must contact receptors to cause an effect. Contact is quantified as the amount of chemical ingested, inhaled, or in material contacting the body (e.g., skin). Some stressors must not only be contacted, but also must be internally absorbed.

6.1 EXPOSURES AT MARSDEN LAKE (NORTH)

In this assessment, Marsden Lake (North) refers to that portion of Marsden Lake immediately surrounding Patrol Road (see figure B3). The north area of the lake was used to characterize the potential impact from contaminants that may have been introduced as a result of former activities at the Grenade Range. The selected surface water COCs for the exposure assessment of Marsden Lake (North) were aluminum, barium, lead, manganese, mercury, and zinc. The sediment COCs selected were antimony, arsenic, vanadium, and SVOCs.* SVOCs are not addressed in the exposure assessment because they were not detected due to detection limit problems (see Section 5); however, they will be discussed in the risk characterization. Exposures of the following ecological entities were characterized at Marsden Lake (North): fish, aquatic invertebrates, algal species, amphibians, waterfowl, wading birds, and aquatic mammals.

Marsden Lake (North) COCs

<u>Surface Water</u>
Aluminum
Barium
Lead
Manganese
Mercury
Zinc

<u>Sediment</u>
Antimony
Arsenic
Vanadium
SVOCs*

6.1.1 Review of COC Source(s)

The specific source of the COCs in this portion of the lake is suspected to be the Grenade Range. This hypothesis is based on information found in available records (Biang et al., 1991) and information in a geophysical survey of the lake (Padar et al., 1995), that showed no other likely source based on a lack of evidence of dumping or use of the lake for waste disposal activities. However, sufficient data are not available to establish a causal link of the elevated presence of the selected COCs due to the Grenade Range. Aluminum, barium, lead, manganese, mercury, and zinc were selected as surface water COCs primarily for their presence in concentrations that were statistically greater than the comparison site and also had a maximum detected concentration that was greater than a toxicological benchmark. Antimony, arsenic, and vanadium were selected as sediment COCs based on a numerical comparison, even though statistical comparisons showed no significant differences between the site concentrations and those at the reference area (see Section 5.3.5.3). These sediment COCs also had a maximum detected concentration that was greater than a toxicological benchmark.

The Range is no longer in operation; therefore any contamination caused by Range activity is historical. Activities that occurred at the Range are known to have caused lead, antimony, and cadmium

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* SVOCs: Semi-volatile organic compounds.

contamination of the Range soils (Alliant Techsystems 2001a). Contaminants related to Range activities could have entered Marsden Lake if projectiles extended beyond the safety fan and degraded in the water, or by runoff of contaminated soil. There is no known source of SVOCs to Marsden Lake. If activities at the Range are the original source of the COCs into Marsden Lake North, then the secondary sources for these COCs to aquatic receptors_would be the sediment, and surface runoff from contaminated soil areas within the Range. The run-off scenario currently is not likely because the Grenade Range soils that were contaminated above remediation goals were excavated and disposed of in 1999 (Alliant Techsystems 2001a). However, seasonal turnover or other chemical processes in the lake, may cause the sediment to become the secondary source of surface water COCs to aquatic receptors.

There are also other sources from outside of the study boundary that contribute to aluminum, antimony, arsenic, barium, manganese, lead, mercury, vanadium, and zinc in Marsden Lake North. Aluminum is a major constituent of the earth's crust and as a result, natural processes exceed anthropogenic releases in the environmental distribution of aluminum (ATSDR 1999). Also, surface water concentrations of aluminum generally increase after acid rain events because runoff contains aluminum released from soil. Marsden Lake is located in the northeast corner of the installation and is bordered by two county roads on the north and east, so inorganic forms of lead and manganese as a result of air deposition from combustion products of gasoline in cars and trucks would not be unlikely (ATSDR 2000). Fossil fuel burning power plants are atmospheric sources of manganese, barium and mercury that eventually become deposited in surface water. Atmospheric deposition of mercury is generally acknowledged as the major source of mercury to aquatic environments (Eisler 2000). As an example in this region, the High Bridge Generating Power Plant, which burns coal and is located in nearby St. Paul, estimated its 1999 mercury emissions to be 36 pounds (Environmental Working Group 2001).

Regardless of the true nature of the source(s) for these COCs, the design of the assessment is based on characterizing the aggregate exposure (i.e., from all sources) as measured in the surface water at this site if the presence of these metals is due in part to historical Army activities at the Range. Relative source attribution can be accomplished as a separate step during Risk Characterization, if needed (USEPA 1998).

6.1.2 Magnitude and Distribution of COCs at Marsden Lake (North)

MAGNITUDE

The measured (unfiltered) COC concentrations in the surface water were used as the primary exposure metric for fish, aquatic invertebrates, algae, and amphibians. The COC concentrations in sediment were used as the primary exposure metric for benthic organisms. The ranges of COC concentrations detected in surface water and sediment are presented in Appendix D. Summary statistics are presented in Tables A21, A22, and A24. The central tendency, detected range and selected percentiles are also presented in Exhibits within this section.

SPATIOTEMPORAL DISTRIBUTION

The spatial and temporal distribution of the COCs is as important as the measured concentrations (USEPA 1998). Because the evaluated site (i.e., the Grenade Range) is no longer in operation, the source of contamination, if any, is historical. The selected COCs are inorganics and are, therefore, environmentally persistent. Their presence through time is expected. The bioavailability or form of the COCs may change with environmental conditions (e.g., pH, temperature, nutrient loads, flow volume), but this aspect of exposure was not evaluated specifically.

A limitation in characterizing the spatial aspect of contamination at Marsden Lake was identified during site sampling and during evaluation of the analytical data. Due to problems with wind and moving cattail mats during the 1999 sampling event, two sample points were not collected as near to the suspected source area as planned. Points ML-13 and ML-15 were collected outside of the sampling area proposed in the Work Plan (USACHPPM 1999). When the analytical data results were received for surface water and sediment, several different analyses were conducted to determine if these two locations (ML-13 and ML-15) were truly representative of the extent of contamination that may be associated with the Grenade Range. Section 4.1.2.1 of this report describes those analyses in further detail. Based on professional judgment when evaluating the available data and graphical presentations, ML-13 and ML-15 were considered to be less characteristic of the potential source area in Marsden Lake North and more characteristic of the potential source area in Marsden Lake South. This is plausible given that the water flows from south to north, and if contaminants were introduced in the southern parts of the lake from the Outdoor Firing Range, they could be carried northward and deposited along the shorelines near the southwestern side of Patrol Road. This information may indicate that the spatial variability of chemical concentrations associated with the Grenade Range area is more limited to the areas closest to Patrol Road and the small and large impact areas. Therefore, based on professional judgment, the data collected from ML-13 and ML-15 were determined to be more similar in concentration and more consistent with hypothesized fate and transport mechanisms to data collected from Marsden Lake South. Based on this judgment, the data from ML-13 and ML-15 were combined with Marsden Lake South data instead of the Marsden Lake North dataset. (See figure D1).

Another limitation that arose during surface water sampling was lack of accessability to surface water due to the floating cattail mats. Holes were cut in the mat at locations ML-16, ML-17, ML-19, and ML-20 in order to expose the underlying surface water. Surface water samples were then collected the next day to allow time for settling of any root or soil particles disturbed by the cutting. The cutting of the cattails and root systems introduces uncertainty into the COC concentrations in surface water samples, such that the true magntitude and distribution of the COCs may be skewed due to a possible introduction of COCs, found in fluid in the roots and stalks that were cut, into the water collected in the hole. This disturbance may also have resulted in high TSS concentrations observed at these locations (see Appendix K, Section K1.1.4.

The spatial distribution of the surface water COCs collected from the remaining eight locations in the north area of Marsden Lake is expected to emanate from the grenade impact areas along the shoreline of Marsden Lake (see figure D1). The concentrations of the COCs may decrease with increasing distance away from these areas, but this was not evaluated due to small sample sizes. The slow flow of the aquatic system is toward the north and is expected to limit the distribution of the COCs, specifically more highly concentrating them on the north side of Patrol Road. Since one medium of concern is the surface water, the exact spatial distribution of the elevated levels of the inorganics is difficult to determine given the sampling design used in the field investigation. Generally, the highest detections of the COCs occurred in the samples to the north of Patrol road (sample locations 16, 17, 19, and 20).

The spatial distribution of the sediment COCs collected from the remaining eight locations in the north area of Marsden Lake is more difficult to predict. However, it is expected to be similar to the surface water distribution to some extent. Mainly, it is expected that sediment concentrations of COCs would be highest along the shorelines of Patrol Road and the Grenade Range, and may decrease with increasing distance away from these areas. Generally, the concentrations of sediment COCs showed little variability between sampling locations. The low variability in sediment concentrations of COCs appears to indicate a more homogeneous distribution of COCs around the impact areas. However, the surface water data suggest that the primary area of contamination, if there truly were one, would be the area north of the road just offshore from the two historical impact areas. This area of the site is completely covered by cattails, with no open water habitat.

BIOLOGICAL DISTRIBUTION

Depending upon their biochemical characteristics, in addition to their presence in surface water these COCs may be found in biological tissues at elevated concentrations. Of primary concern are COCs that tend to concentrate or accumulate in tissues rather than those that are readily metabolized and/or excreted. Section 5.3.3.1 evaluated one aspect of this phenomenon (i.e., biomagnification) during COC selection in order to ensure that chemicals known to significantly bioaccumulate would not be eliminated from further consideration unnecessarily.

The majority of information on the biological distribution of metals is concerned with the heavy metals (Cd, Cu, Fe, Pb, Mn, Hg and Zn) (Rainbow 1996). Aluminum and barium are not heavy metals and limited information was found on specific uptake of these metals by biological entities. Plant species differ considerably in their ability to take up and translocate aluminum to aboveground parts, and some plants only concentrate aluminum in the root tissue (ATSDR 1999). Aquatic invertebrate and insect information show that aluminum is taken up and bioconcentrates in the body of aquatic invertebrates and insects, in particular mayfly nymphs had the majority of aluminum deposited in the outer layers of the integument (ATSDR 1999). Fish are known to bioconcentrate aluminum in their gill tissue, but one study showed that the majority of whole-body aluminum was in the gill mucus, not sorbed to the gill tissue (ATSDR 1999). There have been assumptions made about the biomagnification potential of aluminum in the food chain, but no factual evidence was found in this search to show that birds or mammals eating fish or invertebrates with accumulated aluminum would show large body-burdens of aluminum.

Monitoring data on the levels of antimony in plants and animals is limited. Studies have shown that there was no bioaccumulation of antimony in small mammals compared with their food. Extensive studies at a smelter site also indicated that the uptake of antimony from soil in grass and subsequent translocation to the shoots is slight (ATSDR 1992). Sediment is known to be a sink for antimony but it does not appear that there is much uptake by biota.

Plants may accumulate arsenic via root uptake from soil solution, but uptake apparently is related to species differences, arsenic concentration, and other factors. In animals, evidence shows that fat accumulates more arsenic than muscle tissue (Callahan et al., 1979). It has also been shown that arsenic uptake by liver, gut, and muscle increases with arsenic concentration and other factors.

Information on barium uptake by plants is also limited. Some plants are known to bioconcentrate barium from the soil but it is unclear as to what part of the terrestrial plant stores this metal (ATSDR 1992). However, many reports indicate that aquatic macrophyte roots, such as those of cattails, accumulate larger concentrations of barium than other plant organs (Wang et al 1997). According to the Toxicity Profile for Barium (ATSDR 1992), barium is absorbed by animals and fish following ingestion, but not enough information is available to determine if barium bioaccumulates or biomagnifies in the freshwater aquatic food web.

Lead is usually found in higher concentrations in the roots of aquatic plants than in any other plant structure (Wang et al., 1997). Lead is also known to bioaccumulate in bone, but has not shown evidence of biomagnification in the food chain (Eisler 2000).

Manganese and mercury are heavy metals and general information is available concerning their distribution in biological entities. The roots and rhizomes of aquatic macrophytes are generally known to be the areas of greatest metal accumulation (Wang et al 1997). Studies of inorganic manganese accumulation by plants showed that plants nearest highways accumulate more manganese than plants near local roadways (ATSDR 2000). Algae and plankton are also known to bioconcentrate manganese and methyl mercury (ATSDR 2000; ATSDR 1994). Freshwater insect larvae appear to be net accumulators of heavy metals (i.e. manganese and mercury) with larvae of different insect families showing a range of accumulated metals concentrations bound to proteins (Rainbow 1996). Once bound to proteins these metals are not cleared from the body. Manganese has been found in fish muscle tissue but potential for biomagnification of manganese from lower trophic levels to higher ones is low (ATSDR 2000). On the contrary, organic forms of mercury quickly enter all parts of the aquatic food chain and biomagnify (ATSDR 1994).

Vanadium is taken uptake up by plant roots and will be found in plant parts that require phosphorus because plants cannot differentiate between the two elements. Vanadium has been shown to accumulate in the bone, liver, and lungs of rats (WHO 2000).

The ability of an organism to accumulate these metals does not directly show that toxic effects are occurring. Different forms of the metals (i.e., inorganic vs. organic) may change the uptake ability or body storage of each metal in each biological entity.

6.1.3 Refined Conceptual Site Model

The initial conceptual site model for Marsden Lake consisted of five risk hypotheses and was presented in the Work Plan (USACHPPM 1998), and represented in this Problem Formulation (Section 3). The refined model for Marsden Lake North still includes all five risk hypotheses (listed below), but has a refined explanation of COCs and fate and transport pathways. Figure B3-1 illustrates the hypothesized links between the possible contaminant source and potential adverse ecological effects.

1. Based upon the aquatic toxicity of surface water COCs (aluminum, barium, lead, manganese, mercury, and zinc); if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

2. Based upon the toxicity of sediment COCs (antimony, arsenic, vanadium, and SVOCs); if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment.

4. The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COCs in the surface waters, sediments, and food residues.

5. The health of aquatic mammals, such as mink or muskrat, may be impaired if they are excessively exposed to COCs in the surface waters, sediments, or residues in food.

6.1.4 Exposure Profile for Fish, Aquatic Invertebrates, and Algae at Marsden Lake (North)

Resident fish, aquatic invertebrates, and algal species co-occur with the COCs due to the presence of the COCs in the surface waters. The spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Marsden Lake (North), which was described in the previous section. These receptors will also contact the COCs while inhabiting the aquatic environment—in essence; they inhabit in the waters that contain the COCs such that they are continuously exposed. The possibility that fish may swim in and out of areas of contamination exists; however, this level of detail was not included in the analysis.

The primary mechanism for the selected COCs to induce effects in these organisms is through contact. Contact was quantified as environmental concentrations in surface water, with the assumptions that the COCs are well mixed and/or the organisms move randomly through the water.

The following attributes characterize the exposures for fish, aquatic invertebrates, algae, and benthic organisms.

- <u>Exposure duration</u>
  Entire life-cycle for most; non-emerged (i.e., juvenile) forms of some invertebrates

- <u>Exposure frequency</u>
  Continuous; however, concentration fluctuations by season are expected (but have not been quantified)

- <u>Population portion "at-risk"</u>
  The portion of the local populations that are actually exposed is not known

- <u>Contact intensity</u>
  COC concentrations in surface water are provided in Exhibit 6-1.

Exhibit 6-1. Review of Important Summary Statistics of Marsden Lake (North) COCs

Statistics	Unfiltered Surface Water Concentrations (µg/L)					
	Aluminum	Barium	Lead	Manganese	Mercury	Zinc
Distribution	Normal	Normal	Lognormal	Normal	Normal	Normal
25th percentile	21.7	57.3	0.283	329	0.0015	2.08
50th percentile	118.8	77.5	1.25	650	0.0029	3.81
75th percentile	360.8	137.5	2.63	1025	0.0074	6.84
Maximum	540. 3	200.0	7.0	1300	0.013	15.1
Central Tendency †	205.7	98.8	2.90	649	0.004	5.15
Upper Confidence Limit ‡	406.4	151.3	52.3	1070	0.0191	9.62

† Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

‡ Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentile if the distribution is unknown.

6.1.5 Exposure Profile for Benthic Organisms

Resident benthic organisms co-occur with the COCs due to the presence of the COCs in the sediments. The spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Marsden Lake North, which was described in the previous subsection. These receptors will also contact the COCs while inhabiting the benthic environment—in essence; they inhabit in the sediments that contain the COCs such that they are continuously exposed.

The primary mechanism for the selected COCs to induce effects in these organisms is through contact. Contact was quantified as environmental concentrations in sediment.

The following attributes characterize the exposures for benthic organisms.

- Exposure duration
  Entire life-cycle for most; non-emerged (i.e., juvenile) forms of some invertebrates
- Exposure frequency
  Continuous; however, concentration fluctuations by season are expected (but have not been quantified)
- Population portion "at-risk"
  The portion of the local populations that are actually exposed is not known
- Contact intensity
  COC concentrations in sediment are provided in Exhibit 6-2

Exhibit 6-2. Review of Important Summary Statistics of Marsden Lake (North) Sediment COCs

Statistics	Sediment Concentration (mg/kg)			
	Antimony	Arsenic	Vanadium	SVOCs
Distribution	Unknown	Normal	NA	NA
25th Percentile	0.50	1.73	NA	NA
50th percentile	0.50	5.75	NA	NA
75th percentile	0.55	7.05	NA	NA
Maximum	48	8.90	NA	NA
Central Tendency †	0.50	4.91	NA	NA
Upper Confidence Limit ‡	31.45	7.54	NA	NA

† Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

‡ Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentile if the distribution is unknown.

NA: Information not available (Vanadium was not analyzed for in the Tier II sampling, and there were detection limit issues with SVOCs resulting in all non detects

6.1.6 Exposure Profile for Waterfowl and Wading Birds at Marsden Lake (North)

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of surface water, sediment, and food that contain residues of the COCs. Wading bird co-occurrence in time and space with the COCs is expected to occur, because individuals of the local population may be attracted to the marshy and dense cattail areas of this site. There may also be co-occurrence for waterfowl, but it is less likely that the waterfowl species known to visit Marsden Lake North would spend as much time feeding and nesting in the area as wading birds. Some of the waterfowl known to visit the lake nest in the marshy area; however, several of them prefer to nest in trees and on dry land. It is expected that waterfowl would be found only occasionally feeding at the site, and particularly on the south side of Patrol Road, where there is open water and the cattails are not as dense. Rationale for these conclusions is provided in the subsections below.

LOCAL POPULATIONS OF WATERFOWL AND THEIR EXPOSURE CHARACTERISTICS

The exposure profile for waterfowl (Taxonomic Family *Anatidae*) was characterized using the Mallard duck (*Anas platyrhynchos*) as a representative species. Some important biological attributes of these animals are provided in Appendix N, Section N2.4.

Marsden Lake, as a whole, is used by a number of waterfowl species including Canada geese (*Branta canadensis*) and Mallard (*Anas platyrhynchos*). Common loons (*Gavia immer*) were heard but not seen during field investigations. There were also several other species of waterfowl expected or seen at Marsden Lake (see Appendix N, Section N2.4). Sufficient field information is not available to describe the populations of waterfowl at Marsden Lake. However, it is unlikely that large numbers of birds will be found at the Marsden Lake (North) site due to the lack of areas of open water. Marsden Lake (North) is predominately covered by cattails and can be considered more of a bog/marsh area than an open water lake. Mallards frequently live on small ponds or lakes bordered by trees or aquatic plants such as cattails, reeds, bulrushes, etc., where open water is limited (Terres 1996). This would indicate that Mallards, as well as several other waterfowl species might be nesting in the cattail areas. However, in dense cattail areas, such as that found in Marsden Lake (North), mallards are not likely to be found feeding because feeding habitat selection by mallards has been shown to favor large openings or access areas to open water rather than small openings in the cattail areas. (Ball and Nudds 1989). Other dabbling ducks such as the wood duck and blue-winged teal are not expected to feed in the northern part of Marsden Lake (on the north side of Patrol Road), but are most likely to be found nesting there. Of the diving ducks, most will not be found feeding in the northern section of Marsden Lake but are expected to nest there in the dense cattails along Patrol Road. A large number of loons are not likely to favor Marsden Lake (North) because they cannot fly from land and need open water in order to get a long running start for take off, a requirement that Marsden (North) does not provide (Terres 1996). Open water areas also supply the majority of food for waterfowl. These birds feed on nonvascular vegetation and aquatic insects, which are found in larger quantities in open water areas, not available in most of Marsden Lake (North). Seeds are another major food source for waterfowl, which may not be abundant at Marsden Lake (North), especially on the north side of Patrol Road.

Contact was quantified as a potential average daily dose over a given season, which is the amount of the COC ingested in food, water, and sediment per day with the use of an exposure algorithm. The algorithm incorporated measured surface water concentrations, estimated food residues, and models describing the ingestion rate. The modeling procedures and results are presented in Appendix N. In summary, the following attributes characterize the exposures for waterfowl.

- <u>Exposure frequency</u>
  The frequency of exposure for waterfowl is expected to be intermittent because the birds are not expected to spend much time feeding at the site. The ingestion pathway is of most concern for this receptor but its land use habits for the area indicate that it would utilize the site mostly for nesting and not for feeding. If the waterfowl are only nesting at the site an ingestion exposure pathway is not likely to be complete. For those birds that were to be exposed, exposure is

expected to be intermittent. It was assumed for modeling purposes, that an individual bird might visit the area to feed once or twice each season.

- Exposure duration
  In the possibility that individual waterfowl would utilize the area for feeding, the exposure duration would most likely be a few days within their seasonal stay at TCAAP.

- Population portion "at-risk"
  The portion of the local population that actually could be exposed during each season is expected to be very small. Likely to number less than ten, based on professional judgment.

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N). Only the mallard was modeled. This species was chosen as the surrogate for all waterfowl during the development of the Work Plan (USACHPPM 1998), mainly based on its presence at the site and the availability of data for modeling purposes. Modeling only this species of waterfowl does present some uncertainty. This uncertainty is a result of different site utilization preferences among duck species and slight differences in dietary preferences among the species. The differences may result in underestimation or overestimation of the potential for risk. However, despite the differences and the potential for an underestimation of potential for risk by not modeling the maximally exposed species, modeling the mallard is an acceptable approach due to the lack of necessary input values for the other species, and the overestimation of potential for risk resulting from use of other conservative assumptions.

LOCAL POPULATIONS OF WADING BIRDS AND THEIR EXPOSURE CHARACTERISTICS

The exposure profile for wading birds (Taxonomic Family *Ardeidae*) was characterized using the Great Blue Heron (*Ardea herodias*) as a representative species for that group. Some important biological attributes of these animals are provided in Appendix N, Section N2.3.

When feeding, wading birds prefer to use more open marsh areas where lower plant stem densities and dropping water levels during the dry season make them particularly attractive foraging areas (USFWS 2000). The area surrounding Patrol Road and the Grenade Range, which we refer to as Marsden Lake (North), is predominantly cattails and does not have a large open marsh area for foraging (See figure D1). When nesting, herons and egrets prefer to nest in colonies in trees 30-40 feet off the ground. There is no evidence of colonies of herons or egrets nesting at TCAAP.

Marsden Lake is bordered on the east by Lexington Ave and to the North by a residential area To the west are an open field area and the Area B wetlands. The wetlands may provide good foraging habitat and may attract wading birds that may stop at Marsden Lake (North) during their visits to Area B. _It is also highly probable that wading birds such as the American and least bittern would use the area north of Patrol Road for protective cover and roosting. In addition, bitterns in particular are expected to feed in the north area of Patrol Road to a greater extent than herons and egrets. To the south of Patrol Road is an area that has more open water and may attract more wading birds than the area north of the road. Based on the available information it is not known if the northern area of the lake will support colonies of wading birds but it is know that individual wading birds do frequent the area to forage. During the amphibian study at TCAAP (see Appendix M) a great egret and green heron were observed in Marsden Lake. At this time no determination can be made as to the wading bird population that forages at Marsden Lake.

Contact was quantified as a potential average daily dose over a given season, which is the amount of the COC ingested in food, water, and sediment per day with the use of an exposure algorithm. The algorithm incorporated measured surface water concentrations, estimated food residues, and models describing the ingestion rate. The modeling procedures and results are presented in Appendix N. In summary, the following attributes characterize the exposures for wading birds.

- Exposure frequency
  Herons have been reported in the area from March through November but most leave the area for the winter (USGS 1995). They do not nest at the site, however bitterns and other wading birds may (see Appendix N, Section N2.3). It is expected that herons and egrets will forage there occasionally. However bitterns, unlike the herons and egrets, are expected to spend more time foraging in the dense cattail area on the north side of Patrol Road. The frequency of exposure for wading birds is expected to be moderate. It was assumed for modeling purposes, that an individual bird (as represented by the heron) may visit the area once or twice in any given season (for further explanation of this statement see the Contact Intensity discussion below).

- Exposure duration
  The duration of exposure is dependent on the habitat usage of each species. In the possibility that individual herons or egrets would visit the area, the exposure duration would most likely be less than a day, and most likely no more than an hour or two. However, for the bitterns, the exposure duration is expected to be more frequent and it is estimated that they will be exposed for the majority of their stay at TCAAP.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known. Because great blues nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). Based on information in the *Wildlife Exposure Factors Handbook* (USEPA 1993) and information from various studies at TCAAP, it can be expected that 1-2 herons may frequent Marsden Lake (North). Adult herons are known to feed solitarily unless large densities of prey are found, and some are known to defend feeding territories, thus limiting the number of herons frequenting the area (USEPA 1993). Bitterns found at TCAAP also prefer a solitary lifestyle and would be expected to be present in the same density as herons at Marsden Lake (North).

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N). Only the great blue heron was modeled. This species was chosen as the surrogate for wading birds during the development of the Work Plan (USACHPPM 1999), mainly based on its presence at the site and availability of data for modeling purposes. Modeling only this species of wading bird at Marsden Lake (North) presents some uncertainty. This uncertainty is a result of differences in site usage between the herons and the bitterns, as discussed above. There are also slight differences in dietary preferences among the wading birds, such that bitterns may eat more invertebrates and frogs than herons. Despite the existence of these differences and the potential for an underestimation of potential for risk by not modeling the maximally exposed species, modeling the heron is an acceptable approach due to the lack of necessary input values for the other species, and the overestimation of potential for risk resulting from use of other conservative assumptions.

6.1.7 Exposure Profile for Aquatic Mammals at Marsden Lake (North)

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of surface water, sediment, and food that contain residues of the COCs. Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. The spatial extent of co-occurrence can be approximated by the spatial extent of the COC "footprint" within Marsden Lake (North). These wildlife will contact the COCs while inhabiting the aquatic environment. Muskrat will swim and feed on vegetation (primarily cattails) in the waters and use the sediments for building their dens. Mink will swim and feed on muskrat and other animals in the waters. The presence of the COCs in the food resources of these receptors may result in COC contact via the food chain. While the home range of an individual muskrat is small, indicating that some individuals of the Muskrat population may be continuously exposed, Mink will not be continuously exposed because they are expected to move in and out of areas of contamination.

LOCAL POPULATIONS OF MUSKRAT AND THEIR EXPOSURE CHARACTERISTICS

No Muskrat (*Ondatra zibethicus*) dens were observed in the northern portion of the lake, but a true survey was not conducted. Cattail communities are known to support high densities of muskrats, but in lake habitats shoreline length is also an important factor in muskrat density (USEPA 1993). The water level is known to fluctuate seasonally in the northern section of the lake and may not be good habitat for muskrats, even though the cattails are dense. Low water and drought make muskrats more vulnerable to predators because they need to venture farther from their lodges and in more open space to forage for food. One method of estimating the population of muskrats is to count the number of dens, then multiply by a factor between 2.8 and 5.0 (USEPA 1993), but this is a questionable method. Based on the available information about the type of habitat at Marsden Lake, no observation of dens in the area, and little information about muskrat population densities, no estimate of actual population size can be made. The area is largely covered by cattails but without a formal den count or population study at the lake, there is not enough information available to determine how many muskrats may be present. Some important biological attributes of this species are provided in Appendix N.

Contact was quantified as a potential dose, which is the amount of the COC ingested in food and surface water per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured surface water and sediment concentrations, estimated food residues, and models describing the ingestion rate. The modeling procedures and results are presented in Appendix N. In summary, the following attributes characterize the exposures for Muskrat.

- <u>Exposure duration</u>
  The Muskrat that are exposed are likely to be exposed for the majority of their life cycle because their home range is small.

- <u>Exposure frequency</u>
  For those that are exposed, they will be continuously exposed.

- <u>Population portion "at-risk"</u>
  The portion of the local population that is actually exposed is not known.

- <u>Contact intensity</u>
  The exposure modeling results are presented in Section N4 (in Appendix N).

LOCAL POPULATIONS OF MINK AND THEIR EXPOSURE CHARACTERISTICS

Marsden Lake is quite large (89.1 ha) and has less open water than surrounding lakes (e.g., Round Lake, Sunfish Lake, offsite lakes) and more cattails. Mink (*Mustela vison*) population density depends on available cover and prey (USEPA 1993). Based on the size of Marsden Lake, the size of the mink's home range, and the availability of sufficient habitat and food, it can be expected that 2-3 adult females and 1-2 adult male mink will be found inhabiting Marsden Lake, as a whole. Signs of mink activity at the lake have been observed. (e.g., tracks in the winter, etc.). Some important biological attributes of this species are provided in Appendix N.

Contact was quantified as a potential dose, which is the amount of the COC ingested in food and surface water per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured surface water and sediment concentrations, estimated food residues, and models describing the ingestion rate. The modeling procedures and results are presented in Appendix N. In summary, the following attributes characterize the exposures for Mink.

- Exposure frequency
  Since the size of Marsden Lake (North) site is small relative to the size of known Mink home ranges and the large Marsden Lake habitat area available, it is expected that Mink would most likely be exposed on a recurring, intermittent basis during any given season.

- Exposure duration
  Of the Mink that may be exposed, they are expected to either visit or locate their den at the site, indicating that the exposure duration could vary to a large degree. The duration of the intermittent exposures should last the entire season.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known. Based on literature regarding Mink habits and population densities, it is projected that one adult male, and one adult female and her young could be exposed at the site each year.

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N).

6.1.8 Exposure Profile for Amphibians at Marsden Lake (North)

Resident amphibians will co-occur with the COCs due to the presence of the COCs in the surface waters. Again, the spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Marsden Lake (North), which was described in the previous section. These receptors will also contact the COCs while inhabiting the aquatic environment—in essence; they exist, during a large portion of their life cycle, in the waters that contain the COCs such that they are continuously exposed for those periods. Juvenile amphibians will be relatively more exposed than adults due to their movement habits.

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.

LOCAL POPULATIONS OF AMPHIBIANS AND THEIR EXPOSURE CHARACTERISTICS

As part of the amphibian study conducted at TCAAP (see Appendix M) Marsden Lake as a whole was examined to determine amphibian species richness and relative abundance. Visual surveys were difficult at Marsden Lake given the extensive cattail mats concentrating along the shores. However, three species of frogs were observed by aural surveys, visual encounter surveys (VES), and/or trapping: *Bufo americanus, Rana spp,* and *Rana pipiens*. Overall, relatively few amphibians were found at TCAAP. There may be several reasons for low relative abundance due to limitations and uncertainty in the study, as well as environmental problems (see Table 6.1 in Appendix M).

- Exposure duration
  The entire Embryo and juvenile portions of the life-cycle

- Exposure frequency
  Continuous during the juvenile life stages; intermittent at most for adult amphibians.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known

- Contact intensity
  COC concentrations in surface water are provided in Exhibit 6-1.

6.2 MARSDEN LAKE (SOUTH) ANALYSIS

In this assessment, Marsden Lake (South) refers to the portion of the lake directly adjacent to the Outdoor Firing Range (see Figure B3). The south area was used to characterize the potential impact from contaminants that could have come from activities at the Outdoor Firing Range. The selected surface water COC for the exposure assessment of Marsden Lake (South) was manganese. Aluminum, cadmium, lead, vanadium, and SVOCs were retained as sediment COCs. However, SVOCs will not be addressed in the exposure assessment because they were not detected due to detection limit problems (see Section 5); however, they will be discussed in the risk characterization. Exposures of the following ecological entities were characterized: fish, aquatic invertebrates, algal species, amphibians, waterfowl, wading birds, and aquatic mammals.

Marsden Lake (South) COCs

Surface Water
Manganese

Sediment
Aluminum
Cadmium
Lead
Vanadium
SVOCs

6.2.1 Review of COC Source(s)

The specific source of any COC was suspected to be the Outdoor Firing Range. This hypothesis is based on information found in available records (Biang et al., 1991) and information in a geophysical survey of the lake (Padar et al., 1995), that showed no other likely source based on a lack of evidence of dumping or use of the lake for waste disposal activities. Manganese was selected as a surface water COC due to its presence in concentrations that were statistically greater than the comparison site and also because it had a maximum detected concentration that was greater than a toxicological benchmark. Aluminum, cadmium, lead, and vanadium were selected as sediment COCs based on a numerical comparison, even though statistical comparisons showed no significant differences between the site concentrations and those at the reference area (see Section 5.3.5.3).

The Outdoor Firing Range is no longer in operation; therefore any contamination is historical. Activities that occurred at the Range are known to have caused antimony, copper, and lead contamination of the Range soil (Alliant Techsystems 2001b). If activities at the Range were the original source of the COCs into Marsden Lake South (by introduction of small arms ammunition beyond the safe range line), then the secondary source for them would be the sediment due to surface runoff from contaminated upland Range soil areas. The run-off scenario is no longer a current source concern_because the Outdoor Firing Range soils that were contaminated above remediation goals were excavated and disposed of in 1999 (Alliant Techsystems 2001b). However, seasonal turnover or other chemical processes in the lake may cause the sediment to be a secondary source of surface water COCs. Additionally, there is no known TCAAP source of SVOCs into Marsden Lake.

In addition to Range activities, there are other sources outside of the study boundary that contribute to aluminum, cadmium, lead, manganese, and vanadium in Marsden Lake South. The most significant sources of atmospheric cadmium are smelting of nonferrous materials, incineration of municipal waste and sewage treatment sludge, which can then be deposited to water bodies (ATSDR 1999). Fossil fuel emissions and residential burning of oil or petroleum products also contributes to atmospheric cadmium, which can also be deposited to water bodies and surface soils. Further explanation of sources of aluminum, lead, and manganese in the surface water are reviewed in section 6.1.1.

Regardless of the true nature of the source(s) for the COCs, the design of the assessment is based on characterizing the aggregate exposure (i.e., from all sources) as measured in the surface water at this site if the presence of these metals is due in part to historical Army activities at the Range. Relative source attribution can be accomplished as a separate step, if needed (USEPA 1998).

6.2.2 Magnitude and Distribution of COCs at Marsden Lake (South)

MAGNITUDE

The measured (unfiltered) COC concentrations in the surface water were used as the primary exposure metric for fish, aquatic invertebrates, algae, and amphibians. The COC concentrations in sediment were used as the primary exposure metric for benthic organisms. The range of COC concentrations detected is presented in Appendix E. Summary statistics are presented in Tables A27, A28, and A30. The central tendency, detected range and selected percentiles are also presented in Exhibits found in this section.

SPATIOTEMPORAL DISTRIBUTION

The spatial and temporal distribution of aluminum, cadmium, lead,_manganese, and vanadium are as important as the measured concentrations (USEPA 1998). Because the evaluated site (i.e., the Outdoor Firing Range) is no longer in operation, the contamination, if any, is historical. The COCs are inorganics and are environmentally persistent, so their presence through time is expected. The bioavailability or form of the COCs may change with environmental conditions (e.g., pH, temperature, nutrient loads, flow volume), but this aspect of exposure was not evaluated specifically; however, the uptake and toxicity of the COCs were evaluated for wildlife.

The spatial distribution of COCs is expected to emanate from the range impact areas along the western shoreline of Marsden Lake (see figure B3 and also see Appendix E). The concentrations may decrease with increasing distance away from these areas, but this was not evaluated due to small sample sizes. The slow flow of the aquatic system is toward the north and is expected to limit the distribution of excess concentrations of COCs to areas of the western shoreline adjacent to the Firing Range. Unfortunately, the exact spatial distribution of the elevated levelof manganese is difficult to determine given the sampling design used in the field investigation, because manganese is a surface water COC.

BIOLOGICAL DISTRIBUTION

Manganese is a heavy metal and general information is available concerning its distribution in biological entities. The roots and rhizomes of aquatic macrophytes are generally known to be the areas of greatest metal accumulation (Wang et al 1997). Studies of inorganic manganese accumulation by plants showed that plants nearest highways accumulate more manganese than plants near local roadways (ATSDR 2000). Algae and plankton are also known to bioconcentrate manganese (ATSDR 2000; ATSDR 1994). Freshwater insect larvae appear to be net accumulators of heavy metals (i.e. manganese) with larvae of different insect families showing a range of accumulated metals concentrations bound to proteins (Rainbow 1996). Once bound to proteins these metals are not cleared from the body. Manganese has been found in fish muscle tissue but potential for biomagnification of manganese from lower trophic levels to higher ones is low (ATSDR 2000).

Cadmium is a naturally occurring element, and its presence has been detected in more than 1000 species of aquatic and terrestrial flora and fauna. Many reports indicate that aquatic macrophytes accumulate large concentrations of cadmium in the roots as compared to the shoots (Wang et al 1997). There is considerable variation in the ability of teleost tissues to accumulate cadmium, but the gills seem to be the primary site of accumulation (Eisler 2000). As for vertebrates, cadmium tends to concentrate in the viscera, especially the liver and kidneys (Eisler 2000).

See Section 6.1 for a discussion on the biological distribution of aluminum, lead, and vanadium.

6.2.3 Refined Conceptual Site Model

The initial conceptual model for Marsden Lake consisted of five risk hypotheses and was presented in the Work Plan (USACHPPM 1998) and represented in this Problem Formulation (Section 3). The refined model for Marsden Lake South still includes all five risk hypotheses (listed below), but has a refined explanation of fate and transport pathways, as well as specific COCs. Figure B3-2 illustrates the hypothesized links between the possible contaminant sources and potential adverse ecological effects.

1. Based upon the aquatic toxicity of manganese; if it is present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species

2. Based upon the toxicity of the sediment COCs (aluminum, cadmium, lead, vanadium, and SVOCs); if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment.

4. The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COCs in the surface water, sediments, and COC residues in food.

5. The health of aquatic mammals, such as mink or muskrat, may be impaired if they are excessively exposed to COCs in the surface waters and sediments, or COC residues in food.

6.2.4 Exposure Profile for Fish, Aquatic Invertebrates, and Algae, at Marsden Lake (South)

Resident fish, aquatic invertebrates, and algal species co-occur with the manganese due to the presence of it in the surface waters. The spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Marsden Lake (South), which was described in the previous section. These receptors will also contact manganese while inhabiting the aquatic environment—in essence; they inhabit the waters that contain manganese such that they are continuously exposed. The possibility that fish may swim in and out of areas of contamination exists; however, this level of detail was not included in the analysis.

The primary mechanism for manganese to induce effects in these organisms is through contact. Contact was quantified as environmental concentrations in surface water, with the assumptions that the manganese is well mixed and/or the organisms move randomly through the water.

The following attributes characterize the exposures for fish, aquatic invertebrates, and algae.

- Exposure duration
  Entire life-cycle for most; non-emerged (i.e., juvenile) forms of some invertebrates

- Exposure frequency
  Continuous; ,however, concentration fluctuations by season are expected (but have not been quantified)

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known

- Contact intensity
  COC concentrations in surface water are provided in Exhibit 6-3.

Exhibit 6-3. Review of Manganese Surface Water Statistics of Marsden Lake (South)

Statistic	Unfiltered Water Concentration (µg/L)
	Manganese
Distribution	Unknown
25th percentile	27.8
50th percentile	48.0
75th percentile	50.0
Maximum	370
Central Tendency †	48.0
Upper Confidence Limit ‡	197.3

† Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

‡ Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

6.2.5 Exposure Profile for Benthic Invertebrates at Marsden Lake (South)

Resident benthic organisms co-occur with the COCs due to the presence of the COCs in the sediments. The spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Marsden Lake South, which was described in the previous subsection. These receptors will also contact the COCs while inhabiting the benthic environment—in essence; they inhabit in the sediments that contain the COCs such that they are continuously exposed.

The primary mechanism for the selected COCs to induce effects in these organisms is through contact. Contact was quantified as environmental concentrations in sediment.

The following attributes characterize the exposures for benthic organisms.

- Exposure duration
  Entire life-cycle for most; non-emerged (i.e., juvenile) forms of some invertebrates
- Exposure frequency
  Continuous; however, concentration fluctuations by season are expected (but have not been quantified)
- Population portion "at-risk"
  The portion of the local populations that are actually exposed is not known
- Contact intensity
  COC concentrations in sediment are provided in Exhibit 6-4.

Exhibit 6-4. Review of Sediment COC Summary Statistics at Marsden Lake (South)

Statistic	Sediment Concentration (mg/kg)				
	Aluminum	Cadmium	Lead	Vanadium	SVOCs
Distribution	Normal	Unknown	Normal	NA	NA
25th Percentile	8,325.0	0.50	38.25	NA	NA
50th Percentile	10,200.0	0.50	44.50	NA	NA
75th Percentile	12,500.0	1.15	48.75	NA	NA
Maximum	15,000.0	1.60	63.0	NA	NA
Central Tendency †	10,166.7	0.50	43.67	NA	NA
Upper Confidence Limit ‡	12,372.7	1.49	51.41	NA	NA

† Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

‡ Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

NA: Not available. (Vanadium was not analyzed for in the Tier II sampling, and there were detection limit issues with SVOCs resulting in all non detects

6.2.6 Exposure Profile for Waterfowl and Wading Birds at Marsden Lake (South)

The exposure profile for waterfowl (Taxonomic Family *Anatidae*) was characterized using the Mallard duck (*Anas platyrhynchos*) as a representative species. The exposure profile for wading birds (Taxonomic Family *Ardeidae*) was characterized using the Great Blue Heron (*Ardea herodias*) as a representative species for that group. Biological attributes of these animals and the groups they represent are provided in Appendix N.

Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. The spatial extent of co-occurrence can be approximated by the spatial extent of the contaminant "footprint" within Marsden Lake (South). These wildlife will contact the COCs while inhabiting the aquatic environment. Waterfowl will swim and feed in the waters. Wading birds will stand and feed in the waters. Also, the presence of the COCs in the food resources of these receptors may result in contaminant contact via the food chain. These species will not be continuously exposed because they are expected to move in and out of areas of contamination.

The primary mechanism for effects in these organisms is through ingestion of surface water, sediment and food that contain residues of COCs. Contact was quantified as a potential dose, which is the amount of the COC ingested in food and surface water per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured surface water concentrations, estimated food residues, and models describing the ingestion rate.

LOCAL POPULATIONS OF WATERFOWL AND THEIR EXPOSURE CHARACTERISTICS

Marsden Lake is used by a number of waterfowl species including Canada geese (*Branta canadensis*) and Mallards (*Anas platyrhynchos*). Common loons (*Gavia immer*) were heard but not seen during field investigations. Trumpeter swans are also known to be present in the south portion of Marsden Lake. Unlike Marsden Lake (North), the south portion of the lake near the western shoreline does contain a fair amount of open water that will attract waterfowl because it provides better foraging area for dabbling, seed eating ducks than Marsden Lake (North). More open water is also more attractive to common loons. Sufficient field information is not available to describe the population size of waterfowl at Marsden Lake (South). However, it is expected that more waterfowl will be found in Marsden Lake (South) than in Marsden Lake (North). Dabbling ducks such as the wood duck and blue-winged teal are expected to feed in the waters of Marsden Lake (South) and nest in the cattails near the shoreline or on the floating cattail mats in the lake. Diving ducks, particularly loons, are also expected in this portion of the lake due to the availability of more open water.

The following attributes characterize the exposures for waterfowl.

- <u>Exposure duration</u>
  Canada geese mallard ducks, and the other waterfowl species listed in Appendix N Section N2.4 would be exposed for the majority of their life cycle because these birds have been observed nesting in the TCAAP area throughout the year. Common loons are expected to be exposed seasonally during their juvenile and adult life because they are known to frequent the area during migration to and from breeding grounds (March-May and September-November) (USGS 1995).

- <u>Exposure frequency</u>
  Exposure of mallards, Canada geese, and other waterfowl species is expected to be continuous, but for the common loon exposure is expected to be intermittent, only a few times during their migratory season.

- <u>Population portion "at-risk"</u>

The portion of the local populations that are actually exposed is not known. Without field data an accurate population estimate cannot be made.

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N). Only the mallard was modeled. This species was chosen as the surrogate for all waterfowl during the development of the Work Plan (USACHPPM 1998), mainly based on its presence at the site and the availability of data for modeling purposes. Modeling only this species of waterfowl does present some uncertainty. This uncertainty is a result of different site utilization preferences among duck species and slight differences in dietary preferences among the species. The differences may result in underestimation or overestimation of the potential for risk. However, despite the differences and the potential for an underestimation of potential for risk by not modeling the maximally exposed species, modeling the mallard is an acceptable approach due to the lack of necessary input values for the other species, and the overestimation of potential for risk resulting from use of other conservative assumptions.

LOCAL POPULATIONS OF WADING BIRDS AND THEIR EXPOSURE CHARACTERISTICS

The exposure profile for wading birds (Taxonomic Family *Ardeidae*) was characterized using the Great Blue Heron (*Ardea herodias*) as a representative species for that group. Some important biological attributes of these animals are provided in Appendix N, Section N2.3. All of the wading bird species discussed in appendix N are expected to be present at Marsden Lake (South).

When feeding, herons and egrets prefer to utilize more open marsh areas, where lower plant densities and dropping water levels during the dry season make them particularly attractive foraging areas (USFWS 2000). However, bitterns are known to prefer the more secluded, dense cattail areas. The wading birds do not prefer deep water but rather areas where they can walk through because their method of obtaining food is by stabbing it with their beaks. Fish is the preferred food of most wading birds (USEPA 1993), and may be more abundant in the area adjacent to the Outdoor Firing Range, which we refer to as Marsden Lake (South), due to more permanent volumes of water being retained in that area. There are large areas of cattails in the south section of the lake but not as dominant as that in the northern section. There are also floating mats of cattails, which are easily moved around by the wind and change the outline of open water. It is not known how much this change in open water will affect bird populations at Marsden Lake (South). Marsden Lake (South) is bordered on the east by Lexington Avenue and a residential area, which is not likely habitat for wading birds. To the west is the firing range, which is also not likely to be preferred habitat for wading birds. Even though wading birds are not known to nest at Marsden Lake (South) they have been observed there (see Appendix M) and do feed at the lake. However, the available information is limited and it is not known how many birds the southern area of the lake will support. At this time no determination can be made as to the size of the wading bird population at Marsden Lake.

The following attributes characterize the exposures for wading birds.

- Exposure duration
  The duration of exposure is dependent on the habitat usage required by each species of wading bird. The great blue heron is known to nest in the vicinity of TCAAP and is expected to be exposed throughout its life cycle (USGS 1995). However, due to foraging needs of the heron and egret, the exposure duration is expected to be less than a day, for no more than one or two hours, during one season in the TCAAP vicinity. In addition, the exposure duration for the bitterns is

expected to be more frequent than that of the herons, and is estimated that they will be exposed for the majority of their seasonal stay at TCAAP.

- Exposure frequency
  Herons have been reported in the area from March through November and most leave the area for the winter (USGS 1995). Also, there is no evidence of herons nesting on TCAAP. However, bitterns and other wading birds may nest at Marsden Lake (South) in the cattail areas. It is expected that herons and egrets will forage there occasionally during their stay in the TCAAP vicinity. For those birds that will be exposed, exposure is expected to be intermittent. Because herons do not nest at the site they are expected to forage there only occasionally. However, bitterns may have a more frequent exposure if in fact they are nesting at Marsden Lake (South).

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known. Because great blues nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). It is difficult to determine the amount of foraging habitat available at Marsden Lake (South) without field data. However, based on the available information it is possible that 4-5 wading birds may dominate at Marsden Lake (South). Because bitterns prefer a solitary lifestyle there are expected to be a smaller number, possible only one pair in Marsden Lake (South).

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N). Only the great blue heron was modeled. This species was chosen as the surrogate for wading birds during the development of the Work Plan (USACHPPM 1999), mainly based on its presence at the site and availability of data for modeling purposes. Modeling only this species of wading bird at Marsden Lake (South) does present some uncertainty. This uncertainty is a result of differences in site usage between the herons and the bitterns, as discussed above. There are also slight differences in dietary preferences among the wading birds, such that bitterns may eat more invertebrates and frogs than herons. Despite the existence of these differences and the potential for an underestimation of potential for risk by not modeling the maximally exposed species, modeling the heron is an acceptable approach due to the lack of necessary input values for the other species, and the overestimation of potential for risk resulting from use of other conservative assumptions.

6.2.7 Exposure Profile for Aquatic Mammals at Marsden Lake (South)

The exposure profile for aquatic mammals was characterized with models for the Muskrat (*Ondatra zibethicus*) and the Mink (*Mustela vison*). Exposure of muskrats to all COCs was evaluated, while exposure of mink was only intended to evaluate exposure to mercury (see Appendix N). However, mercury was not selected as a COC at Marsden Lake (South), so mink were not evaluated here. Biological attributes of these animals and the groups they represent are provided in Appendix N.

Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. The spatial extent of co-occurrence can be approximated by the spatial extent of the COC "footprint" within Marsden Lake (South). These wildlife will contact COCs while inhabiting the aquatic environment. Muskrat will swim and feed on vegetation (primarily cattails) in the waters and use the sediments for building their dens. Mink will swim and feed on muskrat and other animals in the waters. The presence of COCs in the food resources of these receptors may result in COC contact via the food chain. While the home range of an individual muskrat is small, indicating that some individuals of the Muskrat population may be continuously exposed, Mink will not be continuously exposed because they are expected to move in and out of areas of contamination.

The primary mechanism for COCs to induce effects in these organisms is through ingestion of surface water and food that contain residues of the COCs. Contact was quantified as a potential dose, which is the amount of COCs ingested in food and surface water per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured surface water concentrations, estimated food residues, and models describing the ingestion rate.

LOCAL POPULATIONS OF MUSKRAT AND THEIR EXPOSURE CHARACTERISTICS

Several dens were observed in the southern portion of the lake, but a true survey was not conducted. Cattail communities are known to support high densities of muskrats, but in lake habitats shoreline length is also an important factor in muskrat density (USEPA 1993). One method of estimating the population of muskrats is to count the number of dens, then multiply by a factor between 2.8 and 5.0 (USEPA 1993), but this is a questionable method. Based on habitat preference information, it appears that more muskrats can be expected in Marsden (South) than in Marsden (North), and using the above estimation method it is possible that between 12 and 20 muskrats may be present. However, because the available information about the type of habitat at Marsden Lake and about muskrat population densities is limited, no accurate estimate of actual population size can be made. The area is largely covered by cattails, which is preferred habitat, but without a formal den count or population study at the lake, there is not enough information available to determine how many muskrats may be present.

The following attributes characterize the exposures for Muskrat.

- Exposure duration
  The Muskrat that are exposed are likely to be exposed for the majority of their life cycle because their home range is small.

- Exposure frequency
  For those that are exposed, they will be continuously exposed.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known.

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N).

LOCAL POPULATIONS OF MINK AND THEIR EXPOSURE CHARACTERISTICS

Marsden Lake is quite large (89.1 ha) and has less open water than surrounding lakes (e.g., Round Lake, Sunfish Lake, offsite lakes) and more cattails. Mink population density depends on available cover and prey (USEPA 1993). Based on the size of Marsden Lake, the size of the mink's home range, and the availability of sufficient habitat and food, it can be expected that 2-3 adult females and 1-2 adult male mink will be found inhabiting Marsden Lake as a whole. Signs of mink activity at the lake have been observed. (e.g., tracks in the winter, etc.). However, exposure of mink to mercury will not be model because mercury was not selected as a COC for this site.

6.2.8 Exposure Profile for Amphibians at Marsden Lake (South)

Resident amphibians will co-occur with mercury due to the presence of mercury in the surface waters. Again, the spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Marsden Lake (South), which was described in the previous section. These receptors will also contact mercury while inhabiting the aquatic environment—in essence; they exist, during a large portion of their life cycle, in the waters that contain mercury such that they are continuously exposed for those periods. Juvenile amphibians will be relatively more exposed than adults due to their movement habits.

The primary mechanism for mercury 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 mercury exposure is random within the site in that the populations of amphibians would likely lay eggs in a random spatial pattern.

LOCAL POPULATIONS OF AMPHIBIANS AND THEIR EXPOSURE CHARACTERISTICS

As part of the amphibian study conducted at TCAAP (see Appendix M) Marsden Lake as a whole, not north and South sections was examined to determine amphibian species richness and relative abundance. Visual surveys were difficult at Marsden Lake given the extensive cattail mats concentrating along the shores. However, three species of frogs were observed by aural surveys, visual encounter surveys (VES), or trapping: *Bufo americanus, Rana spp,* and *Rana pipiens*. Overall, relatively few amphibians were found at TCAAP. There may be several reasons for low relative abundance due to limitations and uncertainty in the study, as well as environmental problems (see Table 6.1 in Appendix M).

- Exposure duration
  The entire Embryo and juvenile portions of the life-cycle

- Exposure frequency
  Continuous during the juvenile life stages; intermittent at most for adult amphibians.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known

- Contact intensity
  COC concentrations in surface water and sediment are provided in Exhibit 6-3 and Exhibit 6-4.

6.3 POND G ANALYSIS

The selected COCs for the exposure assessment of Pond G include 13 surface water inorganics, 6 sediment inorganics, as well as Aroclor 1248, SVOCs and OC pesticides in sediment (Table A55). Exposures of the following ecological entities were characterized: Red fox/Coyote and wading birds.

6.3.1 Review of COC Source(s)

Pond G COCs

Surface Water
Aluminum
Antimony
Arsenic
Barium
Beryllium
Cadmium
Chromium
Cobalt
Copper
Lead
Manganese
Selenium
Thallium

Sediment
Aluminum
Antimony
Arsenic
Chromium
Copper
Lead
Vanadium
SVOCs
OC Pesticides
Aroclor 1248

The specific source(s) of the COCs (listed in the textbox to the right) in Pond G is suspected to be Site G (uncontrolled landfill), Site F (former burning area), or both. The source areas are no longer in operation; therefore the contamination is historical. The secondary sources for these COCs would be the sediment and surface runoff from contaminated upland soil areas. Groundwater seeps have also been suggested as a possible source.

Soils and sediments naturally contain many of the detected inorganic COCs. Since a comparable reference location for Pond G was not used in the selection of COCs (USACHPPM 1999), it is difficult to determine whether many of the inorganics present in the Pond are actually contaminants. Regardless of the true nature of the source(s) for these COCs, the design of the assessment is based on characterizing the aggregate exposure (i.e., from all sources) as measured in the surface water at this site. Relative source attribution can be accomplished as a separate step, if needed (USEPA 1998). Due to detection limit problems, SVOCs will not be quantitatively discussed in this section, but will be discussed in Section 8.

6.3.2 Magnitude and Distribution of COCs at Pond G

MAGNITUDE

The measured COC concentrations in sediment and unfiltered surface water were used as the primary exposure metric for Red foxes and wading birds that may use the Pond as drinking water source. The range of COC concentrations detected is presented in Appendix F. Summary statistics are presented in Table A33. Because the sample size was so small and the pond is also very small (20 x 80 ft), the central tendency, detected range and selected percentiles are not presented as for other sites.

SPATIOTEMPORAL DISTRIBUTION

Based on the small size of the Pond and its physical characteristics, it is assumed that the spatial and temporal distribution of the COCs is basically uniform. The entire Pond is assumed contaminated and concentrations of these COCs are not likely to change significantly for the foreseeable future because they all are environmentally persistent and a continuing source of the COCs is not believed to be present. The bioavailability or form of the COCs may change with environmental conditions (e.g., pH, temperature, nutrient loads, flow volume), but this aspect of exposure was not evaluated specifically.

BIOLOGICAL DISTRIBUTION

Because the pond is very small in size, it does not support any fish. However, it may support some amphibians and invertebrates. Even if these organisms are occasionally present, their numbers would not support the normal dietary requirements of wading birds that may use the pond as a drinking water source. Therefore, exposure through the foodchain is expected to be insignificant and was not evaluated in this assessment.

6.3.3 Refined Conceptual Site Model

The initial conceptual model for Pond G consisted of one risk hypothesis (restated below) and was presented in the Work Plan (USACHPPM 1998) and represented in this Problem Formulation (Section 3). The revised conceptual model for Pond G is the same as before with the exception of more information on fate and transport and a list of specific COCs. Figure B3-3 illustrates the links between the hypothesized contaminant source(s) and potential adverse ecological effects.

- Based upon the toxicity of metals, OC Pesticides, SVOCs, and Aroclor 1248 present in the water or sediment of the pond; if they exist at sufficient concentrations then adverse toxic health effects may occur in foxes and wading birds that use the pond as a drinking water source, and may incidentally ingest sediment.

6.3.4 Exposure Profile for Wading Birds that May Visit Pond G

The exposure profile for wading birds (Taxonomic Family *Ardeidae*) was characterized using the Great Blue Heron (*Ardea herodias*) as a representative species for that group. Biological attributes of these animals and the groups they represent are provided in Appendix N.

Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. On an occasional basis, wading birds have been observed at the Pond. These birds will contact the COCs while standing (and possibly drinking from) the Pond. These species will not be continuously exposed because they are expected to frequent the pond on a very limited basis because of the availability of other higher quality habitat nearby.

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of surface water while drinking and incidental ingestion of sediment that may be suspended in the water during drinking activities. Contact was quantified as a potential dose, which is the amount of the COC ingested in surface water per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured (unfiltered) surface water concentrations with a model describing the ingestion rate.

The following attributes characterize the exposures for wading birds at Pond G.

- Exposure duration
  Actual duration of exposure relative to the lifetime is unknown. For modeling purposes, it can be assumed that individuals may visit the site for a large portion of their lifespan.

- Exposure frequency
  The frequency at which individual birds will visit the pond is not known. It is assumed to be intermittent.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known. It is assumed that very few individuals, if any, of different wading bird species will visit the site. It is expected that 1-3 herons may frequent Pond G in any given year.

- Contact intensity

The exposure modeling results are presented in Section N4 (in Appendix N).

6.3.5 Exposure Profile for Red Foxes / Coyotes

During the development of the Tier II risk assessment work plan (USACHPPM 1999) the Red fox was selected as a species of concern. At that time a Red fox population existed on the facility and foxes had been observed to frequent the Pond G area. Biological attributes of the Red fox are provided in Appendix N. However, in recent years, 1999, 2000, and 2001, there were no reported Red fox sightings, no active dens, and no reported counts at scent stations (Speaker, personal communication). The last year for recorded Red fox sightings at TCAAP was 1998. It is speculated that an increased coyote population has resulted in a decrease in Red fox numbers (Speaker, personal communication). The risk assessment model was implemented for a Red fox population should there be one in the future. A separate model for Coyotes was not developed because the output for a Coyote would be very similar to that of Red fox, as they are both the top mammalian carnivores at the installation.

Individuals from the installation's local population of Red fox are attracted to this aquatic site, indicating that some degree of co-occurrence in time and space does occur. These fox will contact the COCs while standing and possibly drinking from the Pond. These species will not be continuously exposed because they are expected to frequent the pond on a limited basis because of the availability of other quality habitat nearby and the size of their home range.

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of surface water while drinking and incidental ingestion of sediment that may be suspended in the water during drinking activities and that may be ingested while grooming their coats to free sediments adhered to their fur. Contact was quantified as a potential dose, which is the amount of the COC ingested in surface water and sediment per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured sediment and (unfiltered) surface water concentrations with a model describing the ingestion rate.

The following attributes characterize the exposures for Red fox at Pond G.

- Exposure duration
  It is expected that individuals of the Red fox population on the installation will utilize the site for their entire lifetime.

- Exposure frequency
  The frequency at which individual foxes will use the Pond is not known. It is assumed to be intermittent.

- Population portion "at-risk"
  The portion of the local populations actually exposed is assumed to be 100%, since the home range includes the entire installation.

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N).

6.4 RICE CREEK ANALYSIS

Surface water was analyzed for previously selected Tier II COCs (aluminum, barium, cadmium, copper, mercury, and silver). Based on the collected data (Section 4) and the implementation of the revised COC selection criteria (Section 5), each of these previous COCs were eliminated from further consideration. This means that no chemical stressors due to TCAAP-related activities have been identified and further risk assessment is not warranted at this study site.

6.5 SUNFISH LAKE ANALYSIS

Surface water was analyzed for previously selected Tier II COCs (aluminum, barium, cadmium, copper, mercury, silver, and zinc). Based on the collected data (Section 4) and the implementation of the revised COC selection criteria (Section 5), cadmium, mercury, and zinc were retained for further consideration as surface water COCs. In addition, the Tier I assessment recommended that aluminum, chromium, copper, lead, vanadium, and zinc be examined as sediment COCs during this Tier II assessment.

Exposures of the following ecological entities were characterized: fish, aquatic invertebrates, and algal species, benthic organisms, amphibians, wading birds, and aquatic mammals.

6.5.1 Review of COC Source(s)

The specific source of the COCs in Sunfish Lake is Site H-1, which contains an old unpermitted landfill that also included metal debris scattered on the bottom of a portion of the sediments along the north central shore of the lake. The site is no longer used and the upland areas have been remediated in large part because they contained concentrations of metals (i.e., arsenic, antimony, copper, and lead) that posed excessive risks to human health. Therefore, any contamination is historical in nature. Aluminum, chromium, lead, vanadium, and zinc were selected as sediment COCs based on a numerical re-evaluation of the sediment data (see Section 5.3.5). Additional sediment chemistry analysis was not conducted for the Tier II assessment. Instead, the data used in the original Tier I assessment was compared with the new reference site information (Snail Lake), as well as the information from the original sediment reference screen (see Section 5.3.5.6).

Sunfish Lake COCs

Surface Water
Cadmium
Mercury
Zinc

Sediment
Aluminum
Chromium
Lead
Vanadium
Zinc

There are also other sources from outside of the study boundary that contribute to aluminum, cadmium, chromium, lead, mercury, vanadium, and zinc. The major anthropogenic source of chromium, lead, vanadium, and zinc to the sediment is atmospheric deposition. The Twin Cities area is industrialized and lead, copper, and steel smelters were scattered throughout Minnesota, as well as trash burning facilities. These industries emit(ed) metals to the atmosphere that are/(were) eventually deposited/precipitated to surface water and end(ed) up in the sediment. Research on anthropogenic sources of chromium to the environment show that atmospheric emissions from coal combustion, metal-producing industries, municipal incinerators, cement production, and cooling towers contribute 4 to 6 times more chromium to the aquatic environment than do liquid wastes (Eisler 2000). Once it enters (neutral) water, chromium is most likely to precipitate out to the sediment particles. A major source of lead to the environment is through lead smelter emissions and previously from vehicle emissions. Once in the water most lead is precipitated to the sediment bed as carbonates and hydroxides (Eisler 2000). The major source of vanadium to the environment is re-smelting of steel scrap (WHO 1988). Lastly, contributions of zinc to the sediment result from road surface runoff, erosion of agricultural soils, and combustion of fossil fuels and solid wastes (Eisler 2000).

Regardless of the true nature of the source(s) for these COCs, the design of the assessment is based on characterizing the aggregate exposure (i.e., from all sources) as measured in the surface water at this site.

Relative source attribution can be accomplished as a separate step during Risk Characterization, if needed (USEPA 1998).

6.5.2 Magnitude and Distribution of COCs at Sunfish Lake

MAGNITUDE

The measured COC concentrations in surface water were used as the primary exposure metric for fish, aquatic invertebrates, and algae. The measured COC concentrations in the surficial sediment (0-1 ft deep) were used as the primary exposure metric for benthic organisms. Measured COC concentrations from both media were used as components in the wildlife exposure model. The range of COC concentrations detected is presented in Appendix H. Summary statistics are presented in Tables A38, A39, and A43.

SPATIOTEMPORAL DISTRIBUTION

The spatial and temporal distribution of the COCs is as important as the measured concentrations (USEPA 1998). Because the evaluated site is no longer in operation, the contamination is historical. These COCs are inorganics and are, therefore, environmentally persistent. Their presence through time is expected. The bioavailability or form of the COCs may change with environmental conditions (e.g., pH, temperature, nutrient loads, flow volume), but this aspect of exposure was not evaluated specifically. The spatial distribution of the sediment and surface water_COCs is expected to emanate from the banks along the north-central side of the lake, near Site H.

BIOLOGICAL DISTRIBUTION

Depending upon their biochemical characteristics, in addition to their presence in sediment these COCs may be found in biological tissues at elevated concentrations. Of primary concern are COCs that tend to concentrate or accumulate in tissues rather than those that are readily metabolized and/or excreted. Section 5.3.3.5 evaluated one aspect of this phenomenon (i.e., biomagnification) during COC selection.

The majority of information on the biological distribution of metals is concerned with the heavy metals (Cd, Cu, Fe, Pb, Mn, Hg and Zn) (Rainbow 1996). Depending on soil type, plants differ considerably in the amount of chromium they uptake. However, higher chromium content is observed in the roots as compared to the leaves and shoots (Eisler 2000). Chromium does not appear to biomagnify in aquatic or terrestrial food chains; however, each trophic level is expected to have some concentration of chromium in the tissue (Eisler 2000). Chromium III is an essential nutrient to organism and one of its functions is in RNA synthesis (WHO 1988). Small fish have been shown to concentrate chromium in the spleen, bronchi, and intestine 10 to 30 times more than that in the heart, skin, muscle, and scales. In animals, soluble forms of chromium are cleared from the body rather rapidly (in the feces), but colloid or protein bound forms (chromite, chromic chloride) accumulate in the bone marrow, liver, and spleen (WHO 1988).

The great variation of the lead contents of plants is influenced by several environmental factors, such as the presence of geochemical anomalies, pollution, seasonal variation, and genotype ability to accumulate lead (Kabata-Pendias and Pendias 1984). Lead is believed to be the most highly accumulated metal (of all bioavailable metals) in root tissue and only about 3% of lead accumulated in the roots gets translocated to the shoots (Kabata-Pendias and Pendias 1984). Among aquatic biota, lead concentrations are usually the highest in algae and benthic organism and lowest in upper trophic level predators. However, fish do show elevated blood lead levels (Eisler 2000). Thus it is assumed that there is no significant biomagnification in aquatic food chains (Eisler 2000). In mammals, concentrations of lead are usually highest in bone, hematopoietic tissues and hair (Eisler 2000).

Vanadium occurs in small amounts in all plants, with the majority of vanadium being found in the roots. The root concentrations are nearly the same as the level in the soil with the exception of agaric mushrooms, which are accumulators of vanadium (WHO 1988). As for concentrations in aquatic biota, little information was found during this literature search. However, one source did state that levels of vanadium in fish muscle have been found to be high (WHO 1988). In mammals, vanadium is shown to accumulate in the liver, kidney, and lung, but there is a low level of absorption in the gastrointestinal tract, and ingested vanadium is mainly eliminated unabsorbed, with the feces (WHO 1988).

In plants, zinc is thought to be relatively mobile and is likely to be concentrated in roots and mature leaves (Kabata-Pendias and Pendias 1984). In aquatic organisms, concentrations of zinc in the tissues are usually in excess of that required for normal metabolism, with zinc concentrations in fish highest in the eggs, viscera, a liver, and lowest in the muscle (Eisler 2000). Zinc concentrations in mammals can be highest in the pancreas, liver, and kidney, but specific organ accumulation can vary depending on species (Eisler 2000). Bioaccumulation of zinc in the aquatic food chain has not been directly shown.

The ability of an organism to accumulate these metals does not directly show that toxic effects are occurring. Different forms of the metals (i.e., inorganic vs. organic) may change the uptake ability or body storage of each metal in each biological entity. Careful consideration was given when considering these COCs.

6.5.3 Refined Conceptual Site Model

The initial conceptual model for Sunfish Lake consisted of five risk hypotheses (restated below) and was presented in the Work Plan (USACHPPM 1998) and represented in this Problem Formulation (Section 3). The refined conceptual model further explains the COCs and contaminant pathways. Figure B3-4 illustrates the hypothesized links between the contaminant source and potential adverse ecological effects.

1. Based upon the aquatic toxicity of surface water COCs (cadmium, mercury, and zinc); if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

2. Based upon the toxicity of sediment COCs (chromium, copper, lead, vanadium, and zinc); if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment. (An adequate evaluation of potential problems for the amphibian community of the lake was a major data gap in the Tier I assessment).

4. The survival, growth, and reproduction of wading birds may be at risk if they are excessively exposed to COCs in surface water, sediment, and residues in food from the lake.

5. The health of aquatic mammals, such as muskrat, may be impaired if they are excessively exposed to sediments and vegetation containing elevated levels of aluminum and vanadium. In addition, aquatic mammals may also be exposed to chromium, lead, and zinc.

6.5.4 Exposure Profile for Fish, Aquatic Invertebrates, and Algae

Resident fish, aquatic invertebrates, and algal species co-occur with the COCs due to the presence of the COCs in the surface waters. The spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Sunfish Lake. These receptors will also contact the COCs while inhabiting the aquatic environment—in essence; they inhabit in the waters that contain the COCs such that they are continuously exposed. The possibility that fish may swim in and out of areas of contamination exists; however, this level of detail was not included in the analysis.

The primary mechanism for the selected COCs to induce effects in these organisms is through contact. Contact was quantified as environmental concentrations in surface water, with the assumptions that the COCs are well mixed and/or the organisms move randomly through the water.

The following attributes characterize the exposures for fish, aquatic invertebrates, algae, and benthic organisms.

- <u>Exposure duration</u>
  Entire life-cycle for most; non-emerged (i.e., juvenile) forms of some invertebrates

- <u>Exposure frequency</u>
  Continuous; however, concentrations fluctuate by season (see Exhibits 27 through 29)

- <u>Population portion "at-risk"</u>
  The portion of the local populations that are actually exposed is not known

- <u>Contact intensity</u>
  COC concentrations in surface water are provided in Exhibit 6-5 through Exhibit 6-7.

Exhibit 6-5. Review of Cadmium Summary Statistics in Sunfish Lake Surface Water

Statistics	Cadmium Surface Water Concentrations (µg/L) by quarter				Total
	1st■	2nd	3rd	4th	
Distribution	Normal				
25th percentile	NA	0.017	0.017	0.009	0.015
50th percentile	NA	0.018	0.017	0.009	0.015
75th percentile	NA	0.019	0.017	0.011	0.017
Maximum	NA	0.020	0.017	0.014	0.020
Central Tendency (Median)	NA	0.0177	0.0173	0.0105	0.015
Upper Confidence Limit	NA	0.0201	0.0178	0.0134	0.017

■Cadmium was not detected in the first quarter

† Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

‡ Upper Confidence Limit : The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

Exhibit 6-6. Review of Mercury Summary Statistics in Sunfish Lake Surface Water

Statistics	Mercury Surface Water Concentrations (µg/L) by quarter				Total
	1st	2nd	3rd	4th	
Distribution	Unknown				
25th percentile	0.0009	0.0009	0.0011	0.0016	0.0011
50th percentile	0.0011	0.0012	0.0012	0.0016	0.0012
75th percentile	0.0039	0.0012	0.0014	0.0018	0.0016
Maximum	0.0067	0.0012	0.00158	0.0020	0.0067
Central Tendency (Median)[†]	0.0011	0.0012	0.00118	0.0016	0.0012
Upper Confidence Limit[‡]	0.00614	0.0012	0.00154	0.0020	0.0020

[†] Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

[‡] Upper Confidence Limit : The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

Exhibit 6-7 Review of Zinc Summary Statistics in Sunfish Lake Surface Water

Statistics	Zinc Surface Water Concentrations (µg/L) by quarter				Total
	1st	2nd	3rd	4th	
Distribution	Normal				
25th percentile	0.035	1.887	1.718	0.612	0.612
50th percentile	0.035	2.337	1.718	0.612	1.578
75th percentile	1.418	2.337	1.818	0.652	2.023
Maximum	2.8	2.337	1.918	0.692	2.8
Central Tendency (Median)[†]	0.957	2.037	1.785	0.639	1.354
Upper confidence limit[‡]	2.493	2.537	1.896	0.683	1.792

[†] Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

[‡] Upper Confidence Limit : The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

6.5.5 Exposure Profile for Benthic Organisms at Sunfish Lake

Resident benthic organisms co-occur with the COCs due to the presence of the COCs in the sediments. The spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Sunfish Lake, which was described in the previous subsection. These receptors will also contact the COCs while inhabiting the benthic environment—in essence; they inhabit in the sediments that contain the COCs such that they are continuously exposed.

The primary mechanism for the selected COCs to induce effects in these organisms is through contact. Contact was quantified as environmental concentrations in sediment.

The following attributes characterize the exposures for benthic organisms.

- Exposure duration
  Entire life-cycle for most; non-emerged (i.e., juvenile) forms of some invertebrates

- Exposure frequency
  Continuous; however, concentration fluctuations by season are expected (but have not been quantified)

- Population portion "at-risk"
  The portion of the local populations that are actually exposed is not known

- Contact intensity
  COC concentrations in sediment are provided in Exhibit 6-8.

Exhibit 6-8. Review of Important Summary Statistics of Sunfish Lake COCs

Statistics	Sediment Concentrations (mg/kg)				
	Aluminum	Chromium	Lead	Vanadium	Zinc
Distribution	Normal	Normal	Normal	Normal	Normal
25th percentile	11375	43	97.08	44.28	237.5
50th percentile	13050	140.35	119	47.85	307.5
75th percentile	13525	198.25	136.5	51.6	340
Maximum	18000	264	202	61.5	501
Central Tendency	12612	137	124	46.7	302
95th Upper Confidence Limit on the Mean	14542	203	154.3	53.7	374.6

6.5.6 Exposure Profile for Amphibians at Sunfish Lake

Resident amphibians will co-occur with the COCs due to the presence of the COCs in the surface waters and sediments. Again, the spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within the lake, which was described in the previous section. These receptors will also contact the COCs while inhabiting the aquatic environment—in essence; they exist, during a large portion of their life cycle, in the waters and sediments that contain the COCs such that they are continuously exposed for those periods. Juvenile amphibians will be relatively more exposed than adults due to their movement habits.

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.

LOCAL POPULATIONS OF AMPHIBIANS AND THEIR EXPOSURE CHARACTERISTICS

As part of the amphibian study conducted at TCAAP (see Appendix M) Sunfish Lake as a whole was examined to determine amphibian species richness and relative abundance. Visual surveys were difficult due to habitat constraints. However, five species of frogs were observed by aural surveys, visual encounter surveys (VES), and/or trapping: *Bufo americanus, Rana clamitans, Rana pipiens, Pseudacris triseriata and Rana ssp.* Overall, relatively few amphibians were found at the lake. There may be several reasons for low relative abundance due to limitations and uncertainty in the study, as well as environmental problems (see Table 6.1 in Appendix M).

- <u>Exposure duration</u>
  The entire Embryo and juvenile portions of the life-cycle
- <u>Exposure frequency</u>
  Continuous during the juvenile life stages; intermittent at most for adult amphibians.
- <u>Population portion "at-risk"</u>
  The portion of the local populations actually exposed is not known
- <u>Contact intensity</u>
  COC concentrations in sediment are provided in Exhibit 6-8.

6.5.7 Exposure Profile for Wading Birds at Sunfish Lake

The exposure profile for wading birds (Taxonomic Family *Ardeidae*) was characterized using the Great Blue Heron (*Ardea herodias*) as a representative species for that group. Biological attributes of this heron and the group it represents are provided in Appendix N.

Individuals of local populations of these species may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. The spatial extent of co-occurrence can be approximated by the spatial extent of the COC "footprint" within Sunfish Lake. These wildlife will contact the COCs while inhabiting the aquatic environment. Wading birds will stand and feed in the waters. Also, the presence of the COCs in the food resources of these receptors may result in COC contact via the food chain. These species will not be continuously exposed because they are expected to move in and out of areas of contamination.

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of sediment and food that contain residues of the COCs. Contact was quantified as a potential dose, which is the amount of the COC ingested in food and sediment per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured sediment concentrations, estimated food residues, and models describing the ingestion rate.

LOCAL POPULATIONS OF WADING BIRDS AND THEIR EXPOSURE CHARACTERISTICS

The wading birds evaluated at TCAAP, with the exception of bitterns, prefer to utilize more open marsh areas when feeding, where lower plant densities and dropping water levels during the dry season make them particularly attractive foraging areas (USFWS 2000). These birds do not prefer deep water but rather areas where they can walk through because their method of obtaining food is by stabbing it with their beaks. It is not known if nesting occurs at Sunfish Lake by any of the species discussed in Appendix N. Because Great Blue Herons nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). There is very little tree cover at, or surrounding Sunfish Lake. Therefore, colonies are not likely due to insufficient habitat. Bitterns are also not expected to nest at Sunfish Lake and due to their secretive habits they may not be found feeding or even visiting at the lake (See Figure H1). However, herons may forage at Sunfish Lake, and it can be expected that 1 to 3 herons may utilize Sunfish Lake from March through November.

The following attributes characterize the exposures for wading birds.

- Exposure duration
  The great blue heron is known to nest in the vicinity of TCAAP and is expected to be exposed throughout its life cycle (USGS 1995).

- Exposure frequency
  For those birds that will be exposed, exposure is expected to be intermittent. Herons have been reported in the 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 occasionally.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known. Because great blues nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). It is difficult to determine the amount of foraging habitat available at Sunfish Lake without field data. However, based on the available information it is possible that 1 to 3 herons may dominate at Sunfish Lake during any given season.

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N). Only the great blue heron was modeled. This species was chosen as the surrogate for wading birds during the development of the Work Plan (USACHPPM 1999), mainly based on its presence at the site and availability of data for modeling purposes. Modeling only this species of wading bird at Sunfish Lake presents some uncertainty. This uncertainty is a result of differences in site usage between the herons and the bitterns, as discussed above. There are also slight differences in dietary preferences among the wading birds, such that bitterns may eat more invertebrates and frogs than herons. Despite the existence of these differences and the potential for an underestimation of potential for risk by not modeling the maximally exposed species, modeling the heron is an acceptable approach. Mainly due to the lack of necessary input values for the other species, and the overestimation of potential for risk resulting from use of other conservative assumptions.

6.5.8 Exposure Profile for Aquatic Mammals at Sunfish Lake

The exposure profile for aquatic mammals was characterized with models for the Muskrat (*Ondatra zibethicus*) and the Mink (*Mustela vison*). Exposure of muskrats to all COCs was evaluated, while exposure of mink was only evaluated for mercury (see Appendix N). Biological attributes of these animals and the groups they represent are provided in Appendix N.

Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. The spatial extent of co-occurrence can be approximated by the spatial extent of the COC "footprint" within Marsden Lake (South). These wildlife will contact the COCs while inhabiting the aquatic environment. Muskrat will swim and feed on vegetation (primarily cattails) in the waters and use the sediments for building their dens. Mink will swim and feed on muskrat and other animals in the waters. The presence of the COCs in the food resources of these receptors may result in COC contact via the food chain. While the home range of an individual muskrat is small, indicating that some individuals of the Muskrat population may be continuously exposed, Mink will not be continuously exposed because they are expected to move in and out of areas of contamination.

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of sediment and food that contain residues of the COCs. Contact was quantified as a potential dose, which is the amount of the COC ingested in food and sediment per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured sediment concentrations, estimated food residues, and models describing the ingestion rate.

LOCAL POPULATIONS OF MUSKRAT AND THEIR EXPOSURE CHARACTERISTICS

During past field sampling activities, several to many muskrat dens were observed in the eastern portion of the lake. There does not appear to be suitable habitat at other locations in Sunfish Lake, however, no formal population/den count was conducted at the site. Cattail communities are known to support high densities of muskrats, but in lake habitats shoreline length is also an important factor in muskrat density (USEPA 1993). One method of estimating the population of muskrats is to count the number of dens, then multiply by a factor between 2.8 and 5.0 (USEPA 1993), but this is a questionable method. Sunfish Lake is much smaller than Round Lake and is expected to have a much smaller muskrat population. Based on the availability of preferred muskrat habitat at Sunfish Lake (i.e., cattails in the eastern portions), it can be expected that Sunfish Lake is capable of supporting 20 adult muskrats. This is a very conservative estimate; and it assumes that there is ample food and that the whole of the northeastern portion of the lake is inhabitable.

The following attributes characterize the exposures for Muskrat.

- <u>Exposure duration</u>
  The Muskrat that are exposed are likely to be exposed for the majority of their life cycle because their home range is small.

- <u>Exposure frequency</u>
  For those that are exposed, they will be continuously exposed.

- <u>Population portion "at-risk"</u>
  The portion of the local populations actually exposed is not known.

- <u>Contact intensity</u>
  The exposure modeling results are presented in Section N4 (in Appendix N).

LOCAL POPULATIONS OF MINK AND THEIR EXPOSURE CHARACTERISTICS

Signs of mink activity at the lake have been observed. (e.g., tracks in the winter, etc.). Sunfish Lake is approximately 5.7 ha size and assuming that there is sufficient habitat and food, then it can be expected that Sunfish Lake may support 1 female mink. The home range of female minks has been estimated to be 7.8 ha in heavily vegetated areas and 20.4 ha in sparsely vegetated areas (USEPA 1993). One to two male mink may spend time between Sunfish Lake and Marsden Lake, most likely making their dens at Marsden Lake.

The following attributes characterize the exposures for Mink.

- Exposure duration
  Of the Mink that may be exposed, they are expected to either visit or locate their den at the site, indicating that the exposure duration could vary to a large degree. It can be assumed that for a large portion of their life cycle they will be exposed to COCs while at Sunfish Lake.

- Exposure frequency
  Since the size of Sunfish Lake is small relative the size of known Mink home ranges and the large TCAAP habitat area available, it is expected that Mink would most likely be exposed on an intermittent basis during any given year.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known. Based on literature regarding Mink habits and population densities, it is projected that one adult male, and one adult female and her young could be exposed at the site each year.

- Contact intensity
  The exposure modeling results are presented in Section N4 (in Appendix N).

6.6 ROUND LAKE ANALYSIS: CURRENT SCENARIO

Surface water was analyzed for previously selected Tier II COCs (barium, cadmium, copper, mercury, silver, and zinc). Based on the collected data (Section 4) and the implementation of the revised COC selection criteria (Section 5), barium, cadmium, and zinc were retained for further consideration as surface water COCs. The Tier I assessment recommended that aluminum, cadmium, chromium, copper, lead, silver, vanadium, and zinc be examined as sediment COCs during this Tier II assessment. These sediment COCs were reassessed, and aluminum was eliminated from further consideration (see Section 5). Additionally, during the Tier II work plan development, a hazard evaluation on the existing PCB information in Round Lake was completed (see Appendix P).

Round Lake COCs: Current Scenario

Surface Water
Barium
Cadmium
Zinc

Sediment
Cadmium
Chromium
Copper
Lead
Silver
Vanadium
Zinc
PCBs

Exposures of the following ecological entities to these COCs were characterized: fish, aquatic invertebrates, benthic organisms, and algal species, amphibians, wading birds, piscivorous birds, and aquatic mammals.

6.6.1 Review of COC Source(s)

The specific source of the COCs in Round Lake includes activities at Site I, where artillery shell forgings were produced in the past. All contamination of Round Lake due to TCAAP activities is historical; that is, a continuing TCAAP source does not exist.

Barium, cadmium, and zinc in surface water were selected as surface water COCs because they were present in concentrations that were statistically greater than the comparison site and also had a maximum detected concentration that was greater than a toxicological benchmark. Cadmium, chromium, copper, lead, silver, vanadium, and zinc were selected as sediment COCs based on the COC selection criteria (Section 5). Additional sediment chemistry analysis was not conducted for the Tier II assessment. The PCB hazard evaluation (see Appendix P) concluded that PCBs related to former TCAAP activities are found at concentrations above toxicity benchmarks in localized areas in the northern littoral sediments of the lake.

There are also other sources that contribute to aluminum, barium, cadmium, chromium, copper, lead, silver, vanadium, and zinc in Round Lake. For a discussion of additional sources of chromium, copper, vanadium, and zinc please see Section 6.5.1.

Regardless of the true nature of the source(s) for these COCs, the design of the assessment is based on characterizing the aggregate exposure (i.e., from all sources) as measured in the surface water at this site. Relative source attribution can be accomplished as a separate step during Risk Characterization, if needed (USEPA 1998).

6.6.2 Magnitude and Distribution of COCs at Round Lake: Current Scenario

MAGNITUDE

The measured COC concentrations in surface water were used as the exposure metric for barium, cadmium, and zinc while the measured COC concentrations in the surficial sediment (0-1 ft deep) were used as the primary exposure metric for benthic organisms and as a component in the wildlife exposure model. The range of COC concentrations detected is presented in Appendix I. Summary statistics for surface water are provided for selected chemicals in Tables A45, A46, and A47. Summary statistics for sediment COCs are presented in Table A50.

SPATIOTEMPORAL DISTRIBUTION

The spatial and temporal distribution of the COCs is as important as the measured concentrations (USEPA 1998). Because a continuing TCAAP source does not exist, the contamination is historical. These COCs are inorganics and are, therefore, environmentally persistent. There presence through time is expected. The bioavailability or form of the COCs may change with environmental conditions (e.g., pH, temperature, nutrient loads, flow volume), but this aspect of exposure was not evaluated specifically.

The spatial distribution of the COCs emanates from the northeastern banks, where TCAAP drain systems historically released water and other materials. The concentrations decrease with increasing distance away from these areas and the sample data support this. Lead is an exception, having two high concentration locations not in the northern portion of the lake (see Figure I2). For PCBs, it is important to note that while previous sampling showed detections in the northern littoral zone, the most recent and extensive sampling effort (1992) found no valid detections of PCBs in the sediments of the lake at the locations of the historical detections (see Appendix P).

BIOLOGICAL DISTRIBUTION

Depending upon their biochemical characteristics, in addition to their presence in surface water and sediment these COCs may be found in biological tissues at elevated concentrations. Of primary concern are COCs that tend to concentrate or accumulate in tissues rather than those that are readily metabolized and/or excreted. Section 5.3.3.6 evaluated one aspect of this phenomenon (i.e., biomagnification) during COC selection.

Information on PCB uptake by organisms has shown that PCBs can be translocated from soil to various parts of plants (Wang et al., 1997). In addition, PCBs are Persistent, Bioaccumulative, and Toxic (PBT) substances and are found in fish and animal tissues.

The majority of information on the biological distribution of metals is concerned with the heavy metals (Cd, Cu, Fe, Pb, Mn, Hg and Zn) (Rainbow 1996).

Information on barium uptake by plants is also limited. Some plants are known to bioconcentrate barium from the soil but it is unclear as to what part of the terrestrial plant stores this metal (ATSDR 1992). However, many reports indicate that aquatic macrophyte roots, such as those of cattails, accumulate larger concentrations of barium than other plant organs (Wang et al 1997). According to the Toxicity Profile for Barium (ATSDR 1992), barium is absorbed by animals and fish following ingestion, but not enough information is available to determine if barium bioaccumulates or biomagnifies in the freshwater aquatic food web.

The great variation of the lead contents of plants is influenced by several environmental factors, such as the presence of geochemical anomalies, pollution, seasonal variation, and genotype ability to accumulate lead (Kabata-Pendias and Pendias 1984). Lead is believed to be the most highly accumulated metal (of all bioavailable metals) in root tissue and only about 3% of lead accumulated in the roots gets translocated to the shoots (Kabata-Pendias and Pendias 1984). Among aquatic biota, lead concentrations are usually the highest in algae and benthic organism and lowest in upper trophic level predators. However, fish do show elevated blood lead levels (Eisler 2000). Thus it is assumed that there is no significant biomagnification in aquatic food chains (Eisler 2000). In mammals, concentrations of lead are usually highest in bone, hematopoietic tissues and hair (Eisler 2000).

Silver is a naturally occurring element, with characteristics similar to copper but its concentration in rock is about 1000 times lower (Kabata-Pendias and Pendias 1984). Dissolved silver speciation and bioavailability are important in determining silver uptake and retention by aquatic plants. Freshwater plants were shown to have a bioconcentration factor of 200, and phytotoxic effects were seen in duckweed at concentrations of 270 µg Ag^+ /L (Eisler 2000). The amount of silver absorbed by terrestrial plants is related to the amount of metal in the soil, with most silver being retained in the roots (Kabata-Pendias and Pendias 1984). In rainbow trout, silver concentrations in gills and liver increase with increasing exposure to silver (Eisler 2000). In mammals, silver is retained by all body tissue, with the highest concentrations usually in the liver and spleen (Eisler 2000).

Cadmium is a naturally occurring element, and its presence has been detected in more than 1000 species of aquatic and terrestrial flora and fauna. Many reports indicate that aquatic macrophytes accumulate large concentrations of cadmium in the roots as compared to the shoots (Wang et al 1997). There is considerable variation in the ability of teleost tissues to accumulate cadmium, but the gills seem to be the primary site of accumulation (Eisler 2000). As for vertebrates, cadmium tends to concentrate in the viscera, especially the liver and kidneys (Eisler 2000).

Depending on soil type, plants differ considerably in the amount of chromium they uptake. However, higher chromium content is observed in the roots as compared to the leaves and shoots (Eisler 2000). Chromium does not appear to biomagnify in aquatic or terrestrial food chains; however, each trophic level is expected to have some concentration of chromium in the tissue (Eisler 2000). Chromium III is an essential nutrient to organism and one of its functions is in RNA synthesis (WHO 1988). Small fish have been shown to concentrate chromium in the spleen, bronchi, and intestine 10 to 30 times more than that in the heart, skin, muscle, and scales. In animals, soluble forms of chromium are cleared from the body rather rapidly (in the feces), but colloid or protein bound forms (chromite, chromic chloride) accumulate in the bone marrow, liver, and spleen (WHO 1988).

Vanadium occurs in small amounts in all plants, with the majority of vanadium being found in the roots. The root concentrations are nearly the same as the level in the soil with the exception of agaric mushrooms, which are accumulators of vanadium (WHO 1988). As for concentrations in aquatic biota, little information was found during this literature search. However, one source did state that levels of vanadium in fish muscle have been found to be high (WHO 1988). In mammals, vanadium is shown to accumulate in the liver, kidney, and lung, but there is a low level of absorption in the gastrointestinal tract, and ingested vanadium is mainly eliminated unabsorbed, with the feces (WHO 1988).

In plants, zinc is thought to be relatively mobile and is likely to be concentrated in roots and mature leaves (Kabata-Pendias and Pendias 1984). In aquatic organisms, concentrations of zinc in the tissues are usually in excess of that required for normal metabolism, with zinc concentrations in fish highest in the eggs, viscera, a liver, and lowest in the muscle (Eisler 2000). Zinc concentrations in mammals can be highest in the pancreas, liver, and kidney, but specific organ accumulation can vary depending on species (Eisler 2000). Bioaccumulation of zinc in the aquatic food chain has not been directly shown.

The ability of an organism to accumulate these metals does not directly show that toxic effects are occurring. Different forms of the metals (i.e., inorganic vs. organic) may change the uptake ability or body storage of each metal in each biological entity. Careful consideration was given when considering these COCs.

6.6.3 Refined Conceptual Site Model

This refined CSM further explains contaminant pathways and specific COCs. Figure B3-5 illustrates the links between the hypothesized contaminant source and potential adverse ecological effects under the current (or "no action") scenario. Originally, two conceptual models were developed in the Work Plan: one for most COCs (see Section 3.5) and another specific to PCBs (see Appendix P). The following risk hypothesis provide an integrated set of hypotheses.

1. (a) Based upon the aquatic toxicity of surface water COCs (barium, cadmium, and zinc); if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

   (b) Based upon exposure to PCBs in food and sediments, if PCBs are present above acceptable levels, then developmental and reproductive problems may occur in fish (e.g. black bullheads).

2. If concentrations of sediment COCs (cadmium, chromium, copper, lead, silver, vanadium, zinc, and PCBs) are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may be at risk due to contaminants present in the lake.

4. The health of aquatic mammals, such as muskrat, may be impaired if they are excessively exposed to sediments and vegetation containing elevated levels of cadmium, chromium, copper, lead, silver, vanadium and zinc.

5. If PCBs are present above acceptable levels in sediments and fish, then developmental and reproductive problems may occur in exposed mammals and birds.

The initial Problem Formulation (Section 3.5.4) contained a hypothesis related to wading birds and mink exposures to mercury, because the Tier I risk assessment failed to properly screen for mercury in surface water (i.e., the detection limits were too high). Based on the COC selection process (Section 5), mercury was ultimately not selected as a COC for further evaluation.

6.6.4 Exposure Profile for Fish, Aquatic Invertebrates, Algae, and Benthic Organisms at Round Lake Under the Current Scenario

The primary mechanism for the selected COCs to induce effects in these organisms is through contact. Contact was quantified as environmental concentrations in surface water, with the assumptions that the COCs are well mixed and/or the organisms move randomly through the water or sediment.

Under the current scenario (i.e., no water draw downs), resident fish, aquatic invertebrates, algal species, and benthic organisms co-occur with the COCs due to the presence of the COCs in the surface waters and sediments. The spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Round Lake, which was described in the previous sections. These receptors will also contact the COCs while inhabiting the aquatic environment—in essence; they inhabit in the waters and sediment that contain the COCs such that they are continuously exposed. The possibility that fish may swim in and out of areas of contamination exists; however, this level of detail was not included in the analysis.

The following attributes characterize the exposures for fish, aquatic invertebrates, algae, and benthic organisms.

- <u>Exposure duration</u>
  Entire life-cycle for most; non-emerged (i.e., juvenile) forms of some invertebrates

- <u>Exposure frequency</u>
  Continuous; however, concentration fluctuations by season are expected (but have not been quantified)

- <u>Population portion "at-risk"</u>
  The portion of the local populations that are actually exposed is not known

- <u>Contact intensity</u>
  COC concentrations in surface water are provided in Exhibit 6-9 through Exhibit 6-11. COC concentrations in sediment are provided in Exhibit 6-12. PCB-specific concentrations are provided in Exhibit 6-13.

Exhibit 6-9. Review of Surface Water Barium Statistics of Round Lake

Statistics	Barium Surface Water Concentrations (μg/L) by quarter				Total
	1st	2nd	3rd	4th	
Distribution	Unknown[*]				
25th percentile	62	88	85.1	70	67.25
50th percentile	62	88	86.4	70	76.45
75th percentile	62	89	86.6	70	86.73
Maximum	62	91	87	71	91
Central Tendency (Median)	62.0	88.0	86.0	70.	76.45
Upper Confidence Limit	62	90.6	87.0	70.8	89.1

[†] Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

[‡] Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

[*] Due to small sample size distribution could not be determined per quarter and was determined across all quarters

Exhibit 6-10. Review of Surface Water Cadmium Statistics of Round Lake

Statistics	Cadmium Surface Water Concentrations (μg/L) by quarter				Total
	1st[■]	2nd	3rd	4th	
Distribution	Lognormal				
25th percentile	0.015	0.020	0.020	0.011	0.015
50th percentile	0.015	0.021	0.021	0.012	0.015
75th percentile	0.015	0.026	0.022	0.012	0.021
Maximum	NA	0.054	0.026	0.013	0.054
Central Tendency (Median)[†]	NA	0.0275	0.0214	0.0114	0.0186
Upper Confidence Limit[‡]	NA	0.0561	0.025	0.013	0.022

[■] Cadmium was not detected in the first quarter, value shown is the detection limit

[†] Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

[‡] Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

Exhibit 6-11 Review of Surface Water Zinc Statistics of Round Lake

Statistics	Zinc Surface Water Concentrations (μg/L) by quarter				Total
	1st[▪]	2nd	3rd	4th	
Distribution	Normal				
25th percentile	0.035	2.737	1.718	0.612	0.363
50th percentile	0.035	2.737	1.718	0.718	1.205
75th percentile	0.035	3.237	1.818	0.812	1.973
Maximum	NA	4.040	1.820	0.992	4.04
Central Tendency (Median)[†]	NA	3.040	1.70	0.720	1.37
Upper Confidence Limit[‡]	0.035	3.427	1.80	0.842	1.933

[▪] Zinc was not detected in the first quarter, value shown is the detection limit

[†] Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

[‡] Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

Exhibit 6-12. Review of Important Summary Statistics of Round Lake Sediment COCs (from Tier I data)

Statistics	Sediment Concentrations (μg/g)						
	Cadmium	Chromium	Copper	Lead	Silver	Vanadium	Zinc
Distribution	Unknown	Lognormal	Normal	Unknown	Unknown	Normal	Lognormal
25th percentile	0.427	25.65	153.25	10.0	2.105	31.75	159.5
50th percentile	0.427	60.85	374.5	10.0	30.475	37.7	370
75th percentile	2.154	71.58	586.25	10.0	65.575	44.375	443.25
Maximum	11.2	172.0	1250	299.0	86.8	63.0	860.0
Central Tendency[†]	0.4	77	240.17	10.0	30.5	34.8	422.1
Upper Confidence Limit[‡]	8.1	248	482.42	263.3	82.6	42.68	1,064.3

[†] Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

[‡] Upper Confidence Limit: The 95th upper confidence limit on the mean for normal data, the 95th upper confidence limit on the geometric mean for lognormal data, and 95th percentiles if the distribution is unknown.

Exhibit 6-13. Concentrations of PCBs in Surface Water, Surface Sediment, and Fish at Round Lake[†]

Exposure point medium [‡]	Total PCB Concentration		n	Remarks
	[maximum]	[mean]		
Surface water (1992)	< 0.9 µg/L	< 0.9 µg/L	5	No detections
Surface sediment (1992)	< 0.04 mg/kg	< 0.04 mg/kg	17	15 nondetects and 2 samples rejected
Surface sediment (1987 & 1990)	13.0 mg/kg	2.3 mg/kg	17	most samples were < 0.05 mg/kg
Black bullhead (1981 & 1988)	0.27 mg/kg wwt	0.20 mg/kg wwt	5	wet weight concentrations; samples are composites of 5 fish each

[‡]: Other points exist (i.e., macroinvertebrates, amphibians, vegetation, muskrat, etc.) however site-specific residue concentrations are unavailable.

[†] This table was reproduced from Appendix P.

6.6.5 Exposure Profile for Wading Birds and Belted Kingfishers at Round Lake Under the Current Scenario

The exposure profile for wading birds (Taxonomic Family *Ardeidae*) was characterized using the Great Blue Heron (*Ardea herodias*) as a representative species for that group. Biological attributes of these animals and the groups they represent are provided in Appendix N. The exposure profile and biological attributes of the belted kingfisher are presented below. Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. The spatial extent of co-occurrence can be approximated by the spatial extent of the COC "footprint" within Round Lake. These wildlife will contact the COCs while inhabiting the aquatic environment. Wading birds will stand and feed in the waters. Also, the presence of the COCs in the food resources of these receptors may result in COC contact via the food chain. These species will not be continuously exposed because they are expected to move in and out of areas of contamination.

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of surface water and food that contain residues of the COCs. Contact was evaluated as a concentration in the food item consumed by these receptors. For further details on how exposure was evaluated see Appendix P.

LOCAL POPULATIONS OF WADING BIRDS AND THEIR EXPOSURE CHARACTERISTICS

It is not known if nesting occurs at Round Lake by any of these species. Colonies are not likely due to insufficient habitat. Because Great Blue Herons nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). Round Lake is part of the larger Minnesota Valley National Wildlife Refuge where Great Blue Herons, American Bitterns, and Great Egrets are known to nest (USGS 1995). These birds may not nest at Round Lake but they can be expected to frequent the area. Herons have been known to forage up to 24.4 km from their nesting colonies, with population densities of 2.3 birds/km along streams (USEPA 1993). With this in mind, it can be expected that during March through November, Round Lake could possibly support 4-6 adult herons.

The following attributes characterize the exposures for wading birds.

- <u>Exposure duration</u>
  The great blue heron is known to nest in the vicinity of TCAAP and would be exposed throughout its life cycle (USGS 1995).

- <u>Exposure frequency</u>
  For those birds that will be exposed, exposure is expected to be intermittent. Herons have been reported in the 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.

- <u>Population portion "at-risk"</u>
  The portion of the local populations actually exposed is not known. Because great blues nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). Based on information in the *Wildlife Exposure Factors Handbook* (USEPA 1993), Round Lake could possibly support 4-6 adult herons. Adult herons are known to feed solitarily unless large densities of prey are found, and some are known to defend feeding territories, thus limiting the number of herons frequenting the area (USEPA 1993).

- <u>Contact intensity</u>
  Exposures to PCBs were developed as a part of the Work Plan and are presented in Appendix P.

LOCAL POPULATIONS OF BELTED KINGFISHERS AND THEIR EXPOSURE CHARACTERISTICS

This species has been observed on TCAAP and/or Round Lake and eats primarily fish. Kingfishers are stocky, short-legged birds with large heads and bills. They exist on a diet mostly of fish, which they catch by diving headfirst from a perch or the air. They nest in burrows in earthen banks that they dig using their bills and feet. The following test on their food habits is an excerpt from the USEPA *Wildlife Exposure Factors Handbook*:

> Belted kingfishers generally feed on fish that swim near the surface or shallow water. [Some] believe that these 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; [some] found fish caught in Ohio streams to range from 4 to 14 cm in length. Several studies indicate that belted kingfishers usually catch the prey that are most available. Diet therefore varies among different water bodies and with season. Although kingfishers feed primarily on fish, they also sometimes consume large numbers of crayfish, and in shortages of their preferred foods, have been known to consume crabs, mussels, lizards, frogs, toads, small snakes, turtles, insects, salamanders, newts, young birds, mice, and berries.

During the breeding season pairs establish territories for nesting and fishing; otherwise they are solitary. Breeding densities of 2-6 pairs per 10 km of river shoreline have been recorded. These birds are sensitive to disturbance and usually do not nest in areas near human activity. Given this information, it is unlikely that more than 3 or 4 pairs inhabit the Round Lake system, if any inhabit the lake at all. Given the general refuge nearby (i.e., TCAAP) they may feed, at least in part, in Round Lake.

The following attributes characterize the exposures for kingfishers.

- Exposure duration
  The duration is dependent on habitat usage. Because these birds prefer to perchi 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.

- Exposure frequency
  Information pertaining to the time of year and nesting activity of belted kingfishers observed at Round Lake is unavailable, so it is not clear if they are nesting at the site. If the birds are not nesting at the site, then the exposure frequency is expected to be occasional. However, if the birds are nesting there, then exposure frequency is expected to be continual.

- Population portion "at-risk"
  The portion of the local population actually exposed is not known. During the breeding season pairs establish territories for nesting and fishing; otherwise they are solitary. Breeding densities of 2-6 pairs per 10 km of river shoreline have been recorded. Given this information, it is unlikely that more than 3 or 4 pairs inhabit the Round Lake system, if any inhabit the lake at all

- Contact intensity
  Exposures to PCBs were developed as a part of the Work Plan and are presented in Appendix P.

6.6.6 Exposure Profile for Aquatic Mammals at Round Lake Under the Current Scenario

The exposure profile for aquatic mammals was characterized with models for the Muskrat (*Ondatra zibethicus*) and the Mink (*Mustela vison*). Biological attributes of these animals and the groups they represent are provided in Appendix N.

Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. The spatial extent of co-occurrence can be approximated by the spatial extent of the COC "footprint" within Round Lake. These wildlife will contact the COCs while inhabiting the aquatic environment. Muskrat will swim and feed on vegetation (primarily cattails) in the waters and use the sediments for building their dens. Mink will swim and feed on muskrat and other animals in the waters. The presence of the COCs in the food resources of these receptors may result in COC contact via the food chain. While the home range of an individual muskrat is small, indicating that some individuals of the Muskrat population may be continuously exposed, Mink will not be continuously exposed because they are expected to move in and out of areas of contamination.

The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of surface water and food that contain residues of the COCs. Contact was quantified as a potential dose, which is the amount of the COC ingested in food and surface water per day with the use of an exposure algorithm (USEPA 1998). The algorithm incorporated measured surface water concentrations, estimated food residues, and models describing the ingestion rate.

LOCAL POPULATIONS OF MUSKRAT AND THEIR EXPOSURE CHARACTERISTICS

According to the *Wildlife Exposure Factors Handbook* (USEPA 1993), in pond and lake habitats, shoreline length is a more important factor than overall habitat area in determining muskrat density. Many investigators estimate muskrat density by counting the number of houses or push-ups and multiplying by a factor ranging from 2.8 to 5.0; however, this method is questionable.

In addition, a population of muskrats is very dependent on the water cycle of the lake they live in. In times of drought, populations dwindle, mainly as a result of lack of food and also from predation. When the marshy areas re-flood, there is an abundant supply of new plants. The abundance of food fuels the female's reproductive potential of 14 – 21 young per year. Food abundance also allows better than average survival of the young and adults. Under these conditions, a population of only 20 muskrats can grow to more than 100 in a single summer (Illinois DNR, 2001).

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. Considering the size of Round Lake (51 ha), the amount of suitable habitat, the expected 8 meters average distance between dens, the solitary lifestyle of mating pairs, and the home range (0.17 ha) of muskrats, it can be expected that Round Lake is capable of supporting approximately 450 adult muskrats. This is a very conservative estimate, assuming that there is ample food and that the whole of the northern, eastern and southeastern portions of the lake is inhabitable.

The following attributes characterize the exposures for Muskrat.

- Exposure duration
  The Muskrat that are exposed are likely to be exposed for the majority of their life cycle because their home range is small.

- Exposure frequency
  For those that are exposed, they will be continuously exposed.

- Population portion "at-risk"
  The portion of the local population that is actually exposed is not known.

- Contact intensity
  The exposure modeling results for all COCs other than PCBs are presented in Section N4 (in Appendix N). The evaluation of PCB exposures was restricted to mink, because mink are known to be highly sensitive to PCBs and are generally more exposed because PCBs accumulate in fish, a primary food item for mink. Exposures for PCBs are presented in Appendix P.

LOCAL POPULATIONS OF MINK AND THEIR EXPOSURE CHARACTERISTICS

Signs of mink activity at the lake have been observed. (e.g., tracks in the winter, etc.). Considering the size of the Round Lake habitat (51 total hectares, including areas of open water and an estimated 25 near-shore hectares), and assuming that habitat and food abundance are sufficient, then Round Lake may be expected to support around 2 to 3 adult females. On the other hand, the Round Lake system is a small fragment of habitat within a larger industrialized landscape 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 fragment may not be sufficient to support mink over the long term, though they have been observed there.

The following attributes characterize the exposures for Mink.

- Exposure duration
  Of the Mink that may be exposed, they are expected to either visit or locate their den at contaminated areas of the site, indicating that the exposure duration could vary to a large degree.

- Exposure frequency
  Since the size of the contaminated areas of Round lake are small relative the size of known Mink home ranges and the rest of the Round Lake habitat area available, it is expected that Mink would most likely be exposed on an intermittent basis during any given year.

- Population portion "at-risk"
  The portion of the local populations actually exposed is not known. Based on literature regarding Mink habits and population densities, it is projected that one adult male, and one adult female and her young could be exposed at the site each year.

- Contact intensity
  Exposures to PCBs were developed as a part of the Work Plan and are presented in Appendix P.

6.6.7 Exposure Profile for Amphibians at Round Lake Under the Current Scenario

Resident amphibians will co-occur with the COCs due to the presence of the COCs in the surface waters. Again, the spatial extent of co-occurrence for the local populations is expected to be equivalent to the spatial extent of the contamination "footprint" within Round Lake, which was described in the previous section. These receptors will also contact the COCs while inhabiting the aquatic environment—in essence; they exist, during a large portion of their life cycle, in the waters that contain the COCs such that they are continuously exposed for those periods. Juvenile amphibians will be relatively more exposed than adults due to their movement habits.

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.

LOCAL POPULATIONS OF AMPHIBIANS AND THEIR EXPOSURE CHARACTERISTICS

As part of the amphibian study conducted at TCAAP (see 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 clamitans, Pseudacris crucifer,* and *Rana pipiens*. Overall, relatively few amphibians were found at Round Lake. There may be several reasons for low relative abundance due to limitations and uncertainty in the study, as well as environmental problems (see Table 6.1 in Appendix M).

- Exposure duration
  The entire Embryo and juvenile portions of the life-cycle

- Exposure frequency
  Continuous during the juvenile life stages; intermittent at most for adult amphibians.

- <u>Population portion "at-risk"</u>
  The portion of the local populations actually exposed is not known

- <u>Contact intensity</u>
  COC concentrations are provided in Exhibit 6-9 through Exhibit 6-12.

6.7 ROUND LAKE ANALYSIS: FUTURE SCENARIO

The proposed future scenario for Round Lake is a water level manipulation and sediment consolidation plan that is described below. No new TCAAP-source of COCs is expected; any contamination is historical. In the future scenario, the COCs are the same as those in the Round Lake current scenario. Barium, cadmium, and zinc are the surface water COCs, and cadmium, chromium, copper, lead, silver, vanadium, zinc, and PCBs are the sediment COCs.

6.7.1 Description of Proposed Active Water Level Management

The current USFWS management plan for Round Lake (USFWS 1982 and 1998) identifies two potential strategies for improving habitat quality to attract higher numbers of migrating wildlife, especially waterfowl and shorebirds. In summary, the preferred course calls for the implementation of actions to increase the ratio of emergent vegetation to open water from the existing 10:90 to a more productive ratio of about 50:50 by initiating a complete draw down of the lake. The USFWS refers to this plan as their *optimum wildlife alternative*. This completed draw down will de-water the lake, which will consolidate the sediments and allow for emergent plants to germinate and take root. Water levels would then be brought up with the emergent growth to normal pool levels. This pool level would be maintained for several years until emergent plants become sparse again. Additional drawdowns would not de-water the lake to the same extent.

Drawdowns are a wetland management tool in which water levels are manipulated in a strategic manner to promote plant growth and concentrate food sources (i.e., benthic invertebrates) for waterfowl (Fredrickson 1991). The time period of the major drawdown (Spring) proposed by USFWS is expected to promote excellent growth of swamp timothy grass, crabgrass, panic grass, wild millet grass, dock buckwheat, cocklebur and beggarticks because it exposes seeds of these aquatic plants, allowing for germination (Frederickson 1991).

Such actions at Round Lake would be expected to provide a more attractive mix, of open water and cattail-brush, for breeding and migratory waterfowl. The improved habitat would also benefit numerous other marsh birds and animals including muskrats, mink, heron, and marsh wrens (USFWS 1982). The second potential action, though less desirable from the USFWS point of view, would be partial drawdowns that would increase the coverage of submerged vegetation; however, it would not allow as broad a consolidation of bottom sediments as the preferred actions would. Regardless of the active management strategy, the level of potential ecological risk associated with the characteristics and extent of the contaminants in the system may influence the future use of the site.

In order to further explain the plan, the USFWS provided additional information to USACHPPM (Schreiner 2000, also included in Appendix I). The plan is a six-year water level manipulation management plan that will implement a gradual full drawdown of Round Lake in the first year (to 883.5 mean sea level (msl)) starting around May 15. The water level will be lowered approximately one foot every 8-10 days until 883.5 msl is reached. Around August 15, the water level will be raised by one foot every 12-14 days until the normal pool level (888.5 msl) is reached. During years two and three, the normal pool elevation of 888.5 msl will be maintained. In year four, a partial drawdown will be conducted beginning around May 20. The water level will be lowered by approximately one foot every 8-10 days to reach 886 msl. Beginning around July 15, the water level will be raised 0.5 foot every 10 days until a normal pool elevation of 888.5 msl is reached. In years five and six, the normal pool elevation will be maintained until September 1. The water level will then be lowered 1 foot to 887.5 msl and held at that level until November 10, at which time the pool will be raised to normal elevation.

Appendix I presents a map (figure I3) showing the water level at maximum pool elevation and at the USFWS minimum pool elevation after full-drawdown.

6.7.2 Review of COCs for Future Conditions

Since a continuing TCAAP source does not exist for contaminants, evaluated COCs for the future scenario were the same as those identified for the current scenario with the exception of the possibility of additional sediment contaminants present in sediments deeper than the surface sediments (0-1ft).

Appendix I contains tables showing data collected during the OU-2 Feasibility Study for various sediment depth intervals up to a depth of 6 feet below the sediment surface. Based on a review of these tables, concentrations of the majority of the chemicals either get smaller or stay relatively equivalent in the deeper sediment intervals. This condition indicates that if the sediments are disturbed such that the deeper sediments are exposed or otherwise transport their chemicals to the surface, the concentrations would not be expected to increase. However, several heavy metals were detected in the deeper sediments at concentrations greater than those found in the top one foot. These chemicals (cadmium, copper, lead, manganese, nickel, vanadium, and zinc) may potentially pose an exposure problem in the future and were added to the COC list for this scenario if they were not already current scenario COCs. PCBs are also known to exist in the deep sediment (see Appendix P for discussion on the investigation of this issue).

Round Lake COCs: Future Scenario

Surface Water
Barium
Cadmium
Zinc

Sediment
Cadmium
Chromium
Copper
Lead
Manganese
Nickel
Silver
Vanadium
Zinc
PCBs

6.7.3 Expected Changes in Sediment Physiochemical Characteristics

The future physiochemical characteristics of the sediment in Round Lake are some of the most important factors in qualitatively determining if there will be an increase in ecological risk from exposure to sediment COCs, as a result of the future management practice. In the first year of the management plan, there will be a complete drawdown, exposing a large majority of sediment in the northern section of the lake to the ambient air (see Figure I3 in Appendix I). The main concern is that COCs in the sediment will then become available to ecological receptors either directly from the ingestion of sediment or from their transport to the surface water.

Four chemical processes are believed to cause metals to move out of the sediment and possibly become available to ecological receptors: elevation of salinity, changes in oxidation-reduction state of sediments, reductions in pH, and the presence of complexing agents (Rand 1995). Some of these processes are expected to occur when sediments are exposed to oxygen in the air due to drawdown.

Currently, it is expected that the sediments in Round Lake are mostly anoxic, (more so in summer than winter) (USACHPPM 1998b). Anoxic sediments provide a condition for optimum sulfate reduction rates (i.e., AVS formation). Acid volatile sulfides (AVS) are a reactive pool of solid-phase sulfide that is available to bind divalent metals (cadmium, copper, nickel, lead, and zinc) and render that portion unavailable to biota (Suter et al 2000). Therefore, it is thought that the more AVS that is formed, the more metals can be bound and be biologically unavailable. In some areas of Round Lake sediment metals may be relatively unavailable due to binding with AVS (USACHPPM 1998b). In addition, the cation exchange capacity of the sediments in the northern area of Round Lake (48.7 to 77.7 meg/100g) (USACHPPM 1998b) indicates that the northern sediments are organic. In a fashion similar to binding to AVS, metals can bind to this organic matter in the sediment and may become unavailable to benthic organisms and aquatic plants. Other areas are less organic/more sandy (i.e. central portion), and COCs are not concentrated there.

Even though the current status of the lake indicates that metals may be bound to some extent and unavailable to biota, drawdown of the lake could alter this situation. One of the results of drawdown is an acceleration of the oxidation (change in the oxidation-reduction state) and breakdown of the muck that accumulates on the bottom of the lake as the sediment becomes exposed to the ambient air (City of Tallahassee 2001). As the muck (organic matter) breaks down, metals that were once bound can be released and become bioavailable. Also, when the AVS is oxidized, the metals that it bound can be released and become available. It should be noted that the AVS formation typically follows a seasonal trend, such that in summer when temperatures are higher and organic matter degradation rates are higher, the AVS formation rate is usually higher (Van Den Berg et al., 1998). If drawdown is initiated in May, before a greater amount of AVS is formed, then it is likely that metals may be in the surface water, rather than bound to AVS in sediment, and will be carried out of the lake in the drawdown water. This would then result in less exposure to metals via sediment ingestion. Although we have no direct evidence that the trend in quarterly Round Lake surface water concentrations (see Tables I6 – I9) of cadmium, copper, and zinc are directly related to AVS binding, it appears possible that the increase in surface water concentrations of these COCs during the second quarter sampling (January 2000), and then a gradual decrease in quarters three (April 2000) and four (June 2000) may be attributable to AVS binding of surface water metals.

There are also environmental factors other than AVS and oxidation that may have an effect on sediment physiochemistry as a result of drawdowns. These factors (i.e., sediment drying, bioturbation, pH, and total organic carbon) may affect the bioavailability and spatiotemporal distribution of sediment COCs. The environmental factors affecting sediment drying properties can include the ambient air velocity, amount of rainfall during drawdown, sediment organic carbon content, and algal and plant infestations. The events that will influence whether or not COCs bound to the sediment will become available to biota are the oxidation of the sediment and how the sediment dries. There are three stages in the evaporation of water from soil. The first stage does not differ greatly from how water normally evaporates from the surface water of the lake, the second stage is controlled by the soil's ability to transport liquid water to the drying surface and is a function of capillary rise within the soil (Castelli 1998). The third stage is characterized by slow drying when water movement to the soil surface occurs by vapor diffusion. If the soil surface is subject to a relatively high evaporation rate, the capillary transport may be broken and a dry semi-permeable crust would form on the top layer, hindering movement of water to the surface from lower depths (Castelli 1998). If this semi-permeable crust forms, it is not expected that any contaminants from the deep sediment may be brought to the surface and made available. This crust would act as a natural cap to lock in deep sediment contaminants (Castelli 1998). However, if the sediment dries slowly it is likely that contaminants from the deep sediment may be transported to the surface sediment where they may become available to ecological receptors. If the soil becomes mostly desiccated, cracks will form that may penetrate to deep sediment and that will become oxidized and be a potential source of release of historic contaminants (see Appendix I for tables listing the deep sediment contaminants). It is difficult to predict the depth of cracks that may form in Round Lake sediment during the drawdown process. A 1978 survey of Round Lake indicated that the bottom types at the lake are muck and detritus at 80% of the sampled stations and sand/boulders at 20% of the stations (USFWS 1982). A sandy surface substrate is expected in the middle of the lake, and deep sediment is expected to be mostly sand, so it is thought that crack length should be small. Also, due to the high organic content of the sediment (associated with the muck and detritus), it is not expected that the sediment could dry enough during the 3 month drawdown period to form substantial cracks that would allow upward transport of deep sediment constituents.

Kadlec (1989), evaluated the effects of drawdowns on freshwater marsh sediments during a five year period. He found that neither drawdown nor flooding changed the water content of the sediments or the organic matter content, but sediment nutrients increased during drawdown. However, he did find that the bottom sediment surface sank (consolidated) in the first year of drawdown but did not change significantly in the second year. He attributed no change in water content of the sediment to heavy accumulation of litter and mats of filamentous algae, which retard the drying. Kadlec also stated that organic sediments dry slowly and retain large amounts of water, 200% or more by weight, even after prolonged (up to 2 years) absence of surface water. The high organic carbon content in Round Lake sediment (average concentration = 144,000 mg/kg (USACHPPM 1998a)), may not permit drying of the sediment, such that it remains a wet muck. It would then be unlikely that sediment COCs would be available to ecological receptors. Another factor that may hinder sediment drying and oxidation is the freshwater plant Eurasian Water milfoil. As of April 24, 2000 Round Lake is listed as infested with this plant by the Minnesota Department of Natural Resources (MDNR 2000). Infestations of Eurasian Water milfoil are known to rob oxygen from water by preventing the wind from mixing the oxygenated surface water to deeper water. Also, the dense mats it forms can also increase sedimentation rate by trapping sediments (WSDE 2000). Upon drawdown, as this mat of vegetation collapses on the sediment, it may not allow for oxidation of the sediment. As this vegetation collapses onto the sediment, it also may reduce the contact of the waterfowl, wading birds, and aquatic mammals with the sediment._Should this be the case it is highly unlikely that any COCs in the sediment would be available to adversely affect any of the ecological receptors.

Based on the current information we have about Round Lake sediments such as the high organic carbon content that may prevent sediment drying and not allow for oxidation of the sediment and release of bound metals, the sandy composition of subsurface sediments that is related to sediment drying and crack formation, the accumulation of algae and plant material that will collapse on the sediment during drawdown, and the information available in the literature about sediment drying, it seems unlikely that the sediments in the northern section of Round Lake would release deep sediment constituents to the surface water in concentrations greater than what is currently present. However, there is still the potential that exposed sediment could become oxidized due to wind mixing in shallow water areas and wind movement over exposed sediments, and release surface sediment COCs to the surface water. This sediment then becomes a secondary source of COCs from the sediment to the surface water and then an exposure concern for biota.

There is one other confounding factor when trying to assess the sediment in Round Lake and it is related to the variation in lake depth. The shallower northern end of the lake supports a large majority of cattails such that the organic matter content of the northern lake sediment is high. In the middle deeper portion of the lake, the sediment is sandier and there is less organic matter as a result of less vegetation. As drawdown occurs, sediments toward the middle of the lake where there is no vegetation and where sediment is relatively sandier may become oxidized and also dry to the point were deep sediment constituents would be transported to the surface via capillary action and cracking (exposing deep sediment). Because the COC concentrations in the deep sediment in the middle of the lake are lower than in other sections of the lake, there may not be concern for increased bioavailability of metals. On the contrary, around the perimeter of the lake where the organic matter content is high and collapsing vegetation will cover sediment, drying to the extent that cracking would occur and release deep sediment constituents will most likely not occur (See map I3). This variation in lake sediment drying characteristics between deep water and shallow water areas may not pose a problem because the majority of contaminants appear to be in the shallower northern section of the lake (see map I2).

6.7.4 Magnitude and Distribution of COCs at Round Lake Under the Future Scenario

MAGNITUDE

If the sediments become oxidized during the drawdown scenario, it is expected that sediment COCs (metals) would then become bioavailable to sediment dwelling organisms, aquatic plants, and more accessible to biota that may accidentally ingest sediment. The empirical concentrations of COCs in the sediment are not expected to change under the oxidizing condition, only the COC availability, and thus the potential for adverse biological effect. If the sediments dry, forming cracks that allow deep sediment constituents to be transported to the surface, then the COC concentration in sediment is expected to increase. If these newly transported COCs are oxidized, then it is expected that the bioavailable fraction of these COCs will increase, as will the potential for adverse biological effect to biota. Once the water level begins to be increased after the drawdown, surface water concentrations of what are currently sediment COCs are expected to increase because what was once bound to sediment would likely be oxidized and released into the dissolved phase. If this release of sediment COCs occurs upon refilling the lake, barium, cadmium, and zinc would no longer be the only surface water COCs. Other metals listed in the textbox at the beginning of this section would also have to be considered. Due to the speculative nature of what may occur upon drawdown, it is difficult to predict what effect any of the above scenarios would have on biota at Round Lake in the short time period that drawdown would occur and refilling would begin (3 months).

SPATIOTEMPORAL DISTRIBUTION

The spatial and temporal distribution of the COCs is as important as the measured concentrations (USEPA 1998). Because the evaluated source site (Site I) is no longer in operation, and the contamination is historical, any temporal variability will be seasonal, unless there is drawdown. The COCs are inorganics and are therefore environmentally persistent. Their presence through time is expected, and the bioavailable amount of the COCs may change due to variation in sediment conditions (e.g., pH, temperature, nutrient loads, flow volume, oxidation). This aspect of exposure was not directly evaluated. However, when drawdown occurs the temporal pattern of distribution may change. During drawdown, COCs may not be present in surface water, but upon refilling COCs may increase in surface water for a time, until they can be complexed with suspended organic matter that may be stirred up in the refilling process. If drying of the sediments occurs, COCs in deep sediment may increase in surface sediment due to capillary action processes transporting COCs through cracks that may form in the surface sediment during the drawdown phase, and then increase in surface water during the refilling phase.

The spatial distribution of the surface water COCs is expected to emanate from the northern area of the lake where the storm sewer emptied into the lake. The concentrations of COCs may decrease with increasing distance away from these areas, but this was not evaluated. The exact spatial distribution of the elevated levels of inorganics in the surface water is difficult to determine given the sampling design used in the field investigation (based on the information from the current scenario). With the future plan of a full drawdown at the lake, the current spatial distribution of surface water COCs may change when the lake is refilled. The change may be caused by metals that were once bound to anoxic deep sediment being re-released when cracks that form in the drying sediment are inundated with water upon refilling . Also, oxidation of surficial sediments may introduce contaminants to the surface water as a result of changing redox potentials associated with fluctuating water levels. Again, the actual spatial distribution cannot be determined.

The spatial distribution of sediment COCs is also expected to emanate from the northern area of the lake due to the storm sewer outfall. The exact spatial distribution of sediment COCs was not evaluated. With the future drawdown plan, the current spatial distribution of COCs in the sediment may change if deep sediment contaminants move to the surface sediment by the process of capillary action during sediment depuration. Oxidation of surface sediment may release contaminants to the surface water, thus changing the spatial distribution in the surficial sediment.

BIOLOGICAL

As a result of the future management plan, biological distribution of contaminants is not expected to change during the drawdown process; however, upon refilling it is expected that if the sediment dries then more COCs will be available in the surface water. If more COCs are available in the surface water than higher concentrations will be expected in biota. See Section 6.6.2 for a discussion of biological distribution in Round Lake.

6.7.5 Refined Conceptual Site Model

The initial conceptual model for Round Lake consisted of five risk hypotheses based on the proposed future use of the site and was presented in the Work Plan (USACHPPM 1998) and represented in this Problem Formulation (Section 3). The refined CSM further explains contaminant pathways and specific COCs. Figure B3-5 illustrates the links between the hypothesized contaminant source and potential adverse ecological effects under the current (or "no action") scenario.

1. Based on the aquatic toxicity of surface water COCs, if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

2. If sediment concentrations of COCs are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

3. Amphibian survival, development, and reproduction may be at risk due to contaminants present in the lake.

4. The health of aquatic mammals, such as mink and muskrat, may be impaired if they are excessively exposed to sediments containing elevated levels of COCs.

5. The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COPCs that may occur in the benthic invertebrates and fish that they consume at the lake.

6.7.6 Exposure Profile for Fish, Aquatic Invertebrates, and Algae at Round Lake Under the Future Scenario

The exposure profile for the future scenario is generally the same as that for the current scenario (see Section 6. 6.3). The only difference is that if the concentrations of COCs increase in sediment, water, and biota, then the exposure to fish, invertebrates, and algae increases. Also, if the spatial extent of the COCs increases, then the exposure increases. The spatial extent may increase if sediment COCs are released to the surface water and become available to biota via uptake from the surface water and are then distributed around the lake. During drawdown, aquatic invertebrates may move out of the northern sediment (following the water), to the middle of the lake. The middle of the lake appears to have a lower presence of COCs than the northern section of the lake. This movement of invertebrates would decrease their exposure to sediment COCs during drawdown. However, there may also be a negative physical effect of the drawdown, killing many invertebrates by trapping them in small pools of water that may evaporate. Floating cattail mats may also settle on many invertebrates, trapping them in the process. Fish will be concentrated in the middle of the lake as well, during drawdown, such that they would be less exposed to sediment COCs. However, fish may then be stressed not only by COCs but also dissolved oxygen decreases due to fish density and less plants photosynthesizing. Upon refilling of the lake, depending on oxidation of the sediment, fish, invertebrates, and algae may be more exposed to water and sediment COCs.

6.7.7 Exposure Profile for Benthic Organisms at Round Lake Under the Future Scenario

In the *Bioavailability of Sediment-Metals in Round and Sunfish Lakes* (USACHPPM 1998b), five benthic species were found to dominate Round Lake. The following species were found in abundance: *Crangonyx gracils* (amphipod), *Polypedilius fallax* (shredder), *Glyptotendipes lobiferus* (midge), *Physella gryrina* (freshwater snail), and *Chaoborus punctipennis* (midge). Benthic organisms such as those mentioned above, are believed to accumulate metals mainly through direct ingestion of sediment and from solution (Lee et al. 2000 and Beyer et al. 1996). The drawdown plan is scheduled to occur during the breeding season of some waterfowl known to frequent Round Lake. During drawdown, aquatic benthic organisms may move out of the northern sediment (following the water), to the middle of the lake. The middle of the lake appears to have a lower presence of COCs than the northern section of the lake. This movement of benthics would decrease their exposure to COCs during drawdown. Waterfowl rely heavily on these organisms during breeding and drawdowns create optimum foraging conditions of these organisms. However it is difficult to assess the potential toxicity of the sediment to these invertebrates in the future scenario because several studies agree that physical aspects of water level drawdowns severely affect benthic invertebrate life (Castelli 1998). One example of effects may be the loss of vegetative cover in the water, which would decrease the amount of preferential habitat. There is no direct qualitative method to determine if effects to invertebrates would be a result of increased exposure to contaminants or a result of physical effects caused by the drawdown. However, if invertebrates do experience increased exposure to contaminants as a result of the drawdown, then it is possible that waterfowl ingesting them will be negatively affected as well.

6.7.8 Exposure Profile for Waterfowl and Wading Birds at Round Lake Under the Future Scenario

A previous ecological assessment at TCAAP (USAEHA 1991), discussed an unpublished study by University of Minnesota students from 1981 in which 18 species of waterfowl were found to be using the lake during some part of the spring. The Pied-billed Grebe, American Coot Mallard, Ruddy Duck, Wood Duck, and Canada Goose were seen at Round Lake on enough survey dates to be considered potential breeders. However, lack of food and cover were presented as possible limiting factors in the number of birds using the lake. The USAEHA (1991) report indicated that the bird community at TCAAP was healthy, but there was no quantitative information available. As for the future scenario there also is not quantitative information available. Please see Figure 14 that shows the current species that are present at Round Lake, highlighting those species expected to be present during the major drawdown period. The presence of these birds does not necessarily mean they are nesting at the lake. However, after drawdown is complete and habitat is improved, birds are expected to be nesting and raising young.

Without site-specific wading bird activity data, it is difficult to predict how wading birds will be utilizing Round Lake. However, based on life history characteristics some generalizations can be made. Because great blues nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). It is difficult to determine the amount of foraging habitat available at Round Lake without field data. However, based on the available information it is possible that 4-5 wading birds may dominate at Round Lake. It is likely that in the future scenario, no more wading birds will be attracted to the lake due to the solitary hunting style of the birds. Also, the habitat for these birds is not the focus of the drawdown and will not directly be improved.

If bioavailable concentrations of COCs increase in the sediment and surface water, exposure to waterfowl will be increased, mainly through increased COC concentrations in their food items (i.e., invertebrates and vegetation). Also, changes in the spatial distribution of contaminants will also change the exposure to waterfowl. During the first year of drawdown, waterfowl may not yet be attracted to the lake because the water volume may be too small, the ideal vegetation may not yet be present, and the mucky condition of exposed sediments may not be ideal for the birds. However, those birds such as mallards that do frequent the lake would most likely be less exposed to sediment COCs if they are feeding in the middle of the lake, where COC concentrations are lower than in the northern section. Once the lake is refilled and vegetation has improved, more birds are expected to be attracted to the lake and thus exposure of COCs to the waterfowl population would increase.

6.7.9 Exposure Profile for Aquatic Mammals at Round Lake Under the Future Scenario

According to the *Wildlife Exposure Factors Handbook* (USEPA 1993), in pond and lake habitats, shoreline length is a more important factor than overall habitat area in determining muskrat density. Many investigators estimate muskrat density by counting the number of houses or push-ups and multiplying by a factor ranging from 2.8 to 5.0; however, this method is questionable.

In addition, a population of muskrats is very dependent on the water cycle of the lake they live in. In times of drought, populations dwindle, mainly as a result of lack of food and also from predation. When the marshy areas re-flood, there is an abundant supply of new plants. The abundance of food fuels the female's reproductive potential of 14 – 21 young per year. Food abundance also allows better than average survival of the young and adults. Under these conditions, a population of only 20 muskrats can grow to more than 100 in a single summer (Illinois DNR, 2001).

It can be expected that Round Lake is capable of supporting approximately 450 adult muskrats under the current scenario. This is a very conservative estimate, assuming that there is ample food and that the whole of the northern, eastern and southeastern portions of the lake is inhabitable. However, once drawdown is initiated it is expected that the muskrat population will decrease from its current number due to loss of protected dens and increased predation. Upon refilling of the lake the number of muskrats may again increase. Exposure of muskrats to the sediment may also increase during drawdown as they consume cattail roots that are no longer submerged. During drawdown, the plants may also be able to take up more COCs as the COCs become bioavailable under oxic conditions. This would also increase muskrat exposure to COCs. Mucky conditions of the sediments may also present increased exposures when the animals are grooming. Upon refilling of the lake the exposure to the COCs is expected to either stay the same or increase as compared to the current Round Lake scenario.

It is possible that during the drawdown period the mink population may increase from its current number due to increased availability of food (muskrats). However, no quantitative estimate of population increase can be made. As with the muskrat, mucky conditions of the sediments may present increased exposures when the animals are grooming. Upon refilling of the lake the exposure to the COCs is expected to either stay the same or increase as compared to the current Round Lake scenario.

6.7.10 Exposure Profile for Amphibians at Round Lake Under the Future Scenario

The exposure profile in the future scenario is expected to be the same as that in the current scenario (see Section 6.6.6).

However, it is expected that if COC concentrations increase as a result of drawdown, then exposure would increase. Also, physical factors of the drawdown may effect the amphibian population. Egg laying of most amphibians occurs in March through June with some amphibians laying even July. Shallow areas of calm water would not be present for egg laying, as the water is drawing down. Eggs that are laid may be trapped in shallow pools that could evaporate before amphibians develop out of the tadpole stage. Settling floating cattail mats may also trap amphibians, thus reducing amphibian numbers. After full drawdown, exposure to amphibians may be increased or stay the same as the current scenario.

7. EFFECTS ANALYSIS

7.1 ANALYSIS ELEMENTS

The effects analysis of an ecological risk assessment describes the nature of the toxic effects of the chemicals of concern (COCs) and their magnitude as a function of exposure. This effects assessment examines three primary elements: (1) the relationship between chemical concentrations and ecological effects, (2) linkages between measures of effect and the chosen assessment endpoints, and (3) the cause-effect relationship between the chemicals and the effects (USEPA 1998).

7.1.1 Measures of Effect (i.e., stressor-response relationships)

The nature of the stressor-response relationships used in a particular assessment depends upon the purpose and scope of the project. This assessment was based on the measures of effect and ecosystem or receptor characteristics selected in the Work Plan (USACHPPM 1999). The measure of effect selected for each ecological entity evaluated in the assessment are described below.

7.1.2 Strength of the Link Between Measures of Effect and the Assessment Endpoints

As discussed previously, assessment endpoints represent the environmental values of concern for a risk assessment and are the effects that decisions will be based upon. However, assessment endpoints cannot always be measured directly. When measures of effect differ from assessment endpoints, sound and explicit linkages between them are needed (USEPA 1998). Each of the following sections reviews the measures of effect and their linkages to the assessment endpoints. Often the strength of the linkage has a direct influence on risk management decisions based on any given assessment endpoint. If linkages are strong, then basing risk management action on the measures of effect can be relatively easy to justify. If linkages are weak, then basing risk management actions on such measures can become difficult because uncertainty can be high.

7.1.3 Causality

Causality is the relationship between cause and effect. Without a sound basis for linking cause and effect, uncertainty in conclusions of an ecological risk assessment are likely to be high (USEPA 1998). It should also be noted that causal inference is often impossible unless there is an impairment to the aquatic system, and that impairment is defined in terms of a specific effect that is quantified as a count or continuous variable, such as the prevalence of dermal lesions or the abundance of key fish species(Suter et al., 2002). Evidence of causality may be derived from observational data (e.g., bird kills are associated with field application of a pesticide) or experimental data (e.g., laboratory tests with the pesticide showing bird kills at levels similar to those found in the field), and causality can be strengthened when both types of evidence are available (USEPA 1998). When observational data such as biological surveys are used which indicate an association between stressor and effect, then there are several sets of standard causality criteria with which to judge whether the observed association is due to a cause-effect relationship. When experimental data are used, then the differences between laboratory conditions and field conditions must be considered (e.g., extrapolation techniques). Criteria that strongly affirm causality are strength of association, predictive performance, demonstration of a stressor-response relationship, and consistency of association. Each of the following sections reviews the measures of effect and their potential to demonstrate causality.

7.2 EFFECTS IN FISH, AQUATIC INVERTEBRATES, AND ALGAE

The assessment endpoint for this ecological entity is the survival, growth, and reproduction of fish, aquatic invertebrates, and algal species at Marsden Lake, Sunfish Lake, and Round Lake.

The survival, growth, and reproduction of fish, aquatic invertebrates, and algal species were not measured directly. The measures of effect used were as follows:

- Minnesota Water Quality Standards, or
- Other effects-based benchmarks for fish, invertebrates, and algal species.

The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. Toxicity benchmarks are chemical concentrations associated with a specified level of toxicological effect; for example, a concentration at which no adverse effects are observed in the laboratory. Another commonly used benchmark is the LC_{50} or the lethal concentration for 50 percent of the tested organisms. Such benchmarks are point estimates of a specific effect level along a stressor-response curve. For fish, aquatic invertebrates, and algae, the Minnesota Water Quality Standards for Class 2B waters were used, which are based on the protection of several aquatic species from adverse toxic effects. These toxicity-based criteria must have a minimum of 8 “genus mean acute values” representing 5 biological groups (MPCA 2000a). The acute and chronic standards were considered in the assessment. At concentrations below the chronic and low-effect values, some magnitude and type of effect is rarely expected to occur in any species, and usually expected to occur at concentrations above the chronic and low-effect values. Acute effects are often measured in terms of mortality or other debilitating effects and chronic effects are usually measured as mortality, reduced growth, reproduction impairment, harmful changes in behavior, and other nonlethal effects (MPCA 2000). The temporal pattern for acute water quality tests is generally 96 hours or less, and chronic tests are usually one-tenth the life span of the organism or more (generally lasting 2 weeks or longer) (MPCA 2000).

Minnesota uses the USEPA National Ambient Water Quality Criteria unchanged unless more scientifically defensible data becomes available or site-specific conditions warrant a recalculation. When no appropriate Minnesota Water Quality Standards were available for a chemical, then alternative sources were used based on a standard protocol for selecting benchmarks from the available reference literature. We used the current USACHPPM protocol (*Standard Screening-Level Measures of Ecotoxicological Effects, Revision 3,* 1 October 2000), which is included in Appendix O. In the Risk Characterization (Section 8), the benchmarks shown in the appendix are integrated with the field-data to determine the potential for effects.

Additionally, for the PCB assessment in Round Lake, residue-effects data for aquatic organisms – the *Environmental Residue-Effects Database* (ERED) - was searched. The ERED is a compilation of data, taken from the literature, where biological effects (e.g., reduced survival, growth, etc) and tissue contaminant concentrations were simultaneously measured in the same organism. The database was limited to those instances where body burdens are associated with observed biological effects for a variety of contaminant concentrations in sediment. Residue-effect data was found for PCBs and compared to PCB concentrations measured in black bullhead tissue collected from Round Lake by USFWS (see Appendix P) to determine the potential for effect.

7.2.1 Linkage to Assessment Endpoint(s)

Water quality criteria used in this assessment are linked to this endpoint because survival, growth, and development of at least 95 percent of a population of aquatic organisms should be protected by a water quality standard (MPCA 2000a and USEPA 1999). However, by using the benchmarks we extrapolate from the laboratory to the field and between different species (i.e., between taxa). These extrapolations introduce uncertainty. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more, or less sensitive to the chemicals. In addition, the species tested in the laboratories may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar.

Residue-effects data are directly linked to the assessment endpoint because they provide a dose-response relationship, such that a measurable fish tissue concentration is linked to an observable adverse effect in fish. There is some uncertainty associated with these measures because the species presented in the database may not be the same as that at the site, and the data may be from a laboratory study rather than from field conditions. However, we assume that the effect levels will be similar.

7.2.2 Causality

The only line of evidence used as a link to determine causality of effects to these receptors was an HQ calculation based on site water concentration comparisons to water quality criteria. Water quality criteria used in this assessment to perform a toxicological screen with site sample concentrations are promulgated as standard values based on a series of 8 acute tests with different species and families of organisms, and 3 chronic tests. These tests are analyzed using hypothesis testing statistics, such that responses of each tested medium are determined to be either statistically significantly different from those in reference or control media, or not. However, 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). 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.

Because no toxicity tests were performed on site surface water samples, the best indicator of possible impairment to the system was this simple comparison of the concentration to the criteria value. No field observations are available that could better determine if surface water in Marsden Lake North and South, Pond G, Round Lake, Rice Creek, and Sunfish Lake could be causing adverse effects in fish, aquatic invertebrates, and algae. Results of the comparison of selected COC concentration data to water quality criteria are presented in Section 5 of this report.

For Round Lake, Sunfish Lake, and Rice Creek, surface water samples were taken quarterly for one year to better understand the temporal pattern of chemical concentrations. For fish, aquatic invertebrates, and algae, this measure of effect (i.e., HQ based on water concentration and water quality criteria) does not directly indicate that concentrations measured in the surface water will cause adverse effects in fish, invertebrates, and algae. Marsden Lake (North and South) and Pond G were sampled only one time in 1999 and were not subject to temporal evaluations. However, calculation of an HQ for these two sites still has the same implications as those for Round, and Sunfish Lakes and Rice Creek.

7.3 EFFECTS IN BENTHIC ORGANISMS

The assessment endpoint for this ecological entity is the survival, growth, and reproduction of benthic organisms at Marsden Lake North and South, Sunfish Lake, and Round Lake. The survival, growth, and reproduction of benthic organisms at these sites were not measured directly. The measures of effect used were as follows.

- Effects-based benchmarks for benthic organisms
- Sediment toxicity tests using benthic organisms at Sunfish Lake (sediment collected in 1999) and Round Lake (sediment collected in 1995 and 1999)
- Benthic community evaluations at Sunfish Lake(1993) and Round Lake (1993 & 1995)
- Metal bioavailability tests at Sunfish Lake and Round Lake (1995)

The potential for adverse toxicological effects in these organisms was inferred by comparing site sediment concentration data to toxicity benchmarks, as was described for other aquatic organisms. For benthic organisms, regulatory agencies in this country do not have consistent, chemical-specific standards for chemicals in the sediments, such as that available for chemicals in the water column. We used the current USACHPPM protocol (*Standard Screening-Level Measures of Ecotoxicological Effects, Revision 3*, 1 October 2000), which is included in Appendix O, to derive benchmark values to be used in Army risk assessments. In the Risk Characterization (Section 8) the benchmarks in this appendix are integrated with the field-data to determine the potential for effects.

In the Fall of 1999, sediment toxicity testing (14- and 30-day chronic toxicity tests with *Chironomus riparius* and *Hyalella azteca*) was performed on field-collected sediments at Sunfish Lake and the northern portion of Round Lake to reduce uncertainties with evaluating risks to benthic organisms at these two sites. Appendix L presents the evaluation of the *Chironomus riparius* tests (the *Hyalella azteca* tests were rejected so those results where not included) as well as information relating to issues with the tests. Sediment samples were collected from the Northern portion of Round Lake, at all water monitoring locations in Sunfish Lake, and from three locations (north end, middle, and south end). in Snail Lake. Timing of the *Chironomus* tests was such that first instar stage midges were used in order to determine emergence time of sensitive juveniles. In the 14-day tests, growth and development were measured as the number of emerged organisms. For the 30-day test, which was only carried out to 22 days due to a large percentage of emergence in the controls, survival of *C. riparius* was measured by evidence of emergence.

In 1995, 14-day and 30-day sediment toxicity tests using *Hyallela azteca* and *Chironomus riparius* were performed on sediments collected from the southern portion of Round Lake, near the lake outlet. These sediment evaluations were performed to determine whether or not the sediments were toxic, and if they were the cause of the low species diversity in the benthic macroinvertebrate community observed from the Tier I data.

Benthic community surveys were conducted at Round Lake (1993 and 1995) and Sunfish Lake (1993). The information collected in 1993 was used in the 1997 Tier I assessment. The information collected in 1995 was first reported in the *Bioavailability of Sediment-Metals in Round and Sunfish Lakes* report (USACHPPM 1998a).

The sediment data that were collected in September 1995, were used to determine the bioavailability of divalent metals (cadmium, copper, lead, and zinc) in the sediments of Round Lake and Sunfish Lake. The study was also designed to provide information on the potential toxicity of identified metals in the sediment, the transfer of the metals in the sediment to biota, and the biological diversity of the associated benthic organisms.

7.3.1 Linkage to Assessment Endpoint(s)

Sediment quality benchmarks can be reasonably linked to this endpoint because survival, growth, and development are often the endpoints used to derive these benchmarks. Many sediment benchmarks have been derived from chemical concentrations detected in whole sediments that are associated with adverse effects to lab tested and/or *in situ* tested benthic organisms. Some benchmarks, however, are based on presence/absence of a particular species when comparing field sediments across multiple locations. Such benchmarks are less relevant to the assessment endpoint.

By using the benchmarks we extrapolate from the laboratory to the field, between different field sites, and between taxa. These extrapolations introduce uncertainty. The species tested in the laboratories or observed in the field-collected sediments of reference areas may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more or less sensitive to the chemicals. When we use benchmarks based on reference sediments, then we assume that environmental and ecological processes between the reference sites and the study site do not differ in significant ways.

In the sediment toxicity tests, emergence and mortality were used as test endpoints; growth and reproduction were not measured because the invertebrates emerged before the test was completed (see Appendix L, Memo of the acceptance of *Chironomus riparius* tests). Though growth and reproduction, *per se*, were not measured in the test, the linkage of the toxicity test to the assessment endpoint is stronger than standard toxicity benchmarks because actual toxicity of the site sediments were measured using standardized protocols under controlled laboratory conditions. However, it should be noted that the sediments were thoroughly homogenized before the test began. This procedure could have oxidized the sediments, causing previously bound sediment metals to be released to overlying water. This would then represent a worst case evaluation of toxicity, because in the field it is not expected that the sediments would be thoroughly mixed.

Benthic community evaluations can serve as estimates of the assessment endpoint. The community data can be linked to the system's characteristics, which can be used as a line of evidence when comparing the study site to the reference site.

The evaluation of the bioavailability of divalent metals in sediments is linked to the assessment endpoint in that, if the acid volatile sulfide (AVS) concentration in the sediment is greater than the simultaneously extracted metal (SEM) concentration, then the metal is considered to be unavailable to benthic organisms. Therefore, the metal in the sediment should not be causing toxic effects. However, the SEM:AVS molar ratio is dependent on temporal considerations, and when measured during the winter months it provides the worst case scenario of bioavailability.

7.3.2 Causality

The lines of evidence used to assess causality of adverse effects to benthic organisms were HQ calculations based on a comparison of site sediment chemistry data to sediment toxicity benchmarks, AVS:SEM ratios, benthic macroinvertebrate community surveys, and site specific sediment toxicity tests. Sediment toxicity benchmarks can be based on laboratory tests or field studies that show a stressor - response relationship between the chemical concentrations in whole sediment and associated adverse effects in benthic organisms or altered benthic community structure. However, 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). Use of site-specific sediment samples can affirm causality. When using site samples in toxicity evaluations, the bioavailability of contaminants is realistically represented as shown through SEM:AVS ratios, the form of contaminants is realistic, and combined toxic effects are elicited (Suter et al 2000). Thus co-location of sediment toxicity data with sediment chemical concentration data can provide more information to show causality of adverse effect from site sediment. It is also important to perform toxicity tests on an uncontaminated reference site in order to better understand if site sediment chemical contamination is the cause of adverse effects or if is based on other environmental characteristics. Additionally, macroinvertebrate community surveys do not provide a direct link between COCs and adverse effects, but they do provide information on community structure that can be used to indicate ecosystem health.

As with any scientific study, there can be uncertainties associated with the methods used to derive sediment benchmarks. There can also be uncertainties in contaminated site media testing, such that the samples may be unrepresentative, sample collection procedures could modify the media (such as oxidizing anoxic sediments, which could release bound metals), the exact cause of the toxicity is generally unknown, and use of an inappropriate reference site may show toxicity that is not truly indicative of the study site (Suter 2000).

7.4 EFFECTS IN AMPHIBIANS

The assessment endpoint for this ecological entity is the survival, growth, and reproduction of amphibians at Marsden Lake, Sunfish Lake, Round Lake, and Rice Creek. There is very little information available in the published literature on benchmark values for amphibians that would be similar to those used for the other aquatic organisms described above. Therefore, an amphibian study was conducted to test whether amphibians exposed to substances from TCAAP sites were adversely affected as a result of military activity. The measures of effect used were as follows.

- Mortality and development of frogs exposed to field-collected surface water
- Measures of immune function in frogs exposed to field-collected surface water
- Amphibian species richness at each site
- Amphibian species relative abundance at each site

The study attempted to determine (1) if there was a difference in amphibian species richness between TCAAP study sites and the reference sites, (2) if there was a difference in amphibian species relative abundance between the reference sites and the study sites, (3) if mortality and developmental toxicity of offspring differ between reference sites and the study sites, and (4) if surface water from the study sites adversely affects immune function. The temporal pattern for the FETAX test was 96 hours. The phagocyte response tests were conducted with frogs exposed to TCAAP surface water for 14 days. Biological surveys took place on 5-8 May 1998, 21-23 July 1998, and 19-20 April 1999. The immunological characterization was conducted 114 days after the last reproductive event of the Xenopus used in the FETAX test. Additionally, histopathological examinations were conducted on sections of thymus, lung, kidney, and liver from male and female frogs exposed to site surface water for 14 days.
The study is presented in Appendix M. In the Risk Characterization (Section 8) the results of the amphibian study are reviewed.

7.4.1 Linkage to Assessment Endpoint(s)

The FETAX (*Frog Embryo Teratogenesis Assay-Xenopus*) test was used to determine mortality and developmental toxicity from exposure to surface water at Marsden Lake. Thus, survival was measured directly. Growth was not measured, however, the developmental bioassay can be used to infer effects on organism growth; and may, in fact, be a more sensitive indicator of toxicity. Tests of immunulogical function in the frogs were conducted to infer whether immune system suppression is degraded in frogs that develop while exposed to field-collected surface water. Immunolological suppression can directly influence mortality, growth, and reproduction. The immunological response of adult *Xenopus* frogs was characterized by measuring phagocytosis of fluorescent microspheres and the production of radical oxygen intermediates.

Direct measurements of reproduction of amphibians in the field or laboratory were not made due to feasibility constraints. Additionally, exposure to sediment can be a relevant measure of exposure in some cases (i.e., through dermal exposures to wintering tadpoles or adults, or through food chain exposures); however, this study only evaluated exposure to surface water. This was the case due to the inherent difficulties in dealing with sediment in amphibians assays in controlled laboratory experiments (i.e., with dejellied eggs in FETAX, fouling the water in adult exposure assays). The possible effects of sediment were considered such that water soluble constituents would be expected to partition to the water and then to the organism. Therefore, exposure (dermal) to sediment COCs was assumed to be at equalibrium between water and sediment (pore water).

The field-based measures of species richness and species relative abundance (as measured by visual encounter surveys, aural estimates, and trapping) can be used in combination with site specific habitat information and historical information on amphibian abundance at the site to infer whether exposures to any COCs at the sites have had a community-level impact on the amphibians at the sites.

7.4.2 Causality

From the TCAAP amphibian study, Visual Encounter Survey (VES) data can be used to indicate if a site can support amphibians, but does not provide a direct link to adverse effects or causation of low density observed during surveys. The FETAX tests, histopathological examinations, and immunological examinations are better indicators of site specific chemically caused adverse effects in amphibians, because they were based on amphibian exposure to site water. These tests do not indicate exactly which COC could cause an adverse effect, but they do provide a link to assessing if there is a site-specific effect.

7.5 EFFECTS IN MAMMALS, WATERFOWL, AND WADING BIRDS

The assessment endpoints for these ecological entities vary by site, as shown.

- Marsden: Reproductive potential and productivity of aquatic mammals, waterfowl, and wading birds
- Sunfish Lake and Round Lake: Reproductive potential and productivity of aquatic mammals and wading birds
- Pond G: Water and sediment quality for mammals and wading birds using the pond

Reproductive potential and productivity in these organisms was not measured directly at Marsden Lake, Round Lake, or Sunfish Lake. The water and sediment in Pond G was measured directly, but the quality of that water for wildlife is inferred in the same way. The measures of effect used were as follows.

- Laboratory-derived chronic effect levels,
- Or other appropriate toxicity benchmarks

The potential for adverse toxicological effects in these ecological entities is inferred, in part, by determining if estimated field exposures will be greater than the laboratory-derived toxicity benchmarks. The available reference literature mainly provides single chemical toxicity data for a few laboratory mammals and birds where estimates of effect-levels are provided or can be calculated from the study data. The current approach in wildlife risk assessments is to use the following effect-levels to characterize toxicity: the no-observed-adverse-effect-level (NOAEL) and the lowest-observed-adverse-effect-level (LOAEL). The use of the NOAEL or LOAEL does not indicate how effects increase with increasing exposure, so the effects of slightly exceeding a NOAEL or LOAEL are not qualitatively or quantitatively distinguishable from those of greatly exceeding it (Suter et al 2000). Exceedance of a NOAEL-based benchmark does not suggest that adverse effects are likely; it simply indicates contamination that is sufficiently high to warrant further investigation (Sample et al 1996).

A standard protocol for selecting toxicity reference values (TRVs) from the available reference literature was used. We used the current USACHPPM protocol (*Standard Screening-Level Measures of Ecotoxicological Effects, Revision 3*, 1 October 2000), which is included in Appendix O. In the Risk Characterization (Section 8), the TRVs in this appendix are integrated with the field-data to determine the potential for effects.

7.5.1 Linkage to Assessment Endpoint(s)

Studies with reproductive endpoints were chosen first when selecting appropriate studies for TRV values because such effects are considered the most ecologically relevant and are linked to the assessment endpoints. 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. Those TRVs based on reproductive endpoints show a stronger linkage to the assessment endpoint than those TRVs based on other health effects.

By using TRVs, we extrapolate from the laboratory to the field and between taxa. These extrapolations introduce uncertainty. The species tested in the laboratories are usually not the same species of concern in the field. However, we assume that the effect levels will be similar and we have used tools to protect against differential sensitivity between the laboratory species and the wild species. Nonetheless, when we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more or less sensitive to the chemicals. For example, in laboratory toxicity tests, "fresh" chemicals are typically applied to the food, where in the field setting, any contaminating chemicals will be quite weathered, rendering them less toxic due to various reasons related to soil or water chemistry. When we use the TRVs without site-specific adjustments, we assume that environmental and ecological processes that occur in the field setting do not alter exposure or toxicity in important ways. This assumption introduces a significant amount of uncertainty in the risk assessment of wildlife, which is greater than for other ecological entities like benthic organisms. Field-studies that can measure absolute or relative bioavailability of chemicals in water and soil can be conducted, however, in most cases acceptable risk management decisions can be based on less specific characterizations of risk.

7.5.2 Causality

The main line of evidence used to assess causality to these receptors was the calculation of a HQ that was derived by modeling the dietary dose of the receptor and dividing that value by a TRV. In general, only studies that showed a clear dose-response relationship were used to calculate the TRVs used in this assessment. That is, the vast majority of the studies, upon which the TRVs are based, examined differing degrees of toxic effects in organisms exposed to different doses of the toxicant under controlled conditions. However, as with any scientific study there can be uncertainties associated with the methods used to derive TRVs. Single chemical toxicity test benchmarks found in published literature may be biased in that the forms of the chemical used in the tests are likely to be more toxic than that found at contaminated sites, 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.

8. RISK CHARACTERIZATION

Risk characterization, the final phase of the ERA, allows for the clarification of the relationships between the stressors (i.e., the chemicals of concern), effects, and ecological entities in order to reach conclusions about the occurrence of exposure and the adversity of existing or anticipated effects (USEPA 1998).

For each assessment endpoint and risk hypothesis at each of the study sites (as described in Section 3), risk estimates are provided according to each line of evidence. If risks are suspected to be present, then they are described in context of the significance of any adverse effects and lines of evidence supporting their likelihood. The procedures used to estimate risk for each line of evidence are reviewed in the next section (Section 8.1) and the approach for describing the risks is provided in the following section (Section 8.2). Site-specific risk characterizations are provided in Sections 8.3 through 8.9. The risk characterization also identifies and summarizes the important uncertainties, assumptions, and qualifiers in the risk assessment.

8.1 RISK ESTIMATION PROCEDURES

Risk estimation is the process of integrating exposure and effects data and evaluating any associated uncertainties (USEPA 1998). In this assessment, we prepared a risk estimate for each line of evidence. According to USEPA guidance (1997, 1998), risk estimates can be developed using one or more of the following techniques:

(1) Field observational studies and/or site-specific toxicity tests
(2) Categorical rankings
(3) Comparisons of single-point estimates of exposure and effects estimates
(4) Comparisons of exposure and effects estimates that explicitly incorporate variability and uncertainty
(5) Comparisons incorporating the entire stressor-response relationship
(6) Process models that rely on theoretical approximations of exposures and effects

This risk assessment relied on a combination of techniques, including field observational studies and/or site-specific toxicity tests (technique 1) and comparisons using single-point estimates of exposure and effect that highlight the variability in the collected site data (a blend of techniques 3 and 4).

8.1.1 Estimation Procedures for Fish, Aquatic Invertebrates, and Algae

The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. Risks for fish, aquatic invertebrates, and algae at Marsden Lake (North and South), Sunfish Lake and Round Lake were characterized using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (a blend of techniques 3 and 4).

For each COC, the risk estimate contains three pieces of information: (1) a hazard quotient based on the central tendency water concentration and the chronic water quality benchmark, (2) the percent of sample concentrations greater than the chronic water quality benchmark, and (3) the percent of sample concentrations greater than the acute water quality benchmark. Sample concentrations across the sites were compared to the water quality benchmarks in bar graphs. A bar for each sample location at the site was presented in the graph and the benchmarks were indicated in the graph, so the relation between the site concentrations and the concentrations of interest for predicting ecological effects could be viewed. In some cases, alternative benchmarks were placed on the graphs to illustrate other, alternative effect-levels that may be more appropriate for decision-making.

An additional line of evidence was prepared specific to fish exposed to PCBs in Round Lake. Residue-effect data was found for PCBs and compared to PCB concentrations measured in black bullhead tissue collected from Round Lake by USFWS (see Appendix P) to determine the potential for effect. Residue-effects data for aquatic organisms was taken from the *Environmental Residue-Effects Database* (ERED). The ERED is a compilation of data, taken from the literature, where biological effects (e.g., reduced survival, growth, etc) and tissue contaminant concentrations were simultaneously measured in the same organism. The database was limited to those instances where body burdens are associated with observed biological effects for a variety of contaminant concentrations in sediment.

8.1.2 Estimation Procedures for Benthic Invertebrates

The assessment endpoint (survival, growth, and reproduction of benthic organisms) was measured both directly and indirectly. Various lines of evidence were used to infer adverse effects in benthic organisms:

1. Comparison of sediment COC concentrations to effects-based benchmarks for benthic organisms
2. Benthic community evaluations at Sunfish Lake and Round Lake
3. Metal bioavailability studies at Sunfish Lake and Round Lake
4. Sediment toxicity tests at Sunfish Lake and Round Lake

At the Marsden Lake Sites, the potential for adverse toxicological effects in these ecological entities was inferred by evaluating the first line of evidence listed above. At Sunfish and Round Lakes, all four lines of evidence were available for inferring the potential for adverse toxicological effects.

Risk estimates for the first line of evidence, using effects-based benchmarks, were developed using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (a blend of techniques 3 and 4). For each COC, the risk estimate contains three pieces of information: (1) a hazard quotient based on the central tendency sediment concentration and the low effect sediment benchmark, (2) the percent of sample concentrations greater than the low effect sediment benchmark, and (3) the percent of sample concentrations greater than the moderate effect sediment benchmark. Sample concentrations across the sites were compared to the sediment benchmarks in bar graphs. A bar for each sample location at the site was presented in the graph and the benchmarks were indicated in the graph, so the relation between the site concentrations and the concentrations of interest for predicting ecological effects could be viewed.

Risk estimates for the second line of evidence were developed using the results of the benthic community surveys (a form of technique 1) that were conducted at Round Lake and Sunfish Lake in 1993 and 1995. The information collected in 1993 was first reported in the Tier I risk assessment (USACHPPM 1997). The information collected in 1995 was reported in the *Bioavailability of Sediment-Metals in Round and Sunfish Lakes* report (USACHPPM 1998b) and the *Sediment Toxicity Evaluation at Round Lake* (USACHPPM 1998a). A qualitative description of the benthic community is provided for each site (Round and Sunfish Lakes) that indicates relative species diversity and the abundance of pollution tolerant species.

Risk estimates for the third line of evidence (a form of technique 1) were developed using the results of the *Bioavailability of Sediment-Metals in Round and Sunfish Lakes* report (USACHPPM 1998b). This report evaluated the bioavailability of divalent metals (cadmium, copper, lead, and zinc) in sediments sampled in September 1995 at Round and Sunfish Lakes using an SEM/AVS approach. The study was also designed to provide information on the potential toxicity of identified metals in the sediment, the transfer of the metals in the sediment to biota, and the biological diversity of the associated benthic organisms. A qualitative description is provided for each site (Round and Sunfish Lakes) that indicates the potential for these divalent metals to be bioavailable and potentially toxic to benthic organisms.

The fourth line of evidence is based on sediment toxicity testing performed in Round and Sunfish lakes (technique 1). There are two sets of toxicity testing results: 1995 and 1999. In the late summer of 1995, 14- and 30-day sediment toxicity tests measuring survival, growth, and reproduction of *Hyalella azteca* and *Chironomus riparius* were performed on sediments collected from the southern portion of Round Lake, near the lake outlet. The results of the 1995 tests have been previously reported and reviewed by the regulatory agencies (USACHPPM 1998b). In the fall of 1999, 14- and 30-day sediment toxicity tests measuring survival, growth, and reproduction of *Chironomus riparius* were performed on sediments collected from the northern portion of Round Lake and throughout Sunfish Lake. Unfortunately, the 1999 tests using an additional species, *Hyalella azteca*, were found to be invalid and were rejected for risk assessment purposes. The results of the 1999 *Chironomus riparius* tests are reported in this report for the first time (see Appendix L). The potential for survival, growth, and reproductive effects in benthic organisms is inferred by comparing the endpoint results in site sediments to those in the reference sediments. Qualitative risk estimates for each test are based on these comparisons.

8.1.3 Estimation Procedures for Amphibians

Field observational studies and site-specific toxicity tests were used to assess potential ecological effects for amphibians (see Appendix M). The amphibian study was conducted to test whether amphibians exposed to substances from TCAAP sites (Marsden Lake, Sunfish Lake, Round Lake) were adversely affected as a result of military activity. A reference site (Snail Lake) was also evaluated. The measures of effect used as lines of evidence were as follows.

- Mortality and development of frogs exposed to field-collected surface water
- Measures of immune function in frogs exposed to field-collected surface water
- Amphibian species richness and relative abundance at each site (measured using visual encounter surveys, aural estimates, and trapping)

The study attempted to determine (1) if there was a difference in amphibian species richness between TCAAP study sites and the reference sites, (2) if there was a difference in amphibian species relative abundance between the reference sites and the study sites, (3) if mortality and developmental toxicity of offspring differ between reference sites and the study sites, and (4) if surface water from the study sites adversely affects immune function. Species richness and relative abundance were measured using visual encounter surveys (VES), aural estimates, and trapping. Mortality and toxicity were measured using the Frog Embryo Teratogenesis Assay-Xenopus (FETAX). Immune function was determined by examining phagocyte response. The temporal pattern for the (FETAX) test was 96 hours. The phagocyte response tests (Radical Oxygen Intermediate Production and Phagocytosis of fluorescent microspheres), and histopathological evaluations were conducted with frogs exposed to TCAAP surface water for 14 days. This immunological characterization was conducted 114 days after the last reproductive event of the Xenopus used in the FETAX test. Biological surveys took place on 5-8 May 1998, 21-23 July 1998, and 19-20 April 1999.

8.1.4 Estimation Procedures for Waterfowl, Wading Birds, Muskrat, Mink, and Red Foxes

The assessment endpoint (reproductive potential and productivity of aquatic mammals, waterfowl, and wading birds) was not measured directly. 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 for most COCs and to a set of health effects levels for PCBs. A hazard quotient (HQ) is the ratio between the exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. We used single-point estimates of exposure and effect that highlight the variability in the collected site data (a blend of techniques 3 and 4). Exposure was estimated for an individual animal using a potential daily dose algorithm to predict estimates of doses averaged over a given season (see Appendix N). 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 Appendix O). Risk estimates consisting of a hazard ratio matrix, similar to that shown in Exhibit 8-1 is provided for each receptor-COC combination. In the matrix, each value (indicated by the letters A – D in the exhibit) is a hazard quotient, where the calculated exposure dose is based on either the maximum or central tendency detected site concentration 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, and HQ 2 values are based on the low-effect TRV. Values less than one indicate that the calculated exposure is less than a selected level of concern. In cases where a TRV is not available an ND is presented, for "no data." This indicates that an appropriate toxicity value could not be identified and the significance of the exposure for an individual animal is not known. Hazard values were rounded to the nearest whole number when they were greater than or equal to 1, and were reported to the first no zero digit when they were less than 1, because these values are estimates and not exact values due to the uncertainty associated with them.

Exhibit 8-1. Hazard Ratio Matrix Template

Chemical X:	**Receptor**	
	HQ 1	HQ 2
Maximum	A	B
Central Tendency	C	D

The rank of the HQ values A – D should be as follows:

- A > B
- A > C
- C > D
- A >> D

As more of these four hazard quotients exceed unity, the greater is the likelihood that excessive risk may exist for the exposed population. An exceedance of unity by hazard quotient D requires further management consideration than an exceedance by hazard quotient A alone.

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 risk 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 site population will experience a daily dose greater than the lowest level associated with observable health effects in a laboratory population.

8.2 RISK DESCRIPTION PROCEDURE

The risk description is used to interpret and discuss the available information about risks to the assessment endpoints, including an evaluation of the lines of evidence used to derive the risk estimates and an interpretation of the significance of the adverse effects on the assessment endpoints (USEPA 1998). For each hypothesis at each site, risk descriptions are provided to give an overall synthesis of the risk to the receptor from all lines of evidence, and discuss uncertainties. These descriptions also include assessments of the degree to which site concentrations of COCs are greater than background conditions and of the probable spatial distribution of the COCs based on the field investigations.

As identified in Section 8.1, there are several types of risk estimates, depending on the available data used for each line of evidence. How each type of risk estimate is generally interpreted is discussed below.

1. Hazard quotients are used for fish, aquatic invertebrates, algae, benthic organisms, and wildlife. An HQ ratio indicates whether or not an estimated exposure is greater than the selected toxicological criterion. At best, HQs only represent an indicator of concern and should not be confused as actual risk estimates, since they are not a forecast of the probability (or frequency) of an event. However, they are commonly used in risk assessments, since this approach provides an efficient, inexpensive means of confidently identifying low risk situations. Their conservative nature allows risk management decisions of no further action to be made without the need for additional information or analysis (USEPA 1998). The HQ method has several limitations, some of which have been identified in regulatory guidance (USEPA 1998). A calculated HQ or HI that exceeds a target level of unity (or 1) does not, in and of itself, indicate that the site conditions are not "safe" or that it presents an unacceptable risk. A ratio that exceeds the target level triggers further careful consideration of the underlying scientific basis of the estimate.

2. For fish, aquatic invertebrates, algae, and benthic organisms the portion of sample locations greater than benchmarks are used in combination with the HQ. These measures provide information about the frequency of benchmark exceedance and the spatial distribution of elevated concentrations. These measures in combination with sampling maps can be used to infer the extent of risk across a site.

3. Risk estimates for amphibians and some estimates for benthic organisms are based in part on the results of toxicity tests. Toxicity tests using field-collected water or sediments and laboratory-cultured organisms can provide strong evidence for or against the presence of ecological effects at a site. In many cases, such tests also directly measure the health endpoints contained within the assessment endpoint. However, the conditions and species tested in the laboratories may not be fully representative of the conditions and species that occur at the sites. However, we assume that the effect levels observed in lab tests would produce similar effects in the field at specific contaminant concentrations. Nonetheless, when we extrapolate laboratory data to the field, we do not directly consider environmental and ecological processes that may render the organisms more or less sensitive to the chemicals in the field.

4. In addition to toxicity tests, risk estimates for amphibians and some for benthic organisms are based in part on the results of supporting investigations involving species diversity measures, metal bioavailability

measures, and/or laboratory-based assays of health markers in toxicity test organisms. Though these data are considered secondary to the other measures mentioned above, they can provide corroborating evidence for the presence or absence of risk. Such supporting evidence can reduce uncertain in some cases.

Additionally, to further describe the presence of risk to the receptors we use four classifications to explain the risk. These classifications are discussed below.

The classification of "*Not Present*" indicates that no COCs were selected based on the COC selection process in Section 5.3. The two major criteria used to determine COC selection were comparisons of site media concentrations to toxicity benchmarks and comparisons to background. Therefore, if there were no COCs, there would be no expectation that risk should occur.

"*Possible Increase*" is used to describe risk only in the future Round Lake scenario and is determined by considering the current condition HQs and information about what may occur to chemical concentrations in the media if site conditions are manipulated. Because the manipulations have not occurred we can only surmise what may happen.

Below are the remaining classifications of the presence of risk that are intended to represent a scale of increasing confidence in the expectation of the occurrence of adverse effects:

- "Not Apparent"
- "Possible"
- "Potential"
- "Confirmed"

The classifications for risk presence will be based on evaluation of two ranges of HQs; those based on the maximum exposure concentration term and those based on the central tendency exposure concentration term. The magnitude and uncertainty attributes are considered when assigning the classification for risk presence. An explanation of the remaining risk presence classifications is provided below.

"*Not Apparent*" indicates that the ranges of calculated HQs (maximum and central tendency NOAELs and LOAELs) are all less than one. However, for some COCs HQs could not be calculated because toxicity data was not available. Because of these data gaps it is not appropriate to conclude that overall risk is not present for the receptor population. Therefore, considering this information and knowledge of the site and receptors, it is our interpretation that adverse effects do not appear to be occurring. This interpretation is supported by the levels of uncertainty and magnitude of the evidence for the receptor.

"*Possible*" is applied when NOAEL based HQs (maximum and central tendency) exceeded one, but no LOAEL based HQs exceeds one. Magnitude of the NOAEL exceedance is also considered. If the difference between the resulting HQ and criteria value of one is less than an order of magnitude, the occurrence of risk is considered possible (albeit not likely). Also, since there are some COCs with data gaps for toxicity values it can not be concluded that the classification of presence of risk is *Not Apparent*.

"*Potential*" is applied in those cases where the LOAEL HQs (either the maximum or central tendency based) exceeds one. This classification is used to indicate that there is a greater degree of likelihood that adverse effects could be observed at the population/community level. "Potential" is mostly used to describe risks to community receptors (i.e., benthic invertebrates and fish, invertebrates and algae) in which spatial aspects of exceedance of point estimates are used to determine whether the exceedance is widespread or localized. A classification of potential is also used to designate risk when the magnitude of the HQs (both NOAEL and LOAEL based) is greater than an order of magnitude above 1.

"Confirmed" is used when lines of evidence other than the HQ calculation demonstrate adverse effects at current COC concentrations. Results from sediment toxicity tests using site specific media are the primary line of evidence for this determination.

8.3 ECOLOGICAL RISKS AT MARSDEN LAKE (NORTH)

In this assessment, Marsden Lake (North) refers to that portion of Marsden Lake immediately surrounding Patrol Road (see Figure B1-3). The north area of the lake was defined as a distinct area to focus the assessment in terms of characterizing any potential risk due to contaminants that may have been introduced as a result of former activities at the Grenade Range. The selected surface water COCs evaluated in the risk characterization of Marsden Lake (North) are aluminum, barium, lead, manganese, mercury, and zinc. The selected sediment COCs are antimony, arsenic, vanadium, and SVOCs. These COCs were selected based on the process presented in Section 5. These COCs (except for SVOCs) were found at concentrations greater than background and toxicological benchmark concentrations. SVOCs were not detected; however, the target detection limits were not achieved in the analysis of sediment samples. This limitation prevents SVOCs from being eliminated as sediment COCs.

Marsden Lake (North) COCs

Surface Water
Aluminum
Barium
Lead
Manganese
Mercury
Zinc

Sediment
Antimony
Arsenic
Vanadium
SVOCs*

During the development of the risk assessment Work Plan in 1998 (USACHPPM 1999), it was suspected that the Grenade Range could be a specific source of COPCs detected in the northern portion of the lake. Contaminants in Grenade Range soils may have been transported via runoff into the lake, resulting in metals contamination in the surface water and sediment. Section 6.1.1 provides further detail on the possible sources of COCs to Marsden Lake (North). A recent report (Alliant Techsystems 2001a) has established that historical Range activities did cause lead, antimony, and cadmium contamination in the Range soils. Even though SVOCs remain potential COCs, they are not strongly suspected to be true contaminants associated with the Grenade Range (Alliant Techsystems 2001a). Known soil contaminants at the Grenade Range have recently been remediated to an industrial land use standard (Alliant Techsystems 2001a).

A limitation in the characterization of the spatial aspect of potential contamination at Marsden Lake was identified during site sampling and during evaluation of the analytical data. Two of the ten sampling locations in Marsden Lake (North) (ML-13 and ML-15) were collected outside of the original sampling area proposed in the Work Plan, because moving cattail mats prevented access during the field investigation. Chemical analysis results for the two sampling locations (ML-13 and ML-15) were considered to be more similar in concentration and more consistent with the hypothesized fate and transport mechanisms to that for Marsden Lake (South). Therefore, in the risk assessment these locations were removed from the Marsden Lake (North) dataset and combined with the Marsden Lake (South) dataset. See Sections 4.1.2.2 and 6.1.2 for further detail on the rationale used to evaluate this sampling limitation. Spatially, the 8 sampling locations for sediment appear to show slightly different spatial distributions than for surface water. The spatial distribution of sediment COCs showed little variability between sampling locations, suggesting that sediment COCs may be homogenously distributed around the impact areas. However, the spatial distribution of the surface water COCs indicated that the area north of

Patrol Road just offshore from the two historical impact sites might be where the majority of surface water COCs are found.

A revised conceptual site model was developed for Marsden Lake (North) to show the specific COCs being evaluated and to indicate fate and transport pathways (See Figure B3-1).

8.3.1 Hypothesis 1: Effects in Fish, Aquatic Invertebrates, and Algae

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint — Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species
>
> Risk Hypothesis — Based upon the aquatic toxicity of surface water COCs (aluminum, barium, lead, manganese, mercury, and zinc); if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

8.3.1.1 Risk Estimates

Risks for these ecological entities were estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Exhibit 8-2 presents the quantitative risk estimates, which are then described below.

For their entire life cycle, these species are continuously exposed to aluminum, barium, lead, manganese mercury, and zinc; however, concentrations are expected to fluctuate by season and as a result of other changes in the microhabitat (e.g., changes in the amount suspended solids over time). The portion of the community and its populations at Marsden Lake (North) that are actually exposed to levels greater than effects benchmarks is not known precisely, due to limited resources available for sampling the entire Marsden Lake (North) area.

Exhibit 8-2. Marsden Lake (North) Risk Estimates for Fish, Aquatic Invertebrates, and Algae

COC	Hazard Quotient*	Portion Greater Than Chronic Benchmark†	Portion Greater Than Acute Benchmark†
Aluminum	2	50 %	0 %
Barium	30	100 %	38 %
Lead	1	25 %	0 %
Manganese	5	75 %	0 %
Mercury	< 1	0 %	0 %
Zinc	< 1	0 %	0 %

* HQ is calculated using central tendency concentration from Table A22 and chronic water quality benchmark from Appendix O and rounded to 1 significant figure.

† Percentages based on 8 sample locations.

Aluminum: Based on the risk estimate provided in Exhibit 8-2, adverse toxic effects in fish, aquatic invertebrates, and algal species due to aluminum may be occurring because the HQ is greater than 1 and 50% of the measured aluminum water concentrations at the site are greater than chronic water quality benchmark (125 µg/L). This water quality benchmark is a Minnesota Water Quality Standard. However, none of the site concentrations are greater than the acute benchmark (1072 µg/L), indicating that more serious effects are not expected (see Figure B4-1).

Unfiltered water concentrations of aluminum in the surface water at Marsden Lake (North) ranged from 6 to 540 µg/L. Because only unfiltered water column samples were collected, the aluminum detected in the analysis was a combination of the dissolved fraction in the water column and the fraction attached to sediment particles. High TSS in water is known to result in higher concentrations of total aluminum in the water column. Most of the samples indicated high TSS at the site. All metals tend to attach to suspended particles and aluminum will tend to remain attached to such particles when the water pH is greater than 5.5. Aluminum is less biologically available when it is bound to such particles. The average pH in Marsden Lake (North) was 6.8, indicating that much of the total aluminum detected is likely to be particle-bound. Even at neutral pH levels, higher total aluminum concentrations have been found in lakes with a high humic acid content (ATSDR 1999). However, humic acids are not toxic and can complex with aluminum, rendering it not bioavailable (Horne 1994). If the majority of aluminum in the surface water is attached to suspended solids and is not bioavailable, it should not cause adverse effects in biota inhabiting the surface waters of Marsden Lake (North).

Additionally, the TCAAP soils are derived from glacial formations, with two of the major formations at TCAAP being the Hillside Sands and the Arsenal Sands (Montgomery Watson 1997). These sandy soils may contain larger concentrations of aluminum because the sand fraction derived from glacial erosion contains much aluminum (Allen 1995). Analytical results showed that sediment concentrations of aluminum were high in Marsden Lake (North) and Marsden Lake (South), but were also high in the reference site, with neither the aluminum concentration in Marsden Lake (North) or (South) being statistically different than the concentration at the reference site (Snail Lake). Therefore, natural conditions at TCAAP can cause total aluminum levels to be increased in the surface water, based on suspension of solids that would complex aluminum (because unfiltered water column samples were used).

In light of the uncertainty associated with the form of aluminum present at the lake, it may be premature to conclude that aluminum in Marsden Lake (North) surface waters poses a significant risk to fish, aquatic invertebrates, and algal species. The Minnesota Water Quality Standards are based on total recoverable aluminum in the water column. The USEPA (1999) has noted that when particulate aluminum is primarily aluminum hydroxide particles, then a benchmark based on total recoverable aluminum may be more appropriate than one based on the dissolved fraction. In natural surface waters, the procedure that measures total recoverable aluminum might measure aluminum associated with clay particles, which might be less toxic than aluminum associated with aluminum hydroxide (USEPA 1999).

Barium: Based on the risk estimate provided in Exhibit 8-2, adverse toxic effects in fish, aquatic invertebrates, and algal species due to barium may be occurring because the HQ is greater than 1 and all of the measured barium water concentrations at the site are greater than the chronic water quality benchmark (3.9 µg/L) (see Figure B4-2). Three of the eight sample locations (38%) even exceed the acute benchmark (110 µg/L), indicating the possibility for more serious effects in some areas. This risk estimate contains a relatively high degree of uncertainty due to the very limited toxicity database used to generate the water concentration benchmarks. Only one study was found regarding aquatic toxicity of barium. The benchmarks are based on Suter and Tsao's interpretation (1996) of a 21-day *Daphnia magna* test that found a 16 % reproductive impairment in the tested animals.

Lead: Based on the risk estimate provided in Exhibit 8-2, adverse toxic effects in fish, aquatic invertebrates, and algal species due to lead may not be occurring. The HQ is 1 and 25% of the samples had lead detections greater than the chronic water quality benchmark based on a hardness of 100 mg/L

(3.2 μg/L). This chronic water quality benchmark is a MWQC that is hardness dependent. No lead concentrations were greater than the acute water quality benchmark (82 μg/L) that is also hardness dependent, indicating that more serious effects are not expected. (See Figure B4-3).

The two sample locations with lead concentrations exceeding the chronic benchmark had hardness values of 244 mg/L (ML-17) and 340 mg/L (ML-20). If a hardness value of 200 mg/L is used to calculate the benchmark, then the resulting benchmark would be 7.7 μg/L (MPCA 2000b), and neither location (ML-17 = 7.0 μg/L or ML-20 = 5.4 μg/L) would exceed the benchmark value. The resultant HQ would be less than one and 0% of the concentrations would exceed the chronic and acute benchmarks.

Manganese: Based on the risk estimate provided in Exhibit 8-2 adverse toxic effects in fish, aquatic invertebrates, and algal species due to manganese may be occurring because six of eight (75%) measured manganese water concentrations at the site are greater than the chronic water quality benchmark (120 μg/L) developed in Section 7. However, none of the site concentrations are greater than the acute benchmark (2300 μg/L), indicating that more serious effects are not expected (see Figure B4-4).

Only two samples out of eight had concentrations greater than or about equal to the next highest alternative water benchmark (<1100 μg/L) from Suter and Tsao (1996), the original source for the initial benchmark of 120 μg/L. The lowest chronic value for fish (1780 μg/L) from the same source is not exceeded, indicating that fish may not be affected.

Mercury: Based on the risk estimate provided in Exhibit 8-2, adverse toxic effects in fish, aquatic invertebrates, and algal species due to mercury are not expected because the HQ is less than one and no sample had a concentration greater than the chronic water quality benchmark. The maximum mercury water concentration at the site (0.013 μg/L) is much less than the chronic water quality benchmark (0.77 μg/L) developed in Section 7 (see Figure B4-5), and all samples are below the acute water quality benchmark (2.4 μg/L).

Zinc: Based on the risk estimate provided in Exhibit 8-2, adverse toxic effects in fish, aquatic invertebrates, and algal species due to zinc are not expected to occur because the HQ is less than one and no sample concentrations exceeded the chronic water quality benchmark (106 μg/L) (see Figure B4-6). Additionally, no sample concentrations exceeded the acute water quality benchmark (117 μg/L).

8.3.1.2 Risk Description

Based on the line of evidence used to estimate risk to fish, aquatic invertebrates, and algae at Marsden Lake (North), it appears that some adverse toxicological effects on the survival, growth, and reproduction of the receptors are possible due to aluminum, barium, and manganese. However, there remains uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. Based on the uncertainty associated with the form of aluminum present at the lake, it may be premature to conclude that aluminum poses a significant risk. The barium risk estimate contains a high degree of uncertainty due to the lack of a sufficient toxicity database; only one study was found regarding aquatic toxicity of barium. While the manganese chronic water quality benchmark was exceeded in most of the samples, the fish-specific benchmark was not exceeded. Additionally, the suspected source area (the Grenade Range soils) has been remediated to industrial scenario standards based on the protection of human health.

The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. When available for a COC, the Minnesota Water Quality Standards for Class 2B waters were used, which are based on the protection of several aquatic species from adverse toxic effects. Other effects-based benchmarks for fish, invertebrates, and algal species were then used. Water quality criteria used in this assessment are linked to the assessment endpoint because survival, growth, and development of at least 95 percent of a population of aquatic organisms should be protected by a water quality standard (MPCA 2000a and USEPA 1999). However, by using the benchmarks we extrapolate from the laboratory to the field and between different species (i.e., between taxa). These extrapolations introduce uncertainty. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more, or less sensitive to the chemicals. In addition, the species tested in the laboratories may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar. 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).

A comparison of the surface water COC concentrations found at Marsden Lake (North) to those found at the reference site (Snail Lake) indicated statistically higher concentrations of aluminum, barium, and manganese at Marsden Lake (North). This difference is likely partially attributable to the total suspended solids (TSS) concentration in the water. Surface water samples in this investigation were unfiltered, with Marsden Lake (North) having a much higher central tendency TSS concentration (84 mg/L) than the reference lake (1 mg/L) (see Table A21). As explained under the risk estimate for aluminum above, metals tend to bind with particles in the water. Therefore, with unfiltered water column samples the concentration of metals detected is directly related to the concentration of TSS, and does not give an indication of the amount of metal that is bioavailable (freely dissolved) to cause adverse health effects in aquatic organisms. This same TSS issue of metals binding with particles also applies to barium. However, even though the barium water concentration in Snail Lake was less than the concentration in Marsden Lake (North), the Snail Lake concentration was 10.8 times greater than the chronic water quality benchmark. This may indicate that barium concentrations are naturally high in the area and are not likely to cause adverse effects to aquatic organisms.

The spatial distribution of COC concentrations at Marsden Lake (North) can be visualized by comparing Figure D1 (showing sampling locations) to Figures B4-1 through B4-6 (bar graphs indicating the sample locations and the surface water benchmarks for each COC). The majority of surface water samples that exceed water quality benchmarks were from locations ML-16, ML-17, ML-19, and ML-20. These locations also had the highest TSS concentrations of all the Marsden Lake (North) samples. High TSS in water is known to result in higher concentrations of metals in unfiltered water samples because metals tend to attach to suspended particles and will be detected during analysis. The elevated concentrations of metals detected in surface water samples from these locations may be an artifact of disturbance caused by cutting the cattail mats to access the surface water below (see Appendix K, Section K1.1.4 and Appendix C photo 5). Additionally, sample locations ML-16, ML-17, ML-19, and ML-20 are located on the North side of Patrol Road and are on the periphery of two of the impact areas. For some COCs (aluminum, barium, and manganese), location ML-12 also exceeded the water quality benchmarks. Location ML-12 is located on the western periphery of the smaller, most southern impact area. This may be an indication that surface runoff of contaminated soils entering the Lake from the former Grenade Range may have been transported slightly northward, in the direction of water flow.

8.3.2 Hypothesis 2: Effects in Benthic Organisms

The following assessment endpoint and risk hypothesis were evaluated.

Assessment Endpoint – Survival, growth, and reproduction of benthic organisms

Risk Hypothesis - Based upon the toxicity of sediment COCs (antimony, arsenic, vanadium, and SVOCs); if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

8.3.2.1 Risk Estimate

The potential for adverse toxicological effects in these ecological entities were estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.2). Exhibit 8-3 presents the quantitative risk estimates, which are then described below.

For their entire life cycle, these species are continuously exposed to antimony, arsenic, vanadium, and SVOCs; however, concentrations of the metals are expected to fluctuate by season and as a result of other changes in the microhabitat (e.g., changes in SEM:AVS, organic matter content, and dissolved oxygen). The portion of the community and its population at Marsden Lake (North) that are actually exposed to levels greater than effects benchmarks is not known precisely, due to limited resources available for sampling the entire northern portion of the lake.

Exhibit 8-3. Marsden Lake (North) Risk Estimate for Benthic Organisms

COC	Hazard Quotient*	Portion Greater Than Low-effect Benchmark†	Portion Greater Than Moderate-Effect Benchmark†
Antimony	<1	13 %	13 %
Arsenic	<1	25 %	0 %
Vanadium	unavailable	unavailable	unavailable
SVOCs	unavailable	unavailable	unavailable

* HQ is calculated using central tendency concentrations from Table A24 and Low-effect sediment benchmarks from Appendix O and rounded to 1 significant figure.
† Percentages based on 8 sample locations.
SVOCs: Semivolatile organic compounds

Antimony: Based on the risk estimate provided in Exhibit 8-3, adverse toxic effects to benthic organisms due to antimony may be occurring even though the HQ is less than one, because at one isolated location (ML-17), the sediment concentration is four times the low-effect benchmark and 1.9 times the moderate-effect benchmark (see Figure B4-14). No other samples exceeded the benchmarks, with sample concentrations 2 orders of magnitude lower (0.7 mg/kg) than the concentration at ML-17 (48 mg/kg). The low-effect benchmark is a Practical Quantification Limit (PQL) for freshwater sediments as adopted by the USEPA Region IV as a screening value (see Appendix O).

Arsenic: Based on the risk estimate provided in Exhibit 8-3, adverse toxic effects to benthic organisms due to arsenic may not be occurring. The HQ is less than one, and two sampling locations exceed the low-effect benchmark (7.24 mg/kg), but no locations exceed the moderate-effect benchmark (17 mg/kg) (see Figure B4-15). The two locations that do exceed the benchmark (ML-12 and ML-19) only slightly exceed the value (8.1 and 8.9 mg/kg, respectively). The benchmark is a threshold effect level (TEL) as provided by USEPA region IV (see Appendix O).

Vanadium: No risk estimate could be calculated for vanadium because sediments were not analyzed for vanadium. Additionally, no sediment toxicological benchmark is available for vanadium. Therefore, a risk estimate cannot be provided for this COC.

SVOCs: No risk estimate could be calculated for SVOCs due to a problem of meeting the detection limit during sediment analysis. All results reported nondetects for SVOCs at Marsden Lake (North). Therefore, without a concentration a risk estimate could not be calculated.

8.3.2.2 Risk Description

Based on the line of evidence used to estimate risk to benthic organisms at Marsden Lake (North), it appears that adverse toxicological effects are not expected on the population level. However, one sample location showed an antimony concentration that is 2-times higher than the moderate-effect benchmark. Two other locations showed arsenic concentrations that were slightly higher than the low-effect benchmark. Adverse effects to benthic organisms from arsenic may not be occurring. Based on a lack of data, the possibility of effects from vanadium and SVOCs cannot be ruled out. However, there remains uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. Additionally, the suspected source area has been remediated to industrial scenario standards based on the protection of human health.

The assessment endpoint (survival, growth, and reproduction in benthic organisms) was not measured directly for this line of evidence. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site sediment concentration data to toxicity benchmarks. There are no state or federally recognized sediment benchmarks promulgated for use. Sediment quality benchmarks can be reasonably linked to this endpoint because survival, growth, and development are often the endpoints used to derive these benchmarks. Many sediment benchmarks have been derived from chemical concentrations detected in whole sediments that are associated with adverse effects to lab tested and/or *in situ* tested benthic organisms. Some benchmarks, however, are based on presence/absence of a particular species when comparing field sediments across multiple locations. Such benchmarks are less relevant to the assessment endpoint.

By using the benchmarks we extrapolate from the laboratory to the field, between different field sites, and between taxa. These extrapolations introduce uncertainty. The species tested in the laboratories or observed in the field-collected sediments of reference areas may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more or less sensitive to the chemicals. When we use benchmarks based on reference sediments, then we assume that environmental and ecological processes between the reference sites and the study site do not differ in significant ways.

The spatial distribution of sediment COC concentrations at Marsden Lake (North) can be visualized by comparing Figure D1 (showing sediment sampling locations) to Figures B4-14 and B4-15 (bar graphs indicating the sample locations and the sediment benchmarks for COCs with available data). No apparent spatial pattern exists for the sediment COCs. However, the antimony concentration that exceeded the benchmark was located on the periphery of the western most impact area (ML-17), and the two arsenic concentrations that exceed the benchmark were located along the eastern shoreline (ML-12 and ML-19), both on the northwestern edge of an impact area. Activities at the Grenade Range are known to have caused antimony contamination in the range soils so it is likely that runoff from the western most impact area introduced antimony to the sediment at ML-17. Arsenic was not included as a COC in the grenade range soils.

8.3.3 Hypothesis 3: Effects in Amphibians

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, development, and reproduction of amphibians
>
> Risk Hypothesis –Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment. (An adequate evaluation of potential problems for the amphibian community of the lake was a major data gap in the Tier I risk assessment).

8.3.3.1 Risk Estimates

The potential for adverse toxicological effects in amphibians was inferred by evaluating three lines of evidence; Frog Embryo Teratogenesis Assay-Xenopus (FETAX), species richness and relative abundance estimates, and immune function. Risk estimates based on each of these lines are provided in this subsection. A synthesis is provided in the Risk Description subsection.

It is expected that these organisms will be exposed during the embryo and juvenile portions of their life cycle. Additionally, exposure for adults is expected to be on an intermittent basis. However, sufficient information is not available to determine the size of the population exposed to the COCs.

FETAX RESULTS

Risk estimates for the toxicity test line of evidence, using surface water from Marsden Lake for the FETAX, were developed based on technique 1. Exhibit 8-4 presents the quantitative risk estimates for this line of evidence, which are then described below.

Exhibit 8-4. Marsden Lake (North) Risk Estimates for Amphibians based on results of the 1999 FETAX Tests[†]

Site	Petri Dish ID	Initial Number of Embryos	Mortality Count	Mortality Rate (%)	Number Malformed	Malformation Rate (%)
Marsden Lake	3	25	6	24	6	35
Marsden Lake	4	25	4	16	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
- 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.

Based on the risk estimate provided in Exhibit 8-4, amphibians in Marsden Lake appear to have acceptable occurrences of mortality and malformations when compared to the controls and the reference site. The report, containing supporting information on this conclusion, is found in Appendix M. The types and incidences of malformations observed in Marsden Lake samples were not statistically different from controls or the reference site. Also, mortality was not found to be statistically different in Marsden Lake samples than in the reference samples. However, there was a slightly higher incidence of mortality in Marsden Lake compared to the controls. These negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected, and may be unacceptable for decision-making in some cases. The FETAX test was conducted on a composite surface water sample consisting of six samples from around the perimeter of all of Marsden Lake, in order to obtain an overall characterization of the site (Marsden Lake as a whole, not (North) and (South)). Therefore, effect(s) could not be linked to a potential "hot spot" or suspected source area in the lake. Additionally, surface water chemical analysis for metals was not conducted at the same time. FETAX was not conducted with sediments from Marsden Lake due to the inherent difficulties in dealing with sediment in amphibian assays (i.e. with dejellied embryos in FETAX, fouling the water in adult exposure assays). The possible effects of sediment were considered such that water-soluble constituents would be expected to partition to the water and then to the organism. Therefore, exposure (dermal) to sediment COCs was assumed to be at equilibrium between water and sediment (pore water).

IMMUNE FUNCTION, ORGAN WEIGHTS, AND HISTOPATHOLOGY

For this line of evidence qualitative risk estimates are provided based on data collected on organ weights, two immune function markers, and histopathologic evaluations of important blood conditioning organs. Comparisons were made in adult frogs (*X. laevis*) exposed 14-days to surface water from Snail and Marsden Lakes and also from a laboratory control solution used in the FETAX assay. The two markers, based on phagocyte response tests, are sensitive indicators of adverse immune responses. Organ weights can be useful indicators of toxicity. Appendix M presents the full results.

- The production of radical oxygen intermediates (ROI) in adult frogs exposed to Marsden Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- The phagocytosis of foreign particles by splenic macrophages in adult frogs exposed to Marsden Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- Ratios of organ weight to body weight in adult frogs exposed to Marsden Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- No notable histological differences were found between sections of thymus, lung, kidney, and liver when examined using light microscopy for male and female frogs exposed to Marsden Lake surface water, compared to Snail Lake and the laboratory control solution.

SPECIES RICHNESS AND RELATIVE ABUNDANCE

Risk estimates for this line of evidence were developed using visual encounter surveys, aural estimates, and trapping. Exhibit 8-5 presents the qualitative risk estimates for this line of evidence.

Exhibit 8-5. Marsden Lake Risk Estimate for Amphibians Based on Species Richness

Site	Number of Species Seen or Heard in April '99, May '98, and July '98	Species Observed
Marsden Lake	3	*Bufo americanus, Rana pipiens, Pseudacris triseriata*
Snail Lake	6	*Bufo americanus, Rana clamitans, Rana pipiens, Ranid species, Pseudacris triseriata, Hyla chrysoscelis*
Risk Estimate	• No statistically significant differences were found between the reference site and Marsden Lake for the number of species observed. • It was not possible to accurately report relative abundance due to difficulties in accessing the sites because of the dense cattail mats. Species observed may have been listed because they were heard but not seen, so a definitive count was not possible.	

Based on the risk estimate provided in Exhibit 8-5, the diversity of amphibians in Marsden Lake appears to be no different than that in the reference lake (Snail Lake). Due to habitat constraints and the relatively quick Anuran reaction reflex, many frogs detected by sight during the VES surveys could not be reliably identified. Therefore, more species may exist at each site than was originally reported. The study concluded that overall, relatively few amphibians were found.

8.3.3.2 Risk Description

The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species for FETAX, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. The potential for adverse toxicological effects in amphibians at Marsden Lake is inferred using three lines of evidence. Risk estimates based on each of these lines were provided in the previous subsection. However, it is noted that there are uncertainties associated with the amphibian study including but not limited to; no multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, laboratory analysis for metals concentrations in surface water was not conducted, and laboratory tests used species not found at the site. These uncertainties along with others can be found in Table 6.1 in the amphibian report (see Appendix M). Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations, and site surface water was not analyzed for COC concentrations at the time of the amphibian study.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of the assessment.

Laboratory data investigating embryo mortality (a sensitive life stage), developmental effects, and sensitive indicators of stress (immunocharacterization assays) suggest that surface water from Marsden Lake is not toxic to frogs, yet sediment exposures were not specifically evaluated. However, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 63%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation (see Tables 5 and 6 in Appendix M). The field data were ambiguous, yet amphibians were noted at all locations.

Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to metals from the Outdoor Firing Range or the Grenade Range is not apparent. Additionally, data collected by Jannett (1997) and others (as discussed in Appendix M) appear to corroborate a conclusion that adverse impacts of chemical exposure to amphibians are unlikely.

8.3.4 Hypothesis 4: Effects in Waterfowl and Wading Birds

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Reproductive potential and productivity of waterfowl and wading birds
>
> Risk Hypothesis – The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COCs in surface water, sediment, and residues in food from the lake.

8.3.4.1 Risk Estimates

The potential for adverse toxicological effects in waterfowl and wading birds was estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Risk estimates are provided in this subsection. A synthesis is provided in the Risk Description subsection.

WATERFOWL

The frequency of exposure for waterfowl at Marsden Lake (North) is expected to be intermittent because the birds are not expected to spend much time feeding at the Lake. The ingestion pathway is of most concern for this receptor but its land use habits for the area indicate that it would utilize the site mostly for nesting and not for feeding. If the waterfowl are only nesting at the site an ingestion exposure pathway is not likely to be complete. For those birds that were to be exposed, exposure is expected to be intermittent. The portion of the local population that actually could be exposed during each season is expected to be very small. It is likely to number less than ten, based on professional judgment. Only the mallard was modeled. This species was chosen as the surrogate for all waterfowl during the development of the Work Plan (USACHPPM 1998), mainly based on its presence at the site and the availability of data for modeling purposes. Modeling only this species of waterfowl does present some uncertainty. This uncertainty is a result of different site utilization preferences among duck species and slight differences in dietary preferences among the species. The differences may result in underestimation or overestimation of the potential for risk. However, despite the differences and the potential for an underestimation of potential for risk by not modeling the maximally exposed species, modeling the mallard is an acceptable approach due to the lack of necessary input values for the other species, and the overestimation of potential for risk resulting from use of other conservative assumptions.

Exhibit 8-6. Avian Hazard Ratio Matrix for Marsden Lake (North)†

Mallard

	HQ1	HQ2
Aluminum:		
Maximum	3	nd
Central Tendency	2	nd
Antimony:		
Maximum	ND	ND
Central Tendency	ND	ND
Arsenic:		
Maximum	0.1	0.0
Central Tendency	0.0	0.0
Barium:		
Maximum	0.2	0.2
Central Tendency	0.1	0.1
Lead:		
Maximum	4	0.5
Central Tendency	2	0.3
Manganese:		
Maximum	0.006	ND
Central Tendency	0.003	ND
Mercury:		
Maximum	0.05	0.009
Central Tendency	0.04	0.007

Vanadium:		
Maximum	0.1	ND
Central Tendency	0.1	ND
Zinc:		
Maximum	0.3	0.03
Central Tendency	0.2	0.02
SVOCs		
Maximum	ND	ND
Central Tendency	ND	ND

ND: Not determined, one or more parameters were not available to calculate the HQ.

Arsenic, Barium, Manganese, Mercury, Vanadium, and Zinc: None of the calculated hazard quotients for these COCs are greater than unity (see Exhibit 8-6). This indicates that adverse health effects due to these COCs are not expected for waterfowl utilizing Marsden Lake (North).

Aluminum: Two HQs exceeded unity based on modeling for the mallard. The HQ of 3 (based on the maximum site concentrations and an estimate of the no-adverse-effect level) indicates that a maximally exposed bird in the population of waterfowl may experience an exposure 3-times greater than that considered to be safe. The HQ of 2 (based on the central tendency concentration and an estimate of the no-adverse-effect level) indicates that on average, individual waterfowl in the population might experience exposures 2-times greater than that considered to be safe. HQ ratios based on the lowest-observed-adverse-effect-level could not be calculated because a LOAEL based TRV was not available for avians.

Lead: Two HQs exceeded unity based on modeling for the mallard. The HQ of 4 (based on the maximum site concentrations and an estimate of the no-adverse-effect level) indicates that a maximally exposed bird in the population of waterfowl may experience an exposure 3-times greater than that considered to be safe. The HQ of 2 (based on the central tendency concentration and an estimate of the no-adverse-effect level) indicates that on average, individual waterfowl in the population might experience exposures 2-times greater than that considered to be safe. Because the HQ ratios based on the lowest-observed-adverse-effect-level are not greater than unity (0.5 and 0.3), estimated exposures in the waterfowl population are less than exposures known to be associated with adverse health effects.

SVOCs: Hazard quotients for SVOCs could not be calculated due to detection limit problems when analyzing the sediment data. Uncertainty remains as to the presence of SVOCs in the Marsden Lake sediments, and thus whether or not there would be an adverse effect on waterfowl.

WADING BIRDS

Herons have been reported in the area from March through November but most leave the area for the winter (USGS 1995). They do not nest at the site, however bitterns and other wading birds may (see Appendix N, Section N2.3). It is expected that herons and egrets will forage at Marsden Lake (North) occasionally. However bitterns, unlike the herons and egrets, are expected to spend more time foraging in the dense cattail area on the north side of Patrol Road. The frequency of exposure for wading birds is expected to be moderate. Based on information in the *Wildlife Exposure Factors Handbook* (USEPA 1993) and information from various studies at TCAAP, it can be expected that 1-2 herons may frequent Marsden Lake (North). Adult herons are known to feed solitarily unless large densities of prey are found, and some are known to defend feeding territories, thus limiting the number of herons frequenting the area

(USEPA 1993). Bitterns found at TCAAP also prefer a solitary lifestyle and would be expected to be present in the same density as herons at Marsden Lake (North).

Exhibit 8-7. Wading Hazard Ratio Matrix for Marsden Lake (North)[†]

	Heron (no sediment ingestion)		**Heron** (10% sediment ingestion)	
	HQ1	HQ2	HQ1	HQ2
Aluminum:				
Maximum	0.0	nd	0.0	ND
Central Tendency	0.0	nd	0.0	ND
Antimony:				
Maximum	ND	ND	ND	ND
Central Tendency	ND	ND	ND	ND
Arsenic:				
Maximum	0.0	0.0	0.0	0.0
Central Tendency	0.0	0.0	0.0	0.0
Barium:				
Maximum	0.1	0.04	0.1	0.0
Central Tendency	0.0	0.02	0.0	0.0
Lead:				
Maximum	0.0	0.00	0.1	0.0
Central Tendency	0.0	0.00	0.0	0.0
Manganese:				
Maximum	0.0	ND	0.0	ND
Central Tendency	0.0	ND	0.0	ND
Mercury:				
Maximum	0.1	0.01	0.1	0.0
Central Tendency	0.0	0.004	0.0	0.0
Vanadium:				
Maximum	0.0	ND	0.0	ND
Central Tendency	0.0	ND	0.0	ND
Zinc:				
Maximum	0.04	0.004	0.0	0.0
Central Tendency	0.01	0.001	0.0	0.0
SVOCs				
Maximum	ND	ND	ND	ND
Central Tendency	ND	ND	ND	ND

[†] Exposure to the heron was modeled using zero percent sediment ingestion, and a worst-case scenario ingestion of 10%. See Appendix N Section N3.3

None of the calculated hazard quotients for the COCs for wading birds, using both sediment ingestion scenarios, are greater than unity (see Exhibit 8-7). This indicates that adverse health effects are not expected for wading birds that frequent Marsden Lake (North). No sediment ingestion data for wading birds was found in the literature. However, it is possible that wading birds may be ingesting sediment to some degree. It was therefore decided to produce hypothetical values of sediment ingestion rates across a range of portions of sediment in the diet (1, 5, and 10 percent of the diet), in order to support a sensitivity analysis.

Additionally, there is uncertainty with regard to exposure and effect of SVOCs to wading birds. Due to target detection limits not being reached during the sediment analysis, SVOCs in sediment were not detected. Confident conclusions cannot be made concerning SVOCs.

8.3.4.2 Risk Description

Based on the line of evidence used to estimate risk to waterfowl at Marsden Lake (North), it appears that population-level adverse toxicological effects are possible, albeit unlikely due to aluminum and lead (see Exhibit 8-6). The HQ ratios for lead based on the lowest-observed-adverse-effect-level are not greater than unity, indicating that the estimated maximum and average exposures in the waterfowl population are less than exposures known to be associated with adverse health effects. No comparable toxicity data (i.e., LOAELs) exist for aluminum, but the unknown LOAEL values for aluminum are also expected to be less than unity given the magnitude of the NOAEL-based HQ ratios for aluminum. The possibility of effects from antimony cannot be ruled out because there are no toxicity benchmarks for birds exposed to antimony.

Adverse toxicological effects are not expected in wading birds at Marsden Lake (North): all HQ ratios are less than unity. The possibility of effects from antimony cannot be ruled out because there are no toxicity benchmarks for birds exposed to antimony. Additionally, effects from SVOCs cannot be ruled out because due to a lack of data.

The assessment endpoint for these entities (reproductive potential and productivity of waterfowl and wading birds) was not measured directly. 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—ratios between the exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. A ratio that exceeds unity triggers further careful consideration of the underlying scientific basis of the prediction.

8.3.5 Hypothesis 5: Effects in Mink and Muskrat

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint –Reproductive potential and productivity of aquatic mammals
>
> Risk Hypothesis –The health of aquatic mammals such as mink or muskrat, may be impaired if they are excessively exposed to COCs in the surface waters, sediments, and food residues.

8.3.5.1 Risk Estimates

The potential for adverse toxicological effects in aquatic mammals was estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Since the initiation of the risk assessment, the muskrat has been the primary mammal of concern. However, in addition to evaluating the risk of COC exposure to the muskrat, risk to the mink was also evaluated, but only for exposure to mercury. It is recognized that other metals and possibly organics could be present in dietary items of the mink; however, the use of the muskrat to evaluate risk to aquatic mammals provides an estimator for risk to COC other than mercury. Only mercury was evaluated for the mink because there was concern that mink would be at more risk than the muskrat to biomagnification of mercury mainly through consumption of fish.

MUSKRAT

No Muskrat (*Ondatra zibethicus*) dens were observed in the northern portion of the lake, but a true survey was not conducted. Cattail communities are known to support high densities of muskrats, but in lake habitats shoreline length is also an important factor in muskrat density (USEPA 1993). The water level is known to fluctuate seasonally in the northern section of the lake and may not be good habitat for muskrats, even though the cattails are dense. Low water and drought make muskrats more vulnerable to predators because they need to venture farther from their lodges and in more open space to forage for food. One method of estimating the population of muskrats is to count the number of dens, then multiply by a factor between 2.8 and 5.0 (USEPA 1993), but this is a questionable method. Based on the available information about the type of habitat at Marsden Lake, no observation of dens in the area, and little information about muskrat population densities, no estimate of actual population size can be made. The area is largely covered by cattails but without a formal den count or population study at the lake, there is not enough information available to determine how many muskrats may be present. Some important biological attributes of this species are provided in Appendix N.

Risk estimates are provided in Exhibit 8-8. A synthesis is provided in the Risk Description subsection.

Exhibit 8-8. Aquatic Mammal Hazard Ratio Matrix for Marsden Lake (North)

	Muskrat	
Aluminum as $AlCl_3$:	HQ1	HQ2
Maximum	10	2
Central Tendency	6	1
Aluminum as $Al(OH)_3$		
Maximum	1	NA
Central Tendency	0.9	NA
Antimony:		
Maximum	10	2
Central Tendency	0.1	0
Arsenic:		
Maximum	0.9	0.2
Central Tendency	0.5	0.1
Barium:		
Maximum	0.9	0.3
Central Tendency	0.5	0.2
Lead:		
Maximum	0	0
Central Tendency	0	
Manganese:		
Maximum	0.2	0.1
Central Tendency	0.1	0
Mercury:		
Maximum	0.1	0
Central Tendency	0.1	0
Vanadium:		
Maximum	0.8	0.2
Central Tendency	0.6	0.1
Zinc:		
Maximum	0	0
Central Tendency	0	0

NM: Not modeled. (Mink was only modeled for exposure to mercury)
NA: Not available

Arsenic, Barium, Lead, Manganese, Mercury, Vanadium, and Zinc: None of the hazard quotients for these COCs are greater than unity. This indicates that adverse health effects due to these COCs are not expected for mammals at Marsden Lake (North).

Aluminum: Three of the four HQs based on the muskrat model exceed unity. These HQs were calculated based on an allometrically scaled NOAEL (3.05 mg/kg/day) resulting from dosing laboratory animals with aluminum chloride ($AlCl_3$) (see appendix O). However, there is uncertainty as to what form of aluminum is predominant in the sediments at the lake. Based on site-specific information and professional judgment, it is thought that the form of aluminum found in the waterbodies at TCAAP is not solely aluminum chloride, but rather a combination of aluminum complexes such as aluminum hydroxide ($Al(OH)_3$), aluminum bicarbonate, or complexes with other ligands. This leads us to believe that the current TRV (based on aluminum chloride) may be over conservative. The toxicity of aluminum is intimately associated with pH in that the metal is soluble and biologically available in acidic (pH <5.5) soils and waters but relatively innocuous in pH conditions between 5.5 – 7.5 (Sparling and Lowe 1996). Also, concentrations of dissolved aluminum in water vary not only with pH but also with the humic-derived acid content of the water (ATSDR 1999). The TRV value for wildlife is based on a laboratory study in which $AlCl_3$ was administered to mice. Alternatively, two oral exposure studies of aluminum hydroxide discussed in the Aluminum profile (ATSDR 1999), one to rats and one to mice were considered. The study on mice (Domingo et al 1989) was a chronic study in which mated female mice were given by gavage daily doses of $Al(OH)_3$ on gestation days 6 through 15. The study found no evidence of maternal toxicity, embryo/fetal toxicity or teratogenicity even at levels as high as 266 mg Al/kg/day as aluminum hydroxide. The other study (Greger and Donnaubauer 1986), was an intermediate exposure (subchronic) study in which no changes in body weight were found in male Sprague-Dawley rats that ingested up to 158 mg Al/kg/day as aluminum hydroxide in the diet for 16 days. When developing a TRV, studies of chronic duration are more favorable. Based on the Domingo et al (1989) study, a TRV could be generated following the USACHPPM (2000) protocol (see appendix O). The chronic NOAEL TRV derived from the Domingo et al (1989) study would be 26.6 mg/kg/d aluminum. It is more appropriate to develop a TRV based on the form of the chemical that the organisms will be exposed to and an exposure duration that would be more representative of field conditions. Using the $Al(OH)_3$ based NOAEL TRV of 26.6 mg/kg/d would decrease the HQ values by an order of magnitude at the maximum exposure concentration HQ (10 mg/kg-day for $AlCl_3$ to 1 mg/kg-day for $Al(OH)_3$), resulting in an HQ equal to unity. See Exhibit 8-8. This information may indicate that muskrats are not at risk of adverse effects due to aluminum at Marsden Lake (North).

Antimony: Two of the four HQs based on the muskrat model exceed unity. The HQ of 10 (based on the maximum site concentration and an estimate of the no-observable-adverse-effect-level) indicates that a maximally exposed muskrat in the population may experience an exposure 10-times greater than that considered to be safe. The HQ of 2 (based on the maximum site concentration and an estimate of the lowest-observable-adverse-effect-level) 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. However, the HQs based on the central tendency exposures were both below unity, indicating that a population level effect will most likely not be observed.

MINK

Based on the size of Marsden Lake, the size of the mink's home range, and the availability of sufficient habitat and food, it can be expected that 2-3 adult females and 1-2 adult male mink will be found inhabiting Marsden Lake, as a whole. Signs of mink activity at the lake have been observed. (e.g., tracks in the winter, etc.).

Risk estimates are provided in Exhibit 8-9. A synthesis is provided in the Risk Description subsection.

Exhibit 8-9. Mink Hazard Ratio Matrix for Marsden Lake (North)

Mercury:	**Mink** HQ1	HQ2
Maximum	2	0.4
Central Tendency	0.7	0.1

Mercury: One HQ for the mink model exceeded unity. The HQ of 2 (based on the maximum site concentration and an estimate of the no-observable-adverse-effect-level) indicates that a maximally exposed mink in the population may experience an exposure 2-times greater than that considered to be safe. However, because the other HQ ratios were less than unity (0.7, 0.4, and 0.1) estimated exposures in the mink population are less than exposures known to be associated with adverse health effects.

8.3.5.2 Risk Description

Based on the line of evidence used to estimate risk to aquatic mammals at Marsden Lake (North), it appears that adverse toxicological effects are possible, albeit unlikely in the population of aquatic mammals as a result of exposure to aluminum, antimony, arsenic, barium, lead, manganese, mercury, vanadium, and zinc.

However, there remains some uncertainty as to whether adverse effects are occurring due to limitations in the evidence. The assessment endpoint for aquatic mammals (reproductive potential and productivity of aquatic mammals) was not measured directly. 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 – ratios between the exposure dose and the dose associated with either NOAEL or LOAEL laboratory studies. A ratio that exceeds unity triggers further careful consideration of the underlying scientific basis of the prediction.

8.4 ECOLOGICAL RISKS AT MARSDEN LAKE (SOUTH)

In this assessment, Marsden Lake (South) refers to the portion of Marsden Lake directly adjacent to the Outdoor Firing Range (see Figure B1-3). The south area was defined as a distinct area to focus the assessment in terms of characterizing any potential risks due to contaminants that could have come from activities at the Outdoor Firing Range. The selected surface water COC for the risk characterization of Marsden Lake (South) was manganese. Aluminum, cadmium, lead, vanadium, and SVOCs were retained as sediment COCs. These COCs were selected based on the COC selection process presented in Section 5. These COCs (except for SVOCs) were found at concentrations greater than background and toxicological benchmark concentrations. SVOCs were not detected; however, the target detection limits were not achieved in the analysis of sediment samples. This limitation prevents SVOCs from being eliminated as sediment COCs.

Marsden Lake (South) COCs

Surface Water
Manganese

Sediment
Aluminum
Cadmium
Lead
Vanadium
SVOCs

During the development of the Work Plan in 1998 (USACHPPM 1999), it was suspected that the Outdoor Firing Range could be a specific source of COPCs detected in Marsden Lake. Contaminants in the Outdoor Firing Range soils may have been transported via runoff into the lake, resulting in metals contamination in the surface water and sediment in the area defined as Marsden Lake (South). Activities that occurred at the Range are known to have caused antimony, copper, and lead contamination of the Firing Range soil (Alliant Techsystems 2001b). Section 6.2.1 provides further detail on the possible sources of COCs to Marsden Lake (South). Even though SVOCs remain potential COCs, they are not strongly suspected to be true contaminants associated with the Outdoor Firing Range (Alliant Techsystems 2001b). Known soil contaminants at the Outdoor Firing Range have recently been remediated to an industrial land use standard (Alliant Techsystems 2001b).

As explained in the previous section for Marsden Lake (North), there was a limitation in characterizing the spatial aspect of potential contamination related to Marsden Lake (North). As a result, two additional samples (ML-13 and ML-15 from Marsden Lake (North)) were combined with the original ten Marsden Lake (South) samples so that the data set for Marsden Lake (South) consisted of 12 samples (see Section 4.1.2.2 for further detail). The spatial distribution of the contamination in the sediment was expected to emanate from the range impact areas, along the western shoreline of the lake (refer to Figure E1). It is more difficult to predict the spatial distribution of surface water COCs given the sampling design used in the field investigations. However, the analytical data show little spatial variability in surface water concentrations.

A revised conceptual site model was developed for Marsden Lake (South) to show the specific COCs being evaluated and to indicate fate and transport pathways (see Figure B3-2)

8.4.1 Hypothesis 1: Effects in Fish, Aquatic Invertebrates, and Algae

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species
>
> Risk Hypothesis – Based upon the aquatic toxicity of manganese; if it is present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

8.4.1.1 Risk Estimates

Risks for these ecological entities were estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Exhibit 8-10 presents the quantitative risk estimates, which are then described below.

For their entire life cycle, these species are continuously exposed to manganese; however, concentrations are expected to fluctuate by season and as a result of other changes in the microhabitat (e.g., changes in the amount of suspended solids over time). The portion of the community and its populations that are actually exposed to levels greater than effects benchmarks in Marsden Lake (South) is not known precisely due to limited resources available for sampling the entire Marsden Lake (South) area.

Exhibit 8-10. Marsden Lake (South) Risk Estimate for Fish, Aquatic Invertebrates, and Algae

COC	Hazard Quotient*	Portion Greater Than Chronic Benchmark[†]	Portion Greater Than Acute Benchmark[†]
Manganese	3	8 %	0 %

* HQ is calculated using central tendency concentration from Table A28 and chronic water quality benchmark from Appendix O and rounded to 1 significant figure.
[†] Percentages based on 12 sample locations.

<u>Manganese</u>: Based on the risk estimate provided in Exhibit 8-10, adverse toxic effects to fish, aquatic invertebrates, and algal species due to manganese may be occurring because the HQ is 3 and 8 % of the samples had a concentration greater that the chronic water quality benchmark (120 μg/L). However, none of the sample locations had a manganese concentration greater than the acute water quality benchmark, indicating that more serious effects are not expected (see Figure B4-7).

No samples had concentrations greater than the next highest alternative water benchmark based on effects in daphnids, from Suter and Tsao (1996). The lowest chronic value for fish developed by Suter and Tsao (1996) was also not exceeded, indicating that fish may not be affected.

8.4.1.2 Risk Description

Based on the line of evidence used to estimate risk to fish, aquatic invertebrates, and algae at Marsden Lake (South), it appears that adverse toxicological effects due to manganese are possible, but probably unlikely, given the magnitude of the risk estimates in light of alternative effects benchmarks. However, there remains uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. Additionally, the suspected source area (the Outdoor Firing Range soils) has been remediated to industrial scenario standards based on the protection of human health.

The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. A Minnesota Water Quality Standard for Class 2B waters was not available for manganese, so other effects-based benchmarks for fish, invertebrates, and algal species were used. The chronic and acute water quality benchmarks presented for manganese are Tier II values calculated by EPA (1993b) and taken from Suter and Tsao (1996). The Tier II values are derived from fewer studies than the NAWQCs, but are still considered to be protective of aquatic organisms. Suter and Tsao (1996) also provide the lowest chronic values for fish and daphnids, as shown on Figure B4-7. The water quality criteria used in this assessment for manganese are linked to the assessment endpoint because survival and development of a population of aquatic organisms should be protected by this conservative water quality standard. However, by using the benchmarks we extrapolate from the laboratory to the field and between different species (i.e., between taxa). These extrapolations introduce uncertainty. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more, or less sensitive to the chemicals. In addition, the species tested in the laboratories may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar. 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).

A statistical comparison of the surface water COC concentrations in Marsden Lake (South) to those at the reference site (Snail Lake) indicated a difference in manganese concentrations (see Table A29). A comparison of the manganese concentrations at the reference location to the water quality benchmark shows that manganese in surface water at Snail Lake is below the chronic benchmark. However, the benchmark does not account for the total suspended solids (TSS) concentrations in the lakes because it is based on the dissolved metal concentration from filtered water samples. Surface water samples in this investigation were unfiltered, with Marsden Lake (South) having a slightly higher central tendency TSS concentration (5.58 mg/L) than the reference lake (1 mg/L) (see Table A21). The TSS concentration is not expected to result in extremely different metals concentrations between the lakes.

The spatial distribution of surface water COC concentrations at Marsden Lake (South) can be visualized by comparing Figure E1 (showing sampling locations) to Figure B4-7 (bar graph indicating the sample locations and the surface water benchmark for manganese). The only sample location that exceeded the benchmark was ML-08. ML-08 is located directly east of the 1900-yard range bullet catcher. As indicated in a recent TCAAP report (Alliant Techsystems, 2001b), the 1900-yard range soil was an area of concern only for to-be-evaluated (TBE) compounds that included lead and PAHs. Therefore, it does not appear that the 1900-yard range soil was the source of manganese to this location. However, the safety fan of the outdoor firing range did extend into Marsden Lake (Alliant Techsystems, 2001b) and bullets containing manganese that may have landed in the lake can be one source to the surface water. This one sample location out of 12 that exceeded the benchmark indicates that the adversity of effects may be low since the spatial scale examined indicates an isolated area of benchmark exceedance. In addition to manganese, aluminum and lead concentrations in surface water at ML-08 also exceeded their chronic benchmarks. However, aluminum and lead were not selected as surface water COCs in Marsden Lake South based on a statistical comparison with background.

8.4.2 Hypothesis 2: Effects in Benthic Organisms

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, growth, and reproduction of benthic organisms
>
> Risk Hypothesis - Based upon the toxicity of sediment COCs (aluminum, cadmium, lead, vanadium, and SVOCs); if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

8.4.2.1 Risk Estimate

The potential for adverse toxicological effects in these ecological entities were estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.2). Exhibit 8-11 presents the quantitative risk estimates, which are then described below.

For their entire life cycle, these species are continuously exposed to aluminum, cadmium, lead, vanadium, and SVOCs; however, concentrations of the metals are expected to fluctuate by season and as a result of other changes in the microhabitat (e.g., changes in SEM:AVS, organic matter content, and dissolved oxygen). The portion of the community and its population at Marsden Lake (South) that are actually exposed to levels greater than effects benchmarks is not known precisely, due to limited resources available for sampling the entire northern portion of the lake.

Exhibit 8-11. Marsden Lake (South) Risk Estimates for Benthic Organisms

COC	Hazard Quotient*	Portion Greater Than Low-effect Benchmark[†]	Portion Greater Than Moderate-Effect Benchmark[†]
Aluminum	<1	0 %	0 %
Cadmium	<1	33 %	0 %
Lead [‡]	1	75 %	0 %
Vanadium	unavailable	unavailable	unavailable
SVOCs	unavailable	unavailable	unavailable

* HQ is calculated using central tendency concentrations from Table A30 and low-effect sediment benchmark from Appendix O. Values are rounded to 1 significant figure.

[†] Percentages based on 12 sample locations.

[‡] For lead, the unrounded HQ is 1.25. This explains the apparent inconsistency between having an HQ of 1 even though 75 percent of the samples have concentrations greater than the low-effect benchmark.

SVOCs: Semivolatile organic compounds

Aluminum: Based on the risk estimate provided in Exhibit 8-11, adverse toxic effects in benthic organisms due to aluminum are not expected because the HQ is less than one, and no sample locations exceeded either the low-effect or moderate-effect benchmark (see Figure B4-16).

Cadmium: Based on the risk estimate provided in Exhibit 8-11, adverse toxic effects in benthic organisms due to cadmium may not be occurring. The HQ (based on the central tendency concentration) was less than one, and four of twelve samples exceeded the low-effect benchmark (1 mg/kg). However, the four samples only slightly exceeded the value (ranging from 1.1 to 1.6 mg/kg) see Figure B4-17. The low-effect benchmark is a USEPA Region IV sediment quality screening value and is the Contract Laboratory Program's Practical Quantitation Limit (see Appendix O). The value was based on a toxicity study of Florida coastal water sediments USEPA (2000). USEPA Region IV (2000) notes that the sediment screening values they provide have been developed from a database containing information from studies conducted predominantly in marine environments, and that personal communication with the authors of the studies indicate that corresponding values being developed from a freshwater database are within a factor of three of the marine based numbers. Additionally, no samples exceeded the moderate-effect benchmark (3.5 mg/kg).

Lead: Based on the risk estimate provided in Exhibit 8-11, adverse toxic effects in benthic organisms due to lead may be occurring. The HQ was one, and 75% of the sample locations exceeded the low-effect benchmark (see Figure B4-18). However, no sample locations exceeded the moderate-effect benchmark (91.3 mg/kg).

Vanadium: No risk estimate could be calculated for vanadium because sediments were not analyzed for vanadium. Additionally, no sediment toxicological benchmark is available for vanadium. Therefore, a risk estimate cannot be provided for this COC.

SVOCs: No risk estimate could be calculated for SVOCs due to an analytical problem of meeting the detection limit. All results reported nondetects for SVOCs at Marsden Lake (South). Therefore, without a concentration a risk estimate could not be calculated.

8.4.2.2 Risk Description

Based on the line of evidence used to estimate risk to benthic organisms at Marsden Lake (South), it appears that some adverse toxicological effects are possible due to lead because the low-effect benchmark is exceeded in the majority of samples. Adverse effects to benthic organisms from cadmium are possible, albeit unlikely. The possibility of effects from vanadium and SVOCs cannot be ruled out. However, there remains some uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. Additionally, the suspected source area has been remediated to industrial scenario standards based on the protection of human health.

The assessment endpoint (survival, growth, and reproduction of benthic organisms) was not measured directly for this line of evidence. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site sediment concentration data to toxicity benchmarks. There are no state or federally recognized sediment benchmarks promulgated for use. Sediment quality benchmarks can be reasonably linked to this endpoint because survival, growth, and development are often the endpoints used to derive these benchmarks. Many sediment benchmarks have been derived from chemical concentrations detected in whole sediments that are associated with adverse effects to lab tested and/or *in situ* tested benthic organisms. Some benchmarks, however, are based on presence/absence of a particular species when comparing field sediments across multiple locations. Such benchmarks are less relevant to the assessment endpoint.

By using the benchmarks we extrapolate from the laboratory to the field, between different field sites, and between taxa. These extrapolations introduce uncertainty. The species tested in the laboratories or observed in the field-collected sediments of reference areas may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more or less sensitive to the chemicals. When we use benchmarks based on reference sediments, then we assume that environmental and ecological processes between the reference sites and the study site do not differ in significant ways.

A statistical comparison of the sediment COC concentrations at Marsden Lake (South) to those at Snail Lake indicated no significant difference. The null hypothesis tested was that site concentrations of COCs are greater than the reference concentrations. Therefore, no sediment COCs at Marsden Lake (South) were determined to be statistically greater than the concentrations detected at the reference location (Snail Lake).

The spatial distribution of sediment COC concentrations at Marsden Lake (South) can be visualized by comparing Figure E1 (showing sediment sampling locations) to Figures B4-16 through B4-18 (bar graphs indicating the sample locations and the sediment benchmarks for each COC). No distinct spatial pattern is evident. However, locations ML-04, ML-05, and ML-06 appear to have typically lower concentrations of sediment COCs than other sites. These three locations are near the western shoreline and appear to be just down range of the 1900-yard bullet catcher. Of the sediment COCs selected, lead is the only COC that was also a soil COC at the Outdoor Firing Range. Additionally, the Grenade Range soil COCs were cadmium, antimony, and lead, but there is no definitive information on the transport of Grenade Range soil COCs to the southern area of Marsden Lake.

8.4.3 Hypothesis 3: Effects in Amphibians

The following assessment endpoint and risk hypothesis were evaluated.

Assessment Endpoint – Survival, development, and reproduction of amphibians

Risk Hypothesis –Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment. (An adequate evaluation of potential problems for the amphibian community of the lake was a major data gap in the Tier I risk assessment).

8.4.3.1 Risk Estimate

The potential for adverse toxicological effects in amphibians was inferred by evaluating three lines of evidence. Risk estimates based on each of these lines are provided in this subsection. A synthesis is provided in the Risk Description section. The information provided below is the same information provided for Hypothesis 3 in Marsden Lake (North). All samples and field data collected at Marsden Lake were combined to provide an overall assessment of Marsden Lake rather than dividing it by specific source areas (i.e. North and South).

It is expected that these organisms will be exposed during the embryo and juvenile portions of their life cycle. Additionally, exposure for adults is expected to be on an intermittent basis. However, sufficient information is not available to determine the size of the population exposed to the COCs.

FETAX RESULTS

Risk estimates for the toxicity test line of evidence, using surface water from Marsden Lake for the FETAX, were developed based on technique 1. Exhibit 8-12 presents the quantitative risk estimates for this line of evidence, which are then described below.

Exhibit 8-12. Marsden Lake Risk Estimates for Amphibians based on results of the 1999 FETAX Tests†

Site	Petri Dish ID	Initial Number of Embryos	Mortality Count	Mortality Rate (%)	Number Malformed	Malformation Rate (%)
Marsden Lake	3	25	6	24	6	35
Marsden Lake	4	25	4	16	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 • 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.

Based on the risk estimate provided in Exhibit 8-12, amphibians in Marsden Lake appear to have acceptable occurrences of mortality and malformations when compared to the controls and the reference site. The report, containing supporting information on this conclusion, is found in Appendix M. The types and incidences of malformations observed in Marsden Lake samples were not statistically different from controls or the reference site. Also, mortality was not found to be statistically different in Marsden Lake samples than in the reference samples. However, there was a slightly higher incidence of mortality in Marsden Lake compared to the controls. These negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected, and may be unacceptable for decision-making in some cases. The FETAX test was conducted on a composite surface water sample consisting of six samples from around the perimeter of all of Marsden Lake, in order to obtain an overall characterization of the site (Marsden Lake as a whole, not (North) and (South)). Therefore, effect(s) could not be linked to a potential "hot spot" or suspected source area in the lake. Additionally, surface water chemical analysis for metals was not conducted at the same time. FETAX was not conducted with sediments from Marsden Lake due to the inherent difficulties in dealing with sediment in amphibian assays (i.e. with dejellied embryos in FETAX, fouling the water in adult exposure assays). The possible effects of sediment were considered such that water-soluble constituents would be expected to partition to the water and then to the organism. Therefore, exposure (dermal) to sediment COCs was assumed to be at equilibrium between water and sediment (pore water).

IMMUNE FUNCTION, ORGAN WEIGHTS, AND HISTOPATHOLOGY

For this line of evidence qualitative risk estimates are provided based on data collected on organ weights, two immune function markers, and histopathologic evaluations of important blood conditioning organs. Comparisons were made in adult frogs (*X. laevis*) exposed 14-days to surface water from Snail and Marsden Lakes and also from a laboratory control solution used in the FETAX assay. The two markers, based on phagocyte response tests, are sensitive indicators of adverse immune responses. Organ weights can be useful indicators of toxicity. Appendix M presents the full results.

- The production of radical oxygen intermediates (ROI) in adult frogs exposed to Marsden Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).

- The phagocytosis of foreign particles by splenic macrophages in adult frogs exposed to Marsden Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).

- Ratios of organ weight to body weight in adult frogs exposed to Marsden Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).

- No notable histological differences were found between sections of thymus, lung, kidney, and liver when examined using light microscopy for male and female frogs exposed to Marsden Lake surface water, compared to Snail Lake and the laboratory control solution.

SPECIES RICHNESS AND RELATIVE ABUNDANCE

Risk estimates for this line of evidence were developed using visual encounter surveys, aural estimates, and trapping. Exhibit 8-13 presents the qualitative risk estimates for this line of evidence.

Exhibit 8-13. Marsden Lake Risk Estimate for Amphibians Based on Species Richness

Site	Number of Species Seen or Heard in April '99, May '98, and July '98	Species Observed
Marsden Lake	3	*Bufo americanus, Rana pipiens, Pseudacris triseriata*
Snail Lake	6	*Bufo americanus, Rana clamitans, Rana pipiens, Ranid species, Pseudacris triseriata, Hyla chrysoscelis*
Risk Estimate	• No statistically significant differences were found between the reference site and Marsden Lake for the number of species observed. • It was not possible to accurately report relative abundance due to difficulties in accessing the sites because of the dense cattail mats. Species observed may have been listed because they were heard but not seen, so a definitive count was not possible.	

Based on the risk estimate provided in Exhibit 8-13, the diversity of amphibians in Marsden Lake appears to be no different than that in the reference lake (Snail Lake). Due to habitat constraints and the relatively quick Anuran reaction reflex, many frogs detected by sight during the VES surveys could not be reliably identified. Therefore, more species may exist at each site than was originally reported. The study concluded that overall, relatively few amphibians were found.

8.4.3.2 Risk Description

The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species for FETAX, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. The potential for adverse toxicological effects in amphibians at Marsden Lake is inferred using three lines of evidence. Risk estimates based on each of these lines were provided in the previous subsection. However, it is noted that there are uncertainties associated with the amphibian study including but not limited to; no multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, and laboratory tests used species not found at the site. These uncertainties along with others can be found in Table 6.1 in the amphibian report (see Appendix M). Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations, and site surface water was not analyzed for COC concentrations at the time of the amphibian study. Therefore, effects from specific surface water COCs were not reported, because it was not part of the study design.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of the assessment.

Laboratory data investigating embryo mortality (a sensitive life stage), developmental effects, and sensitive indicators of stress (immunocharacterization assays) suggest that surface water from Marsden Lake is not toxic to frogs, yet sediment exposures were not specifically evaluated. However, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 63%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation (see Tables 5 and 6 in Appendix M). The field data were ambiguous, yet amphibians were noted at all locations.

Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to metals from the Outdoor Firing Range or the Grenade Range is not apparent. Additionally, data collected by Jannett (1997) and others (as discussed in Appendix M) appear to corroborate a conclusion that adverse impacts of chemical exposure to amphibians are unlikely.

8.4.4 Hypothesis 4: Effects in Waterfowl and Wading Birds

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Reproductive potential and productivity of waterfowl and wading birds
>
> Risk Hypothesis – The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COCs in surface water, sediment, and residues in food from the lake.

8.4.4.1 Risk Estimates

The potential for adverse toxicological effects in waterfowl and wading birds was estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Risk estimates are provided in this subsection. A synthesis is provided in the Risk Description subsection.

WATERFOWL

Marsden Lake is used by a number of waterfowl species including Canada geese (*Branta canadensis*) and Mallards (*Anas platyrhynchos*). Common loons (*Gavia immer*) were heard but not seen during field investigations. Trumpeter swans are also known to be present in the south portion of Marsden Lake. Unlike Marsden Lake (North), the south portion of the lake near the western shoreline does contain a fair amount of open water that will attract waterfowl because it provides better foraging area for dabbling, seed eating ducks than Marsden Lake (North). More open water is also more attractive to common loons.

Sufficient field information is not available to describe the population size of waterfowl at Marsden Lake (South). However, it is expected that more waterfowl will be found in Marsden Lake (South) than in Marsden Lake (North). Dabbling ducks such as the wood duck and blue-winged teal are expected to feed in the waters of Marsden Lake (South) and nest in the cattails near the shoreline or on the floating cattail mats in the lake. Diving ducks, particularly loons, are also expected in this portion of the lake due to the availability of more open water. Exposure of mallards, Canadian geese, and other waterfowl species is expected to be continuous, but for the common loon exposure is expected to be intermittent, only a few times during their migratory season.

Risk estimates are provided in Exhibit 8-14. A synthesis is provided in the Risk Description section.

Exhibit 8-14. Waterfowl Hazard Ratio Matrix for Marsden Lake (South)

	Mallard	
Aluminum:	HQ1	HQ2
Maximum	8	ND
Central Tendency	5	ND
Cadmium:		
Maximum	0	0
Central Tendency	0	0
Lead:		
Maximum	13	1
Central Tendency	9	1
Manganese:		
Maximum	0	ND
Central Tendency	0	ND
Vanadium:		
Maximum	0.1	ND
Central Tendency	0.1	ND
SVOCs		
Maximum	ND	ND
Central Tendency	ND	ND

ND: Not determined, a component was not available for calculation of the HQ.

Cadmium, Manganese, Vanadium: None of the hazard quotients for these COCs are greater than unity. This indicates that adverse health effects due to these COCs are not expected for avians at Marsden Lake (South). Lowest-observable-adverse-effect-levels were not available for manganese and vanadium, so LOAEL based HQs could not be calculated for these COCs. However, NOAEL based HQs were below unity for manganese and vanadium so it is expected that LOAEL based HQs would also be below unity.

Aluminum: Two of the four HQs for the modeled mallard exceeded unity. The other two HQs (based on a LOAEL TRV) could not be calculated because the data was not available. The HQ of 8 (based on the maximum site concentration and an estimate of the no-adverse-effect-level) indicates that a maximally exposed waterfowl in the population may experience an exposure 8-times greater than that considered to be safe. The HQ of 5 (based on the central tendency concentration and the NOAEL) indicates that on average, individual waterfowl in the population may experience exposures 5-times greater than that considered to be safe. In many cases, LOAEL based HQs are an order of magnitude lower than NOAEL based HQs. Therefore, we expect that if a LOAEL based TRV were available for aluminum, the resulting

HQs would be an order of magnitude lower than the NOAEL values (i.e. equal to or less than unity in this case).

Lead: Two of the four HQs for the modeled mallard exceed unity. The HQ of 13 (based on the maximum site concentration and an estimate of the NOAEL) indicates that a maximally exposed waterfowl in the population may experience an exposure 13-times greater than that considered to be safe. The HQ of 9 (based on the central tendency concentration and the NOAEL) indicates that on average, individual waterfowl in the population may experience exposures 9-times greater that that considered to be safe. However, because the HQ ratios based on the LOAEL are not greater than unity (1 and 1), estimated exposures in the waterfowl population are less than exposures known to be associated with adverse health effects.

SVOCs: Hazard quotients for SVOCs could not be calculated due to detection limit problems when analyzing the sediment data. Uncertainty remains as to the presence of SVOCs in the Marsden Lake sediments, and thus whether or not there would be an adverse effect on waterfowl.

WADING BIRDS

The duration of exposure is dependent on the habitat usage required by each species of wading bird. The great blue heron is known to nest in the vicinity of TCAAP and is expected to be exposed throughout its life cycle (USGS 1995). The exposure duration for the bitterns is expected to be more frequent than that of the herons, and is estimated that they will be exposed for the majority of their seasonal stay at TCAAP. Herons have been reported in the area from March through November and most leave the area for the winter (USGS 1995). Also, there is no evidence of herons nesting on TCAAP. However, bitterns and other wading birds may nest at Marsden Lake (South) in the cattail areas. It is expected that herons and egrets will forage there occasionally during their stay in the TCAAP vicinity. For those birds that will be exposed, exposure is expected to be intermittent.

Risk estimates are provided in Exhibit 8-15. A synthesis is provided in the Risk Description section.

Exhibit 8-15. Wading Bird Hazard Ratio Matrix for Marsden Lake (South)[†]

		Heron (no sediment ingestion)		Heron (10 % sediment ingestion)	
		HQ1	HQ2		
Aluminum:					
	Maximum	0.1	ND	0.1	ND
	Central Tendency	0	ND	0.1	ND
Cadmium:					
	Maximum	0	0.0	0	0.0
	Central Tendency	0	0.0	0	0.0
Lead:					
	Maximum	0.1	0.0	0.3	0.0
	Central Tendency	0.1	0.0	0.2	0.0
Manganese:					
	Maximum	0	ND	0	ND
	Central Tendency	0	ND	0	ND
Vanadium:					
	Maximum	0	ND	0	ND
	Central Tendency	0	ND	0	ND

SVOCs				
Maximum	ND	ND	ND	ND
Central Tendency	ND	ND	ND	ND

†Exposure to the heron was modeled using zero percent sediment ingestion and a worst-case scenario ingestion of 10%. See Appendix N, Section N3.3.
ND: Not determined, a component was not available for calculation of the HQ.

None of the calculated hazard quotients for the COCs are greater than unity. This indicates that adverse health effects are not expected for wading birds that frequent Marsden Lake (South). Estimates are provided for two exposure scenarios: one with sediment ingestion and one without sediment ingestion. No sediment ingestion data for wading birds was found in the literature. However, it is possible that wading birds may be ingesting sediment to some degree. It was therefore decided to produce hypothetical values of sediment ingestion rates across a range of portions of sediment in the diet (1, 5, and 10 percent of the diet), in order to support a sensitivity analysis.

Additionally, there is uncertainty with regard to exposure and effect of SVOCs to wading birds. Due to target detection limits not being reached during the sediment analysis, SVOCs in sediment were not detected. Confident conclusions cannot be made concerning SVOCs.

8.4.4.2 Risk Description

Based on the line of evidence used to estimate risk to waterfowl at Marsden Lake (South), it appears that population-level adverse toxicological effects are possible but not likely due to aluminum and lead (see Exhibit 8-14). The HQ ratios for lead based on the lowest-observed-adverse-effect-level are not greater than unity, indicating that the estimated maximum and average exposures in the waterfowl population are less than exposures known to be associated with adverse health effects. No comparable toxicity data (i.e., LOAELs) exist for aluminum, but the unknown LOAEL values for aluminum are also expected to be less than unity given the magnitude of the NOAEL-based HQ ratios for aluminum. The possibility of effects from SVOCs cannot be ruled out because of limitations in the sample analysis for these compounds.

Adverse toxicological effects are not expected in wading birds at Marsden Lake (South): all HQ ratios are less than unity. The possibility of effects from SVOCs cannot be ruled out because of limitations in the sample analysis for these compounds.

The assessment endpoint for these entities (reproductive potential and productivity of waterfowl and wading birds) was not measured directly. 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—ratios between the exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. A ratio that exceeds unity triggers further careful consideration of the underlying scientific basis of the prediction.

8.4.5 Hypothesis 5: Effects in Mink and Muskrat

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint –Reproductive potential and productivity of aquatic mammals
>
> Risk Hypothesis –The health of aquatic mammals such as mink or muskrat, may be impaired if they are excessively exposed to COCs in the surface waters, sediments, and food residues.

8.4.5.1 Risk Estimates

The potential for adverse toxicological effects in aquatic mammals was estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Since the initiation of the risk assessment, the muskrat has been the primary mammal of concern. In addition to the risk of COC exposure to the muskrat, risk to the mink was also to be evaluated, but only for exposure to mercury. It is recognized that other metals and possibly organics could be present in dietary items of the mink; however, the use of the muskrat to evaluate risk to aquatic mammals provides an estimator for risk to COC other than mercury. Note that mercury was not selected as a final COC for this site so risk estimates were not produced for the mink.

MUSKRAT

Several dens were observed in the southern portion of the lake, but a true survey was not conducted. Based on habitat preference information, it appears that more muskrats can be expected in Marsden (South) than in Marsden (North), and using an estimation method it is possible that between 12 and 20 muskrats may be present. However, because the available information about the type of habitat at Marsden Lake and about muskrat population densities is limited, no accurate estimate of actual population size can be made. The area is largely covered by cattails, which is preferred habitat, but without a formal den count or population study at the lake, there is not enough information available to determine how many muskrats may be present. The Muskrat that are exposed are likely to be exposed for the majority of their life cycle because their home range is small.

Risk estimates are provided in Exhibit 8-16. A synthesis is provided in the Risk Description section.

Exhibit 8-16. Aquatic Mammal Hazard Ratio Matrix for Marsden Lake (South)

	Muskrat	
Aluminum as $AlCl_3$	HQ1	HQ2
Maximum	22	5
Central Tendency	15	3
Aluminum as $Al(OH)_3$		
Maximum	3	ND
Central Tendency	2	ND
Cadmium:		
Maximum	0.1	0
Central Tendency	0	0
Lead:		
Maximum	0	0
Central Tendency	0	0
Manganese:		
Maximum	0.3	0.1
Central Tendency	0.2	0.1
Vanadium:		
Maximum	0.8	0.2
Central Tendency	0.6	0.1
SVOCs		
Maximum	ND	ND
Central Tendency	ND	ND

NA: Not applicable
ND: Not determined

Cadmium, Lead, Manganese, Vanadium: None of the calculated hazard quotients for the COCs are greater than unity. This indicates that adverse health effects due to these COCs are not expected for aquatic mammal at Marsden Lake (South).

Aluminum: All four HQs based on the muskrat model exceed unity. These HQs were calculated based on an allometrically scaled NOAEL (3.05 mg/kg/day) resulting from dosing laboratory animals with aluminum chloride ($AlCl_3$) (see appendix O). However, there is uncertainty as to what form of aluminum is predominant in the sediments at the lake. Based on site-specific information and professional judgment, it is thought that the form of aluminum found in the waterbodies at TCAAP is not solely aluminum chloride, but rather a combination of aluminum complexes such as aluminum hydroxide ($Al(OH)_3$), aluminum bicarbonate, or complexes with other ligands. This leads us to believe that the current TRV (based on aluminum chloride) may be over conservative. The toxicity of aluminum is intimately associated with pH in that the metal is soluble and biologically available in acidic ($pH < 5.5$) soils and waters but relatively innocuous in pH conditions between 5.5 – 7.5 (Sparling and Lowe 1996). Also, concentrations of dissolved aluminum in water vary not only with pH but also with the humic-derived acid content of the water (ATSDR 1999). The TRV value for wildlife is based on a laboratory study in which $AlCl_3$ was administered to mice. Alternatively, two oral exposure studies of aluminum hydroxide discussed in the Aluminum profile (ATSDR 1999), one to rats and one to mice were considered. The study on mice (Domingo et al 1989) was a chronic study in which mated female mice were given by gavage daily doses of $Al(OH)_3$ on gestation days 6 through 15. The study found no evidence of maternal toxicity, embryo/fetal toxicity or teratogenicity even at levels as high as 266 mg Al/kg/day as aluminum hydroxide. The other study (Greger and Donnaubauer 1986), was an intermediate exposure (subchronic) study in which no changes in body weight were found in male Sprague-Dawley rats that ingested up to 158 mg Al/kg/day as aluminum hydroxide in the diet for 16 days. When developing a TRV, studies of chronic duration are more favorable. Based on the Domingo et al (1989) study, a TRV could be generated following the USACHPPM (2000) protocol (see appendix O). The chronic NOAEL TRV derived from the Domingo et al (1989) study would be 26.6 mg/kg/d aluminum. It is more appropriate to develop a TRV based on the form of the chemical that the organisms will be exposed to and an exposure duration that would be more representative of field conditions. Using the $Al(OH)_3$ based NOAEL TRV of 26.6 mg/kg/d would decrease the HQ values by an order of magnitude at the maximum exposure concentration HQ (22 mg/kg-day for $AlCl_3$ to 3 mg/kg-day for $Al(OH)_3$), resulting in an HQ slightly greater than unity. See Exhibit 8-8. The HQ of 3 (based on the maximum site concentration and an estimate of the no-observable-adverse-effect-level) indicates that a maximally exposed muskrat in the population may experience an exposure 3-times greater than that considered to be safe. This information may indicate that the population of muskrats is not at risk of adverse effects due to aluminum at Marsden Lake (South).

SVOCs: Hazard quotients for SVOCs could not be calculated due to detection limit problems when analyzing the sediment data. Uncertainty remains as to the presence of SVOCs in the Marsden Lake sediments, and thus whether or not there would be an adverse effect on aquatic mammals.

8.4.5.2 Risk Description

Based on the line of evidence used to estimate risk to aquatic mammals at Marsden Lake (South), it appears that the form of aluminum found at the site may influence the outcome (see Exhibit 8-16). Assuming that mammals are exposed to aluminum chloride, then the data suggest that population-level adverse effects are possible because the average exposure would be 3-times greater than exposures known to be associated with adverse health effects. On the other hand, assuming they are exposed to aluminum hydroxide, then the data suggest that population-level adverse effects may not be expected given the magnitude of the available HQ ratios associated with "safe" exposures. (Note: exposure levels known to be associated with effects for this form are not known). Based on supporting information provided earlier in the report regarding the expected form of aluminum at the site, the aluminum hydroxide-based conclusion is considered to be more site-specific. However, there remains uncertainty as to whether adverse effects are occurring due to limitations in the evidence. The possibility of effects from SVOCs cannot be ruled out because of limitations in the sample analysis for these compounds.

The assessment endpoint for aquatic mammals (reproductive potential and productivity of aquatic mammals) was not measured directly. 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 – ratios between the exposure dose and the dose associated with either NOAEL or LOAEL laboratory studies. A ratio that exceeds unity triggers further careful consideration of the underlying scientific basis of the prediction.

8.5 ECOLOGICAL RISKS AT POND G

The selected COCs evaluated in the risk characterization of Pond G include 13 surface water inorganics, 6 sediment inorganics, as well as Aroclor 1248, SVOCs, and OC pesticides in the sediment. The COCs were selected based on the process presented in Section 5. These COCs were found at concentrations greater than toxicological benchmark concentrations.

The specific source(s) of the COCs (listed in the textbox to the right) in Pond G is suspected to be Site G (uncontrolled landfill), Site F (former burning area), or both. The source areas are no longer in operation; therefore the contamination is historical. The secondary sources for these COCs would be the_sediment, and surface runoff from contaminated upland soil areas. Groundwater seeps have also been suggested as a possible source.

Soils and sediments naturally contain many of the detected inorganic COCs. Since a comparable reference location for Pond G was not used in the selection of COCs (USACHPPM 1999), it is difficult to determine whether many of the inorganics present in the Pond are actually contaminants. Regardless of the true nature of the source(s) for these COCs, the design of the assessment is based on characterizing the aggregate exposure (i.e., from all sources) as measured in the surface water at this site.

Based on the small size of the Pond and its physical characteristics, it is assumed that the spatial and temporal distribution of the COCs is basically uniform. The entire Pond is assumed to contain concentrations of these COCs which are not likely to change significantly for the foreseeable future because they all are environmentally persistent and a continuing source of the COCs is not believed to be present.

Pond G COCs

Surface Water
Aluminum
Antimony
Arsenic
Barium
Beryllium
Cadmium
Chromium
Cobalt
Copper
Lead
Manganese
Selenium
Thallium

Sediment
Aluminum
Antimony
Arsenic
Chromium
Copper
Lead
Vanadium
SVOCs
OC Pesticides
Aroclor 1248

A revised conceptual site model was developed for Pond G to show specific COCs being evaluated and to indicate fate and transport pathways (see Figure B3-3).

8.5.1 Hypothesis: Effects in Mammals and Wading Birds

The following assessment endpoint and risk hypothesis were evaluated.

Assessment Endpoint – Water and sediment quality for wildlife using the pond

Risk Hypothesis - Based upon the toxicity of metals, OC Pesticides, SVOCs, and Aroclor 1248 present in the water or sediment of the pond; if they exist at sufficient concentrations then adverse toxic health effects may occur in foxes and wading birds that use the pond as a drinking water source, and may incidentally ingest sediment.

8.5.1.1 Risk Estimate

Risks for these ecological entities were estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). The average COC concentrations were used in the exposure model because Pond G is so small, it is assumed that the COC concentrations should be uniform.

WADING BIRDS

Individuals of local populations of these groups may be attracted to this aquatic site, indicating that some degree of co-occurrence in time and space would occur. On an occasional basis, wading birds have been observed at the Pond. These birds will contact the COCs while standing (and possibly drinking from) the Pond. These species will not be continuously exposed because they are expected to frequent the pond on a very limited basis because of the availability of other higher quality habitat nearby and the inability of the site to provide a sustainable supply of food.

Risk estimates are provided in Exhibit 8-17. A synthesis is provided in the Risk Description subsection.

Exhibit 8-17. Wading Bird Hazard Ratio Matrix for Pond G Based on Average Exposure Doses

	Heron (no sediment ingestion)		**Heron** (10% sediment ingestion)	
	HQ1	HQ2	HQ1	HQ2
Aluminum	0	ND	0	ND
Antimony	ND	ND	ND	ND
Arsenic	0	0	0	0
Barium	0	0	0	0
Beryllium	ND	ND	ND	ND
Cadmium	0	0	0	0
Chromium	0	0	0	0
Cobalt	ND	ND	ND	ND
Copper	0	0	0	0
Lead	0	0	0.5	0
Manganese	0	ND	0	ND
Selenium	0	0	0	0
Thallium	ND	ND	ND	ND
Vanadium	0	ND	0	ND
Aroclor 1248	0	0	0	0

ND: Not determined due to missing data needed for the HQ calculation

Arsenic, Barium, Cadmium, Chromium, Copper, Lead, Selenium, and Aroclor 1248: None of the calculated hazard quotients for these COCs are greater than unity. This indicates that adverse health effects are not expected for wading birds that use Pond G as a drinking water source. Estimates are provided for two exposure scenarios: one with and one without incidental sediment ingestion. No sediment ingestion data for wading birds was found in the literature. However, it is possible that wading birds may be ingesting sediment to some degree. It was therefore decided to produce hypothetical values of sediment ingestion rates across a range of portions of sediment in the diet (1, 5, and 10 percent of the diet), in order to support a sensitivity analysis.

Antimony, Beryllium, Cobalt, and Thallium: HQs could not be calculated for these COCs because toxicity benchmark values were not available for avians. Any potential for risk associated with these COCs remains a data gap in the assessment.

Aluminum, Manganese, and Vanadium: The HQs based on the average concentration and the no-observable-adverse-effect level are less than unity. Lowest-observable-adverse-effect values were not available to calculate HQs. However, because LOAEL based HQs are less than NOAEL based HQs, the population of wading birds using Pond G is expected to be free from adverse effects from these COCs because the NOAEL based values are less than unity.

RED FOX

At that time the Tier II ERA Work Plan was being developed, a Red fox population existed on the facility and foxes had been observed to frequent the Pond G area. However, in recent years, 1999, 2000, and 2001, there were no reported Red fox sightings, no active dens, and no reported counts at scent stations (Speaker, personal communication). The last year for recorded Red fox sightings at TCAAP was 1998. It is speculated that an increased coyote population has resulted in a decrease in Red fox numbers (Speaker, personal communication). The risk assessment model was implemented for a Red fox population should there be one in the future. A separate model for Coyotes was not developed because the output for a Coyote would be very similar to that of Red fox, as they are both the top mammalian carnivores at the installation. Individuals from the installation's local population of Red fox are attracted to this aquatic site, indicating that some degree of co-occurrence in time and space does occur. These fox will contact the COCs while standing and possibly drinking from the Pond. These species will not be continuously exposed because they are expected to frequent the pond on a limited basis because of the availability of other quality habitat nearby and the size of their home range. The primary mechanism for the selected COCs to induce effects in these organisms is through ingestion of surface water while drinking and incidental ingestion of sediment that may be suspended in the water during drinking activities and that may be ingested while grooming their coats to free sediments adhered to their fur.

Risk estimates are provided in Exhibit 8-18. A synthesis is provided in the Risk Description subsection.

Exhibit 8-18. Red Fox Hazard Ratio Matrix for Pond G

	Fox	
	HQ1	HQ2
Aluminum	0	0
Antimony	0	0
Arsenic	0	0
Barium	0	0
Beryllium	0	ND
Cadmium	0	0
Chromium	0	0
Cobalt	0	0
Copper	0	0
Lead	0	0
Manganese	0	0

Selenium	0	0
Thallium	0	0
Vanadium	0	0
Aroclor 1248	0	0

ND: Not determined due to missing data needed for the HQ calculation

None of the calculated hazard quotients for the COCs are greater than unity. This indicates that adverse health effects are not expected for fox that may use Pond G as a drinking water source.

8.5.1.2 Risk Description

Based on the line of evidence used to estimate risk to wading birds at Pond G, adverse toxicological effects are not expected due to arsenic, barium, cadmium, chromium, copper, lead, selenium, aluminum, manganese, and vanadium. The possibility of effects from antimony, beryllium, cobalt, thallium, and Aroclor 1248 is an uncertainty because toxicity values were not available to calculate HQs.

Based on the same line of evidence, that adverse toxicological effects are not expected in fox that use Pond G as a drinking water source (see Exhibit 8-18). All HQs are less than unity.

The assessment endpoint for these entities (Water and sediment quality for wildlife using the pond) was not measured directly. 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 a hazard ratio. A ratio that exceeds unity triggers further careful consideration of the underlying scientific basis of the prediction.

8.6 ECOLOGICAL RISKS AT RICE CREEK

Rice Creek is a tributary of the Mississippi River, which flows through the northwest corner of the installation. The TCAAP accounts for approximately 2 percent of Rice Creek's 474-square-kilometer basin and is near the downstream end of the creek. The State of Minnesota classifies Rice Creek as a "Class 1C, 2BD, 3B water".

Rice Creek potentially receives groundwater from the perched unit 1 aquifer under site K, and was once thought to potentially receive volatile organic compounds (VOCs) from this source (with discharge between OU2-FS sampling locations RCK04SE and RCK05SE). Site K consists primarily of building 103 and surrounding property and occupies approximately 21 acres. The site has been used mainly for munitions manufacturing and assembly operations.

Surface water was analyzed for previously selected Tier II COPCs (aluminum, barium, cadmium, copper, mercury, and silver). Based on the collected data (see Section 4.3) and the implementation of the revised COC selection criteria (Section 5), none of these previous COPCs was retained as COCs. This means that no chemical stressors due to TCAAP-related activities have been identified.

The following assessment endpoint and risk hypothesis was evaluated.

> Assessment Endpoint – Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species

> Risk Hypothesis – Based upon the aquatic toxicity of aluminum; barium, cadmium, copper, silver, and mercury, if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

The COC selection process described in Section 5 resulted in no surface water COCs being selected for Rice Creek. Based on the results of the analytical data and the objective to compare Rice Creek downstream to Rice Creek upstream, while accounting for quarter differences and location differences, the statistical analysis indicated that no surface water COCs were significantly greater in downstream Rice Creek versus upstream Rice Creek. Although water quality benchmarks were exceeded during some sampling quarters for aluminum, barium, and mercury, the benchmarks were exceeded upstream of the Building 103 source area, as well as down stream. This indicates that if any adverse effects in fish, aquatic invertebrates, and algal species are occurring, they are not a result of TCAAP activities.

8.7 ECOLOGICAL RISKS AT SUNFISH LAKE

The selected surface water COCs evaluated in the risk characterization of Sunfish Lake are cadmium, mercury, and zinc. The selected sediment COCs are aluminum, chromium, lead, vanadium, and zinc. The COCs were selected based on the process presented in Section 5. These COCs were found at concentrations greater than background and toxicological benchmark concentrations.

During the development of the Tier II Work Plan in 1998 (USACHPPM 1999), it was suspected that Site H-1, which contains an old unpermitted landfill and also included metal debris scattered on the bottom of a portion of the Sunfish Lake sediments along the northcentral shore, could be the specific source of COPCs detected in the lake during the OU-2 Feasability Study. Site H-1 is no longer used and the upland soil areas of the site have been remediated in large part because they contained concentrations of metals (i.e., arsenic, antimony, copper, lead, and manganese) that posed excessive risks to human health (Stone & Webster 2002). These contaminated soils may have been transported via runoff into the lake, and metal contaminated leachate from the landfill may have migrated into the lake, causing metals contamination in the sediment and surface water. Section 6.5.1 provides further detail on the possible sources of COCs to Sunfish Lake.

Based on collected data reviewed in this report, the spatial distribution of the sediment and surface water COCs does not necessarily emanate from the banks along the northcentral shoreline (near site H-1). There is little spatial variability in the concentrations of surface water COCs around the lake and the same appears to be true for sediment aluminum, vanaduim, and zinc (see Appendix H). However, chromium and lead concentrations appear to be higher in the middle section of the lake adjacent to the unpermitted landfill. The temporal distribution of surface water concentrations (collected over a one-year period between 1999-2000) did show significant differences in the data during certain quarters of sampling. Also, analytical results from the second quarter of sampling (January 2000), showed all metals in surface water had slightly higher concentrations than during any other quarter.

Sunfish Lake COCs

Surface Water
Cadmium
Mercury
Zinc

Sediment
Aluminum
Chromium
Lead
Vanadium
Zinc

A revised conceptual site model was developed for Sunfish Lake to show specific COCs being evaluated and to indicate fate and transport pathways (see Figure B3-4).

8.7.1 Hypothesis 1: Effects in Fish, Aquatic Invertebrates, and Algae

The following assessment endpoint and risk hypothesis were evaluated.

Assessment Endpoint – Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species

Risk Hypothesis – Based upon the aquatic toxicity of surface water COCs (cadmium, mercury, and zinc); if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

8.7.1.1 Risk Estimates

Risks for these ecological entities were estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). For their entire life cycle, these species are continuously exposed to cadmium, mercury, and zinc; however, concentrations fluctuate by season and as a result of other changes in the microhabitat. Water samples were collected quarterly between 1999 and 2000 so that seasonal changes in concentration could be evaluated. The community of these organisms and their populations are not expected to be exposed to levels greater than exposure benchmarks. Exhibit 8-19 presents the quantitative risk estimates, which are then described below.

Exhibit 8-19. Sunfish Lake Risk Estimate for Fish, Aquatic Invertebrates, and Algae

COC	Hazard Quotient*	Portion Greater Than Chronic Benchmark†	Portion Greater Than Acute Benchmark†
Cadmium	<1	0 %	0 %
Mercury	<1	0 %	0 %
Zinc	<1	0 %	0 %

* HQ is calculated using central tendency concentration from Table A38 and chronic water quality benchmark from Appendix O and rounded to 1 significant figure.
† Percentages based on 12 sample locations.

Cadmium: Based on the risk estimate provided in Exhibit 8-19, adverse toxic effects in fish, aquatic invertebrates, and algal species due to cadmium are not expected to occur because the HQ is less than one and no concentrations exceeded the chronic water quality benchmark (1.1 ug/L). This water quality benchmark is a Minnesota Water Quality Standard that was calculated based on a water hardness of 100 mg/L. Additionally, no concentrations exceeded the acute water quality benchmark. These values are based on the protection of the aquatic community from adverse effects.

Mercury: Based on the risk estimate provided in Exhibit 8-19, adverse toxic effects in fish, aquatic invertebrates, and algal species due to mercury are not expected to occur because the HQ is less than one and no concentrations exceeded the chronic water quality benchmark (0.77 μg/L). This water quality benchmark is a NAWQC value. The chronic Minnesota Water Quality Standard was based on human health information and was not used. Additionally, no concentrations exceeded the acute water quality benchmark (2.4 μg/L). The acute benchmark was a Minnesota Water Quality Standard based on the protection of the aquatic community from adverse toxic effects.

Zinc: Based on the risk estimate provided in Exhibit 8-19, adverse toxic effects in fish, aquatic invertebrates, and algal species due to zinc are not expected to occur because the HQ is less than one and no concentrations exceeded the chronic water quality benchmark (106 μg/L). This water quality benchmark is a Minnesota Water Quality Standard that was calculated based on a water hardness of 100 mg/L. Additionally, no concentrations exceeded the acute water quality benchmark (117 μg/L). These benchmark values are based on the protection of the aquatic community from adverse effects.

8.7.1.2 Risk Description

Based on the line of evidence used to estimate risk to fish, aquatic invertebrates, and algal species at Sunfish Lake, adverse toxicological effects are not expected due to cadmium, mercury, and zinc. Additionally, the suspected source area has been remediated so future site-related impacts to the Lake are not expected.

The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. When available for a COC, the Minnesota Water Quality Standards (MWQSs) for Class 2B waters were used, which are based on the protection of several aquatic species from adverse toxic effects. Other effects-based benchmarks for fish, invertebrates, and algal species were then used when MWQSs are not available. Water quality criteria used in this assessment are linked to the assessment endpoint because survival, growth, and development of at least 95 percent of a population of aquatic organisms should be protected by a water quality standard (MPCA 2000a and USEPA 1999). However, by using the benchmarks we extrapolate from the laboratory to the field and between different species (i.e., between taxa). These extrapolations introduce uncertainty. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more, or less sensitive to the chemicals. In addition, the species tested in the laboratories may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar. 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).

A comparison of the surface water COC concentrations in Sunfish Lake to those in the reference site (Snail Lake) showed that cadmium, mercury, and zinc were statistically greater at Sunfish Lake compared to Snail (see Tables A39 and A41). However, concentrations were within the same order of magnitude between the two sites and below the water benchmarks. Additionally, the COCs in the soil at Site H-1 (the suspected source area) are not the same as the surface water COCs in Sunfish Lake. There may be several different explanations for this occurrence.

The spatial distribution of surface water COC concentrations at Sunfish Lake can be visualized by comparing Figure H1 (showing sampling locations) to Figure B4-8 through B4-10 (bar graph indicating the sample locations and the surface water benchmarks). Figures B4-8 through B4-10 indicate sample location by quarter where "a" = quarter one (September 1999), "b"= quarter two (January 2000), "c" = quarter three (April 2000), and "d" = quarter four (June 2000). No COCs exceeded their respective benchmarks.

8.7.2 Hypothesis 2: Effects in Benthic Organisms

The following assessment endpoint and risk hypothesis were evaluated.

Assessment Endpoint – Survival, growth, and reproduction of benthic organisms

Risk Hypothesis – Based upon the aquatic toxicity of sediment COCs (chromium, copper, lead, vanadium, and zinc); if sediment concentrations are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

8.7.2.1 Risk Estimates

The potential for adverse toxicological effects in these ecological entities was inferred by evaluating four lines of evidence (see Section 8.2.1). Risk estimates based on each of these four lines are provided in this subsection. A synthesis is provided in the Risk Description subsection.

These species are continuously exposed to aluminum, chromium, lead, vanadium, and zinc for their entire life cycle. However, concentrations fluctuate by season and as a result of other changes in the microhabitat. The community of these organisms and their populations are not expected to be exposed to levels greater than exposure benchmarks.

SITE CONCENTRATIONS COMPARED TO BENCHMARKS

Risk estimates for the first line of evidence, using effects-based benchmarks, were developed using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (a blend of techniques 3 and 4). Exhibit 8-20 presents the quantitative risk estimates for this line of evidence, which are then described below.

Exhibit 8-20. Sunfish Lake Risk Estimates for Benthic Organisms Comparing Site Concentrations to Benchmarks

COC	Hazard Quotient*	Portion Greater Than Low-effect Benchmark†	Portion Greater Than Moderate-effect Benchmark†
Aluminum	<1	0 %	0 %
Chromium	4	92 %	67 %
Lead	4	100 %	67 %
Vanadium	unavailable	unavailable	unavailable
Zinc	2	92 %	50 %

* HQ is calculated using central tendency concentration from Table A43 and low-effect sediment benchmark from Appendix O and rounded to 1 significant figure.
† Percentages based on 12 sample locations.

Aluminum: Based on the risk estimate provided in Exhibit 8-20, adverse toxic effects in benthic organisms due to aluminum are not expected to occur because the HQ is less than one and no concentrations exceeded the low-effect sediment quality benchmark (58,030 mg/kg). Additionally, no concentrations exceeded the moderate-effect benchmark (73,160 mg/kg). These values are found in Jones et al. (1997) that cites studies performed with the amphipod *Hyalella azteca* and the midge *Chironomus riparius* in the USEPA Assessment and Remediation of Contaminated Sediment (ARCS) project (USEPA 1996). The low-effect benchmark is the ARCS probable effect concentration (PEC) and the moderate-effect benchmark is the ARCS high no effect concentration (NEC).

Chromium: Based on the risk estimate provided in Exhibit 8-20, adverse toxic effects in benthic organisms due to chromium may be occurring because the HQ is 4 and 92 % of the samples are greater than the low-effect benchmark (37.3 mg/kg). Additionally, the moderate-effect benchmark (90 mg/kg) is exceeded in 67 % of the samples (see Figure B4-20). Six of the twelve (50%) samples even exceeded the severe effect benchmark (110 mg/kg (dry)) (Persaud, et al 1992), indicating that more serious effects such as potential elimination of most benthic organisms is likely to occur.

Lead: Based on the risk estimate provided in Exhibit 8-20, adverse toxic effects in benthic organisms due to lead may be occurring because the HQ is 4 and all measured concentrations at the site are greater than the low-effect benchmark (35 mg/kg) (see Figure B4-21). Nine of twelve (75%) sediment concentrations are greater than the moderate-effect benchmark (91.3 mg/kg), indicating the possibility for more moderate effects for a smaller portion of the community. However, no sediment lead concentration was above the Canadian severe-effect benchmark (250 mg/kg) (Persaud, et al 1992), indicating that most of the benthic organisms will not be eliminated as a result of sediment lead concentrations.

Vanadium: Risk estimates for vanadium could not be developed because no data were found during the literature search on sediment benchmarks for vanadium. This remains a data gap.

Zinc: Based on the risk estimate provided in Exhibit 8-20, adverse toxic effects in benthic organisms due to lead may be occurring because the HQ is 2 and 92 % of the samples had zinc concentrations greater than the low-effect benchmark (123 mg/kg). Six of twelve measured sediment zinc concentrations exceed the moderate-effect benchmark (315 mg/kg), indicating the possibility for moderate effects for a smaller portion of the community. However, no sediment zinc concentrations exceeded the Canadian severe effect benchmark (820 mg/kg) (Persaud, et al 1992), indicating that most of the benthic organisms will not be eliminated as a result of sediment zinc concentrations.

BENTHIC COMMUNITY EVALUATIONS

The second line of evidence that was used to evaluate risk to benthic organisms was benthic community evaluations that were conducted on two separate occasions. During the 1993 study (USACHPPM 1997), samples were collected at the inlet, offshore of the landfill, and at the lake outlet. Canadian geese were observed swimming, feeding, and nesting on and around the lake, and goose droppings were common along the shore. Other observations concluded that Sunfish Lake is a eutrophic lake bordering on over enrichment. The species of benthic organisms found in the lake were common for a eutrophic pond, and the dominant species were moderately to very tolerant of organic over enrichment. Diversity was characterized as moderately low to moderate. Species present either required an organic environment or were tolerant of adverse water quality conditions. The overall conclusion was that the lake shows signs of stress from overeutrophication, especially at the inlet.

During the 1995 benthic community diversity study (USACHPPM 1998b), four samples were collected from around Sunfish Lake. Five species of benthic organisms dominated the sediments of the lake, making up 74% of the individuals collected. All of the five dominant species are indicative of conditions of organic overenrichment. The fingernail clam *Sphaerium striatinum* and the phantom midge *Chaoborus punctipennis* were identified. The latter species migrates through the water column on a daily basis and may not be directly affected by sediment contaminants because its primary exposure medium is not likely to be interstitial water. The insect *Glyptotendipes lobiferus* occurred, although most of these individuals were found at location SFL0401 (see Figure H2). This species is common and often abundant in slow moving waters polluted by sewage waste. Another pollution tolerant species found at all Sunfish Lake locations was the snail *Physella gyrina*. The tube dwelling tubificid *Amphichaeta americanus* was found in Sunfish Lake as well and is moderately tolerant of organically enriched sediments. This survey indicates that there are benthic organisms present in Sunfish Lake and although they are the more pollution tolerant species, there is no evidence of risk to these organisms as a result of sediment COCs. The presence of more pollution tolerant species can be a result of increased organic matter and more anoxic conditions and other physical properties of the lake.

Taken together these two studies demonstrate that the benthic community in Sunfish Lake is dominated by pollution tolerant species indicating less than optimum conditions (i.e., degraded or enriched sediments.) However, based on the limitations of these studies, a causal link between the COCs and this condition has not been demonstrated nor ruled out.

For comparison purposes, a reference location (Snail Lake) was also sampled. The results from the 1993 Snail Lake sampling indicated that the six most abundant species found in the reference lake required an organic environment, and that the diversity was determined to be moderate (USACHPPM 1997). Snail Lake samples from the 1995 study (USACHPPM 1998b) showed that the macroinvertebrate species richness and diversity index were not different than that found in Sunfish Lake. The Snail Lake samples were dominated by oligochaeta (*Aulodrilus pluriseta* and *Amphichaeta Americans*), which are indicators of organically enriched environments.

SEDIMENT-METALS BIOAVAILABILTY STUDY

The third line of evidence used was information on the presence of acid volatile sulfide in the sediments at Sunfish Lake. The study measured the simultaneously extractable metal/acid volatile sulfide (SEM/AVS) ratio at 6 sediment locations in Sunfish Lake during September 1995 (USACHPPM 1998b). The study focused on the divalent transition metals presumed at the time to be potential COCs (i.e., cadmium, copper, nickel, lead, and zinc). In anoxic sediments, when the sum concentration of divalent metals (i.e., SEM) exceeds the binding of the acid volatile sulfide (AVS) pool, these metals are bioavailable to benthic organisms (Rand 1995). The SEM/AVS measurements taken in Sunfish Lake indicated that the divalent transition metals do not appear to be generally bioavailable because the AVS pool was of sufficient quantity to bind all available divalent metals. However, concentrations of AVS have been shown to vary by 2 orders of magnitude and can be correlated with changes in the temperature of the overlying water.

SEDIMENT TOXICITY TESTS

The fourth line of evidence is based on sediment toxicity testing performed with Sunfish Lake sediments (technique 1). For their entire life cycle, these species are continuously exposed to the COCs; however, concentrations fluctuate by season and as a result of other changes in the microhabitat. The community of these organisms and their populations may or may not be exposed to levels greater than exposure benchmarks, such that toxic effects are evident.

Exhibit 8-21 presents the toxicity test results and subsequent qualitative risk estimates.

Exhibit 8-21. Sunfish Lake Risk Estimates for Benthic Organisms Based on 1999 Sediment Toxicity Test Results for *Chironomus riparius* †

Sample Location *	14-Day Test		30-Day Test		
	Survival Rate (%)	Growth	Survival Rate (%)	Emergence Rate (%)	Mean Time to Emergence (d) ‡
Sunfish Lake					
W1-SF-SD-01	96	na	99	88	15.7
W1-SF-SD-02	98	na	83	79	14.6
W1-SF-SD-03	86	na	74	63	15.3
Snail Lake					
W1-SL-SD-01	95	na	88	79	14.3
W1-SL-SD-06	93	na	94	84	14.1
W1-SL-SD-09	94	na	84	81	13.6
Lab Control A	95	na	89	89	14.0
Lab Control B	88	na	90	88	15.2
Risk Estimate	• Cumulative survival rate was no different statistically than at reference lake • Growth was not measured because it was not feasible due to significant emergence of organisms (see text)		• Cumulative survival rate was no different statistically than at the reference lake • Cumulative emergence rate was no different statistically than at the reference lake • Mean time to emergence appears to be slightly delayed compared to the reference lake		

† The data presented here is summarized across replicates for each sample location. Appendix L presents the testing report.

‡ The mean time to emergence (days) was only calculated for those organisms that actually emerged.

na: Not available

* Sample locations refer to samples identified on Figure H1 (Sunfish Lake) and Figure J1 (Snail Lake).

** Indicates a statistically significant difference when compared to the pooled results from sample locations in the reference site (Snail Lake).

Based on the risk estimates provided in Exhibit 8-21, benthic organisms in Sunfish Lake appear to have acceptable survival rates, cumulative emergence rates, but slightly delayed emergence times when compared to the reference site (Snail Lake). The laboratory report is found in Appendix L. Cumulative survival rates in the 14- and 30-day tests were no different statistically than at the reference lake. The cumulative emergence rate in the 30-day test was no different statistically than at the reference lake. The mean time to emergence of *C. riparius* organisms were delayed compared to the reference site. The ASTM method (E 1706) for conducting sediment toxicity tests with *C. riparius* indicates that under optimal conditions larvae will pupate and emerge as adults after 15 to 21 days at 20°C. However, these tests were conducted at an average temperature of 23° C, which may have contributed to faster emergence times. Other determinants of emergence times include quantity of food provided and organic matter content of the sediment (De Haas et al, 2002). Nonetheless, the biological significance of this small difference in time to emergence between the lakes is not known.

Growth was not acceptably measured in the short-term tests (14-day). Growth is typically measured as length or dry weight of organisms that have not pupated or emerged as adults. Because the majority of larvae emerged during the 14–day test, sufficient larvae were not available to report a growth measurement.

8.7.2.2 Risk Description

The assessment endpoint (survival, growth, and reproduction of benthic organisms) was measured both directly and indirectly. The potential for adverse toxicological effects in benthic organisms at Sunfish Lake was evaluated using four lines of evidence. Risk estimates based on each of these four lines were provided in the previous subsection and a synthesis of the findings is provided here.

Elevated concentrations of chromium, lead, and zinc pose a potential risk for benthic organisms, however other data suggest that there may be sufficient AVS and other organic material to bind lead and zinc rendering them biologically unavailable. Available toxicity test results are consistent with a conclusion of little to no adverse effects; however, there are important limitations on those test results. Conclusions cannot be made about the potential risk posed by vanadium, if any, because vanadium-specific sediment quality benchmarks are unavailable, the SEM/AVS evaluation does not address the bioavailability of vanadium, and the toxicity tests were not designed to link specific COCs to test observations. Further details are provided in the following paragraphs.

Findings from two benthic community surveys indicate that the benthic community in Sunfish Lake is dominated by pollution tolerant species indicating less than optimum conditions (i.e., degraded or enriched sediments.) However, when compared to the reference location, it appears that both sites are affected by organic overenrichment and that Sunfish Lake is not different than the reference locations in terms of diversity and pollution tolerant species. Unfortunately, macroinvertebrate community surveys do not provide a direct link between COCs and adverse effects related to the risk hypothesis. However, they do provide information on community structure that can be used to indicate ecosystem health. The presence of more pollution tolerant species can be a result of increased organic matter, anoxic conditions and other physical properties of the lake, or historical contamination. A causal link between the COCs and this condition has not been demonstrated nor ruled out at Sunfish Lake.

The comparison of sediment concentrations to sediment quality benchmarks, suggests that ecological effects may be occurring due to elevated concentrations of chromium, lead, and zinc. Since promulgated sediment quality criteria were not available from either Minnesota or the USEPA, we used benchmarks derived from various reference sources. Of the sediment sampling locations, 67 percent of chromium samples, 75 percent of lead concentrations, and 50 percent of zinc concentrations were above the Smith et al. (1996) Probable Effect Level, (the level above which toxicity can be frequently observed). An important limitation of this line of evidence is that such effects-benchmarks do not take site-specific conditions under consideration.

Based on a preliminary bioavailability study using SEM/AVS measurements, lead and zinc did not appear to be generally bioavailable because the AVS pool was of sufficient quantity to bind all available divalent metals. However, concentrations of AVS have been shown to vary by 2 orders of magnitude and have been correlated with changes in the temperature of the overlying water. The optimum sulfate reduction rates (i.e., AVS formation) occur in the summer when water temperatures are the highest and oxygen consumption rates increase, forcing the sediment to become anoxic. Therefore, the lowest AVS concentrations—and by extension, the most vulnerable time for toxic effects in organisms—would be expected to occur in the colder winter months. The available site-specific data suggest agreement with this prediction. Data collected by the MPCA in March 1994 (Appendix C in USACHPPM 1998b) showed lower AVS concentrations than did the September 1995 USACHPPM data. Considering the uncertainties of the SEM/AVS technique, these results suggest that if any excess risk exists, then it may be characterized by increasing mobilization and bioavailability of these particular metals in winter and spring, followed by inhibited mobility and decreased bioavailability in the summer and fall. This scenario could be described as a seasonal "pulse" of potentially adverse exposures (USACHPPM 1998b). The quarterly water monitoring data from Sunfish Lake appear consistent with this hypothesis (samples collected between September 1999 – June 2000). The second quarter of sampling (January 2000) showed that cadmium and zinc had slightly higher surface water concentrations than during any other quarter (see Table A39). Without an SEM/AVS evaluation performed during the most vulnerable time of the year (i.e., winter), a hypothesized link between lead and zinc and this degraded condition cannot be fully explored.

Sediment toxicity tests of field-collected sediments were conducted using two standard test species: *Chironomus riparius* and *Hyalella azteca*. Based on the *Chironomus riparius* sediment toxicity tests, benthic organisms in Sunfish Lake would appear to have both acceptable survival and cumulative emergence rates, but slightly delayed emergence times when compared to the reference (Snail Lake). However the biological significance of this small difference in time to emergence between the lakes is not known. Unfortunately, growth was not acceptably measured in the short-term tests (14-day).

Unfortunately, the *Hyalella azteca* tests were found by USACHPPM to be invalid and were rejected for risk characterization purposes. Even though toxicity tests are considered stronger lines of evidence than chemistry and species composition data alone, testing in more than one species is usually necessary to adequately characterize risk. Additionally, there are other data quality concerns regarding the toxicity tests (see Appendix R). These uncertainties prohibit the risk characterization from placing sufficient weight on the tests to counter indications of ecological risk from the other, traditionally weaker lines of evidence.

The spatial distribution of risk within Sunfish Lake can be estimated by combining the sample-specific findings from each of the lines of evidence with the sample location maps provided in Figures H1 and H2. (Note: The Figures could not be accurately combined because GPS information was not available for all sampling locations.) The benthic community survey data do not demonstrate a noticeable gradient or difference in species composition across the lake. However, the 1993 survey did conclude that the area near the inlet (the southwest portion) showed the most signs of overeutrophication. In contrast to the above lines of evidence, data from the metals bioavailability study indicate that AVS concentrations appear to be sufficiently high across the whole lake (i.e., no heterogeneity) to bind a large portion of lead and zinc, potentially rendering them biologically unavailable and less toxic.

Chemistry data indicate that the middle of the lake near the old landfill shoreline has the highest concentrations of chromium and lead, while zinc and vanadium concentrations appear to be relatively homogenous throughout the lake. The *C. riparius* toxicity test results showed that the sampling location near the middle of the lake nearest to the old landfill shore-line (sample no. SF-SD03) produced a slightly lower emergence rate in the 30-day test and slightly lower survival rates in both the 14- and 30-day tests. However, the toxicity tests were not designed to identify differences between locations within the lake.

8.7.3 Hypothesis 3: Effects in Amphibians

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, development, and reproduction of amphibians
>
> Risk Hypothesis –Amphibian survival, development, and reproduction may also be at risk due to contaminants present in the surface water and sediment. (An adequate evaluation of potential problems for the amphibian community of the lake was a major data gap in the Tier I risk assessment).

8.7.3.1 Risk Estimates

The potential for adverse toxicological effects in amphibians was inferred by evaluating three lines of evidence; Frog Embryo Teratogenesis Assay-Xenopus (FETAX), species richness and relative abundance estimates, and immune function. Risk estimates based on each of these lines are provided in this subsection. A synthesis is provided in the Risk Description subsection.

It is expected that these organisms will be exposed during the embryo and juvenile portions of their life cycle. Additionally, exposure for adults is expected to be on an intermittent basis. However, sufficient information is not available to determine the size of the population exposed to the COCs.

FETAX RESULTS

Risk estimates for the toxicity test line of evidence, using surface water from Sunfish Lake for the FETAX, were developed based on technique 1. Exhibit 8-22 presents the quantitative risk estimates for this line of evidence, which are then described below.

Exhibit 8-22. Sunfish Lake Risk Estimates for Amphibians based on results of the 1999 FETAX Tests[†]

Site	Petri Dish ID	Initial Number of Embryos	Mortality Count	Mortality Rate (%)	Number Malformed	Malformation Rate (%)
Sunfish Lake	9	25	2	8	1	4.5
Sunfish Lake	10	25	1	4	3	12.5
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
- 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.

Based on the risk estimate provided in Exhibit 8-22, amphibians in Sunfish Lake appear to have acceptable occurrences of mortality and malformations when compared to the controls and the reference site. The report, containing supporting information on this conclusion, is found in Appendix M. The types and incidences of malformations observed in Sunfish Lake samples were not statistically different from controls or the reference site. Also, mortality was not found to be statistically different in Sunfish Lake samples than in the controls and reference samples. However, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected, and may be unacceptable for decision-making in some cases. The FETAX test was conducted on a composite surface water sample consisting of four samples from around the perimeter of Sunfish Lake, in order to obtain an overall characterization of the site. Therefore, effect(s) could not be linked to a potential "hot spot" or suspected source area in the lake. Additionally, surface water chemical analysis for metals was not conducted at the same time. FETAX was not conducted with sediments from Sunfish Lake due to the inherent difficulties in dealing with sediment in amphibian assays (i.e. with dejellied embryos in FETAX, fouling the water in adult exposure assays). The possible effects of sediment were considered such that water-soluble constituents would be expected to partition to the water and then to the organism. Therefore, exposure (dermal) to sediment COCs was assumed to be at equilibrium between water and sediment (pore water).

IMMUNE FUNCTION, ORGAN WEIGHTS, AND HISTOPATHOLOGY

For this line of evidence qualitative risk estimates are provided based on data collected on organ weights, two immune function markers, and histopathologic evaluations of important blood conditioning organs. Comparisons were made in adult frogs (*X. laevis*) exposed 14-days to surface water from Snail and Sunfish Lakes and also from a laboratory control solution used in the FETAX assay. The two markers, based on phagocyte response tests, are sensitive indicators of adverse immune responses. Organ weights can be useful indicators of toxicity. Appendix M presents the full results.

The following bullets present the risk estimates.

- The production of radical oxygen intermediates (ROI) in adult frogs exposed to Sunfish Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- The phagocytosis of foreign particles by splenic macrophages in adult frogs exposed to Sunfish Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- Ratios of organ weight to body weight in adult frogs exposed to Sunfish Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- No notable histological differences were found between sections of thymus, lung, kidney, and liver when examined using light microscopy for male and female frogs exposed to Sunfish Lake surface water, compared to Snail Lake and the laboratory control solution.

SPECIES RICHNESS AND RELATIVE ABUNDANCE

Risk estimates for this line of evidence were developed using visual encounter surveys, aural estimates, and trapping. Exhibit 8-23 presents the qualitative risk estimates for this line of evidence.

Exhibit 8-23. Sunfish Lake Risk Estimate for Amphibians Based on Species Richness

Site	Number of Species Seen or Heard in April '99, May '98, and July '98	Species Observed
Sunfish	5	*Bufo americanus, Rana clamitans, Rana pipiens, Ranid species, Pseudacris triseriata*
Snail	6	*Bufo americanus, Rana clamitans, Rana pipiens, Ranid species, Pseudacris triseriata, Hyla chrysoscelis*
Risk Estimate	• No statistically significant differences were found between the reference site and Sunfish Lake for the number of species observed. • It was not possible to accurately report relative abundance due to difficulties in accessing the sites because of the dense cattail mats. Species observed may have been listed because they were heard but not seen, so a definitive count was not possible.	

Based on the risk estimate provided in Exhibit 8-23, the diversity of amphibians in Sunfish Lake appears to be no different than that in the reference lake (Snail Lake). Due to habitat constraints and the relatively quick Anuran reaction reflex, many frogs detected by sight during the VES surveys could not be reliably identified. Therefore, more species may exist at each site than was originally reported. The study concluded that overall, relatively few amphibians were found.

8.7.3.2 Risk Description

The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species for FETAX, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. The potential for adverse toxicological effects in amphibians at Sunfish Lake is inferred using three lines of evidence. Risk estimates based on each of these lines were provided in the previous subsection. However, it is noted that there are uncertainties associated with the amphibian study including but not limited to; no multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, and laboratory tests used species not found at the site. These uncertainties along with others can be found in Table 6.1 in the amphibian report (see Appendix M). Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations, and site surface water was not analyzed for COC concentrations at the time of the amphibian study. Therefore, effects from specific surface water COCs were not reported, because it was not part of the study design.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of the assessment.

Laboratory data investigating embryo mortality (a sensitive life stage), developmental effects, and sensitive indicators of stress (immunocharacterization assays) suggest that surface water from Sunfish Lake is not toxic to frogs, yet sediment exposures were not specifically evaluated. However, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 41%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation (see Tables 5 and 6 in Appendix M). The field data were ambiguous, yet amphibians were noted at all locations.

Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to metals from Site H (the suspected source area) is not apparent. Additionally, data collected by Jannett (1997) and others (as discussed in Appendix M) appear to corroborate a conclusion that adverse impacts of chemical exposures to amphibians are unlikely.

8.7.4 Hypothesis 4: Effects in Wading Birds

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Reproductive potential and productivity of wading birds
>
> Risk Hypothesis – The survival, growth, and reproduction of wading birds may be at risk if they are excessively exposed to COCs in surface water, sediment, and residues in food from the lake.

8.7.4.1 Risk Estimates

The potential for adverse toxicological effects in wading birds was estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1).

Because Great Blue Herons nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). There is very little tree cover at, or surrounding Sunfish Lake. Therefore, colonies are not likely due to insufficient habitat. Bitterns are also not expected to nest at Sunfish Lake and due to their secretive habits they may not be found feeding or even visiting at the lake (See Figure H1). However, herons may forage at Sunfish Lake, and it can be expected that 1 to 3 herons may utilize Sunfish Lake from March through November. The frequency of exposure of wading birds is expected to be intermittent. Herons have been reported in the area from March through November but most leave the area for the winter (USGS 1995). However, some herons do nest in the vicinity, so for modeling purposes exposure duration is expected to be the life cycle of the wading bird. The portion of the local population exposed is not known. However, based on the available information it is possible that 1-3 wading birds may dominate at Sunfish Lake during any given season.

Risk estimates are provided in Exhibit 8-24. A synthesis is provided in the Risk Description subsection.

Exhibit 8-24. Wading Bird Hazard Ratio Matrix for Sunfish Lake[†]

		Heron (no sediment ingestion)		**Heron** (10% sediment ingestion)	
		HQ1	HQ2	HQ1	HQ2
Aluminum:					
	Maximum	0	nd	0	ND
	Central Tendency	0	nd	0	ND
Cadmium:					
	Maximum	0	0	0	0
	Central Tendency	0	0	0	0
Chromium:					
	Maximum	0	0	0	0
	Central Tendency	0	0	0	0
Lead:					
	Maximum	0	0	0.2	0
	Central Tendency	0	0	0.1	0
Mercury:					
	Maximum	0.02	0.0	0.02	0.0
	Central Tendency	0.01	0.0	0.01	0.0
Vanadium:					
	Maximum	0	ND	0	ND
	Central Tendency	0	ND	0	ND
Zinc:					
	Maximum	0	0	0	0
	Central Tendency	0	0	0	0

[†] **Exposure to the heron was modeled using zero percent sediment ingestion and a worst-case scenario ingestion of 10%.** See Appendix N Section N3.3

None of the calculated hazard quotients for the COCs are greater than unity. This indicates that adverse health effects are not expected for wading birds that frequent Sunfish Lake. Estimates are provided for two exposure scenarios: one with and one without incidental sediment ingestion. No sediment ingestion data for wading birds was found in the literature. However, it is possible that wading birds may be ingesting sediment to some degree. It was therefore decided to produce hypothetical values of sediment ingestion rates across a range of portions of sediment in the diet (1, 5, and 10 percent of the diet), in order to support a sensitivity analysis.

8.7.4.2 Risk Description

Based on the line of evidence used to estimate risk to wading birds at Sunfish Lake, adverse toxicological effects are possible, albeit unlikely due to exposure to aluminum, cadmium, chromium, lead, mercury, vanadium, and zinc.

The assessment endpoint for wading birds (reproductive potential and productivity of wading birds) was not measured directly. 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—ratios between the exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. A ratio that exceeds unity triggers further careful consideration of the underlying scientific basis of the prediction.

8.7.5 Hypothesis 5: Effects in Aquatic Mammals

The following assessment endpoint and risk hypothesis were evaluated.

> <u>Assessment Endpoint</u> – Reproductive potential and productivity of aquatic mammals
>
> <u>Risk Hypothesis</u> – The health of aquatic mammals, such as the muskrat, may be impaired if they are excessively exposed to sediments and vegetation containing elevated levels of aluminum and vanadium. In addition, aquatic mammals may also be exposed to chromium, lead, and zinc.

8.7.5.1 Risk Estimates

The potential for adverse toxicological effects in aquatic mammals was estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Since the initiation of the risk assessment, the muskrat has been the primary mammal of concern. However, in addition to evaluating the risk of COC exposure to the muskrat, risk to the mink was also evaluated, but only for exposure to mercury. It is recognized that other metals and possibly organics could be present in dietary items of the mink; however, the use of the muskrat to evaluate risk to aquatic mammals provides an estimator for risk to COC other than mercury. Only mercury was evaluated for the mink because there was concern that mink would be at more risk than the muskrat to biomagnification of mercury mainly through consumption of fish.

MUSKRAT

During past field sampling activities, several to many muskrat dens were observed in the eastern portion of the lake. Based on the availability of preferred muskrat habitat at Sunfish Lake (i.e., cattails in the eastern portions), it can be expected that Sunfish Lake is capable of supporting 20 adult muskrats. This is a very conservative estimate; and it assumes that there is ample food and that the whole of the northeastern portion of the lake is inhabitable. The frequency of exposure for muskrats to these chemicals is expected to be continuous. For those muskrats that are exposed, it is expected to be for the majority of their life cycle. Based on the current information, the portion of the muskrat population exposed at Sunfish Lake is unknown.

Risk estimates are provided in Exhibit 8-25. A synthesis is provided in the Risk Description subsection.

Exhibit 8-25. Aquatic Mammal Hazard Ratio Matrix for Sunfish Lake

	Muskrat		**Mink**	
Aluminum as Al(Cl_3):	HQ1	HQ2	HQ1	HQ2
Maximum	26	5	NM	NM
Central Tendency	19	4	NM	NM
Aluminum as Al($(OH)_3$)				
Maximum	4	ND	NM	NM
Central Tendency	3	ND	NM	NM
Cadmium:				
Maximum	0.1	0	NM	NM
Central Tendency	0	0	NM	NM
Chromium:				
Maximum	3	1	NM	NM
Central Tendency	2	0.7	NM	NM
Lead:				

Maximum	0	0	NM	NM
Central Tendency	0	0	NM	NM
Mercury:				
Maximum	0.1	0	1	0
Central Tendency	0.1	0	1	0
Vanadium:				
Maximum	1	0.3	NM	NM
Central Tendency	1	0.2	NM	NM
Zinc:				
Maximum	0.1	0.1	NM	NM
Central Tendency	0.1	0	NM	NM

Na: Not applicable
NM: Not modeled

Cadmium, Lead, Mercury, Vanadium, and Zinc: None of the calculated hazard quotients for these COCs are greater than unity. This indicates that adverse health effects due to these COCs are not expected for mammals at Sunfish Lake.

Aluminum: All four HQs for the muskrat exceed unity. These HQs were calculated using a TRV based on aluminum chloride ($AlCl_3$). However, there is uncertainty as to what form of aluminum is predominant in the sediments at the lake. Based on site-specific information and professional judgment, it is thought that the form of aluminum found in the waterbodies at TCAAP is not solely aluminum chloride, but rather a combination of aluminum complexes such as aluminum hydroxide ($Al(OH)_3$), aluminum bicarbonate, or complexes with other ligands. This leads us to believe that the current TRV (based on aluminum chloride) may be over conservative. The toxicity of aluminum is intimately associated with pH in that the metal is soluble and biologically available in acidic (pH <5.5) soils and waters but relatively innocuous in pH conditions between 5.5 – 7.5 (Sparling and Lowe 1996). Also, concentrations of dissolved aluminum in water vary not only with pH but also with the humic-derived acid content of the water (ATSDR 1999). The TRV value for wildlife is based on a laboratory study in which $AlCl_3$ was administered to mice. Alternatively, two oral exposure studies of aluminum hydroxide discussed in the Aluminum profile (ATSDR 1999), one to rats and one to mice were considered. The study on mice (Domingo et al 1989) was a chronic study in which mated female mice were given by gavage daily doses of $Al(OH)_3$ on gestation days 6 through 15. The study found no evidence of maternal toxicity, embryo/fetal toxicity or teratogenicity even at levels as high as 266 mg Al/kg/day as aluminum hydroxide. The other study (Greger and Donnaubauer 1986), was an intermediate exposure (subchronic) study in which no changes in body weight were found in male Sprague-Dawley rats that ingested up to 158 mg Al/kg/day as aluminum hydroxide in the diet for 16 days. When developing a TRV, studies of chronic duration are more favorable. Based on the Domingo et al (1989) study, a TRV could be generated following the USACHPPM (2000) protocol (see appendix O). The chronic NOAEL TRV derived from the Domingo et al (1989) study would be 26.6 mg/kg/d aluminum. It is more appropriate to develop a TRV based on the form of the chemical that the organisms will be exposed to and an exposure duration that would be more representative of field conditions. Using the $Al(OH)_3$ based NOAEL TRV of 26.6 mg/kg/d could decrease the HQ values by two orders of magnitude for the maximum concentration HQ (26 mg/kg-day for $AlCl_3$ to 0.4 mg/kg-day), resulting in HQs less than unity. See Exhibit 8-25. This information may indicate that muskrats are not at risk of adverse effects due to aluminum at Sunfish Lake.

Chromium: Two of the four HQs for the muskrat exceed unity. The HQ of 3 (based on the maximum site concentrations and an estimate of the no-adverse-effect-level) indicates that a maximally exposed muskrat in the population may experience an exposure 3-times greater than that considered to be safe. The HQ of

2 (based on central tendency concentrations and an estimate of the no-adverse-effect-level) indicates that on average, individual muskrats in the population may experience exposures 2-times greater than that considered to be safe. However, because the HQ ratios based on the lowest-observable-adverse effect level are not greater than unity (1 and 0.7) estimated exposures in the muskrat population are less than exposures known to be associated with adverse health effects.

These HQs were based on the most toxic form of chromium (Cr^{6+}) and may not accurately reflect thresholds of toxicity in Sunfish Lake. Chromium can exist in oxidation states ranging from 2^+ to 6^+. However, only 3^+ and 6^+ are normally found within the range of pH and redox potentials common in environmental systems, and of these, Cr 3^+ is generally considered to be the more stable form (Losi et al 1994). Chromium III is known to form numerous complexes, with both organic and inorganic ligands. Also, under natural conditions, Cr 6^+ is easily reduced by living matter to Cr 3^+ (WHO 1988). At pH values from 5.5 to 12, Cr 3^+ precipitates almost completely as $Cr(OH)_3$ resulting in low aqueous concentrations of Cr 3^+ (Losi et al 1994). Unless solubilized by significant organic complexation, Cr 3^+ can be expected to remain in a solid form in most environmental systems and can be physically immobilized within the soil matrix or be subject to sedimentation in aqueous environments (Losi et al. 1994).

TCAAP was designed as a facility for the fabrication, loading and proof testing of small arms ammunition. As such, some of the activities and the wastes they produced were characteristic of a metal finishing operation. However, there was no wastewater discharge to Sunfish Lake. Surface water flows to the south in the vicinity of Site H and probably flows toward Sunfish Lake (Montgomery Watson 1997). A considerable portion of the area around Sunfish Lake has been used as a dump (Site H) containing industrial sludge, paint residue, incineration ash and solvents (Stone & Webster 2001). Chromium is known to be a component of a majority of the above mentioned dump contents (Allen et al 1995). Aerial photographs show that an entire northern bay-like area of the lake was filled in during WWII (USAEHA 1991). Solvents, incinerator clinkers and ash from burning cages were some of the materials dumped at Site H. It is also possible that contaminated storm water runoff from Site H flowed into Sunfish lake, thus increasing the concentrations of metals in the lake. If any waste from the production of ammunition was dumped around Sunfish Lake it was most likely cadmium dichromate or calcium chromate. These two forms of chromium are as Cr 6^+ and are used for metal finishing and as metal primers and corrosion inhibitors, respectively (Losi et al 1994). However, given the potential for Cr 6^+ to be reduced to Cr 3^+ (Allen et al 1995; Allen 1995; Kabata-Pendias and Pendias 1984), it is most likely that the majority of chromium currently in Sunfish Lake is in the form of Cr 3^+. The mammal toxicity values for this form of chromium indicate that Cr3+ is much less toxic than Cr6+.

Taken together, this indicated that a substantial risk of adverse effects due to chromium is most likely not present.

MINK

Signs of mink activity at the lake have been observed. (e.g., tracks in the winter, etc.). Sunfish Lake is approximately 5.7 ha is size and assuming that there is sufficient habitat and food, then it can be expected that Sunfish Lake may support 1 female mink. The home range of female minks has been estimated to be 7.8 ha in heavily vegetated areas and 20.4 ha in sparsely vegetated areas (USEPA 1993). One to two male mink may spend time between Sunfish Lake and Marsden Lake, most likely making their dens at Marsden Lake. Of the Mink that may be exposed, they are expected to either visit or locate their den at the site, indicating that the exposure duration could vary to a large degree. It can be assumed that for a large portion of their life cycle they will be exposed while at Sunfish Lake.

Risk estimates are provided in Exhibit 8-26. A synthesis is provided in the Risk Description subsection.

Exhibit 8-26. Mink Hazard Ratio Matrix for Sunfish Lake

	Mink	
Mercury:	HQ1	HQ2
Maximum	1	0
Central Tendency	1	0

8.7.5.2 Risk Description

Based on the line of evidence used to estimate risk at Sunfish Lake, adverse toxicological effects due to cadmium, lead, mercury, vanadium, and zinc are not expected in aquatic mammals because the HQ ratios are not greater than unity, although there is a possibility that adverse effects may occur (see Exhibit 8-25). Population-level effects are not expected for chromium exposures because the average exposure in the population is less than exposures known to be associated with adverse health effects. However, for aluminum, it appears that the form of aluminum found at the site may influence the outcome. Assuming that mammals are exposed to aluminum chloride, then the data suggest that population-level adverse effects are possible because the average exposure would be 4-times greater than exposures known to be associated with adverse health effects. On the other hand, assuming they are exposed to aluminum hydroxide, then the data suggest that population-level adverse effects may not be expected given the magnitude of the available HQ ratios associated with "safe" exposures. (Note: exposure levels known to be associated with effects for this form are not known). Based on supporting information provided earlier in the report regarding the expected form of aluminum at the site, the aluminum hydroxide-based conclusion is considered to be more site-specific. However, there remains uncertainty as to whether adverse effects are occurring due to limitations in the evidence.

The assessment endpoint for aquatic mammals was not measured directly. 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 careful consideration of the underlying scientific basis of the prediction.

8.8 ECOLOGICAL RISKS AT ROUND LAKE (CURRENT SCENARIO)

The selected surface water COCs evaluated in the risk characterization of Round Lake are barium, cadmium, and zinc. The selected sediment COCs are cadmium, chromium, copper, lead, silver, vanadium, zinc, and PCBs. The COCs were selected based on the process presented in Section 5. These COCs were found at concentrations greater than background and toxicological benchmark concentrations.

It was clear during the development of the Tier II Work Plan that 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. In 1969, the tenant of the building noticed that many of the floor drains were still connected to the storm sewer and has since remedied the situation. Due to the nature of the production process used at building 502, PCBs and metals were expected to be the main COPCs.

Round Lake COCs: Current Scenario

Surface Water
Barium
Cadmium
Zinc

Sediment
Cadmium
Chromium
Copper
Lead
Silver
Vanadium
Zinc
PCBs

The spatial distribution of the COCs emanates from the northeastern banks, where TCAAP drain systems historically released water and other materials. The COC concentrations decrease with increasing distance away from these areas and the sample data support this. Lead in the sediment is an exception, having two high concentration locations (RL-18 and RL-19) not in the northern portion of the lake (see Figure I2). For PCBs, it is important to note that while previous sampling showed detections in the northern littoral zone, the most recent and extensive sampling effort (1992) found no valid detections of PCBs in the sediments of the lake at the locations of the historical detections. Further information on the PCBs sampling and evaluation can be found in Appendix P. The temporal distribution of barium, cadmium, and zinc in surface water did show significant variability during certain quarters of recent sampling (see Table A48). Also, analytical results from the second quarter of sampling (January 2000), showed that all metals (except mercury) in surface water had slightly higher concentrations than during any other quarter.

The revised conceptual models were introduced in Section 6.6.3. The revised diagram for Round Lake can be referred to for information pertaining to the fate and transport of COCs (see Figure B3-5). Risk estimates and descriptions are provided for each hypothesis described in the following subsections.

8.8.1 Hypothesis 1: Effects in Fish, Aquatic Invertebrates, and Algae

The following assessment endpoint and risk hypotheses were evaluated. Risk hypothesis 1 was presented in Problem Formulation, while hypothesis 2 was developed from the PCB evaluation presented in Appendix P.

> Assessment Endpoint – Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species
>
> Risk Hypothesis (1)– Based upon the aquatic toxicity of surface water COCs (barium, cadmium, and zinc); if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

Risk Hypothesis (2) – Based upon exposure to PCBs in food and sediments, if PCBs are present above acceptable levels, then developmental and reproductive problems may occur in fish (e.g. black bullheads).

8.8.1.1 Risk Estimates

Risks for these ecological entities were estimated using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (see Section 8.1.1). Exhibit 8-27 presents the quantitative risk estimates, which are then described below. For their entire life cycle, these species are continuously exposed to the surface water COCs; however, concentrations fluctuate by season and as a result of other changes in the microhabitat. Water samples were collected quarterly (during 1999 and 2000) so that seasonal changes in concentration could be evaluated. Historical fish sample data from two USFWS studies, presented in Appendix P, were used for residue-based calculations.

Exhibit 8-27. Round Lake (Current Scenario) Risk Estimate for Fish, Aquatic Invertebrates, and Algae

COC	Hazard Quotient *	Portion Greater Than Chronic Benchmark †	Portion Greater Than Acute Benchmark †
Water-based benchmark comparisons			
Barium	20	100 %	0 %
Cadmium	<1	0 %	0 %
Zinc	<1	0 %	0 %
Residue-based comparisons (for fish only)			
PCBs	<1	0 %	0 %

* For Ba, Cd, and Zn, the HQ is calculated using central tendency surface water concentration from Table A38 and chronic water quality benchmarks from Appendix O. For PCBs, the HQ is calculated using the average tissue residue wet weight concentration found in black bullhead from Appendix P (0.2 mg/kg) and the lowest, no effect level based on sediment contaminants as found in the Environmental Residue-Effects Database (10 mg/kg). All values are rounded to 1 significant figure or shown as less than one.

† For Ba, Cd, and Zn, percentages are based on 12 sample locations from the 1999-2000 sampling. For PCBs, percentages are based on 25 fish composited into 5 samples (5 fish each) from the 1981 and 1988 USFWS studies (see Appendix P).

Na: Not applicable.

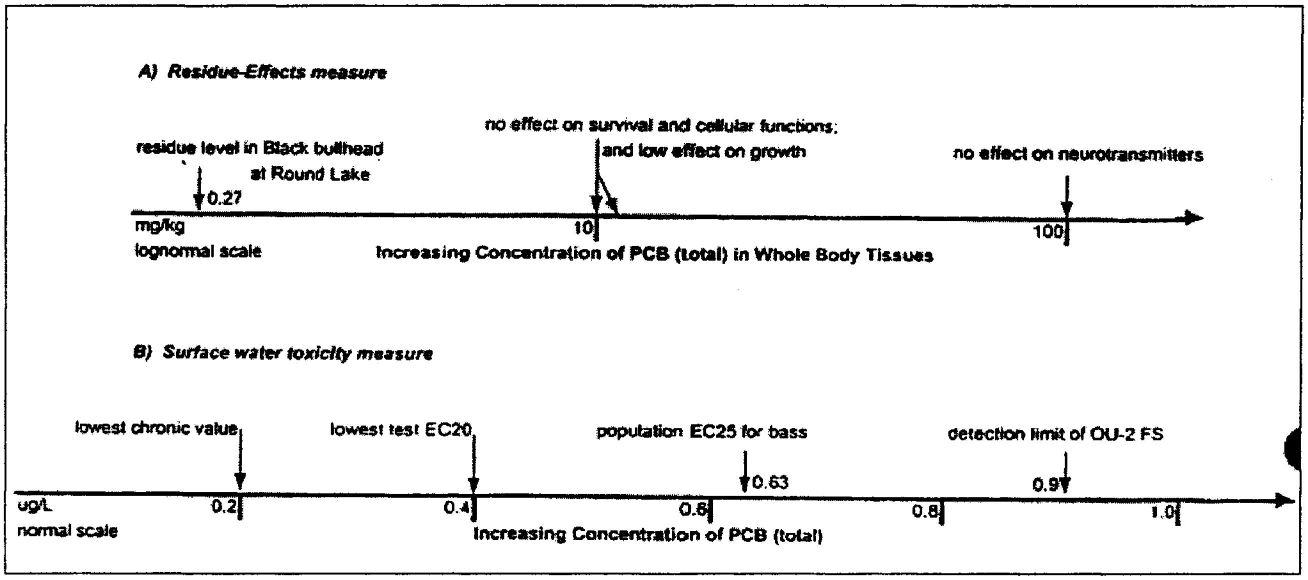
Barium: Based on the risk estimate provided in Exhibit 8-27, adverse toxic effects in fish, aquatic invertebrates, and algal species due to barium may be occurring because the HQ is 20 and 100% of the samples are greater than the chronic water quality benchmark (3.8 μg/L). However, no samples exceeded the acute benchmark (110 μg/L), indicating that more serious effects are not expected.

Cadmium: Based on the risk estimate provided in Exhibit 8-27, adverse toxic effects in fish, aquatic invertebrates, and algal species due to cadmium are not expected to occur because the HQ is less than 1 and none of the samples were greater than the chronic water quality benchmark (1.1 μg/L), which is hardness dependent. The chronic benchmark is a Minnesota Water Quality Standard for Class 2B waters and was calculated based on a hardness of 100 mg/L. The chronic value is estimated to be protective of at least 95% of a population of aquatic organisms (MPCA 2000). The acute water quality benchmark was also not exceeded, and is a Minnesota Water Quality Standard for Class 2B waters that was calculated based on a hardness of 100 mg/L.

Zinc: Based on the risk estimate provided in Exhibit 8-27, adverse toxic effects in fish, aquatic invertebrates, and algal species due to zinc are not expected to occur because the HQ is less than 1 and none of the samples were greater than the chronic water quality benchmark (106 μg/L), which is hardness dependent. The chronic benchmark is a Minnesota Water Quality Standard for Class 2B waters and was calculated based on a hardness of 100 mg/L. The chronic value is estimated to be protective of at least 95% of a population of aquatic organisms (MPCA 2000). The acute water quality benchmark was also not exceeded, and is a Minnesota Water Quality Standard for Class 2B waters that was calculated based on a hardness of 100 mg/L.

PCBs: Based on the risk estimate provided in Exhibit 8-27, adverse toxic effects in fish due to PCB exposures from water, sediment, and food are not expected to occur because the HQ is less than 1 and none of the samples were greater than the lowest residue benchmark (10 mg/kg) found in the US Army Corps of Engineers' Environmental Residue-Effects Database (ERED). The benchmark is based on survival and growth endpoints. This estimate can be viewed in context with other residue benchmarks in Exhibit 8-28 A. The Environmental Residue-Effects Database (ERED) was searched to determine a PCB concentration in fish tissue that is linked to an observable biological effect. The value from the ERED was then compared to the maximum composite PCB concentration detected in black bullhead tissue collected from Round Lake by the USFWS (see Appendix P). The composite fish tissue PCB concentrations (total PCBs) detected in Round Lake bullhead (0.16 – 0.27 mg/kg wwt) are much less than tissue concentrations known to be associated with no survival impact and low effects on growth in channel catfish (10 mg/kg wwt). Due to a lack of necessary data (i.e., reproductive endpoints), this is not a direct measure of population or reproductive health risks. However, it is a persuasive argument for low risk for major health effects (e.g., reduced survival and growth) because it uses actual PCB levels in fish, rather than indirect measures of exposure.

Exhibit 8-28. Supplemental Round Lake (Current) Risk Estimate for Fish on Exposure to PCBs



Based on the diagram provided in Exhibit 8-28 B for the surface water toxicity measure, if PCBs are present at water concentrations between the detection limit and benchmark, then it could be argued that adverse health effects might exist for the fish population. However, the detection limit for PCBs in water (0.9 μg/L) was greater than the toxicity benchmarks provided and all analytical results were less than the detection limit. The benchmarks provided in the exhibit are based on direct water exposures in the laboratory, but it should be noted that for fish the primary exposure route to PCBs is via the sediment/food pathway and not directly from the water.

8.8.1.2 Risk Description

Based on the line of evidence used to estimate risk to fish, aquatic invertebrates, and algae at Round Lake under the current scenario, it appears that adverse toxicological effects are possible due to elevated barium concentrations in the surface water, and some effects to fish may be possible from PCB concentrations in the northern littoral sediments. However, there remains uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. The barium and PCB risk estimates contain a high degree of uncertainty due to the lack of a sufficient toxicity database. For barium only one study was found regarding aquatic toxicity, and for PCBs the tissue concentration associated with adverse biological effects was based on one study with channel catfish. Though PCBs were not detected in the surface water using a detection limit greater than the toxicity benchmark, this uncertainty is of little relevance because sediment and food-based exposure routes are more important for PCB exposure. Additionally, the primary source of COCs has been eliminated; so further contamination to the lake from TCAAP is not expected. However, other sources such as auto emissions and roadsalt from the nearby highways may still be contributing to problems at Round Lake.

The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water and fish concentration data to toxicity benchmarks. The Minnesota Water Quality Standards for Class 2B waters were used for cadmium and zinc, which are based on the protection of several aquatic species from adverse toxic effects. For barium, other effects-based benchmarks for fish, invertebrates, and algal species were then used because a Minnesota value was not available. The chronic barium benchmark was derived using USEPA Tier II criteria (based mostly on acute data) and does not have as robust a data set as a NAWQC would have. Regardless of the source of the benchmark, the water quality benchmarks used in this assessment are linked to the assessment endpoint because they were developed to be protective of survival, growth, and development of at least 95 percent of a population of aquatic organisms (MPCA 2000a and USEPA 1999).

Uncertainty exists in the estimate of the health risk of developmental and reproductive problems for the lake's black bullhead. Given the current information and analysis, the best estimate of the threat is that it is low. The threat for these fish is limited to the northern littoral areas, so in this sense the extent of the threat is small. Given the PCB residues found in fish tissue, it can be predicted that growth and survival are not impacted. This is corroborated by the health and sizes of the fish collected by the USFWS for tissue analysis (i.e. full grown adults). No residue-based reproductive endpoints exist, however, and a definitive conclusion about reproduction using this measure cannot be made.

By using the benchmarks we extrapolate from the laboratory to the field and between different species (i.e., between taxa). These extrapolations introduce uncertainty. When we extrapolate laboratory data to the field, we do not consider environmental and ecological processes that may render the organisms more, or less sensitive to the chemicals. In addition, the species tested in the laboratories may not be similar to the species that occur at the sites. However, we assume that the effect levels will be similar. 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).

A comparison of the surface water barium concentration in Round Lake to that in Snail Lake showed statistically greater concentrations at Round Lake compared to Snail (see Tables A45 and A48).

However, it should be noted that exposure does not necessarily indicate effect, and that the central tendency concentration for barium at the reference site was itself 12.3 times higher than the chronic benchmark.

The degree to which regional conditions contribute to the PCB load in Round Lake remains unknown. During the OU-2 feasibility study and Tier I risk assessment it was agreed that sediment concentrations of total PCBs less than 0.05 mg/kg would represent a regional condition and not TCAAP-related contamination. Of the 17 surface sediment samples analyzed for PCBs, most results were < 0.05 ug/g (i.e., not detected) with a mean of 2.3 mg/kg and maximum of 13 mg/kg (Appendix P). There is little useful information on total PCB concentrations in fish tissue for reference lakes even though bullhead fillet (skin-off) data across the state are available. These data can be accessed in the Minnesota Fish Consumption Advisory Fish Tissue Reside Database. Unfortunately, whole body concentrations from this database are not available for comparison. This is problematic because the largest portion of the PCB burden should be found in tissues other than the fillet. An empirically-based algorithm which estimates whole body PCB residues from fillet data in bottom-feeder fish like bullhead was not found. Given the fillet data provided by the MPCA (see Appendix P), if the whole body concentration were 4-times this fillet concentration it would be approximately 0.07 mg/kg. Yet, even this concentration would not indicate that potential health risk from PCBs at Round Lake is much greater than that associated with background conditions.

The spatial distribution of surface water COC concentrations can be visualized by comparing Figure I1 (showing sampling locations and source areas) to Figures B4-11 through B4-13 (bar graphs indicating the sample locations and the surface water benchmarks for each COC). Figures B4-11 through B4-13 indicate sample location by quarter where "a" = quarter one (September 1999), "b"= quarter two (January 2000), "c" = quarter three (April 2000), and "d" = quarter four (June 2000). The samples were collected in the northern portion of the lake, where the majority of high concentrations of sediment metals were previously affirmed. During each sampling event concentrations across sites were very similar. Only locations RL-04 and RL-05 had slightly higher concentrations of metals compared to the other locations during quarters two and three. TCAAP had two outfalls to Round Lake, one was a storm run-off and cooling water outlet (from Line E) nearest to the intersection of Highways 10 and 96, and the other was an emergency gravity outlet for sewage from the plant (by-pass or standby system) that was located slightly to the west of the cooling water outfall. The by-pass line was to be used only in the event that the existing forcemains or facilities for pumping to the Minneapolis sewer system would fail simultaneously (Minnesota Department of Health, 1964). During field investigations in the 1990's, no evidence of the location of the by-pass outfall could be found. Site I cooling water was directed into the storm water outfall and was known to contain cadmium, zinc, chromium, and cyanide (ANL 1991). Locations RL-04 and RL-05 appear slightly closer to the location of where the by-pass outfall was expected to be located. However, this does not indicate that the by-pass may have been the source of COCs to these two locations.

Though the spatial distribution of surface water COCs appears to be relatively uniform, there was a marked temporal pattern to barium, cadmium, and zinc concentrations (see Figures B4-11 to B4-13). Quarters two (winter) and three (spring) showed higher concentrations among the sampling locations compared to quarters one and four. Project reports did not indicate any deviation in sampling procedure or analysis. Although we have no direct evidence that the trend in quarterly surface water concentrations being higher in these two quarters is related to AVS binding, it appears possible that the increase in surface water concentrations of barium, cadmium, and zinc during quarters two (January 2000) and three (April 2000) may be attributable to AVS binding of surface water metals. During the colder months, less surface water metals bind to AVS in sediment, thus resulting in higher metals concentrations in the surface water. However, in the warmer months when the sediment tends to be more anoxic, divalent metals in the surface water bind to AVS in the sediment and thus the metals concentrations in the surface water are decreased. This pattern is consistent with one conclusion from the USACHPPM *Bioavailability of Sediment-metals in Round and Sunfish Lakes Report* (1998). That report indicated that there would be increased "mobilization and bioavailability of these particular metals [cadmium and zinc, in this case] in winter and spring, followed by inhibited mobility and decreased bioavailability in the summer and fall. This scenario could be described as a seasonal 'pulse' of potentially adverse exposures."

8.8.2 Hypothesis 2: Effects in Benthic Organisms

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, growth, and reproduction of benthic organisms
>
> Risk Hypothesis – If concentrations of sediment COCs (cadmium, chromium, copper, lead, silver, vanadium, zinc, and PCBs in the current scenario) are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

8.8.2.1 Risk Estimates

For the current scenario, the potential for adverse toxicological effects in these ecological entities was inferred by evaluating four lines of evidence (see Section 8.2.1). Risk estimates based on each of these four lines are provided in this subsection. A synthesis and comparison of these lines of evidence is provided in the Risk Description subsection. The frequency of exposure of benthic organisms is expected to be continuous. However, concentrations of COCs fluctuate seasonally. The duration of exposure is expected to be the entire life-cycle for most; non-emergent (i.e., juvenile) forms of some invertebrates. Based on the current information, the portion of the local population exposed is not known.

SITE CONCENTRATIONS COMPARED TO BENCHMARKS

Risk estimates for the first line of evidence, using effects-based benchmarks, were developed using comparisons of single-point estimates of exposure and effect that highlight the variability in the collected site data (a blend of techniques 3 and 4). Exhibit 8-29 presents the quantitative risk estimates for this line of evidence, which are then described below.

Exhibit 8-29. Round Lake Risk Estimates for Benthic Organisms Using the First Line of Evidence

COC	Hazard Quotient*	Portion Greater Than Low-effect Benchmark†	Portion Greater Than Moderate-effect Benchmark†
Cadmium	<1	28 %	22 %
Chromium	2	56 %	22 %
Copper	7	72 %	44 %
Lead	<1	17 %	17 %
Silver	31	22 %	11 %
Zinc	3	61 %	44 %
PCBs (1992)	<1	0 %	0 %
PCBs (1987 & 1990)	70	63 %	50 %

* For the inorganics, the HQ is calculated using central tendency concentration from Tables A50 and A51 and low-effect sediment benchmark from Appendix O. All values are rounded to 1 significant figure or shown as less than one.

† Percentages based on 18 sample locations.

Cadmium: Based on the risk estimate provided in Exhibit 8-29, adverse effects in benthic organisms due to cadmium may be occurring. The HQ (based on the central tendency) was less than one, 5 of 18 samples exceeded the low-effect benchmark (1.0 mg/kg), and 4 of 18 samples exceeded the moderate-effect benchmark (3.53 mg/kg).

Chromium: Based on the risk estimate provided in Exhibit 8-29, adverse toxic effects in benthic organisms due to chromium may be occurring because the HQ was 2, 56% of the samples exceeded the low-effect benchmark (37.3 mg/kg), and 22% exceeded the moderate-effect benchmark (90 mg/kg). (See Figure B4-24). Additionally, two samples exceeded the Canadian Environmental Quality Guideline severe-effect benchmark (110 mg/kg). Sediment concentrations above the severe effect benchmark are expected to be frequently associated with adverse biological effects (Environment Canada, 1998).

Copper: Based on the risk estimate provided in Exhibit 8-29, adverse effects in benthic organisms are expected to occur due to copper. The HQ is 7, 72% of the samples exceeded the low-effect benchmark (35.7 mg/kg), and 44% of the samples exceeded the moderate-effect benchmark (197 mg/kg). (See Figure B4-25). These benchmarks are freshwater sediment values provided by Smith et al. (1996), (see Appendix O).

Lead: Based on the risk estimate provided in Exhibit 8-29, adverse toxic effects in benthic organisms may be occurring due to lead. The HQ (based on the central tendency) was less than one, but 17% of the samples were greater than the low-effect benchmark and 17% were greater than the moderate effect benchmark. (See Figure B4-26). However, a majority of the samples (14 of 18) were reported as non detects (see Table I10 in Appendix I).

Silver: Based on the risk estimate provided in Exhibit 8-29, adverse toxic effects in benthic organisms due to silver may be occurring. The HQ was 31, 22% of the samples exceeded the low-effect benchmark (1.0 mg/kg), and 17% of the samples exceeded the moderate-effect benchmark (3.7 mg/kg). (See Figure B4-27). However, a majority of the samples (13 of 18) were reported as non detects (see Table I10 in Appendix I).

Zinc: Based on the risk estimate provided in Exhibit 8-29, adverse toxic effects in benthic organisms due to zinc may be occurring. The HQ was 3, 61% of the samples exceeded the low-effect benchmark (123 mg/kg), and 44% of the samples exceeded the moderate-effect benchmark (315 mg/kg). (See Figure B4-28). Two samples even exceeded the Canadian Environmental Quality Guideline severe-effect benchmark (820 mg/kg).

PCBs: Exhibit 8-29 provides two risk estimates for PCBs reflecting two different sampling and analysis periods. The historical data (1987 & 1990) was generated by sampling efforts of the Army contracting firms Camp, Dresser, and McKee (CDM) and Federal Cartridge Company (FCC). The most recent sediment data (from 1992) was collected as part of the OU-2 Feasibility Study by the Army's contractor Montgomery Watson. These data are discussed in more detail in Appendix P.

Based on the two risk estimates in the exhibit, it appears that adverse toxic effects in benthic organisms due to PCBs probably occurred in the past. The HQ using the historical dataset from 1987 and 1990 was 70 with 63% of the samples exceeded the low effect benchmark (0.033 mg/kg) and 50% of the samples exceeded the moderate-effect benchmark (0.277 mg/kg).

Based on the most recent sampling data from 1992, it appears that adverse toxic effects in benthic organisms due to PCBs may no longer be occurring. The HQ using the 1992 dataset was less than 1 with no sampling locations exceeding the effect benchmarks. The 1992 sampling and analysis efforts within the OU-2 feasibility study could not verify the past detections of PCBs in the surficial sediment (0-1 ft depths).

BENTHIC COMMUNITY EVALUATIONS

Benthic community evaluations were conducted in 1993 and first reported in the Tier I risk assessment (USACHPPM 1997). During that evaluation samples were taken near the outfalls at the northern portion of Round Lake, off shore of a construction debris landfill, and at the outlet. The depth of the water was approximately five feet and there was abundant submerged aquatic vegetation. Results found that the species found in the lake were typical of a eutrophic pond. At most sample locations there was widespread distribution of species. The species in the lake presented a mix of tolerance to organic enrichment and adverse water quality conditions. No single species dominated the community, and the diversity ranged from moderate to moderately high in samples taken from the northern section of the lake. In the sample taken in the southern portion of the lake, there were fewer species and fewer individuals. There were no aquatic worms or chironomids, and few amphipods at this south sample. The species present were not known to be indicative of water quality, and diversity was calculated to be moderately low.

Because low diversity was expected in the northern portion of the lake, near the suspected sources of contamination (the outfalls), the low diversity observed in the southern portion of the lake was somewhat unexpected. Only one sample was taken from the southern portion, so to further investigate the potential for the presence of toxic sediment in the south, an additional study was conducted in 1995, focusing on the sediments in the southern portion of Round Lake (USACHPPM 1998a). The study on the southern portion of Round Lake also collected samples from the reference area (Snail Lake). Results showed that the majority of species collected at both Round and Snail Lakes had a wide range of tolerance. No extremely pollution sensitive or extremely pollution tolerant species were collected. Compared to the reference lake, Round Lake did have lower abundance and species diversity. In a composite sample, only 5 species and 32 individual organisms were collected in Round Lake. The clam (*Spharium striatinum*), the snail (*Gyralus deflectus*), the amphipod (*Crangonyx gracilis*), and the chironomids (*Chironomus decorus* and *Procladitus sublettei*) were collected. These species are generally tolerant of organic pollution or also poor water quality conditions. The lower abundance and diversity may have been attributable to the lower dissolved oxygen concentrations found in the southern Round Lake samples (4.6 mg/L) and the amount of total suspended solids (5 mg/L). It is also possible that the variation was natural.

Additionally in 1995, a benthic community evaluation was also performed at the northern portion of Round Lake (USACHPPM 1998b). The results showed that the northern portion of Round Lake was dominated by five benthic species. The species found were the common amphipod (*Crangonyx gracilis*), *Polypedilius fallax* found on vascular hydrophytes and is tolerant to poor water quality, *Glyptotendipes lobiferus* and *Physella gryrina* (pollution tolerant species), and the midge *Chaoborus punctipennis*. The presence of these species is typically associated with those found in a eutrophic pond.

For comparison purposes, a reference location (Snail Lake) was also sampled. The results from the 1993 Snail Lake sampling indicated that the six most abundant species found in the reference lake required an organic environment, and that the diversity was determined to be moderate (USACHPPM 1997). Snail Lake samples from the 1995 study (USACHPPM 1998b) showed that the macroinvertebrate species richness and diversity index do not appear different than that found in Round Lake. The Snail Lake samples were dominated by oligochaeta (*Aulodrilus pluriseta* and *Amphichaeta Americans*), which are indicators of organically enriched environments.

SEDIMENT-METALS BIOAVAILABILTY STUDY

The study measured the simultaneously extractable metal/acid volatile sulfide (SEM/AVS) ratio at 4 sediment locations in the northern section of Round Lake during September 1995 (USACHPPM 1998b). In anoxic sediments, when the concentration of metal (SEM) exceeds the binding of the acid volatile sulfide (AVS) pool, metals are available to benthic organisms (Rand 1995). The SEM/AVS measurements taken in Round Lake indicated that at three of the four sample locations sufficient AVS existed in the sediments to bind the divalent transition metals (cadmium, copper, nickel, lead, and zinc) rendering them unavailable to organisms. One of the Round Lake samples (RL0501, see Figure I2) showed a potential for toxicity due to cadmium and/or zinc. On the other hand, additional metal binding phases can play a role in reducing toxicity in addition to AVS (e.g. organic carbon). Therefore, the degree to which organic carbon is present will influence the SEM/AVS ratio to overestimate exposure and risk. This issue has site-specific importance because the samples that contained the highest SEM/AVS ratio were associated with the highest levels of organic carbon.

Therefore, based on this risk estimate of the SEM/AVS, it appears that the majority of benthic organisms may be unaffected during the summer (i.e. when anoxic conditions are expected) by these divalent metals in Round Lake because the metals are not bioavailable.

SEDIMENT TOXICITY TESTS

The fourth line of evidence is based on sediment toxicity testing performed using the top 6 inches sediments collected from Round and Snail lakes (technique 1). Exhibit 8-30 presents the toxicity test results and subsequent qualitative risk estimates associated with the 1999 sampling of the northern sediments in Round Lake. Exhibit 8-31 and Exhibit 8-32 present the toxicity test results and subsequent qualitative risk estimates associated with the 1995 sampling of the southern sediments.

Exhibit 8-30. Round Lake Risk Estimates for Benthic Organisms Based on 1999 Sediment Toxicity Test Results for *Chironomus riparius* †

Sample Location *	14-Day Test		30-Day Test		
	Survival Rate (%)	Growth	Survival Rate (%)	Emergence Rate (%)	Mean Time to Emergence (d) ‡
Northern area of Round Lake					
W1-RL-SD-01	83	na	13 **	11 #	17.2
W1-RL-SD-02	84	na	79	78	18.1
W1-RL-SD-03	86	na	13 **	13 #	15.7
W1-RL-SD-04	41 **	na	40 **	39 #	15.0
W1-RL-SD-05	44 **	na	60 **	58 #	15.6
Snail Lake					
W1-SL-SD-01	95	na	88	79	14.3
W1-SL-SD-06	93	na	94	84	14.1
W1-SL-SD-09	94	na	84	81	13.6
Lab Control A	95	na	89	89	14.0
Lab Control B	88	na	90	88	15.2

Risk Estimate	• Cumulative survival rates were lower than at the reference lake. • At the northern most locations (04 and 05), the survival differences were substantial and reached statistical significance. • Growth was not measured because it was not feasible because there was significant emergence of organisms (see text)	• Cumulative survival rates were lower than at the reference lake. • At 4 of 5 locations, the survival differences were substantial and reached statistical significance. • At 4 of 5 locations, the cumulative emergence rate was substantially different than at the reference lake. • Mean time to emergence is delayed compared to the reference lake

na: Not available

† The data presented here is summarized across replicates for each sample location. Appendix L presents the testing report.

‡ The mean time to emergence (days) was only calculated for those organisms that actually emerged.

* Sample locations refer to samples identified on Figure I1 (Round Lake) and Figure J1 (Snail Lake).

** Indicates a significant statistical difference when compared to the pooled results from sample locations in the reference site (Snail Lake).

\# Indicates a presumption that emergence rates at these locations are significantly different than the pooled results from sample locations in the reference site (Snail Lake). Statistical tests were not performed due to significantly lower survival rates.

Exhibit 8-31. Round Lake Risk Estimates for Benthic Organisms Based on 1995 Sediment Toxicity Test Results for *Chironomus riparius* †

Lake and Test Replicate *	14-Day Test		30-Day Test		
	Survival Rate (%)	Growth, as Dry Weight Mass (mg)	Survival Rate (%)	Emergence Rate (%)	Mean Time to Emergence (d)
Southern area of Round Lake					
Replicate A	40	0.33	100	100	25.6
Replicate B	100	0.21	100	100	25.6
Replicate C	100	0.26	100	100	25.3
Replicate D	10	0.50	100	100	25.9
Snail Lake					
Replicate A	100	0.11	100	100	25.7
Replicate B	100	0.22	100	100	24.5
Replicate C	100	0.27	100	100	24.9
Replicate D	100	0.29	100	100	25.5
Lab Control					
Replicate A	100	1.4	100	100	16.6
Replicate B	100	1.0	100	100	17.5
Replicate C	100	1.3	100	100	16.4
Replicate D	100	0.91	100	100	16.4

Risk Estimate	• The cumulative survival rate (across all replicates) was not statistically different than at the reference lake. • Organism growth, measured as total dry weight, was not statistically different than at the reference lake.	• The cumulative survival rate (across all replicates) was not different than at the reference lake. • Cumulative emergence rates were not different than at the reference lake. • Mean time to emergence was not delayed compared to the reference lake.

[†] The data presented here is summarized from the original report: *Sediment Toxicity Evaluation at Round Lake* (USACHPPM 1998b).

* A composite sediment sample from each lake was split into four replicates for the test. The three sampling locations contributing to the composite sample for the southern sampling area of Round Lake (4-1, 4-2, and 4-3) are presented in Figure I2 (Round Lake). Statistical tests were performed on pooled replicate data.

Exhibit 8-32. Round Lake Risk Estimates for Benthic Organisms Based on 1995 Sediment Toxicity Test Results for *Hyalella azteca* [†]

Lake and Test Replicate *	14-Day Test		30-Day Test		
	Survival Rate (%)	Growth, as Dry Weight Mass (mg)	Survival Rate (%)	Growth, as Dry Weight Mass (mg)	First Day of Observable Reproduction
Southern area of Round Lake					
Replicate A	100	0.18	100	0.36	30
Replicate B	70	0.16	100	0.13	30
Replicate C	100	0.16	100	0.39	30
Replicate D	90	0.14	100	0.40	30
Snail Lake					
Replicate A	70	0.09	100	0.40	30
Replicate B	80	0.06	100	0.40	30
Replicate C	80	0.20	100	0.38	30
Replicate D	90	0.20	100	0.37	30
Lab Control					
Replicate A	80	0.13	100	0.12	30
Replicate B	80	0.14	100	0.20	30
Replicate C	60	0.23	100	0.16	30
Replicate D	90	0.16	100	0.15	30
Risk Estimate	• The cumulative survival rate (across all replicates) was not statistically different than at the reference lake. • Organism growth, measured as total dry weight, was not statistically different than at the reference lake.		• The cumulative survival rate (across all replicates) was not different than at the reference lake. • Organism growth, measured as total dry weight, was not statistically different than at the reference lake. • Signs of reproduction (eggs) were noted for each replicate on day 30.		

[†] The data presented here is summarized from the original report: *Sediment Toxicity Evaluation at Round Lake* (USACHPPM 1998b).

* A composite sediment sample from each lake was split into four replicates for the test. The three sampling locations contributing to the composite sample for the southern sampling area of Round Lake (4-1, 4-2, and 4-3) are presented in Figure I2 (Round Lake). Statistical tests were performed on pooled replicate data.

Based on the risk estimates provided in Exhibit 8-31 and Exhibit 8-32 it appears that sediments in the southern portion of Round Lake are not toxic to benthic organisms. In the 30-day tests, both *C. riparius* and *H. azteca* attained 100% survival. Cumulative emergence rates in the *C. riparius* tests were not different than at the reference lake, and in the *H. azteca* tests organism growth was not statistically different than at the reference lake. In the 14-day tests for both species, the cumulative survival rates were not statistically different than at the reference lake. Additionally, organism growth for either species was not statistically different than at the reference lake.

Risk estimates provide in Exhibit 8-30 for the northern portion of Round Lake indicate that the sediments are toxic to benthic organisms. In the 14-day test, the cumulative survival rates were lower in Round

Lake than at the reference lake (Snail Lake). At the northern most locations (RL-04 and RL-05), the survival differences were substantial and reached statistical significance. In the 30-day test, cumulative survival rates were lower than at the reference lake. At four of the five locations, the survival differences were substantial and reached statistical significance. Additionally, the cumulative emergence rate was substantially different than at the reference lake. The mean time to emergence also appears delayed compared to the reference location. However, due to limitation with the 1999 toxicity study on Round Lake there is uncertainty surrounding the use of time to emergence data (see Appendix R, item 7).

8.8.2.2 Risk Description

The assessment endpoint (survival, growth, and reproduction of benthic organisms) was measured both directly and indirectly. The potential for adverse toxicological effects in benthic organisms at Round Lake was evaluated using four lines of evidence. Risk estimates based on each of these four lines were provided in the previous subsection and a synthesis of the findings is provided here.

Elevated concentrations of chromium, copper, silver, and zinc pose a plausible risk for benthic organisms in the northern portion of Round Lake. The most recent sediment sampling effort could not confirm the presence of PCBs in the surfical sediment (0-1 ft. depth); however, historical data suggest that PCBs posed probable risks for benthic organisms in the past. Additionally, other data suggest that there may not be sufficient AVS and other organic material to render copper and zinc biologically unavailable in some portions of this northern area. The available toxicity test results are consistent with these results. Risk estimates provided by the toxicity tests indicate that the northern sediments cause adverse effects in benthic organisms. However, there are important limitations on those test results. Conclusions cannot be made about the potential risk posed by vanadium, if any, because vanadium-specific sediment quality benchmarks are unavailable, the SEM/AVS evaluation does not address the bioavailability of vanadium, and the toxicity tests were not designed to link specific COCs to test observations.

Findings from two benthic community surveys indicate that the benthic community in Round Lake is a mix of species tolerant to organic enrichment and adverse water quality conditions (typical of a eutrophic pond). Snail Lake (reference site) was not determined to be eutrophic. However, samples from Snail Lake did indicate that two groups dominate its benthic community: species typical of organic environments (oligochaeta), and those requiring organic environments (amphipods). It appears that Round Lake and the reference location may suffer from the same issue of organic enrichment. Unfortunately, macroinvertebrate community surveys do not provide a direct link between COCs and adverse effects related to the risk hypothesis. However, they do provide information on community structure that can be used to indicate ecosystem health. The presence of more pollution tolerant species can be a result of increased organic matter, anoxic conditions due to vegetation and other physical properties of the lake, or historical contamination. A causal link between the COCs and this condition has not been demonstrated nor ruled out at Round Lake.

The comparison of sediment concentrations to sediment quality benchmarks, suggests that ecological effects may be occurring or have occurred in the past due to elevated concentrations of chromium, copper, silver, zinc, and PCBs. Since promulgated sediment quality criteria were not available from either Minnesota or the USEPA, we used benchmarks derived from various reference sources. Of the sediment sampling locations, 22 percent of chromium samples, 44 percent of copper concentrations, 17 percent of silver concentrations, and 44 percent of zinc concentrations were above the Smith et al. (1996) Probable Effect Level, (the level above which toxicity can be frequently observed). For PCBs, 50 percent of the historical samples were above the Smith et al (1996) Probable Effect Level for total PCBs. The PCB low-effect level was derived from a different source (USEPA 2000) and was based on Aroclor 1221. An important limitation of this line of evidence is that such effects-benchmarks do not take site-specific conditions under consideration.

Based on a preliminary bioavailability study using SEM/AVS measurements, cadmium, copper, nickel, lead, and zinc did not appear to be generally bioavailable because the AVS pool was of sufficient quantity to bind these divalent metals. One of the Round Lake samples showed a potential for toxicity due to cadmium and/or zinc. However, concentrations of AVS have been shown to vary by 2 orders of magnitude and have been correlated with changes in the temperature of the overlying water. The optimum sulfate reduction rates (i.e., AVS formation) occur in the summer when water temperatures are the highest and oxygen consumption rates increase, forcing the sediment to become anoxic. Therefore, the lowest AVS concentrations—and by extension, the most vulnerable time for toxic effects in organisms—would be expected to occur in the colder winter months. The available site-specific data suggest agreement with this prediction. Data collected by the MPCA in March 1994 (Appendix C in USACHPPM 1998b) showed lower AVS concentrations than did the September 1995 USACHPPM data. Considering the uncertainties of the SEM/AVS technique, these results suggest that if any excess risk exists, then it may be characterized by increasing mobilization and bioavailability of these particular metals in winter and spring, followed by inhibited mobility and decreased bioavailability in the summer and fall. This scenario could be described as a seasonal "pulse" of potentially adverse exposures (USACHPPM 1998b). The quarterly water monitoring data from Round Lake appear consistent with this hypothesis (samples collected between September 1999 – June 2000). The second quarter of sampling (January 2000) showed that the divalent metals cadmium and zinc had slightly higher surface water concentrations than during any other quarter (see Table A39). Without an SEM/AVS evaluation performed during the most vulnerable time of the year (i.e., winter), a hypothesized link between cadmium and zinc and this degraded condition cannot be fully explored.

Sediment toxicity tests of field-collected sediments were conducted using two standard test species: *Chironomus riparius* and *Hyalella azteca*. Based on the *Chironomus riparius* sediment toxicity tests, benthic organisms in the northern portion of Round Lake appear to have lower cumulative survival rates than controls and reference sediments, and slightly delayed emergence times when compared to the reference (Snail Lake). However, the biological significance of this difference in time to emergence between the lakes is not known. Unfortunately, growth was not acceptably measured in the short-term tests (14-day). Additionally, sediment collected from the top six inches portrays effects that are a result of current as well as historic contamination. The TCAAP source of contaminants to Round Lake has not been operation for a number of years so the toxicity tests represent the aggregate toxicity without differentiating the source.

Unfortunately, the *Hyalella azteca* tests were found by USACHPPM to be invalid and were rejected for risk characterization purposes. Even though toxicity tests are considered stronger lines of evidence than chemistry and species composition data alone, testing in more than one species is usually necessary to adequately characterize risk. Additionally, there are other data quality concerns regarding the toxicity tests (see Appendix R). These uncertainties limit the confidence that can be placed on these tests.

The spatial distribution of risk within Round Lake can be estimated by combining the sample-specific findings from each of the lines of evidence with the sample location maps provided in Figures I1 and I2. (Note: The figures could not be accurately combined because GPS information was not available for all sampling locations.) The majority of the distribution of risk is believed to be located in the northern portion of the lake as indicated by findings in the OU2-FS and historical investigations, so investigations were performed in this portion. The benthic community survey data indicate that at most sample locations there was widespread distribution of species and those species found were indicative of eutrophic conditions. However, the 1993 survey did conclude that the area near the outlet (the southern portion) showed fewer species than in the northern section. This conclusion was also verified in the

1998a USACHPPM study. However, sediment chemical concentration data does not necessarily indicate that the lower species diversity is a result of COC presence in the southern area of the lake.

Chemistry data indicate that the concentrations of COCs are relatively heterogeneous in the northern portion of the lake, with some of the higher sediment concentrations existing in line with the outfall. The *C. riparius* toxicity test results showed that four of the five sampling locations in the northern section of the lake had statistically lower survival rates compared to the reference lake in the 30-day test. In contrast, data from the metals bioavailability study indicate that AVS concentrations appear to be sufficient to bind divalent metals at three of the four locations in the northern section of the lake. However, one location, directly in line with the outfall, had insufficient AVS to bind divalent metals. However, it should be noted that because the toxicity tests were based on a composite sample, they were not designed to identify differences between locations within the lake.

8.8.3 Hypothesis 3: Effects in Amphibians

The following assessment endpoints and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, development, and reproduction of amphibians
>
> Risk Hypothesis – Amphibian survival, development, and reproduction may be at risk due to contaminants present in the lake.

8.8.3.1 Risk Estimate

The potential for adverse toxicological effects in amphibians was inferred by evaluating three lines of evidence; Frog Embryo Teratogenesis Assay-Xenopus (FETAX), species richness and relative abundance estimates, and immune function. Risk estimates based on each of these lines are provided in this subsection. A synthesis is provided in the Risk Description subsection.

It is expected that these organisms will be exposed during the embryo and juvenile portions of their life cycle. Additionally, exposure for adults is expected to be on an intermittent basis. However, sufficient information is not available to determine the size of the population exposed to the COCs.

FETAX RESULTS

Risk estimates for the toxicity test line of evidence, using surface water from Round Lake for the FETAX, were developed based on technique 1. Exhibit 8-33 presents the quantitative risk estimates for this line of evidence, which are then described below.

Exhibit 8-33. Round Lake Risk Estimates for Amphibians based on results of the 1999 FETAX Tests†

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).

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†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.

Based on the risk estimate provided in Exhibit 8-33, amphibians in Round Lake appear to have acceptable occurrences of mortality and malformations when compared to the controls and the reference site. The report, containing supporting information on this conclusion, is found in Appendix M. The types and incidences of malformations observed in Round Lake samples were not statistically different from controls or the reference site. Also, mortality was not found to be statistically different in Round Lake samples than in the controls and reference samples. However, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected, and may be unacceptable for decision-making in some cases. The FETAX test was conducted on a composite surface water sample consisting of four samples from along the western shoreline of Round Lake, due to site access constraints. Therefore, effect(s) could not be linked to a potential "hot spot" or suspected source area in the lake. Additionally, surface water chemical analysis for metals was not conducted on the water samples collected for use in the FETAX analysis. FETAX was not conducted with sediments from Round Lake due to the inherent difficulties in dealing with sediment in amphibian assays (i.e. with dejellied embryos in FETAX, fouling the water in adult exposure assays). The possible effects of sediment were considered such that water-soluble constituents would be expected to partition to the water and then to the organism. Therefore, exposure (dermal) to sediment COCs was assumed to be at equilibrium between water and sediment (pore water).

IMMUNE FUNCTION, ORGAN WEIGHTS, AND HISTOPATHOLOGY

For this line of evidence qualitative risk estimates are provided based on data collected on organ weights, two immune function markers, and histopathologic evaluations of important blood conditioning organs. Comparisons were made in adult frogs (*X. laevis*) exposed 14-days to surface water from Snail and Sunfish Lakes and also from a laboratory control solution used in the FETAX assay. The two markers, based on phagocyte response tests, are sensitive indicators of adverse immune responses. Organ weights can be useful indicators of toxicity. Appendix M presents the full results.

The following bullets present the risk estimates.

- The production of radical oxygen intermediates (ROI) in adult frogs exposed to Round Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- The phagocytosis of foreign particles by splenic macrophages in adult frogs exposed to Round Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- Ratios of organ weight to body weight in adult frogs exposed to Round Lake surface water was no different statistically than that found in frogs exposed to laboratory control solution or surface water from the reference site (Snail Lake).
- No notable histological differences were found between sections of thymus, lung, kidney, and liver when examined using light microscopy for male and female frogs exposed to Round Lake surface water, compared to Snail Lake and the laboratory control solution.

SPECIES RICHNESS AND RELATIVE ABUNDANCE

Risk estimates for this line of evidence were developed using visual encounter surveys, aural estimates, and trapping. Exhibit 8-34 presents the qualitative risk estimates for this line of evidence.

Exhibit 8-34. Round Lake Risk Estimate for Amphibians Based on Species Richness

Site	Number of Species Seen or Heard in April '99, May '98, and July '98	Species Observed
Round	5	*Rana clamitans, Pseudacris crucifer, Rana pipiens, Ranid species, Bufo americanus*
Snail	6	*Bufo americanus, Rana clamitans, Rana pipiens, Ranid species, Pseudacris triseriata, Hyla chrysoscelis*

Risk Estimate	- No statistically significant differences were found between the reference site and Round Lake for the number of species observed. - It was not possible to accurately report relative abundance due to difficulties in accessing the sites. Species observed may have been listed because they were heard but not seen, so a definitive count was not possible.

Based on the risk estimate provided in Exhibit 8-34, the diversity of amphibians in Round Lake appears to be no different than that in the reference lake (Snail Lake). Due to habitat constraints and the relatively quick Anuran reaction reflex, many frogs detected by sight during the VES surveys could not be reliably identified. Therefore, more species may exist at each site than was originally reported. The study concluded that overall, relatively few amphibians were found.

8.8.3.2 Risk Description

The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species for FETAX, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. The potential for adverse toxicological effects in amphibians at Round Lake is inferred using three lines of evidence. Risk estimates based on each of these lines were provided in the previous subsection. However, it is noted that there are uncertainties associated with the amphibian study including but not limited to; no multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, and laboratory tests used species not found at the site. These uncertainties along with others can be found in Table 6.1 in the amphibian report (see Appendix M). Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations, and site surface water was not analyzed for COC concentrations at the time of the amphibian study. Therefore, effects from specific surface water COCs were not reported, because it was not part of the study design.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of the assessment.

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, yet sediment exposures were not specifically evaluated. However, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 10%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation (see Tables 5 and 6 in Appendix M). The field data were ambiguous, yet amphibians were noted at all locations.

Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to the COCs from TCAAP that were transported to Round Lake via the outfall (the suspected source area) is not apparent. Additionally, data collected by Jannett (1997) and others (as discussed in Appendix M) appear to corroborate a conclusion that adverse impacts of chemical exposures to amphibians are unlikely.

8.8.4 Hypothesis 4: Effects in Aquatic Mammals

The following assessment endpoint and risk hypotheses were evaluated.

> Assessment Endpoint – Reproductive potential and productivity of aquatic mammals
>
> Risk Hypothesis – The health of aquatic mammals, such as the muskrat, may be impaired if they are excessively exposed to sediments and vegetation containing elevated levels of cadmium, chromium, copper, lead, silver, vanadium, and zinc.

8.8.4.1 Risk Estimate

For the current scenario, the potential for adverse toxicological effects in aquatic mammals was estimated using comparisons of single point estimates of exposure and effect that highlight the variability in the collected site data. Risk estimates are provided in this subsection. A synthesis is provided in the Risk Description subsection

Muskrat

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. Considering the size of Round Lake (51 ha), the amount of suitable habitat, the expected 8 meters average distance between dens, the solitary lifestyle of mating pairs, and the home range (0.17 ha) of muskrats, it can be expected that Round Lake is capable of supporting approximately 450 adult muskrats. This is a very conservative estimate, assuming that there is ample food and that the whole of the northern, eastern and southeastern portions of the lake is inhabitable. The frequency of exposure for muskrats is expected to be continuous. For those muskrats that are exposed, it is expected to be for the majority of their life cycle. Based on the current information, the portion of the muskrat population exposed at Round Lake is unknown.

Exhibit 8-35. Muskrat Hazard Ratio Matrix for Round Lake

	Muskrat	
	HQ1	**HQ2**
Barium:		
Maximum	2	0.7
Central Tendency	1	0.5
Cadmium		
Maximum	0.5	0
Central Tendency	0	0
Chromium:		
Maximum	2	0.8
Central Tendency	1	0.4
Copper:		
Maximum	5	3
Central Tendency	1	0.6
Lead:		
Maximum	0.1	0
Central Tendency	0	0
Silver:		
Maximum	150	30

Central Tendency	53	11
Vanadium:		
Maximum	1	0.3
Central Tendency	0.7	0.1
Zinc:		
Maximum	0.2	0.1
Central Tendency	0.1	0.1

Cadmium, Lead, Vanadium, Zinc: None of the hazard quotients for these COCs are greater than unity. This indicates that adverse health effects due to these COCs are not expected for mammals at Round Lake.

Barium: One of the four HQs for the modeled muskrat is greater than unity. The HQ of 2 (based on the maximum site concentration and an estimate of the NOAEL) indicates that a maximally exposed muskrat in the population may experience an exposure 2-times greater than that considered to be safe. Because the remaining HQ ratios (especially those based on the LOAEL) are not greater than unity, estimated exposures in the muskrat population are less than exposures known to be associated with adverse health effects.

Chromium: One of the four HQs for the modeled muskrat is greater than unity. The HQ of 2 (based on the maximum site concentration and an estimate of the NOAEL) indicates that a maximally exposed muskrat in the population may experience an exposure 2 times greater than that considered to be safe. Because the remaining HQ ratios (especially those based on the LOAEL) are not greater than unity, estimated exposures in the muskrat population are less than exposures known to be associated with adverse health effects.

Copper: Two of the four HQs for the modeled muskrat are greater than unity. The HQ of 5 (based on the maximum site concentration and an estimate of the NOAEL) indicates that a maximally exposed muskrat in the population may experience an exposure 5-times greater than that considered to be safe. The HQ of 3 (based on the maximum concentration and LOAEL) indicates that individual muskrats in the population may experience exposures 3-times greater than that known to be associated with adverse health effects. However, the HQ ratios based on the central tendency exposure estimates are not greater than unity (1 and 0.6), indicating that on average exposures in the muskrat population are less than exposures known to be associated with adverse health effects.

Silver: All four of the HQs for the modeled muskrat are greater than unity. The HQ of 150 (based on the maximum site concentration and an estimate of the NOAEL) indicates that a maximally exposed muskrat in the population may experience an exposure 150-times greater than that considered to be safe. The HQ of 30 (based on the maximum and LOAEL) indicates a maximumally exposed muskrat in the population may experience exposures 30-times greater than that known to be associated with adverse effects. The HQ ratios based on the central tendency exposure estimates are also greater than unity (53 and 11) indicating that estimated exposures in the muskrat population are greater than exposures known to be associated with adverse health effects.

8.8.4.2 Risk Description

Based on the line of evidence used to estimate risk to aquatic mammals at Round Lake, it appears that population-level toxicological effects are not expected due to exposure to cadmium, lead, vanadium, and zinc. However, it does appear that population level toxicological effects are possible, but probably unlikely for exposure to barium, chromium, and copper. Additionally, the HQ evidence indicates that the possibility for adverse population-level effects is most likely for mammals exposed to silver. The HQ ratios for silver range from 11 to 150. This indicates that on average the population's exposure is estimated to be 11-times greater than exposures known to be associated with adverse health effects. However, the concentrations of two of the 18 sample locations in Round Lake are skewed very high relative to the rest of the samples (see Figure B4-27 and Figure I2). These two locations are not found near the shoreline where muskrat dens at the lake are located. This suggests that the true population-average exposure would be less than exposures known to be associated with adverse health effects. That is, removing these two locations changes the HQ range from 11-150 to 0.2-4. Regardless of the model predictions, there remains uncertainty as to whether adverse effects are occurring due to limitations in this evidence.

The assessment endpoint for aquatic mammals was not measured directly. 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 than exceeds unity triggers further careful consideration of the underlying scientific basis of the prediction.

8.8.5 Hypothesis 5: PCB Exposure in Mink, Wading birds, and Belted Kingfishers

The following assessment endpoint and risk hypothesis were evaluated. This risk hypothesis was developed from the PCB evaluation presented in Appendix P.

> Assessment Endpoint – Reproductive potential and productivity of aquatic mammals and wading and piscivorous birds.
>
> Risk Hypothesis – If PCBs are present above acceptable levels in sediments and fish, then developmental and reproductive problems may occur in exposed mammals and birds.

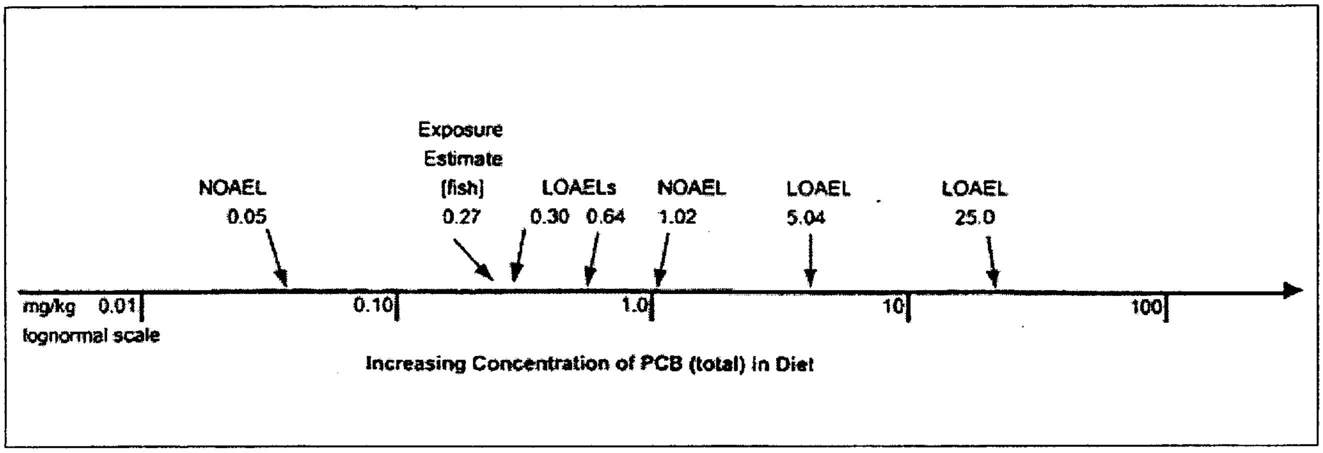
8.8.5.1 Risk Estimates

For the current scenario, the potential for adverse toxicological effects in aquatic mammals and birds was estimated using comparisons of single point estimates of exposure and effect that highlight the variability in the collected site data. The risk estimates for this hypothesis were developed during the Work Plan and were originally presented and approved in 1999 (see Appendix P). Risk estimates are provided in this subsection. A synthesis is provided in the Risk Description subsection

Mink

Signs of mink activity at the lake have been observed. (e.g., tracks in the winter, etc.). Considering the size of the Round Lake habitat (51 total hectares, including areas of open water and an estimated 25 near-shore hectares), and assuming that habitat and food abundance are sufficient, then Round Lake may be expected to support around 2 to 3 adult females. On the other hand, the Round Lake system is a small fragment of habitat within a larger industrialized landscape 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 fragment may not be sufficient to support mink over the long term, though they have been observed there.

Exhibit 8-36. Mink Risk Estimate Diagram for Exposure to PCBs in Fish at Round Lake



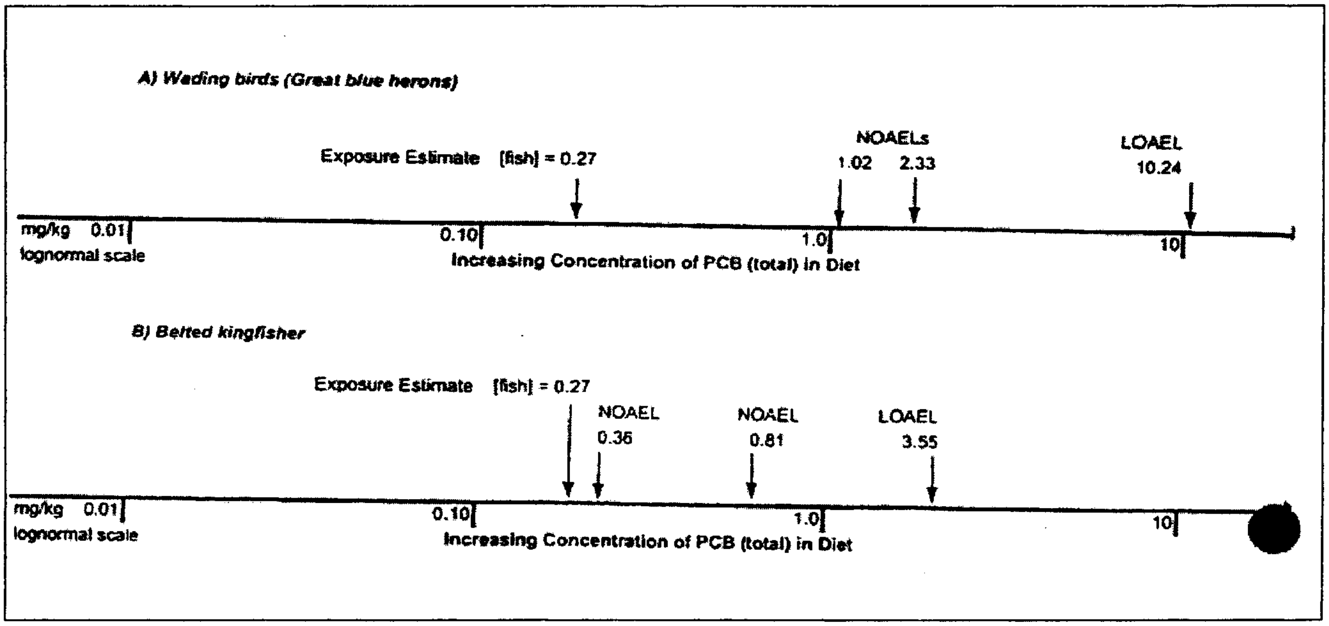
The risk estimate diagram (Exhibit 8-36) shows that the exposure estimate for mink (based on the maximum composited fish concentration: 0.27 mg/kg) is within a range of concentrations where no adverse effects have been observed in laboratory tests (0 – 1.02 mg/kg) and is equilivant to the lowest LOAEL (0.30 mg/kg). The estimation is based, in large part, on the assumption that mink at Round Lake eat only fish (i.e., black bullhead) and the exposure point is best represented by the maximum concentration of total PCBs detected in fish at the lake. This assumption overestimates the risk because the diet of mink does not consist soley of fish, given that the rest of the diet contains less PCB than fish.

Wading birds and Belted Kingfishers

It is not known if wading birds nests at Round Lake. Colonies of herons are not likely due to insufficient habitat. Because Great Blue Herons nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). Round Lake is part of the larger Minnesota Valley National Wildlife Refuge where Great Blue Herons, American Bitterns, and Great Egrets are known to nest (USGS 1995). These birds may not nest at Round Lake but they can be expected to frequent the area. Herons have been known to forage up to 24.4 km from their nesting colonies, with population densities of 2.3 birds/km along streams (USEPA 1993). With this in mind, it can be expected that during March through November, Round Lake could possibly support 4-6 adult herons.

Additionally, kingfishers have been observed on TCAAP and/or Round Lake. They nest in burrows in earthen banks that they dig using their bills and feet. During the breeding season pairs establish territories for nesting and fishing; otherwise they are solitary. Breeding densities of 2-6 pairs per 10 km of river shoreline have been recorded. These birds are sensitive to disturbance and usually do not nest in areas near human activity. Given this information, it is unlikely that more than 3 or 4 pairs inhabit the Round Lake system, if any inhabit the lake at all. Given the general refuge nearby (i.e., TCAAP) they may feed, at least in part, in Round Lake.

Exhibit 8-37.Risk Estimate Diagram for Wading Birds and Belted Kingfishers Exposed to PCBs in Fish at Round Lake



The risk estimate diagram A (Exhibit 8-37) shows that the exposure estimate for wading birds (0.27 mg/kg fish) is below the range of concentrations where no adverse effects are observed in laboratory tests (1.02 – 2.33 mg/kg fish) and is less than the lowest LOAEL (10.24 mg/kg fish). The estimate is based, in large part on the assumption that the wading birds eat only fish (i.e., black bullheads) and the exposure point is best represented by the maximum concentration of total PCBs detected in fish tissue at Round Lake. The assumption overestimates the risk because the diet of wading birds does not soley consist of fish (especially not exclusive to bullhead) (see Appendix N), and the rest of the diet is expected to contain less PCB than the fish. It is assumed that the fish consumed by wading birds will contain relatively higher concentrations of PCBs than would any other component of the diet.

The risk estimate diagram B (Exhibit 8-37) shows that the exposure estimate for the belted kingfish (0.27 mg/kg fish tissue) is below both the NOAEL (0.38 – 0.81 mg/kg) and LOAEL (3.55 mg/kg) based toxicity benchmarks. The assumption overestimates the risk because the diet of the kingfisher does not soley consist of bullhead and the rest of the diet is expected to contain more pelagic fish that would exhibit lower PCB concentrations in the tissue because sediment is not the main exposure route of PCBs to pelagic fish.

8.8.5.2 Risk Description

Based on the line of evidence used to estimate risk for these receptors at Round Lake, adverse toxicological effects due to exposure to PCBs are expected to be low in mink, wading birds, and belted kingfishers for both the littoral and profundal areas of Round Lake because the exposure estimates are not greater than toxicity benchmarks. Population-level effects are possible, albeit unlikely, for mink, wading birds, and belted kingfishers because the maximum exposure to the population is less than exposures known to be associated with adverse health effects.

The assessment endpoint for fish-eating wildlife exposed to PCBs was not measured directly. The potential for adverse changes in the assessment endpoint is inferred by comparing estimates of fish PCB concentrations to estimates of food-based toxicological benchmarks for PCBs. Therefore, uncertainty does exist in the estimate of the health risk of developmental and reproductive problems for these receptors. An exposure estimate that exceeds the benchmark triggers further careful consideration of the underlying scientific basis of the prediction.

8.9 ECOLOGICAL RISKS AT ROUND LAKE (FUTURE SCENARIO)

The proposed future scenario for Round Lake is a water level manipulation and sediment consolidation plan as proposed by USFWS (1998). The surface water COCs for the future scenario are the same as those selected under the current scenario. However, the sediment COCs have changed slightly due to the potential for deep sediment COCs to mobilized to the surface (see Section 6.7.2).

Round Lake COCs: Future Scenario

Surface Water
Barium
Cadmium
Zinc

Sediment
Cadmium
Chromium
Copper
Lead
Manganese
Nickel
Silver
Vanadium
Zinc
PCBs

Appendix I contains tables showing data collected during the OU-2 Feasibility Study for various sediment depth intervals up to a depth of 6 feet below the sediment surface. Based on a review of these tables, concentrations of the majority of the chemicals in the deeper sediment intervals either get smaller or stay relatively equivalent compared to the recently measured surface sediment concentrations. This condition indicates that if the sediments are disturbed such that the deeper sediments are exposed or otherwise transport their chemicals to the surface, the concentrations in the surface sediment would be expected to increase. However, it is possible that some surface sediment COC concentrations would stay the same if that COC is not found in the deep sediment at higher concentrations or if it does not get transported to the surface. Several heavy metals were detected in the deeper sediments at concentrations greater than those found in the top one foot. These chemicals (cadmium, copper, lead, manganese, nickel, vanadium, and zinc) may potentially pose an exposure problem in the future and were added to the COC list for this scenario if they were not already current scenario COCs. PCBs are also known to exist in the deep sediment (see Appendix P for discussion on the investigation of this issue). Since a continuing source of contaminants from TCAAP does not exist, it is expected that the source contamination would be secondary, as released from the sediments if the USFWS drawdown plan is implemented.

The spatial distribution of the surface water COCs is expected to emanate from the northern area of the lake where the storm sewer emptied into the lake. The concentrations of COCs may decrease with increasing distance away from these areas, but this was not evaluated. The exact spatial distribution of the elevated levels of inorganics in the surface water is difficult to determine given the sampling design used in the field investigation because it was based on the present condition of the lake. With the future plan of a full drawdown at the lake, the current spatial and temporal distribution of surface water COCs is expected to change when the lake is refilled. The change may be caused by metals that were once bound to anoxic deep sediment being re-released when cracks that form in the drying sediment are inundated with water upon refilling. Also, oxidation of surficial sediments may make bound metals bioavailable and also introduce contaminants to the surface water as a result of changing redox potentials associated with fluctuating water levels. Again, the actual spatial distribution cannot be determined.

The spatial distribution of sediment COCs is also expected to emanate from the northern area of the lake due to the storm sewer outfall. The exact spatial distribution of sediment COCs was not evaluated. With the future drawdown plan, the current spatial distribution of COCs in the sediment may change if deep sediment contaminants move to the surface sediment by the process of capillary action during sediment depuration. Oxidation of surface sediment may release contaminants to the surface water, thus changing the spatial distribution in the surficial sediment.

The COCs are several inorganics and PCBs, which are environmentally persistent. Their presence through time is expected, and the bioavailable amount of the COCs may change due to variation in sediment conditions (e.g., pH, temperature, nutrient loads, flow volume, oxidation). This aspect of exposure was not directly evaluated. However, when drawdown occurs the temporal pattern of distribution may change. During drawdown, COCs may not be present in surface water, but upon refilling COCs may increase in surface water for a time, until they can be complexed with suspended organic matter that may be stirred up in the refilling process. If drying of the sediments occurs, COCs in deep sediment may increase in surface sediment due to capillary action processes transporting COCs through cracks that may form in the surface sediment during the drawdown phase, and then increase in surface water during the refilling phase. However, the spatial and temporal distribution of COCs under the future scenario cannot be predicted with absolute certainty.

The revised conceptual model was introduced in Section 6.7.5. A revised diagram for Round Lake can be referred to for information pertaining to the fate and transport of COCs (see Figure B3-5). For each hypothesis described in the following sections, a description is provided of how future risks may be different from those determined under the current scenario.

8.9.1 Hypothesis 1: Effects in Fish, Aquatic Invertebrates, and Algae

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, growth, and reproduction of fish, aquatic invertebrates, and algal species
>
> Risk Hypothesis – Based on the aquatic toxicity of surface water COCs, if they are present in the water above acceptable levels for sufficient durations, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species.

Due to the speculative nature of what may occur upon drawdown, it is difficult to predict what toxicological effect the drawdown scenario would have on biota at Round Lake in the short time period that drawdown would occur and refilling would begin (3 months). Based on the information currently available it is assumed that if the USFWS plan is implemented, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species. Under the current scenario, it appears that adverse toxicological effects are possible due to elevated barium concentrations. Without knowing how the Round Lake sediments will oxidize or dry, or how other chemical reactions will occur during drawdown, specific risk estimates identifying which metals may pose excess risk can not be made. It is only expected that there may be an increase of COCs in the surface water upon refilling of the lake. The change may be caused by metals that were once bound to anoxic deep sediment being re-released when cracks that form in the drying sediment are inundated with water upon refilling. Also, oxidation of surficial sediments may introduce contaminants to the surface water as a result of changing redox potentials associated with fluctuating water levels. The actual spatial and temporal distribution cannot be reliably determined.

8.9.2 Hypothesis 2: Effects in Benthic Organisms

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Survival, growth, and reproduction of benthic organisms
>
> Risk Hypothesis – If sediment concentrations of COCs are sufficiently elevated and bioavailable to benthic organisms, then adverse toxic effects may occur.

Due to the speculative nature of what may occur upon drawdown, it is difficult to predict what toxicological effect the drawdown scenario would have on biota at Round Lake in the short time period that drawdown would occur and refilling would begin (3 months). Based on the information currently available it is assumed that if the USFWS plan is implemented, then adverse toxic effects may occur in benthic organisms. Under the current scenario, elevated concentrations of chromium, copper, silver, and zinc pose a plausible risk for benthic organisms in the northern portion of Round Lake. Additionally, other data suggest that there may not be sufficient AVS and other organic material to render copper and zinc biologically unavailable in some portions of this northern area. The available toxicity test results are consistent with these results. Risk estimates provided by the toxicity tests indicate that the northern sediments cause adverse effects in benthic organisms. Under the future scenario, it is expected that the degree of adversity for benthic organisms may increase in the short term. No evidence is available as to whether to expect longer-term toxicity-related effects in benthic organisms. COCs in the sediment may change if deep sediment contaminants move to the surface sediment by the process of capillary action during sediment depuration. Oxidation of surface sediment may release contaminants to the surface water, thus changing the spatial distribution in the surficial sediment. Deep sediment metals may also become bioavailable (see Appendix I for tables showing deep sediment metal concentrations). Without knowing the magnitude of these future processes during drawdown and reflooding, a specific risk description of the long-term adversity that the COCs may pose to benthic organisms under the future scenario remains elusive.

8.9.3 Hypothesis 3: Effects in Amphibians

The following assessment endpoints and risk hypothesis were evaluated.

> <u>Assessment Endpoint</u> – Survival, development, and reproduction of amphibians
>
> <u>Risk Hypothesis</u> – Amphibian survival, development, and reproduction may be at risk due to contaminants present in the lake.

Due to the speculative nature of what may occur upon drawdown, it is difficult to predict what toxicological effect the drawdown scenario would have on biota at Round Lake in the short time period that drawdown would occur and refilling would begin (3 months). It is expected that if COC concentrations increase during drawdown, then exposure would increase. After full drawdown, exposure to amphibians may be increased or stay the same as the current scenario. Under the current scenario, the occurrence of adverse toxicological effects in amphibians due to the COCs from TCAAP that were transported to Round Lake via the outfall (the suspected source area) are not apparent. Additionally, data collected by Jannett (1997) and others (as discussed in Appendix M) appear to corroborate a conclusion that adverse impacts of chemical exposures to amphibians are unlikely. It is not clear whether any adverse effects would occur. If so, it is not certain that the effects would be detectable using a study design similar to that employed in 1999.

8.9.4 Hypothesis 4: Effects in Aquatic Mammals

The following assessment endpoint and risk hypothesis were evaluated.

> <u>Assessment Endpoint</u> – Reproductive potential and productivity of aquatic mammals
>
> <u>Risk Hypothesis</u> – The health of aquatic mammals, such as mink and muskrat, may be impaired if they are excessively exposed to sediments containing elevated levels of COCs.

Due to the speculative nature of what may occur upon drawdown, it is difficult to predict what toxicological effect the drawdown scenario would have on biota at Round Lake in the short time period that drawdown would occur and refilling would begin (3 months). It can be expected that Round Lake is capable of supporting approximately 450 adult muskrats under the current scenario. This is a very conservative estimate, assuming that there is ample food and that the whole of the northern, eastern and southeastern portions of the lake is inhabitable. However, once drawdown is initiated it is expected that the muskrat population will decrease from its current number due to loss of protected dens and increased predation. Upon refilling of the lake the number of muskrats may again increase. Exposure of muskrats to the sediment may also increase during drawdown as they consume cattail roots that are no longer submerged. During drawdown, the plants may also be able to take up more COCs as the COCs become bioavailable under oxic conditions. This would also increase muskrat exposure to COCs. Mucky conditions of the sediments may also present increased exposures when the animals are grooming. Upon refilling of the lake the exposure to the COCs is expected to either stay the same or increase as compared to the current Round Lake scenario.

It is possible that during the drawdown period the mink population may increase from its current number due to increased availability of food (muskrats). However, no quantitative estimate of population increase can be made. As with the muskrat, mucky conditions of the sediments may present increased exposures when the animals are grooming. Upon refilling of the lake the exposure to the COCs is expected to either stay the same or increase as compared to the current Round Lake scenario.

8.9.5 Hypothesis 5: Effects in Waterfowl and Wading Birds

The following assessment endpoint and risk hypothesis were evaluated.

> Assessment Endpoint – Reproductive potential and productivity of waterfowl and wading birds
>
> Risk Hypothesis – The survival, growth, and reproduction of waterfowl and wading birds may be at risk if they are excessively exposed to COPCs that may occur in the benthic invertebrates and fish that they consume at the lake.

Due to the speculative nature of what may occur upon drawdown, it is difficult to predict what toxicological effect the drawdown scenario would have on biota at Round Lake in the short time period that drawdown would occur and refilling would begin (3 months). If bioavailable concentrations of COCs increase in the sediment and surface water, exposure to waterfowl will be increased, mainly through increased COC concentrations in their food items (i.e., invertebrates and vegetation). Also, changes in the spatial distribution of contaminants will also change the exposure to waterfowl. During the first year of drawdown, waterfowl may not yet be attracted to the lake because the water volume may be too small, the ideal vegetation may not yet be present, and the mucky condition of exposed sediments may not be ideal for the birds. However, those birds such as mallards that do frequent the lake would most likely be less exposed to sediment COCs if they are feeding in the middle of the lake, where COC concentrations are lower than in the northern section. Once the lake is refilled and vegetation has improved, more birds are expected to be attracted to the lake and thus exposure of COCs to the waterfowl population would increase.

Without site-specific wading bird activity data, it is difficult to predict how wading birds will be utilizing Round Lake. However, based on life history characteristics some generalizations can be made. Because great blues nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). It is difficult to determine the amount of foraging habitat available at Round Lake without field data. However, based on the available information it is possible that 4-5 wading birds may dominate at Round Lake. It is likely that in the future scenario, no more wading birds will be attracted to the lake due to the solitary hunting style of the birds. Also, the habitat for these birds is not the focus of the drawdown and will not directly be improved.

9. CONCLUDING SUMMARY AND RECOMMENDATIONS

Management goals and objectives as well as available management decisions were defined in section 2. The following is a summary of the results of the risk assessment for each site and how they relate to the site-specific goals and decisions. A detailed, more technical risk characterization is presented in section 8.

9.1 MARSDEN LAKE (NORTH)

Marsden Lake (North) refers to that portion of Marsden Lake immediately surrounding Patrol Road. The north area of the lake was defined as a distinct area to focus the assessment in terms of characterizing any potential risk due to contaminants that may have been introduced as a result of former activities at the Grenade Range. It was originally suspected that the Grenade Range could be a specific source of COPCs detected in the northern portion of the lake. Contaminants in Grenade Range soils may have been transported via runoff into the lake, resulting in metals contamination in the surface water and sediment.

Known soil contaminants at the Grenade Range have recently been remediated to an industrial land use standard (Alliant Techsystems 2001a).

Marsden Lake sites were evaluated in the 1997 Tier I screening-level ecological risk assessment, but the the study areas were not sufficiently characterized for an adequate Tier I screening evaluation (USACHPPM 1997 and 1999). This limitation was addressed within the Tier II risk assessment by performing additional site investigations, including water and sediment sampling and chemical analyses (USACHPPM 1999). The data obtained from these investigations were evaluated using the same Tier I screening process that was finalized in 1997 (USACHPPM 1997), in order to complete that phase of the risk assessment for these sites and identify COPCs that were carried through the Tier II risk evaluation. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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.

9.1.1 Marsden Lake North Findings

The management objective for Marsden Lake is to prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake (see section 2.1). In summary, the risk assessment identified the potential occurrence of some adverse toxicological effects in ecological entities at the site. Of primary concern are adverse changes in survival, growth, and reproduction of fish, aquatic invertebrates, and algal species due to elevated concentrations of aluminum, barium, and manganese in surface water. Some uncertainties remain due to limitations in the evidence.

Chemicals of concern: The selected surface water COCs evaluated in the risk characterization of Marsden Lake (North) were aluminum, barium, lead, manganese, mercury, and zinc. The selected sediment COCs were antimony, arsenic, vanadium, and SVOCs. These COCs (except for SVOCs) were found at concentrations greater than background and toxicological benchmark concentrations. SVOCs were not detected; however, the target detection limits were not achieved in the analysis of sediment samples. This limitation prevents SVOCs from being eliminated as sediment COCs. Even though SVOCs remain potential COCs, they are not strongly suspected to be true contaminants associated with the Grenade Range.

The spatial distribution of sediment COCs showed little variability between sampling locations, suggesting that sediment COCs may be homogenously distributed around the Grenade Range impact areas. However, the spatial distribution of the surface water COCs indicated that the area north of Patrol Road just offshore from the two historical impact sites might be where the majority of surface water COCs are found.

In summary, adverse effects were found to be either not apparent or possible, albeit unlikely, for four of the five endpoints assessed. The risk assessment identified the potential for adverse effects due to aluminum, barium, and manganese for fish, aquatic invertebrates, and algae. However, important uncertainties remain related to the chemical speciation of aluminum and the validity of the toxicological benchmarks used for each of these COCs. Furthermore, it is not clear that these COCs are related to TCAAP-specific activities. These uncertainties question the validity of the identified risks for this endpoint. Each of the assessment endpoint receptors is discussed below.

<u>Fish, aquatic invertebrates, and algae:</u> The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. This line of evidence at Marsden Lake (North) identified the potential that some adverse toxicological effects on the survival, growth, and reproduction of these receptors due to aluminum, barium, and manganese.

Important uncertainties remain as to whether adverse effects have or are occurring due to limitations in the evidence. It may be premature to conclude that aluminum poses a significant risk, based on the uncertainty associated with the form of aluminum present at the lake. The barium risk estimate contains a high degree of uncertainty due to the lack of a sufficient toxicity database; only one study was found regarding aquatic toxicity of barium. While the manganese chronic water quality benchmark (designed to be protective of 95% of aquatic species) was exceeded in most of the samples, the fish-specific benchmark was not exceeded.

<u>Benthic organisms:</u> The assessment endpoint (survival, growth, and reproduction in benthic organisms) was not measured directly for this line of evidence. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site sediment concentration data to toxicity benchmarks. Based on this line of evidence at Marsden Lake (North), it appears that adverse toxicological effects are possible, albeit unlikely, on the population level. Only one sample location showed a potential problem for antimony, with a concentration that was 2-times higher than the moderate-effect benchmark. Two other locations showed arsenic concentrations that were slightly higher than the low-effect benchmark; thus overall, adverse effects to the benthic community from arsenic are unlikely.

There remains a low degree of uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. Based on a lack of data, the possibility of effects from vanadium and SVOCs cannot be ruled out.

<u>Amphibians:</u> The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species using the Frog Embryo Teratogenesis Assay-Xenopus (FETAX) protocol, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to metals from the Outdoor Firing Range or the Grenade Range is not apparent. Additionally, data collected by Jannett (1997) and

others appear to corroborate a conclusion that adverse impacts of chemical exposure to amphibians are unlikely.

Moderate uncertainty remains as to whether adverse effects have or are occurring due to limitations in the evidence including, but not limited to, a lack of multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, laboratory analysis for metals concentrations in surface water used in the assays was not conducted, and laboratory tests used species not found at the site. Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations. The laboratory data investigating embryo mortality (a sensitive life stage), developmental effects, and sensitive indicators of stress (immunocharacterization assays) suggest that surface water from Marsden Lake is not toxic to frogs, yet sediment exposures were not specifically evaluated.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap in most risk assessments. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of this risk assessment. Nonetheless, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 63%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation.

<u>Waterfowl and wading birds:</u> The assessment endpoint for these entities (reproductive potential and productivity of waterfowl and wading birds) was not measured directly. 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—ratios between the exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. Based on this line of evidence at Marsden Lake (North), it appears that population-level adverse toxicological effects in waterfowl are possible, albeit unlikely, due to aluminum and lead. The HQ ratios for lead based on the lowest-observed-adverse-effect-level are not greater than unity, indicating that the estimated maximum and average exposures in the waterfowl population are less than exposures known to be associated with adverse health effects. No comparable toxicity data (i.e., LOAELs) exist for aluminum, but the unknown LOAEL values for aluminum are also expected to be less than unity given the magnitude of the NOAEL-based HQ ratios for aluminum. Adverse toxicological effects are also possible, albeit unlikely, in wading birds at Marsden Lake (North): all HQ ratios are less than unity.

There remains a low degree of uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. The possibility of effects from antimony cannot be ruled out because there are no toxicity benchmarks for birds exposed to antimony. Additionally, effects from SVOCs cannot be ruled out due to a lack of data.

<u>Mink and muskrat:</u> The assessment endpoint for aquatic mammals (reproductive potential and productivity of aquatic mammals) was not measured directly. 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 – ratios between the exposure dose and the dose associated with either NOAEL or LOAEL laboratory studies. Based on this line of evidence at Marsden Lake (North), it appears that adverse toxicological effects are possible, albeit unlikely, in the population of aquatic mammals as a result of exposure to aluminum, antimony, arsenic, barium, lead, manganese, mercury, vanadium, and zinc.

However, moderate uncertainty remains as to whether adverse effects are occurring due to limitations in the evidence. For example, effects from SVOCs cannot be ruled out due to a lack of data.

9.1.2 Marsden Lake North Recommendation

The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

9.2 MARSDEN LAKE (SOUTH)

Marsden Lake (South) refers to the portion of Marsden Lake directly adjacent to the Outdoor Firing Range. The south area was defined as a distinct area to focus the assessment in terms of characterizing any potential risks due to contaminants that could have come from activities at the Outdoor Firing Range. It was suspected that the Outdoor Firing Range could be a specific source of COPCs detected in Marsden Lake. Contaminants in the Outdoor Firing Range soils may have been transported via runoff into the lake, resulting in metals contamination in the surface water and sediment in the area defined as Marsden Lake (South).

Known soil contaminants at the Outdoor Firing Range have recently been remediated to an industrial land use standard (Alliant Techsystems 2001b).

Marsden Lake sites were evaluated in the 1997 Tier I screening-level ecological risk assessment, but the the study areas were not sufficiently characterized for an adequate Tier I screening evaluation (USACHPPM 1997 and 1999). This limitation was addressed within the Tier II risk assessment by performing additional site investigations, including water and sediment sampling and chemical analyses (USACHPPM 1999). The data obtained from these investigations were evaluated using the same Tier I screening process that was finalized in 1997 (USACHPPM 1997), in order to complete that phase of the risk assessment for these sites and identify COPCs that were carried through the Tier II risk evaluation. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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 amphibians.

9.2.1 Marsden Lake South Findings

The management objective for Marsden Lake is to prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake (see section 2.1). In summary, the risk assessment identified the potential occurrence of some adverse toxicological effects in ecological entities at the site. Of primary concern are adverse changes in the survival, growth, and reproduction of benthic organisms due to elevated sediment lead concentrations. Of secondary concern are adverse changes in the reproductive potential and productivity of aquatic mammals due to elevated exposures to aluminum. Some uncertainties remain due to limitations in the evidence.

Chemicals of Concern: The selected surface water COC for the risk characterization of Marsden Lake (South) was manganese. Aluminum, cadmium, lead, vanadium, and SVOCs were retained as sediment COCs. These COCs (except for SVOCs) were found at concentrations greater than background and toxicological benchmark concentrations. SVOCs were not detected; however, the target detection limits were not achieved in the analysis of sediment samples. This limitation prevents SVOCs from being eliminated as sediment COCs. However, they are not strongly suspected to be true contaminants associated with the Outdoor Firing Range (Alliant Techsystems 2001b).

The analytical data show little spatial variability in surface water concentrations. However, there is some variability in sediment concentrations but no pattern is apparent.

In summary, adverse effects were found to be either not apparent or unlikely in four of the five endpoints assessed. The risk assessment identified the potential for adverse effects in benthic organisms due to lead. However, sediment lead concentrations at the site are not statistically greater than nearby Snail Lake, which points to the uncertainty that the lead is related to TCAAP-specific activities. Lead was retained as a COC because it occurs at concentrations that appear greater than relatively pristine Minnesota Valley National Wildlife Refuge Lakes. Lead may be elevated regionally. Each of the assessment endpoint receptors is discussed below.

Fish, aquatic invertebrates,and algae: The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. Based on this line of evidence at Marsden Lake (South), it appears that adverse toxicological effects due to manganese are possible, albeit unlikely, given the magnitude of the risk estimates in light of alternative effects benchmarks. Only one sample location out of 12 exceeded the benchmark indicating that the adversity of effects may be low since the spatial scale examined indicates an isolated area of benchmark exceedance.

There remains relatively low uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence for this site.

Benthic organisms: The assessment endpoint (survival, growth, and reproduction of benthic organisms) was not measured directly for this line of evidence. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site sediment concentration data to toxicity benchmarks. Based on this line of evidence at Marsden Lake (South), it appears that there is the potential for some adverse toxicological effects due to lead because the low-effect benchmark is exceeded in the majority of samples. Adverse effects to benthic organisms from cadmium are unlikely.

Moderate uncertainty remains as to whether adverse effects have or are occurring due to limitations in the evidence. The possibility of effects from vanadium and SVOCs cannot be ruled out due to a lack of data.

Amphibians: The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species using the Frog Embryo Teratogenesis Assay-Xenopus (FETAX) protocol, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to metals from the Outdoor Firing Range or the Grenade Range is not apparent. Additionally, data collected by Jannett (1997) and others appear to corroborate a conclusion that adverse impacts of chemical exposure to amphibians are unlikely.

There remains moderate uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence including, but not limited to, a lack of multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, laboratory analysis for metals concentrations in surface water used in the assays was not conducted, and laboratory tests used species not found at the site. Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations. The laboratory data investigating embryo mortality (a sensitive life stage), developmental effects, and sensitive indicators of stress (immunocharacterization assays) suggest that surface water from Marsden Lake is not toxic to frogs, yet sediment exposures were not specifically evaluated.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap in most risk assessments. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of this risk assessment. Nonetheless, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 63%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation.

<u>Waterfowl and wading birds:</u> The assessment endpoint for these entities (reproductive potential and productivity of waterfowl and wading birds) was not measured directly. 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—ratios between the exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. Based on this line of evidence at Marsden Lake (South), it appears that population-level adverse toxicological effects in waterfowl are possible, albeit unlikely, due to aluminum and lead. The HQ ratios for lead based on the lowest-observed-adverse-effect-level are not greater than unity, indicating that the estimated maximum and average exposures in the waterfowl population are less than exposures known to be associated with adverse health effects. No comparable toxicity data (i.e., LOAELs) exist for aluminum, but the unknown LOAEL-based HQ values for aluminum are also expected to be less than unity given the magnitude of the NOAEL-based HQ ratios for aluminum. Waterfowl HQ ratios for cadmium, manganese, and vanadium were less than unity, indicating that adverse effects in these receptors are also unlikely. Adverse toxicological effects are possible, albeit unlikely, in wading birds at Marsden Lake (South) as well: all HQ ratios are less than unity.

There remains a low degree of uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. The possibility of effects from SVOCs cannot be ruled out because of limitations in the sample analysis for these compounds.

<u>Aquatic Mammals:</u> The assessment endpoint for aquatic mammals (reproductive potential and productivity of aquatic mammals) was not measured directly. 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 – ratios between the exposure dose and the dose associated with either NOAEL or LOAEL laboratory studies. Based on this line of evidence at Marsden Lake (South), it appears that the form of aluminum found at the site may influence the outcome. Assuming that mammals are exposed to aluminum chloride, then the data suggest that population-level adverse effects are possible because the average exposure would be 3-times greater than exposures known to be associated with adverse health effects. On the other hand, assuming they are exposed to aluminum hydroxide, then the data suggest that population-level adverse effects are unlikely given the magnitude of the available HQ ratios associated with "safe" exposures. (Note: exposure levels known to be associated with effects for this form are not known). Based on supporting information provided earlier in the report regarding the expected form of aluminum at the site, the aluminum hydroxide-based conclusion is considered to be more site-specific.

Moderate uncertainty remains as to whether adverse effects are occurring due to limitations in this evidence. Additionally, the possibility of effects from SVOCs cannot be ruled out because of limitations in the sample analysis for these compounds.

9.2.2 Marsden Lake South Recommendation

The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

9.3 POND G

Pond G is a small pond (20 x 80 ft) that received drainage from Site G. The specific source(s) of the COCs in Pond G is suspected to be Site G (uncontrolled landfill), Site F (former burning area), or both. The source areas are no longer in operation; therefore the contamination is historical. The secondary sources for these COCs would be the sediment, and surface runoff from contaminated upland soil areas. Groundwater seeps have also been suggested as a possible source.

Pond G was evaluated in the 1997 Tier I screening-level ecological risk assessment, but the the study area was not sufficiently characterized for an adequate Tier I screening evaluation (USACHPPM 1997 and 1999). This limitation was addressed within the Tier II risk assessment by performing additional water and sediment sampling and chemical analyses (USACHPPM 1999). The data obtained from these investigations were evaluated using the same Tier I screening process that was finalized in 1997 (USACHPPM 1997), in order to complete that phase of the risk assessment for these sites and identify COPCs that were carried through the Tier II risk evaluation.

9.3.1 Pond G Findings

The management objective for Pond G is to protect the health of wildlife using the pond for drinking water (see section 2.1). In summary, the risk assessment did not identify a plausible risk for adverse toxicological effects in the ecological entities evaluated at the site. Some uncertainties remain due to limitations in the evidence.

Chemicals of Concern: The selected COCs evaluated in the risk characterization of Pond G include 13 surface water inorganics, 6 sediment inorganics, as well as Aroclor 1248, SVOCs, and OC pesticides in the sediment. These COCs were found at concentrations greater than toxicological benchmark concentrations. Since a comparable reference location for Pond G is not available and, therefore, was not used in the selection of COCs (USACHPPM 1999), it is difficult to determine whether many of the inorganics present in the Pond are actually contaminants.

Based on the small size of the Pond and its physical characteristics, it is assumed that the spatial and temporal distribution of the COCs is basically uniform. Even though a continuing source of the COCs is not believed to be present, the concentrations of these COCs are not likely to change significantly for the foreseeable future because they all are environmentally persistent.

Wading Birds and mammals: The assessment endpoint for these entities (water and sediment quality for wildlife using the pond) was not measured directly. The potential implication for adverse health effects in Wading birds and mammals is inferred by comparing estimates of exposure to estimates of health effects in the form of a hazard ratio. Based on this line of evidence at Pond G, adverse toxicological effects in for mammals such as the fox are not apparent because all HQ ratios are less than unity. Such effects are also not apparent for wading birds due to exposure to most COCs, yet the possibility of effects due to exposure to antimony, beryllium, cobalt, and thallium can not be ruled out because toxicity values were not available to compare to exposure estimates. Additionally, the possibility of effects from SVOCs cannot be ruled out because of limitations in the sample analysis for these compounds.

9.3.2 Pond G Recommendation

The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

9.4 RICE CREEK

Rice Creek is a tributary of the Mississippi River, which flows through the northwest corner of the installation. The TCAAP accounts for approximately 2 percent of Rice Creek's 474-square-kilometer basin and is near the downstream end of the creek. The State of Minnesota classifies Rice Creek as a "Class 1C, 2BD, 3B water".

Rice Creek potentially receives groundwater from the perched unit 1 aquifer under site K, and was once thought to potentially receive volatile organic compounds (VOCs) from this source (with discharge between OU2-FS sampling locations RCK04SE and RCK05SE). Site K consists primarily of building 103 and surrounding property and occupies approximately 21 acres. The site has been used mainly for munitions manufacturing and assembly operations.

In order to address deficiencies in previous surface water sampling and analysis data used in the 1997 Tier I risk assessment, additional surface water sampling was performed at the creek between September 1999 and June 2000.

9.4.1 Rice Creek Findings

The management objective for Rice Creek is to prevent unacceptable toxic effects (in organisms living) in the creek. Risks of toxicological effects were not identified at this site.

Chemicals of Concern: In the Tier II assessment, surface water was analyzed for COPCs selected in the 1997 Tier I report (aluminum, barium, cadmium, copper, mercury, and silver). Based on the collected data and the implementation of the revised Tier II COC selection criteria, none of these previous COPCs were retained as COCs. This means that no chemical stressors due to TCAAP-related activities have been identified. Therefore, adverse effects were not identified.

9.4.2 Rice Creek Recommendation

The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

9.5 SUNFISH LAKE

Sunfish Lake is a shallow 5.7-hectare lake in the southeast corner of the installation. It was suspected that Site H-1, which contains an old unpermitted landfill and also included metal debris scattered on the bottom of a portion of the Sunfish Lake sediments along the northcentral shore, could be the specific source of COPCs detected in the lake during the OU-2 Feasability Study. Also, contaminated soils from the site may have been transported via runoff into the lake, and metal contaminated leachate from the landfill may have migrated into the lake, causing metals contamination in the sediment and surface water.

Site H-1 is no longer used and the upland soil areas of the site have been remediated in large part because they contained concentrations of metals (i.e., arsenic, antimony, copper, lead, and manganese) that posed excessive risks to human health (Stone & Webster 2002).

The Tier I risk assessment (USACHPPM 1997) identified potential sediment-related risks, which required further investigtion. Therefore, sediment toxicity tests were conducted usign field-collected sediments. In order to address deficiencies in previous surface water sampling and analysis data used in the 1997 Tier I risk assessment, additional surface water sampling was performed at the creek between September 1999 and June 2000. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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 amphibians.

9.5.1 Sunfish Lake Findings

The management objective for Sunfish Lake is to prevent unacceptable toxic effects in the lake and effects that reduce the health of wildlife associated with the lake (see section 2.1). In summary, the risk assessment identified the potential occurence of some adverse toxicological effects in ecological entities at the site. Of primary concern are adverse changes in the survival, growth, and reproduction of benthic organisms due to elevated sediment chromium, lead, and zinc concentrations. Of secondary concern are adverse changes in the reproductive potential and productivity of aquatic mammals due to elevated exposures to aluminum. Some uncertainties remain due to limitations in the evidence.

Chemicals of Concern: The selected surface water COCs evaluated in the risk characterization of Sunfish Lake were cadmium, mercury, and zinc. The selected sediment COCs were aluminum, chromium, lead, vanadium, and zinc. These COCs were found at concentrations greater than background and toxicological benchmark concentrations.

Based on collected data reviewed in this report, the spatial distribution of the sediment and surface water COCs does not necessarily emanate from the banks along the northcentral shoreline (near site H-1). There is little spatial variability in the concentrations of surface water COCs around the lake and the same appears to be true for sediment aluminum, vanaduim, and zinc (see Appendix H). However, chromium and lead concentrations appear to be higher in the middle section of the lake adjacent to the unpermitted landfill. The temporal distribution of surface water concentrations (collected over a one-year period between 1999-2000) did show significant differences between certain quarters of sampling. Also, analytical results from the second quarter of sampling (January 2000), showed all metals in surface water had slightly higher concentrations than during any other quarter.

In summary, adverse effects were found to be either not apparent or unlikely in four of the five endpoints assessed. The risk assessment identified that elevated sediment concentrations of chromium, lead, and zinc pose a potential risk for benthic organisms, but other data suggest that there may be sufficient sulfide and organic material to render lead and zinc biologically unavailable. Toxicity test results are consistent with a conclusion of little to no adverse effects, but limitations in the tests have been identified. Each assessment endpoint is discussed below.

Fish, aquatic invertebrates, and algae: The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. Based on this line of evidence at Sunfish Lake, adverse toxicological effects are possible, albeit unlikely, due to cadmium, mercury, and zinc. No COC concentrations exceeded their respective benchmarks.

Benthic organisms: The assessment endpoint (survival, growth, and reproduction of benthic organisms) was measured both directly and indirectly. The potential for adverse toxicological effects in benthic organisms at Sunfish Lake was evaluated using four lines of evidence, as shown.

- Comparison of sediment COC concentrations to effects-based benchmarks for benthic organisms
- Benthic community evaluations
- Metal bioavailability studies
- Sediment toxicity tests

Findings from two benthic community surveys indicate that the benthic community in Sunfish Lake is dominated by pollution tolerant species indicating less than optimum conditions (i.e., degraded or enriched sediments.) However, when compared to the reference location, it appears that both sites are affected by organic overenrichment and that Sunfish Lake is not different than the reference locations in terms of diversity and pollution tolerant species.

Elevated sediment concentrations of chromium, lead, and zinc indicate a potential risk for benthic organisms, however other data suggest that there may be sufficient AVS and other organic material to bind lead and zinc rendering them biologically unavailable. Available toxicity test results are consistent with a conclusion of little to no adverse effects; however, there are important limitations on those test results.

Moderate uncertainty remains as to whether adverse effects are occurring due to limitations in this evidence. First, concentrations of AVS have been shown to vary by 2 orders of magnitude and have been correlated with changes in the temperature of the overlying water. Without an SEM/AVS evaluation performed during the most vulnerable time of the year (i.e., winter), a hypothesized link between lead and zinc and this degraded condition cannot be fully explored.

Second, the sediment toxicity tests were conducted using two standard test species: *Chironomus riparius* and *Hyalella azteca*. Based on the *Chironomus riparius* sediment toxicity tests, benthic organisms in Sunfish Lake would appear to have both acceptable survival and cumulative emergence rates, but slightly delayed emergence times when compared to the reference (Snail Lake). However the biological significance of this small difference in time to emergence between the lakes is not known. Unfortunately, growth was not acceptably measured in the short-term tests (14-day).

Unfortunately, the *Hyalella azteca* tests were found by USACHPPM to be invalid and were rejected for risk characterization purposes. Even though toxicity tests are considered stronger lines of evidence than chemistry and species composition data alone, testing in more than one species is usually necessary to adequately characterize risk. Additionally, there are other data quality concerns regarding the toxicity tests (see Appendix R). These uncertainties prohibit the risk characterization from placing sufficient weight on the tests to counter indications of ecological risk from the other, traditionally weaker lines of evidence.

Last, conclusions cannot be made about the potential risk posed by vanadium, if any, because vanadium-specific sediment quality benchmarks are unavailable, the SEM/AVS evaluation does not address the bioavailability of vanadium, and the toxicity tests were not designed to link specific COCs to test observations.

Amphibians: The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species using the Frog Embryo Teratogenesis Assay-Xenopus (FETAX) protocol, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to metals from Site H (the suspected source area) is not apparent. Additionally, data collected by Jannett (1997) and others appear to corroborate a conclusion that adverse impacts of chemical exposure to amphibians are unlikely.

Moderate uncertainty remains as to whether adverse effects have or are occurring due to limitations in the evidence including, but not limited to, a lack of multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, laboratory analysis for metals concentrations in surface water used in the assays was not conducted, and laboratory tests used species not found at the site. Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations. The laboratory data investigating embryo mortality (a sensitive life stage), developmental effects, and sensitive indicators of stress (immunocharacterization assays) suggest that surface water from Sunfish Lake is not toxic to frogs, yet sediment exposures were not specifically evaluated.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap in most risk assessments. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of this risk assessment. Nonetheless, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 63%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation.

Wading birds: The assessment endpoint for wading birds (reproductive potential and productivity of wading birds) was not measured directly. 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—ratios between the exposure dose and the dose associated with either no-observable effects or the lowest known observable effects from laboratory studies. Based on this line of evidence at Sunfish Lake, adverse toxicological effects are possible, albeit unlikely, in Wading birds due to exposure to aluminum, cadmium, chromium, lead, mercury, vanadium, and zinc because no hazard quotients exceeded unity.

Aquatic Mammals: The assessment endpoint for aquatic mammals was not measured directly. 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. Based on this line of evidence at Sunfish Lake, adverse toxicological effects due to cadmium, lead, mercury, vanadium, and zinc are possible, albeit unlikely, in aquatic mammals because the HQ ratios are not greater than unity. Population-level effects are possible, albeit unlikely, for chromium exposures because the average exposure in the population is less than exposures known to be associated with adverse health effects. However, for aluminum, it appears that the form of aluminum found at the site may influence the outcome. Assuming that mammals are exposed to aluminum chloride, then the data suggest that population-level adverse effects are possible because the average exposure would be 4-times greater than exposures known to be associated with adverse health effects. On the other hand, assuming they are exposed to aluminum hydroxide, then the data suggest that population-level adverse effects are unlikely given the magnitude of the available HQ ratios associated with "safe" exposures. (Note: exposure levels known to be associated with effects for this form are not known). Based on supporting information provided earlier in the report regarding the expected form of aluminum at the site, the aluminum hydroxide-based conclusion is considered to be more site-specific. However, there remains moderate uncertainty as to whether adverse effects are occurring due to limitations in the evidence.

9.5.2 Sunfish Lake Recommendation

The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

9.6 ROUND LAKE (CURRENT SCENARIO)

Round Lake is approximately 125 acreas and is located just southwest of the TCAAP. The lake is under the management of the U.S. Fish and Wildlife Service as a unit of the Minnesota Valley National Wildlife Refuge. 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 tenant of the building noticed that many of the floor drains were still connected to the storm sewer and has since remedied the situation.

The Tier I risk assessment (USACHPPM 1997) identified potential sediment-related risks, which required further investigtion. Therefore, sediment toxicity tests were conducted using 1999 field-collected sediments. In addition, the sediment-metal bioavailability study and the toxicity tests from the southern portion of Round Lake (USACHPPM 1998a,b) were incorporated into the Tier II risk characterization. In order to address deficiencies in previous surface water sampling and analysis data used in the 1997 Tier I risk assessment, additional surface water sampling was performed at the creek between September 1999 and June 2000. No previous assessment investigated possible effects in the amphibian populations. Therefore, 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 amphibians.

9.6.1 Round Lake (Currrent) Findings

The management objective for Round Lake is to prevent unacceptable toxic effects to benthos, fish, wildlife, and effects that reduce habitat quality in the lake. In summary, the risk assessment identified the occurence of adverse toxicological effects in ecological entities at the site. Of primary concern is adverse changes in the survival, growth, and reproduction of benthic organisms due to elevated sediment chromium, copper, silver, and zinc concentrations. The secondary concern is adverse changes in the survival, growth, and reproduction of fish, aquatic invertebrates, and algal species due to elevated concentrations of barium in surface water. The last concern is adverse changes in the reproductive potential and productivity of aquatic mammals due to elevated exposures to silver. Some uncertainties remain due to limitations in the evidence.

Chemicals of Concern: The selected surface water COCs evaluated in the risk characterization of Round Lake were barium, cadmium, and zinc. The selected sediment COCs were cadmium, chromium, copper, lead, silver, vanadium, zinc, and PCBs. The inorganic COCs were found at concentrations greater than background and toxicological benchmark concentrations. Though PCBs were not detected in the sediment during the last sampling event, historical levels were high and PCBs have been detected in fish inhabiting the lake.

The spatial distribution of the COCs emanates from the northeastern banks, where TCAAP drain systems historically released water and other materials. The COC concentrations decrease with increasing distance away from these areas and the sample data support this. Lead in the sediment is an exception, having two high concentration locations (RL-18 and RL-19) elsewhere in the lake. For PCBs, it is important to note that while previous sampling showed detections in the northern littoral zone, the most recent and extensive sampling effort (1992) found no valid detections of PCBs in the sediments of the lake at the locations of the historical detections. The temporal distribution of barium, cadmium, and zinc in surface water did show significant variability during certain quarters of recent sampling. Also, analytical results from the second quarter of sampling (January 2000), showed that all metals (except mercury) in surface water had slightly higher concentrations than during any other quarter.

In summary, the risk assessment 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 related to these COCs remain. Adverse effects were not apparent for amphibians and are unlikely for waterfowl and wading birds. Each of the assessment endpoints is discussed below. Additionally, there may be a potential for adverse toxicological effects due to exposure to PCBs in mink, wading birds, and belted kingfishers, but the potential is low for both the littoral and profundal areas of Round Lake because the exposure estimates are not greater than toxicity benchmarks

<u>Fish, aquatic invertebrates, and algae:</u> The assessment endpoint (survival, growth, and reproduction of fish, aquatic invertebrates, and algal species) was not measured directly. The potential for adverse toxicological effects in these ecological entities is inferred by comparing site water concentration data to toxicity benchmarks. Based on this line of evidence at Round Lake under the current scenario, it appears that there is a potential for adverse toxicological effects due to elevated barium concentrations in the surface water, and some effects to fish may be possible from PCB concentrations in the northern littoral sediments. However, there remains high uncertainty as to whether adverse effects have or are occurring due to limitations in the evidence. The barium and PCB risk estimates contain a high degree of uncertainty due to the lack of a sufficient toxicity database. For barium only one study was found regarding aquatic toxicity, and for PCBs the tissue concentration associated with adverse biological effects in fish was based on one study with channel catfish. Also, the fish PCB residue data is old (from 1981 and 1988) and may not reflect current conditions. The most recent sampling effort did not detect PCBs in the sediment. The best estimate of the PCB-based health risk of developmental and reproductive problems for the lake's black bullhead is that the threat is low. The threat for these fish is limited to the northern littoral areas, so in this sense the extent of the threat is small. Given the PCB residues found in fish tissue, it can be predicted that growth and survival are not impacted. This is corroborated by the health and sizes of the fish collected by the USFWS for tissue analysis (i.e. full grown adults). No residue-based reproductive endpoints exist, however, and a definitive conclusion about reproduction using this measure cannot be made.

The primary source of COCs has been eliminated; so further contamination to the lake from TCAAP is not expected. A comparison of the surface water barium concentration in Round Lake to that in Snail Lake showed statistically greater concentrations at Round Lake compared to Snail (see Tables A45 and A48). However, it should be noted that exposure does not necessarily indicate effect, and that the central tendency concentration for barium at the reference site was itself 12.3 times higher than the chronic benchmark.

Benthic organisms: -The assessment endpoint (survival, growth, and reproduction of benthic organisms) was measured both directly and indirectly. The potential for adverse toxicological effects in benthic organisms at Round Lake was evaluated using four lines of evidence, as shown.

- Comparison of sediment COC concentrations to effects-based benchmarks for benthic organisms
- Benthic community evaluations
- Metal bioavailability studies
- Sediment toxicity tests

Findings from two benthic community surveys indicate that the benthic community in Round Lake is a mix of species tolerant to organic enrichment and adverse water quality conditions (typical of a eutrophic pond). Snail Lake (reference site) was not determined to be eutrophic. However, samples from Snail Lake did indicate that two groups dominate its benthic community: species typical of organic environments (oligochaeta), and those requiring organic environments (amphipods). It appears that Round Lake and the reference location may suffer from the same issue of organic enrichment.

Elevated concentrations of chromium, copper, silver, and zinc may be the drivers of the confirmed risk for benthic organisms in the northern portion of Round Lake. The most recent sediment sampling effort could not confirm the presence of PCBs in the surfical sediment (0-1 ft. depth); however, historical data suggest that PCBs posed probable risks for benthic organisms in the past. Additionally, other data suggest that there may not be sufficient AVS and other organic material to render copper and zinc biologically unavailable in some portions of this northern area. The available toxicity test results are consistent with these results. Risk estimates provided by the toxicity tests indicate that the northern sediments cause adverse effects in benthic organisms. However, there are important limitations on those test results.

Moderate uncertainty remains as to whether adverse effects are occurring due to limitations in this evidence. First, concentrations of AVS have been shown to vary by 2 orders of magnitude and have been correlated with changes in the temperature of the overlying water. Without an SEM/AVS evaluation performed during the most vulnerable time of the year (i.e., winter), a hypothesized link between lead and zinc and this degraded condition cannot be fully explored.

Second, the sediment toxicity tests were conducted using two standard test species: *Chironomus riparius* and *Hyalella azteca*. Based on the *Chironomus riparius* sediment toxicity tests, benthic organisms in the northern portion of Round Lake appear to have lower cumulative survival and emergence rates than controls and reference sediments, and slightly delayed emergence times when compared to the reference (Snail Lake). However, the biological significance of this difference in time to emergence between the lakes is not known. Unfortunately, growth was not acceptably measured in the short-term tests (14-day). Additionally, sediment collected from the top six inches portrays effects that are a result of current as well as historic contamination. The TCAAP source of contaminants to Round Lake has not been operation for a number of years so the toxicity tests represent the aggregate toxicity without differentiating the source.

Unfortunately, the *Hyalella azteca* tests were found by USACHPPM to be invalid and were rejected for risk characterization purposes. Even though toxicity tests are considered stronger lines of evidence than chemistry and species composition data alone, testing in more than one species is usually necessary to adequately characterize risk. Additionally, there are other data quality concerns regarding the toxicity tests (see Appendix R). These uncertainties limit the confidence that can be placed on these tests.

Last, conclusions cannot be made about the potential risk posed by vanadium, if any, because vanadium-specific sediment quality benchmarks are unavailable, the SEM/AVS evaluation does not address the bioavailability of vanadium, and the toxicity tests were not designed to link specific COCs to test observations.

Amphibians: The assessment endpoint (survival, development, and reproduction of amphibians) was measured indirectly using *Xenopus laevis* (African clawed frog) as the test species using the Frog Embryo Teratogenesis Assay-Xenopus (FETAX) protocol, immunological assays, and histopathological assessments. Additionally, some direct field observations were made to evaluate the presence of an amphibian community at TCAAP. Based on a synthesis of the findings provided by these lines of evidence, the occurrence of adverse toxicological effects in amphibians due to contaminants transported from the TCAAP outfall is not apparent. Additionally, data collected by Jannett (1997) and others appear to corroborate a conclusion that adverse impacts of chemical exposure to amphibians are unlikely.

Moderate uncertainty remains as to whether adverse effects have or are occurring due to limitations in the evidence including, but not limited to, a lack of multiyear data, actual density of amphibians was not measured, season/weather variations, limited access to lakes, laboratory analysis for metals concentrations in surface water used in the assays was not conducted, and laboratory tests used species not found at the site. Additionally, amphibian specific surface water benchmarks are not available for comparison with site water COC concentrations. The 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, yet sediment exposures were not specifically evaluated.

Although uncertainties exist, as with any method, the amphibian study was designed to provide information on a sensitive ecological receptor that is generally not evaluated and simply reported as a data gap in most risk assessments. Thus, even though these data contain limitations and unacceptable statistical power for some comparisons, the availability of this data for characterizing risk can be considered a strength of this risk assessment. Nonetheless, these negative findings must be considered cautiously because the power of the statistical comparisons was lower than expected. The power for the mortality endpoint was 63%, and for malformation the power ranged from 24% to 67%, depending on the type of malformation.

Aquatic Mammals Exposed to Inorganic COCs: The assessment endpoint for aquatic mammals was not measured directly. 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. Based on this line of evidence at Round Lake, it does not appear that population-level toxicological effects are expected due to exposure to barium, cadmium, chromium, lead, vanadium and zinc. However, it does appear that there is a potential for adverse toxicological effects due to exposure to silver. The HQ ratios for silver range from 11 to 150. This indicates that on average the population's exposure is estimated to be 11-times greater than exposures known to be associated with adverse health effects. However, the concentrations of two of the 18 sample locations in Round Lake are skewed very high relative to the rest of the samples. These two locations are not found near the shoreline where muskrat dens at the lake are located, nor where cattails grow. This suggests that the true population-average exposure would be less than exposures known to be associated with adverse health effects. That is, removing these two locations changes the HQ range from 11-150 to 0.2-4. Regardless of the model predictions, high uncertainty as to whether adverse effects are occurring due to limitations in this evidence.

Wildlife Exposed to PCBs: The assessment endpoint for fish-eating wildlife exposed to PCBs was not measured directly. The potential for adverse changes in the assessment endpoint is inferred by comparing estimates of fish PCB concentrations to estimates of food-based toxicological benchmarks for PCBs.

Based on the line of evidence used to estimate risk for these receptors at Round Lake, there may be a potential for adverse toxicological effects due to exposure to PCBs in mink, wading birds, and belted kingfishers, but the potential is low for both the littoral and profundal areas of Round Lake because the exposure estimates are not greater than toxicity benchmarks. Population-level effects are possible, albeit unlikely for mink, wading birds, and belted kingfishers because the maximum exposure to the population is less than exposures known to be associated with adverse health effects.

9.6.2 Round Lake (Current) Recommendation

The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

9.7 ROUND LAKE (FUTURE SCENARIO)

The current USFWS management plan for Round Lake (USFWS 1982 and 1998) identifies two potential strategies for improving habitat quality to attract higher numbers of migrating wildlife, especially waterfowl and shorebirds. In summary, the preferred course calls for the implementation of actions to increase the ratio of emergent vegetation to open water from the existing 10:90 to a more productive ratio of about 50:50 by initiating a complete draw down of the lake. The USFWS refers to this plan as their *optimum wildlife alternative*. This completed draw down will de-water the lake, which will consolidate the sediments and allow for emergent plants to germinate and take root. Water levels would then be brought up with the emergent growth to normal pool levels. This pool level would be maintained for several years until emergent plants become sparse again. Additional drawdowns would not de-water the lake to the same extent.

9.7.1 Round Lake (Future) Findings

The management objective for Round Lake is to prevent unacceptable toxic effects to benthos, fish, wildlife, and effects that reduce habitat quality in the lake. In summary, the risk assessment identified the potential for the occurence of adverse toxicological effects in ecological entities under the future scenario. This potential is driven by a presumption of increased exposures for all ecological entities to bioavailable fractions of sediment-based contaminants. However, significant uncertainty remains due to limitations in the evidence. Without knowing the magnitude of these future processes during drawdown and reflooding, a specific risk description of the long-term adversity that the COCs may pose under the future scenario remains ellusive.

Expected Changes in Sediment Physiochemical Characteristics: The future physiochemical characteristics of the sediment in Round Lake are some of the most important factors to determine if there will be an increase in ecological risk from exposure to sediment COCs as a result of the future management practice. In the first year of the management plan, there will be a complete drawdown, exposing a large majority of sediment in the northern section of the lake to the ambient air. Even though the current status of the lake indicates that metals may be bound to some extent and unavailable to biota, drawdown of the lake could alter this situation.

One of the results of drawdown is an acceleration of the oxidation (change in the oxidation-reduction state) and breakdown of the muck that accumulates on the bottom of the lake as the sediment becomes exposed to the ambient air. There are also environmental factors that may have an effect on sediment physiochemistry as a result of drawdowns. These factors (i.e., sediment drying, bioturbation, pH, and total organic carbon) may affect the bioavailability and spatiotemporal distribution of sediment COCs.

The environmental factors affecting sediment drying properties can include the ambient air velocity, amount of rainfall during drawdown, sediment organic carbon content, and algal and plant infestations. Kadlec (1989) evaluated the effects of drawdowns on freshwater marsh sediments during a five year period. He found that neither drawdown nor flooding changed the water content of the sediments or the organic matter content, but sediment nutrients increased during drawdown. However, he did find that the bottom sediment surface sank (consolidated) in the first year of drawdown but did not change significantly in the second year. He attributed no change in water content of the sediment to heavy accumulation of litter and mats of filamentous algae, which retard the drying. Kadlec also stated that organic sediments dry slowly and retain large amounts of water, 200% or more by weight, even after prolonged (up to 2 years) absence of surface water. The high organic carbon content in Round Lake sediment (average concentration = 144,000 mg/kg (USACHPPM 1998a)), may not permit drying of the sediment, such that it remains a wet muck.

Based on the current information we have about Round Lake sediments such as the high organic carbon content that may prevent sediment drying and not allow for oxidation of the sediment and release of bound metals, the sandy composition of subsurface sediments that is related to sediment drying and crack formation, the accumulation of algae and plant material that will collapse on the sediment during drawdown, and the information available in the literature about sediment drying, it seems unlikely that the sediments in the northern section of Round Lake would release deep sediment constituents to the surface water in concentrations greater than what is currently present. However, there is still the potential that exposed sediment could become oxidized due to wind mixing in shallow water areas and wind movement over exposed sediments, and release surface sediment COCs to the surface water. This sediment then becomes a secondary source of COCs from the sediment to the surface water and then an exposure concern for biota.

Chemicals of Concern: Since a continuing TCAAP source does not exist for contaminants, evaluated COCs for the future scenario were the same as those identified for the current scenario with the exception of the possibility of additional sediment contaminants present in sediments deeper than the surface sediments (0-1ft). Appendix 1 contains tables showing data collected during the OU-2 Feasibility Study for various sediment depth intervals up to a depth of 6 feet below the sediment surface. Based on a review of these tables, concentrations of the majority of the chemicals either get smaller or stay relatively equivalent in the deeper sediment intervals. This condition indicates that if the sediments are disturbed such that the deeper sediments are exposed or otherwise transport their chemicals to the surface, the concentrations would not be expected to increase. However, several heavy metals were detected in the deeper sediments at concentrations greater than those found in the top one foot. These chemicals (cadmium, copper, lead, manganese, nickel, vanadium, and zinc) may potentially pose an exposure problem in the future and were added to the COC list for this scenario if they were not already current scenario COCs. PCBs are also known to have existed in the deep sediment.

If the sediments become oxidized during the drawdown scenario, it is expected that sediment COCs (metals) would then become bioavailable to sediment dwelling organisms, aquatic plants, and more accessible to biota that may accidentally ingest sediment. The empirical concentrations of COCs in the sediment are not expected to change under the oxidizing condition, only the COC availability, and thus the potential for adverse biological effect. If the sediments dry, forming cracks that allow deep sediment constituents to be transported to the surface, then the COC concentration in sediment is expected to increase. If these newly transported COCs are oxidized, then it is expected that the bioavailable fraction of these COCs will increase, as will the potential for adverse biological effect to biota. Once the water level begins to be increased after the drawdown, surface water concentrations of what are currently sediment COCs are expected to increase because what was once bound to sediment would likely be oxidized and released into the dissolved phase. If this release of sediment COCs occurs upon refilling the lake, barium, cadmium, and zinc would no longer be the only surface water COCs. Other metals would also have to be considered. Due to the speculative nature of what may occur upon drawdown, it is difficult to predict what effect any of the above scenarios would have on biota at Round Lake in the short time period that drawdown would occur and refilling would begin (3 months).

Fish, aquatic invertebrates, and algae: Based on the information currently available it is assumed that if the USFWS plan is implemented, then adverse toxic effects may occur in fish, aquatic invertebrates, and algal species. Under the current scenario, it appears that adverse toxicological effects are possible due to elevated barium concentrations. Without knowing how the Round Lake sediments will oxidize or dry, or how other chemical reactions will occur during drawdown, specific risk estimates identifying which metals may pose excess risk can not be made. It is only expected that there may be a possible increase of COCs in the surface water upon refilling of the lake.

Benthic organisms: Under the future scenario, it is expected that the degree of adversity for benthic organisms may possibly increase in the short term. No evidence is available as to whether to expect longer-term toxicity-related effects in benthic organisms. COCs in the sediment may change if deep sediment contaminants move to the surface sediment by the process of capillary action during sediment depuration. Oxidation of surface sediment may release contaminants to the surface water, thus changing the spatial distribution in the surficial sediment. Without knowing the magnitude of these future processes during drawdown and reflooding, a specific risk description of the long-term adversity that the COCs may pose to benthic organisms under the future scenario remains ellusive.

Amphibians: It is expected that if COC concentrations increase during drawdown, then exposure would possibly increase. After full drawdown, exposure to amphibians may be increased or stay the same as the current scenario. Under the current scenario, the occurrence of adverse toxicological effects in amphibians due to the COCs from TCAAP that were transported to Round Lake via the outfall (the suspected source area) are not apparent. Additionally, data collected by Jannett (1997) and others (as discussed in Appendix M) appear to corroborate a conclusion that adverse impacts of chemical exposures to amphibians are unlikely. It is not clear whether any adverse effects would occur. If so, it is not certain that the effects would be detectable using a study design similar to that employed in 1999.

Aquatic Mammals: It can be expected that Round Lake is capable of supporting approximately 450 adult muskrats under the current scenario. This is a very conservative estimate, assuming that there is ample food and that the whole of the northern, eastern and southeastern portions of the lake is inhabitable. However, once drawdown is initiated it is expected that the muskrat population will decrease from its current number due to loss of protected dens and increased predation. Upon refilling of the lake the number of muskrats may again increase. Exposure of muskrats to the sediment may also increase during drawdown as they consume cattail roots that are no longer submerged. During drawdown, the plants may also be able to take up more COCs as the COCs become bioavailable under oxic conditions. This would also possibly increase muskrat exposure to COCs. Mucky conditions of the sediments may also present increased exposures when the animals are grooming. Upon refilling of the lake the exposure to the COCs is expected to either stay the same or increase as compared to the current Round Lake scenario.

It is possible that during the drawdown period the mink population may increase from its current number due to increased availability of food (muskrats). However, no quantitative estimate of population increase can be made.

Waterfowl and Wading birds: If bioavailable concentrations of COCs increase in the sediment and surface water, exposure to waterfowl will possibly be increased, mainly through increased COC concentrations in their food items (i.e., invertebrates and vegetation). Also, changes in the spatial distribution of contaminants will also change the exposure to waterfowl. During the first year of drawdown, waterfowl may not yet be attracted to the lake because the water volume may be too small, the ideal vegetation may not yet be present, and the mucky condition of exposed sediments may not be ideal for the birds. However, those birds such as mallards that do frequent the lake would most likely be less exposed to sediment COCs if they are feeding in the middle of the lake, where COC concentrations are lower than in the northern section. Once the lake is refilled and vegetation has improved, more birds are expected to be attracted to the lake and thus exposure of COCs to the waterfowl population would increase.

Without site-specific wading bird activity data, it is difficult to predict how wading birds will be utilizing Round Lake. However, based on life history characteristics some generalizations can be made. Because great blues nest colonially, local population density varies with the availability of suitable nesting habitat as well as foraging habitat (USEPA 1993). It is difficult to determine the amount of foraging habitat available at Round Lake without field data. However, based on the available information it is possible that 4-5 wading birds may dominate at Round Lake. It is likely that in the future scenario, no more wading birds will be attracted to the lake due to the solitary hunting style of the birds. Also, the habitat for these birds is not the focus of the drawdown and will not directly be improved.

9.7.2 Round Lake (Future) Recommendation

The risk managers should use the risk assessment results to determine if no further action is appropriate, or whether a feasibility study should be conducted to consider a remedy.

10. STUDY PERSONNEL

Efforts from the following staff were integral for the preparation of this report. The main project officers from the Environmental Health Risk Assessment Program were Lia M. Gaizick and Matthew J. McAtee. Staff from other programs at USACHPPM were Colleen J. Brust, Mark S. Johnson, Gene Sinar, Robert J. Valis, and Robyn B. Lee. Our point of contact for technical questions is Ms. Lia M. Gaizick at DSN 584-2953 or commercial (410) 436-2953.

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APPENDIX A

TABLES

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Table A1. Summary of Tier I Re-Assessment for Marsden Lake (North)*

Preliminary COPCs	Non-Detection		Data Quality		Tier I Back-ground Sediment	Eliminated Prior to Toxicity Screen	Tier I Hazard Quotients ** (Based on maximum concentration)					COPCs
	Water	Sed.	Water	Sed.			Aquatic organisms	Benthic Organisms	Wading Birds	Waterfowl	Aquatic Mammals	
Al					×		>1	×	>1	×	>1	Al
Sb							×	>1	nd	nd	nd	Sb
As							×	>1	>1	×	>1	As
Ba					×		>1	×	>1	×	>1	Ba
Be	×	×	×	×	×	×	×	×	×	×	×	×
Cd							×	>1	>1	×	>1	Cd
Cr					×		×	×	>1	×	×	Cr
Co		×		×	×		×	×	×	×	×	×
Cu							×	>1	>1	×	×	Cu
Pb					×		>1	×	>1	nd	nd	Pb
Mg					×		×	×	×	×	×	×
Mn					×		>1	×	nd	nd	nd	Mn
Hg							>1	×	>1	>1	>1	Hg
Ni					×		×	×	×	×	×	×
Se	×		×		×	×	×	×	×	×	×	×
Ag		×		×	×		×	×	nd	nd	×	Ag
Tl		×		×	×		×	×	nd	nd	nd	Tl
Zn					×		×	×	>1	×	×	Zn
SVOCs	×†		×†		nd		×	nd	nd	nd	nd	SVOCs
OC Pesticides	×		×		nd		×	nd	nd	nd	nd	OC Pesticides
PCBs	×	×	×	×	×	×	×	×	×	×	×	×

* Moving from left to right until the toxicity screen, the first "×" indicates when in the process an analyte was *eliminated* from further evaluation as a preliminary chemical of potential concern (COPC); additional "×" marks across the row indicate that the remaining evaluations were not performed. To be eliminated prior to the toxicity screen, the preliminary COPC must have "×" marks for both water and sediment based on earlier criteria.

** The "×" marks indicate that either an HQ was less than 1.0 or that an HQ was not calculated because it was previously eliminated as a water or sediment COPC (for aquatic and benthic organisms only).

nd: Indicates that data was missing; either no Tier I background data for comparison, no site concentration data, no Tier I toxicity data, or missing constants for modeling the dose.

†: Bis(2-ethylhexyl)phthalate) was the exception. It was detected in site water samples, but later eliminated because the detections were determined to be due to laboratory contamination.

Table A2. Summary of Tier I Re-Assessment for Marsden Lake (South)*

Preliminary COPCs	Non-Detection		Data Quality		Tier I Background Sediment	Eliminated Prior to Toxicity Screen	Tier I Hazard Quotients ** (Based on maximum concentration)					COPCs
	Water	Sed.	Water	Sed.			Aquatic organisms	Benthic Organisms	Wading Birds	Waterfowl	Aquatic Mammals	
Al							>1	×	>1	>1	>1	Al
Sb							>1	>1	nd	nd	nd	Sb
As							×	>1	>1	×	>1	As
Ba					×		>1	×	>1	×	>1	Ba
Be	×	×	×	×	×	×	×	×	×	×	×	×
Cd							×	>1	>1	×	>1	Cd
Cr							×	>1	>1	>1	>1	Cr
Co							×	×	nd	nd	×	Co
Cu							×	>1	>1	×	×	Cu
Pb							>1	>1	>1	Na	Na	Pb
Mg					×		×	×	×	×	×	×
Mn					×		×	×	nd	nd	nd	Mn
Hg							>1	×	>1	>1	>1	Hg
Ni					×		×	×	×	×	×	×
Se	×				×	×	×	×	×	×	×	×
Ag		×		×	×		×	×	nd	nd	×	Ag
Tl		×		×	×		×	×	nd	nd	nd	Tl
Zn							×	>1	>1	×	×	Zn
SVOCs	×			†			×	nd	nd	nd	nd	SVOCs
OC Pesticides	×						×	nd	nd	nd	nd	OC Pesticides
PCBs	×	×	×	×	×	×	×	×	×	×	×	×

* Moving from left to right until the toxicity screen, the first "×" indicates when in the process an analyte was *eliminated* from further evaluation as a preliminary chemical of potential concern (COPC); additional "×" marks across the row indicate that the remaining evaluations were not performed. To be eliminated prior to the toxicity screen, the preliminary COPC must have "×" marks for both water and sediment based on earlier criteria.

** The "×" marks indicate that either an HQ was less than 1.0 or that an HQ was not calculated because it was previously eliminated as a water or sediment COPC (for aquatic and benthic organisms only).

nd: Indicates that data was missing; either no Tier I background data for comparison, no site concentration data, no Tier I toxicity data, or missing constants for modeling the dose.

†: Only Butylbenzylphthalate was eliminated. It was detected in site sediment samples, but later eliminated because the detections were found to be due to laboratory contamination.

Table A3. Summary of Tier I Re-Assessment for Pond G*

Preliminary COPCs	Non-Detection		Data Quality		Tier I Back-ground Sediment	Eliminated Prior to Toxicity Screen	Tier I Hazard Quotients ** (Based on maximum concentration)				COPCs
	Water	Sed.	Water	Sed.			Aquatic organisms	Benthic Organisms	Wading Birds	Aquatic Mammals	
Al							>1	×	>1	>1	Al
Sb							×	×	nd	nd	Sb
As							×	>1	×	>1	As
Ba					×		>1	×	×	>1	Ba
Be		×		×	×		nd	nd	nd	×	Be
Cd					×		×	×	×	>1	Cd
Cr							×	>1	×	>1	Cr
Co		×		×	×		×	×	Na	Na	Co
Cu							×	>1	×	×	Cu
Pb							>1	>1	×	>1	Pb
Mg					×		×	×	×	×	×
Mn					×		×	×	nd	nd	Mn
Hg	×		×				×	×	×	×	×
Ni					×		×	×	×	×	×
Se		×		×	×		×	×	nd	nd	Se
Ag	×	×	×	×	×	×	×	×	×	×	×
Tl		×		×	×		×	×	nd	nd	Tl
Zn							×	×	×	×	×
SVOCs	×†		×†				×	nd	nd	nd	SVOCs
§OC Pesticides	×		×		nd		×	>1	nd	nd	OC Pesticides
PCBs	×	×‡	×				×	>1	0.01	0.17	Aroclor-1248

* Moving from left to right until the toxicity screen, the first "×" indicates when in the process an analyte was *eliminated* from further evaluation as a preliminary chemical of potential concern (COPC); additional "×" marks across the row indicate that the remaining evaluations were not performed. To be eliminated prior to the toxicity screen, the preliminary COPC must have "×" marks for both water and sediment based on earlier criteria.

** The "×" marks indicate that either an HQ was less than 1.0 or that an HQ was not calculated because it was previously eliminated as a water or sediment COPC (for aquatic and benthic organisms only).

nd: Indicates that data was missing; either no Tier I background data for comparison, no site concentration data, or no Tier I toxicity data.

†: Bis(2-ethylhexyl)phthalate) was the exception. It was detected in site water samples, but later eliminated because the detections were found to be due to laboratory contamination.

‡: All PCBs except Aroclor-1248 were eliminated

§ Hazard quotients were calculated on chemicals that were detected (DDT, DDD, DDE, and Dieldrin). For sediment HQs dieldrin was less than unity. Due to detection limit problems (see section 5.2.1.3) all OC pesticides could not be eliminated from further consideration as COPCs.

Table A4. Marsden Lake North Tier I Reassessment Surface Water Hazard Quotients (HQs)

Analyte	Sample Locations†									
	ML-11	ML-12	ML-13‡	ML-14	ML-15‡	ML-16	ML-17	ML-18	ML-19	ML-20
Aluminum	0.05	2.49		0.14		4.08	4.32	0.30	1.60	1.60
Antimony	0.00	0.00		0.00		0.00	0.00	0.00	0.00	0.01
Arsenic	0.02	0.04		0.01		0.03	0.08	0.03	0.05	0.05
Barium	11.32	16.32		10.53		24.21	52.63	16.58	42.11	34.21
Cadmium	0.01	0.07		0.01		0.01	0.12	0.01	0.01	0.06
Chromium	0.04	0.14		0.05		0.16	0.21	0.09	0.16	0.13
Cobalt	0.02	0.24		0.02		0.11	0.22	0.05	0.08	0.20
Copper	0.01	0.27		0.02		0.12	0.42	0.03	0.06	0.16
Lead	0.03	0.53		0.08		0.44	2.19	0.09	0.34	1.69
Magnesium	0.08	0.08		0.09		0.11	0.23	0.10	0.16	0.24
Manganese	0.11	1.34		0.03		2.04	2.65	0.86	1.30	2.24
Mercury	0.23	0.90		0.16		0.42	1.88	0.09	0.41	1.59
Nickel	0.00	0.01		0.00		0.01	0.01	0.00	0.00	0.01
Silver	0.03	0.01		0.02		0.02	0.02	0.01	0.01	0.03
Thallium	0.00	0.00		0.00		0.00	0.01	0.00	0.00	0.00
Zinc	0.00	0.03		0.00		0.08	0.14	0.04	0.02	0.05

† HQs calculated to be less than 0.01 are reported in the table as 0.00.

Na: No toxicity value was available for HQ calculations

‡ Data from sample locations ML-13 and ML-15 were moved to the Marsden Lake South data set (see section 4 for explanation)

Table A5. Marsden Lake North Tier I Reassessment Sediment Hazard Quotients (HQs)

Analyte	Sample Locations									
	ML-11	ML-12	ML-13†	ML-14	ML-15†	ML-16	ML-17	ML-18	ML-19	ML-20
Antimony*	0.04	0.04		0.04		0.06	4.00	0.04	0.04	0.04
Arsenic	0.98	1.35		0.93		0.20	0.32	0.17	1.48	1.12
Cadmium	0.82	2.33		0.85		0.83	0.82	0.83	0.83	0.83
Copper	1.75	2.69		1.06		0.21	0.50	0.11	0.38	0.39
Lead	0.48	0.84		0.84		0.18	0.12	0.13	0.55	0.32
Mercury*	0.48	0.70		0.50		0.50	0.50	0.48	0.60	0.50
Zinc	0.53	0.72		0.42		0.13	0.18	0.15	0.42	0.42

*: No background screening values were available for these compounds.

† Data from sample locations ML-13 and ML-15 were moved to the Marsden Lake South data set (see section 4 for explanation)

Table A6. Marsden Lake North Tier I Reassessment Amphibian Hazard Quotients

Analyte	SAMPLE LOCATION									
	ML-11	ML-12	ML-13†	ML-14	ML-15†	ML-16	ML-17	ML-18	ML-19	ML-20
Antimony										
Arsenic										
Cadmium	0.50	1.57		0.51		0.50	1.04	0.50	0.50	0.76
Copper										
Lead										
Mercury										
Zinc										
Magnesium										
Manganese										
Nickel										
Silver										
Thallium										
Aluminum										
Barium										
Chromium										
Cobalt	0.01	0.13		0.01		0.06	0.12	0.02	0.04	0.11

Note: Blank Cells indicate that no toxicity reference values were available for hazard quotient calculations.

† Data from sample locations ML-13 and ML-15 were moved to the Marsden Lake South data set (see section 4 for explanation)

Table A7. Marsden Lake North Tier I Reassessment Wading Bird Hazard Quotients (HQs)

Analyte	SAMPLE LOCATION									
	ML-11	ML-12	ML-13†	ML-14	ML-15†	ML-16	ML-17	ML-18	ML-19	ML-20
Aluminum	0.94	48.11		2.63		78.93	83.57	5.76	30.98	30.98
Antimony	Na	Na		Na		Na	Na	Na	Na	Na
Arsenic	21.24	50.65		16.42		40.97	108.43	33.74	57.88	67.50
Barium	35.08	50.60		32.65		75.06	163.17	51.40	130.54	106.06
Cadmium	0.58	2.87		0.58		0.58	4.95	0.58	0.58	2.63
Chromium	10.32	32.14		11.03		37.67	50.59	22.38	37.67	31.21
Cobalt	Na	Na		Na		Na	Na	Na	Na	Na
Copper	0.31	11.13		1.01		5.10	17.42	1.06	2.47	6.80
Lead	0.92	13.50		2.15		11.05	55.17	2.30	8.73	42.59
Magnesium	Na	Na		Na		Na	Na	Na	Na	Na
Manganese	Na	Na		Na		Na	Na	Na	Na	Na
Mercury	1692.23	6556.87		1163.49		3067.00	13747.83	624.15	2961.30	11632.82
Nickel	0.02	0.15		0.03		0.14	0.22	0.05	0.05	0.12
Silver	Na	Na		Na		Na	Na	Na	Na	Na
Thallium	Na	Na		Na		Na	Na	Na	Na	Na
Zinc	3.44	23.71		0.29		62.84	101.99	27.75	17.66	40.61
SVOCs	NA	NA		NA		NA	NA	NA	NA	NA
OC Pesticides	NA	NA		NA		NA	NA	NA	NA	NA

Na: One or both of the components were not available to calculate the HQ.

SVOCs and OC Pesticides did not have concentration data available; therefore HQs could not be calculated.

† Data from sample locations ML-13 and ML-15 were moved to the Marsden Lake South data set (see section 4 for explanation)

Table A8. Marsden Lake North Tier I Reassessment Waterfowl Hazard Quotients (HQs)

Analyte	SAMPLE LOCATION†									
	ML-11	ML-12	ML-13‡	ML-14	ML-15‡	ML-16	ML-17	ML-18	ML-19	ML-20
Aluminum	0.00	0.04		0.00		0.07	0.07	0.01	0.03	0.03
Antimony	Na	Na		Na		Na	Na	Na	Na	Na
Arsenic	0.15	0.21		0.14		0.04	0.07	0.03	0.23	0.18
Barium	0.03	0.04		0.03		0.07	0.14	0.05	0.12	0.09
Cadmium	0.02	0.06		0.02		0.02	0.02	0.02	0.02	0.02
Chromium	0.01	0.02		0.01		0.03	0.04	0.02	0.03	0.02
Cobalt	Na	Na		Na		Na	Na	Na	Na	Na
Copper	0.04	0.06		0.02		0.00	0.01	0.00	0.01	0.01
Lead	Na	Na		Na		Na	Na	Na	Na	Na
Magnesium	Na	Na		Na		Na	Na	Na	Na	Na
Manganese	Na	Na		Na		Na	Na	Na	Na	Na
Mercury	0.91	1.34		0.95		0.94	0.97	0.90	1.14	0.97
Nickel	0.00	0.00		0.00		0.00	0.00	0.00	0.00	0.00
Silver	Na	Na		Na		Na	Na	Na	Na	Na
Thallium	Na	Na		Na		Na	Na	Na	Na	Na
Zinc	0.27	0.36		0.21		0.07	0.10	0.08	0.21	0.21
SVOCS	NA	NA		NA		NA	NA	NA	NA	NA
OC pesticides	NA	NA		NA		NA	NA	NA	NA	NA

† HQs calculated to be less than 0.01 are reported in the table as 0.00.
Na: One or both of the components were not available to calculate the HQ.
SVOCs and OC Pesticides did not have concentration data available; therefore HQs could not be calculated.
‡ Data from sample locations ML-13 and ML-15 were moved to the Marsden Lake South data set (see section 4 for explanation)

Table A9. Mardsen Lake North Tier I Reassessment Aquatic Mammal Hazard Quotients (HQs)

Analyte	SAMPLE LOCATION†									
	ML-11	ML-12	ML-13‡	ML-14	ML-15‡	ML-16	ML-17	ML-18	ML-19	ML-20
Aluminum	7.86	403.90		22.08		662.66	701.62	48.38	260.06	260.06
Antimony	Na	Na		Na		Na	Na	Na	Na	Na
Arsenic	18.60	43.38		14.55		34.20	90.32	28.17	49.51	57.14
Barium	10.86	15.66		10.11		23.23	50.51	15.91	40.40	32.83
Cadmium	1.92	8.69		1.93		1.93	13.42	1.93	1.93	7.33
Chromium	0.20	0.62		0.21		0.73	0.97	0.43	0.73	0.60
Cobalt	0.00	0.05		0.00		0.02	0.05	0.01	0.02	0.04
Copper	0.03	0.27		0.03		0.11	0.37	0.02	0.06	0.15
Lead	Na	Na		Na		Na	Na	Na	Na	Na
Magnesium	Na	Na		Na		Na	Na	Na	Na	Na
Manganese	Na	Na		Na		Na	Na	Na	Na	Na
Mercury	0.24	0.74		0.20		0.37	1.38	0.14	0.38	1.19
Nickel	0.01	0.05		0.01		0.04	0.07	0.02	0.02	0.04
Silver	0.00	0.00		0.00		0.00	0.00	0.00	0.00	0.00
Thallium	Na	Na		Na		Na	Na	Na	Na	Na
Zinc	0.01	0.04		0.00		0.08	0.13	0.04	0.03	0.05
SVOCs	NA	NA		NA		NA	NA	NA	NA	NA
OC Pesticides	NA	NA		NA		NA	NA	NA	NA	NA

†: HQs calculated to be less than 0.01 are reported in the table as 0.00.

Na: One or both of the components were not available to calculate the HQ.

SVOCs and OC Pesticides did not have concentration data available; therefore HQs could not be calculated.

‡ Data from sample locations ML-13 and ML-15 were moved to the Marsden Lake South data set (see section 4 for explanation)

Table A10. Marsden Lake South Tier I Reassessment Surface Water Hazard Quotients (HQs)

Analyte	Sample Location†											
	ML-01	*ML-02*	*ML-03*	*ML-04*	*ML-05*	*ML-06*	*ML-07*	*ML-08*	*ML-09*	*ML-10*	*ML-13*	*ML-15*
Aluminum	0.02	0.04	0.03	0.04	0.04	0.06	0.05	8.80	0.06	0.19	0.03	0.04
Antimony	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Arsenic	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.07	0.02	0.02	0.01	0.01
Barium	10.79	10.79	11.05	10.26	10.53	11.32	11.05	23.42	11.05	11.84	10.26	9.21
Cadmium	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.17	0.01	0.01	0.01	0.01
Chromium	0.04	0.06	0.05	0.06	0.05	0.05	0.05	0.32	0.06	0.05	0.05	0.04
Cobalt	0.01	0.02	0.02	0.01	0.01	0.01	0.02	0.40	0.03	0.03	0.01	0.01
Copper	0.01	0.01	0.02	0.01	0.01	0.01	0.01	0.92	0.01	0.02	0.02	0.01
Lead	0.02	0.02	0.02	0.02	0.02	0.03	0.05	2.13	0.04	0.08	0.02	0.02
Magnesium	0.08	0.08	0.08	0.07	0.08	0.08	0.08	0.07	0.08	0.08	0.08	0.08
Manganese	0.06	0.06	0.10	0.10	0.04	0.10	0.10	0.75	0.11	0.10	0.04	0.06
Mercury	0.07	0.10	0.06	0.19	0.09	0.23	0.16	2.46	0.06	0.22	0.07	0.10
Nickel	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00
Silver	0.03	0.02	0.02	0.01	0.02	0.01	0.01	0.03	0.01	0.01	0.01	0.02
Thallium	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00
Zinc	0.00	0.15	0.14	0.00	0.00	0.00	0.01	0.21	0.00	0.01	0.04	0.01

†: HQs calculated to be less than 0.01 are reported in the table as 0.00
Na: No toxicity value was available for HQ calculations

Table A11. Marsden Lake South Tier I Reassessment Sediment Hazard Quotients (HQs)

Analyte	Sample Locations											
	ML-01	ML-02	ML-03	ML-04	ML-05	ML-06	ML-07	ML-08	ML-09	ML-10	ML-13	ML-15
Aluminum	0.21	0.11	0.19	0.14	0.06	0.16	0.24	0.21	0.26	0.24	0.15	0.14
Antimony*	0.05	0.08	0.04	0.04	0.06	0.04	0.05	0.04	0.07	0.04	0.08	0.07
Arsenic	1.58	1.22	1.83	1.13	1.38	1.12	1.33	1.63	1.67	1.43	2.00	2.00
Cadmium	2.67	1.35	1.83	0.83	0.83	0.83	0.83	0.83	0.83	0.83	2.33	2.17
Chromium	0.81	0.42	0.85	0.62	0.26	0.62	1.15	0.85	1.00	1.00	0.65	0.69
Cobalt*	0.22	0.10	0.19	0.20	0.19	0.20	0.22	0.20	0.22	0.20	0.20	0.20
Copper	1.56	0.75	1.38	1.06	0.54	1.19	3.25	1.81	1.94	1.81	1.56	1.50
Lead	1.48	1.35	1.65	0.90	0.71	1.06	1.55	1.45	1.42	1.29	2.03	2.00
Mercury*	0.90	0.75	0.80	0.60	0.65	0.70	0.55	0.85	0.90	0.75	0.80	0.85
Zinc	0.83	0.46	0.73	0.55	0.46	0.44	0.73	0.67	0.67	0.61	0.60	1.17
4,4'-DDD	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	11.88	NA

.*: No background sediment screening values were available for these compounds

Table A12. Marsden Lake South Tier I Reassessment Amphibian Hazard Quotients (HQs)

Analyte	SAMPLE LOCATION											
	ML-01	ML-02	ML-03	ML-04	ML-05	ML-06	ML-07	ML-08	ML-09	ML-10	ML-13	ML-15
Aluminum												
Antimony												
Arsenic												
Cadmium	1.46	0.77	1.03	0.50	0.50	0.50	0.50	1.34	0.50	0.50	1.3	1.2
Chromium												
Cobalt	0.22	0.10	0.20	0.20	0.19	0.20	0.23	0.41	0.23	0.21	0.2	0.2
Copper												
Lead												
Mercury												
Magnesium												
Manganese												
Nickel												
Silver												
Thallium												
Zinc												
SVOCs												
4.4'-DDD												

Note: Blank Cells indicate that no toxicity reference values were available for hazard quotient calculations

Table A13. Marsden Lake South Tier I Reassessment Wading Bird Hazard Quotients (HQs)

Analyte	SAMPLE LOCATION											
	ML-01	ML-02	ML-03	ML-04	ML-05	ML-06	ML-07	ML-08	ML-09	ML-10	ML-13	ML-15
Aluminum	2.02	1.58	1.99	1.92	1.17	2.35	2.92	171.82	3.13	5.49	1.76	1.80
Antimony	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Arsenic	18.13	19.08	19.10	18.35	17.40	20.04	22.70	86.80	20.78	23.90	15.73	16.46
Barium	33.45	33.45	34.27	31.82	32.63	35.08	34.27	72.61	34.27	36.71	31.82	28.56
Cadmium	0.71	0.64	0.71	0.67	0.62	0.67	0.76	7.37	0.73	0.73	0.68	0.69
Chromium	10.32	13.65	12.47	13.30	11.31	11.58	11.67	76.66	13.37	12.93	10.94	9.22
Cobalt	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Copper	0.42	0.41	0.73	0.42	0.52	0.48	0.53	38.25	0.35	0.69	0.65	0.44
Lead	0.54	0.60	0.65	0.64	0.50	0.92	1.33	53.65	1.01	2.03	0.71	0.71
Mercury	508.02	730.03	433.95	1375.04	666.53	1692.34	1163.51	17978.03	455.15	1586.61	529.12	708.92
Magnesium	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Manganese	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Nickel	0.02	0.03	0.06	0.02	0.01	0.02	0.02	0.55	0.03	0.03	0.01	0.02
Silver	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Thallium	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Zinc	2.78	108.77	102.05	2.97	2.96	3.09	4.61	156.03	3.66	4.93	32.53	9.66
SVOCs	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
OC Pesticides (4,4'-DDD)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0.47	NA

Na: One or both of the components was not available to calculate the HQ.

Note: SVOCs and OC Pesticides did not have concentration data available (with the exception of 4.4'-DDD); therefore HQs could not be calculated.

Table A14. Marsden Lake South Tier I Reassessment Waterfowl Hazard Quotients (HQs)

Analyte	SAMPLE LOCATION†											
	ML-01	ML-02	ML-03	ML-04	ML-05	ML-06	ML-07	ML-08	ML-09	ML-10	ML-13	ML-15
Aluminum	6.62	3.53	6.07	4.63	1.82	5.19	7.72	6.77	8.27	7.73	4.63	4.47
Antimony	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Arsenic	0.24	0.18	0.28	0.17	0.21	0.17	0.20	0.26	0.25	0.22	0.30	0.30
Barium	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.06	0.03	0.03	0.03	0.03
Cadmium	0.07	0.03	0.05	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.06	0.05
Chromium	1.28	0.68	1.34	0.98	0.42	0.98	1.82	1.38	1.58	1.58	1.04	1.10
Cobalt	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Copper	0.03	0.02	0.03	0.02	0.01	0.02	0.07	0.04	0.04	0.04	0.03	0.03
Lead	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Mercury	1.70	1.42	1.51	1.14	1.23	1.33	1.04	1.65	1.70	1.42	1.51	1.61
Magnesium	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Manganese	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Nickel	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Silver	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Thallium	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Zinc	0.42	0.25	0.38	0.28	0.23	0.22	0.37	0.36	0.33	0.31	0.31	0.59
SVOCs	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
OC pesticides (4,4'-DDD)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

†: HQs calculated to be less than 0.01 are reported in the table as 0.00

Na: One or both of the components was not available to calculate the HQ.

SVOCs and OC Pesticides did not have concentration data available; therefore HQs could not be calculated.

Table A15. Marsden Lake South Tier I Reassessment Aquatic Mammal Hazard Quotients (HQs)

Analyte	SAMPLE LOCATION											
	ML-01	ML-02	ML-03	ML-04	ML-05	ML-06	ML-07	ML-08	ML-09	ML-10	ML-13	ML-15
Aluminum	135.78	76.69	125.65	99.29	42.47	112.79	163.18	1561.36	174.87	184.74	97.99	95.32
Antimony	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Arsenic	16.62	17.04	17.67	16.36	15.81	17.74	20.16	73.67	18.90	21.26	15.04	15.64
Barium	10.35	10.35	10.61	9.85	10.10	10.86	10.61	22.47	10.61	11.36	9.85	8.84
Cadmium	2.86	2.19	2.44	1.93	1.93	1.93	1.93	19.43	1.93	1.93	2.69	2.61
Chromium	0.27	0.30	0.31	0.31	0.24	0.28	0.33	1.55	0.34	0.34	0.27	0.24
Cobalt	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.09	0.01	0.01	0.01	0.01
Copper	0.03	0.02	0.03	0.02	0.02	0.02	0.05	0.83	0.03	0.04	0.03	0.03
Lead	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Mercury	0.20	0.20	0.18	0.23	0.17	0.28	0.20	1.84	0.20	0.28	0.19	0.21
Magnesium	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Manganese	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Nickel	0.00	0.01	0.02	0.01	0.00	0.01	0.01	0.17	0.01	0.01	0.00	0.01
Silver	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Thallium	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na	Na
Zinc	0.01	0.14	0.13	0.01	0.01	0.01	0.01	0.20	0.01	0.01	0.05	0.02
SVOCs	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
OC Pesticides (4,4'-DDD)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

†: HQs calculated to be less than 0.01 are reported in the table as 0.00.

.Na: One or both of the components were not available to calculate the HQ

SVOCs and OC Pesticides did not have concentration data available (with the exception of 4,4'-DDD); therefore HQs could not be calculated

Table A16. Pond G Tier I Reassessment Sediment Hazard Quotients (HQs)†

Analyte	PG 01	PG 02	PG-03	SD-FB	SD-FD
Aluminum	0.22	0.19	0.31	0.00	0.33
Antimony*	0.05	0.04	0.04	0.00	0.04
Arsenic	0.58	0.72	1.27	0.02	1.22
Chromium	0.73	0.65	1.38	0.00	1.35
Copper	1.19	1.13	2.44	0.02	2.38
Lead	0.90	0.77	1.26	0.00	1.23
Mercury*	0.70	0.22	0.65	0.05	0.65
Zinc	0.71	0.52	0.92	0.01	0.83
Aroclor 1248	0.00	0.00	0.00	NA	6.50
4,4'-DDT	NA	4.29	NA	NA	NA
4,4'-DDD	NA	20.00	NA	NA	12.13
4,4'-DDE	NA	10.60	NA	NA	NA
Dieldrin	NA	0.03	NA	NA	NA
4-methylphenol	NA	NA	NA	NA	NA

†: HQs calculated to be less than 0.01 are reported in the table as 0.00

*: No background sediment screening values were available for these compounds

UD: chemical was undetected in the sample

NA: one or both of the components were not available to calculate the HQ

Table A17. Pond G Tier I Reassessment Surface Water Hazard Quotients (HQs)

Analyte	Sample Location				
	PG 01	PG -02	PG-03	FB-01	FD-XX
Aluminum	1.93	3.61	12.81	0.02	6.48
Antimony	0.02	0.02	0.04	0.00	0.04
Arsenic	0.07	0.07	0.10	0.00	0.08
Barium	13.43	14.74	26.32	0.03	19.52
Beryllium	Na	Na	Na	Na	Na
Cadmium	0.02	0.04	0.13	0.01	0.06
Chromium	0.08	0.12	0.32	0.01	0.18
Cobalt	0.17	0.24	0.48	0.00	0.34
Copper	0.15	0.20	0.57	0.02	0.30
Lead	0.75	1.03	2.47	0.10	1.34
Magnesium	0.05	0.05	0.06	0.00	0.01
Manganese	0.65	0.77	0.69	0.00	1.12
Mercury	0.39	0.67	2.46	0.04	1.07
Nickel	0.02	0.02	0.04	0.00	0.03
Selenium	Na	Na	Na	Na	Na
Silver	0.02	0.02	0.03	0.01	0.01
Thallium	0.00	0.00	0.01	0.00	0.00
Zinc	0.01	0.03	0.13	0.02	0.05

†: HQs calculated to be less than 0.01 are reported in the table as 0.00
Na: No Tier I toxicity value was available for HQ calculations

Table A18. Pond G Tier I Reassessment Wading Bird Hazard Quotients (HQs)

	Sample location†				
Analyte	SC-PG-SW-01	SC-PG-SW-02	SC-PG-SW-03	SC-PG-FB-01	SC-PG-SW-FD-XX
Aluminum	0.80	0.79	1.69	0.00	1.38
Antimony	Na	Na	Na	Na	Na
Arsenic	0.08	0.09	0.12	0.00	0.10
Barium	0.12	0.13	0.24	0.00	0.18
Beryllium	Na	Na	Na	Na	Na
Cadmium	0.00	0.00	0.00	0.00	0.00
Chromium	0.16	0.16	0.38	0.01	0.30
Cobalt	Na	Na	Na	Na	Na
Copper	0.00	0.00	0.01	0.00	0.01
Lead	0.07	0.08	0.16	0.00	0.11
Magnesium	Na	Na	Na	Na	Na
Manganese	0.16	0.19	0.17	0.00	0.28
Mercury	0.15	0.08	0.25	0.01	0.18
Nickel	0.00	0.00	0.00	0.00	0.00
Selenium	0.49	0.49	0.51	0.49	0.49
Silver	Na	Na	Na	Na	Na
Thallium	Na	Na	Na	Na	Na
Zinc	0.04	0.03	0.09	0.01	0.06
Aroclor 1248	0.00	0.00	0.00	0.00	0.01
SVOCs					
OC Pesticides:					
4,4'-DDT	NA	0.06	NA	NA	NA
4,4'-DDD	NA	0.31	NA	NA	0.19
4,4'-DDE	NA	0.10	NA	NA	NA
Dieldrin	NA	NA	NA	NA	NA
4-methylphenol	NA	NA	NA	NA	NA

†: HQs calculated to be less than 0.01 are reported in the table as 0.00

Na: A component of the HQ calculation was not available.

Note: SVOCs and OC Pesticides did not have concentration data available; therefore HQs could not be calculated.

Table A19. Pond G Tier I Reassessment Aquatic Mammal Hazard Quotients (HQs)

Analyte	Sample location†				
	PG-01	PG-02	PG-03	PG-FB-01	PG -FD-XX
Aluminum	456.49	585.72	2200.65	2.47	1250.65
Antimony	Na	Na	Na	Na	Na
Arsenic	74.60	74.10	102.73	1.00	85.29
Barium	12.88	14.15	25.26	0.03	18.73
Beryllium	0.10	0.10	0.24	0.10	0.11
Cadmium	2.70	4.10	14.00	1.50	6.10
Chromium	0.45	0.54	1.51	0.06	0.95
Cobalt	0.04	0.05	0.10	0.00	0.07
Copper	0.15	0.18	0.51	0.01	0.29
Lead	0.45	0.56	1.38	0.05	0.79
Magnesium	Na	Na	Na	Na	Na
Manganese	Na	Na	Na	Na	Na
Mercury	0.39	1.06	1.74	0.22	0.85
Nickel	0.09	0.11	0.20	0.00	0.14
Selenium	Na	Na	Na	Na	Na
Silver	0.00	0.00	0.00	0.00	0.00
Thallium	Na	Na	Na	Na	Na
Zinc	0.01	0.02	0.12	0.02	0.05
Aroclor 1248	0.00	0.00	0.00	0.00	0.17
SVOC					
OC Pesticides					
4,4'-DDT	NA	0.00	NA	NA	NA
4,4'-DDD	NA	0.00	NA	NA	0.00
4,4'-DDE	NA	0.00	NA	NA	NA
Dieldrin	NA	NA	NA	NA	NA
4-methylphenol	NA	NA	NA	NA	NA

†: HQs calculated to be less than 0.01 are reported in the table as 0.00.

Na: A component of the HQ calculation was not available.

Note: SVOCs did not have concentration data available; therefore HQs could not be calculated.

Table A20. U.S. Fish and Wildlife Sediment Data Used in the Tier I ERA to Select Sediment Chemicals to Evaluate

	Louisville Swamp		Chaska Lake		Rice Lake West		Grass Lake			Fisher Lake	Gravel Pit	Long Meadow Lake				Statistics and Screening Values		
Substance	001	003	005	007	009	012	014	017	020	026	051	053	056	059	063	mean	SD	(value)
Al	8190.0	4650.0	7880.0	7160.0	7990.0	6760.0	6370.0	6820.0	7620.0	5440.0	13400.0	6860.0	6200.0	6180.0	5300.0	7121.0	2018.0	10674.0
As	<5.0	<4.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	10.0	<6.0	<5.0	<5.0	<5.0	<5.0	-	-	5.0
Ba	150.0	161.0	153.0	138.0	204.0	209.0	133.0	161.0	151.0	270.0	201.0	119.0	129.0	101.0	113.0	160.0	44.6	238.0
B	2.0	2.0	<1.0	2.0	2.0	<1.0	<1.0	<1.0	<1.0	<1.0	2.0	<1.0	2.0	<2.0	<1.0	-	-	2.0
Be	0.6	0.4	0.6	0.6	0.6	0.6	0.5	1.6	0.6	0.5	0.9	0.8	0.8	0.9	1.0	0.7	0.3	1.2
Cd	0.7	0.4	0.6	0.6	0.4	1.0	0.7	0.5	0.1	1.2	0.4	0.6	0.4	0.5	0.5	0.6	0.3	1.0
Cr	16.0	10.0	15.0	14.0	15.0	18.0	13.0	13.0	17.0	13.0	31.0	15.0	13.0	13.0	15.0	15.0	4.7	24.0
Cu	20.0	14.0	20.0	19.0	20.0	21.0	19.0	20.0	21.0	20.0	38.1	19.0	16.0	18.0	18.0	20.0	5.3	30.0
Fe	16200.0	10400.0	15900.0	14500.0	17600.0	15800.0	15200.0	15400.0	16800.0	4100.0	23600.0	1600.0	14300.0	13300.0	15300.0	17420.0	7077.0	29882.0
Pb	15.0	14.0	16.0	15.0	17.0	15.0	16.0	18.0	19.0	23.0	20.0	34.0	25.0	33.0	27.0	20.0	6.6	32.0
Mg	10500.0	12200.0	12600.0	12500.0	9840.0	10200.0	10800.0	9130.0	9100.0	6780.0	13700.0	10700.0	10700.0	7820.0	7880.0	10297.0	1952.0	13734.0
Mn	1190.0	2930.0	922.0	958.0	1210.0	1130.0	751.0	991.0	707.0	1000.0	1070.0	600.0	781.0	411.0	1050.0	1047.0	567.0	2046.0
Ni	20.0	15.0	21.0	21.0	20.0	20.0	21.0	19.0	21.0	16.0	31.0	19.0	16.0	17.0	15.0	19.0	3.9	26.0
Se	<7.0	<6.0	<7.0	<7.0	<8.0	<7.0	<7.0	<7.0	<7.0	<10.0	<9.0	<7.0	<7.0	<6.0	<7.0	-	-	10.0
Ag	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	0.5	0.9	<0.30	<0.2	<0.2	<0.2	<0.2	-	-	0.2
V	5.9	5.4	7.2	6.8	7.6	7.0	8.5	8.0	10.0	9.4	18.1	12.0	9.7	13.0	7.4	9.1	3.3	15.0
Zn	82.0	57.3	72.5	69.4	69.4	69.1	62.8	64.7	69.8	69.3	129.0	71.8	60.3	66.1	62.2	72.0	17.0	101.0

all values are as ug/g dry weight

Table A21. Summary Statistics for Selected Surface Water Quality Parameters in Marsden Lake (North) and Snail Lake

Parameter	Marsden Lake (North)					Snail Lake				
	Dist.	DF	Central Tendency	Confidence Interval	Maximum	Dist.	DF	Central Tendency	Confidence Interval	Maximum
pH	N	8/8	6.74	6.38 – 7.10	6.92	N	10/10	8.66	8.56 – 8.76	8.94
DO (mg/L)	N	8/8	7.49	1.99 – 12.99	17.8	N	10/10	9.69	9.15 – 10.22	11.39
DO (%)	N	8/8	22.0	-5.20 – 49.18	38.1	N	10/10	101.3	95.2 – 107.3	120.5
Conductivity (umhos/cm)	N	8/8	425	307.73 – 541.83	653	N	10/10	299.9	298.5 – 301.3	304
Temperature (°C)	N	8/8	13.13	10.33 – 15.93	17.24	N	10/10	17.36	16.96 – 17.75	17.9
Hardness (mg/L)	N	8/8	187	105.25 – 268.53	340	N	10/10	101.2	99.2 – 103.2	104
TKN (mg/L)	L	8/8	2.73	1.83 – 5.68	7.3	N	10/10	0.80	0.75 – 0.84	0.92
TOC (mg/L)	*	8/8	14	9.1, 38.4	50	*	10/10	8.7	7.9, 11.6	12
Total Phosphorus (mg/L)	N	8/8	0.28	0.12 – 0.44	0.56	*	10/10	0.02	0.01, 0.02	<0.05
TSS (mg/L)	N	8/8	84	9.87 – 157.91	250	*	10/10	1.0	1.0, 1.0	<1.0

Table A22. Summary Statistics for Selected Chemicals in Surface Water for Marsden Lake (North) and Snail Lake

Detected Substance	Marsden Lake (North) (µg/L)					Snail Lake (µg/L)					Tox Screening Value
	Dist.	DF	Central Tendency	Confidence Interval	Maximum	Dist.	DF	Central Tendency	Confidence Interval	Maximum	
Al	N	8/8	205.7	4.95 – 406.4	540.25	*	10/10	6.3	4.3 – 6.9	6.95	125
Sb	L	8/8	0.086	0.062 –0.141	0.21	L	10/10	0.25	0.25– 0.26	0.27	31
As	N	8/8	2.06	0.96 – 3.15	4.5	N	10/10	5.6	5.5 – 5.7	5.90	190
Ba	N	8/8	98.8	46.18 – 151.32	200	*	10/10	41	34.5, 42.0	42.0	3.8
Cd	*	3/8	0.015 †	0.015, 0.111	0.13	*	0/10	Na	Na	Na	0.66
Cr	N	7/8	1.35	0.64 – 0.78	2.3	N	10/10	0.55	0.50 – 0.60	0.68	117
Co	N	8/8	0.58	0.17 – 1.00	1.2	L	10/10	0.055	0.054 – 0.057	0.059	5
Cu	N	8/8	1.34	0.57 – 9.83	4.1	L	10/10	2.2	2.1 – 2.2	2.3	6.4
Pb	L	8/8	2.90	1.10 – 52.27	7	*	10/10	0.086	0.060, 0.090	0.092	1.3
Mg	L *	8/8	11230	8287 – 15637	20000	*	10/10	8950	8800, 9665	9800	82000
Mn	N	8/8	649	227.9 – 1070.3	1300	N	10/10	20.5	16.3 – 24.7	31	80.3
Hg	N	8/8	0.0049	0.0022 – 0.0191	0.013	L	10/10	0.00057	0.00053 – 0.00063	0.00075	0.003
Ni	N	8/8	0.89	0.31 – 1.48	2	N	10/10	0.240	0.21 – 0.27	0.290	80
Ag	N	1/8	0.0187 †	0.012 – 0.026	0.0322	L	8/10	0.0148 †	0.0125 – 0.0186	0.0242	1
Tl	*	3/8	0.0064 †	0.0035, 0.0172	0.021	*	0/10	Na	Na	Na	18
Zn	N	1/8	5.15 †	0.68 – 9.62	15.1	N	10/10	0.67	0.61 – 0.73	0.81	59

Dist.: Distribution of the data (N refers to normal, L refers to lognormal, and the '*' refers to neither normal nor lognormal distribution).
DF: Detection frequency
Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.
Confidence Interval: The 95th confidence interval on the mean for normal data, the 95th confidence interval on the geometric mean for lognormal data, and the 5th and 95th percentiles if the distribution is unknown.
Na: Not applicable
†: The mean presented here was calculated using the detection limit when a sample was not detected.

Table A23. Statistical Comparison of Surface Water Data Between Marsden Lake (North) and Snail Lake

Parameter	Snail Lake		Marsden Lake (North)		Selected Statistical Test	Outcome
	No. of Non-detects	Data Distribution	No. of Non-detects	Data Distribution		
Al	0	Unknown	0	Normal	WRS	M.N. > S.L. (p<0.01)
Ba	0	Unknown	0	Normal	WRS	M.N. > S.L. (p<0.01)
Cd	10	Na	5	Unknown	Test of proportions	N.S.
Pb	0	Unknown	0	Lognormal	WRS	M.N. > S.L. (p<0.01)
Mn	0	Normal	0	Normal	t-test	M.N. > S.L. (p<0.01)
Hg	0	Normal (lognormal)	0	Normal	t-test on logs	M.N. > S.L. (p<0.01)
Zn	1	Normal	1	Normal	t-test	M.N. > S.L. (p<0.01)

WRS: Wilcoxon Rank Sum test
N.S.: non significant
M.N.: Marsden Lake North
S.L.: Snail Lake
Na: Not applicable

Table A24. Summary Statistics for Selected Chemicals in Sediment for Marsden Lake (North) and Snail Lake

Detected Substance	Marsden Lake (North) (mg/kg)					Snail Lake (mg/kg)					Tox Screening Value
	Dist.	DF	Central Tendency	Confidence Interval	Maximum	Dist.	DF	Central Tendency	Confidence Interval	Maximum	
Al	N	8/8	4338	2711 – 5964	6700	N	10/10	9570.0	8299 – 10841	13000	None
Sb	*	3/8	0.50†	0.49, 31.45	48	N	10/10	1.13	0.98 – 1.28	1.40	None
As	N	7/8	4.91†	2.29 – 7.54	8.90	N	10/10	52.4	40.0 – 64.8	74.0	6
Ba	N	8/8	58.6	31.6 – 85.6	100	N	10/10	146	125 – 167	210.0	None
Cd	*	2/8	0.50†	0.49, 1.09	1.4	N	10/10	1.44	0.96 – 1.91	2.90	0.5
Cr	N	8/8	7.74	4.88 – 10.60	14.0	N	10/10	21.3	16.6 – 25.9	31.0	1
Cu	N	8/8	14.1	2.09 – 26.29	43.0	N	10/10	107.00	77.7 –136	160.0	10
Pb	L	8/8	14.4	8.9 – 37.3	26.0	L	10/10	85.0	58.5 – 112	150.0	31
Mg	N	8/8	1830	1088.1 – 2571.9	3300.0	N	10/10	4200.0	3788 – 4612	5200.00	None
Mn	N	8/8	150.0	66.5 – 233.5	330.0	N	10/10	300.0	215 – 385	520.00	460
Hg	*	3/8	0.10†	0.07, 0.11	0.12	L	10/10	0.16	0.12 – 0.19	0.27	0.2
Ni	*	2/8	10.0†	10.0, 14.0	14	N	10/10	17.1	15 – 19	22	16
Se	*	2/8	1.00	0.98, 1.13	1.2	*	8/10	1.20†	0.99, 2.52	2.70	None
Zn	N	8/8	44	23.5 – 65.0	86	N	10/10	117	100 – 134	160	120

Dist.: Distribution of the data (N refers to normal, L refers to lognormal, and the '*' refers to neither normal nor lognormal distribution).

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

Confidence Interval: The 95th confidence interval on the mean for normal data, the 95th confidence interval on the geometric mean for lognormal data, and the 5th and 95th percentiles if the distribution is unknown.

Na: not applicable

†: The mean presented here was calculated using the detection limit when a sample was not detected.

Table A25. Comparison of Background Sediment Data to Marsden Lake North Sediment Data

Substance	Tier I Background			Tier II Snail Lake			1999 Marsden Lake North		
	mean	SD	[value]	mean	SD	[value]	Central Tendency	Confidence Interval	Maximum
Sb	-	-		1.13	0.21	2	0.50†	0.49, 31.45	48
As	-	-	5	52	17	84	4.91†	2.29 – 7.54	8.90
Cd	0.6	0.3	1.0	1.44	0.67	3	0.50	0.49, 1.09	1.4
Cu	20	5.3	30.0	107	41	182	14.1	2.09 – 26.29	43.0
Hg	-	-	-	0.16	0.05	0.25	0.10	0.07, 0.11	0.12
V	9.1	3.3	15	-	-	-	-	-	-
Zn	72	17	101	117	24	160	44	23.5 – 65.0	86

All values are expressed as ug/g

The "[value]" denotes the screening concentration value derived as [value] = x + ($t_{95\%}$ * SD) for Snail Lake data $t_{95\%}$=1.833, for Tier I background $t_{95\%}$=1.761

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions

Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown

"-" no data was available

Table A26. Statistical Comparison of Sediment Data Between Marsden Lake (North) and Snail Lake

Parameter	Snail Lake		Marsden Lake (North)		Selected Statistical Test	Outcome
	No. of Non-detects	Data Distribution	No. of Non-detects	Data Distribution		
Antimony	0	Normal (lognormal)	5	Unknown	Test of proportions	N.S.
Arsenic	0	Normal	1	Normal	t-test	N.S.
Cadmium	0	Normal	5	Unknown	Test of proportions	N.S.
Copper	0	Normal	0	Normal	t-test	N.S.
Mercury	0	Lognormal	6	Unknown	Test of proportions	N.S..
Zinc	0	Normal	0	Normal	t-test	N.S.

WRS: Wilcoxon Rank Sum test
N.S.: non significant
M.N.: Marsden Lake North
S.L.: Snail Lake

Table A27. Summary Statistics for Selected Surface Water Quality Parameters in Marsden Lake (South) and Snail Lake

Parameter	Marsden Lake (South)					Snail Lake				
	Dist.	DF	Central Tendency	Confidence Interval	Maximum	Dist.	DF	Central Tendency	Confidence Interval	Maximum
pH	N	12/12	7.22	7.03 – 7.42	7.64	N	10/10	8.66	8.56 – 8.76	8.94
DO (mg/L)	N	12/12	5.90	4.14 – 7.66	9.71	N	10/10	9.69	9.15 – 10.22	11.39
DO (%)	N	12/12	60.28	42.28 –78.27	96.7	N	10/10	101.3	95.2 – 107.3	120.5
Conductivity (umhos/cm)	N	12/12	296.25	294.11 – 298.39	302.0	N	10/10	299.9	298.5 – 301.3	304
Temperature (°C)	N	12/12	16.20	15.44 – 16.95	18.39	N	10/10	17.36	16.96 – 17.75	17.9
Hardness (mg/L)	N	12/12	107.50	102.28 – 112.72	122.0	N	10/10	101.2	99.2 – 103.2	104
TKN (mg/L)	*	12/12	0.57	0.54, 0.73	0.82	N	10/10	0.80	0.75 – 0.84	0.92
TOC (mg/L)	L	12/12	9.1	8.86 – 9.38	9.80	*	10/10	8.7	7.9, 11.6	12
Total Phosphorus (mg/L)	*	12/12	0.02	0.01, 0.04	<0.05	*	10/10	0.02	0.01, 0.02	<0.05
TSS (mg/L)	N	11/12	5.58	3.81 – 7.36	9.0	*	10/10	1.0	1.0, 1.0	<1.0

Table A28. Summary Statistics for Selected Chemicals in Surface Water for Marsden Lake (South) and Snail Lake

Detected Substance	Marsden Lake (South) (μg/L)					Snail Lake (μg/L)					Tox Screening Value
	Dist.	DF	Central Tendency	Confidence Interval	Maximum	Dist.	DF	Central Tendency	Confidence Interval	Maximum	
Al	*	12/12	4.85	2.9, 507.9	1100.25	*	10/10	6.1	4.3, 6.9	6.7	125
Sb	*	12/12	0.09	0.052, 0.098	0.10	N	10/10	0.25	0.24 – 0.26	0.27	31
As	*	12/12	0.79	0.67, 2.16	3.60	N	10/10	5.6	5.5 – 5.7	5.90	190
Ba	*	12/12	41.5	37.2, 64.8	89.0	*	10/10	41	34.5, 42.0	42.0	3.8
Cd	*	1/12	Na	Na	0.19	*	0/10	Na	Na	Na	0.66
Cr	*	2/12	0.55	0.45, 1.94	0.50	N	10/10	0.55	0.50 – 0.60	0.63	117
Co	*	12/12	0.070	0.062, 0.989	2.0	N	10/10	0.055	0.053 – 0.057	0.052	5
Cu	*	12/12	0.12	0.09, 4.14	9.0	N	10/10	2.2	2.1 – 2.2	2.3	6.4
Pb	*	12/12	0.08	0.060, 3.20	6.80	*	10/10	0.086	0.060, 0.090	0.092	1.3
Mg	*	12/12	6450	5830, 6700	6700	*	10/10	8950	8800, 9665	9800	82000
Mn	*	12/12	48.0	19.6, 197.3	370	N	10/10	20.5	16.3 – 24.7	31	80.3
Hg	*	12/12	0.00068	0.00042, 0.0085	0.017	N	10/10	0.00057	0.00051 – 0.00064	0.00075	0.003
Ni	*	12/12	0.20	0.13, 2.56	5.00	N	10/10	0.240	0.21 – 0.27	0.290	80
Ag	N	0/12	Na	Na	Na	N	8/10	0.0147 †	0.0112 – 0.0183	0.0180	1
Tl	*	1/12	0.0035	0.0035, 0.0199	0.04	*	0/10	Na	Na	Na	18
Zn	*	7/12	0.70	0.42, 19.26	23.1	N	10/10	0.67	0.61 – 0.73	0.70	59

Dist.: Distribution of the data (N refers to normal, L refers to lognormal, and the '*' refers to neither normal nor lognormal distribution).
DF: detection frequency
Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.
Confidence Interval: The 95th confidence interval on the mean for normal data, the 95th confidence interval on the geometric mean for lognormal data, and the 5th and 95th percentiles if the distribution is unknown.
Na: not applicable
†: The mean presented here was calculated using the detection limit when a sample was not detected.
‡: This substance is classified as a possible contaminant, due to possible outlier(s) in the data set. See the text for more discussion.

Table A29. Statistical Comparison of Water Data Between Marsden Lake (South) and Snail Lake

Parameter	Snail Lake		Marsden Lake (South)		Selected Statistical Test	Outcome
	No. of Non-detects	Data Distribution	No. of Non-detects	Data Distribution		
Al	0	Unknown	0	Unknown	WRS	N.S.
Ba	0	Unknown	0	Unknown	WRS	N.S.
Cd	10	Na	9	Na	Test of proportions	N.S.
Cu	0	Normal	0	Unknown	WRS	N.S.
Pb	0	Unknown	0	Unknown	WRS	N.S.
Mn	0	Normal	0	Unknown	WRS	M.S. > S.L., $p<0.01$
Hg	0	Normal	0	Unknown	WRS	N.S.
Zn	1	Normal	5	Unknown	Test of proportions	N.S.

WRS: Wilcoxon Rank Sum test
N.S.: non significant
M.S.: Marsden Lake South
S.L.: Snail Lake
Na: Not applicable

Table A30. Summary Statistics for Selected Chemicals in Sediment for Marsden Lake (South) and Snail Lake

Parameter	Marsden Lake (South) (mg/kg)					Snail Lake (mg/kg)					Tox Screening Value
	Dist.	DF	Central Tendency	Confidence Interval	Maximum	Dist.	DF	Central Tendency	Confidence Interval	Maximum	
Al	N	12/12	10166.67	7960.6 – 12372.7	15000	N	10/10	9570.0	8299 – 10841	13000	None
Sb	*	9/12	0.60†	0.50, 0.99	1.0	N	10/10	1.13	0.98 – 1.28	1.40	None
As	N	12/12	9.17	7.98 – 10.35	12	N	10/10	52.4	40.0 – 64.8	74.0	6
Ba	L	12/12	134	125.0 – 144.7	190	N	10/10	146	125 – 167	210.0	None
Cd	*	6/12	0.50†	0.50, 1.49	1.6	N	10/10	1.44	0.96 – 1.91	2.90	0.6
Cr	N	12/12	19.32	15.14 – 23.49	30.0	N	10/10	21.3	16.6 – 25.9	31.0	26
Co	*	4/12	9.95†	7.5, 11.0	11.0	*	10/10	10.0	9.9, 12.8	15	None
Cu	N	12/12	24.48	17.47 – 31.48	52.0	N	10/10	107.00	77.7 –136	160.0	16
Pb	N	12/12	43.67	35.92 – 51.41	63.0	N	10/10	85.0	58.5 – 112	150.0	31
Mg	N	12/12	3116.67	2579.0 – 3654.3	4300	N	10/10	4200.0	3788 – 4612	5200.00	None
Mn	N	12/12	355.0	261.4 – 448.6	680	N	10/10	300.0	215 – 385	520.00	460
Hg	N	12/12	0.15	0.14 – 0.17	0.18	L	10/10	0.16	0.12 – 0.19	0.27	0.2
Ni	N	9/12	16.7†	13.7 – 19.8	23	N	10/10	17.1	15 – 19	22	16
Se	*	6/12	1.00	0.99, 1.29	1.4	*	8/10	1.20†	0.99, 2.52	2.70	None
Zn	N	12/12	79.1	63.8 – 94.4	140	N	10/10	117	100 – 134	160	120

Dist.: Distribution of the data (N refers to normal, L refers to lognormal, and the '*' refers to neither normal nor lognormal distribution).

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

Confidence Interval: The 95th confidence interval on the mean for normal data, the 95th confidence interval on the geometric mean for lognormal data, and the 5th and 95th percentiles if the distribution is unknown.

Na: not applicable

†: The mean presented here was calculated using the detection limit when a sample was not detected.

Table A31. Comparison of Background Sediment Data to Marsden Lake South Sediment Data

Substance	Tier I Background			Tier II Snail Lake			1999 Marsden Lake South		
	mean	SD	[value]	mean	SD	[value]	Central Tendency	Confidence Interval	Maximum
Al	7121	2018	10674	9570	1776	12826	10166.67	7960.6 – 12372.7	15000
Sb	-	-		1.13	0.21	2	0.60†	0.50, 0.99	1.0
As	-	-	5	52	17	84	9.17	7.98 – 10.35	12
Cd	0.6	0.3	1.0	1.44	0.67	3	0.50	0.50, 1.49	1.6
Cr	15.0	4.7	24.0	21	6	33	19.32	15.14 – 23.49	30.0
Co	-	-	-	10.5	1.6	13	9.95	7.5, 11.0	11.0
Cu	20	5.3	30.0	107	41	182	24.48	17.47 – 31.48	52.0
Pb	20	6.6	32.0	85	37	153	43.67	35.92 – 51.41	63.0
Hg	-	-	-	0.16	0.05	0.25	0.15	0.14 – 0.17	0.18
V	9.1	3.3	15	-	-	-	-	-	-
Zn	72	17	101	117	24	160	79.1	63.8 – 94.4	140

All values are expressed as ug/g

The "[value]" denotes the screening concentration value derived as [value] = $x + (t_{95\%} * SD)$ for Snail Lake data $t_{95\%}=1.833$, for Tier I background $t_{95\%}=1.761$

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions

Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown

"-" no data was available

Table A32. Statistical Comparison of Sediment Data Between Marsden Lake (South) and Snail Lake

Parameter	Snail Lake		Marsden Lake South		Selected Statistical Test	Outcome
	No. of Non-detects	Data Distribution	No. of Non-detects	Data Distribution		
Aluminum	0	Normal	0	Normal	t-test	N.S.
Antimony	0	Normal (lognormal)	3	Unknown	WRS	N.S.
Arsenic	0	Normal	0	Normal	t-test	N.S.
Cadmium	0	Normal	6	Unknown	Test of proportions	N.S.
Chromium	0	Normal	0	Normal	t-test	N.S.
Cobalt	0	Unknown	6	Unknown	Test of proportions	N.S.
Copper	0	Normal	0	Normal	t-test	N.S.
Lead	0	Normal (lognormal)	0	Normal (lognormal)	t-test	N.S.
Mercury	0	Lognormal	0	Normal (lognormal)	t-test on logs	N.S.
Zinc	0	Normal	0	Normal	t-test	N.S.

WRS: Wilcoxon Rank Sum test
N.S.: non significant
M.N.: Marsden Lake North
S.L.: Snail Lake

Table A33. Summary Statistics for Selected Chemicals in Pond G

Parameter	Surface Water (μg/L) †				Sediment (μg/g)			
	DF	Mean	Minimum	Maximum	DF	Mean	Minimum	Maximum
Al	3/3	763	240	1600	3/3	14000	11000	18000
Sb	3/3	0.70	0.49	1.10	1/3	0.54	<0.48	0.63
As	3/3	4.2	3.7	5.1	3/3	5.13	3.5	7.6
Ba	3/3	69	51	100	3/3	153.3	130	190
Be	1/3	0.07	<0.048	0.12	0/3	Na	<0.93	<1.0
Cd	3/3	0.069	0.027	0.14	3/3	0.76	0.73	0.81
Cr	3/3	1.81	0.82	3.4	3/3	24	17	36
Co	3/3	1.49	0.86	2.4	0/3	Na	<9.3	<10
Cu	3/3	3.0	1.5	5.6	3/3	25.3	18	39
Pb	3/3	4.5	2.4	7.9	3/3	30.3	24	39
Mg	3/3	4400	4300	4600	3/3	2866.7	2200	3600
Mn	3/3	347	320	380	3/3	333.3	270	390
Hg	0/3	Na	0.0027	0.017	2/3	0.10	0.044	0.082
Ni	3/3	3.9	2.7	5.8	3/3	16	12	22
Se	1/3	40	Na	0.41	0/3	Na	<0.95	<0.99
Ag	3/3	0.0133	0.0089	0.019	0/3	Na	<0.48	<0.50
Tl	3/3	0.0184	0.0061	0.037	0/3	Na	<0.95	<0.99
Zn	3/3	5.79	0.58	14	3/3	85.7	62	110
BEHP	2/3	8.8	8.3	9.3	0/3	Na	Na	Na
4,4'-DDT	0/3	Na	<0.05	<0.05	1/3	NA	<0.03	0.03
4,4'-DDD	0/3	Na	<0.05	<0.05	1/3	Na	<0.05	0.16
4,4'-DDE	0/3	Na	<0.05	<0.05	1/3	Na	<0.05	0.053
Dieldrin	0/3	Na	<0.05	<0.05	1/3	Na	<0.05	0.071
Aroclor-1016	0/3	Na	<0.1	<0.1	1/3	Na	<0.2	<0.33
Aroclor-1221	0/3	Na	<0.1	<0.1	1/3	Na	<0.1	<0.33
Aroclor-1232	0/3	Na	<0.1	<0.1	1/3	Na	<0.1	<0.33
Aroclor-1242	0/3	Na	<0.1	<0.1	1/3	Na	<0.05	<0.33
Aroclor-1248	0/3	Na	<0.1	<0.1	1/3	Na	<0.1	0.39

†: Surface water data are unfiltered water concentrations.
DF: detection frequency
Na: not applicable
BEHP: bis(2-Ethylhexyl)phthalate

Table A34. Summary Statistics for Selected Metals in Downstream and Upstream Rice Creek Samples During 1999-2000 Water Monitoring

Parameter	Rice Creek (Down Stream)[♦] µg/L						Rice Creek (Up Stream)[♦] µg/L					
	Dist	DF[Φ]	Central Tendency	Std. Dev.	Confidence Interval	Max	Dist	DF	Central Tendency	Std. Dev.	Confidence Interval	Max
Aluminum	N	10/10	127.2	86.52	65.34 – 189.1	240	N	10/10	118	73.8	65.2 – 171	200.5
Barium	N	10/10	78.79	9.87	71.73 – 85.85	96	N	10/10	77.40	8.88	71.1 - 83.8	92
Cadmium	N	8/10	0.0274	0.0102	0.0201 – 0.0347	0.047	N	8/10	0.0272	0.0079	0.0215 – 0.0328	0.035
Copper	N	10/10	1.32	0.321	1.09 – 1.55	1.70	N	10/10	1.202	0.252	1.02 – 1.38	1.60
Mercury	*	9/10	0.00379	NA	0.000679, 0.000756	0.00528	*	9/10	0.00432	NA	0.000722, 0.0253	0.042
Silver	*	0/10	0.00819	NA	0.00339, 0.0180	0.0186	*	5/10	0.0143	NA	0.00328, 0.130	0.212

[Φ] All quarterly data was combined, resulting in a total of 10 samples,because there were only two samples taken from each location per quarter. Two samples is not sufficient to calculate descriptive statistics.

[♦] Downstream=Rice Creek Downstream from building 103 drainage, Up= Rice Creek Upstream from building 103 drainage (reference site)

Dist : Distribution of the data (N refers to Normal, L refers to Lognormal, and "*" refers to unknown)

DF: Detection Frequency

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions

Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown

Max: Maximum detected value during sampling

NA: Not applicable

Table A35. Summary Statistics for Water Quality Parameters in Downstream and Upstream Rice Creek Samples During Quarterly Water Monitoring

Parameter	Rice Creek Downstream						Rice Creek Upstream					
	Dist.	DF[Φ]	Central Tendency	Std. Dev.	Confidence Interval	Max	Dist.	DF	Central Tendency	Std. Dev.	Confidence Interval	Max
pH	N	10/10	8.081	0.582	7.665 – 8.497	8.8	N	10/10	8.048	0.562	7.646 – 8.450	8.8
DO mg/L	N	10/10	9.430	1.918	8.058 – 10.80	12.1	N	10/10	8.349	1.621	7.190 – 9.508	10.98
%DO	N	10/10	93.90	17.97	81.04 – 106.8	122.2	N	10/10	84.53	23.16	67.96 – 101.1	120.2
Conductivity (µmhos/L)	*	10/10	391.0	NA	378.0, 673.0	673	*	10/10	384.0	NA	371.0, 667.0	667
Temperature (°C)	*	10/10	20.10	NA	0.0500, 20.99	20.99	*	10/10	19.62	NA	0.0700, 20.89	20.89
Hardness (mg/L)	*	10/10	153.0	NA	148.0, 272.0	272	*	10/10	161.0	NA	148.0, 272.0	272
TKN (mg/L)	N	10/10	2.650	0.684	2.161 – 3.139	3.8	N	10/10	2.590	0.750	2.053 – 3.127	3.9
TOC (mg/L)	L	10/10	17.81			21	*	10/10	17.00	NA	16.00, 21.00	21
Total Phosphorus (mg/L)	N	10/10	0.1760	0.06867	0.1269 – 0.2251	0.26	N	10/10	0.1740	0.06802	0.1253 – 0.2227	0.27
TSS (mg/L)	N	10/10	52.00	29.54	30.86 – 73.14	94	*	10/10	59.00	NA	2.00, 83.00	83

[Φ] All quarterly data was combined, resulting in a total of 10 samples,because there were only two samples taken from each location per quarter. Two samples is not sufficient to calculate descriptive statistics.
DF: Detection Frequency
Dist.: Distribution of the data (N refers to Normal, L refers to Lognormal and * refers to unknown)
Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions
Std. Dev: Standard deviation of the mean for normal data and of the geometric mean for lognormal data
Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown
NA: Not applicable

Table A36. Statistical Comparison of Surface Water Metal Data Between Upstream and Downstream at Rice Creek

Parameter	Upstream (reference area)		Downstream		Selected Statistical Test	Outcome
	No. of Non-detects	Data Distribution	No. of Non-detects	Data Distribution		
Al	0/10	Normal	0/10	Normal	2-factor repeated measures ANOVA	N.S.
Ba	0/10	Normal	0/10	Normal	2-factor repeated measures ANOVA	N. S.
Cd	2/10	Normal	2/10	Normal	2-factor repeated measures ANOVA & Fisher's Exact Test	N.S.
Hg	1/10	Unknown	1/10	Unknown	2-factor repeated measures ANOVA	N. S.

Na: Not applicable
ANOVA: analysis of variance
N.S.: not significant

Table A37. Statistical Comparison of Surface Water Quality Data Between Upstream and Downstream at Rice Creek

Parameter	Upstream (reference site)		Downstream		Selected Statistical Test	Outcome
	No. non-detects	Data distribution	No. non-detects	Data distribution		
pH	0/10	Normal	0/10	Normal	2-factor repeated measures ANOVA	N. S.
D.O.	0/10	Normal	0/10	Normal	2-factor repeated measures ANOVA	Qtr 1: Down> Up (p<0.05)
% D.O.	0/10	Normal	0/10	Normal	2-factor repeated measures ANOVA	N. S.
Conductivity	0/10	Unknown	0/10	Unknown	2-factor repeated measures ANOVA	N. S.
Temperature	0/10	Unknown	0/10	Unknown	2-factor repeated measures ANOVA	N. S.
Hardness	0/10	Unknown	0/10	Unknown	2-factor repeated measures ANOVA	N. S.
TKN	0/10	Normal	0/10	Normal	2-factor repeated measures ANOVA	N. S.
TOC	0/10	Unknown	0/10	Log normal	2-factor repeated measures ANOVA	N. S.
Total Phosphorus	0/10	Normal	0/10	Normal	2-factor repeated measures ANOVA	N. S.
TSS	0/10	Unknown	0/10	Normal	2-factor repeated measures ANOVA	N. S.

NA: Not available
N.S: Not significant

Table A38. Summary Statistics of Combined Quarterly Sunfish and Snail Lakes Water Monitoring Data

Parameter	Sunfish Lake µg/L						Snail Lake µg/L						
	Dist	DF	Central Tendency	Std. Dev.	Confidence Interval	Max	Dist	DF	Central Tendency	Std. Dev.	Confidence Interval	Max	Tox Screening Value
Aluminum	L	12/12	29.78	31.66	19.29 – 61.05	35.49	L	12/12	6.21	1.57	5.50 – 7.18	8.2	125
Barium	N	12/12	33.84	12.72	27.89 – 39.79	53.01	N	12/12	46.84	7.10	42.33 – 51.35	58.01	3.8
Cadmium	N	8/12	0.015	0.0034	0.014 – 0.017	0.02	N	8/12	0.013	0.0064	0.0089 – 0.017	0.027	0.66
Copper	N	12/12	1.30	0.43	1.094 – 1.501	1.8	U	12/12	2.25	NA	2.1, 6.82	11.0	6.4
Mercury	U	12/12	0.0012	NA	0.00066, 0.00412	0.0067	N	12/12	0.00085	0.00028	0.00068 – 0.00103	0.0014	0.003
Silver	U	0/12	NA	NA	NA	NA	U	0/12	NA	NA	NA	NA	1.0
Zinc	U	12/12	1.35	0.936	0.916 – 1.79	2.8	N	12/12	1.019	0.329	0.810 – 1.23	1.45	59

Dist.: Distribution of the data (N refers to normal, L refers to lognormal, and the '*' refers to neither normal nor lognormal distribution).

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.

Confidence Interval: The 95th confidence interval on the mean for normal data, the 95th confidence interval on the geometric mean for lognormal data, and the 5th and 95th percentiles if the distribution is unknown.

Na: not applicable

Table A39. Summary Statistics for Sunfish Lake and Snail Lake Water Monitoring Data Evaluated Under Criterion 5 of COC Selection

Chemical Detected	Qtr	Sunfish Lake					Snail Lake					Tox Screening Value
		Dist$^{\delta}$	DF	Central Tendency	Confidence Interval	Maximum	Dist$^{\delta}$	DF	Central Tendency	Confidence Interval	Maximum	
Cd	1	N	0/3	NA	NA	NA	N	0/3	NA	NA	NA	0.66
	2		2/3	0.0177	0.0153 - 0.0201	0.0200		2/3	0.0203	0.0145 – 0.0262	0.0270	
	3		3/3	0.0173	0.0167 – 0.0178	0.01795		3/3	0.0116	0.0111 – 0.0122	0.0120	
	4		3/3	0.0105	0.00764 – 0.0134	0.0140		3/3	0.0050	0.00319 – 0.00688	0.0068	
Ba	1	N	3/3	31.7	30.2 – 33.2	33.02	N	3/3	41.3	40.8 – 41.9	42.0	3.8
	2		3/3	49.3	46.3 – 52.4	53.01		3/3	56.0	54.4 – 57.7	58.01	
	3		3/3	38.3	37.8 – 38.9	39.0		3/3	50.3	48.3 – 52.3	52.0	
	4		3/3	16.0	15.0 – 17.0	17.0		3/3	39.7	38.2 – 41.1	41.0	
Hg	1	U	0/3	NA	NA	NA	L	3/3	0.000619	0.000455 – 0.00104	0.000750	0.003
	2		2/3	0.00120	0.00078 – 0.00120	0.00120		3/3	0.000926	0.000590 – 0.00364	0.00130	
	3		3/3	0.00118	0.00108 – 0.00154	0.00158		3/3	0.000876	0.000688 – 0.00132	0.000970	
	4		3/3	0.0016	0.00151 – 0.00196	0.0020		3/3	0.00104	0.000673 – 0.00274	0.00140	
Zn	1	N	3/3	0.957	0 – 2.49	2.80	N	3/3	0.693	0.591 – 0.796	0.81	59
	2		3/3	2.04	1.54 – 2.54	2.34		3/3	1.34	1.24 – 1.43	1.44	
	3		3/3	1.79	1.67 – 1.90	1.92		3/3	1.11	1.10 – 1.12	1.12	
	4		3/3	0.639	0.594 – 0.683	0.692		3/3	0.932	0.462 – 1.40	1.45	

Values are reported to three significant figures in most cases, unless they were reported otherwise by the analyzing laboratory.
δ: Due to small sample size distribution could not be determined per quarter
L=lognormal, N=normal, and U=unknown distirbutions
Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions
Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown
NA: Not applicable

Table A40. Summary Statistics for Water Quality Parameters in Sunfish and Snail Lakes

Parameter	Sunfish Lake						Snail Lake					
	Dist.	DF	Central Tendency	Std. Dev.	Confidence Interval	Max	Dist.	DF	Central Tendency	Std. Dev.	Confidence Interval	Max
pH	*	12/12	7.7900	NA	7.176, 10.25	10.28	*	12/12	8.49	NA	7.74, 8.65	8.65
DO mg/L	N	12/12	7.8808	2.980	5.987 – 9.775	11.5	N	12/12	10.35	1.47	9.42 – 11.29	12.86
%DO	N	12/12	81.48	40.06	56.03 – 106.9	135.4	N	11/11	98.78	10.82	91.91 – 105.6	113.6
Conductivity (μmhos/L)	N	12/12	268.58	31.82	248.4 – 288.8	314	N	11/11	348.36	39.40	323.3 – 373.4	399
Temperature (°C)	*	12/12	16.03	NA	2.448, 23.63	23.9	N	10/10	12.68	8.22	7.458 – 17.90	21.4
Hardness (mg/L)	N	12/12	67.50	13.46	58.95 – 76.05	90	N	12/12	122.50	19.26	110.3 – 134.7	146
TKN (mg/L)	*	6/12	0.51	NA	0.5, 1.515	1.9	NA	9/12	0.685	NA	0.5, 0.956	1.0
TOC (mg/L)	L	12/12	6.773	0.9518	6.169 – 7.378	7.8	L	12/12	8.978	0.8308	8.450 – 9.505	11.0
Total Phosphorus (mg/L)	*	12/12	0.0500	NA	0.0255, 0.136	0.18	*	12/12	0.0200	NA	0.020, 0.040	0.04
TSS (mg/L)	NA	8/12	2	NA	1 – 8.05	13	NA	7/12	1	NA	1 – 2.45	3

DF: Detection Frequency

Dist.: Distribution of the data (N refers to Normal, L refers to Lognormal and * refers to unknown)

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions

Std. Dev: Standard deviation of the mean for normal data and of the geometric mean for lognormal data

Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown

NA: Not applicable

Table A41. Criterion 5:_Statistical Comparison of Surface Water Metal Data Between Sunfish Lake and Snail Lake

Parameter	Snail Lake (SL)		Sunfish Lake (SF)		Selected Statistical Test	Outcome
	No. of Non-detects	Data Distribution	No. of Non-detects	Data Distribution		
Ba	0/12	Normal (lognormal)	0/12	Normal (lognormal)	2-factor repeated measures ANOVA	N. S.
*Cd	1/9	Normal	1/9	Normal	2-factor repeated measures ANOVA	Qtrs 3&4 SF >SL ($p< 0.01$)
Hg	0/12	Normal (lognormal)	4/12	Unknown	2-factor repeated measures ANOVA	Sunfish > Snail
Zn	0/12	Normal	0/12	Normal	2-factor repeated measures ANOVA	Qtr 3 SF > SL ($p<0.05$)

*Cadmium was only assessed for quarters 2 through 4 due to all non detected values during the first quarter sampling at both lakes

Na: Not applicable

ANOVA: analysis of variance

N.S.: not significant

Table A42. Statistical Comparison of Surface Water Quality Data Between Sunfish Lake and Snail Lake

Parameter	Snail Lake (SL)		Sunfish Lake (SF)		Selected Statistical Test	Outcome
	No. non-detects	Data distribution	No. non-detects	Data distribution		
pH	0/12	Unknown	0/12	Unknown	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	4th qtr SF > SL ($p<0.05$)
D.O.	0/12	Normal	0/12	Normal	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	4th qtr SF.> SL ($p<0.05$)
% D.O.	0/11	Normal	0/12	Normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	4th qtr SF > SL ($p<0.01$)
Conductivity	0/11	Normal	0/12	Normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	N. S.
Temperature	0/10	Normal	0/12	Unknown	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	3rd, 4th qtr SF >SL ($p<0.05$)
Hardness	0/12	Normal	0/12	Normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	N. S.
TKN	3/12	NA	6/12	NA	Fisher's Exact Test	N. S.
TOC	0/12	Log-normal	0/12	Log normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	N. S.
Total Phosphorus	0/12	Unknown	0/12	Unknown	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	1st & 4th qtrs SF > SL ($p<0.05$)
TSS	5/12	NA	4/12	NA	Fisher's Exact Test	N. S.

NA: Not available N.S: Not significant

Table A43. Summary Statistics for Selected Chemicals in Sunfish Lake Sediment (from the Tier I data) and Snail Lake (Tier II data)

Detected Substance	Sunfish Lake (mg/kg)					Snail Lake (mg/kg)					Tox Screening Value (µg/g)
	Dist	DF	Central Tendency	Confidence Interval	Maximum	Dist	DF	Central Tendency	Confidence Interval	Maximum	
Al	N	12/12	12612	10682-14542	18000	N	10/10	9570.0	8299 – 10841	13000	None
Cr	N	12/12	137	72 – 203	264	N	10/10	21.3	16.6 – 25.9	31.0	1
Cu	N	12/12	72.1	50.0 – 94.3	126	N	10/10	107.00	77.7 –136	160.0	16
Pb	N	12/12	124.0	93.6 – 154.3	202	N	10/10	85.0	58.5 – 112	150.0	31
V	N	12/12	46.7	39.7 – 53.7	61.5	Na	Na	Na	Na	Na	Na
Zn	N	12/12	302.0	229.5 – 374.6	501	N	10/10	117	100 - 134	160	59

Na: Not available

Table A44. Comparison of Background Sediment Data to Sunfish Lake Sediment

Substance	Tier I Background			Tier II Snail Lake			Tier I Sunfish Lake		
	mean	SD	[value]	mean	SD	[value]	Central Tendency	Confidence Interval	Maximum
Al	7121	2018	10674	9570	1776	12826	12612	10682 - 14542	18000
Cr	15.0	4.7	24.0	21	6	33	137	72 - 203	264
Cu	20	5.3	30.0	107	41	182	72.1	50.0 – 94.3	126
Pb	20	6.6	32.0	85	37	153	124	93.6 – 154.3	202
V	9.1	3.3	15	-	-	-	46.7	39.7 – 53.7	61.5
Zn	72	17	101	117	24	160	302	229.5 – 374.6	501

All values are expressed as ug/g

The "[value]" denotes the screening concentration value derived as [value] = $x + (t_{95\%} * SD)$ for Snail Lake data $t_{95\%}=1.833$, for Tier I background $t_{95\%}=1.761$

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions

Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown

"-" no data was available

Table A45. Summary Statistics of Combined Quarterly Round and Snail Lakes Water Monitoring Data

Parameter	Round Lake µg/L						Snail Lake µg/L					
	Dist	DF	Central Tendency	Std. Dev.	Confidence Interval	Max	Dist	DF	Central Tendency	Std. Dev.	Confidence Interval	Max
Barium	U	20/20	76.45	NA	61.67, 89.11	91.01	N	12/12	46.84	7.10	42.33 – 51.35	58.01
Cadmium	L	14/20	0.0186	0.0076	0.016 – 0.022	0.054	N	8/12	0.013	0.0064	0.0089 – 0.017	0.027
Copper	N	20/20	0.991	0.314	0.844 – 1.14	1.60	U	12/12	2.25	NA	2.1, 6.82	11.0
Mercury	N	18/20	0.001	0.0012	0.001 - 0.002	0.0063	N	12/12	0.00085	0.00028	0.00068 – 0.00103	0.0014
Silver	U	0/20	NA	NA	NA	NA	U	0/12	NA	NA	NA	NA
Zinc	N	20/20	1.38	1.20	0.812 – 1.93	4.037	N	12/12	1.019	0.329	0.810 – 1.23	1.45

Dist.: Distribution of the data (N refers to normal, L refers to lognormal, and the '*' refers to neither normal nor lognormal distribution).
Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median if the distribution is unknown.
Confidence Interval: The 95th confidence interval on the mean for normal data, the 95th confidence interval on the geometric mean for lognormal data, and the 5th and 95th percentiles if the distribution is unknown.
Na: not applicable

Table A46 Summary Statistics for Round Lake and Snail Lake Water Monitoring Data Evaluated Under Criterion 5 of COC Selection

Chemical Detected	Qtr	Round Lake					Snail Lake					Tox Screening Value
		Dist$^{\delta}$	DF	Central Tendency	Confidence Interval	Maximum	Dist$^{\delta}$	DF	Central Tendency	Confidence Interval	Maximum	
Cd	1	L	0/5	NA	NA	NA	N	0/3	NA	NA	NA	0.66
	2		4/5	0.0275	0.0187 – 0.0561	0.0540		2/3	0.0203	0.0145 – 0.0262	0.0270	
	3		5/5	0.0214	0.0189 – 0.0246	0.0295		3/3	0.0116	0.0111 – 0.0122	0.0120	
	4		5/5	0.0114	0.0100 – 0.0132	0.0130		3/3	0.0050	0.00319 – 0.00688	0.0068	
Ba	1	U	5/5	62.02	56.4 – 62.0	62.02	N	3/3	41.3	40.8 – 41.9	42.0	3.8
	2		5/5	88.01	85.6 – 90.6	91.01		3/3	56.0	54.4 – 57.7	58.01	
	3		5/5	86.40	82.5 – 87.0	87.0		3/3	50.3	48.3 – 52.3	52.0	
	4		5/5	70.00	69.2 – 70.8	71.0		3/3	39.7	38.2 – 41.1	41.0	
Hg	1	L	5/5	0.000827	0.000758 – 0.00091	0.00099	L	3/3	0.000619	0.000455 – 0.00104	0.000750	0.003
	2		5/5	0.000663	0.000569 – 0.000757	0.00085		3/3	0.000926	0.000590 – 0.00364	0.00130	
	3		5/5	0.00197	0.000471 – 0.00347	0.00628		3/3	0.000876	0.000688 – 0.00132	0.000970	
	4		5/5	0.0009	0.00810 – 0.00099	0.00110		3/3	0.00104	0.000673 – 0.00274	0.00140	
Zn	1	N	0/5	NA	NA	NA	N	3/3	0.693	0.591 – 0.796	0.81	59
	2		5/5	3.04	2.65 – 3.43	4.04		3/3	1.34	1.24 – 1.43	1.44	
	3		5/5	1.70	1.60 – 1.80	1.82		3/3	1.11	1.10 – 1.12	1.12	
	4		5/5	0.720	0.598 – 0.842	0.992		3/3	0.932	0.462 – 1.40	1.45	

Values are reported to three significant figures in most cases, unless they were reported otherwise by the analyzing laboratory.
δ: Due to small sample size distribution could not be determined per quarter
L=lognormal, N=normal, and U=unknown distirbutions
Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions
Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown
NA: Not applicable

Table A47. Summary Statistics for Water Quality Parameters in Round and Snail Lakes

Parameter	Round Lake						Snail Lake					
	Dist.	DF	Central Tendency	Std. Dev.	Confidence Interval	Max	Dist.	DF	Central Tendency	Std. Dev.	Confidence Interval	Max
pH	N	20/20	8.288	0.3741	8.113 – 8.464	8.9	*	12/12	8.49	NA	7.74, 8.65	8.65
DO (mg/L)	*	20/20	9.815	NA	8.299, 10.60	12.4	N	12/12	10.35	1.47	9.420 – 11.29	12.86
%DO	N	20/20	93.35	17.58	85.12 – 101.6	117.8	N	11/11	98.78	10.82	91.91 – 105.7	113.6
Conductivity (µmhos/L)	N	20/20	556.50	70.09	523.7 – 589.3	669	N	11/11	348.36	39.40	323.3 – 373.4	399
Temperature (°C)	*	19/19	16.44	NA	1.552, 22.164	22.2	N	10/10	12.68	8.22	7.458 – 17.90	21.4
Hardness (mg/L)	N	20/20	155.9	24.57	144.4, 167.4	194	N	12/12	122.50	19.26	110.3 – 134.7	146
TKN (mg/L)	*	19/20	1.2	NA	0.614, 1.585	1.5	NA	9/12	0.685	NA	0.5, 0.956	1.0
TOC (mg/L)	L	20/20	12.51	3.933	10.67 – 14.35	21	L	12/12	8.978	0.8308	8.450 – 9.505	11.0
Total Phosphorus (mg/L)	*	20/20	0.0400	NA	0.03, 0.0825	0.13	*	12/12	0.0200	NA	0.020, 0.040	0.04
TSS (mg/L)	NA	19/19	8	NA	3.9, 26.4	30	NA	7/12	1	NA	1 – 2.45	3

DF: Detection Frequency
Dist.: Distribution of the data (N refers to Normal, L refers to Lognormal and * refers to unknown)
Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions
Std. Dev: Standard deviation of the mean for normal data and of the geometric mean for lognormal data
Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown
NA: Not applicable

Table A48. Statistical Comparison of Surface Water Metal Data Between Round Lake and Snail Lake

Parameter	Snail Lake (SL)		Round Lake (RL)		Selected Statistical Test	Outcome
	No. of Non-detects	Data Distribution	No. of Non-detects	Data Distribution		
Ba	0/12	Normal (lognormal)	0/20	Unknown	2-factor repeated measures ANOVA	All qtrs RL > SL (p<0.01)
Cd	4/12	Lognormal	6/20	Lognormal	2-factor repeated measures ANOVA	Qtrs 2-4 RL.>SL (p< 0.01)
Hg	0/12	Normal (lognormal)	2 /20	Normal (lognormal)	2-factor repeated measures ANOVA	N. S.
Zn	0/12	Normal	0/20	Normal	2-factor repeated measures ANOVA	Qtrs 2 & 3 RL > SL (p< 0.01)

Na: Not applicable
ANOVA: analysis of variance
N.S.: not significant

Table A49. Statistical Comparison of Surface Water Quality Data Between Round Lake and Snail Lake

Parameter	Snail Lake (SL)		Round Lake (RL)		Selected Statistical Test	Outcome
	No. non-detects	Data distribution	No. non-detects	Data distribution		
pH	0/12	Unknown	0/20	Normal	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	N. S.
D.O.	0/12	Normal	0/20	Unknown	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	4th qtr RL > SL ($p<0.05$)
% D.O.	0/11	Normal	0/20	Normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	4th qtr RL > SL ($p<0.01$)
Conductivity	0/11	Normal	0/20	Normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	All qtrs RL > SL ($p<0.01$)
Temperature	0/10	Normal	0/19	Unknown	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	3rd, 4th qtr RL >SL ($p<0.05$)
Hardness	0/12	Normal	0/20	Normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	All qtrs RL > SL ($p<0.01$)
TKN	3/12	NA	1/20	NA	Fisher's Exact Test	N.S.
TOC	0/12	Log-normal	0/20	Log normal	2-factor repeated measures ANOVA (on ranks of data) & Dunnett's test (to compare lakes)	1st, 4th qtr RL > SL ($p<0.01$) 2nd qtr RL > SL ($p<0.05$)
Total Phosphorus	0/12	Unknown	0/20	Unknown	2-factor repeated measures ANOVA (on ranks of data) & Mann-Whitney (to compare lakes)	1st, 2nd, 4th qtrs RL > SL ($p<0.05$)
TSS	5/12	NA	0/19	NA	Fisher's Exact Test	All qtrs RL > SL ($p<0.01$)

NA: Not available N.S: Not significant

Table A50. Summary Statistics for Selected Chemicals in Round Lake Sediment (from Tier I data) and Snail Lake (Tier II data)

Detected Substance	Round Lake (µg/g)					Snail Lake (mg/kg)					Tox Screening Value (µg/g)
	Dist	DF	Central Tendency	Confidence Interval	Maximum	Dist	DF	Central Tendency	Confidence Interval	Maximum	
Al	N	18/18	8526	6489 – 10562	15400	N	10/10	9570.0	8299 – 10841	13000	None
Cd	*	5/18	0.4	0.4, 8.1	11.2	N	10/10	1.44	0.96 – 1.91	2.90	0.5
Cr	L	18/18	77	40 – 248	172.0	N	10/10	21.3	16.6 – 25.9	31.0	1
Cu	N	14/18	240.17	0.0 – 485.42	1250	N	10/10	107.00	77.7 –136	160.0	16
Pb	*	4/18	10.0	10.0, 263.3	299.0	L	10/10	85.0	58.5 – 112	150.0	31
Ag	*	4/18	30.5	1.3, 82.6	86.8	Na	Na	Na	Na	Na	1.0
V	N	16/18	34.8	26.91 – 42.68	63.0	Na	Na	Na	Na	Na	Na
Zn	L	18/18	422.1	243.3 – 1064.3	860.0	N	10/10	117	100-134	160	59

Na: Not available

Table A51. Comparison of Background Sediment Data to Round Lake Sediment

Substance	Tier I Background			Tier II Snail Lake			Tier I Round Lake		
	mean	SD	[value]	mean	SD	[value]	Central Tendency	Confidence Interval	Maximum
Al	7121	2018	10674	9570	1776	12826	8526	6489 – 10562	15400
Cd	0.6	0.3	1.0	1.44	0.67	3	0.4	0.4, 8.1	11.2
Cr	15.0	4.7	24.0	21	6	33	77	40 - 248	172
Cu	20	5.3	30.0	107	41	182	240.17	0.0 – 485.42	1250
Pb	20	6.6	32.0	85	37	153	10	10, 263.3	299
Ag	-	-	0.2	0.49	0.01	1	30.5	1.3, 82.6	86.8
V	9.1	3.3	15	-	-	-	34.8	26.91 – 42.68	63.0
Zn	72	17	101	117	24	160	422.1	243.3 – 1064.3	860
PCBs	-	-	-	-	-	0.33	2.3	NA	13

All values are expressed as ug/g

The "[value]" denotes the screening concentration value derived as [value] = $x + (t_{95\%} * SD)$ for Snail Lake data $t_{95\%}$=1.833, for Tier I background $t_{95\%}$=1.761

Central Tendency: Mean for normal data, geometric mean for lognormal data, and the median for unknown distributions

Confidence Interval: 95% confidence interval on the mean for normal data, 95% confidence interval on the geometric mean for lognormal data, and 5th and 95th percentiles if the distribution is unknown

"-" no data was available

Table A52. Selected Chemicals of Concern(COCs) for the Exposure and Effects Analysis

Medium	Study Site					
	Marsden Lake (North)	Marsden Lake (South)	Pond G	Rice Creek	Sunfish Lake	Round Lake
Surface Water	Aluminum Barium Lead Manganese Mercury Zinc	Manganese	Aluminum Antimony Arsenic Barium Beryllium Cadmium Chromium Cobalt Copper Lead Manganese Selenium Thallium	None	Cadmium Mercury Zinc	Barium Cadmium Zinc
Surface Sediment	Antimony Arsenic Vanadium SVOCs	Aluminum Cadmium Lead Vanadium SVOCs	Aluminum Antimony Arsenic Chromium Copper Lead Vanadium SVOCs OC pesticides Aroclor-1248	None	Aluminum Chromium Lead Vanadium Zinc	Cadmium Chromium Copper Lead Silver Vanadium Zinc PCBs

Table A53. Marsden Lake North Summary of the Tier II COC Selection Process

COCPs*	Criterion 3			Criterion 4	Criterion 5		Final COC list	
	Water	Sediment			Water	Sediment	✦	
Al		NE				NE	w	Al
Sb	✖				✖		s	Sb
As	✖				✖		s	As
Ba		NE				NE	w	Ba
Cd			▲		✖	✖		✖
Cr	✖	NE			✖	NE		✖
Co	✖	NE			✖	✖		✖
Cu	✖				✖	NE		✖
Pb		NE	▲			✖	b	Pb
Mn		NE				NE	w	Mn
Hg			▲			✖	w	Hg
Ag	✖	NE			✖	NE		✖
Tl	✖	NE			✖	NE		✖
V	NE	NE			NE	NE	s	V
Zn			▲			✖	w	Zn
SVOCs	NE		▲		NE			SVOCs
OC Pesticides	NE				NE	✖		✖

*: See Table A1 for how this list of COCPs was determined.

Criterion 3=Comparability with toxicological benchmarks and bioaccumulation potential through the food chain, Criterion 4=Chemical distribution and detection frequency in sampled media, Criterion 5=Essential nutrients at site concentrations, and Criterion 6=Site-relatedness and comparability with background concentrations.

✖: indicates at which step in the process the analyte was eliminated as a COC, once the chemical is eliminated an x will appear across the row. Blank spaces indicate that the chemical was not eliminated at by that criterion.

NE: Not evaluated as a COC for that media.

▲: Indicates a chemical that has the potential to bioaccumulate and must be carried through as a COC.

✦: In the final COC list s=sediment COC, w=water COC, b=COC specific to both media

Table A54. Marsden Lake South Summary of Tier II COC Selection Process

COPCs*	**Criterion 3**		**Criterion 4**	**Criterion 5**		**Final COC list**	
	Water	Sediment		Water	Sediment	✦	
Al				✖		s	Al
Sb	✖			✖	✖		✖
As	✖			✖	✖		✖
Ba		NE		✖	NE	w	✖
Cd		▲		✖		s	Cd
Cr	✖			✖	✖		✖
Co	✖			✖	✖		✖
Cu				✖	✖		✖
Pb				✖		s	Pb
Mn		NE			NE	w	Mn
Hg		▲		✖	✖		✖
Ag	✖	NE		✖	NE		✖
Tl	✖	NE		✖	NE		✖
V	NE	NE		NE	NE	s	V
Zn		▲		✖	✖		✖
SVOCs	NE	▲		NE		s	SVOCs
OC Pesticides	NE	▲		NE	✖		✖

*: See Table A2 for how this list of COCPs was determined.

Criterion 3=Comparability with toxicological benchmarks and bioaccumulation potential through the food chain, Criterion 4=Chemical distribution and detection frequency in sampled media, Criterion 5=Essential nutrients at site concentrations, and Criterion 6=Site-relatedness and comparability with background concentrations.

✖: indicates at which step in the process the analyte was eliminated as a COC, once the chemical is eliminated an x will appear across the row. Blank spaces indicate that the chemical was not eliminated at by that criterion.

NE: Not evaluated as a COC for that media.

▲: Indicates a chemical that has the potential to bioaccumulate and must be carried through as a COC.

✦: In the final COC list s=sediment COC, w=water COC, b=COC specific to both media

Table A55. Pond G Summary of the Tier II COC Selection Process

COPCs*	Criterion 3 Water	Criterion 3 Sediment	Criterion 4	Criterion 5 Water	Criterion 5 Sediment	Final COC list ✦	
Al	NA	NA	NA	NA	NA	b	Al
Sb	NA	NA	NA	NA	NA	b	Sb
As	NA	NA	NA	NA	NA	b	As
Ba	NA	NA	NA	NA	NA	w	Ba
Be	NA	NA	NA	NA	NA	w	Be
Cd	NA	NA	NA	NA	NA	w	Cd
Cr	NA	NA	NA	NA	NA	b	Cr
Co	NA	NA	NA	NA	NA	w	Co
Cu	NA	NA	NA	NA	NA	b	Cu
Pb	NA	NA	NA	NA	NA	b	Pb
Mn	NA	NA	NA	NA	NA	w	Mn
Se	NA	NA	NA	NA	NA	w	Se
Tl	NA	NA	NA	NA	NA	w	Tl
V	NA	NA	NA	NA	NA	s	V
SVOCs	NA	NA	NA	NA	NA	s	SVOCs
OC Pesticides	NA	NA	NA	NA	NA	s	OC Pesticides
Aroclor-1248	NA	NA	NA	NA	NA	s	Aroclor-1248

*: See Table A3 for how this list of COCPs was determined.

Criterion 3=Comparability with toxicological benchmarks and bioaccumulation potential through the food chain, Criterion 4=Chemical distribution and detection frequency in sampled media, Criterion 5=Essential nutrients at site concentrations, and Criterion 6=Site-relatedness and comparability with background concentrations.

✖: indicates at which step in the process the analyte was eliminated as a COC, once the chemical is eliminated an x will appear across the row. Blank spaces indicate that the chemical was not eliminated at by that criterion.

NE: Not evaluated as a COC for that media.

▲: Indicates a chemical that has the potential to bioaccumulate and must be carried through as a COC.

✦: In the final COC list s=sediment COC, w=water COC, b=COC specific to both media

Table A56. Rice Creek Summary of Tier II COC selection process

Analytes	Final 1997 Tier I COPC list$^+$		Criterion 1	Criterion 2	Criterion 3		Criterion 4	Criterion 5		Final COC list	
			Water	Water	Water	Sediment		Water	Sediment	*	
Al	w	Al				NE		✖	NE	w	✖
Sb		✖									
As		✖									
Ba	w	Ba				NE		✖	NE	w	✖
Be		✖									
Cd	w	Cd				NE ▲			NE	w	✖
Ca		✖									
Cr		✖									
Co		✖									
Cu	w	Cu			✖	NE	✖	✖	NE	w	✖
Cn^-		✖									
Fe		✖									
Pb		✖									
Mg		✖									
Mn		✖									
Hg	w	Hg				NE ▲		✖	NE	w	✖
Mo		✖									
Ni		✖									
K		✖									
Se		✖									
Ag	w	Ag	✖	✖	✖	NE	✖	✖	NE	w	✖
Na		✖									
Tl		✖									
V		✖									
Zn		✖									

Criterion 1= Chemicals not detected, Criterion 2= Data quality review and comparison with lab blanks, Criterion 3=Comparability with toxicological benchmarks and bioaccumulation potential through the food chain, Criterion 4=Chemical distribution and detection frequency in sampled media, Criterion 5 =Site-relatedness and comparability with background concentrations.

+: Only the chemicals listed here were carried through the Tier II COC selection process. ✖ in this column indicates chemicals that were eliminated in the 1997 Tier I and not evaluated in the Tier II process.

✖: indicates at which step in the process the analyte was eliminated as a COC, once the chemical is eliminated an x will appear across the row. Blank spaces indicate that the chemical was not eliminated based on that criterion.

NE: Not evaluated as a COC for that media.

▲: Indicates a chemical that has the potential to bioaccumulate and must be carried through as a COC.

*: In the final COC list s=sediment COC, w=water COC, b=COC specific to both media

Table A57. Sunfish Lake Summary of Tier II COC Selection Process

Analytes	Final 1997 Tier I COPC list^{+}		Criterion 1	Criterion 2	Criterion 3		Criterion 4	Criterion 5		Final COC list *	
			Water	Water	Water	Sediment		Water	Sediment		
Al	b	Al			✖	✖	✖	✖		s	Al
Sb		✖									
As		✖									
Ba	w	Ba				NE		✖	NE	w	✖
Be		✖									
Cd	w	Cd				NE ▲			NE	w	Cd
Ca		✖									
Cr	s	Cr	NE	NE	NE			NE		s	Cr
Co		✖									
Cu	b	Cu			✖		✖	✖	✖	w	✖
Cn^{-}		✖									
Fe	s										
Pb	s	Pb	NE	NE	NE	▲		NE		s	Pb
Mg		✖									
Mn		✖									
Hg	w	Hg				NE ▲		✖	NE	w	Hg
Mo		✖									
Ni		✖									
K		✖									
Se		✖									
Ag	w	Ag	✖	✖	✖	NE	✖	✖	NE	w	✖
Na		✖									
Tl		✖									
V	s	V	NE	NE	NE			NE		s	V
Zn	b	Zn								b	Zn

Criterion 1= Chemicals not detected, Criterion 2= Data quality review and comparison with lab blanks, Criterion 3=Comparability with toxicological benchmarks and bioaccumulation potential through the food chain, Criterion 4=Chemical distribution and detection frequency in sampled media, Criterion 5 =Site-relatedness and comparability with background concentrations.

+: Only the chemicals listed here were carried through the Tier II COC selection process. ✖ in this column indicates chemicals that were eliminated in the 1997 Tier I and not evaluated in the Tier II process.

✖: indicates at which step in the process the analyte was eliminated as a COC, once the chemical is eliminated an x will appear across the row. Blank spaces indicate that the chemical was not eliminated based on that criterion.

NE: Not evaluated as a COC for that media.

▲: Indicates a chemical that has the potential to bioaccumulate and must be carried through as a COC.

*: In the final COC list s=sediment COC, w=water COC, b=COC specific to both media

Table A58. Round Lake Summary of the Tier II COC Selection Process

Analytes	Final 1997 Tier I COPC list^{+}		Criterion 1	Criterion 2	Criterion 3		Criterion 4	Criterion 5		Final COC list *	
			Water	Water	Water	Sediment		Water	Sediment		
Al	s	Al	NE	NE	NE	✖	✖	NE	✖		✖
Sb		✖									
As		✖									
Ba	w	Ba								w	Ba
Be		✖									
Cd	b	Cd				▲				b	Cd
Ca		✖									
Cr	s	Cr	NE	NE	NE			NE		s	Cr
Co		✖									
Cu	b	Cu			✖			✖		s	Cu
Cn^-		✖									
Fe		✖									
Pb	s	Pb	NE	NE	NE	▲		NE		s	Pb
Mg		✖									
Mn		✖									
Hg	w	Hg				NE ▲			NE		
Mo		✖									
Ni		✖									
K		✖									
Se		✖									
Ag	b	Ag	✖	✖	✖		✖			s	Ag
Na		✖									
Tl		✖									
V	s	V	NE	NE	NE			NE		s	V
Zn	b	Zn				▲				b	Zn
PCBs	s	PCBs	NE	NE	NE	▲		NE		s	PCBs

Criterion 1= Chemicals not detected, Criterion 2= Data quality review and comparison with lab blanks, Criterion 3=Comparability with toxicological benchmarks and bioaccumulation potential through the food chain, Criterion 4=Chemical distribution and detection frequency in sampled media, Criterion 5 =Site-relatedness and comparability with background concentrations.

+: Only the chemicals listed here were carried through the Tier II COC selection process. ✖ in this column indicates chemicals that were eliminated in the 1997 Tier I and not evaluated in the Tier II process.

✖: indicates at which step in the process the analyte was eliminated as a COC, once the chemical is eliminated an x will appear across the row. Blank spaces indicate that the chemical was not eliminated based on that criterion.

NE: Not evaluated as a COC for that media.

▲: Indicates a chemical that has the potential to bioaccumulate and must be carried through as a COC.

*: In the final COC list s=sediment COC, w=water COC, b=COC specific to both media

APPENDIX B-1

FIGURES

(General Site Maps and CSMs)

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Figure B1 - 1. Ecological Risk Assessment Framework

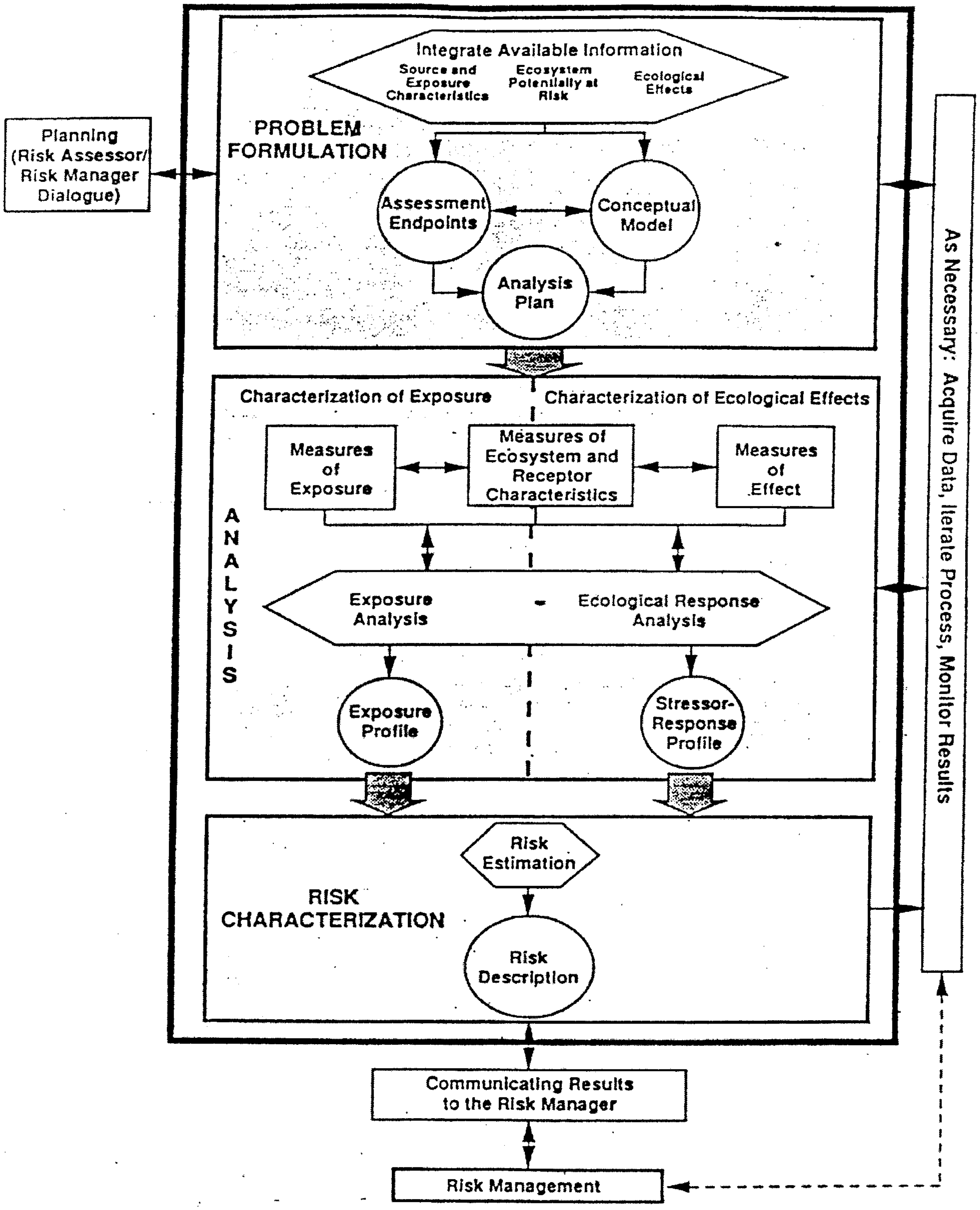


Figure B1 - 2. Twin Cities Army Ammunition Plant Aquatic Sites

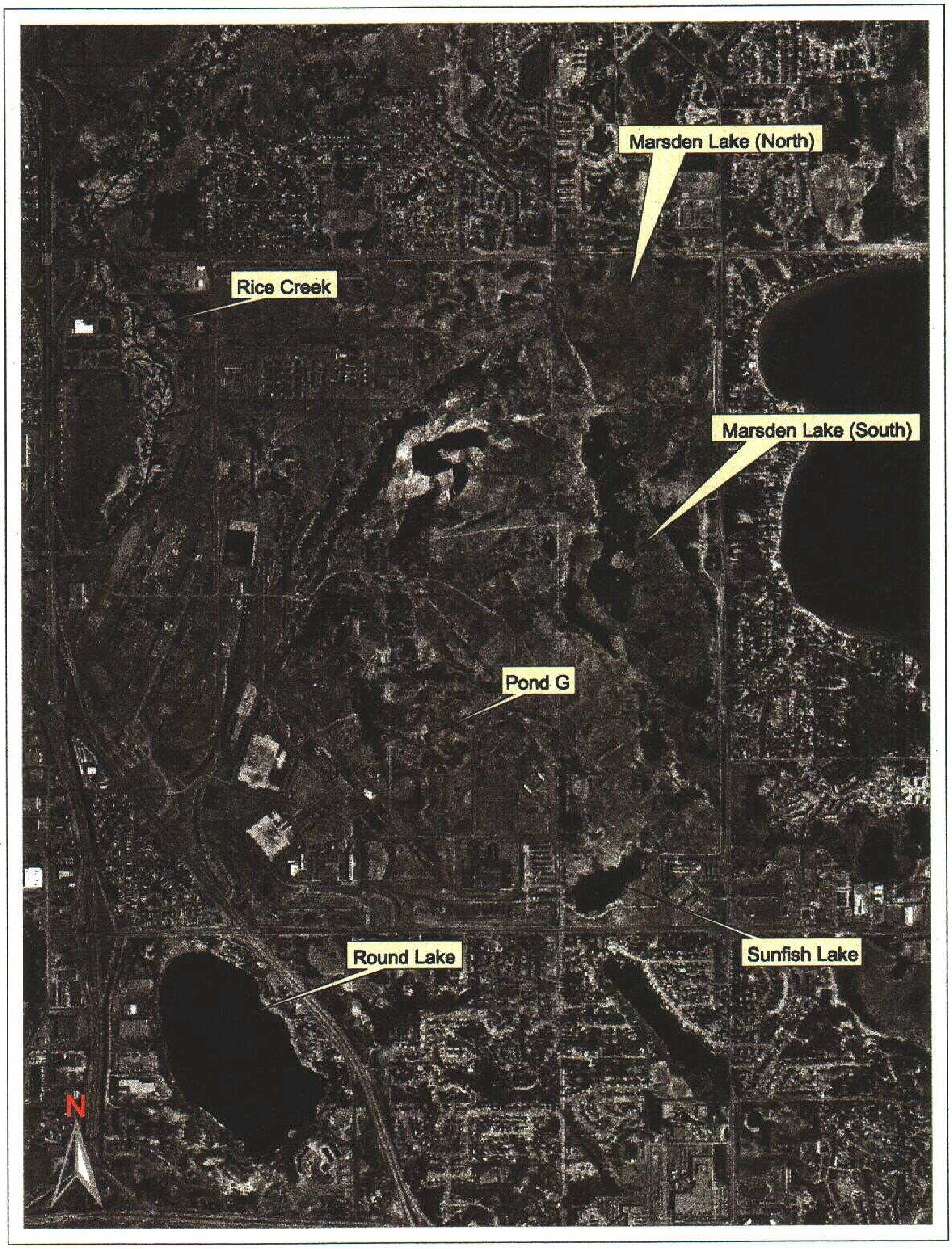


Figure B1 - 3. Marsden Lake

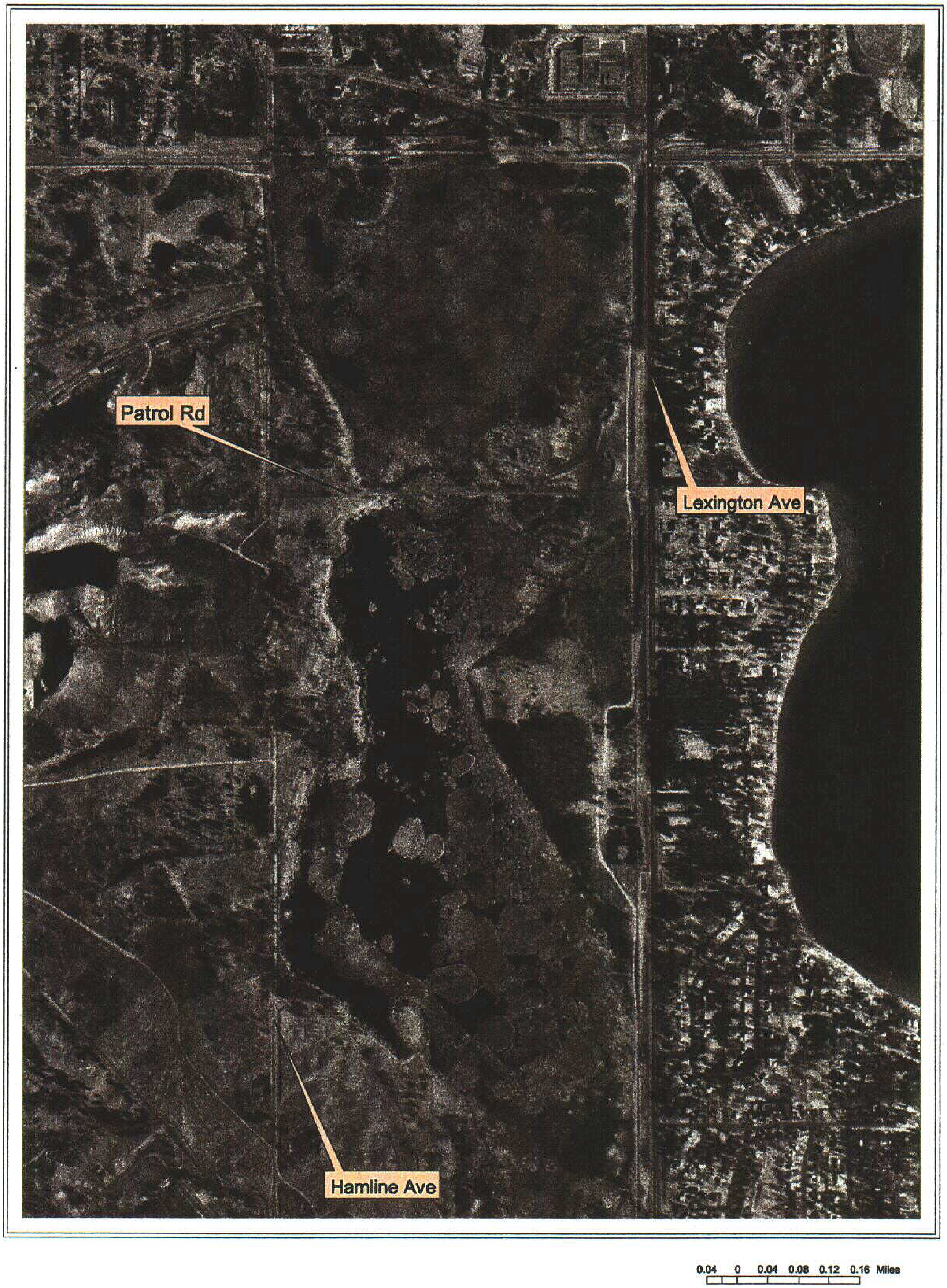


Figure B1 - 4. Marsden Lake Initial Conceptual Site Model Diagram

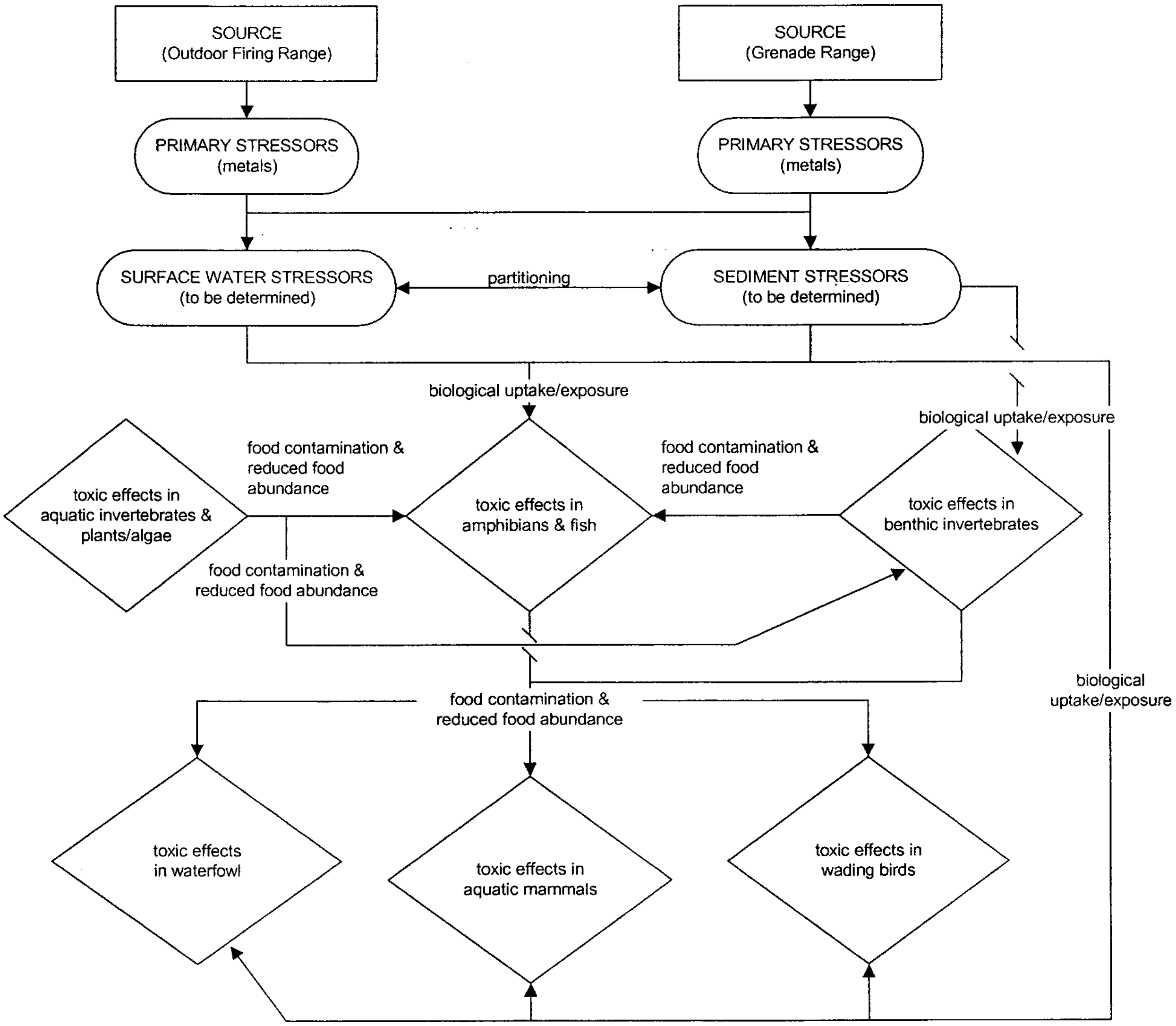


Figure B1 - 5. Pond G

Figure B1 - 6. Pond G Initial Conceptual Site Model Diagram

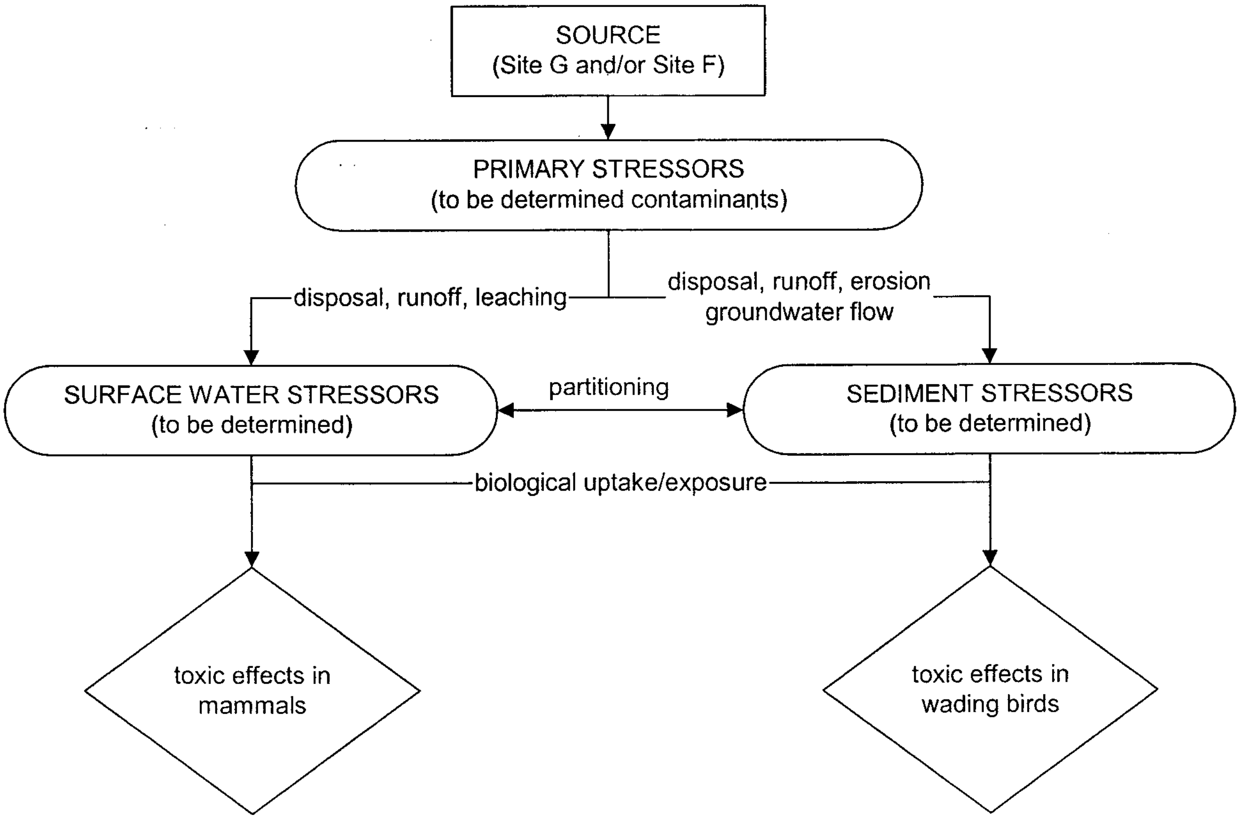


Figure B1 - 7. Rice Creek

Figure B1 - 8. Rice Creek Initial Conceptual Site Model Diagram

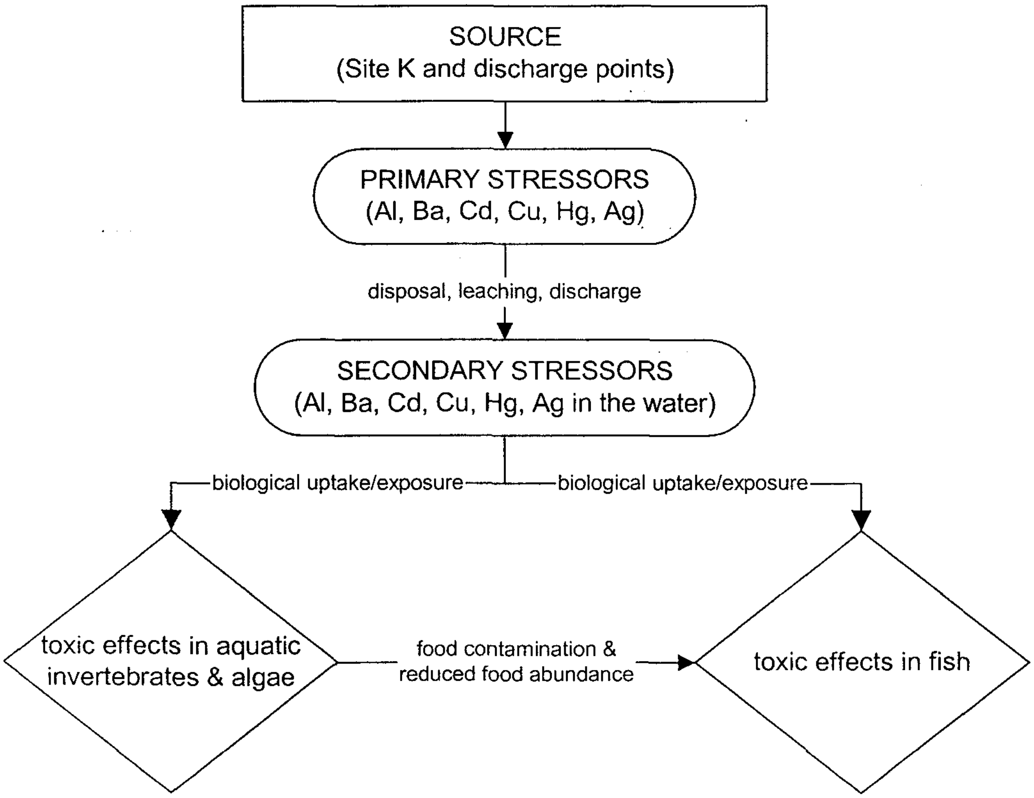


Figure B1 - 9. Sunfish Lake

Figure B1 - 10. Sunfish Lake Initial Conceptual Site Model Diagram

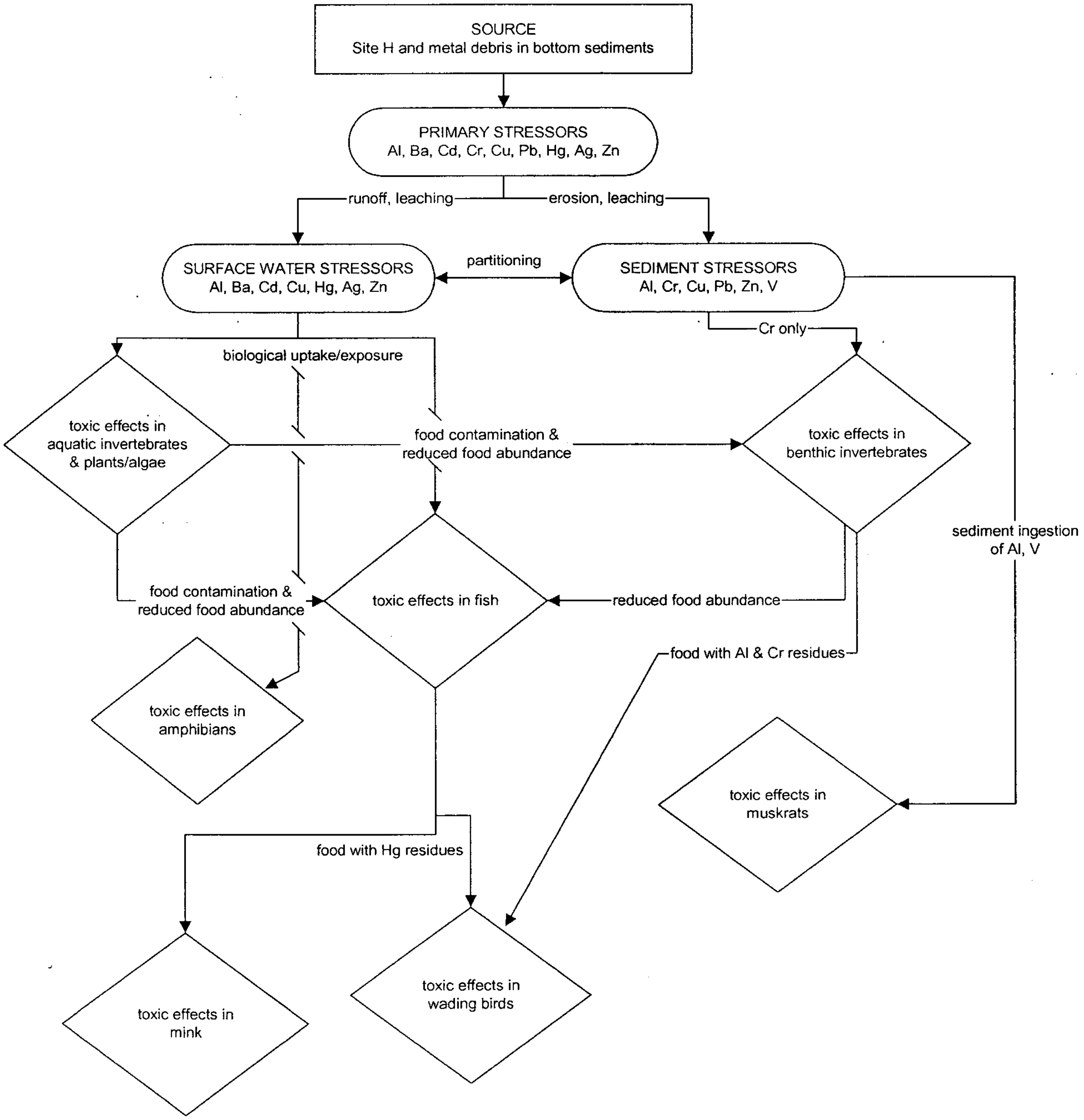
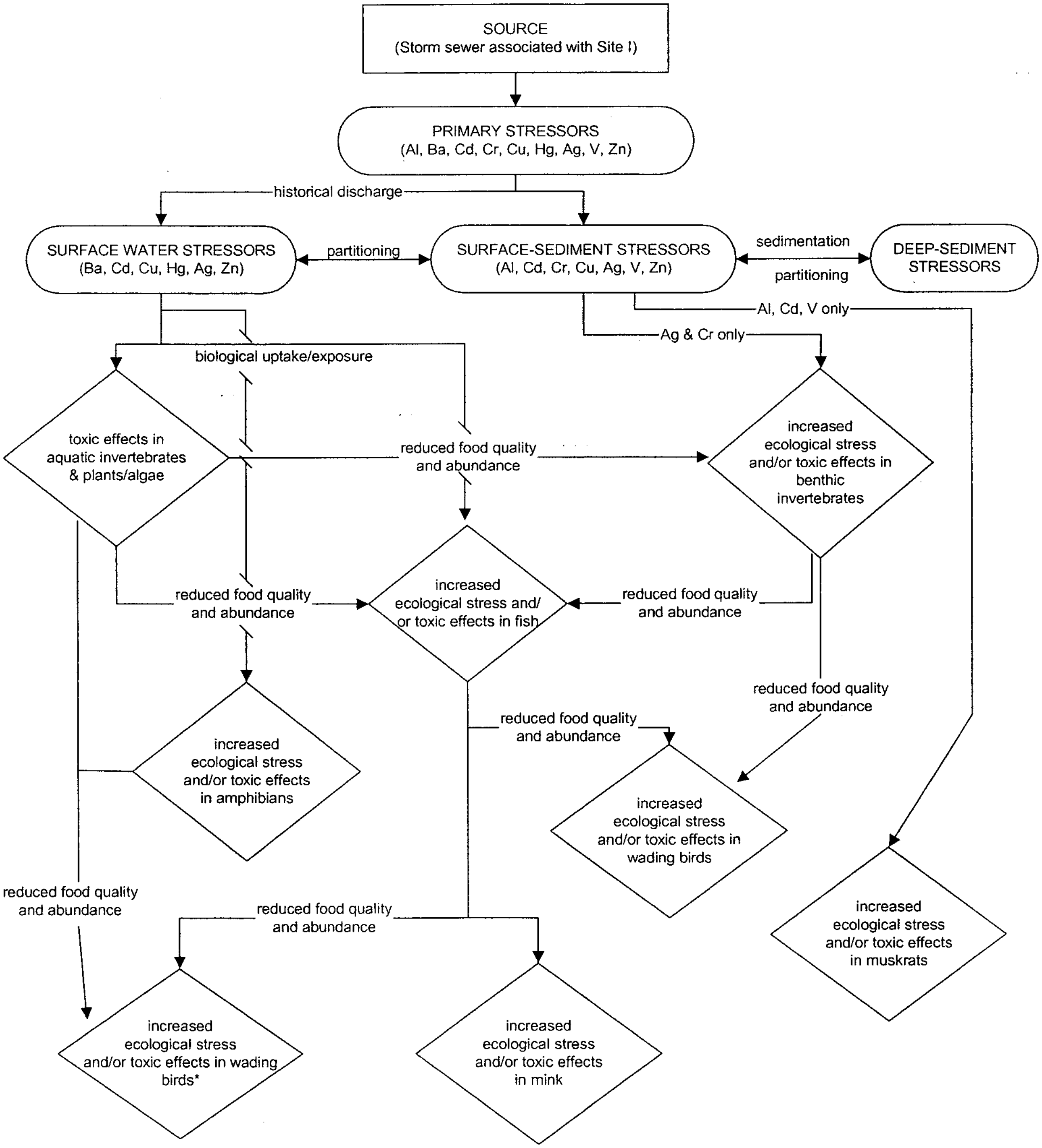


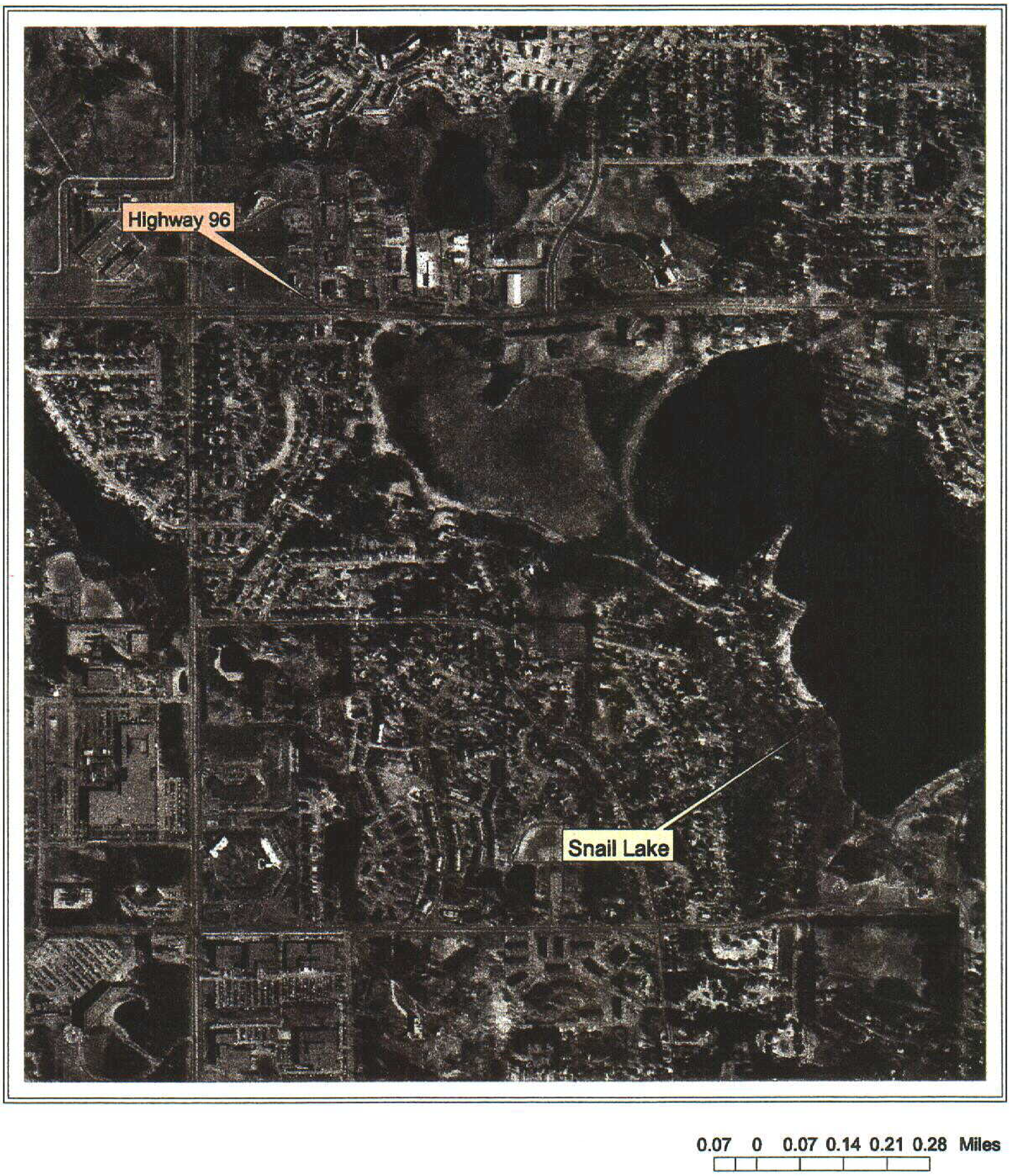
Figure B1 - 11. Round Lake

Figure B1 - 12. Round Lake Initial Conceptual Site Model Diagram



* This scenario applies to the potential future exposure scenarios only.
See text for details.

Figure B1 - 13. Snail Lake (Reference Site)



Appendix B2

Tier II Surface Water and Sediment Data Distributions

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Figure B2 -1. Aluminum surface water data distributions

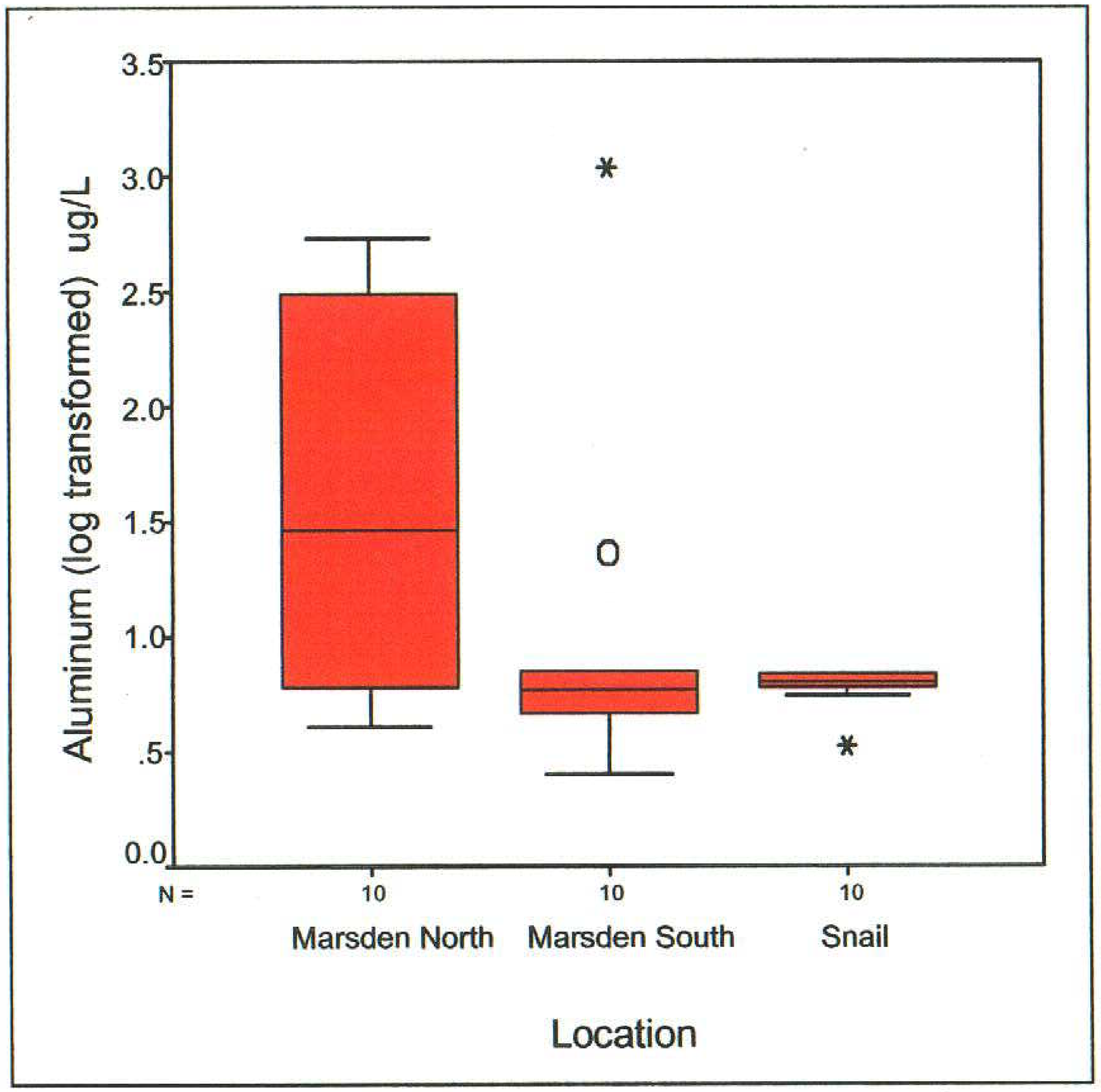


Figure B2 - 2. Antimony surface water data distributions

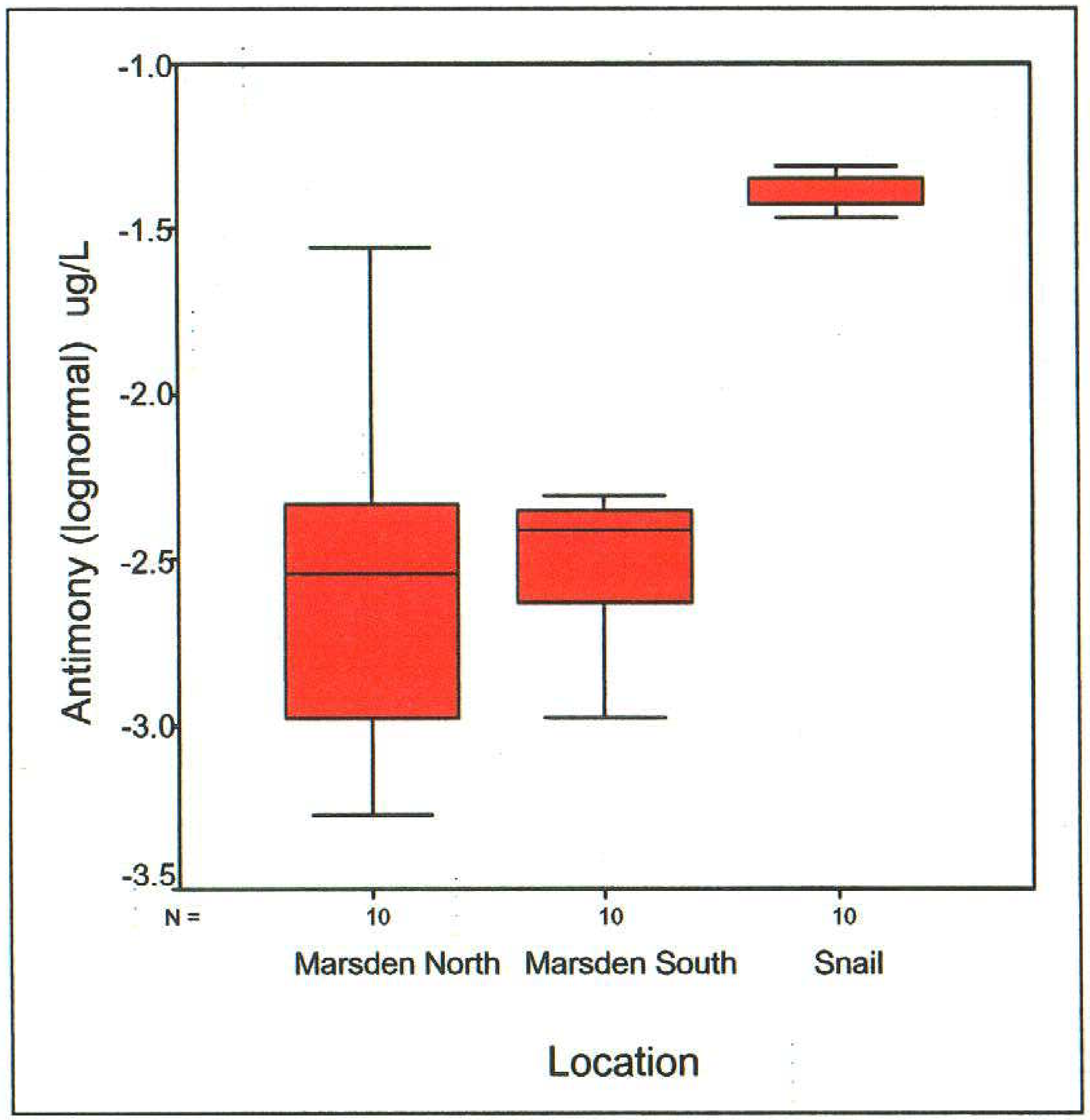


Figure B2 - 3. Arsenic surface water data distributions

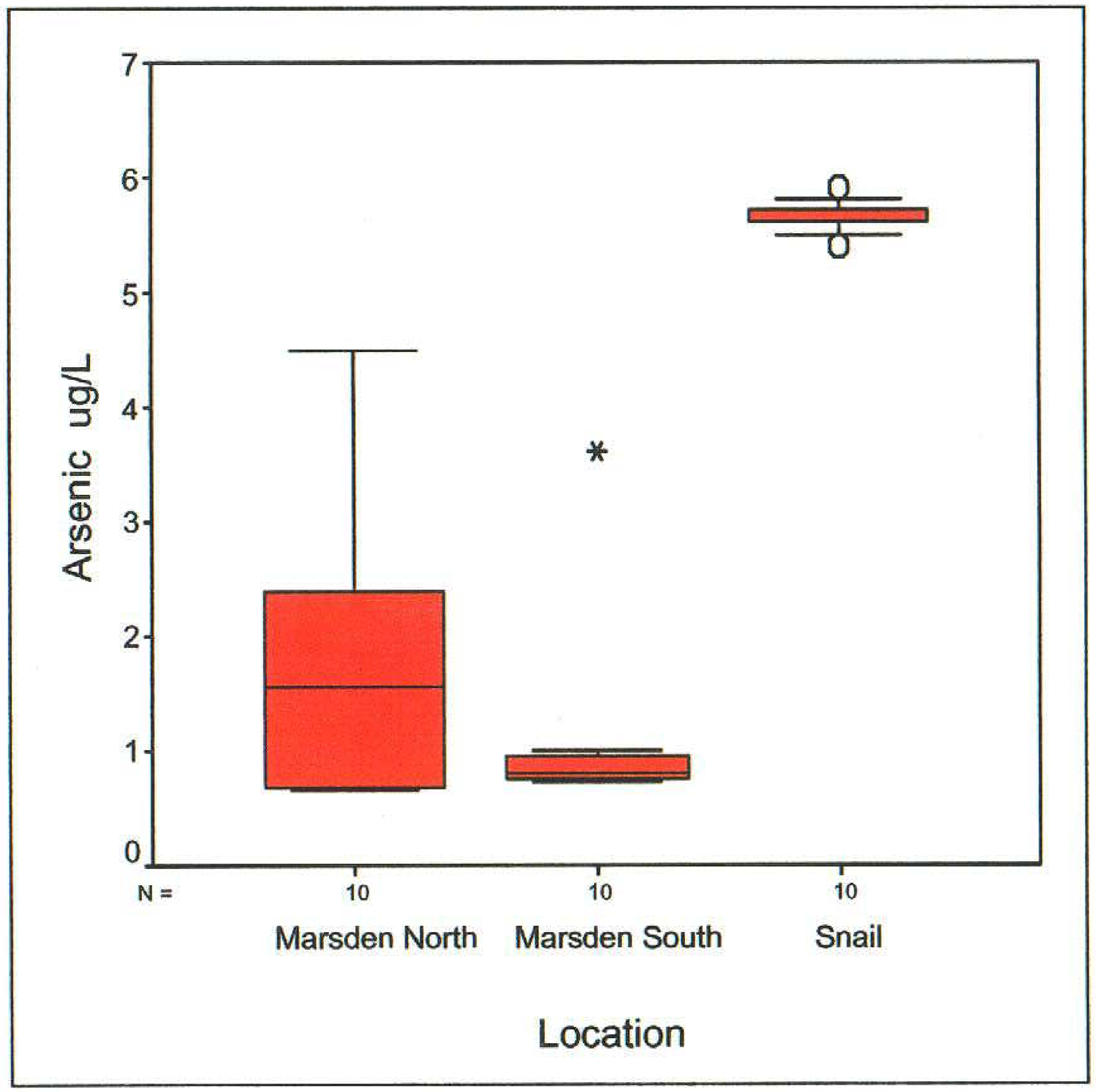


Figure B2 - 4. Barium surface water data distributions

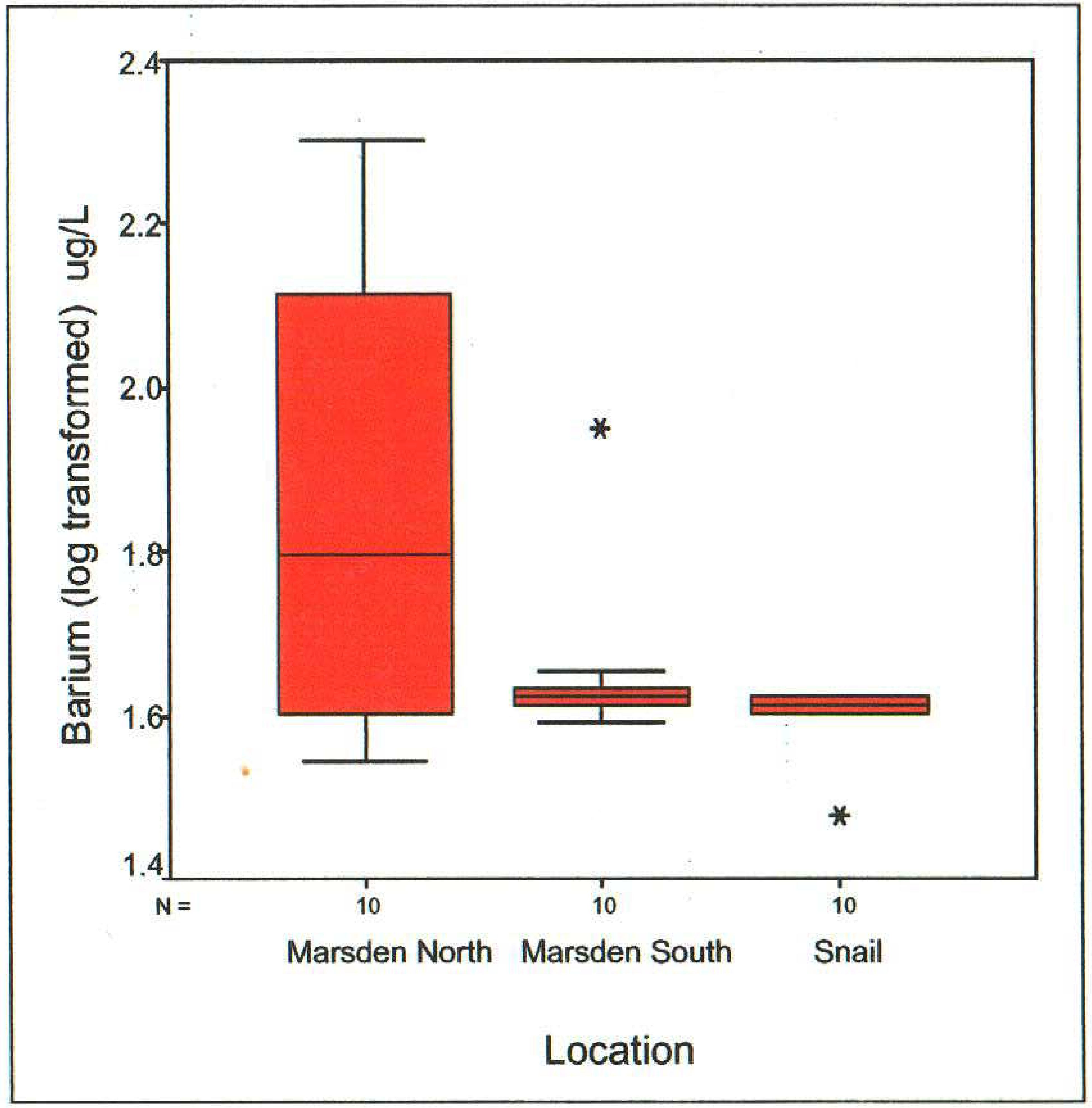
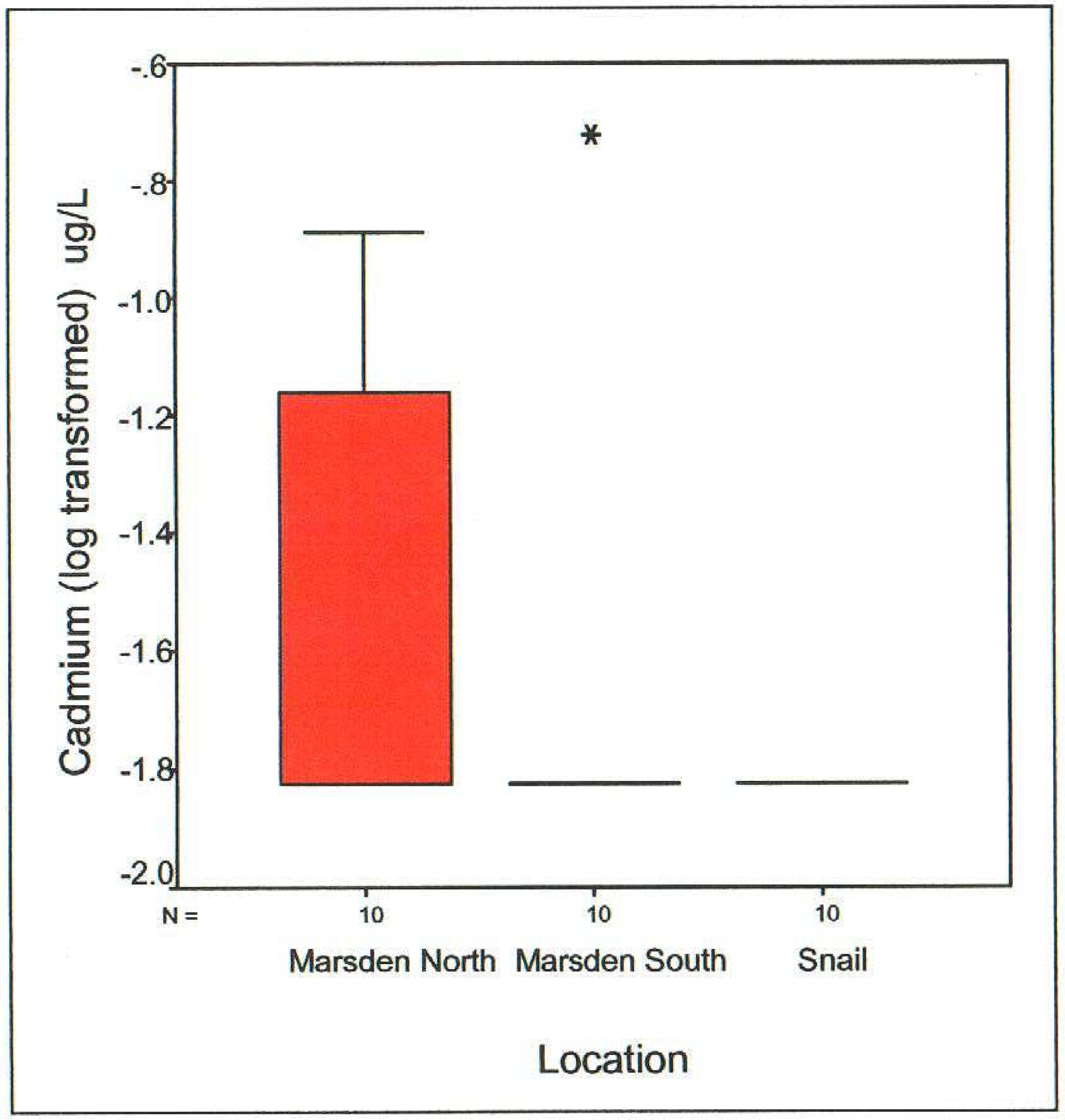


Figure B2 - 5. Cadmium surface water data distributions



FigureB2 - 6. Chromium surface water data distribution

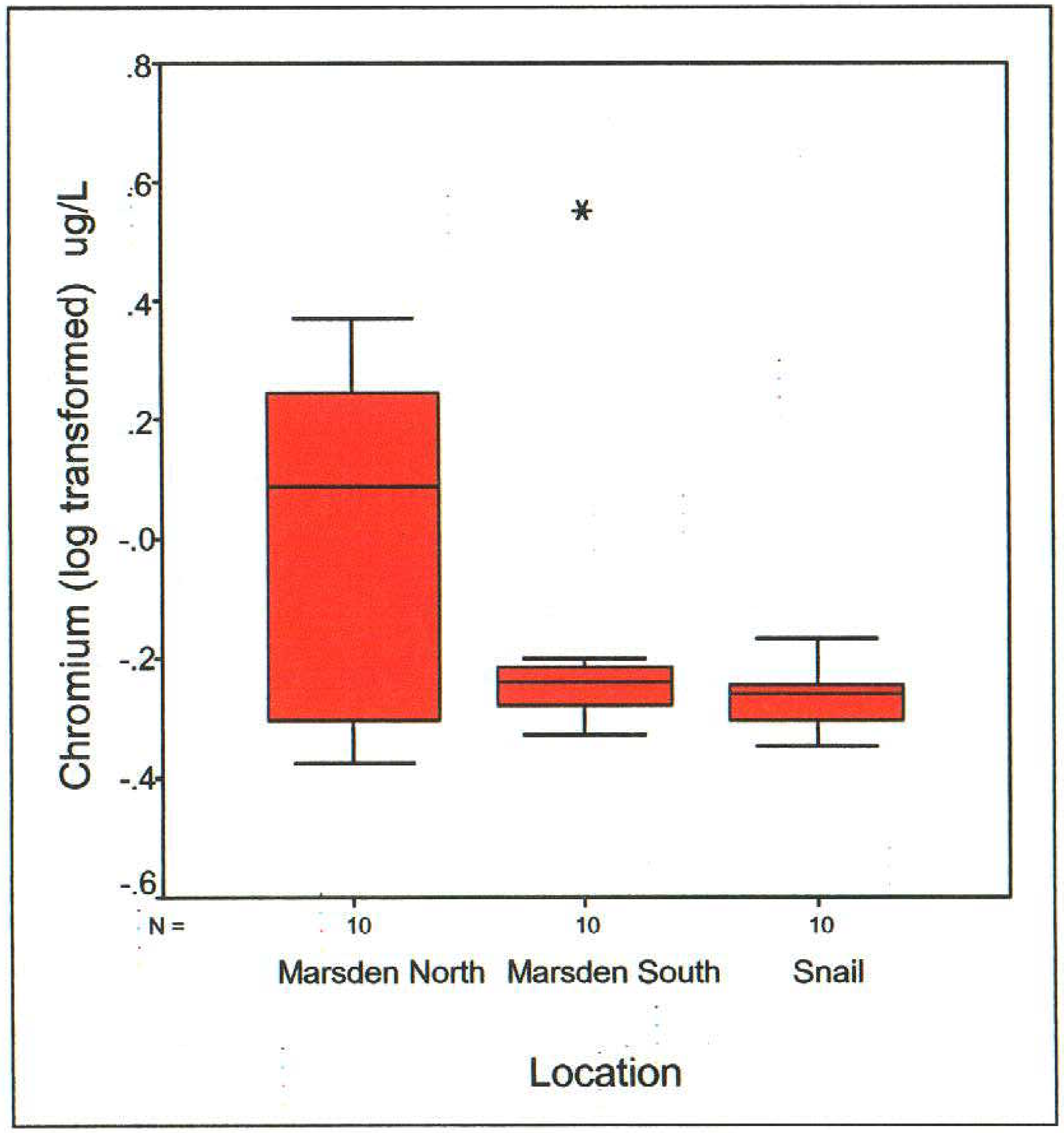


Figure B2 - 7. Cobalt surface water data distributions

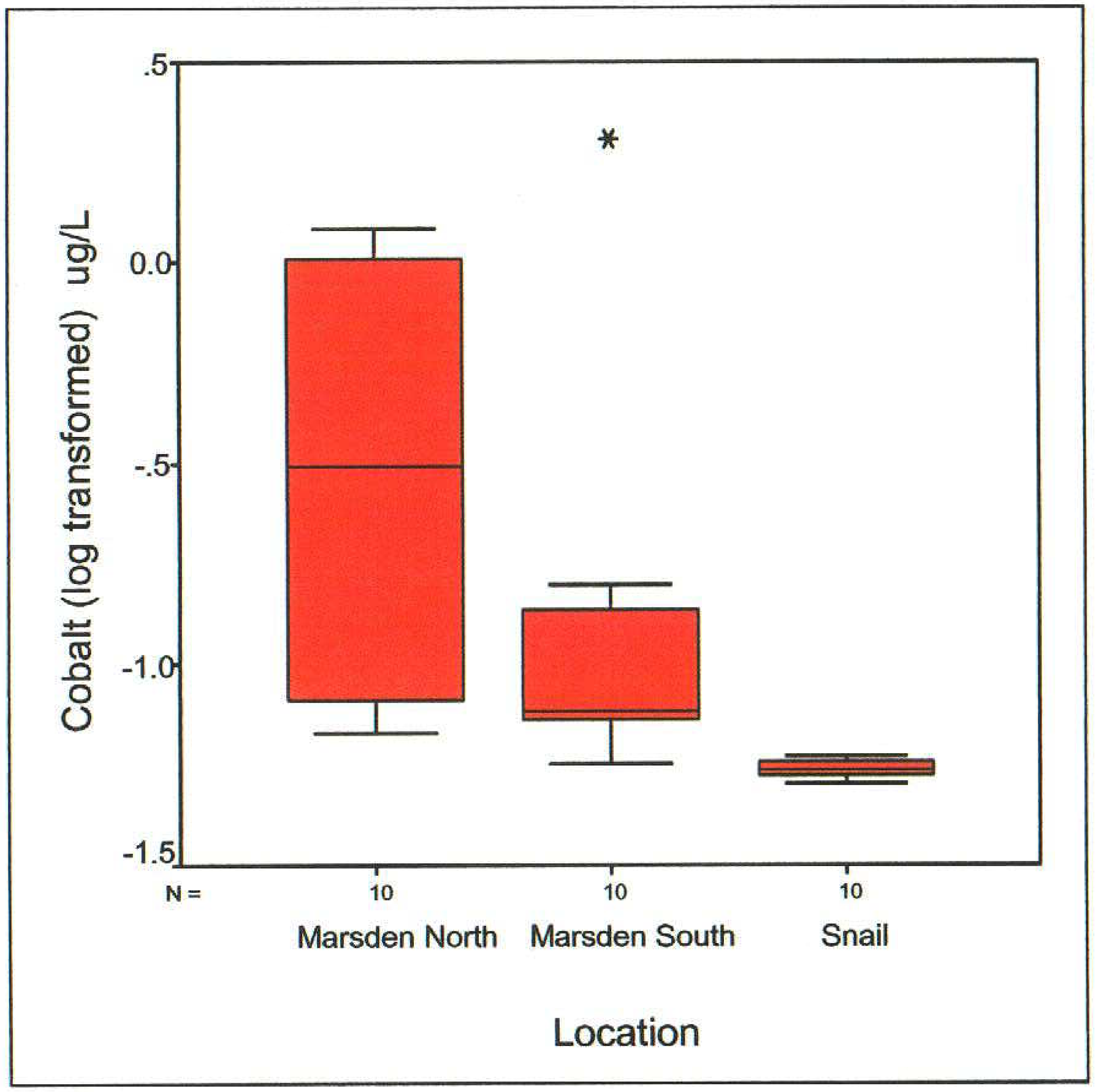


Figure B2 - 8. Copper surface water data distributions

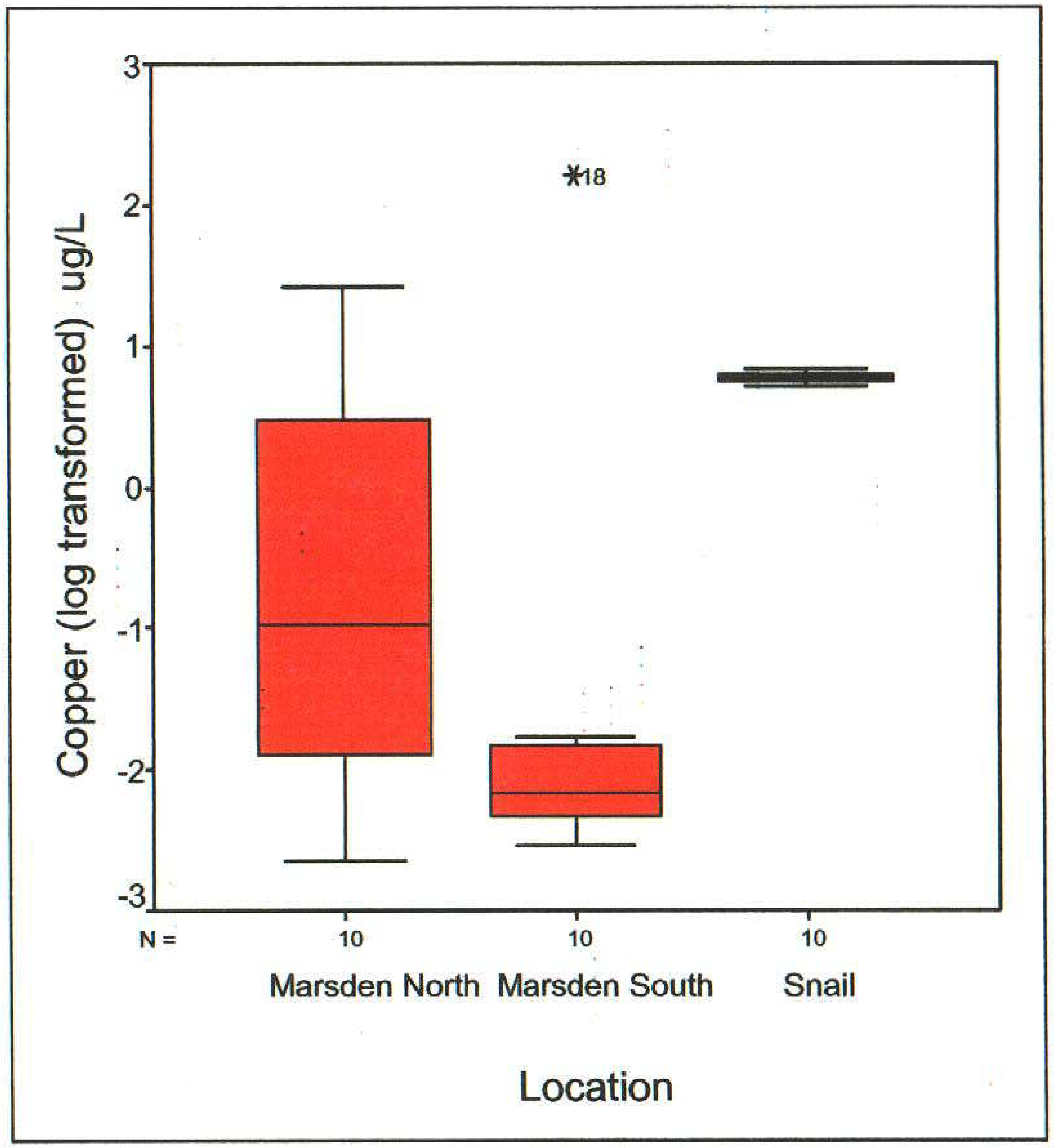


Figure B2 - 9. Lead surface water data distributions

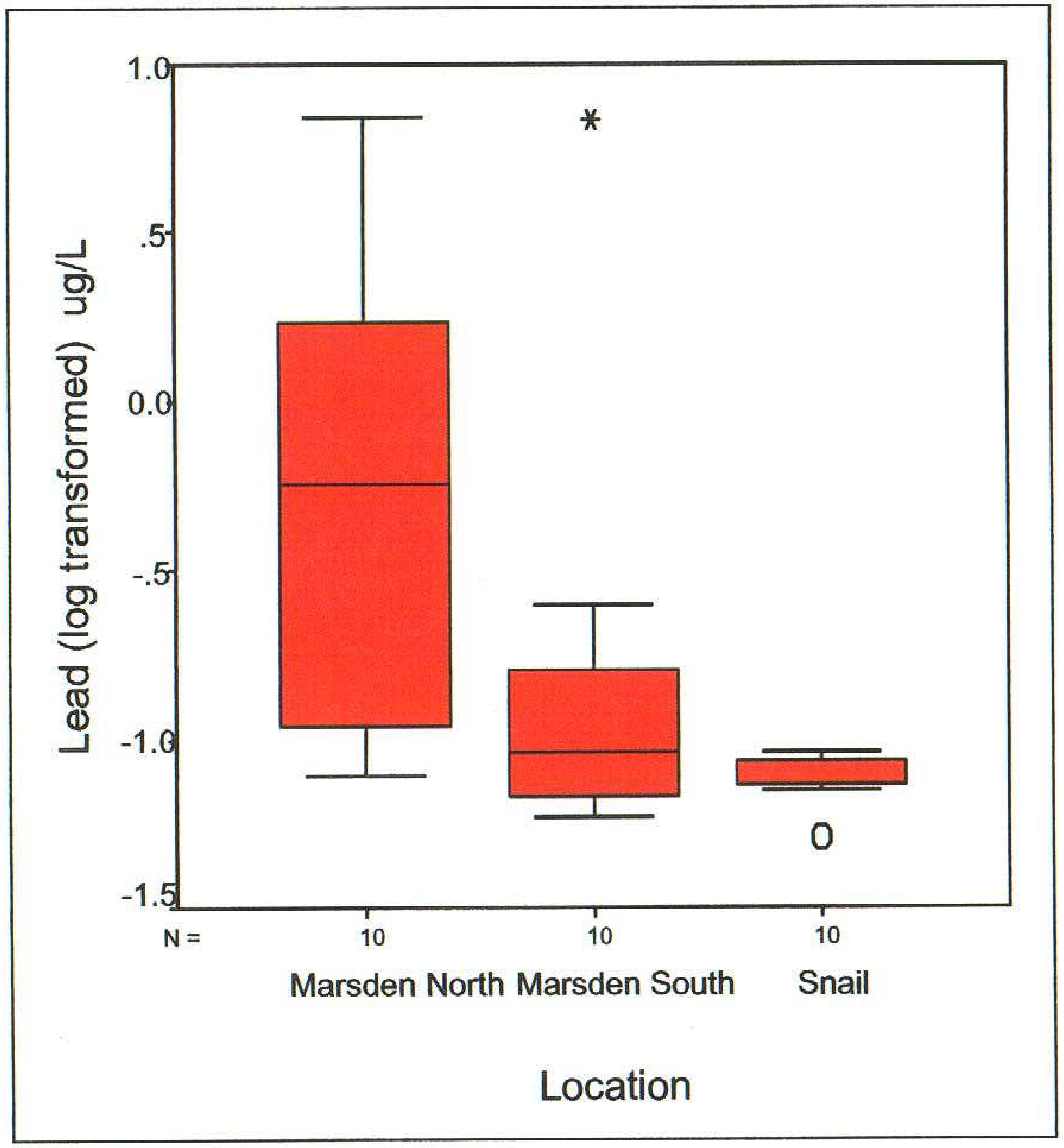


Figure B2 - 10. Magnesium surface water data distributions

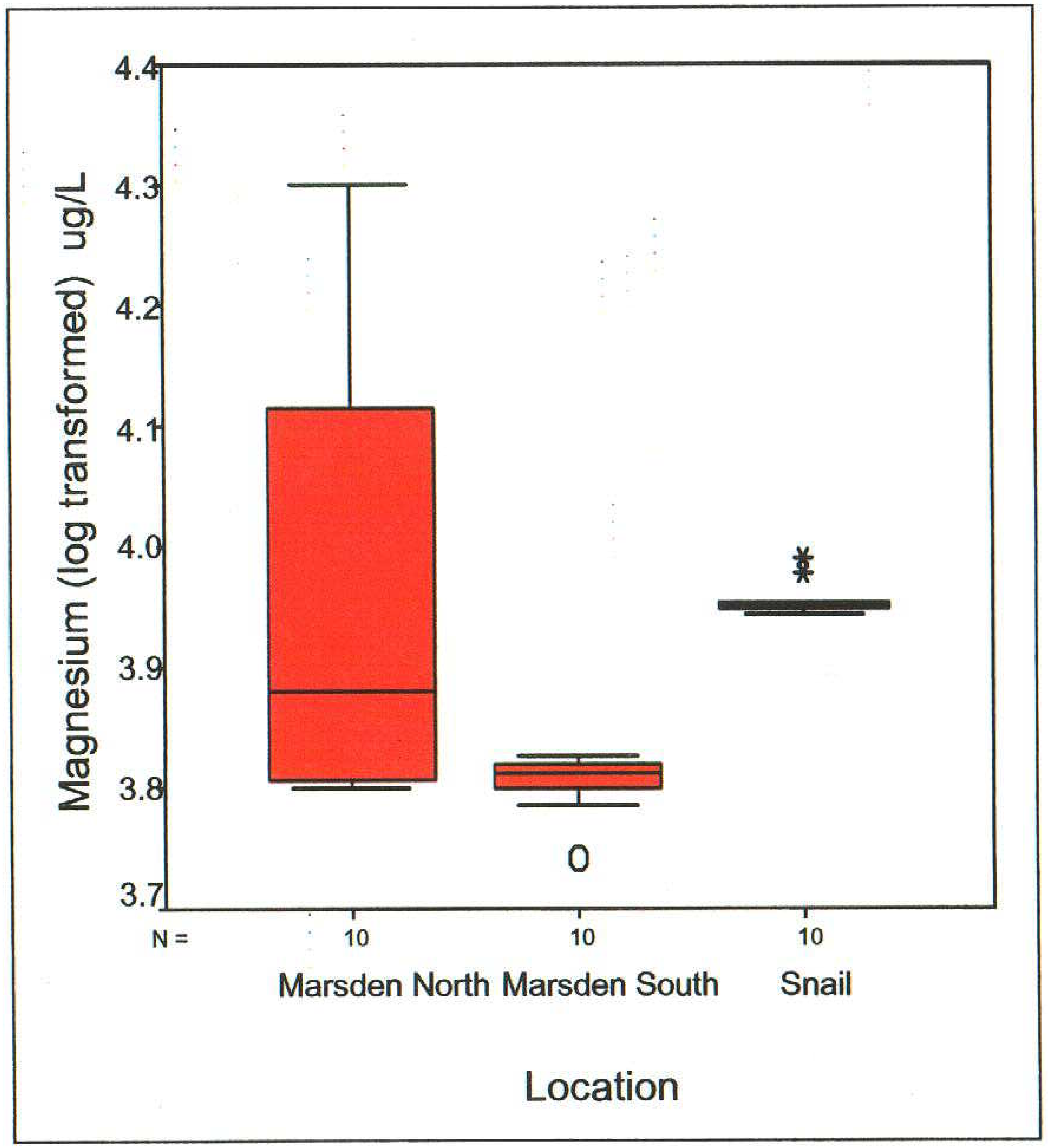


Figure B2 - 11. Manganese surface water data distributions

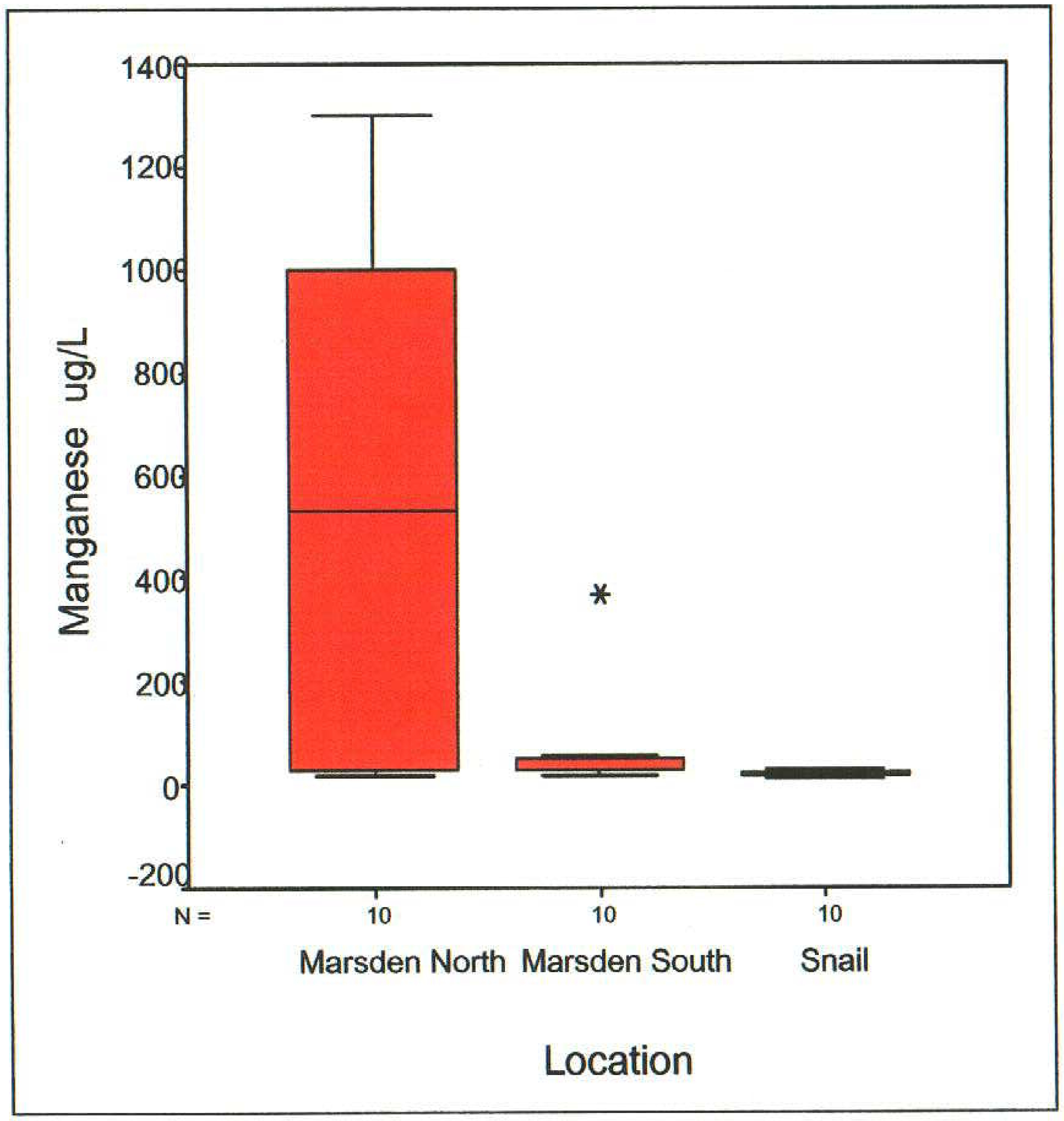


Figure B2 - 12. Mercury surface water data distributions

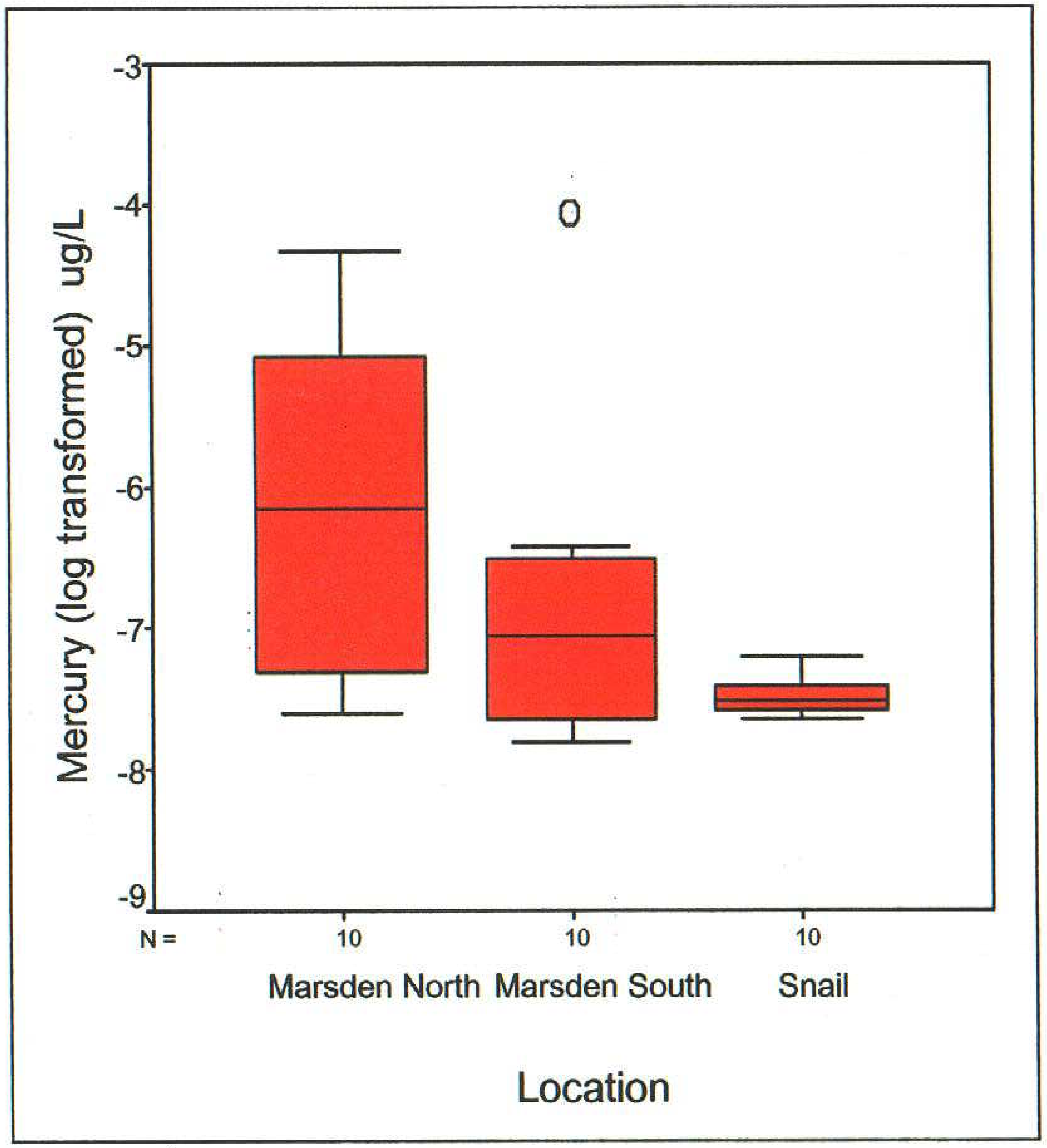


Figure B2 - 13. Nickel surface water data distributions

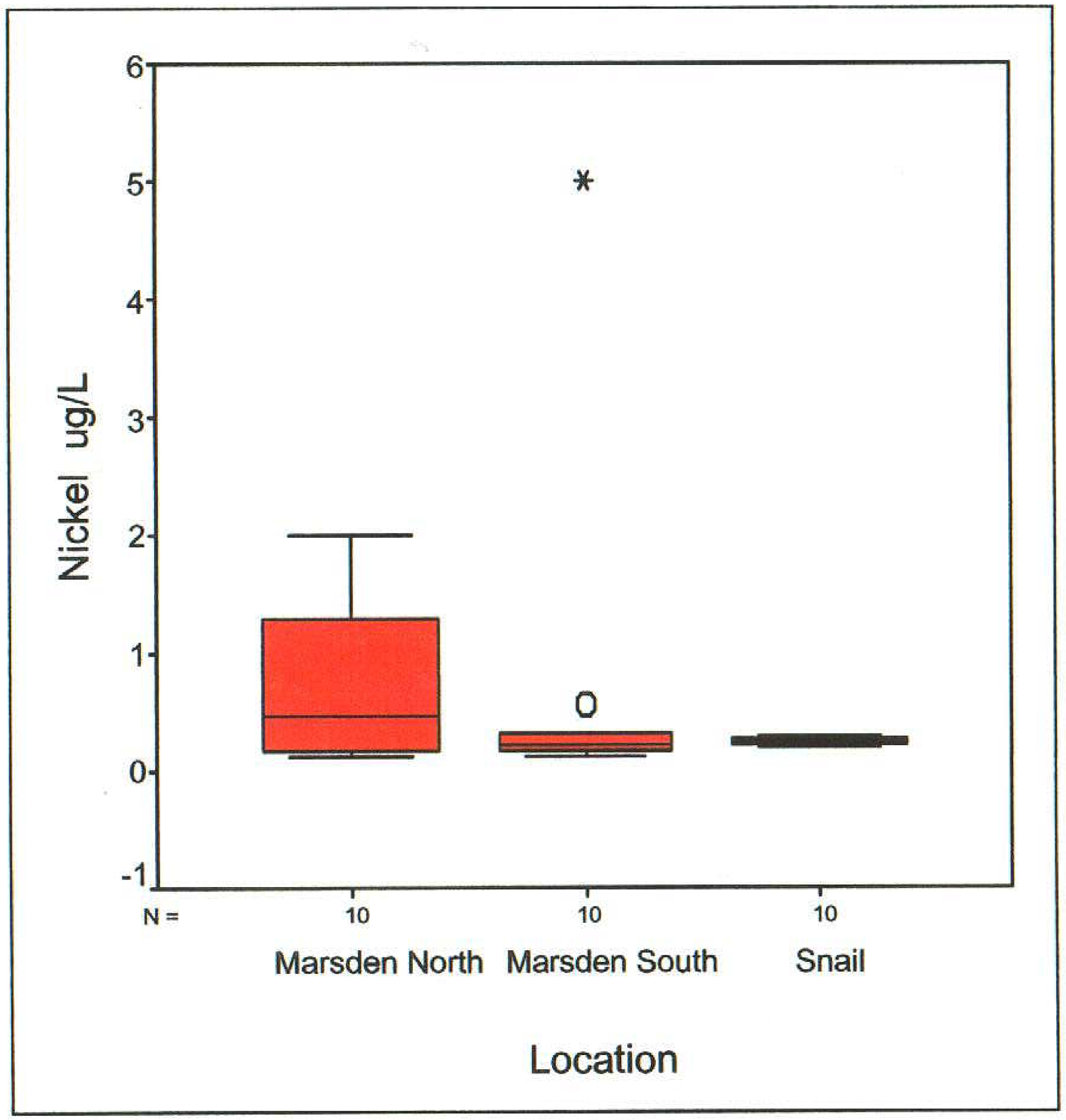


Figure B2 - 14. Silver surface water data distributions

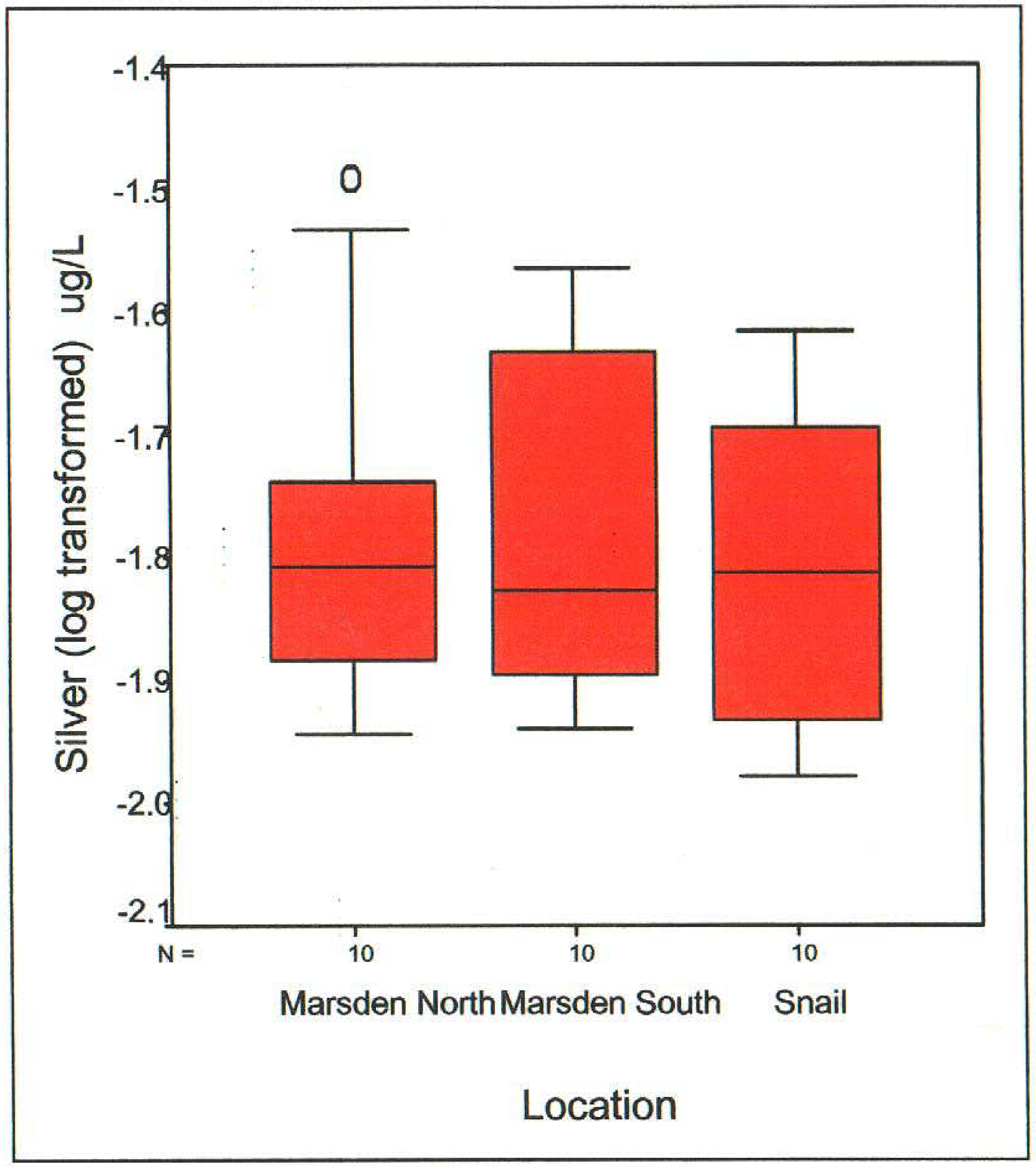


Figure B2 - 15. Thallium surface water data distributions

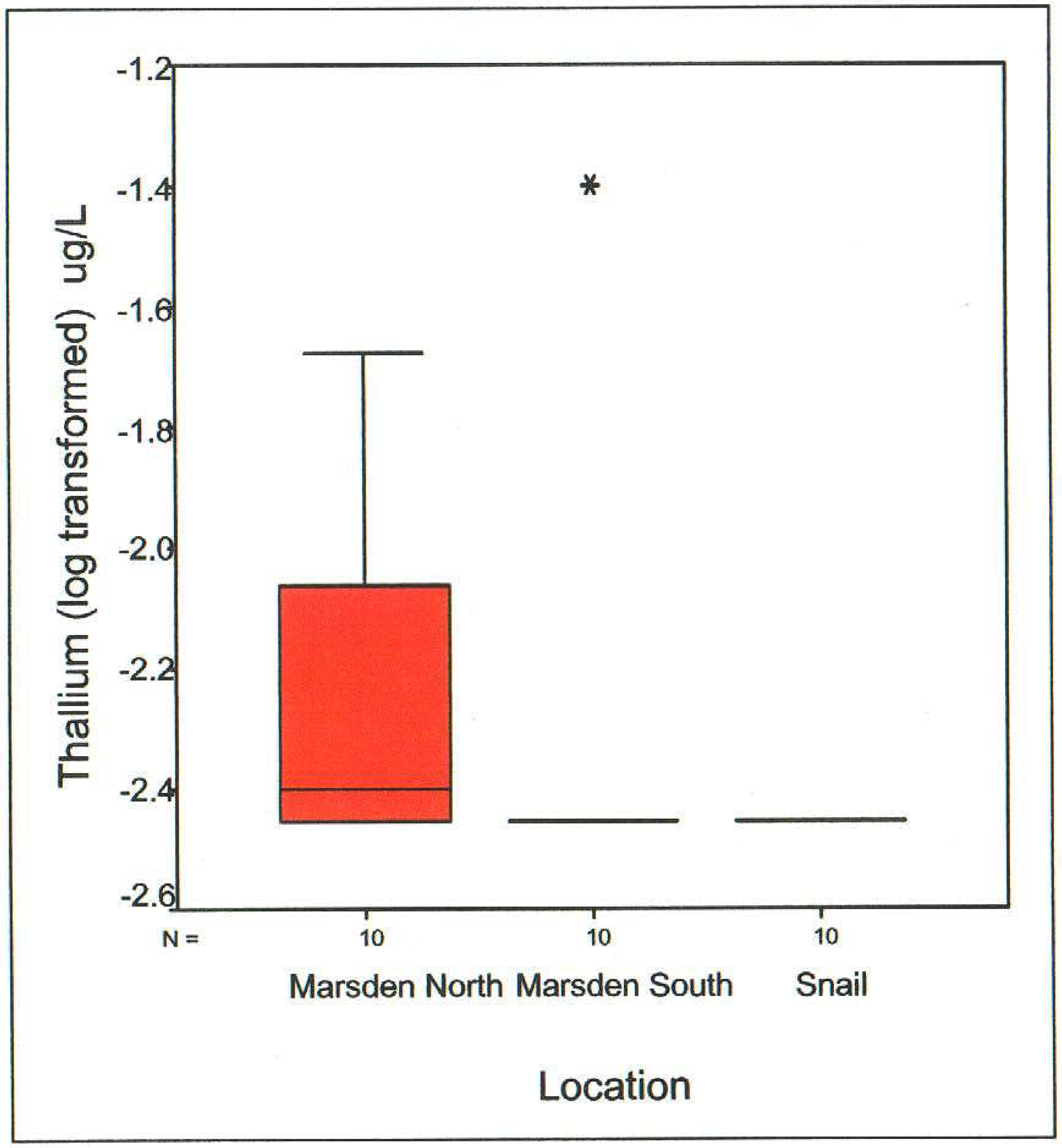


Figure B2 - 16. Zinc surface water data distributions

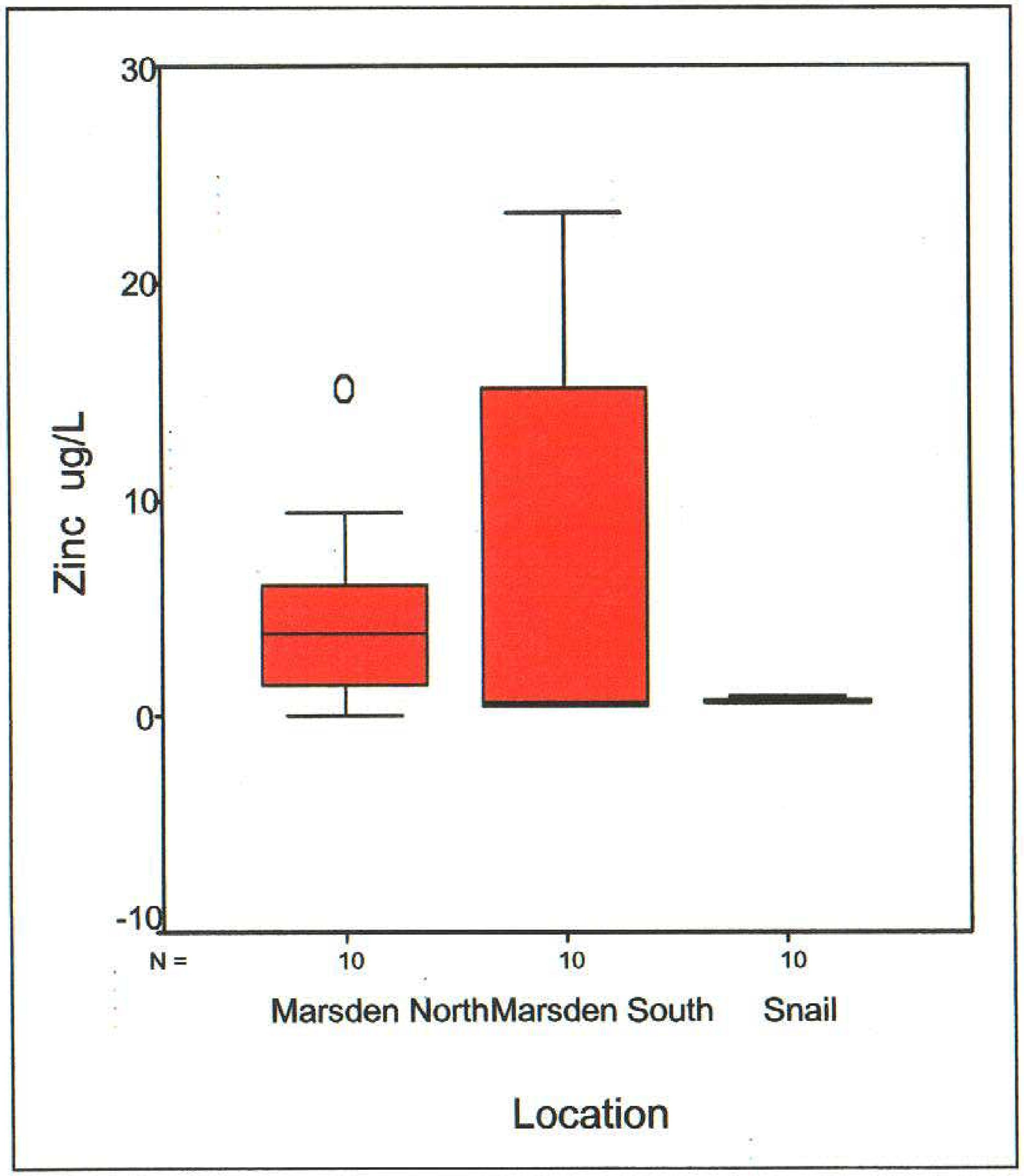
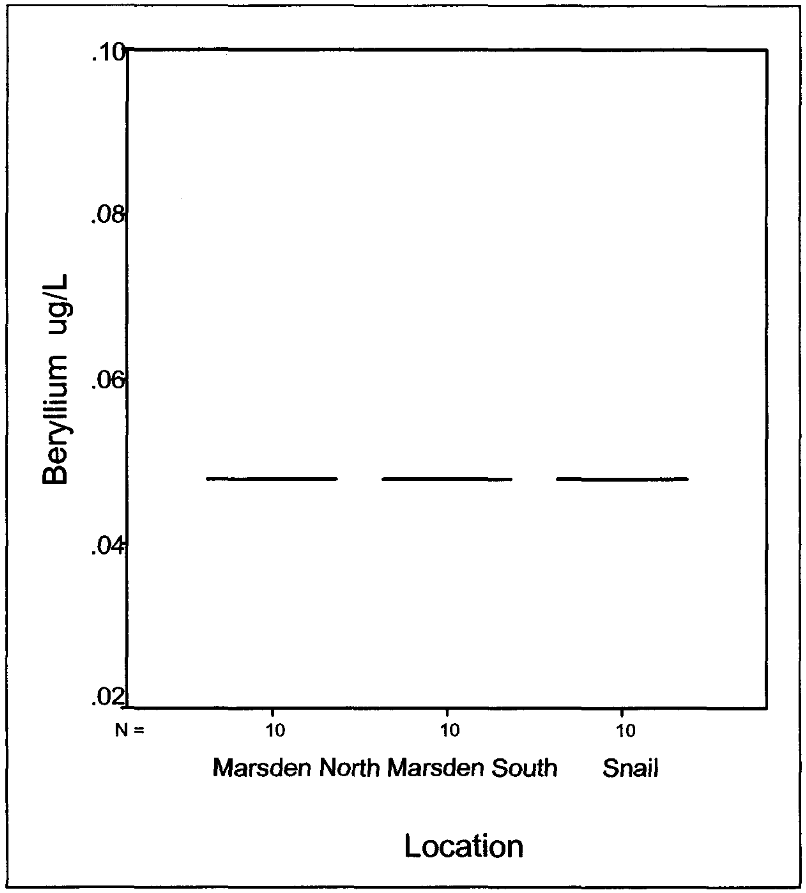


Figure B2 - 17. Beryllium surface water data distributions



FigureB2 - 18. Selenium surface water data distributions

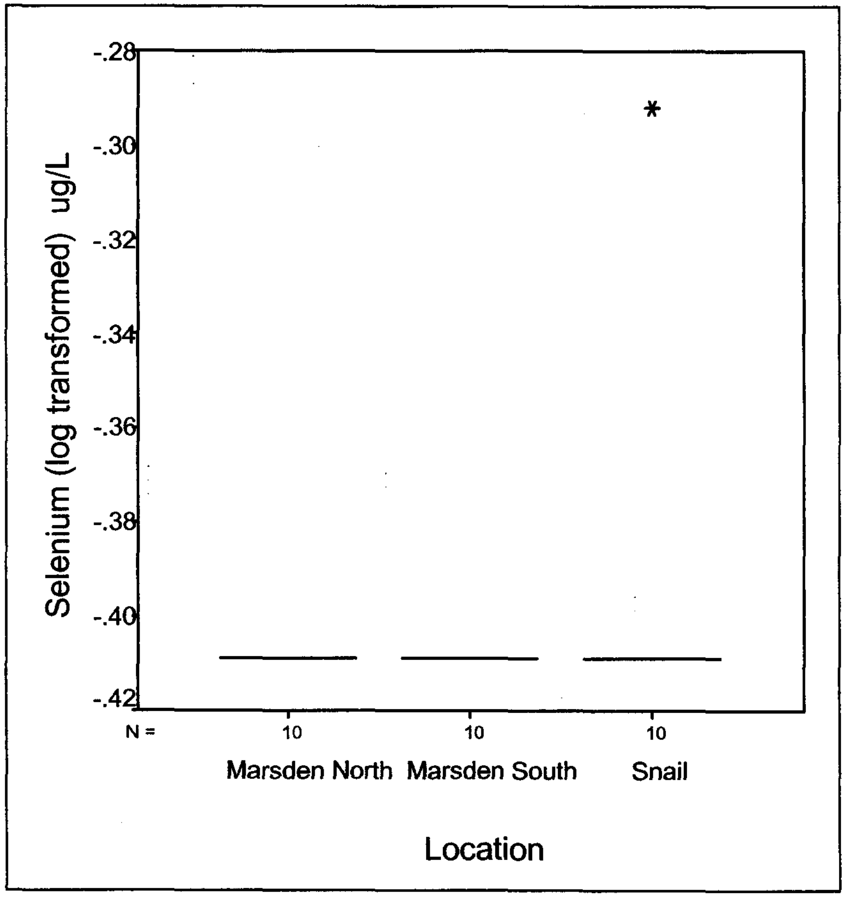


Figure B2 - 19. Aluminum Sediment Data Distribution

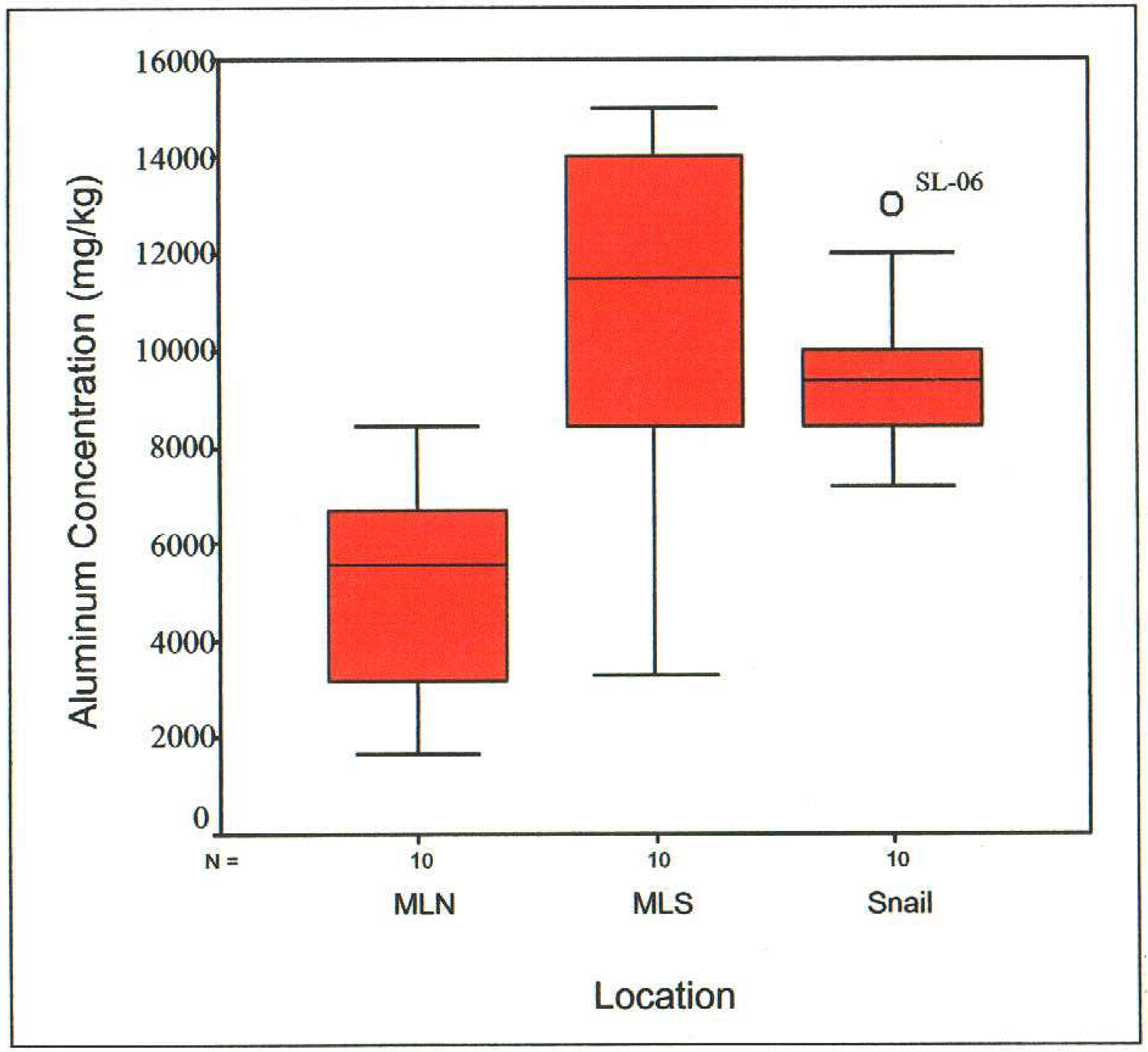


Figure B2 - 20Antimony Sediment Data Distribution

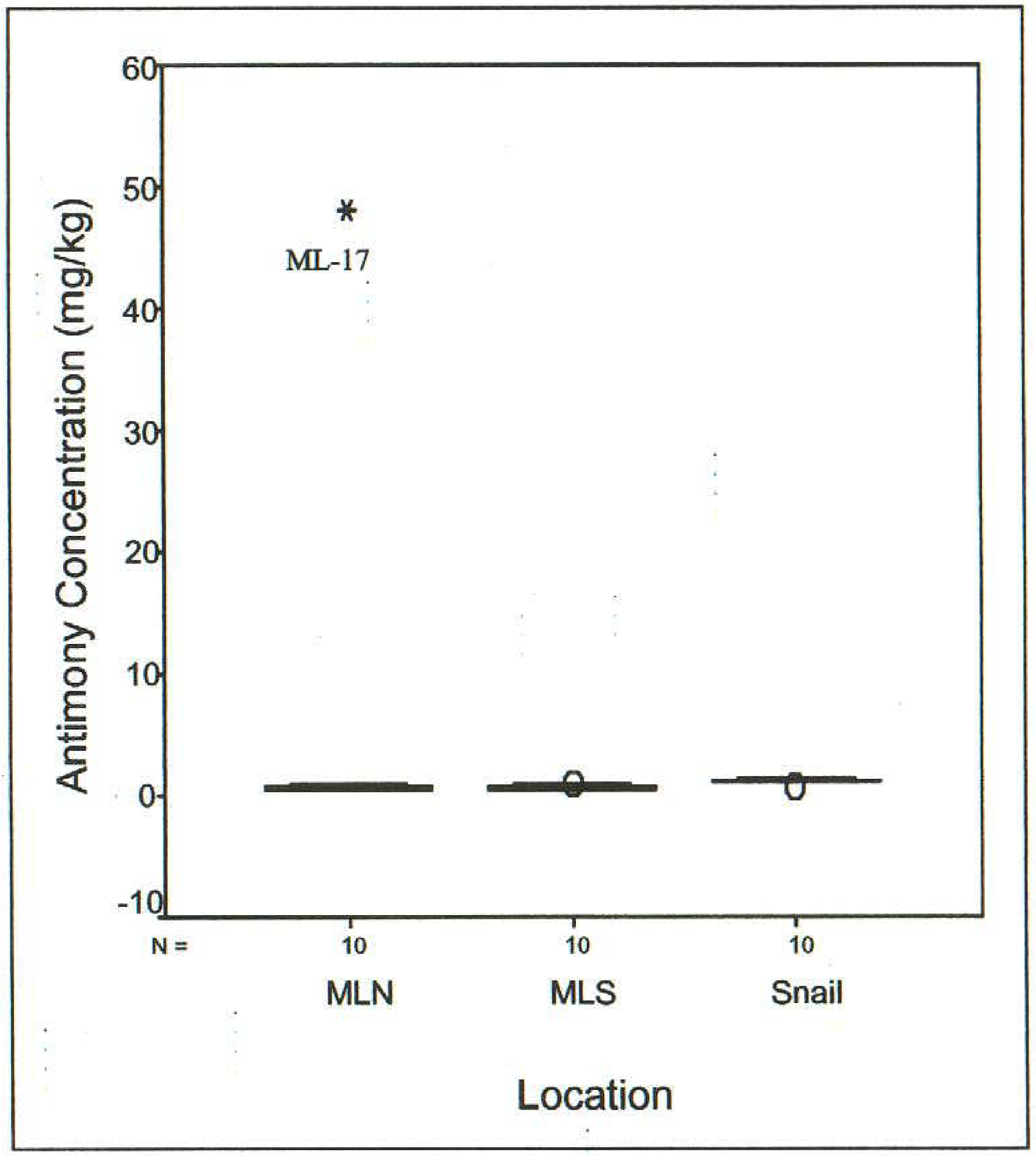


Figure B2 - 21. Sediment Arsenic Data Distribution

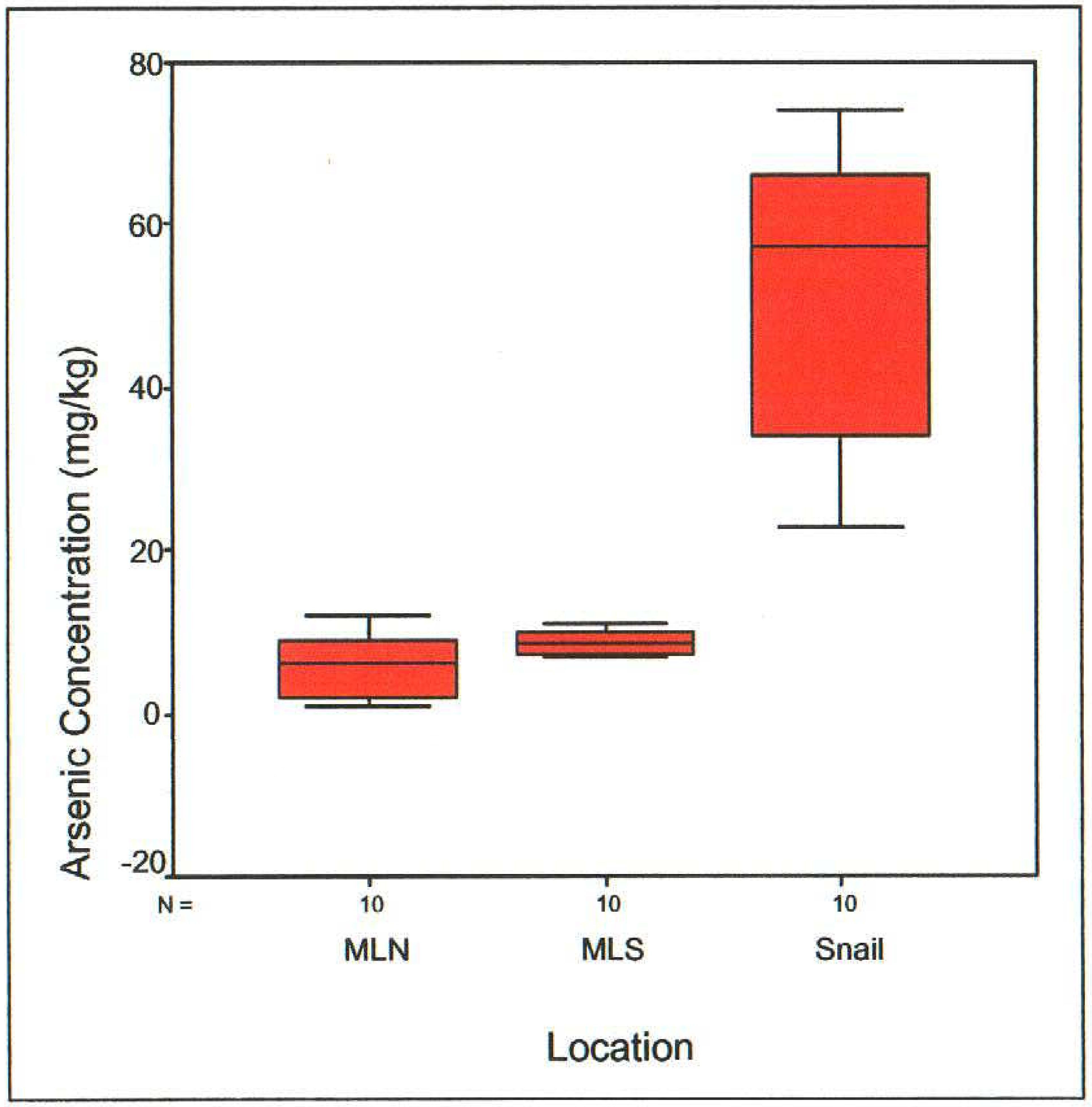


Figure B2 - 22. Sediment Barium Data Distribution

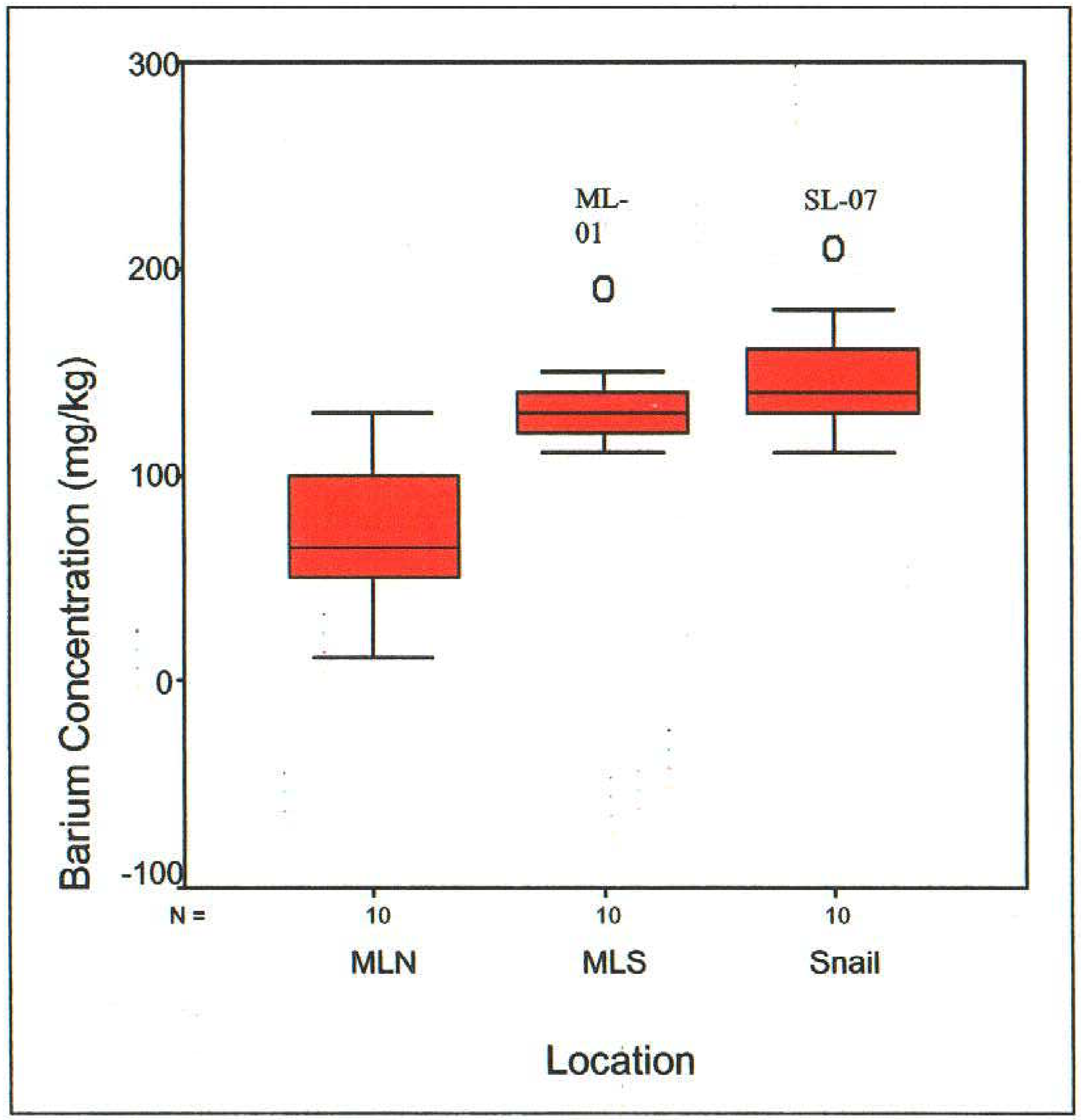


Figure B2 - 23. Sediment Beryllium Data Distribution

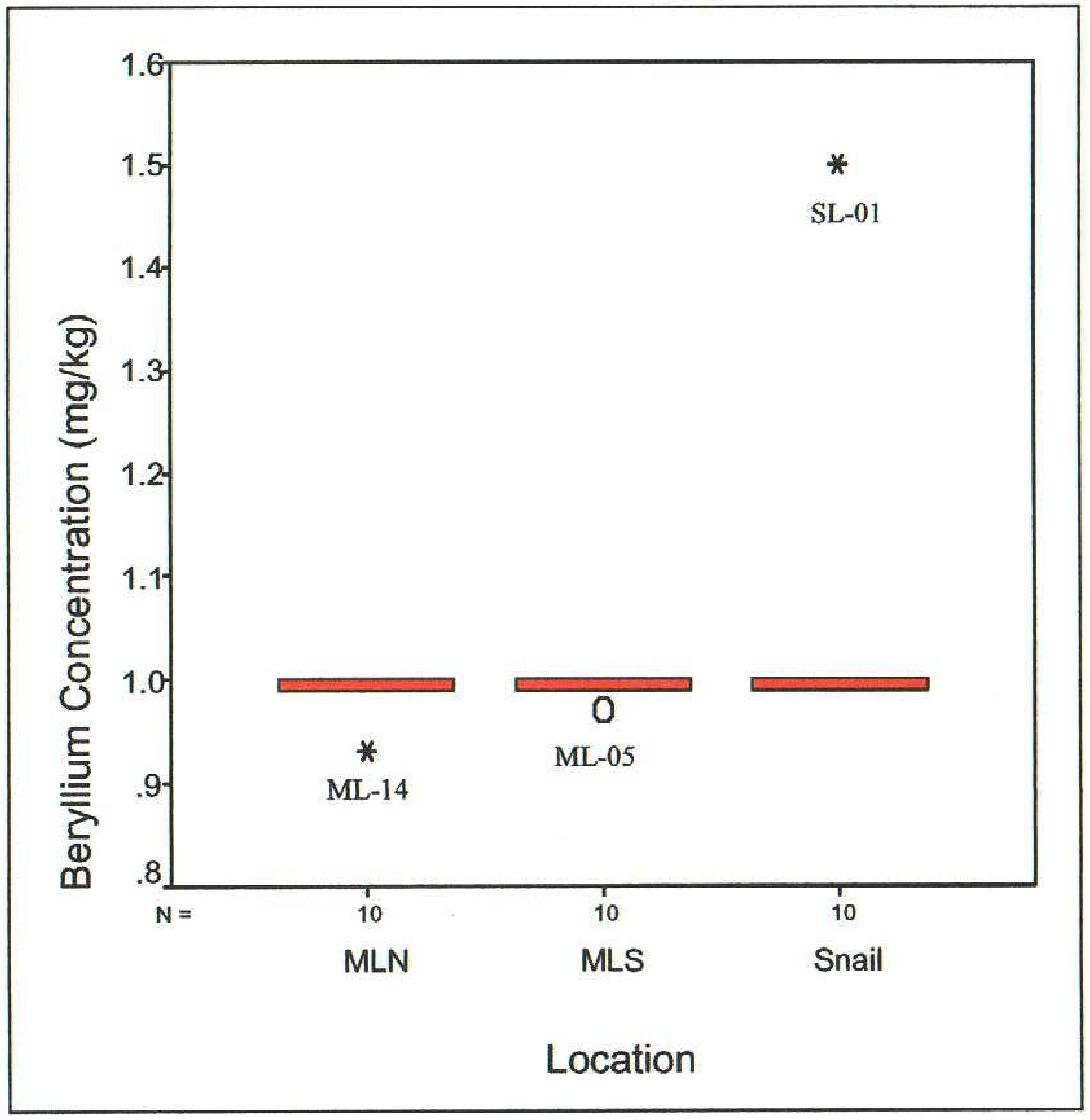


Figure B2 - 24. Sediment Cadmium Data Distribution

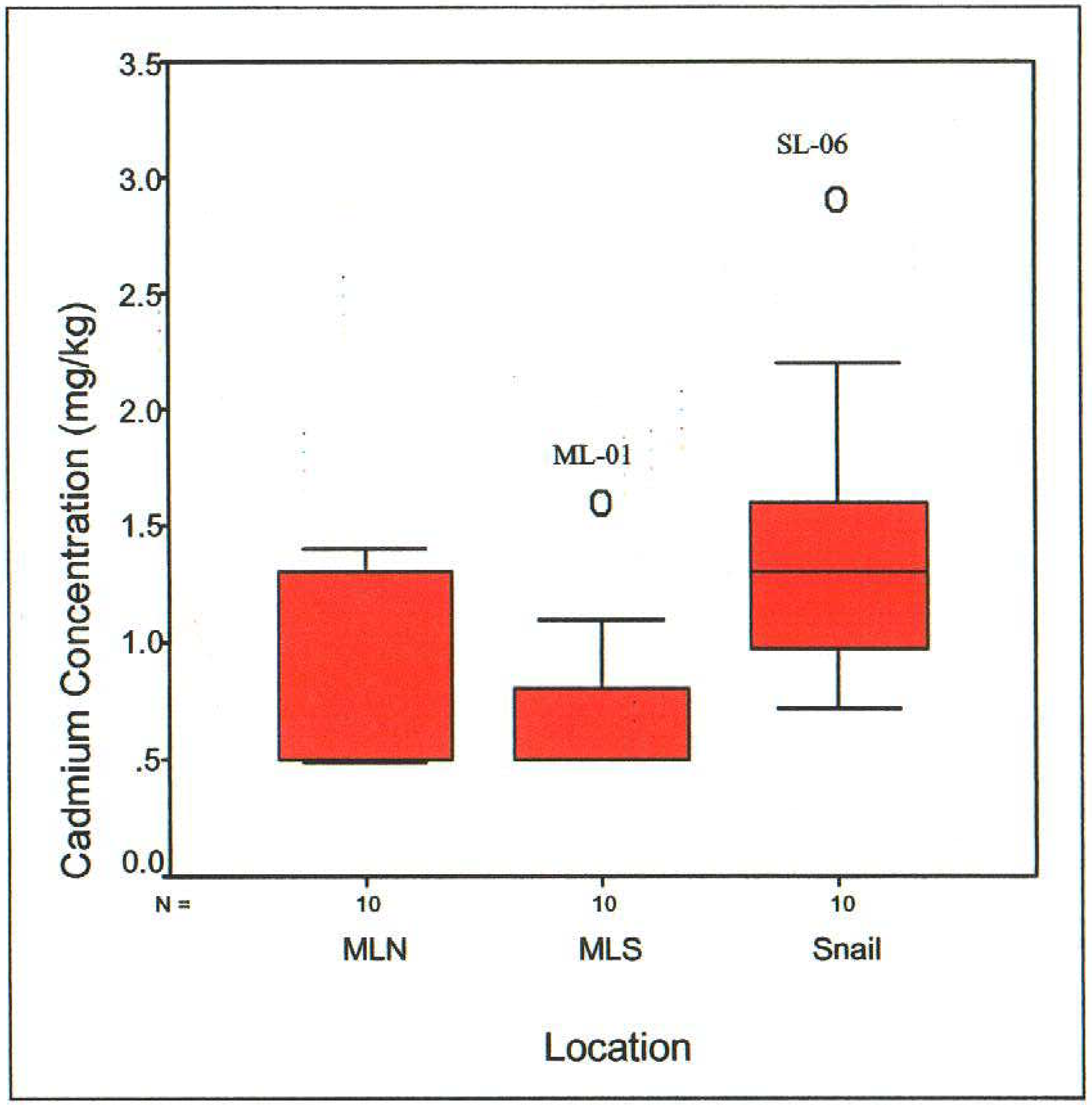


Figure B2 - 25. Sediment Chromium Data Distribution

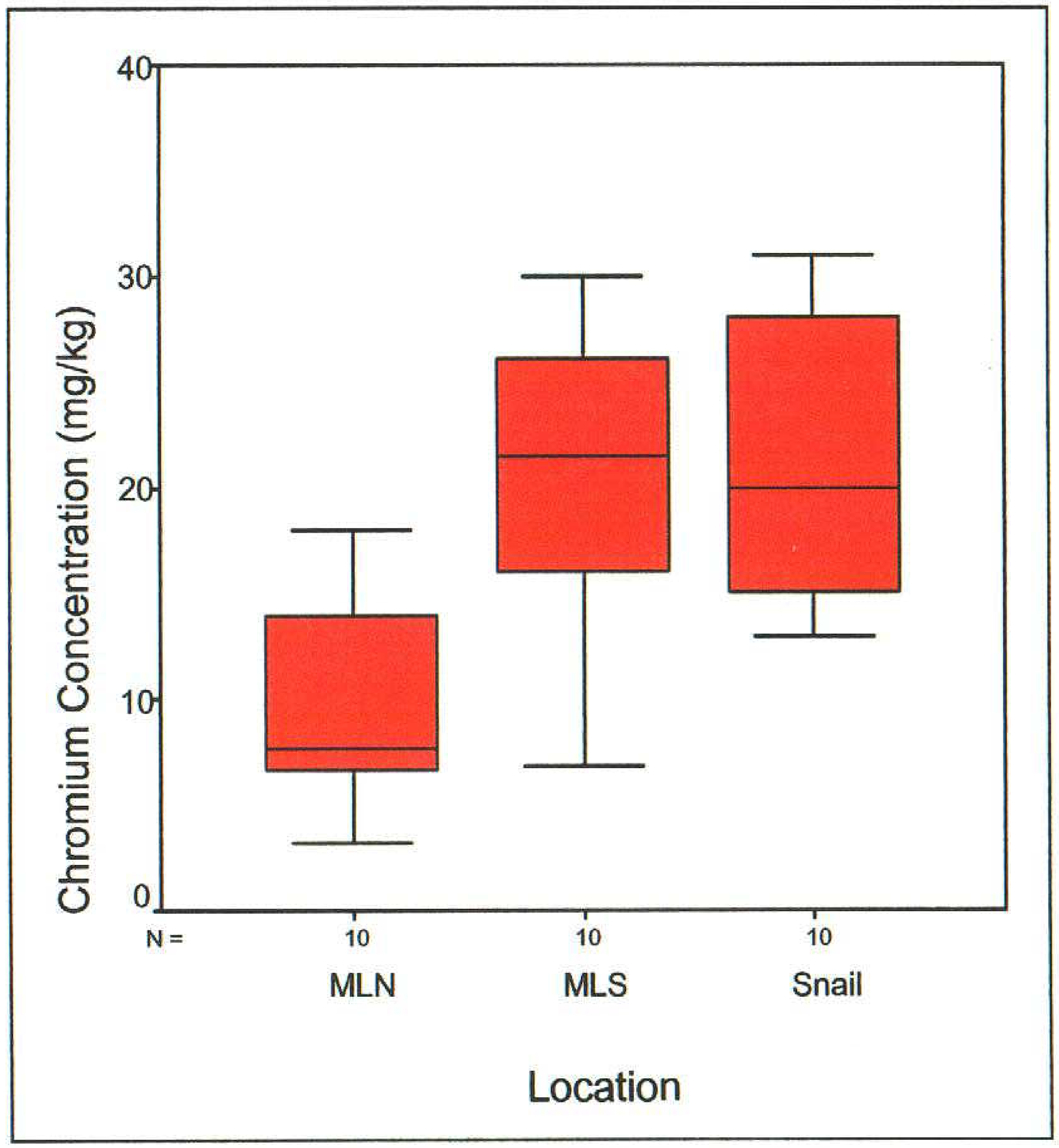


Figure B2 - 26. Sediment Cobalt Data Distribution

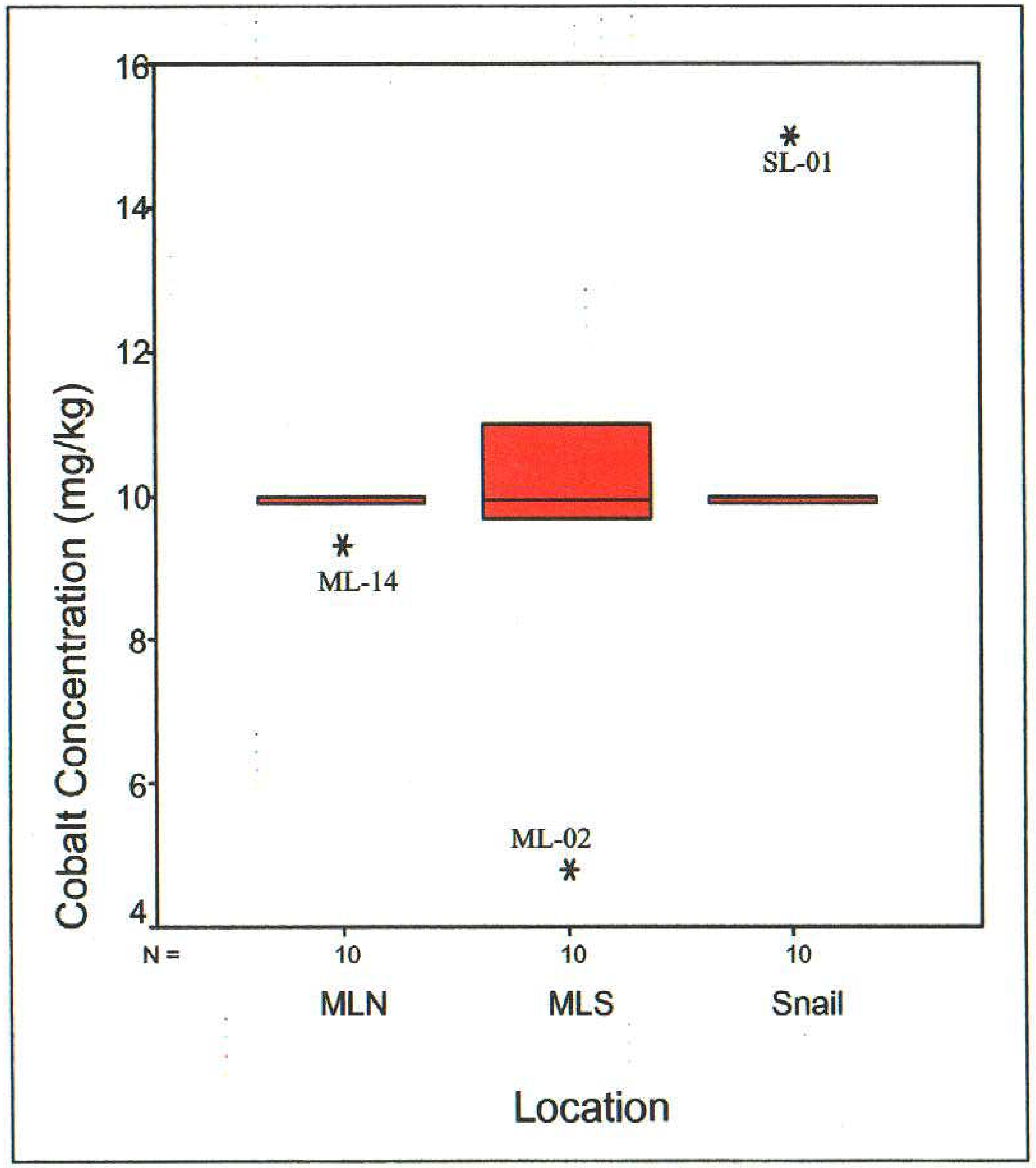
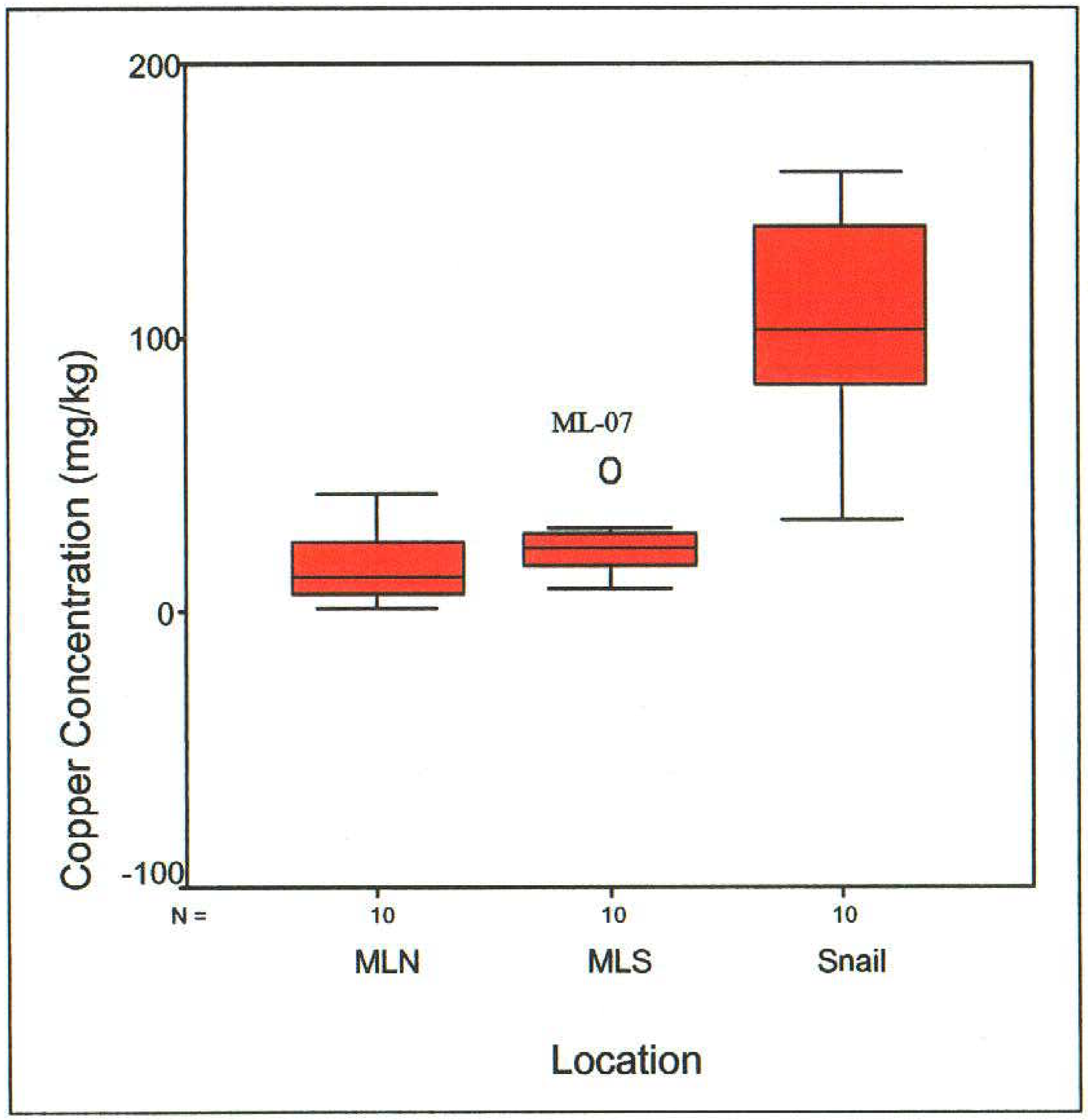


Figure B2 - 27. Sediment Copper Data Distribution



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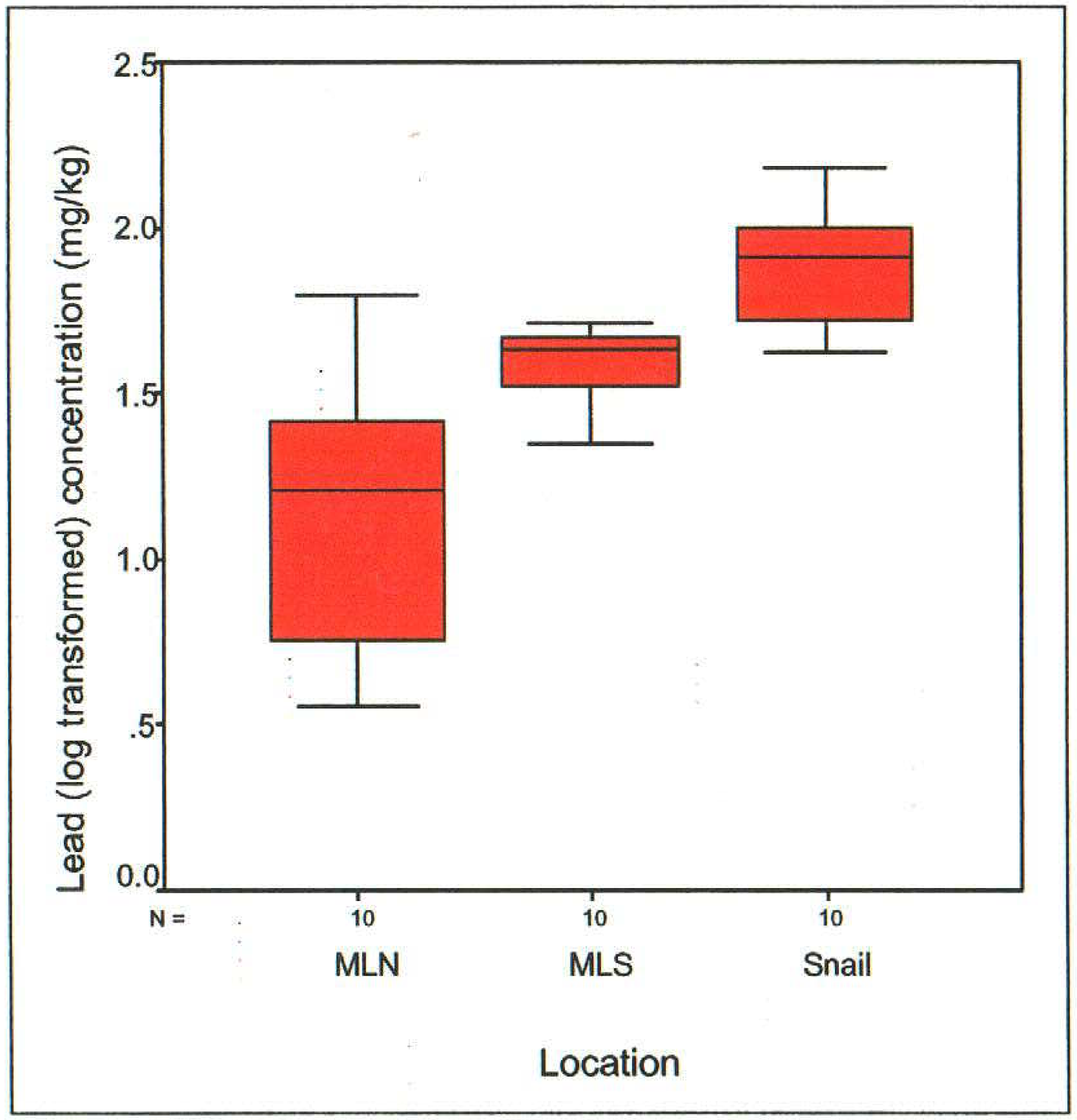

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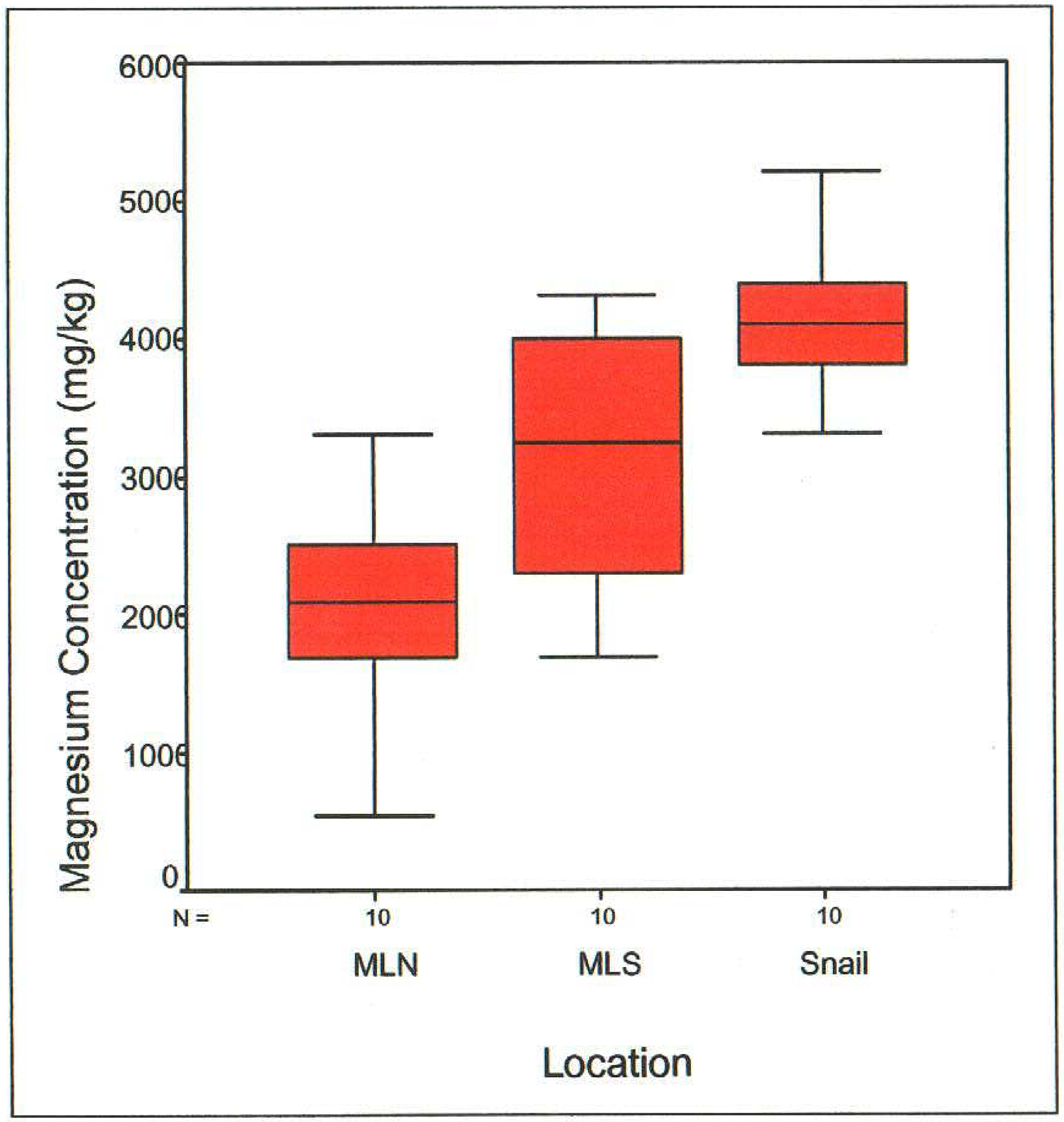

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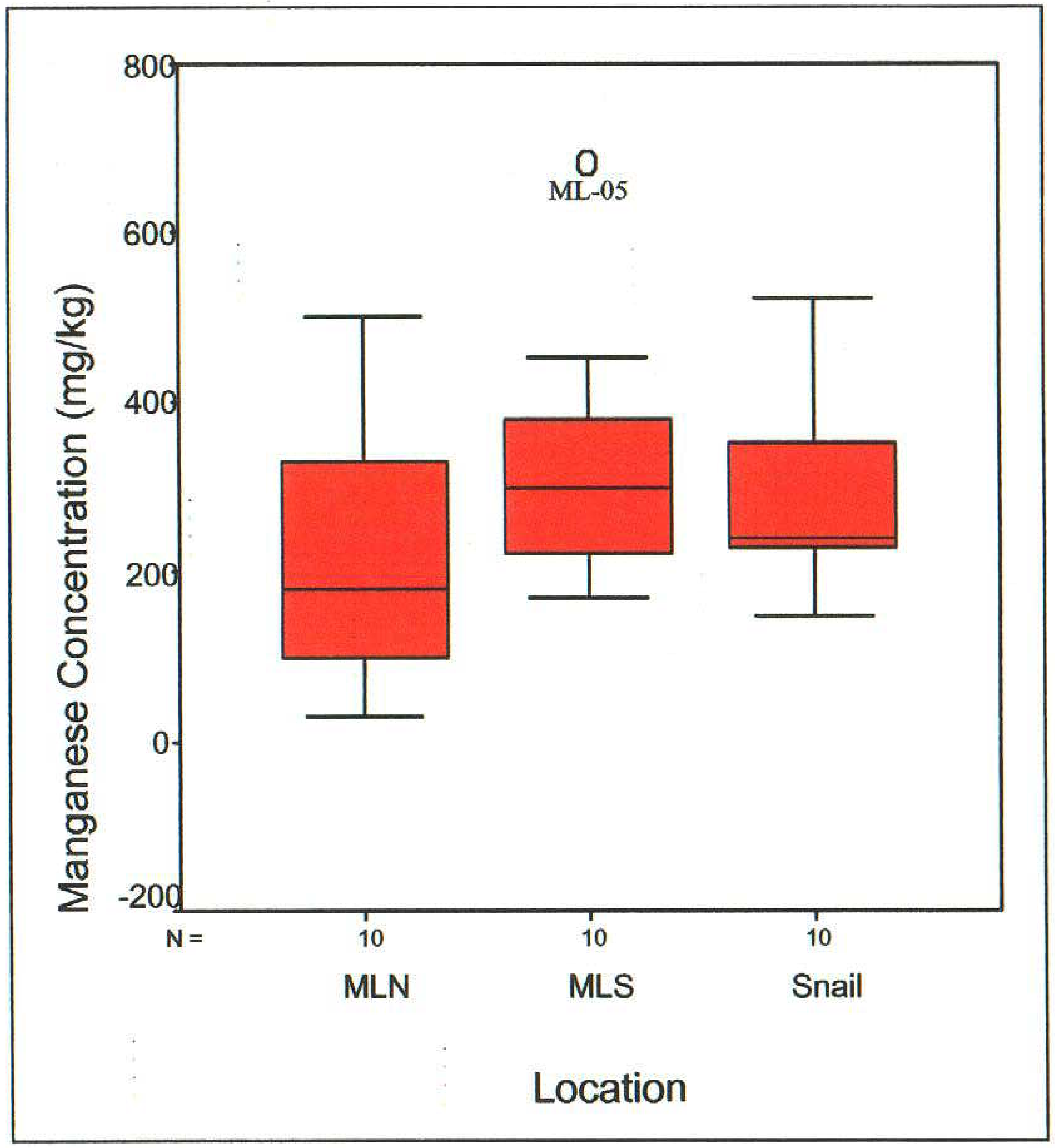

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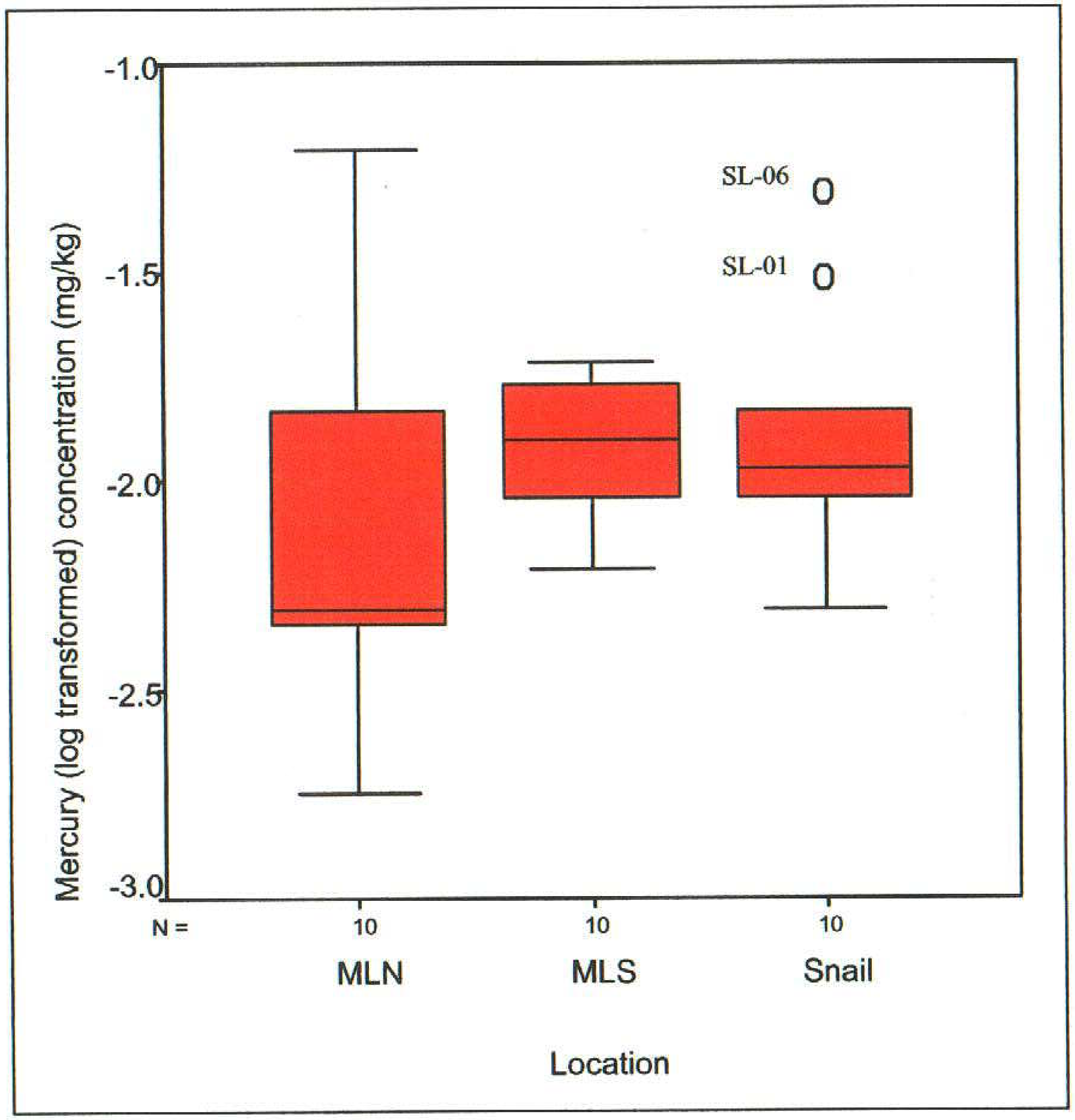


Figure B2 - 32. Sediment Nickel Data Distribution

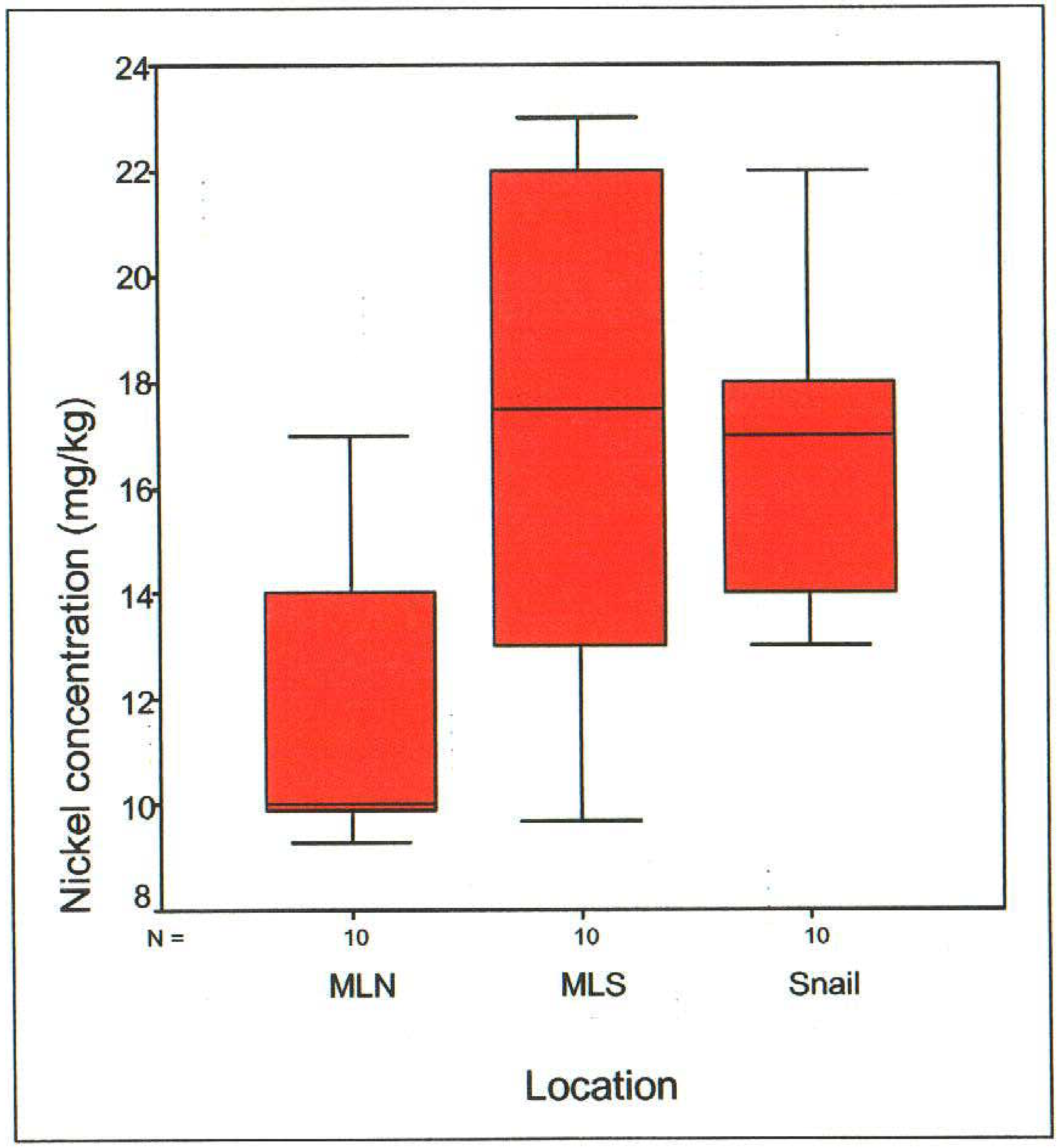


Figure B2 - 33. Sediment Selenium Data Distribution

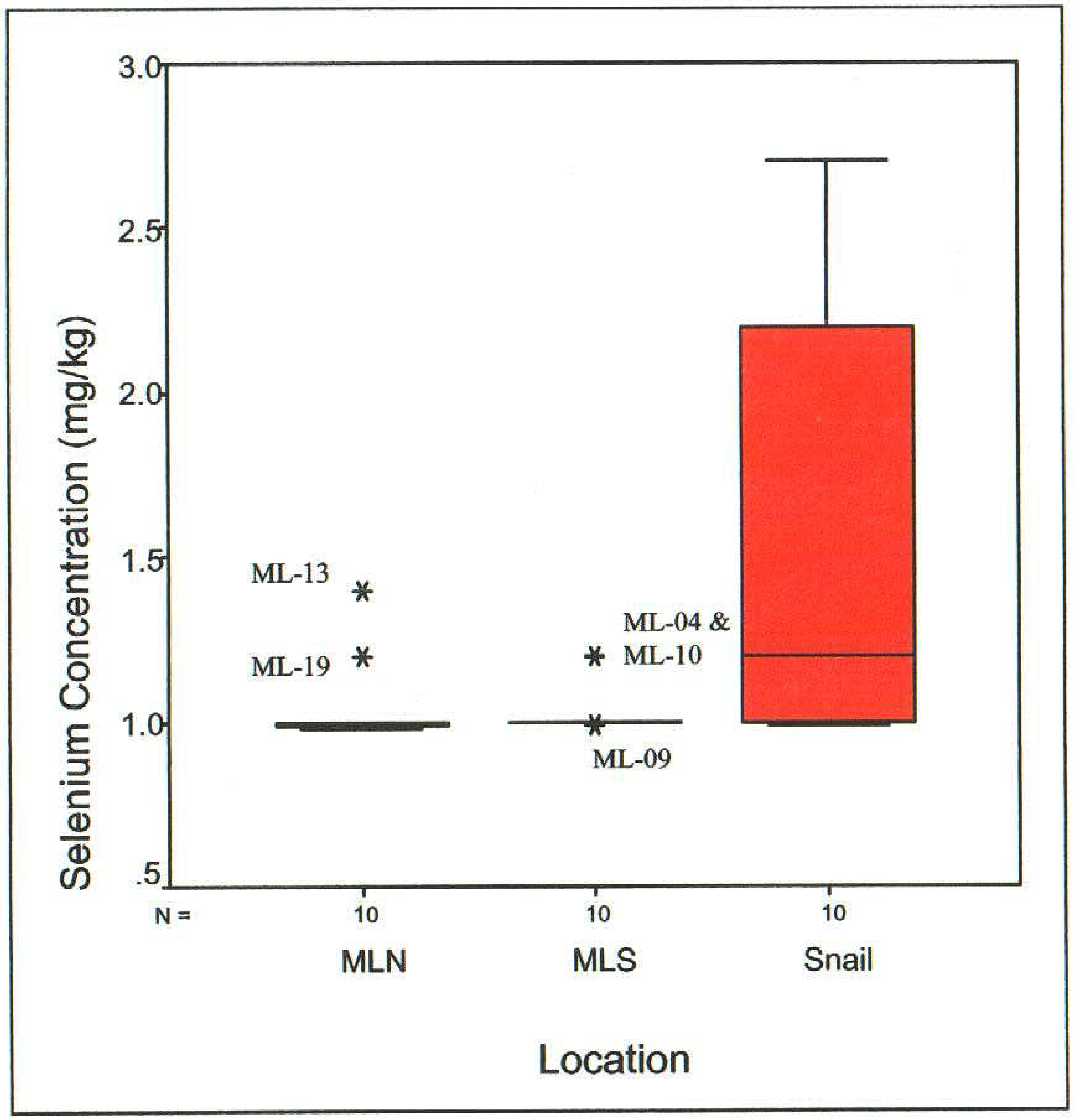


Figure B2 - 34. Sediment Silver Data Distribution

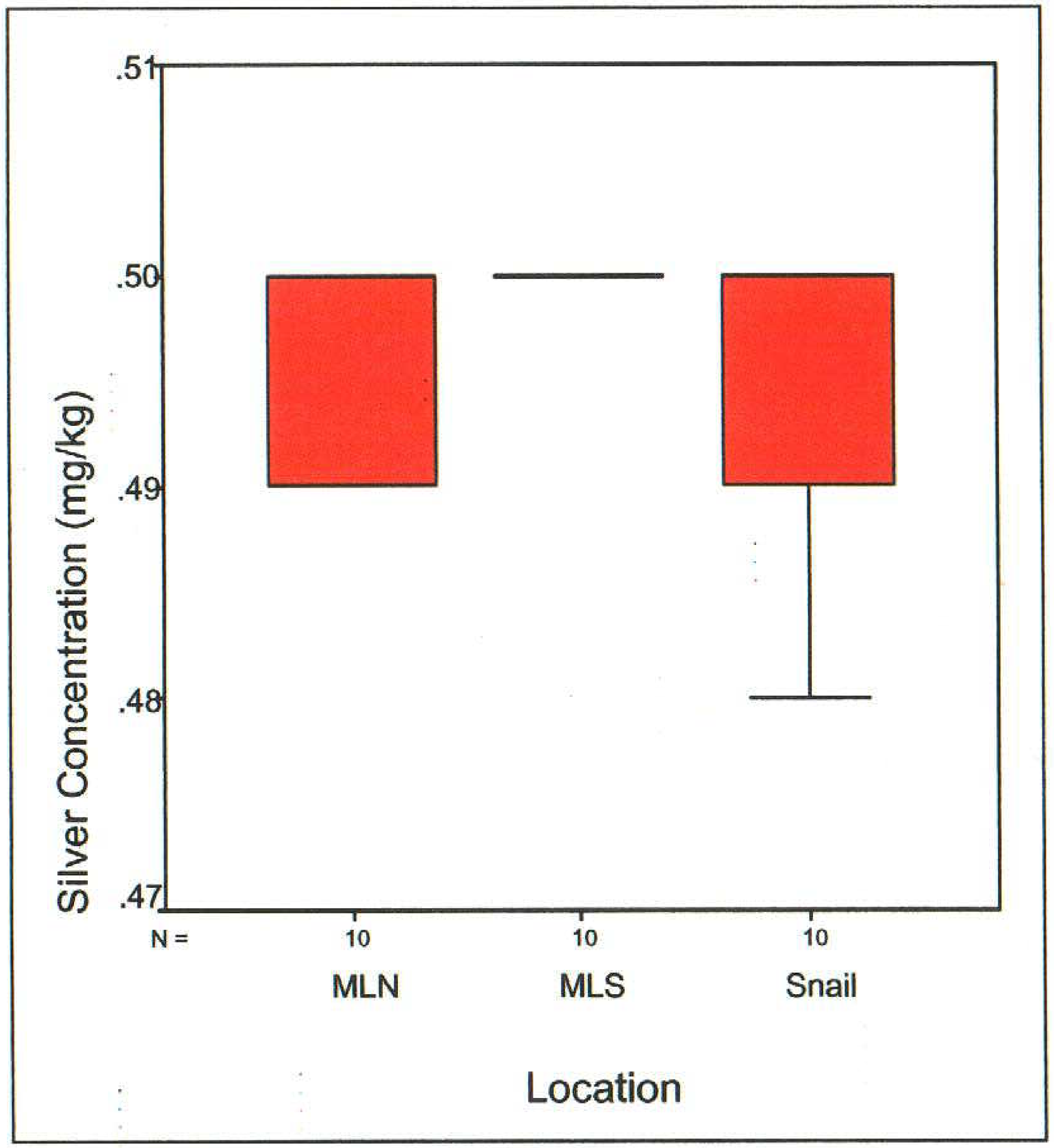


Figure B2 - 35. Sediment Thallium Data Distribution

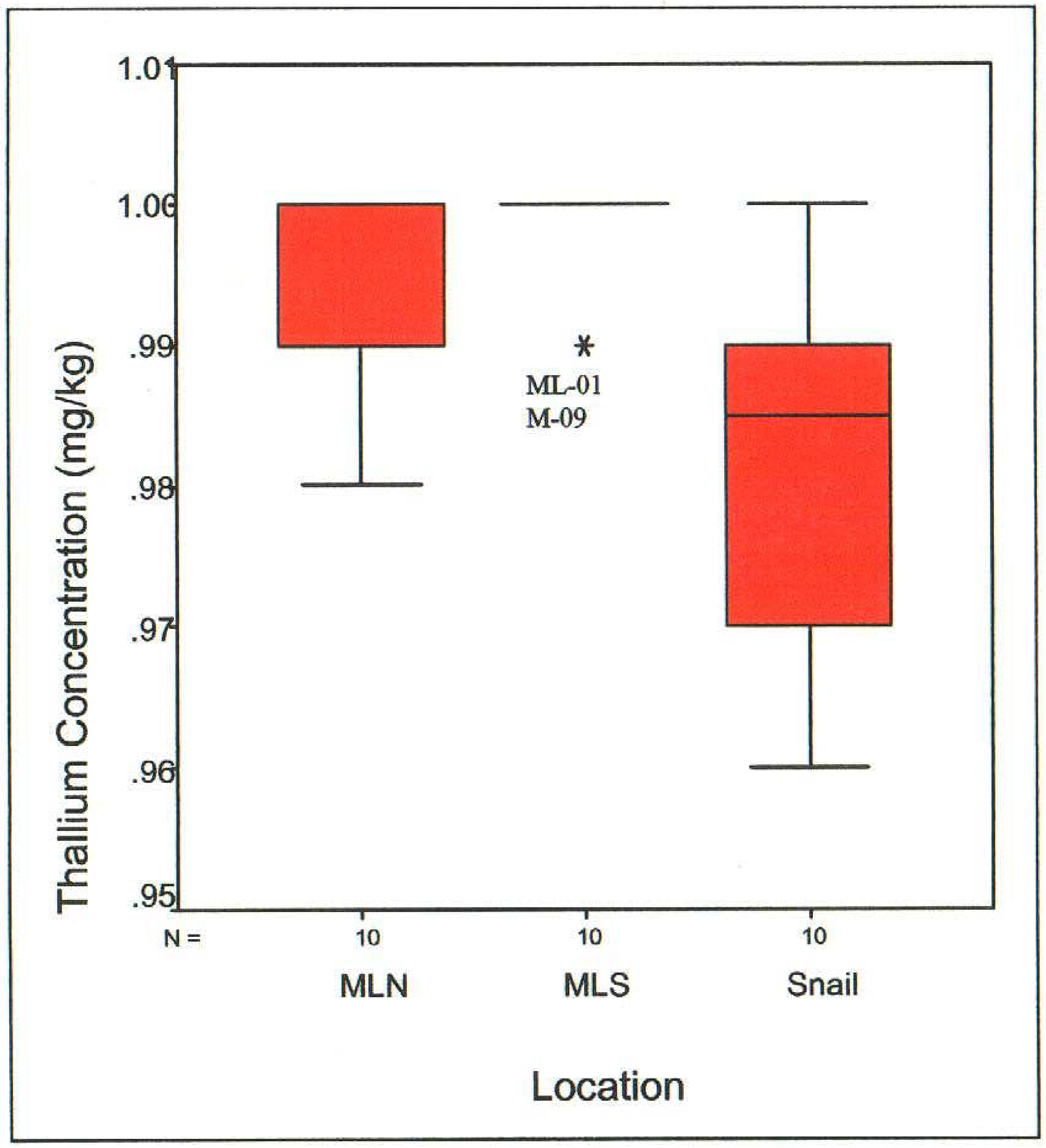
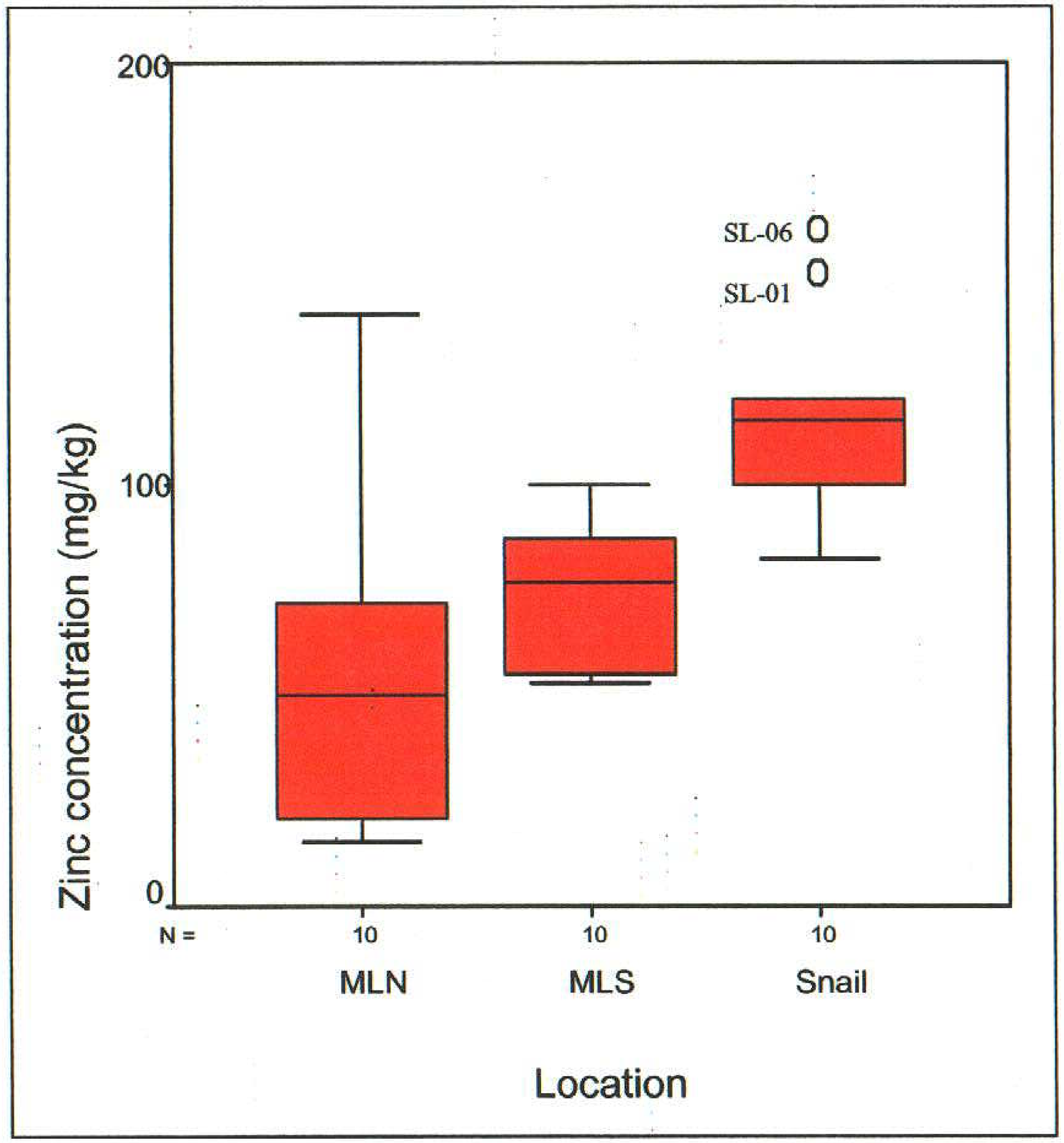


Figure B2 - 36. Sediment Zinc Data Distribution



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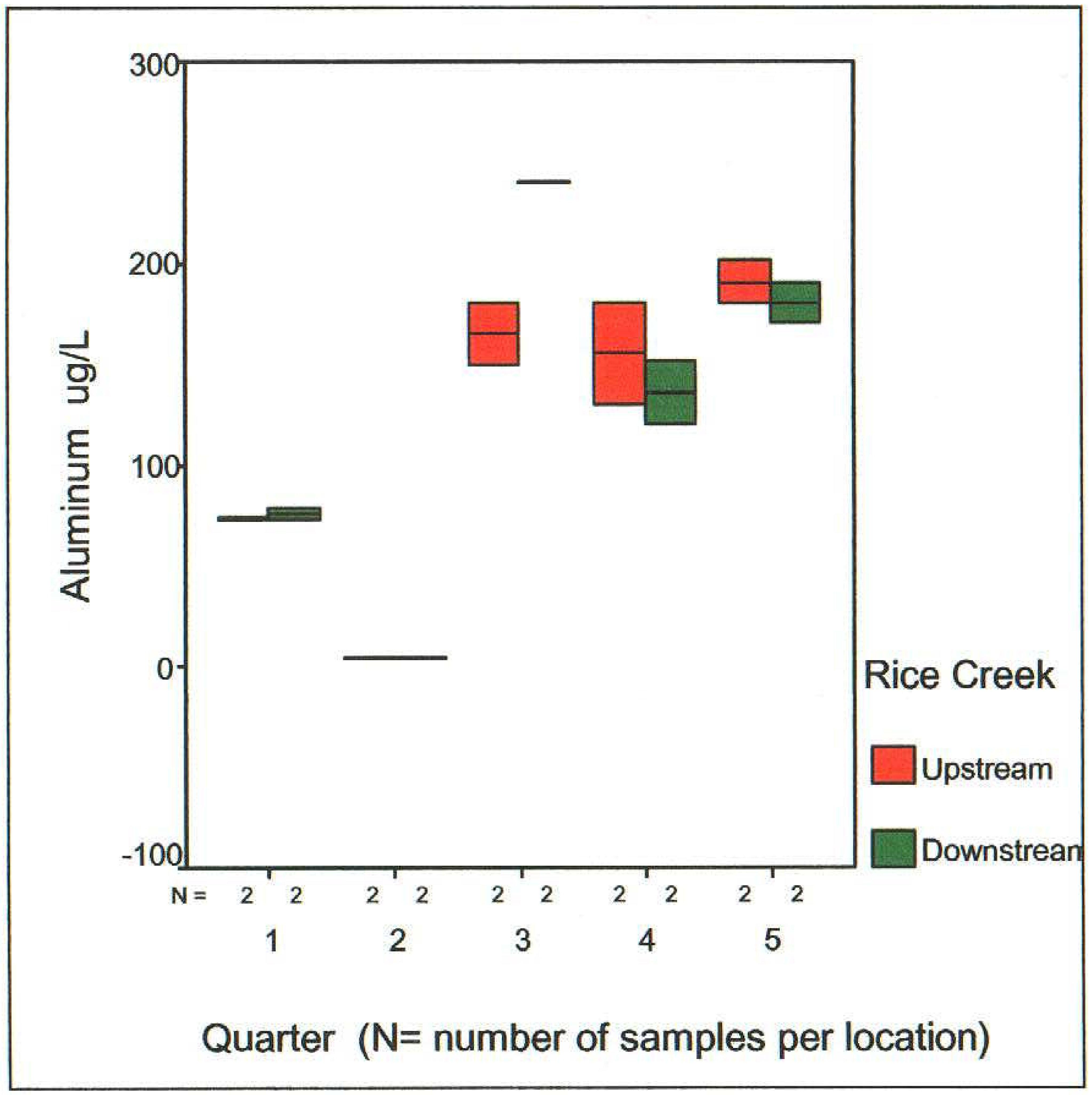

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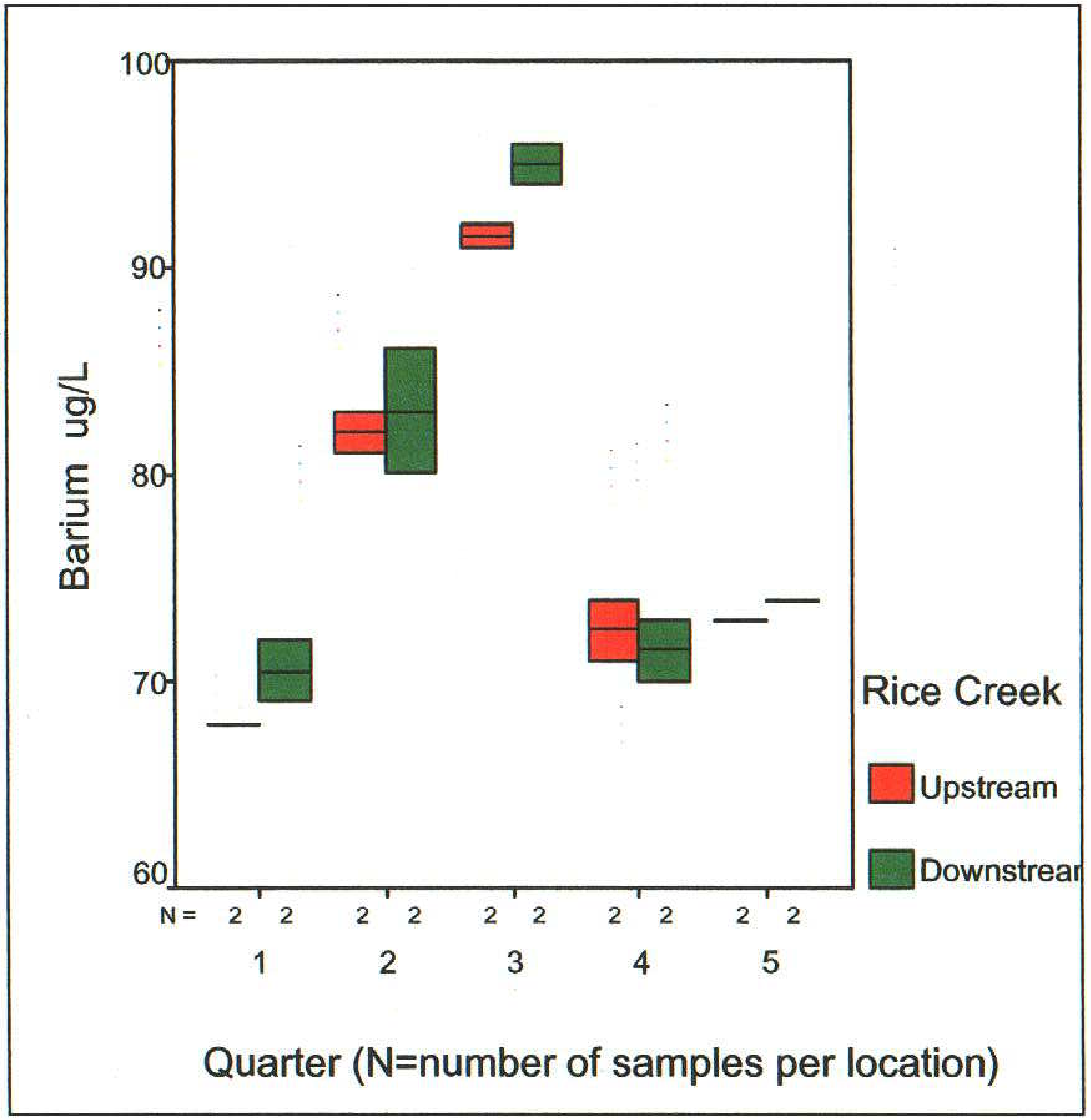

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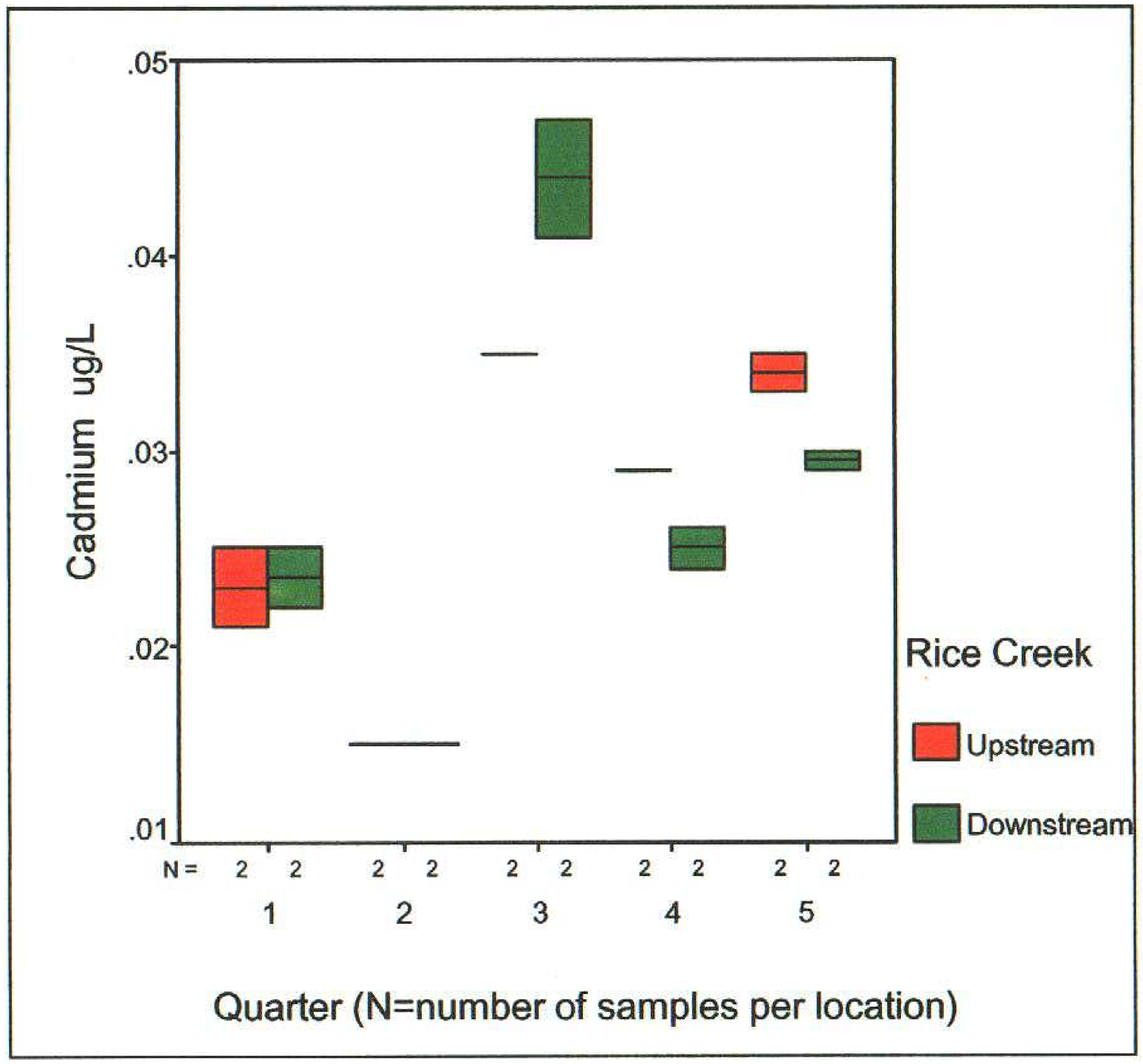

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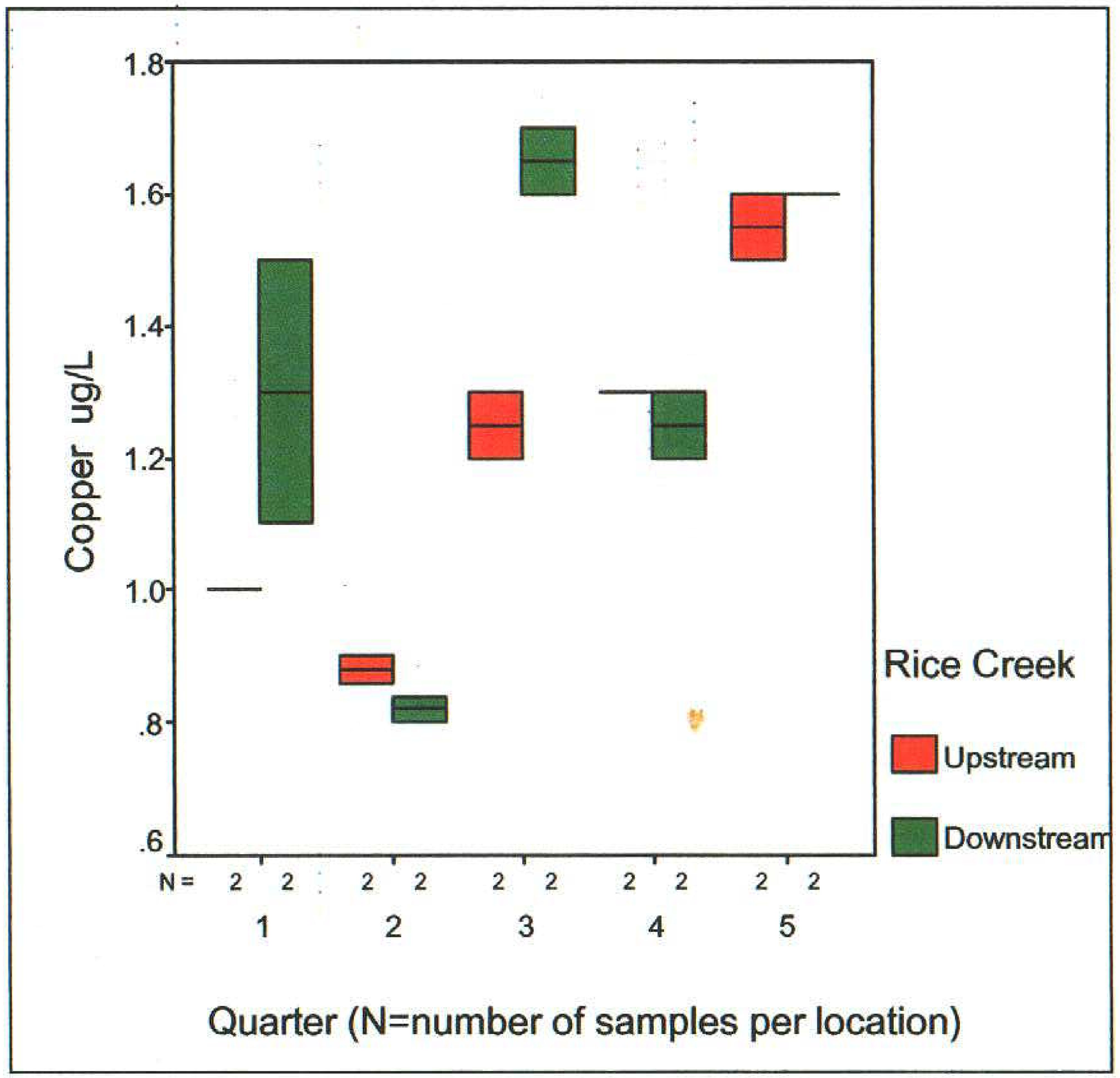


Figure B2 - 41. Rice Creek Water Hg Data Distribution

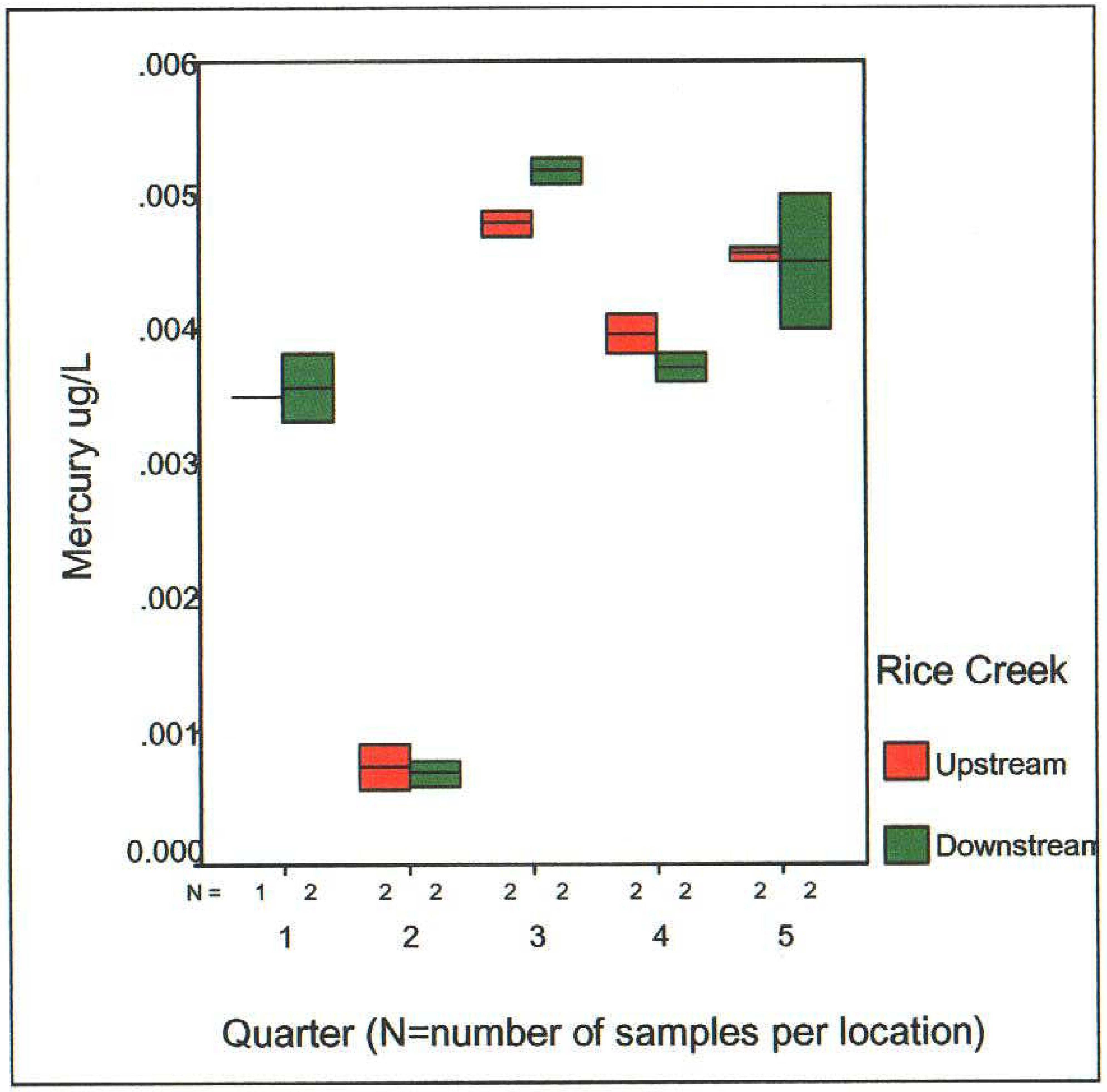
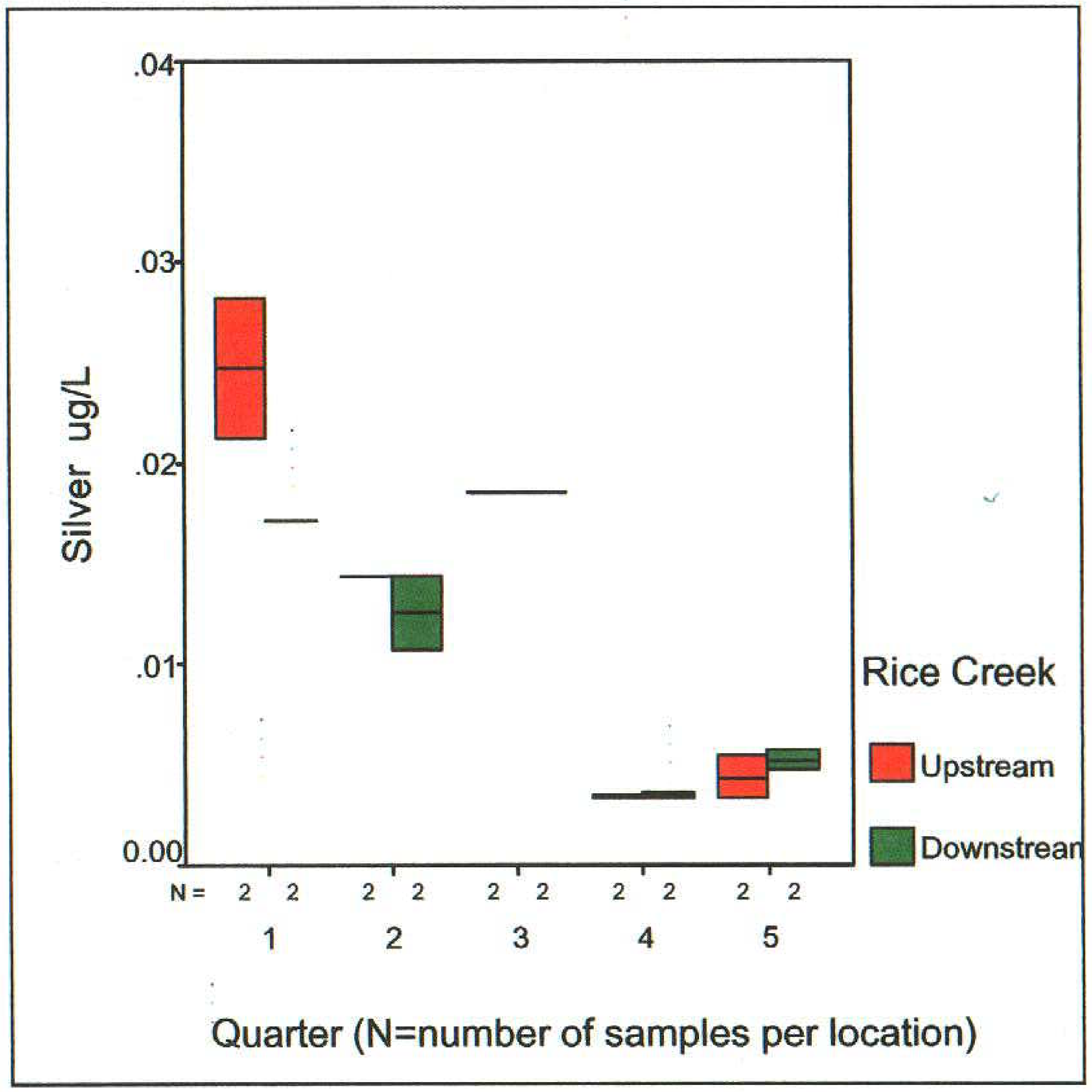


Figure B2 - 42. Rice Creek Water Ag Data Distribution



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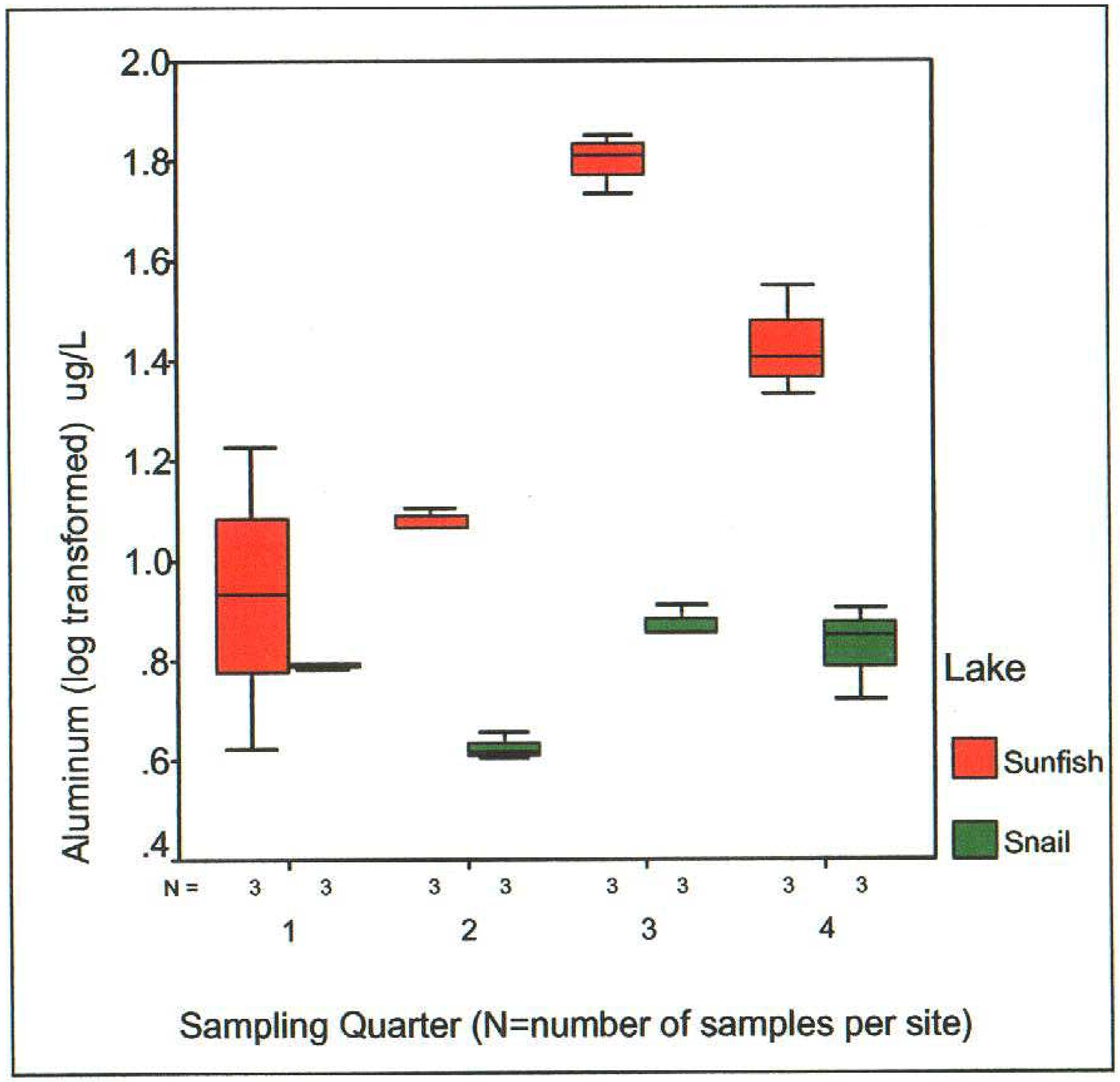

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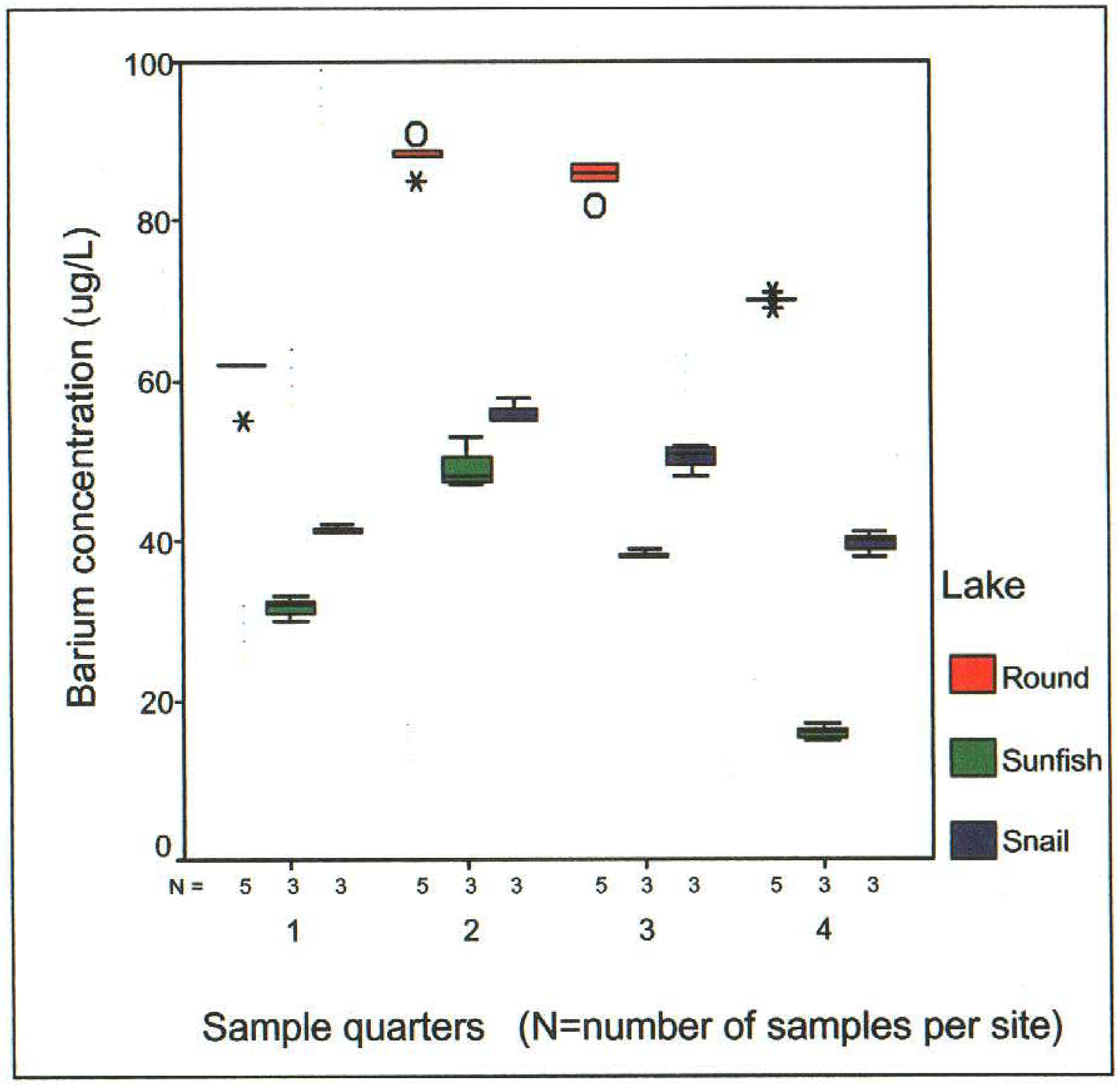

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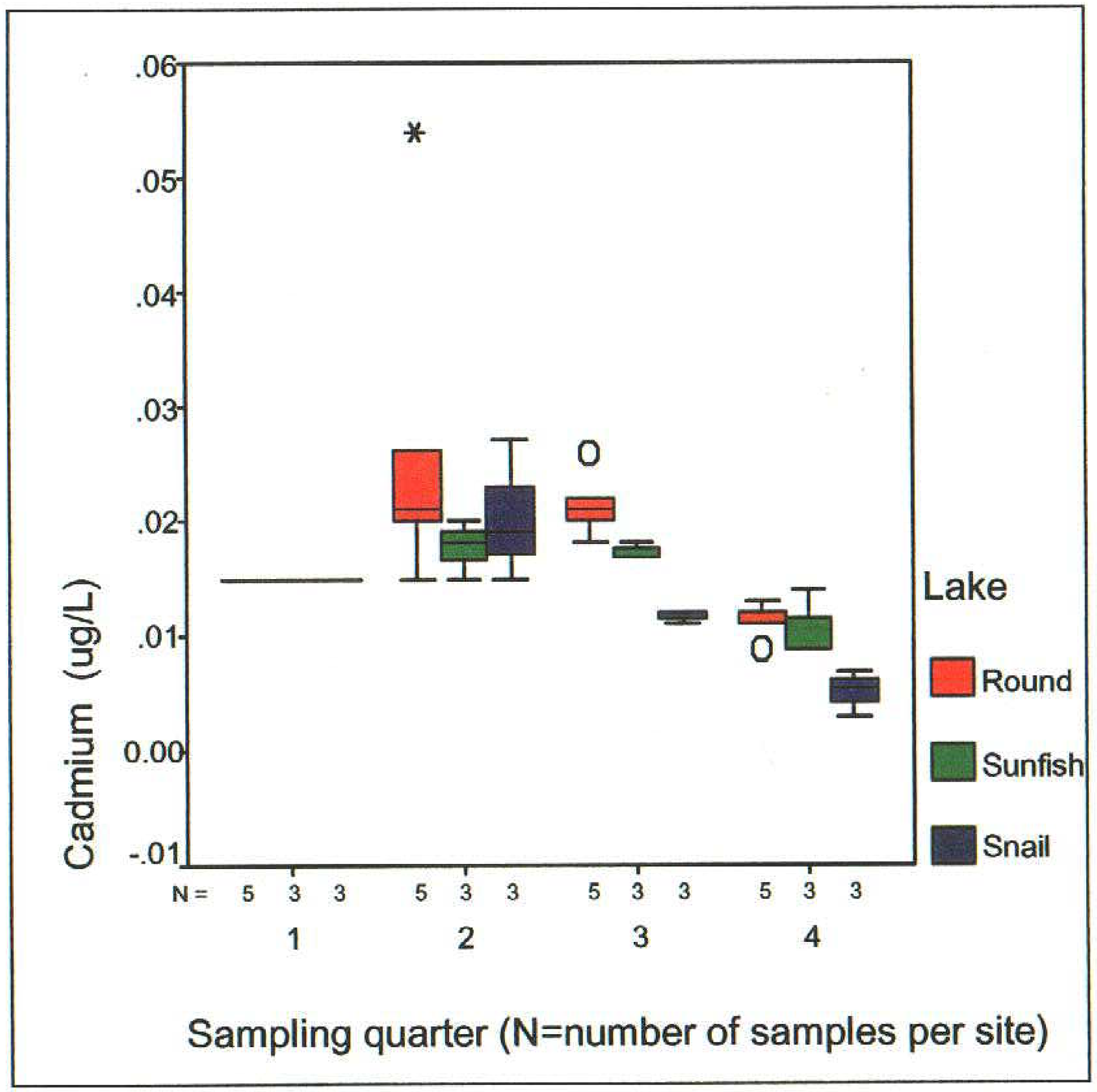

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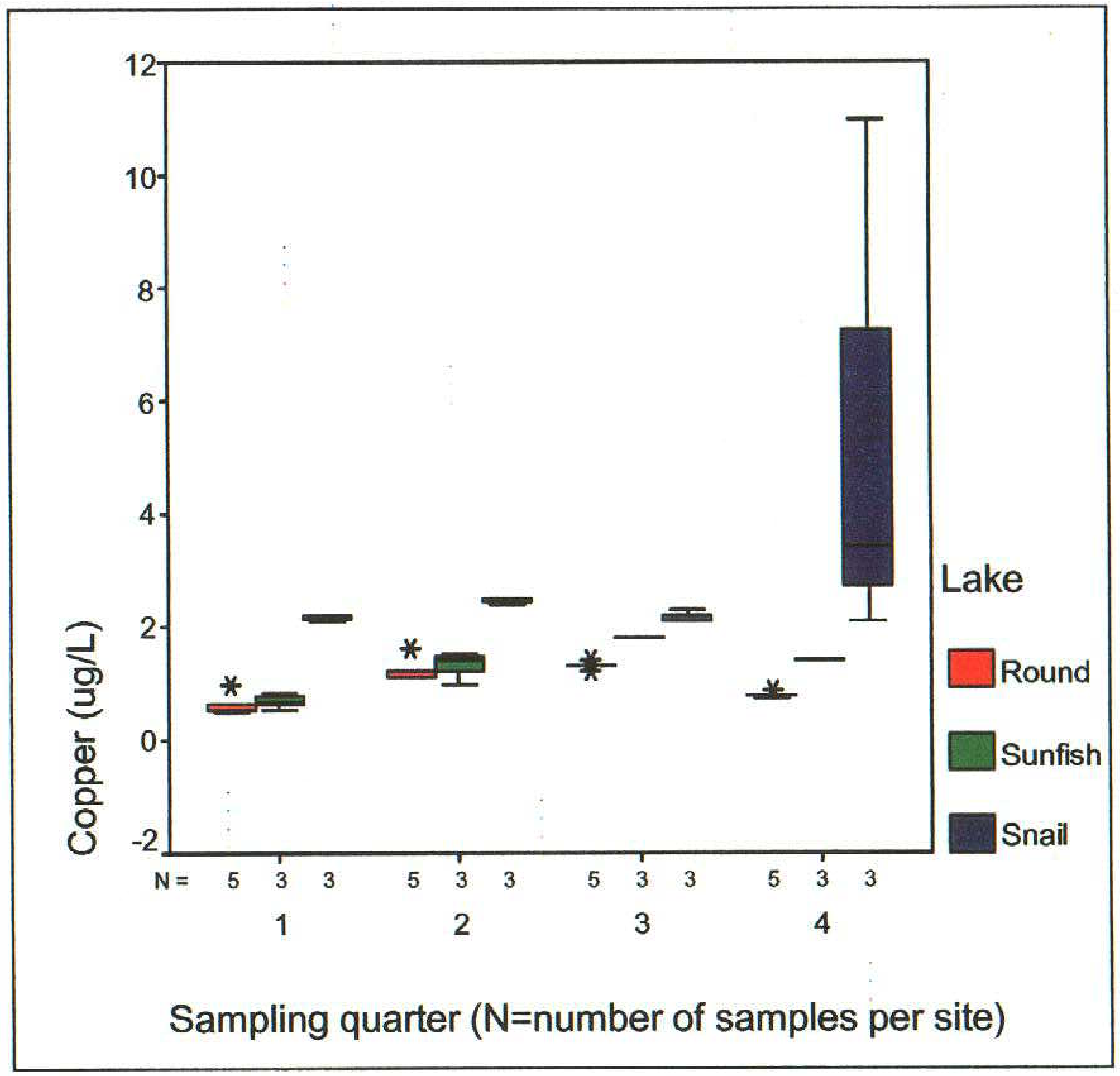

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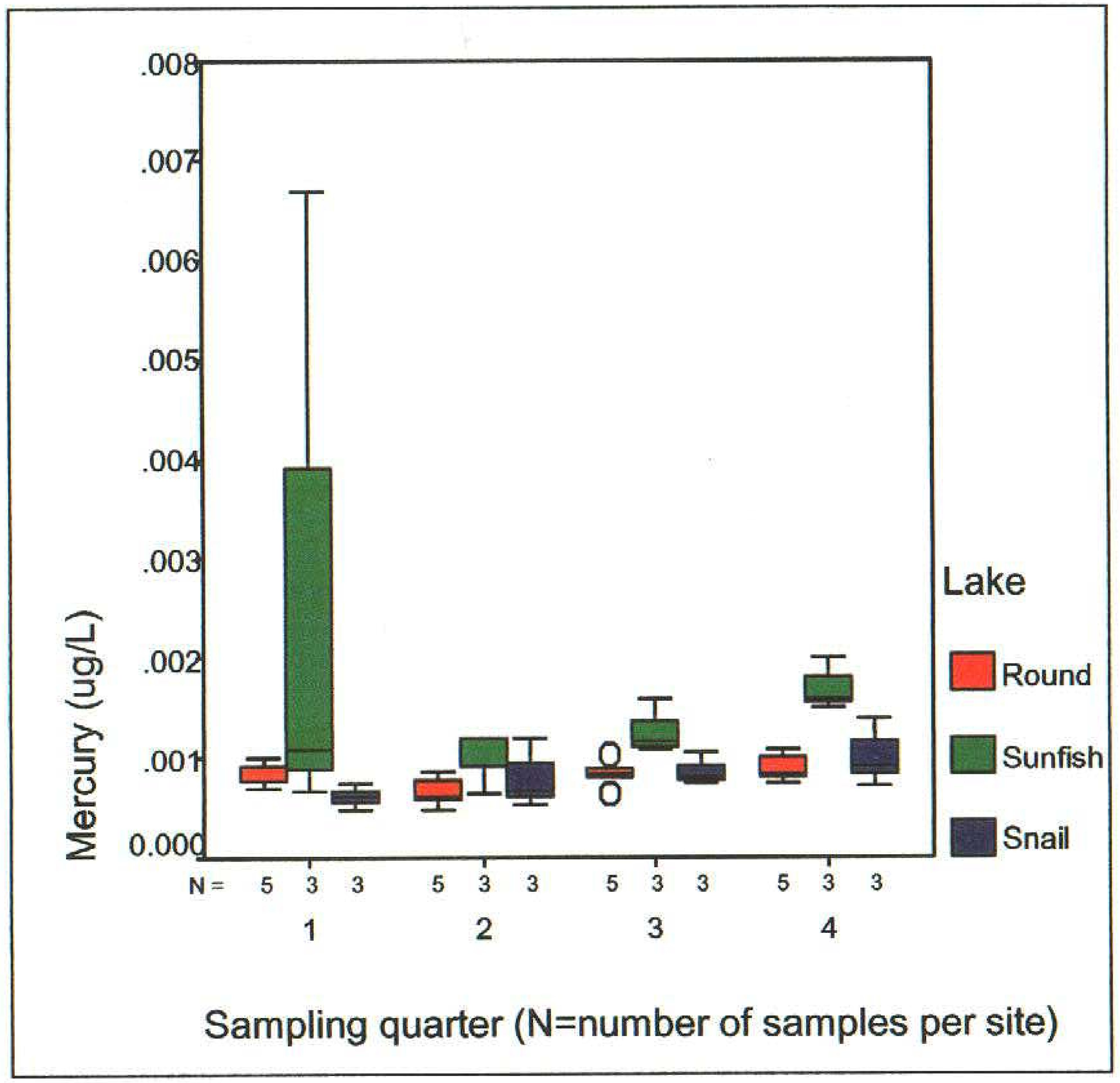

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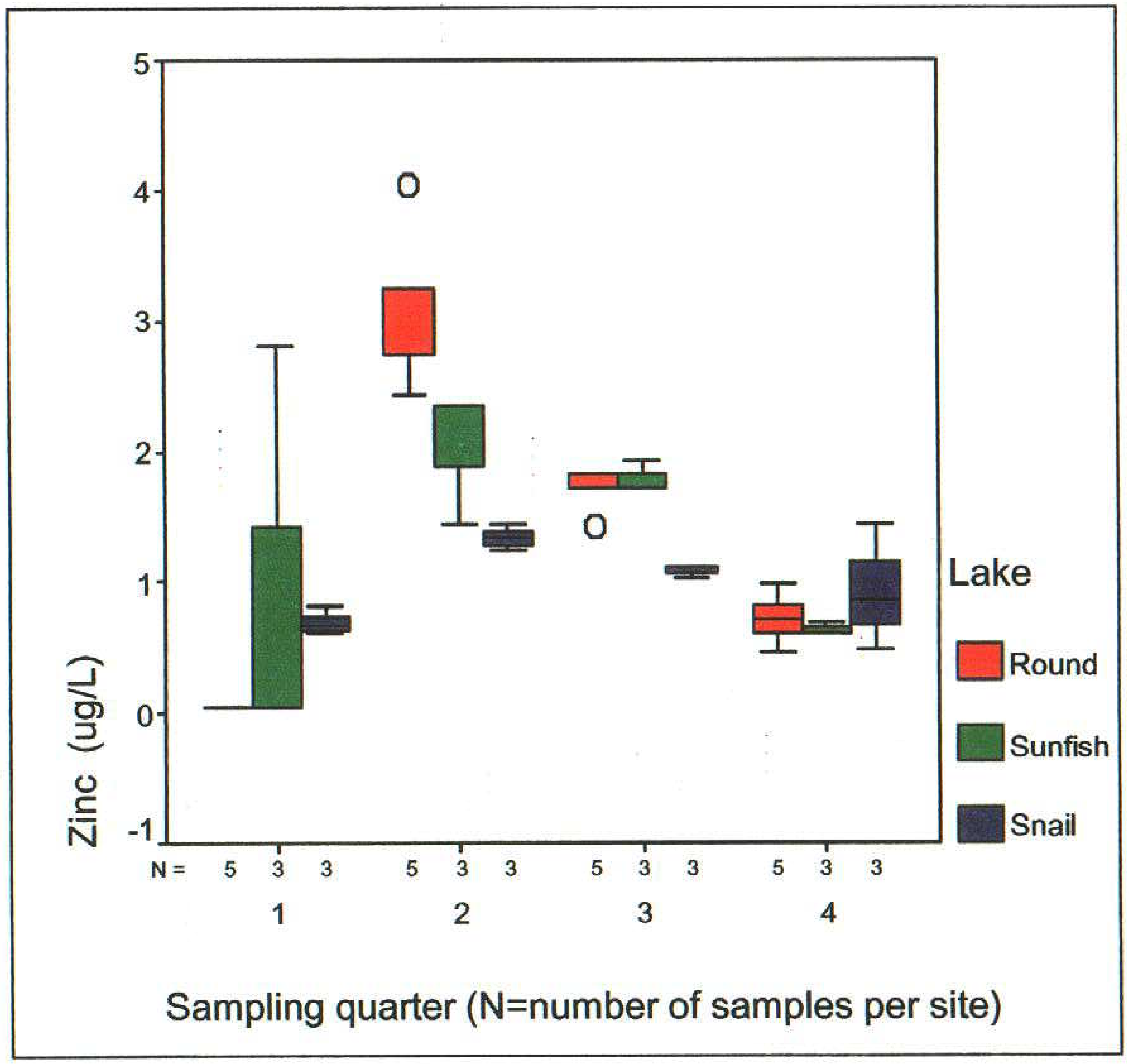
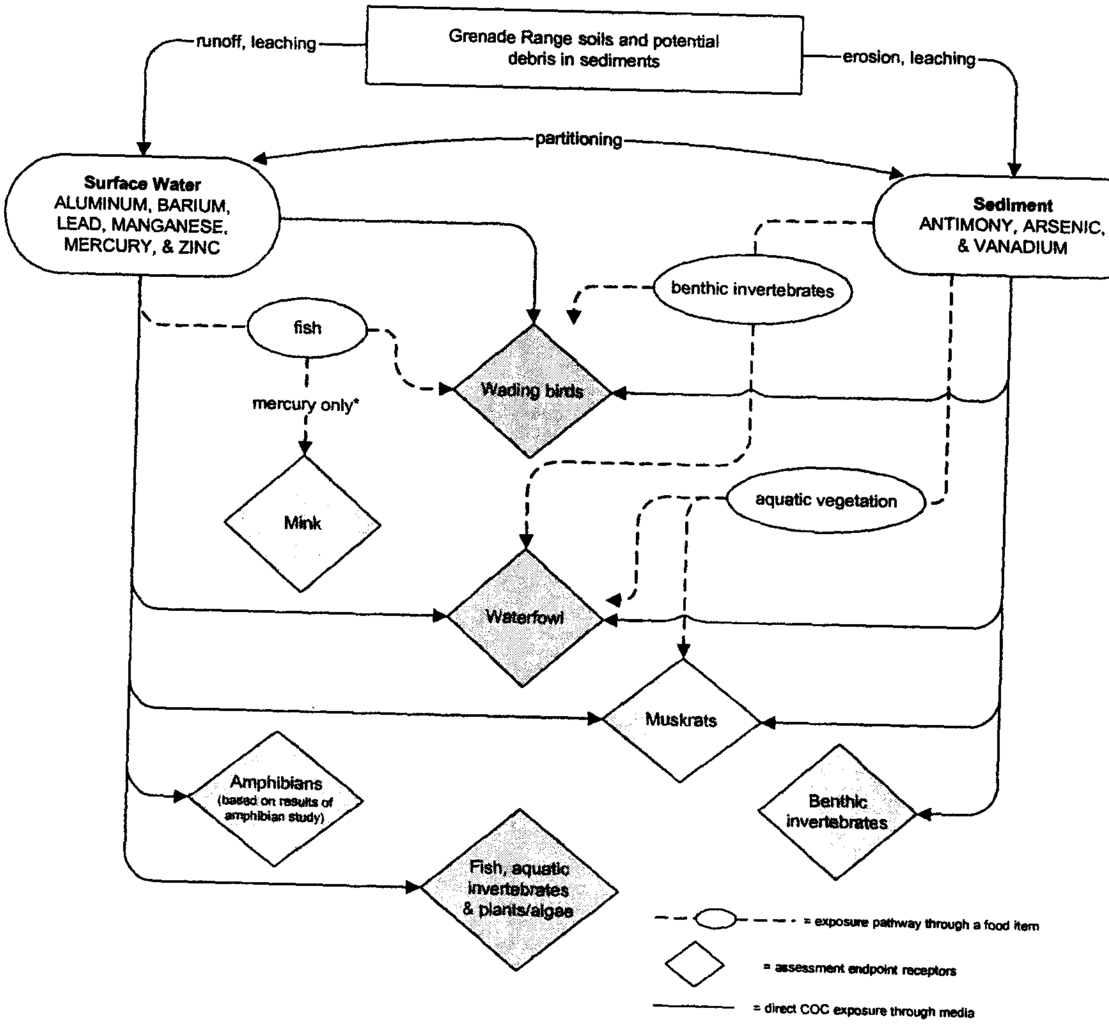


Appendix B-3

Revised Conceptual Site Models

Figure B3 - 1. Marsden Lake North Revised Conceptual Site Model



* Based on the risk hypothesis, only mercury is assessed with regard to mink

Figure B3 - 2. . Marsden Lake South Revised Conceptual Site Model

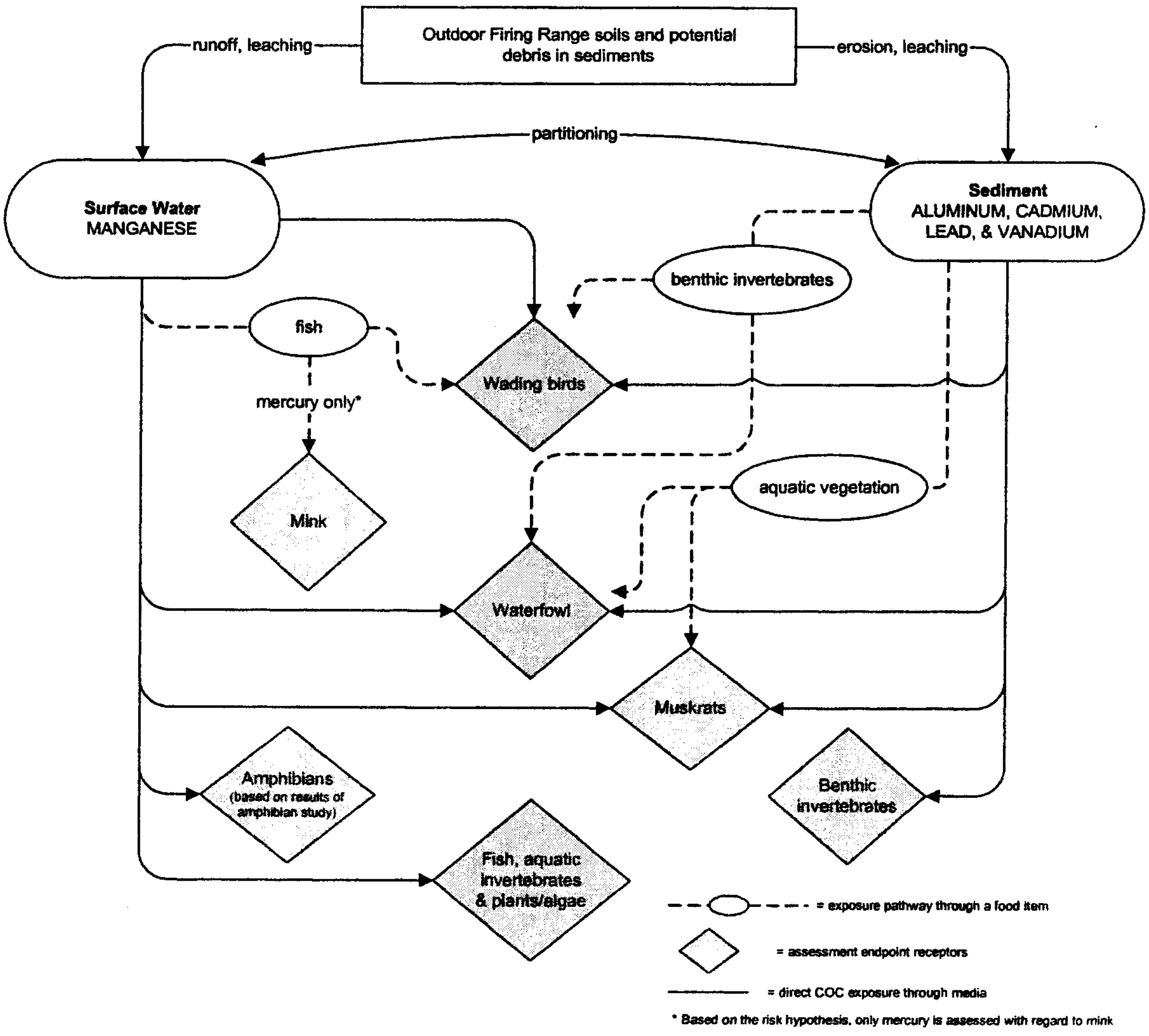


Figure B3 - 3. Pond G Revised Conceptual Site Model

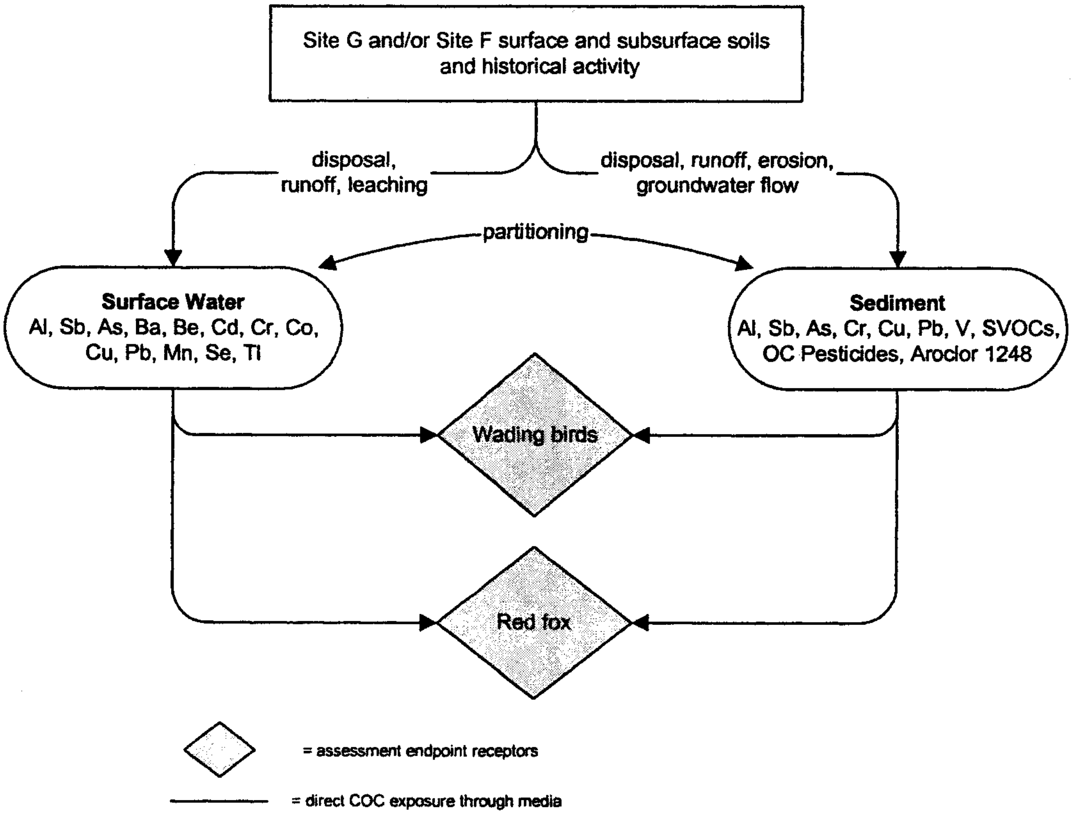
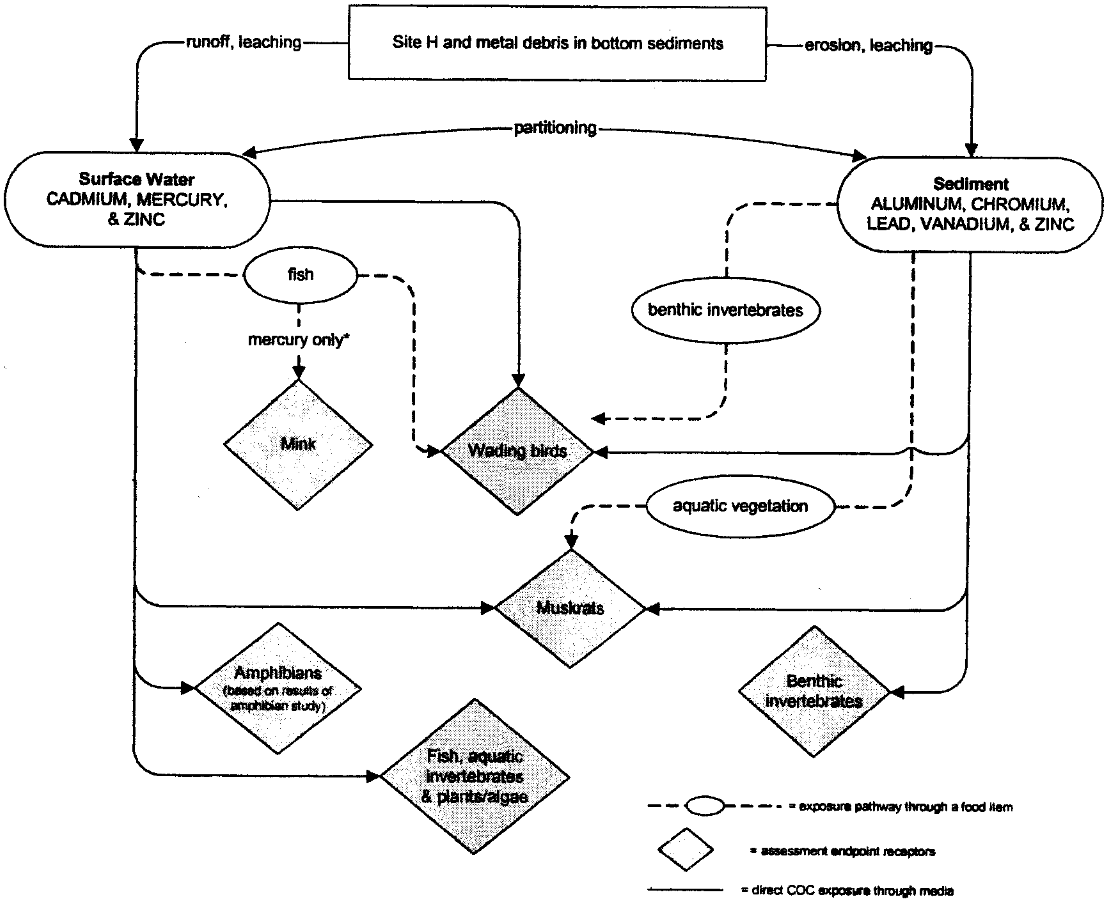
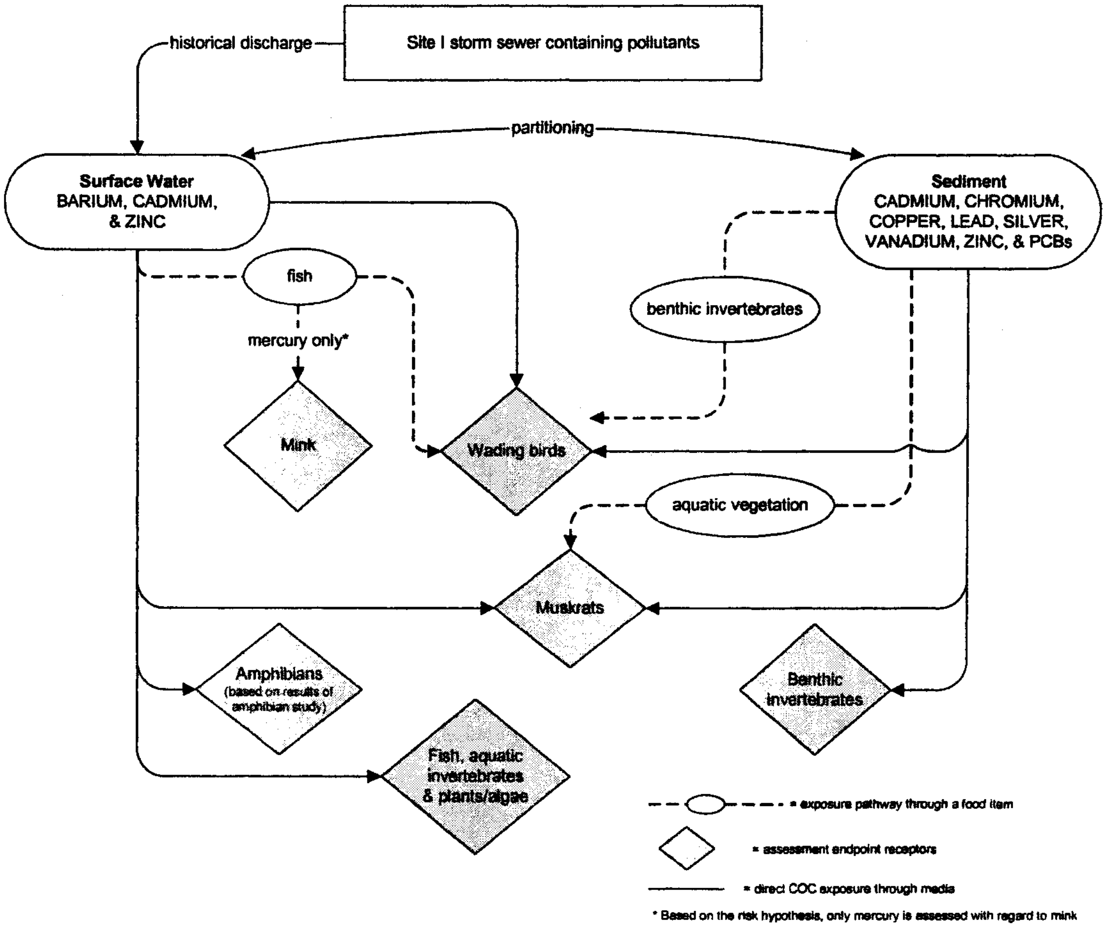


Figure B3 - 4. Sunfish Lake Revised Conceptual Site Model



* Based on the risk hypothesis, only mercury is assessed with regard to mink

Figure B3 - 5. Round Lake Revised Conceptual Site Model



Appendix B-4

Risk Characterization Figures

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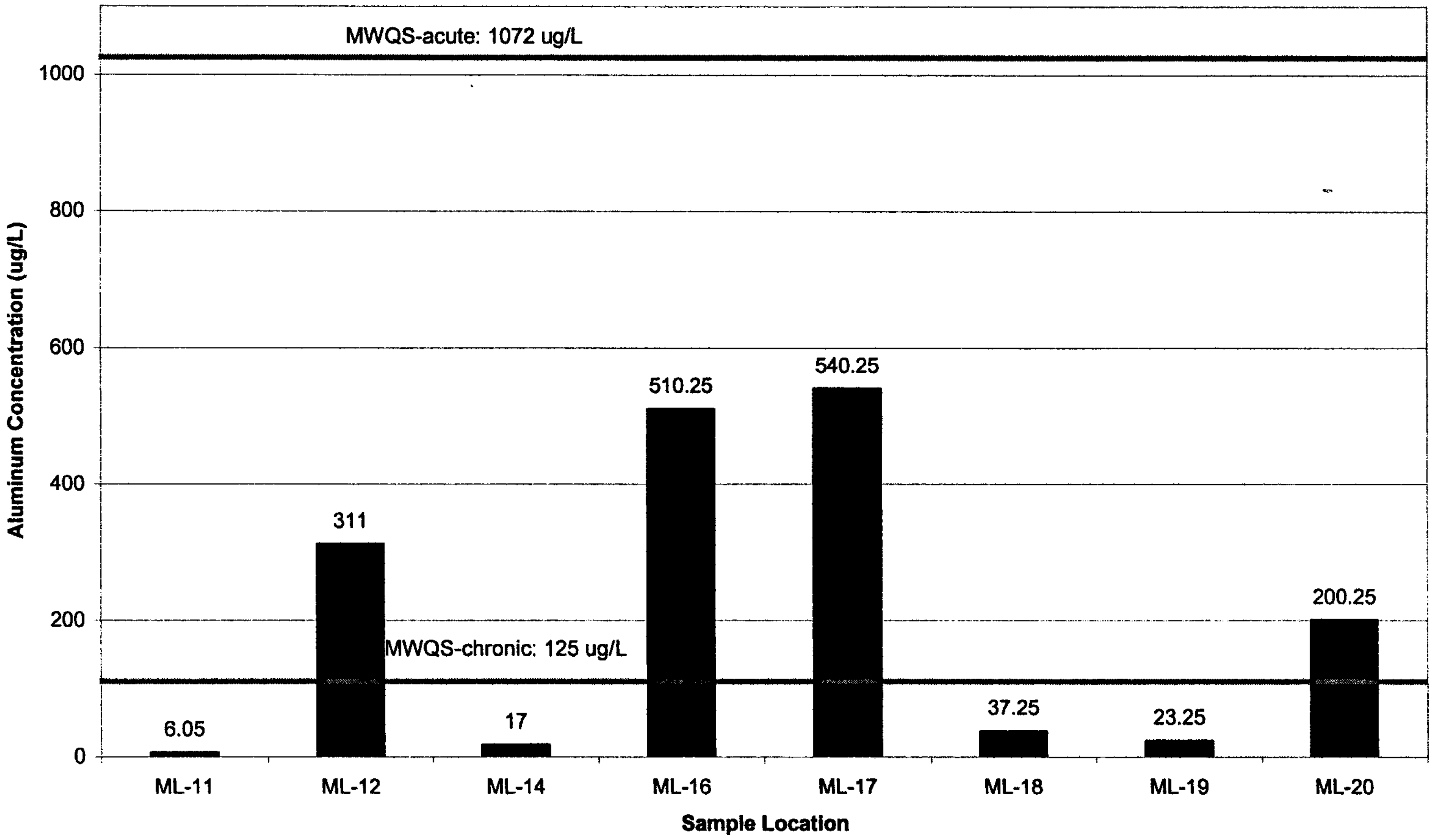

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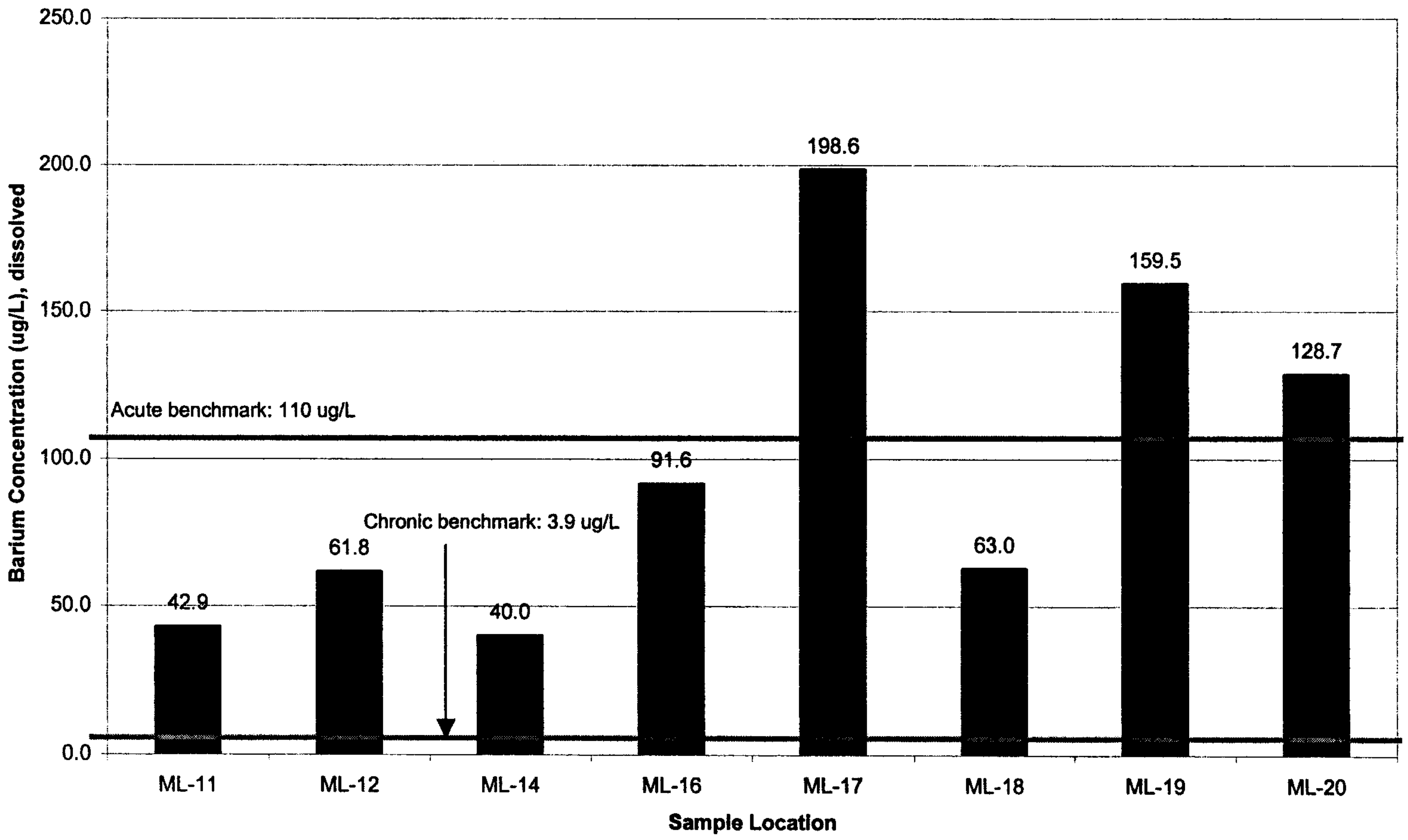

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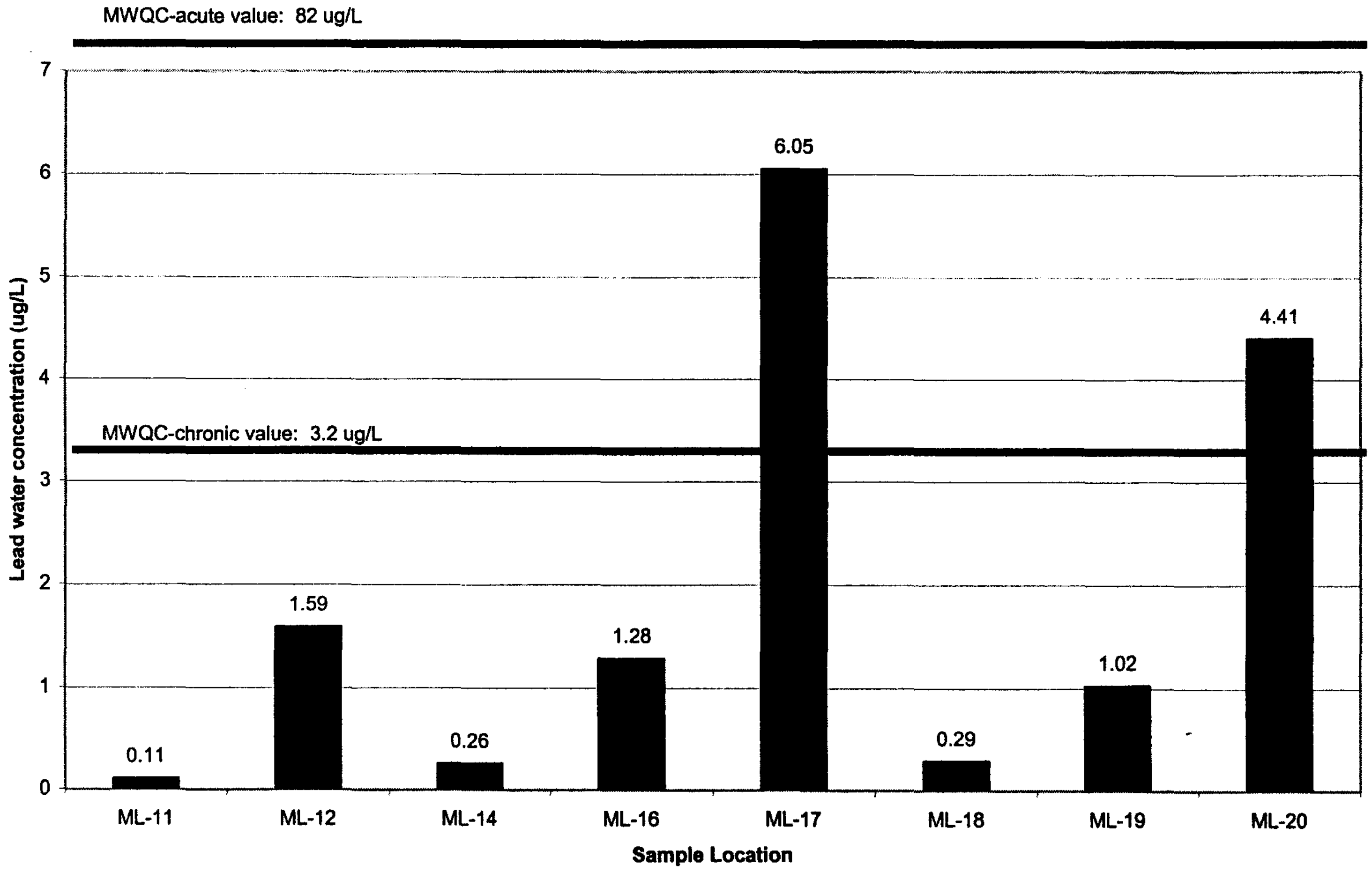

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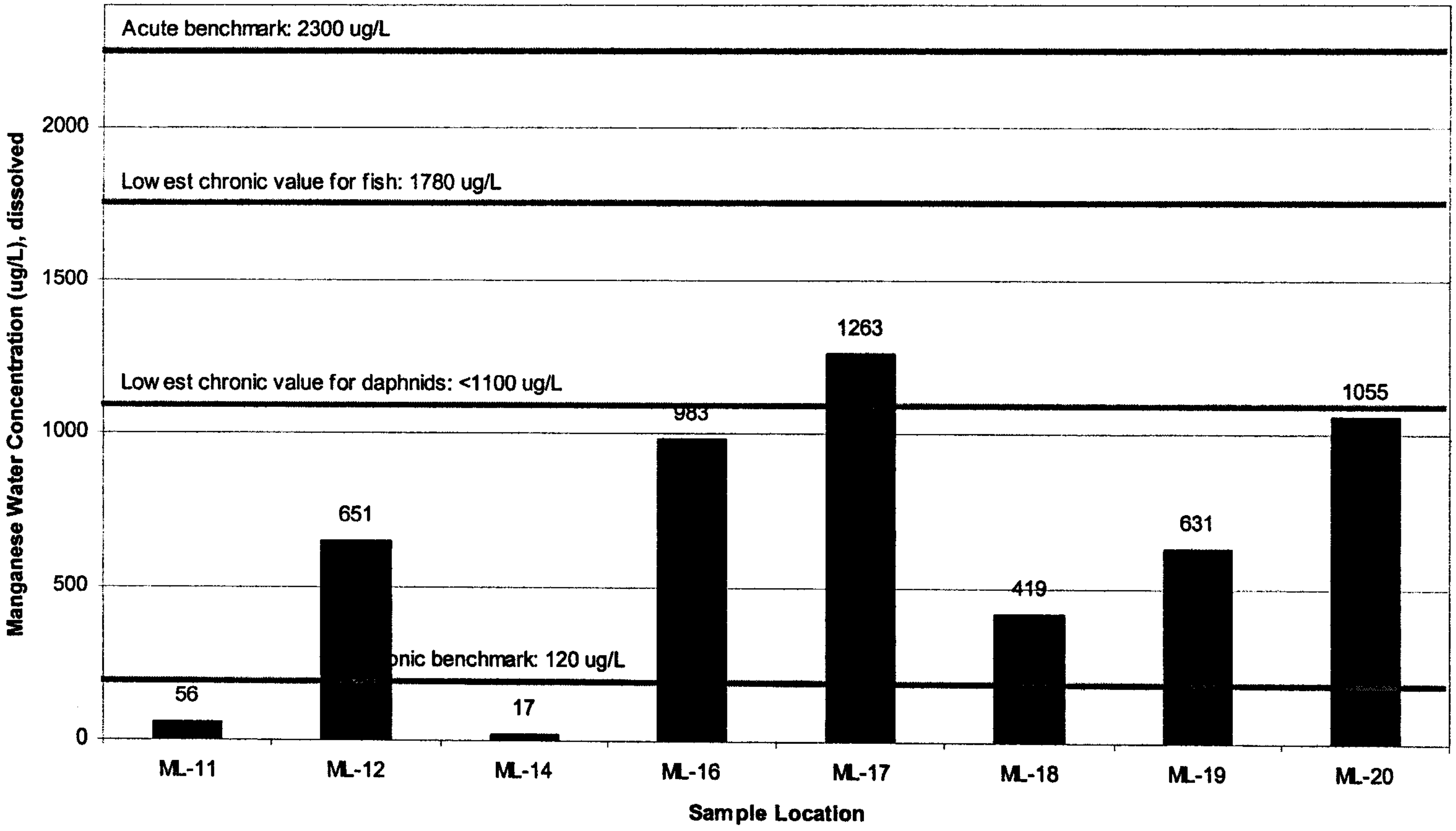

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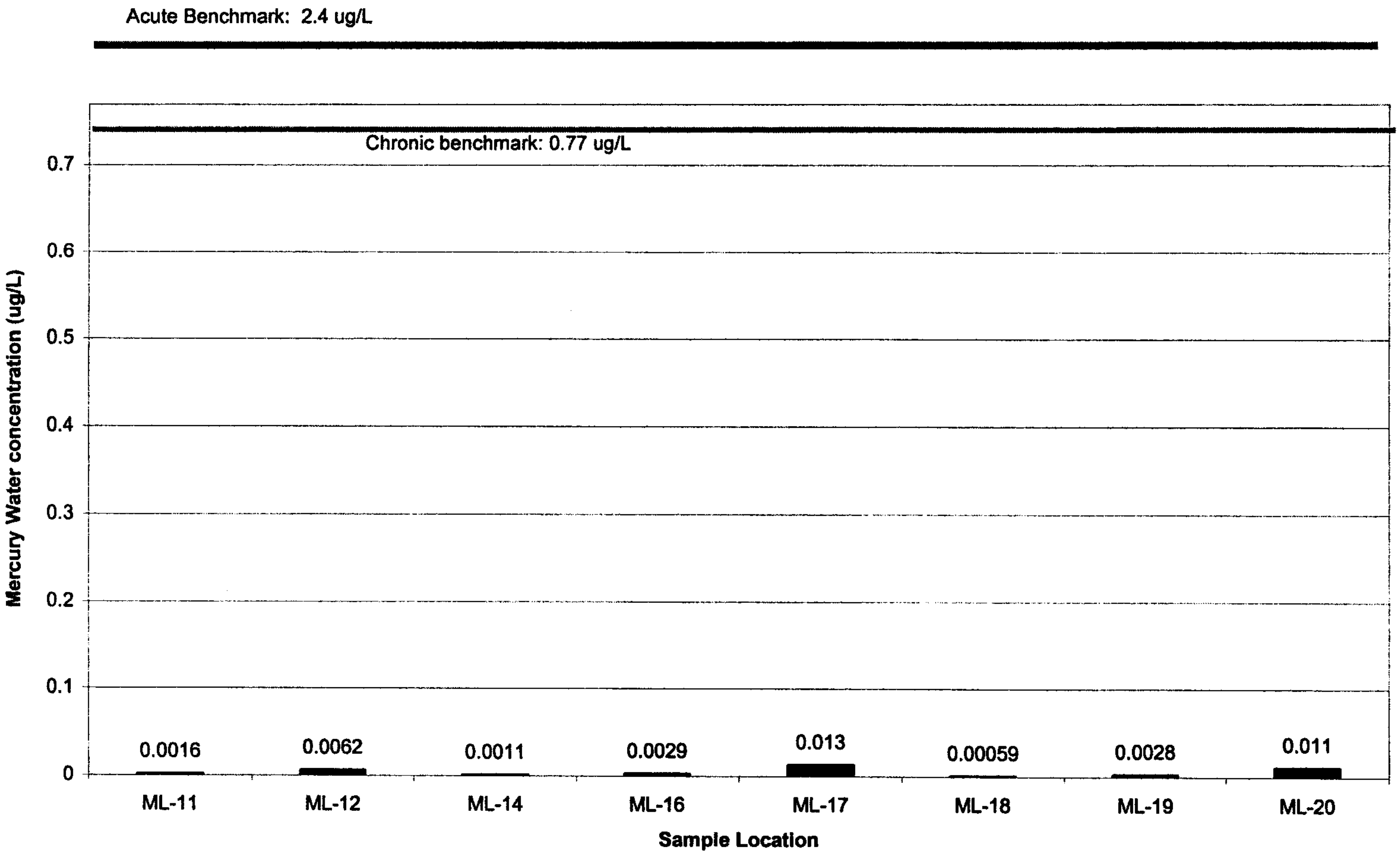

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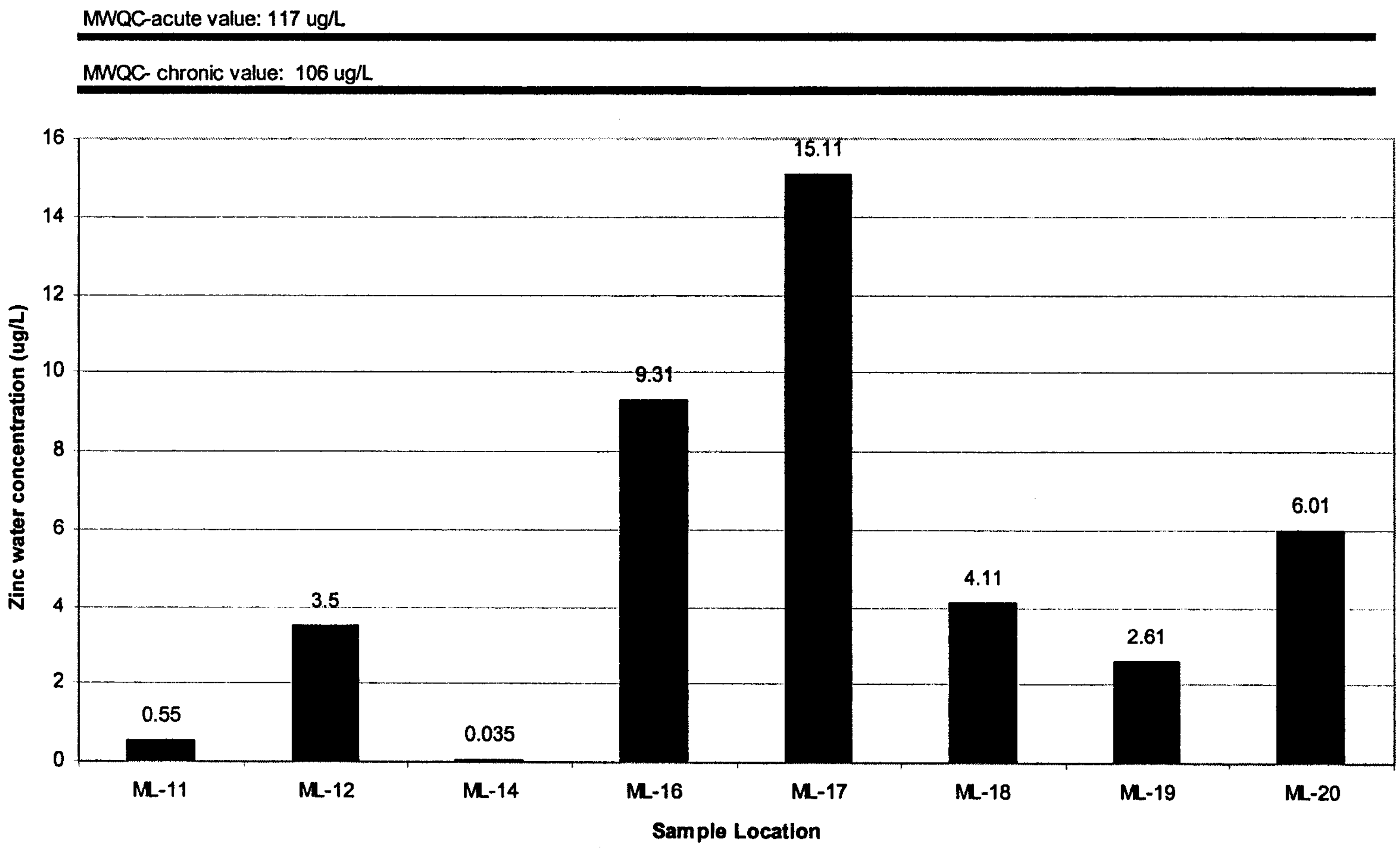

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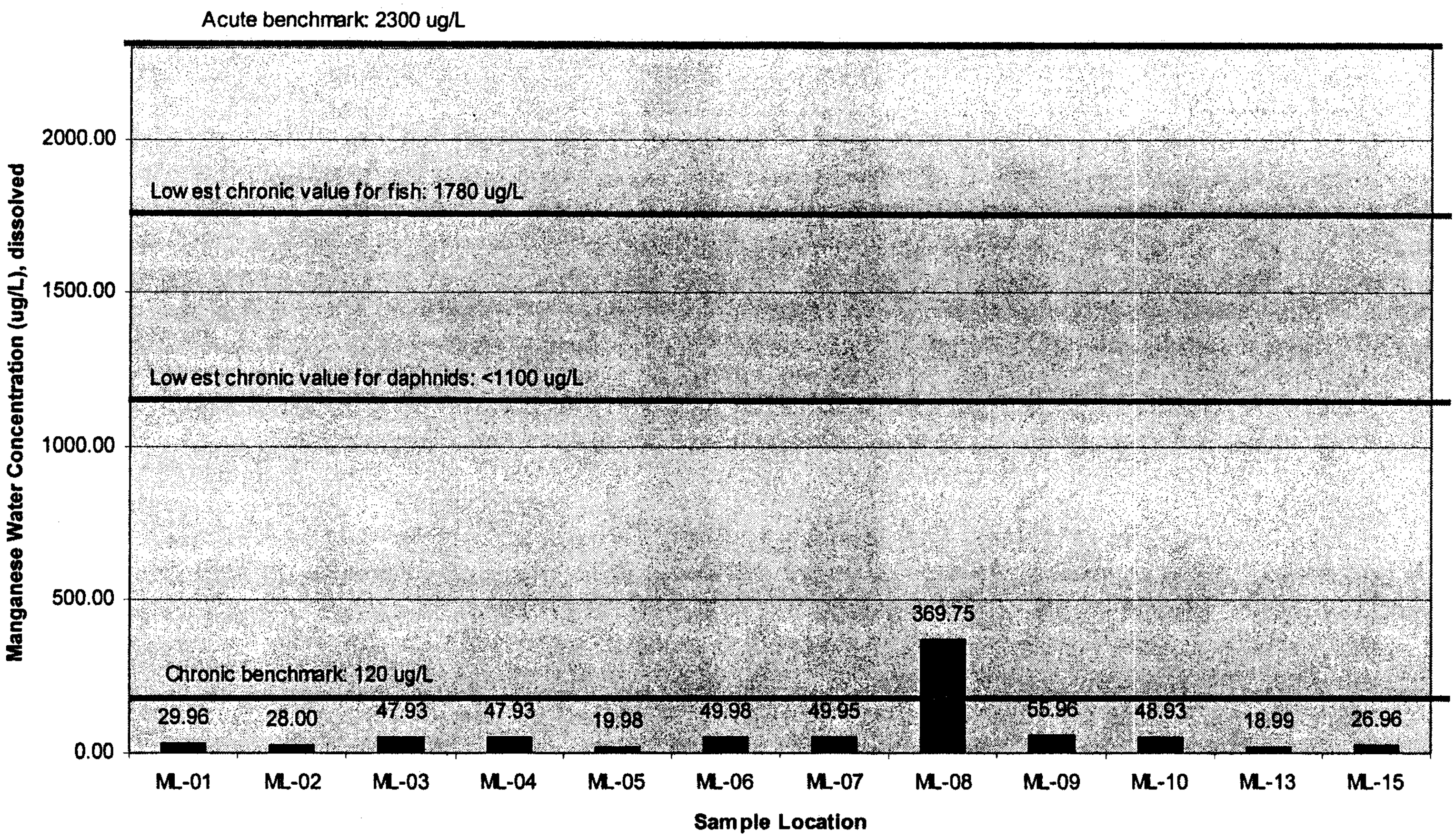

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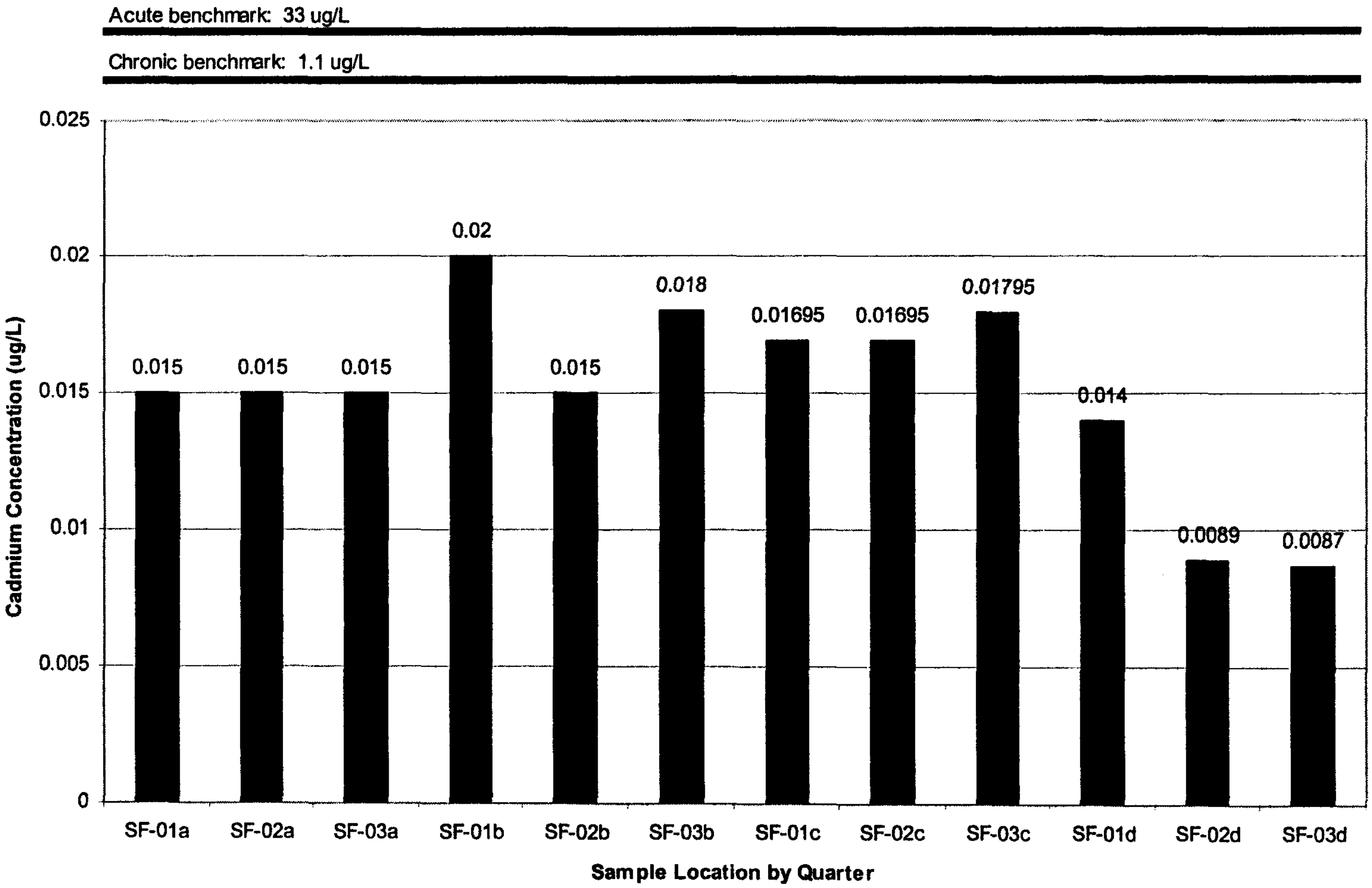

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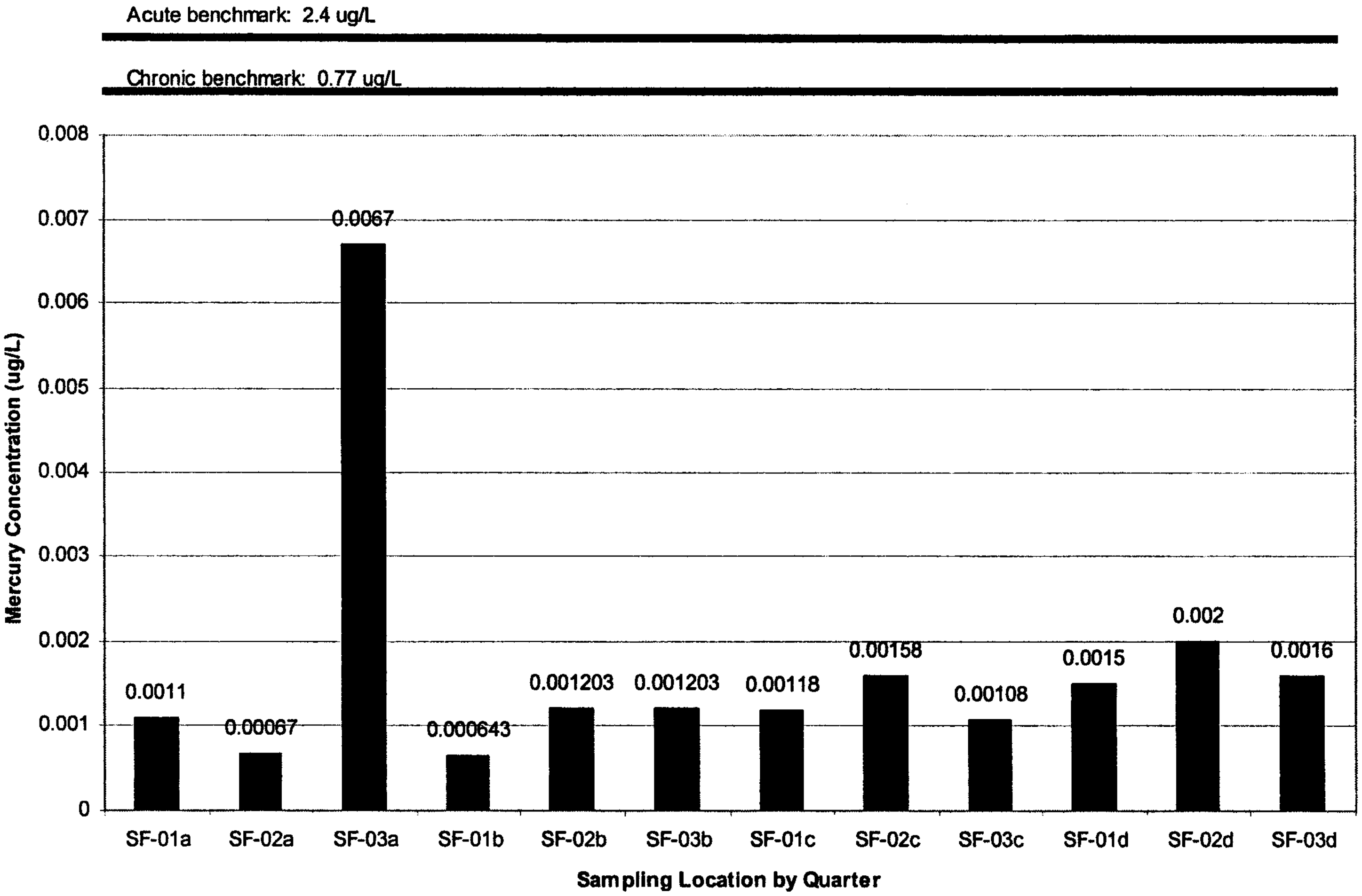

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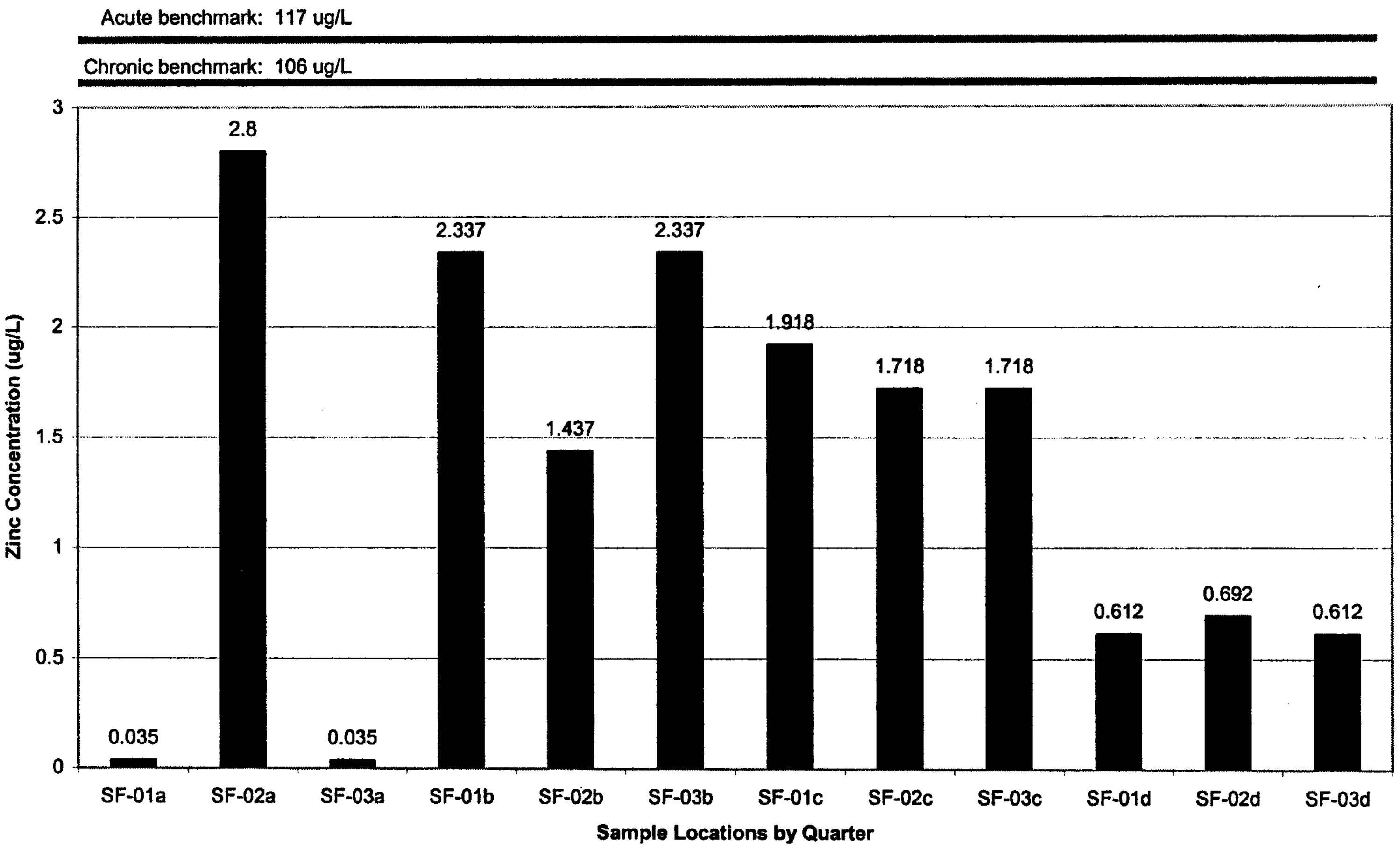

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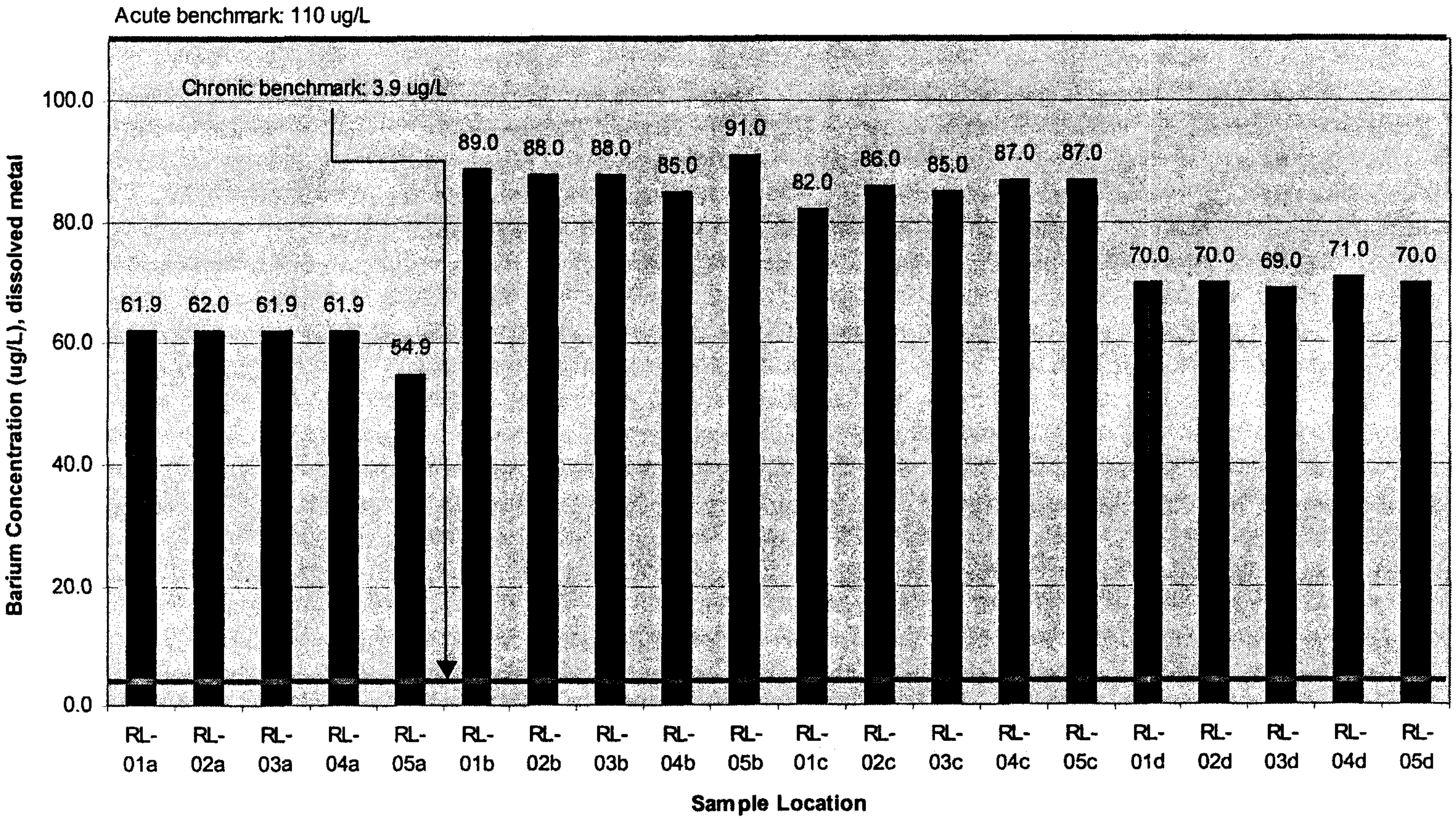

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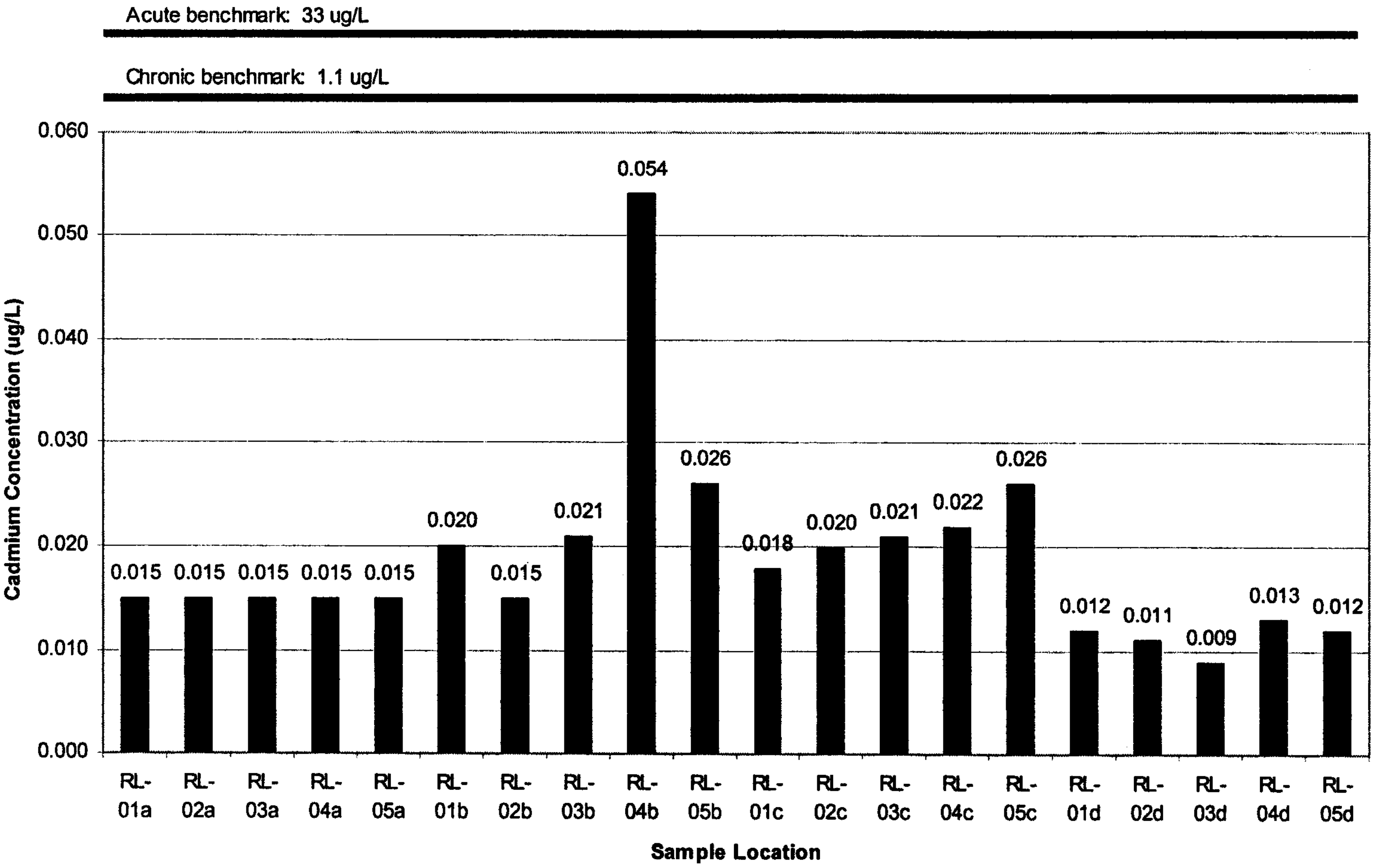

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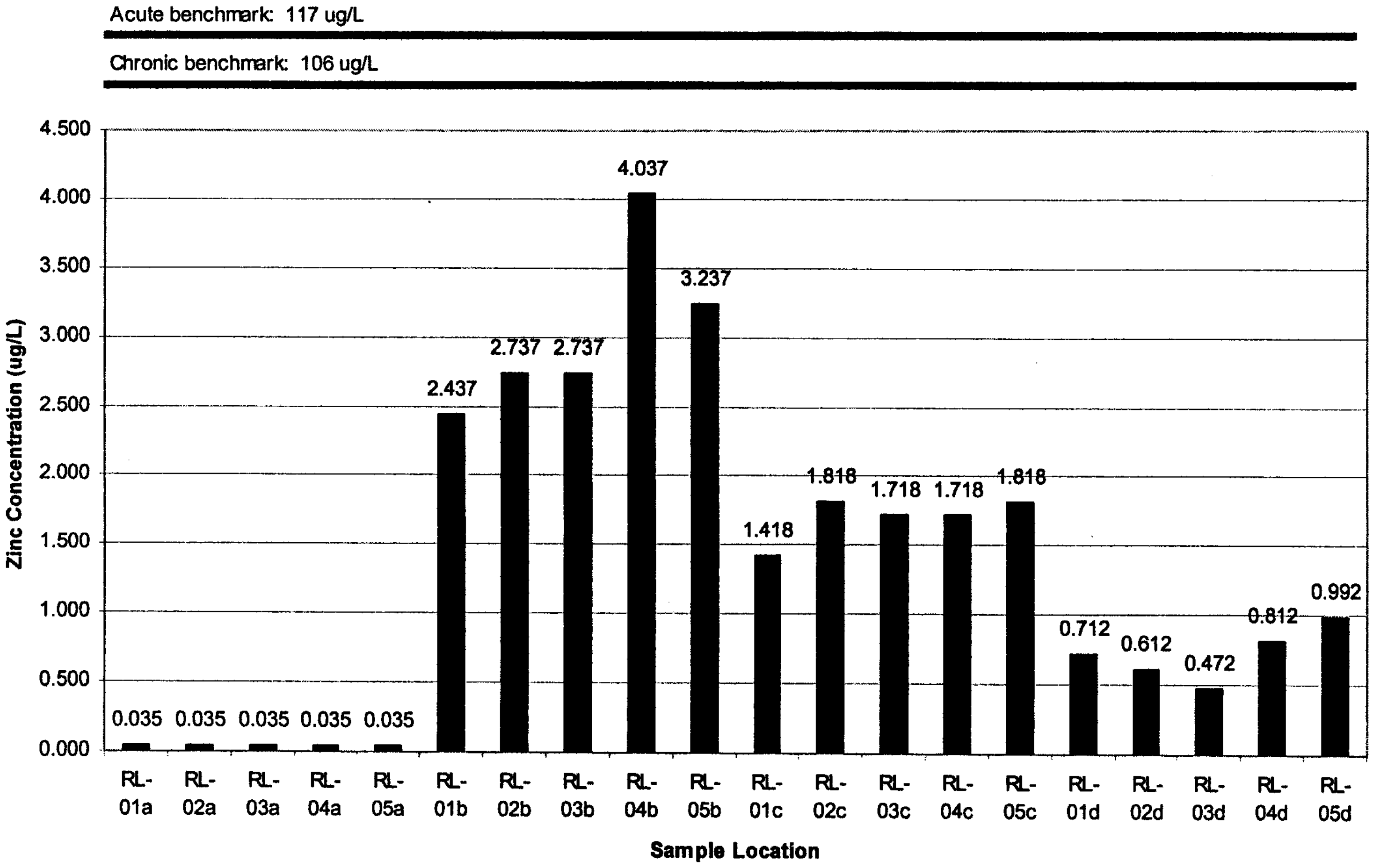

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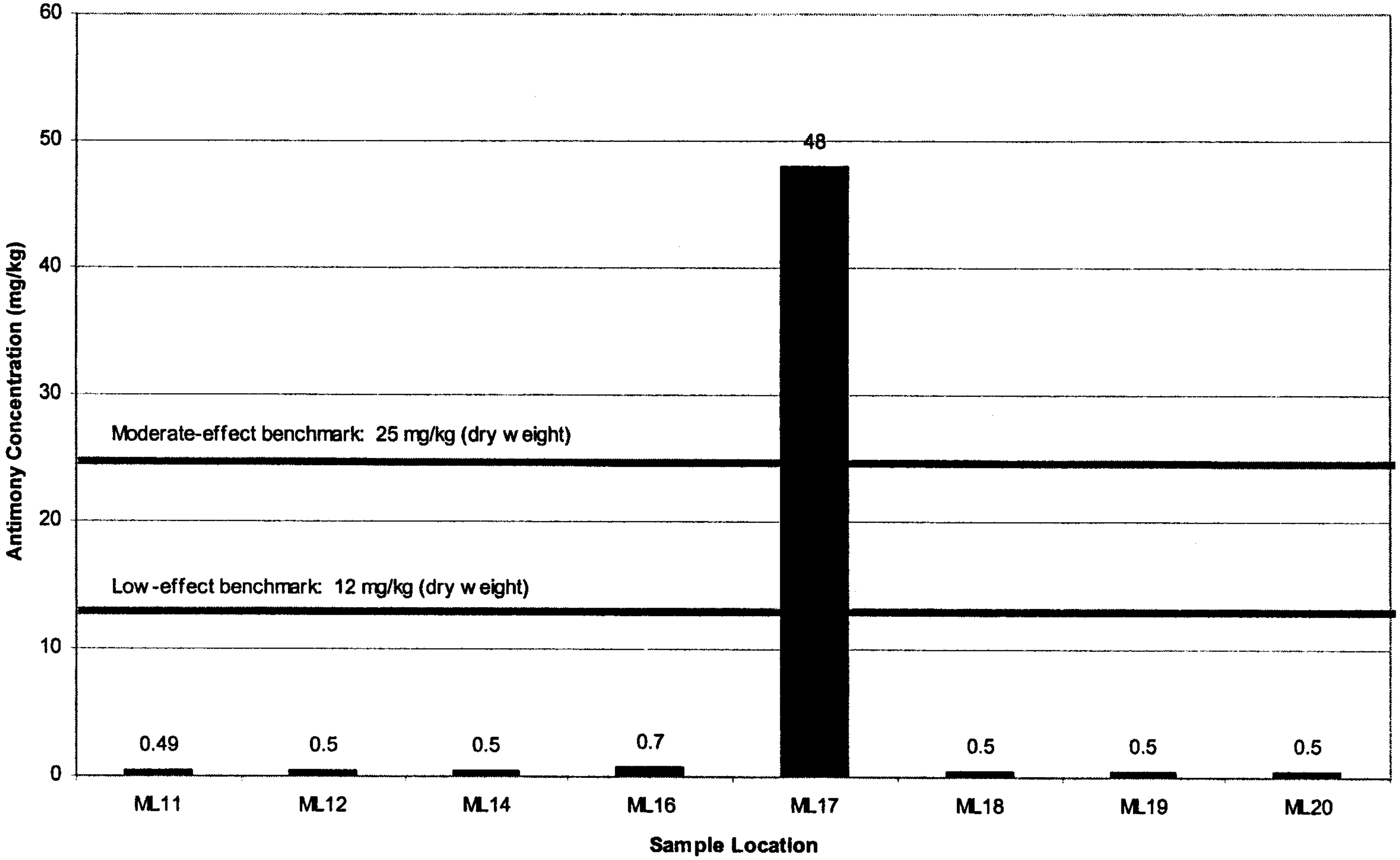

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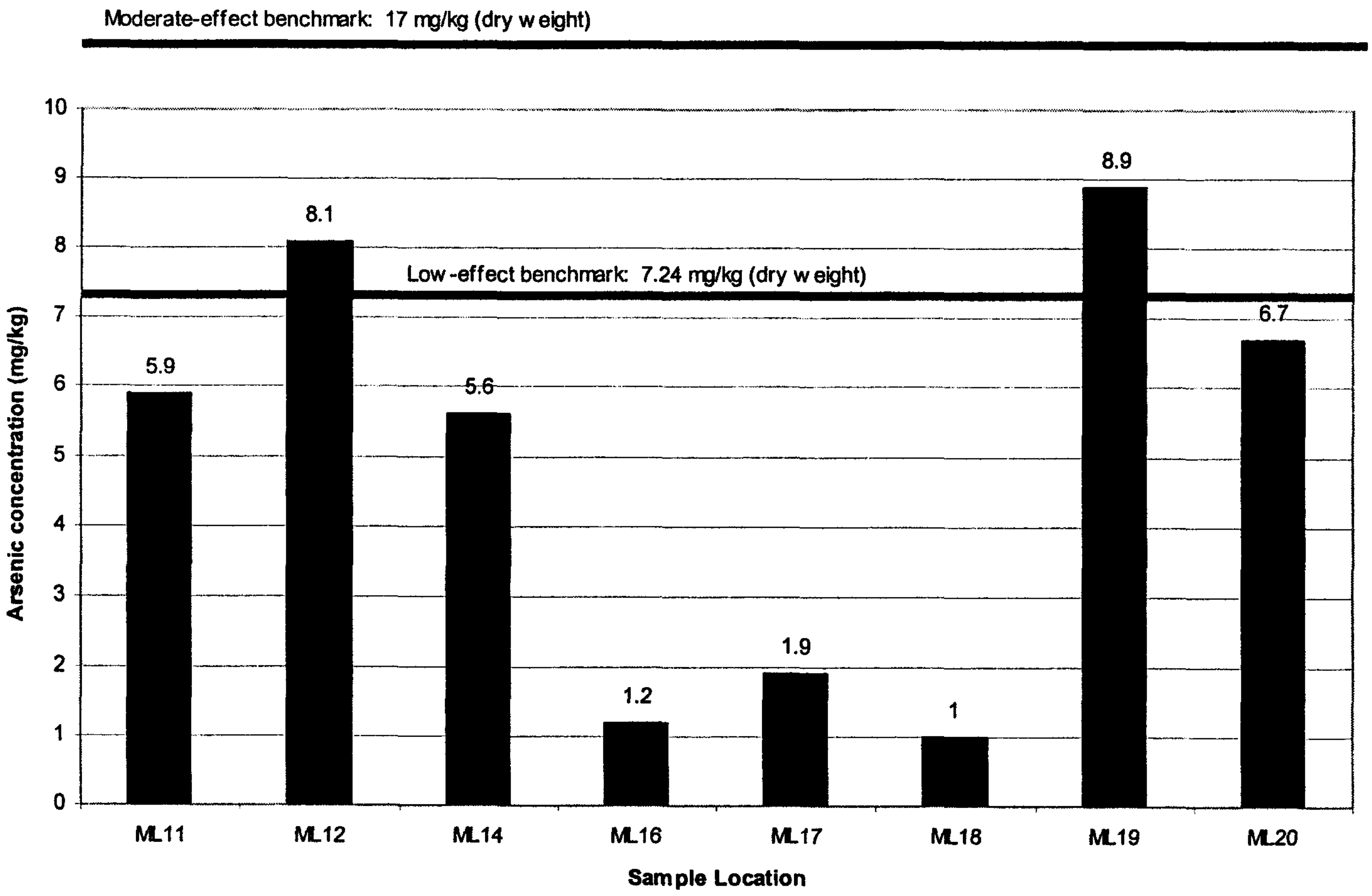

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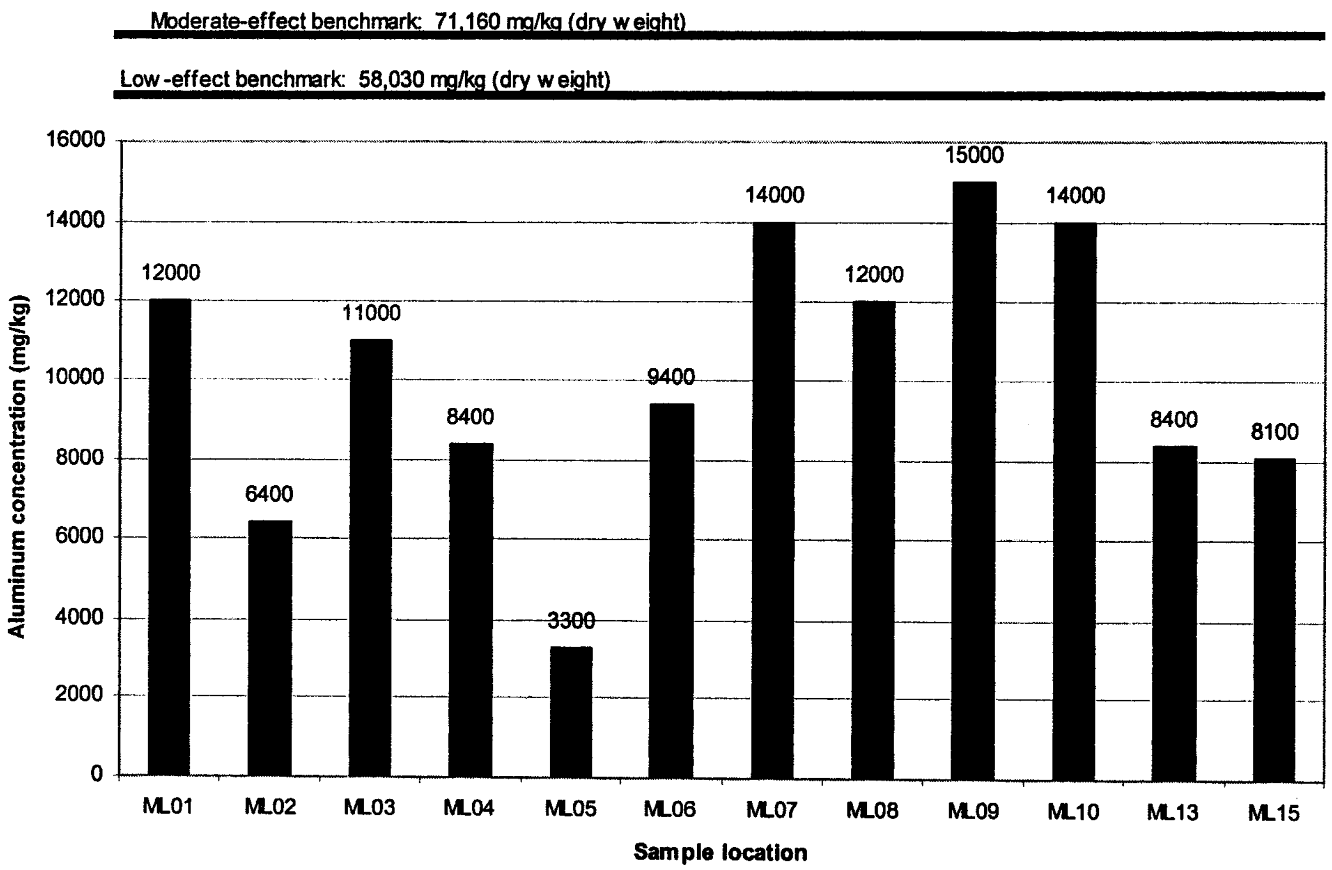

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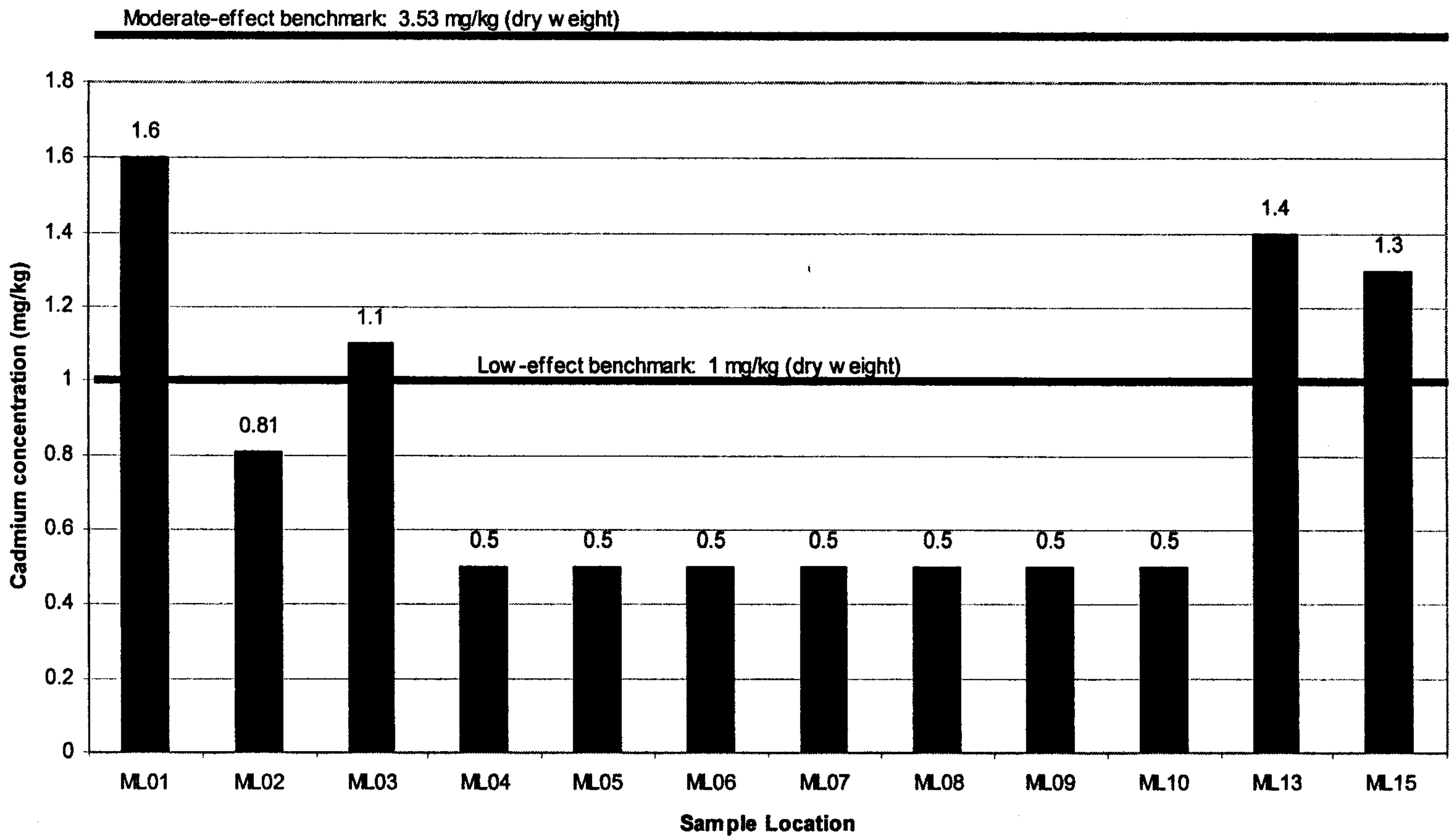

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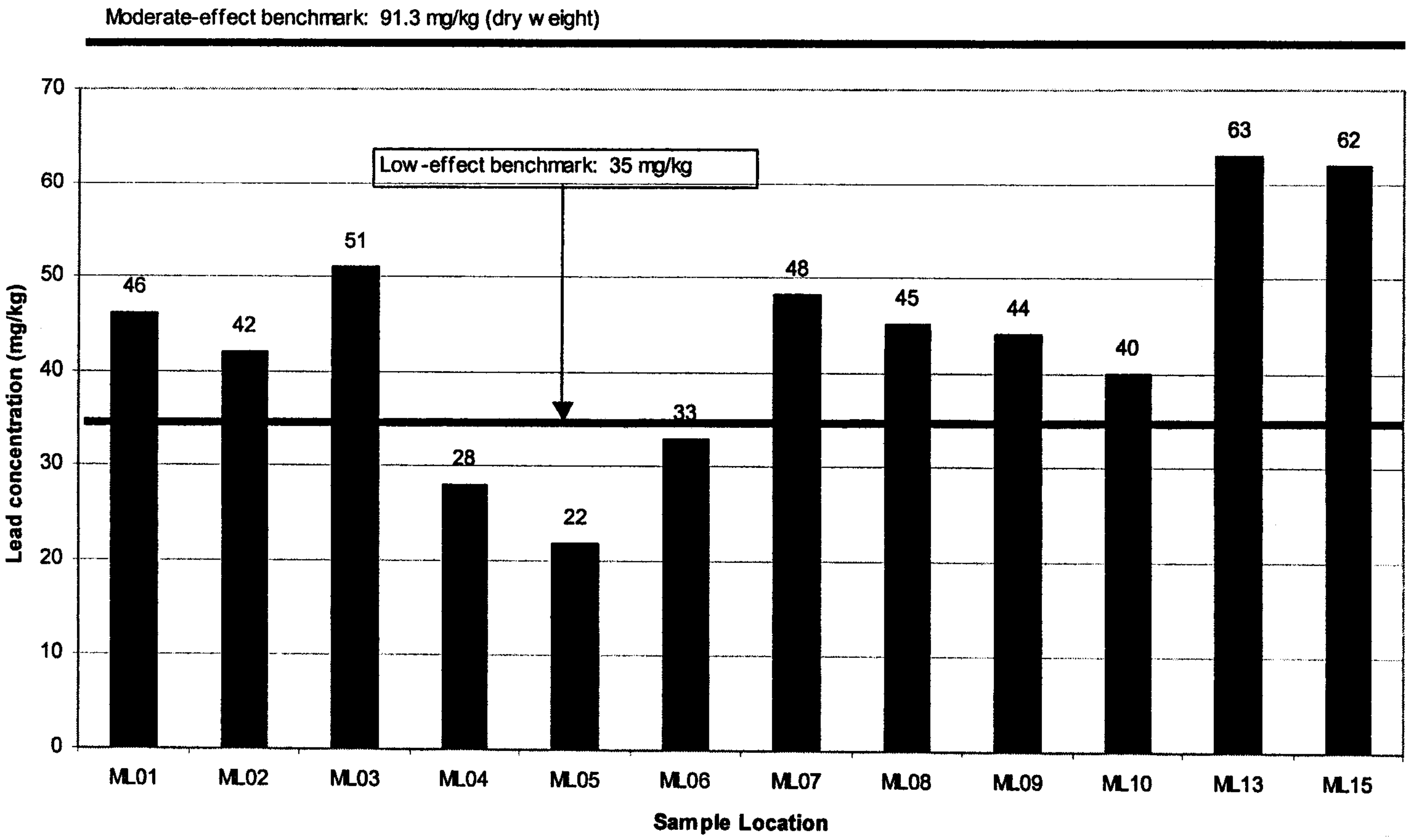

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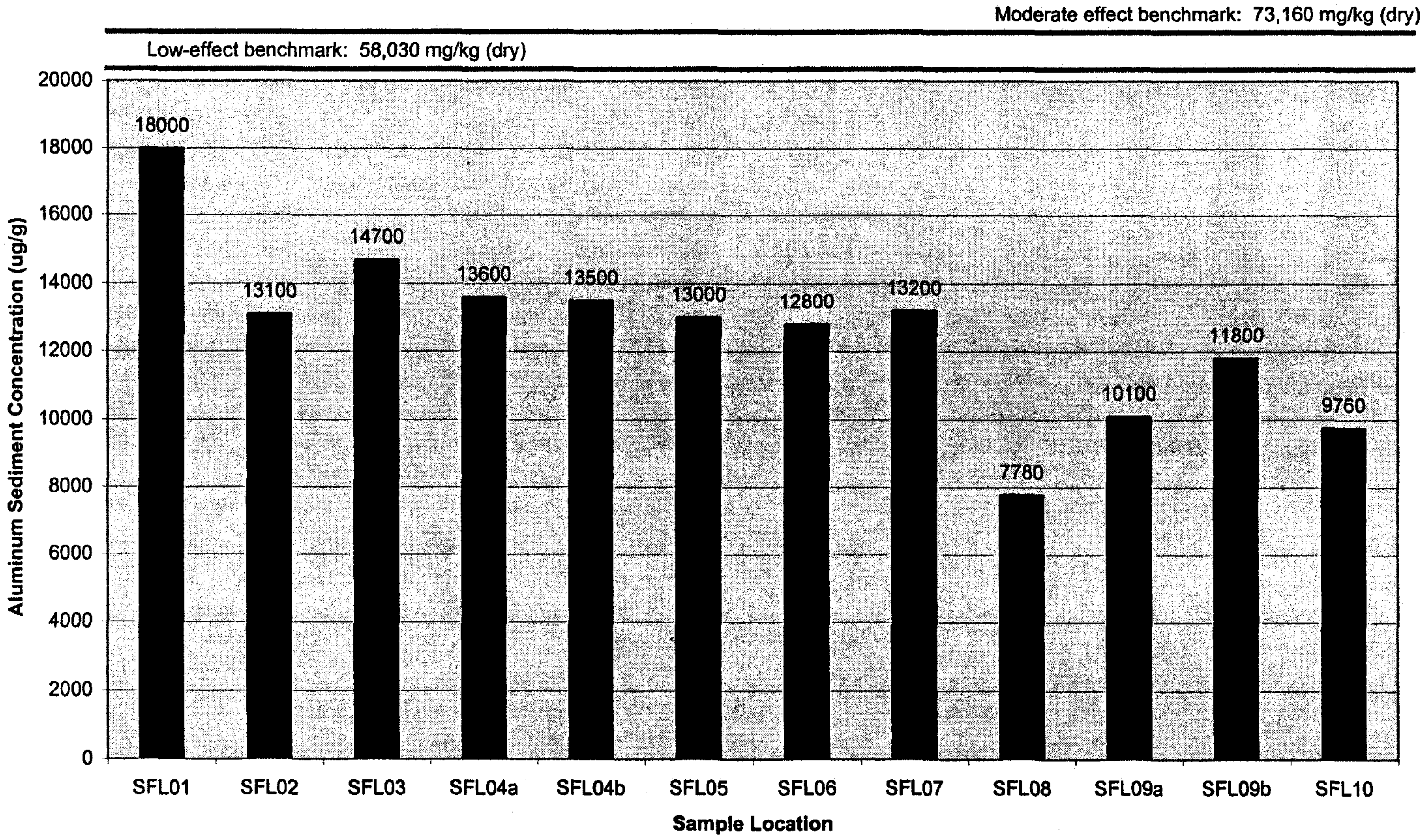

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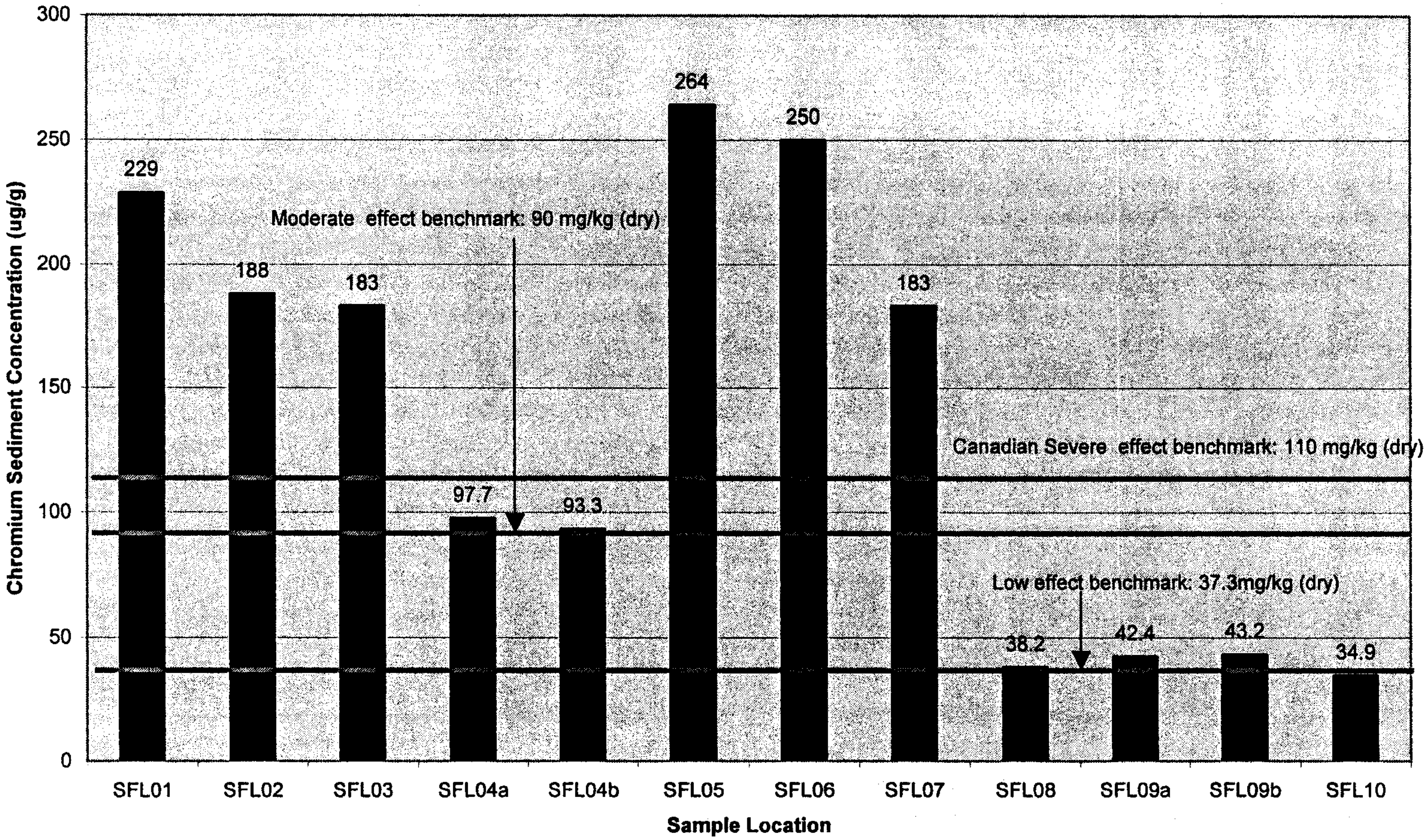

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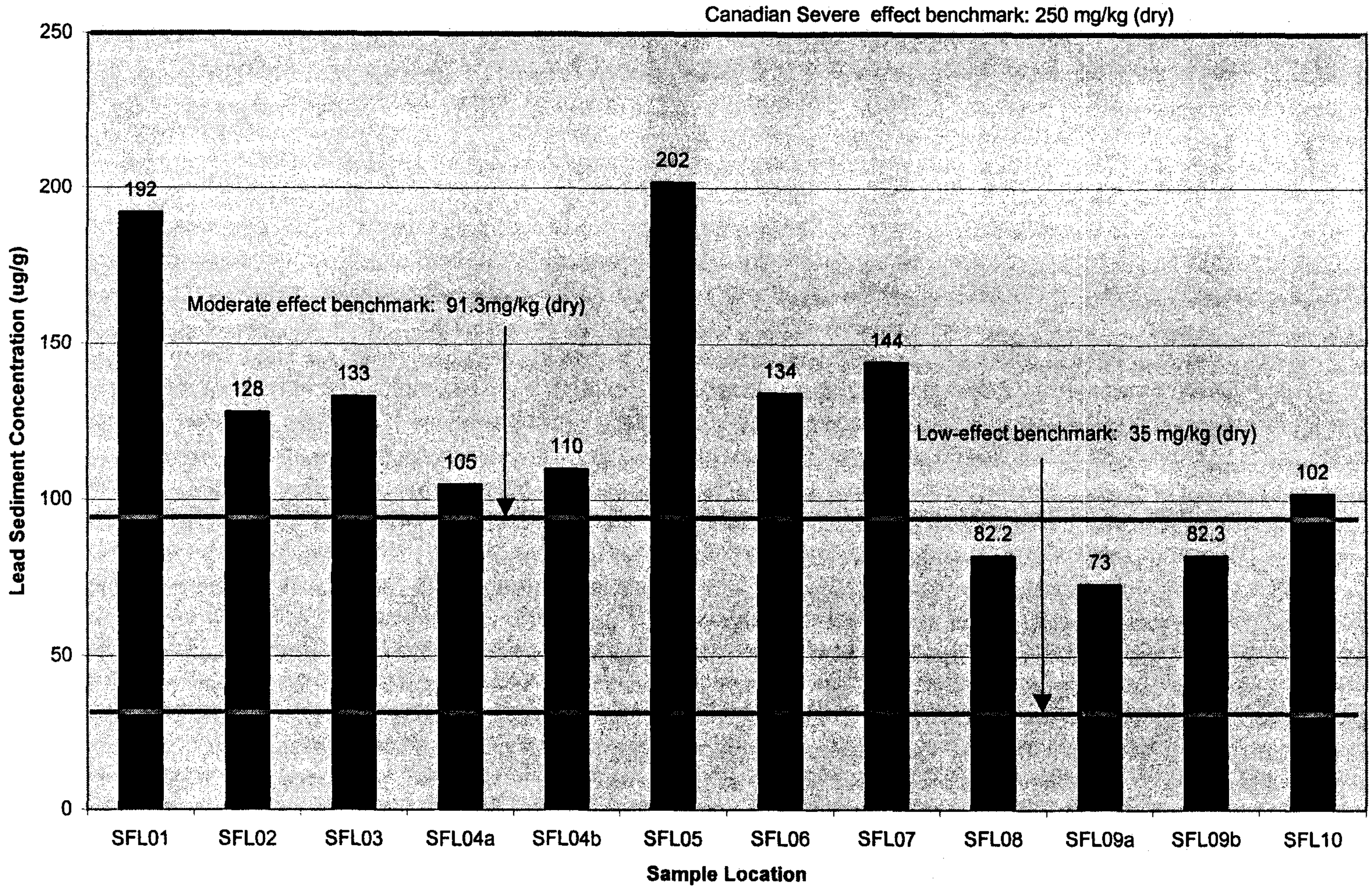

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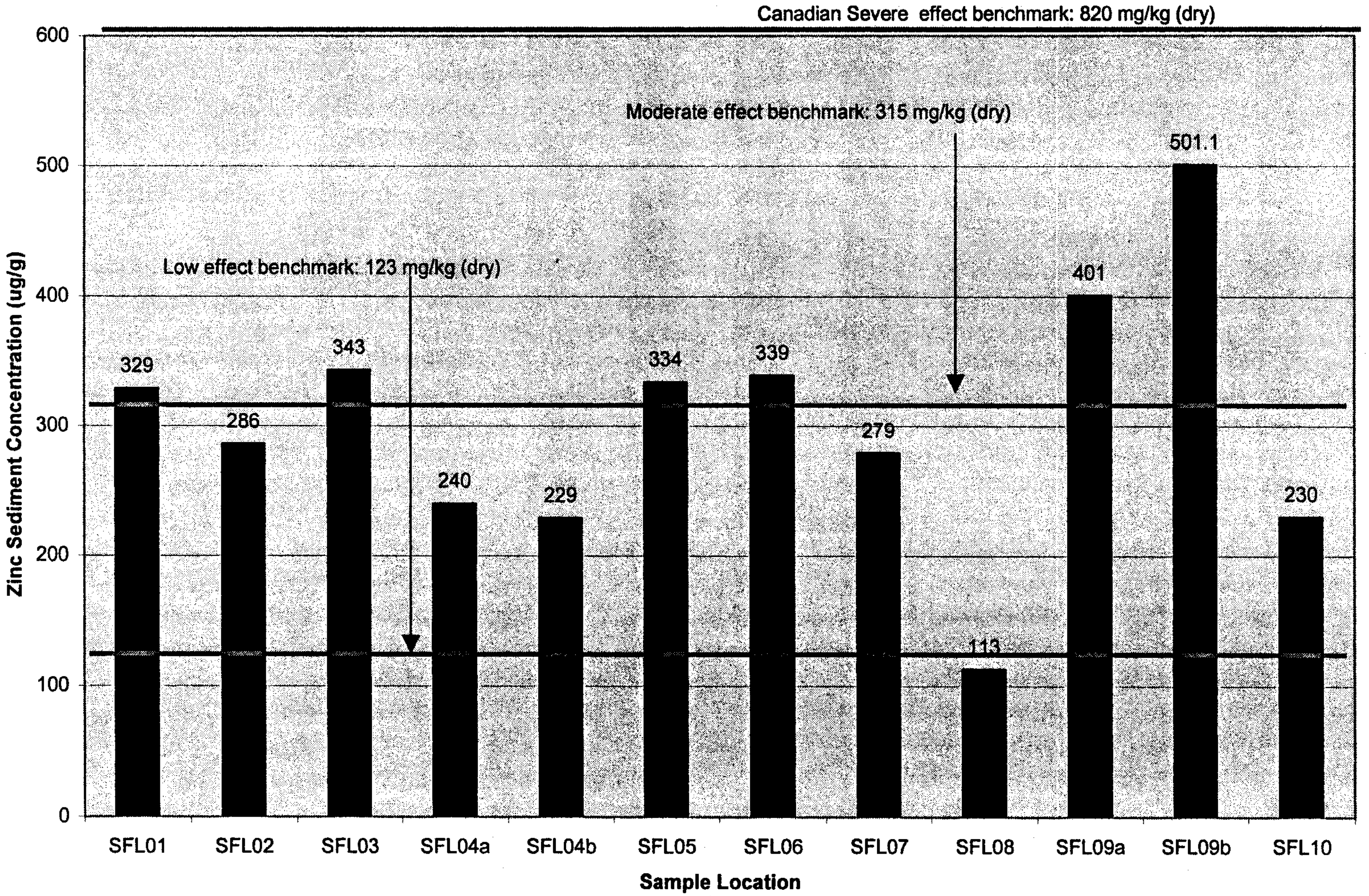

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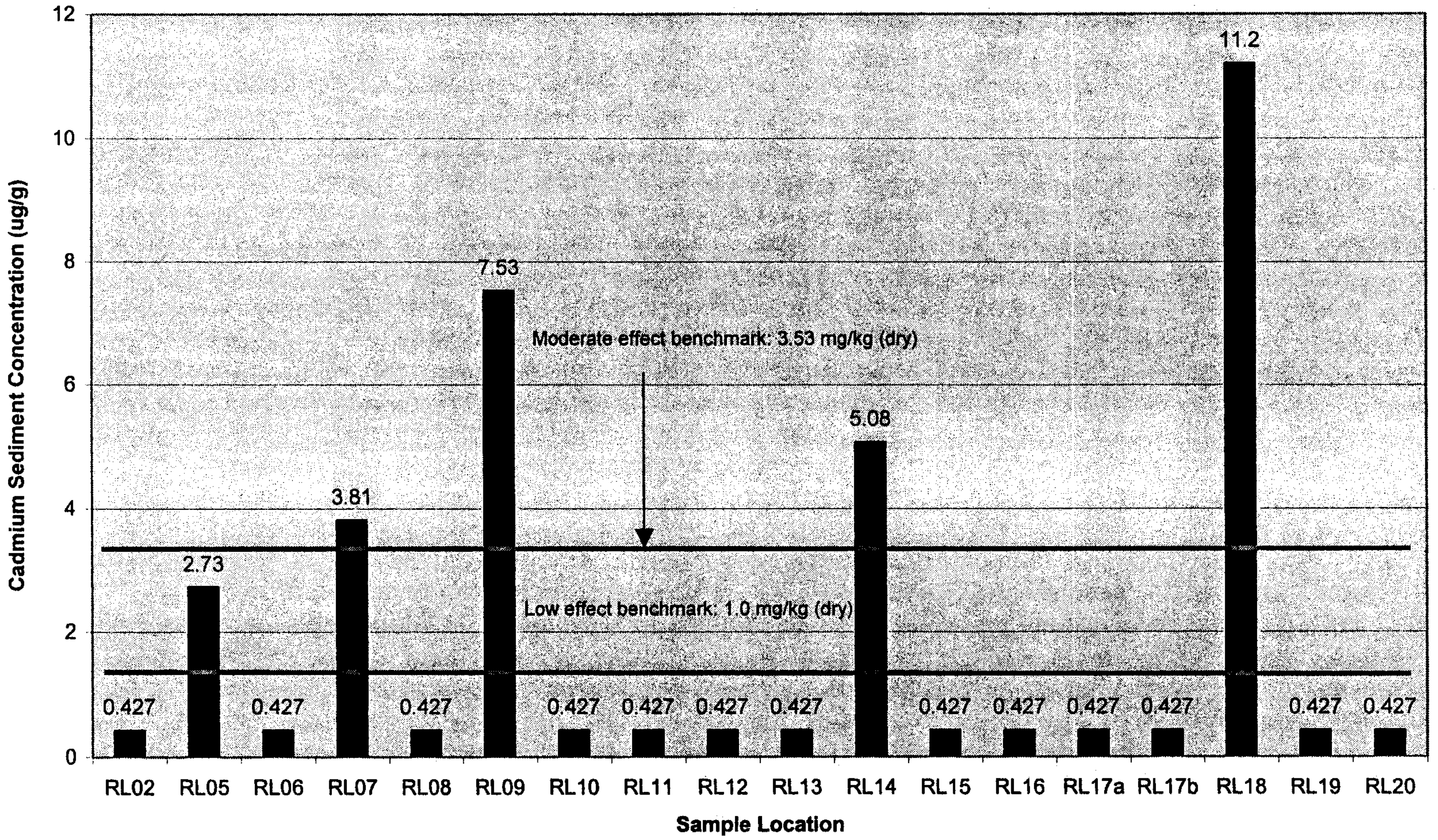

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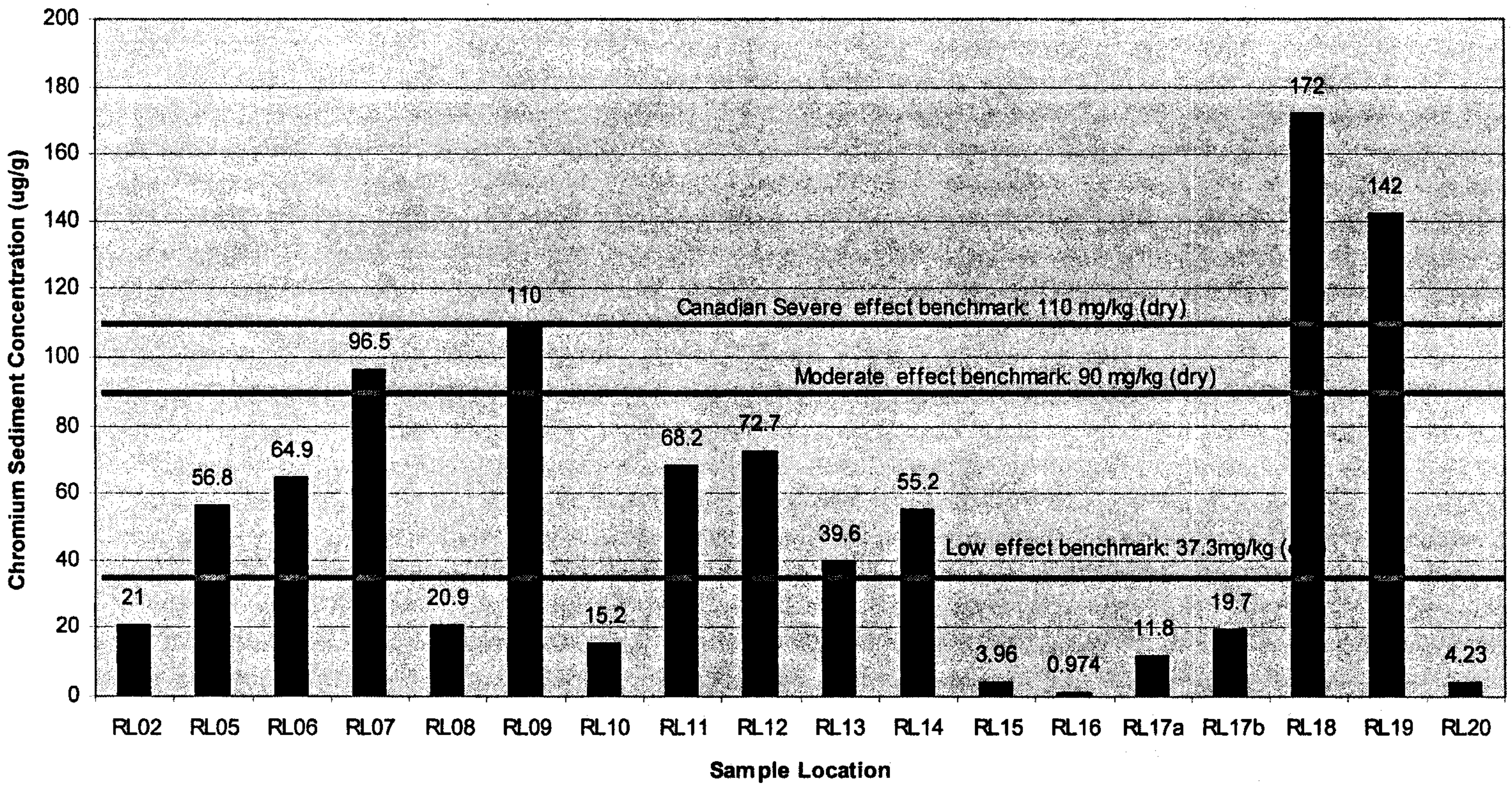

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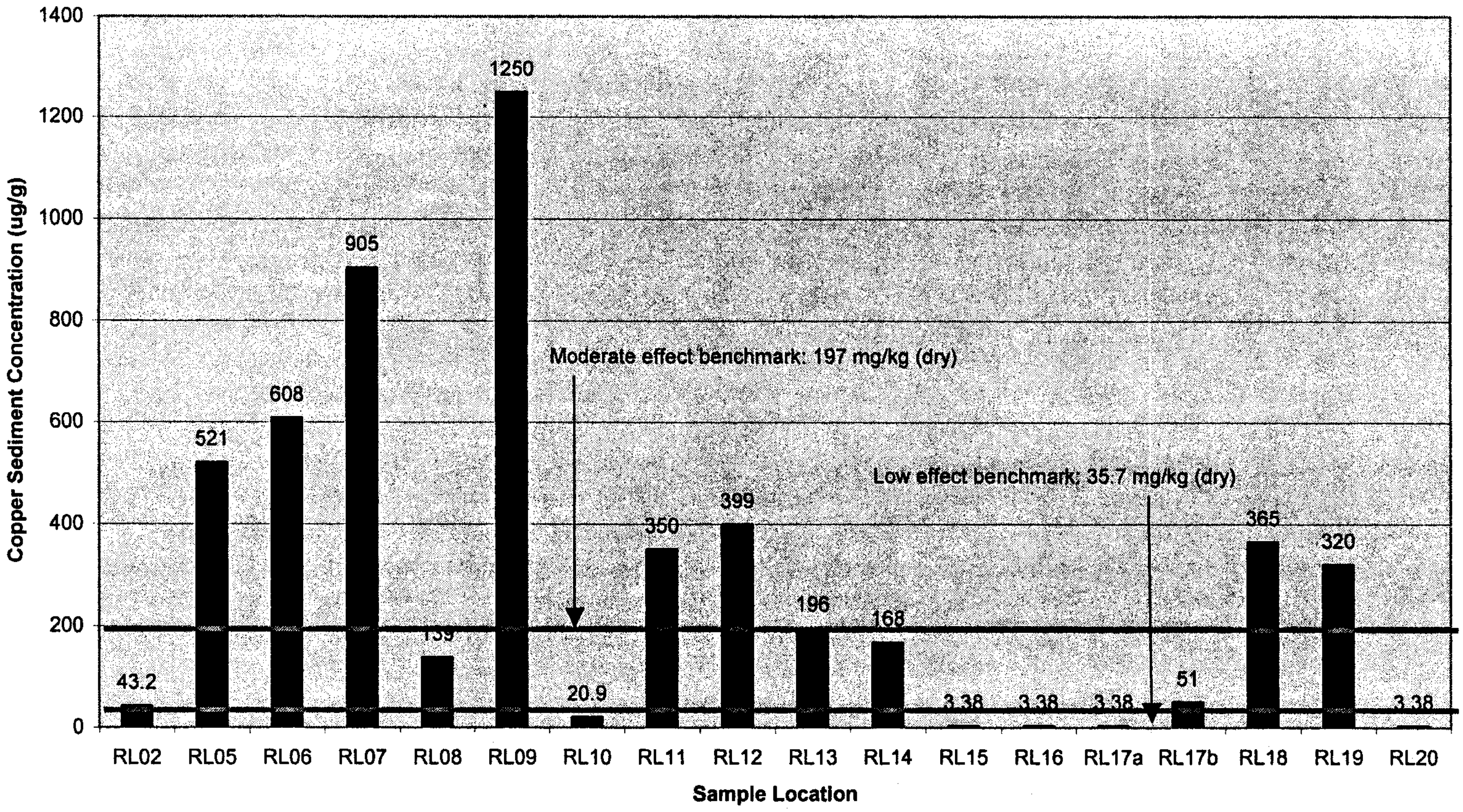

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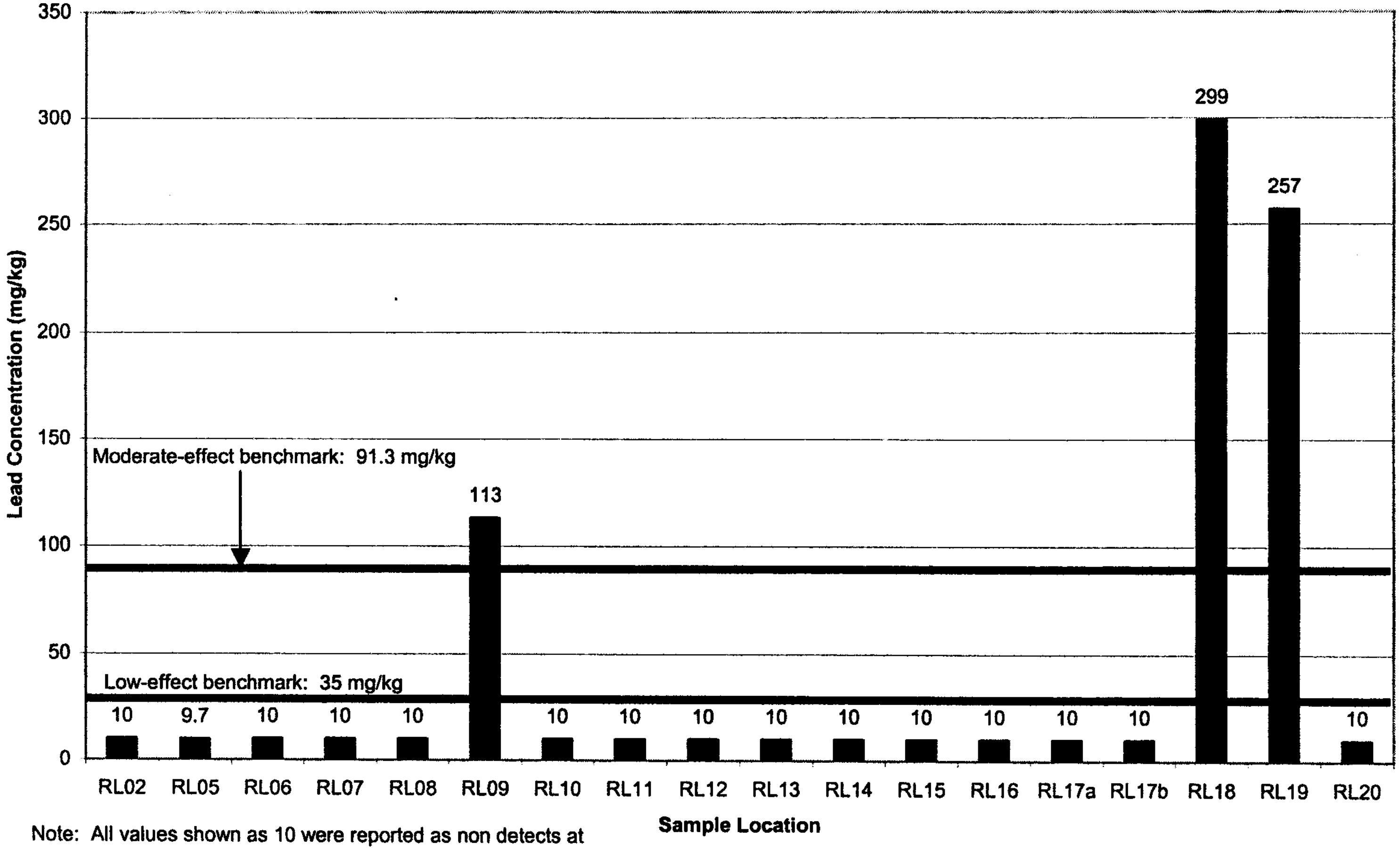

Note: All values shown as 10 were reported as non detects at <10 mg/kg. Values were taken from the OU-2 FS.

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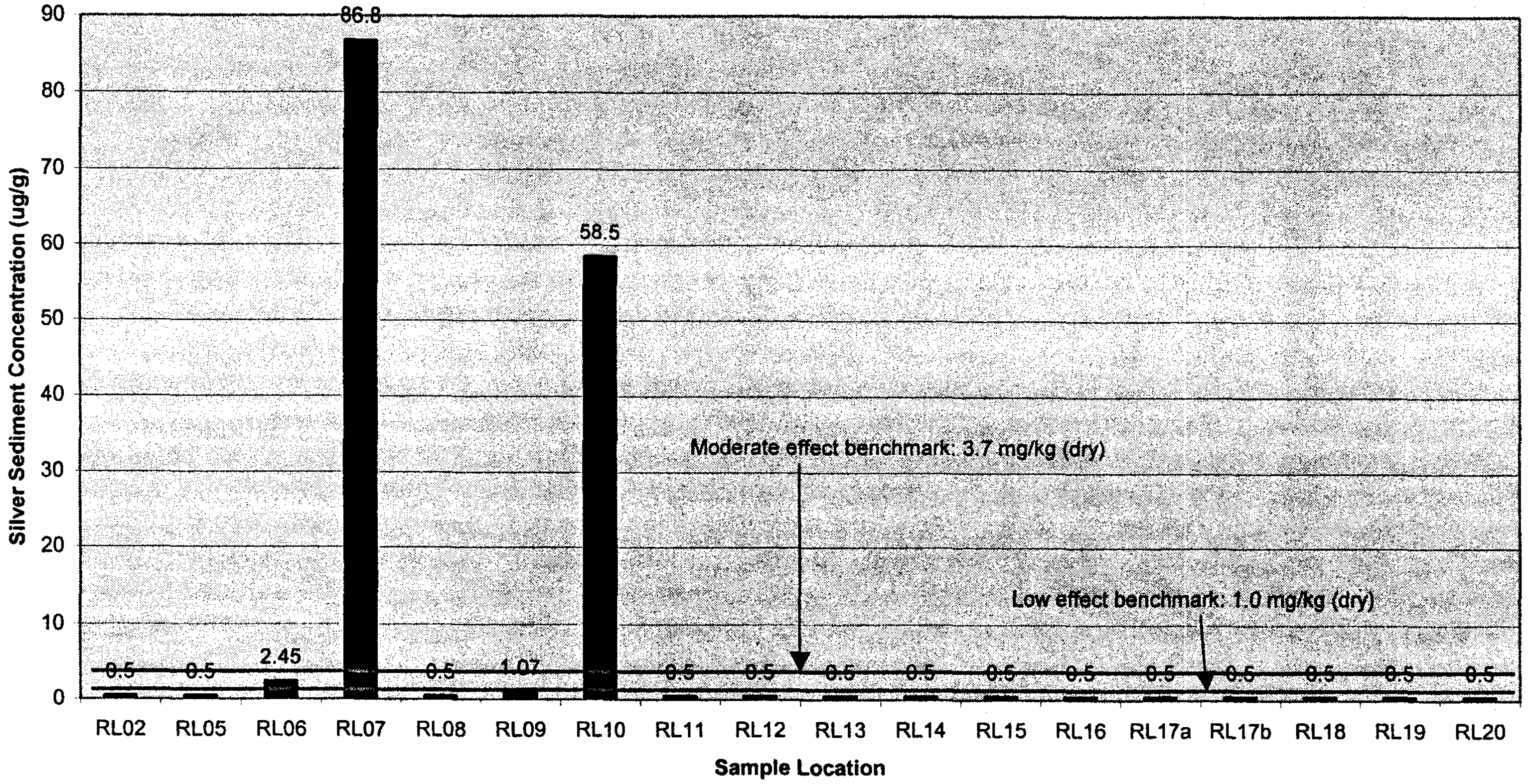

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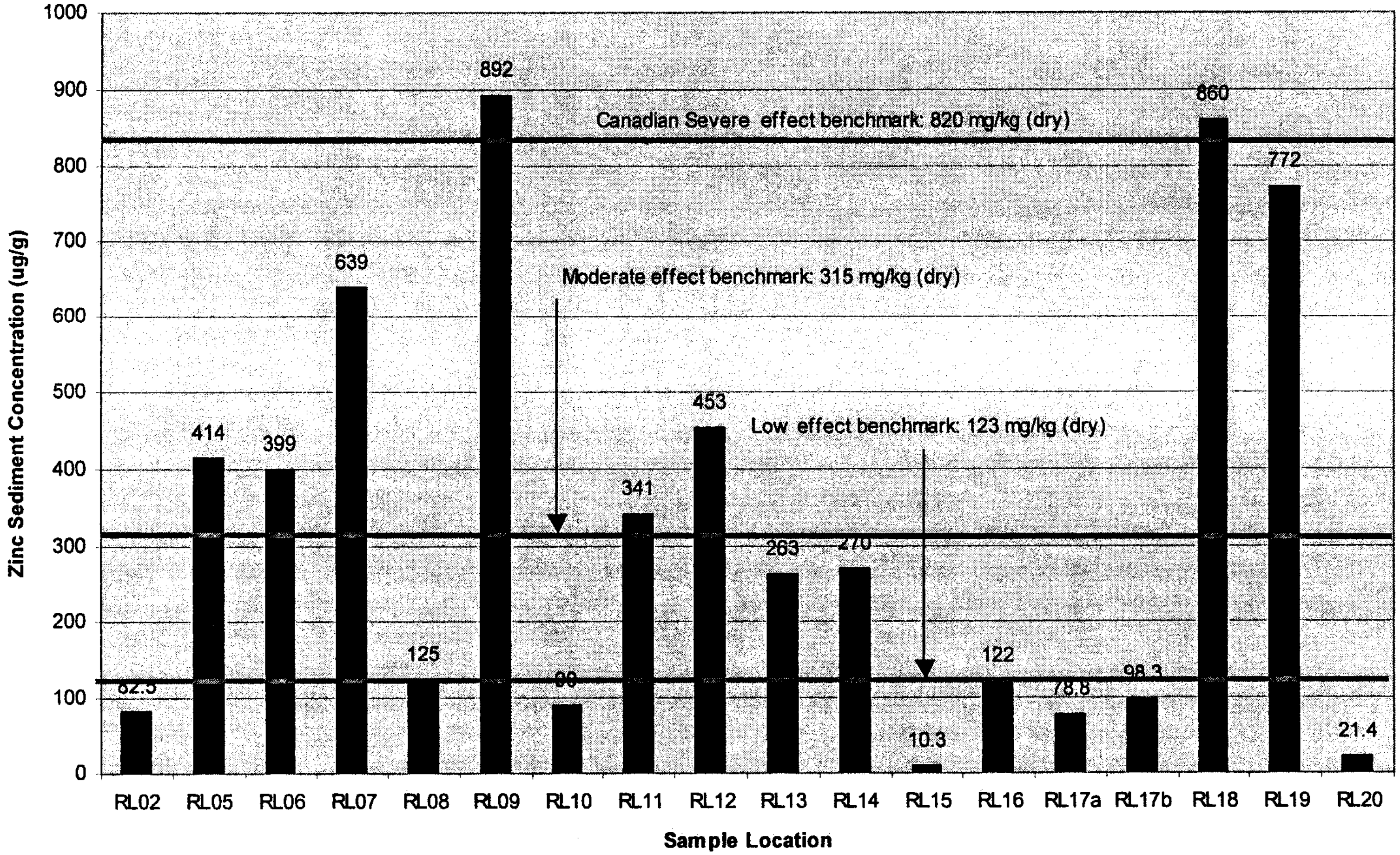

Marsden Lake North Allometrically Scaled HQ matrix

		Muskrat		Mink		Mallard		Heron		Heron 10% sed	
		HQ1	HQ2	HQ1	HQ2	HQ1	HQ2	HQ1	HQ2	HQ1	HQ2
Aluminum:											
	Maximum	10.0	2.0	na	na	2.6	nd	0.0	nd	0.0	ND
	Central Tendency	6.4	1.3	na	na	1.7	nd	0.0	nd	0.0	ND
Antimony:											
	Maximum	9.5	1.9	0.0	NM	ND	ND	ND	ND	ND	ND
	Central Tendency	0.1	0.0	NM	NM	ND	ND	ND	ND	ND	ND
Arsenic:											
	Maximum	0.9	0.2	NM	NM	0.1	0.0	0.0	0.0	0.0	0.0
	Central Tendency	0.5	0.1	NM	NM	0.0	0.0	0.0	0.0	0.0	0.0
Barium:											
	Maximum	0.9	0.3	na	na	0.2	0.2	0.1	0.04	0.1	0.0
	Central Tendency	0.5	0.2	na	na	0.12	0.09	0.0	0.02	0.0	0.0
Lead:											
	Maximum	0.0	0.0	NM	NM	4.24	0.45	0.0	0.00	0.1	0.0
	Central Tendency	0.0	0.0	NM	NM	2.35	0.25	0.0	0.00	0.0	0.0
Manganese:											
	Maximum	0.2	0.1	na	na	0.006	ND	0.0	ND	0.0	ND
	Central Tendency	0.1	0.0	na	na	0.003	ND	0.0	ND	0.0	ND
Mercury:											
	Maximum	0.1	0.0	1.8	0.4	0.05	0.009	0.1	0.01	0.1	0.0
	Central Tendency	0.1	0.0	0.7	0.1	0.04	0.007	0.0	0.004	0.0	0.0
Vanadium:											
	Maximum	0.8	0.2	NM	NM	0.07	ND	ND	ND	ND	ND
	Central Tendency	0.6	0.1	NM	NM	0.06	ND	ND	ND	ND	ND
Zinc:											
	Maximum	0.0	0.0	NM	NM	0.29	0.032	0.04	0.004	0.0	0.0
	Central Tendency	0.0	0.0	NM	NM	0.15	0.017	0.01	0.001	0.0	0.0

Marsden Lake South Allometrically Scaled HQ Matrix

	Muskrat		Mink		Mallard		Heron (no sed ingestion)		Heron (10% sed ingestion)	
Aluminum:	HQ1	HQ2	HQ1	HQ2	HQ1	HQ2	HQ1	HQ2		
Maximum	22	5	NM	NM	8	nd	0.1	nd	0.1	ND
Central Tendency	15	3	NM	NM	5	nd	0	nd	0.1	ND
Cadmium:										
Maximum	0.1	0	NM	NM	0	0	0	0	0	0
Central Tendency	0	0	NM	NM	0	0	0	0	0	0
Lead:										
Maximum	0	0	NM	NM	13	1	0.1	0	0.3	0
Central Tendency	0	0	NM	NM	9	1	0.1	0	0.2	0
Manganese:										
Maximum	0.3	0.1	NM	NM	0	ND	0	ND	0	ND
Central Tendency	0.2	0.1	NM	NM	0	ND	0	ND	0	ND
Vanadium:										
Maximum	0.8	0.2	NM	NM	0.1	ND	ND	ND	ND	ND
Central Tendency	0.6	0.1	NM	NM	0.1	ND	ND	ND	ND	ND

Figure B4 – 31. Sunfish Lake Allometrically Scaled HQ Matrix

	Muskrat		Mink		Heron		Heron 10% sed	
Aluminum:	HQ1	HQ2	HQ1	HQ2	HQ1	HQ2	HQ1	HQ2
Maximum	26.3	5.3	NM	NM	0.0	nd	0.0	ND
Central Tendency	18.5	3.7	NM	NM	0.0	nd	0.0	ND
Cadmium:								
Maximum	0.1	0.0	NM	NM	0.0	0.0	0.0	0.0
Central Tendency	0.0	0.0	NM	NM	0.0	0.0	0.0	0.0
Chromium:								
Maximum	3.4	1.3	NM	NM	0.0	0.0	0.0	0.0
Central Tendency	1.8	0.7	NM	NM	0.0	0.0	0.0	0.0
Lead:								
Maximum	0.0	0.0	NM	NM	0.0	0.0	0.2	0.0
Central Tendency	0.0	0.0	NM	NM	0.0	0.0	0.1	0.0
Mercury:								
Maximum	0.1	0.0	1.3	0	0.02	0.0	0.02	0.0
Central Tendency	0.1	0.0	1.0	0	0.01	0.0	0.01	0.0
Vanadium:								
Maximum	1.3	0.3	NM	NM	0.0	ND	0.0	ND
Central Tendency	1.0	0.2	NM	NM	0.0	ND	0.0	ND
Zinc:								
Maximum	0.1	0.1	NM	NM	0.0	0.0	0.0	0.0
Central Tendency	0.1	0.0	NM	NM	0.0	0.0	0.0	0.0

Note: As a sensitivity analysis, doses were calculated for the Heron assuming a 0% sediment ingestion and 10% (maximum) sediment ingestion.

Round Lake Allometrically Scaled HQ Matrix

	Muskrat HQ1	Muskrat HQ2
Barium:		
Maximum	2	0.7
Central Tendency	1	0.5
Cadmium		
Maximum	0.5	0
Central Tendency	0	0
Chromium:		
Maximum	2	0.8
Central Tendency	1	0.4
Copper:		
Maximum	5	3
Central Tendency	0.9	0.6
Lead:		
Maximum	0.1	0
Central Tendency	0	0
Silver:		
Maximum	150	30
Central Tendency	53	11
Vanadium:		
Maximum	1	0.3
Central Tendency	0.7	0.1
Zinc:		
Maximum	0.2	0.1
Central Tendency	0.1	0.1

APPENDIX C

Photodocumentation

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Photo 1. Daily calibration of YSI prior to sampling

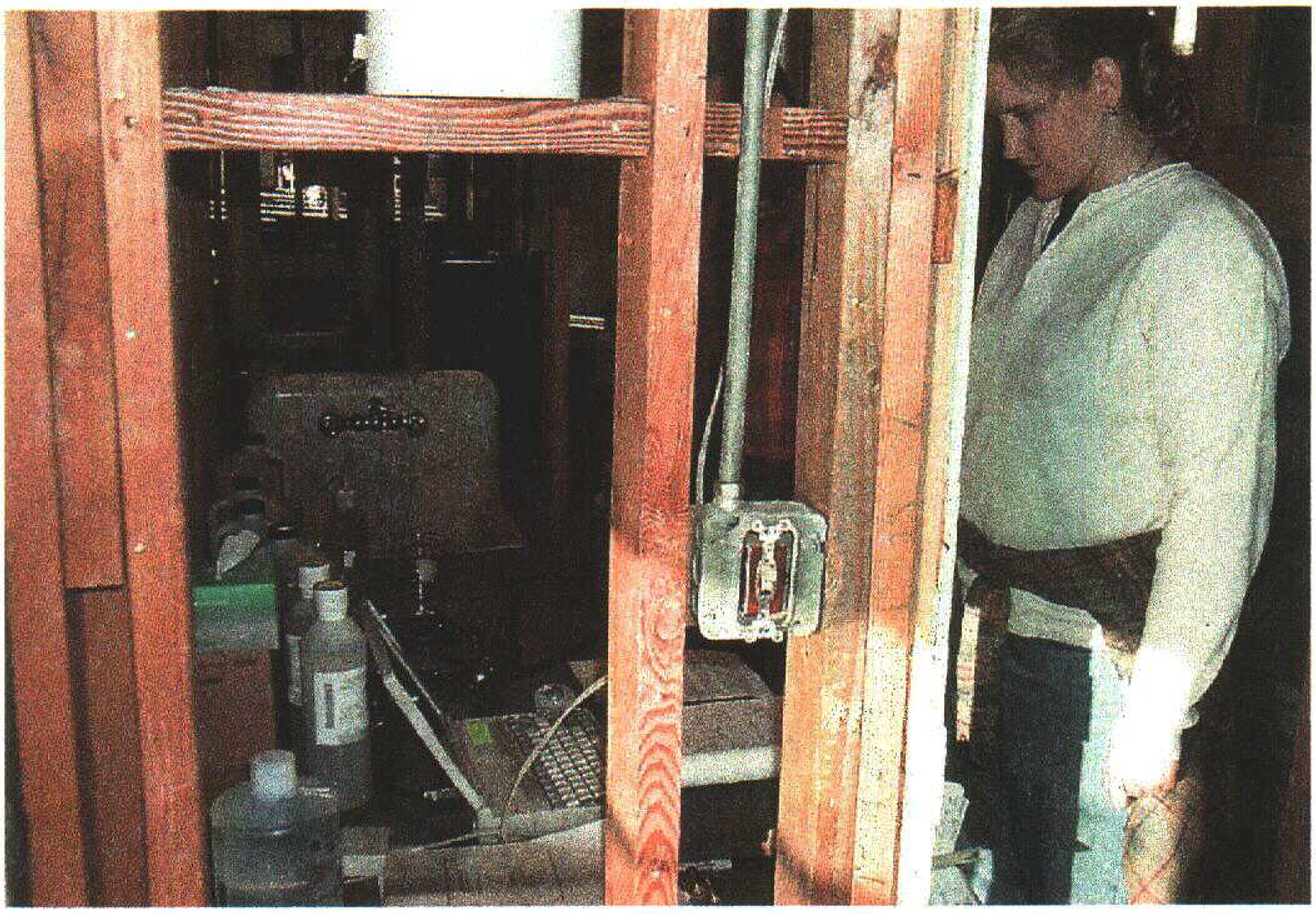


Photo 2. Preparation of boat and sampling equipment at Snail Lake, fall 1999

Photo 3. Launching boat at Marsden Lake, south area, fall 1999

Photo 4. Collecting water sample at Marsden Lake, north area, site 18, fall 1999

Photo 5. Hole cut in cattails with steel machete at Marsden Lake, site 20, fall 1999

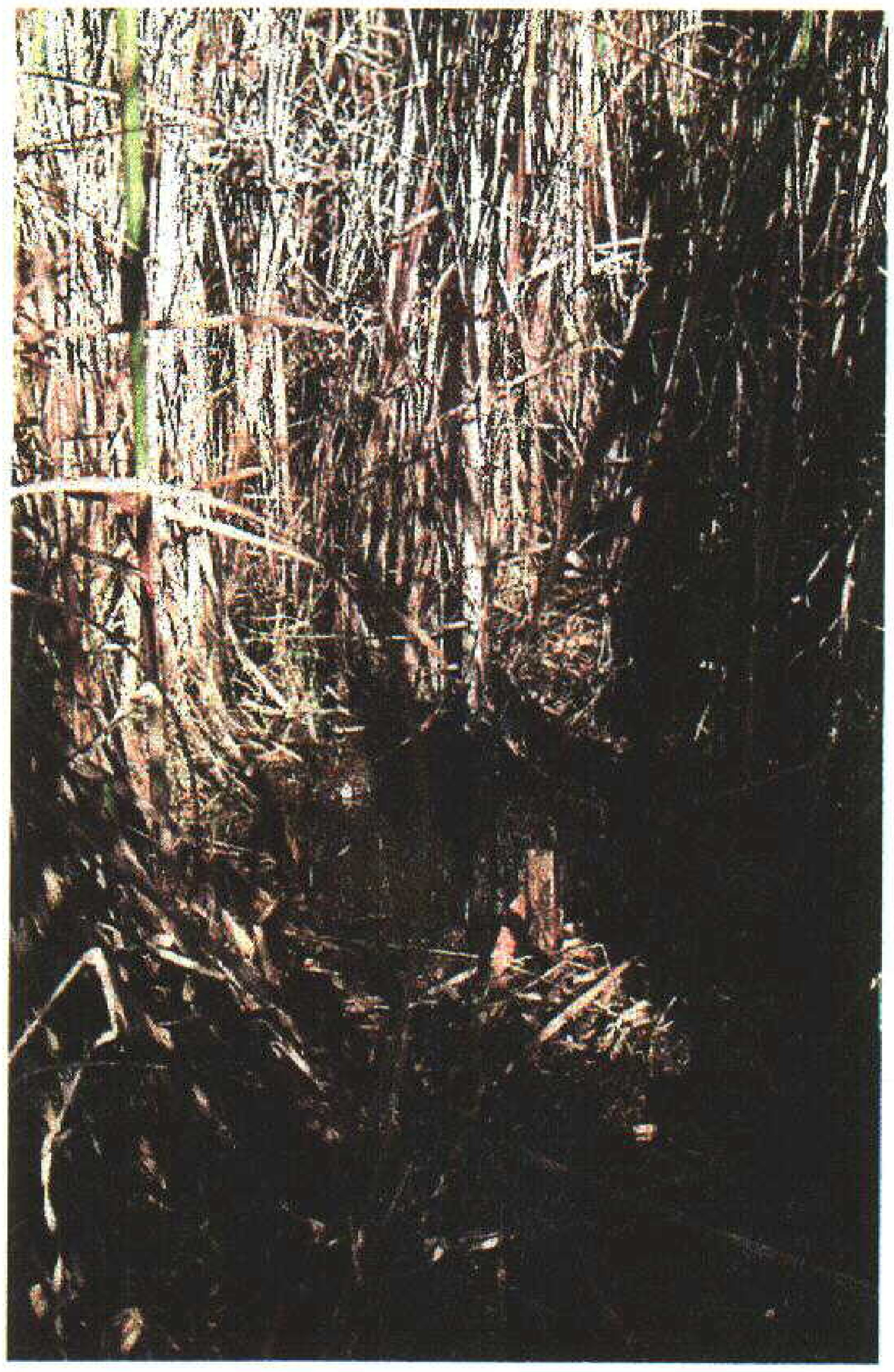


Photo 6. Collection of sediment samples at Snail lake, fall 1999

Photo 7. Round Lake, site 4, summer 2000, collecting water sample

Photo 8. Snail Lake, site 8, fall 1998. Double-bagging water metals sample after collection

Photo 9. Pond G, fall 1999

Photo 10. Rice Creek, summer 2000, collecting water sample

Photo 21. Marsden Lake, south area

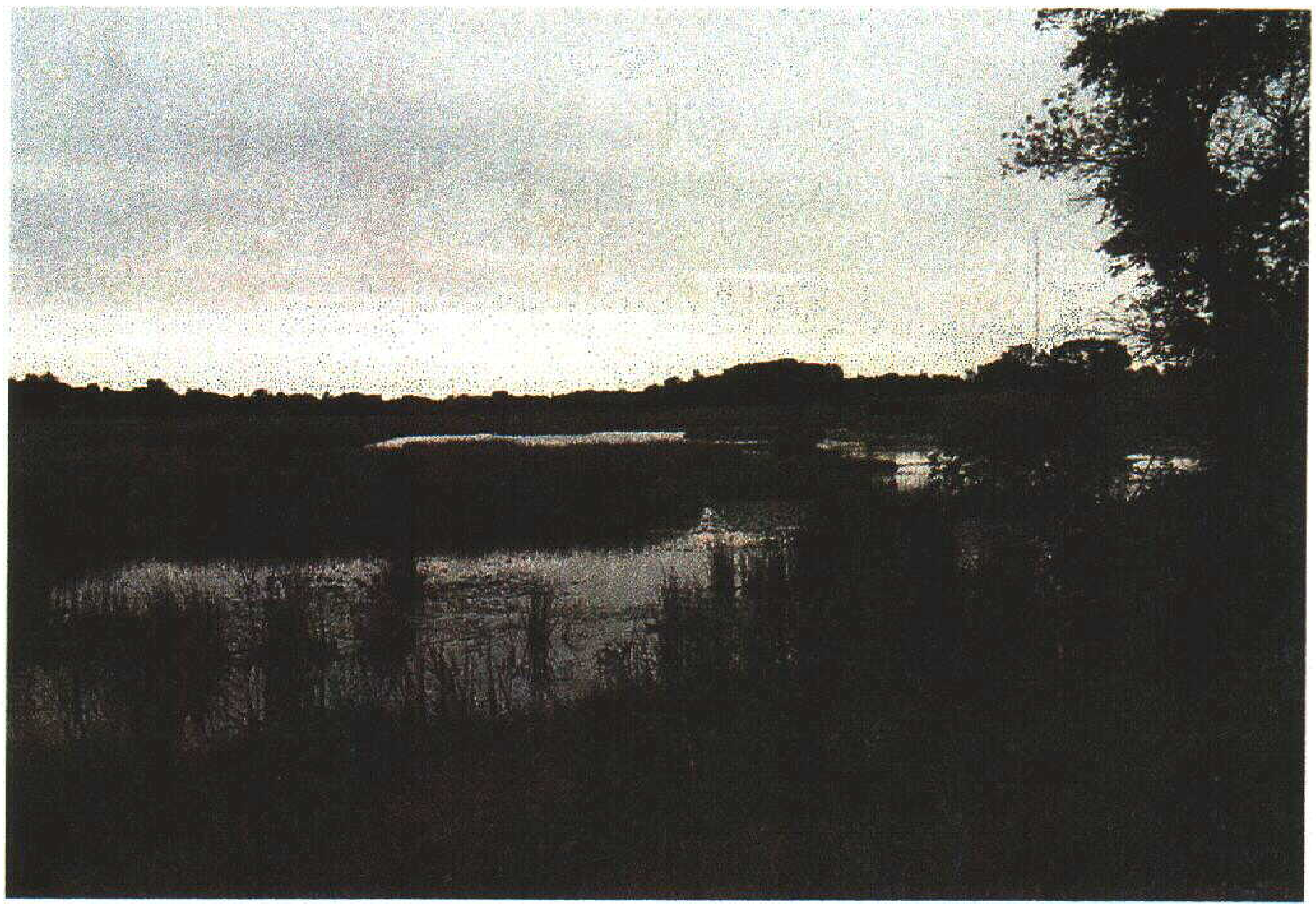


Photo 22. Marsden Lake, north area, south of road

Photo 23. Marsden, north area, looking towards site 20

Photo 24. Marsden, north area, site 11, fall 1999.

Photo 25. Marsden Lake, site 11, north area, fall 1999.

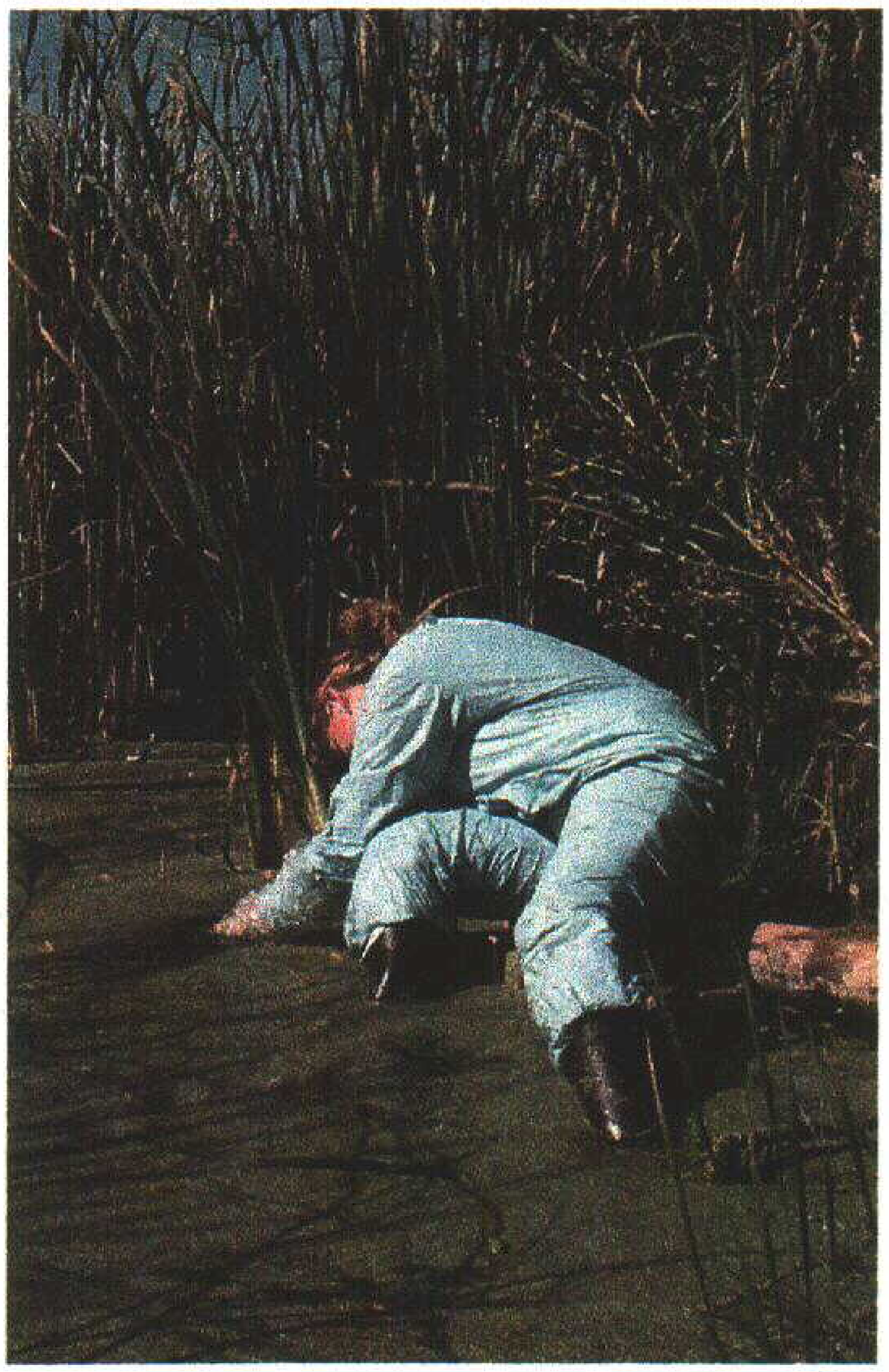


Photo 26. Round Lake, fall 1999, access point.

Photo 27. Round Lake, spring 2000.

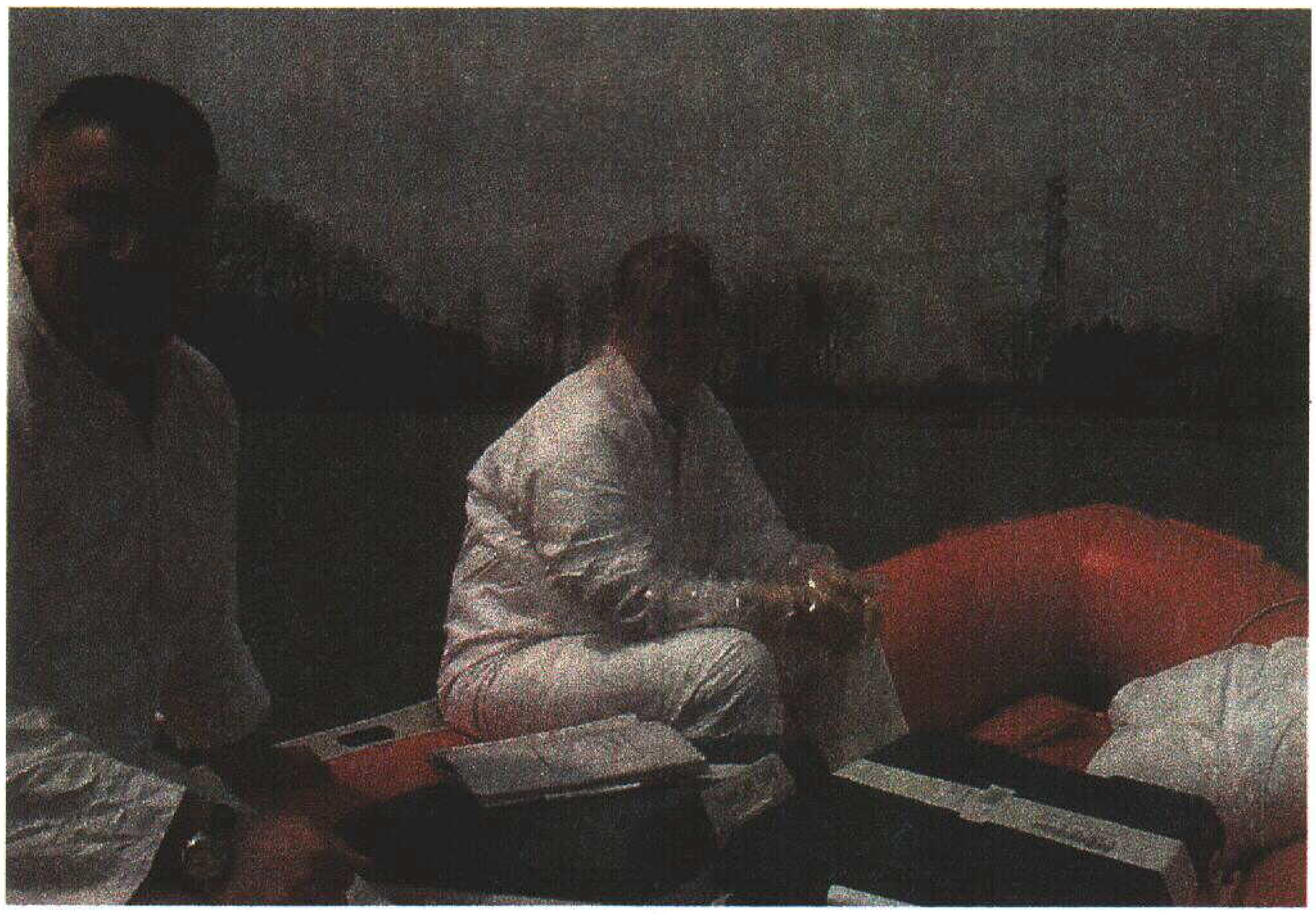


Photo 28. Rice Creek, site 1, spring 2000.

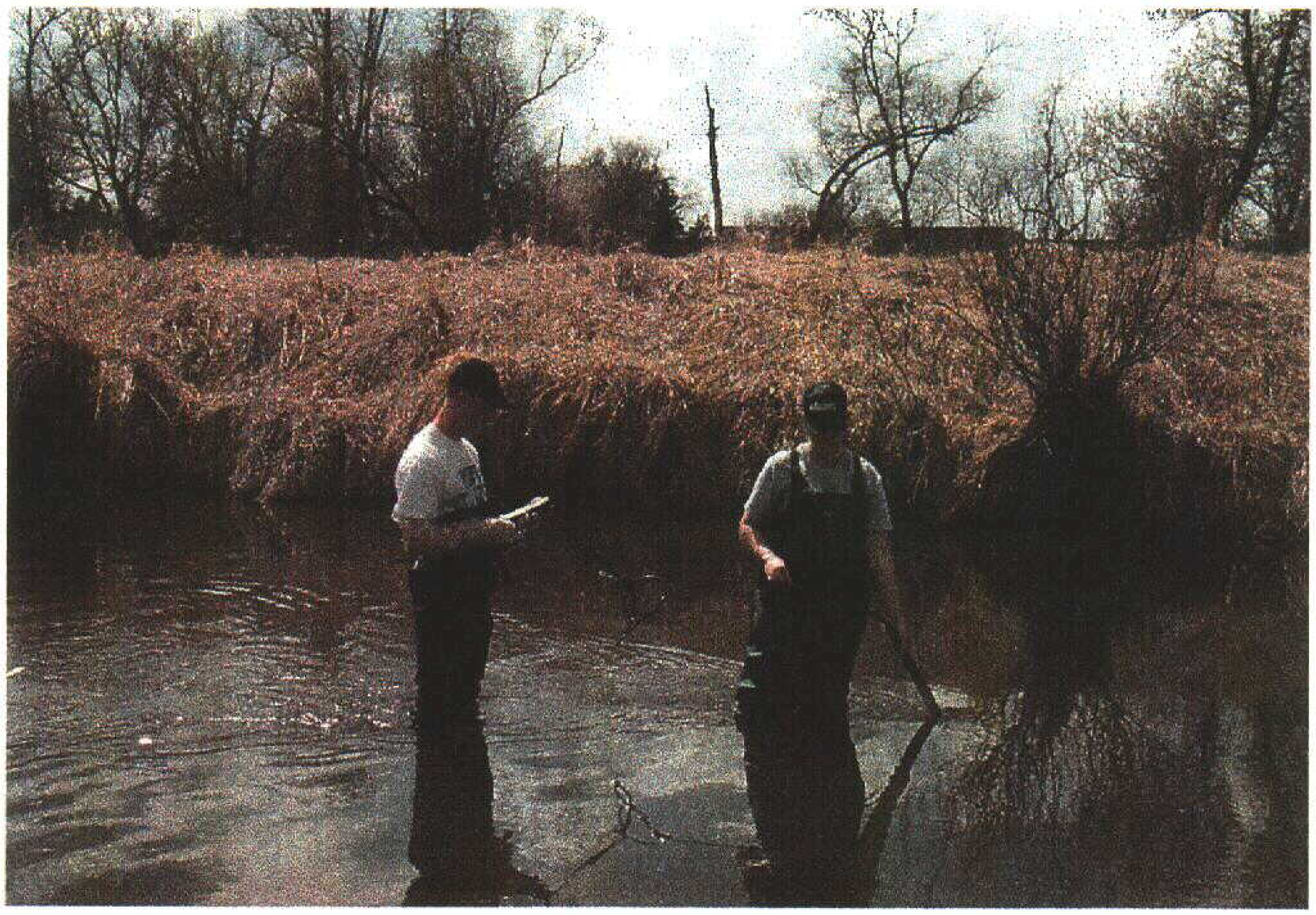


Photo 29. Sunfish, site 3, fall 1999, collecting water sample.

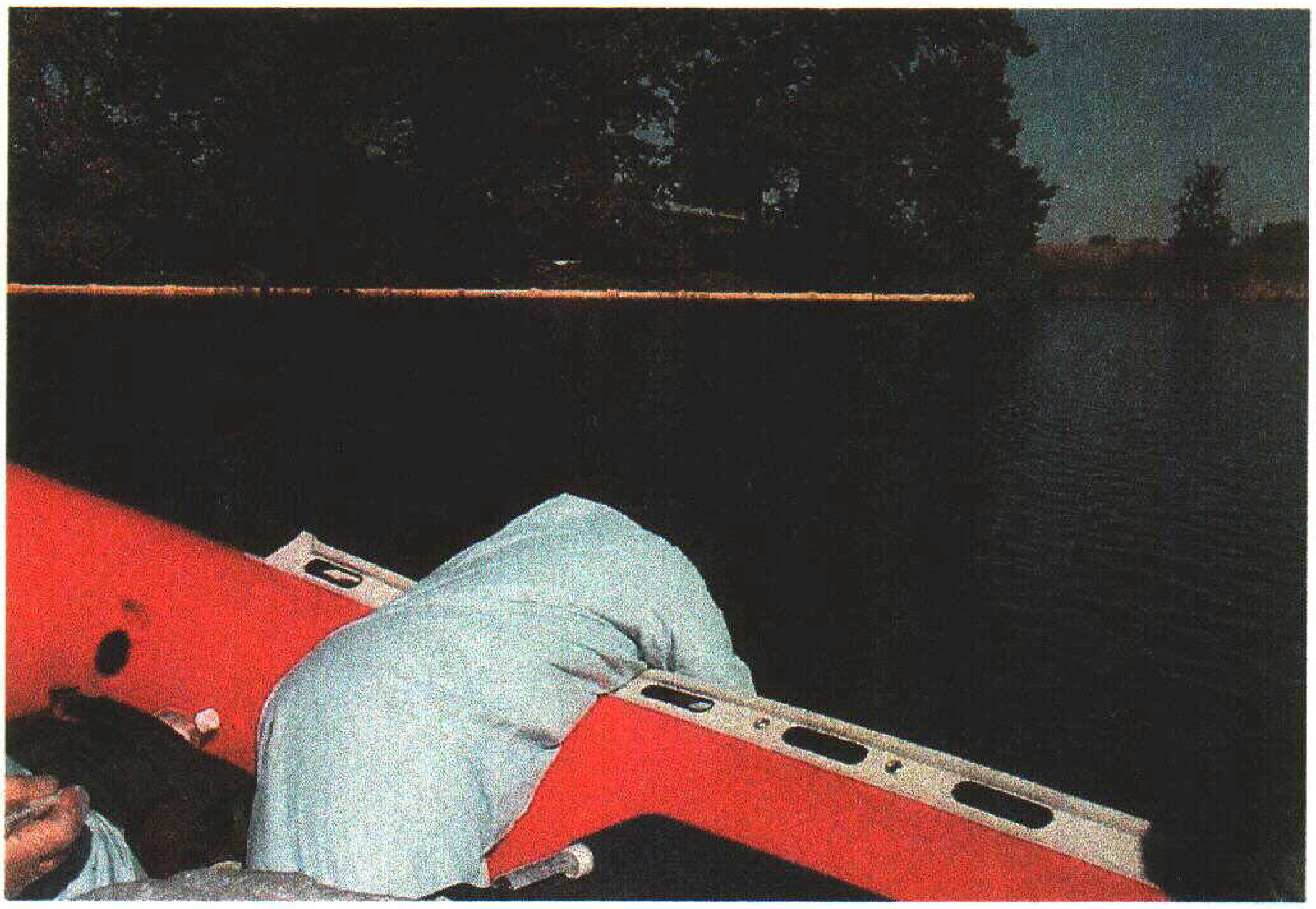


Photo 30. Sunfish, fall 1999, excavation of Site H.

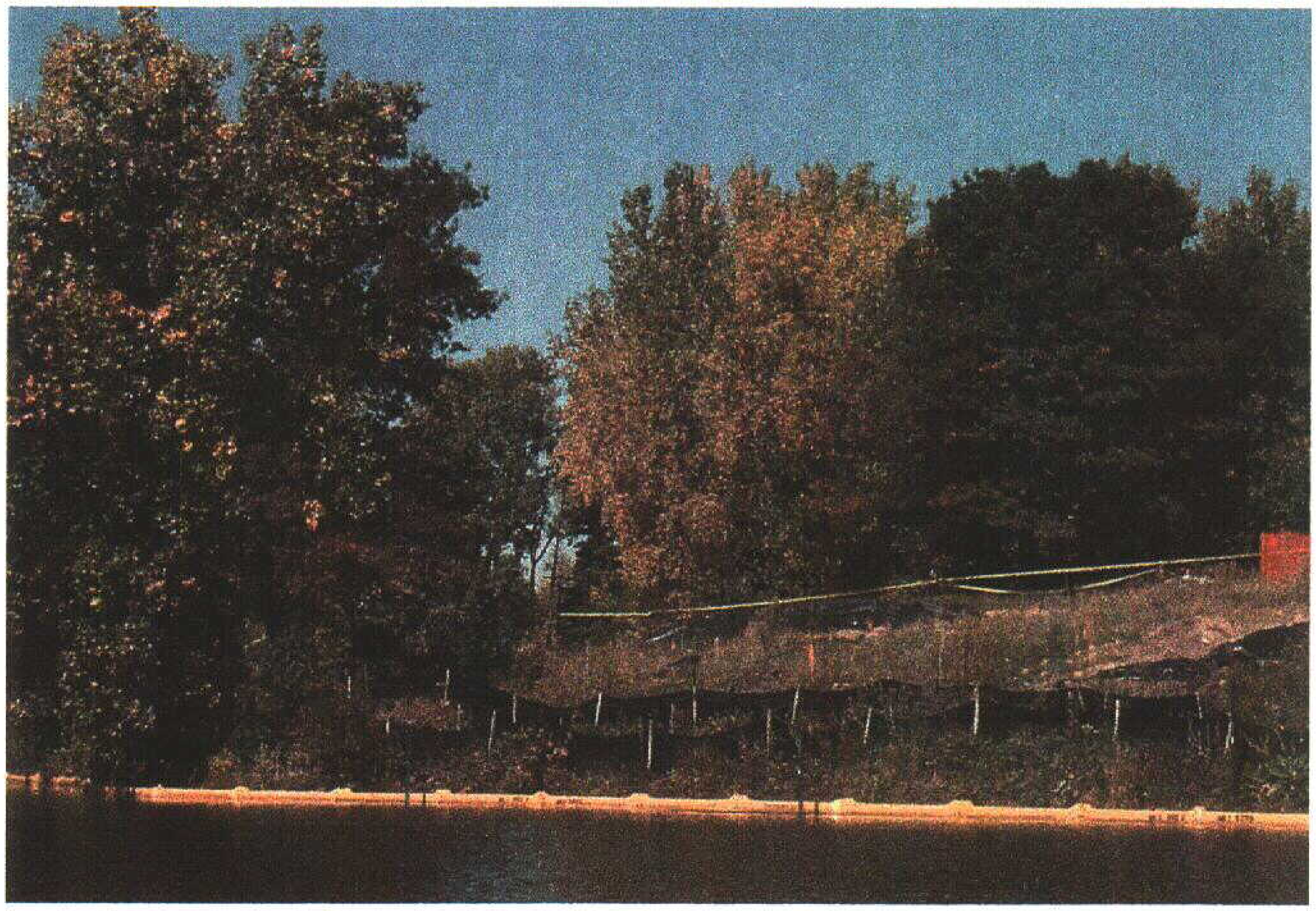


Photo 31. Sunfish, spring 2000, excavation of site H.

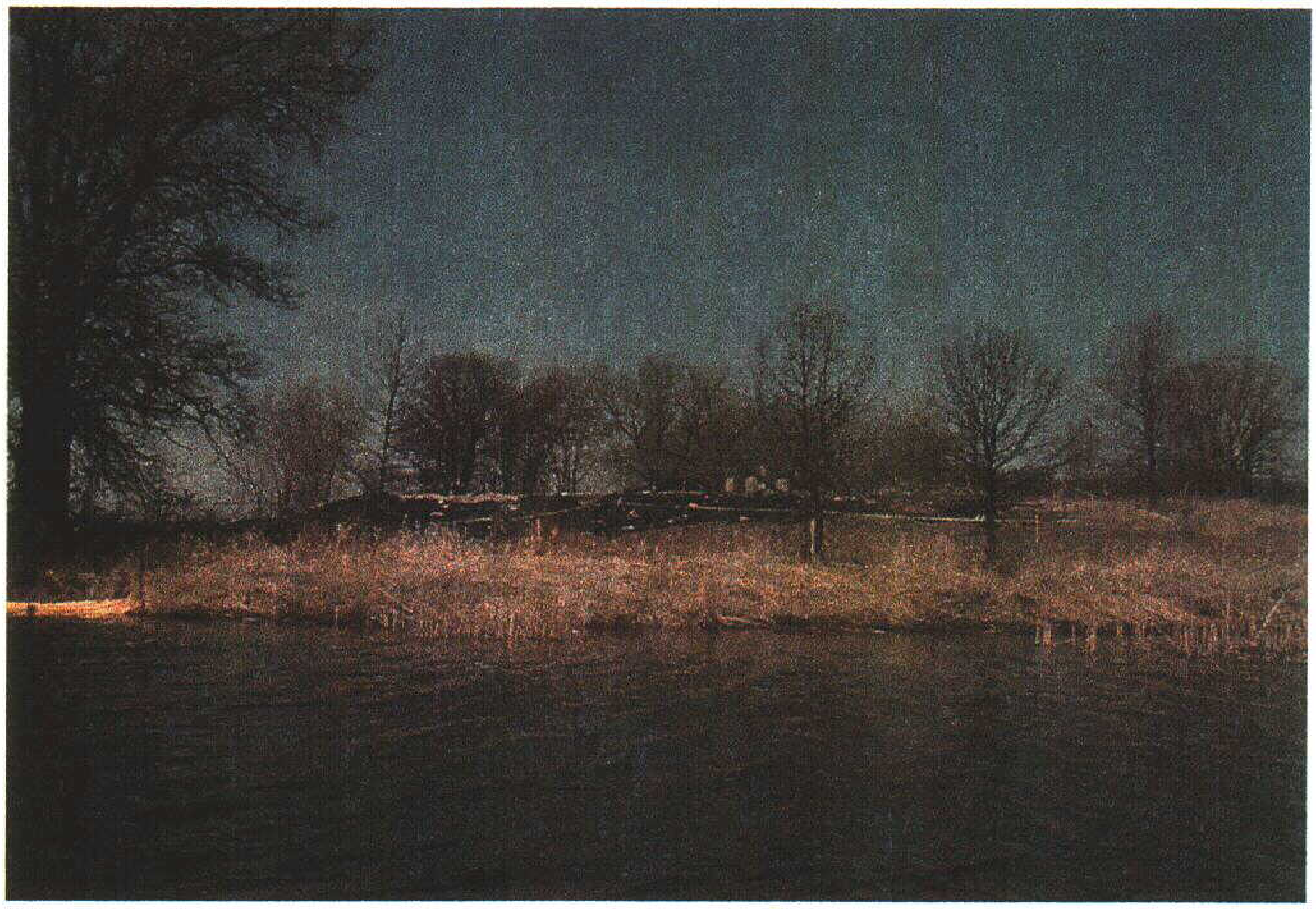
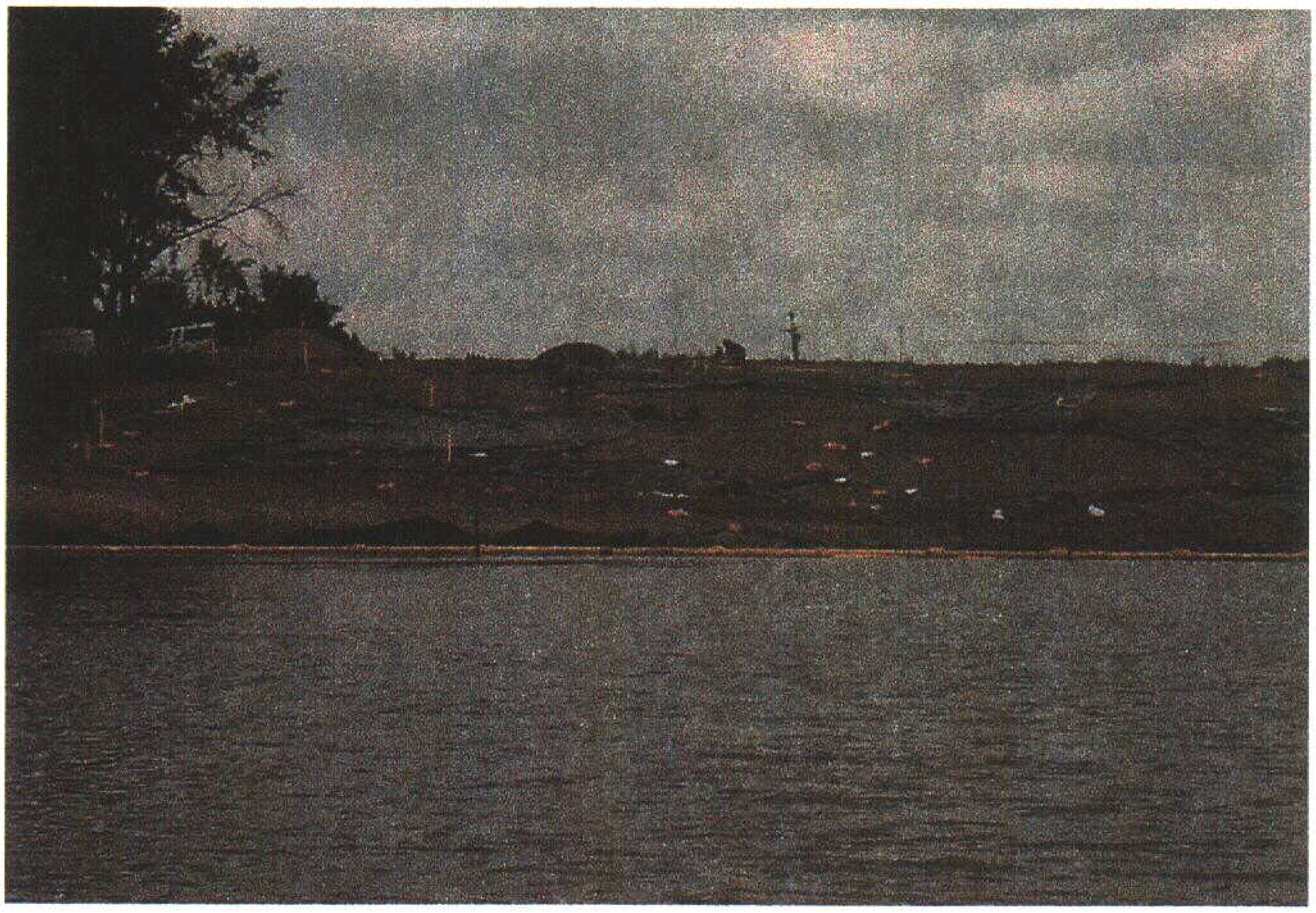


Photo 32. Sunfish Lake, summer 2000, excavation of Site H.



APPENDIX D

Marsden Lake (North) Sampling Locations and Analytical Measurement Results

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TABLES

FIGURES

Marsden Lake (North) Data Presentation Summary

The map of Marsden Lake (North) (Fig.D1), shows the location where each media sample was taken. Global Positioning System (GPS) coordinates were used to plot the locations. These coordinates recorded in the field, during sampling, are reported in tables D1 and D2. The map image is from an orthophoto taken in 1997, obtained from Wenck Associates, Inc., Maple Plain, Minnesota.

The tables D3 through D8 summarize the data for Marsden Lake (North) surface water and sediment samples taken during the site characterization. The summaries include the field and analytical result for each analyte at each sample location. The minima and maxima are reported as well as the detection frequency (DF), risk based action level (RBAL), target detection limit and achieved detection limit. The RBALs and target detection limits are those that were defined in the QAPP. Summary data used for statistical analysis such as the central tendencies, standard deviations and confidence intervals are report in tables found in Appendix A.

The number of digits reported for each value in the following tables does not necessarily reflect the precision and accuracy of the measurement. The data tables present significant figures as they were originally reported by the analyzing laboratory. Significant figures reported for the analytes were based on the analytical detection limit values. However, some sample values have more significant figures than others to demonstrate that blank correction was discontinued, as was requested by USEPA comments on the draft version of this report (see Appendix R). Data quality is discussed in Appendix K.

The results are reported in the following manner to facilitate data review.

- If the detection limit is sample-specific (as it was for the pesticides in sediment), the value reported for the achieved detection limit for a site is the highest detection limit for that parameter for all samples (including the field duplicate) from that site.

- If a parameter was not detected in any samples from a site (e.g. Pond G, water, alpha-BHC) then the minimum and maximum values represent the lowest and highest achieved detection limits for those data.

- Any concentrations that were below the detection limit but still quantified are identified in Volume 2 of this report as estimated values (J-flagged). These values are reported and used uncensored in the following tables.

- If a parameter was undetected in at least one but not all samples from a site, then the minimum reflects the lowest detection limit of the undetected samples. If there was a J-flagged value lower than the highest undetected value, the J-flagged value was reported as the maximum because the value was actually detected but estimated.

- Field duplicates (FD) were used as quality control samples and were not included in the reported minima and maxima columns. However, if an analyte was detected in the FD and not in it's corresponding sample, then the FD detection was counted in the detection frequency column (only in the numerator).

- When laboratory replicate samples (R1, R2) were available, the values from the primary sample (R1) were reported. See Appendix K section K1.2.1 for explanation.

Definition of Data Qualifiers (Flags)

During the data validation process, all laboratory data were assigned appropriate data validation flags and reason codes. Validation flags used in the following tables are defined as follows:

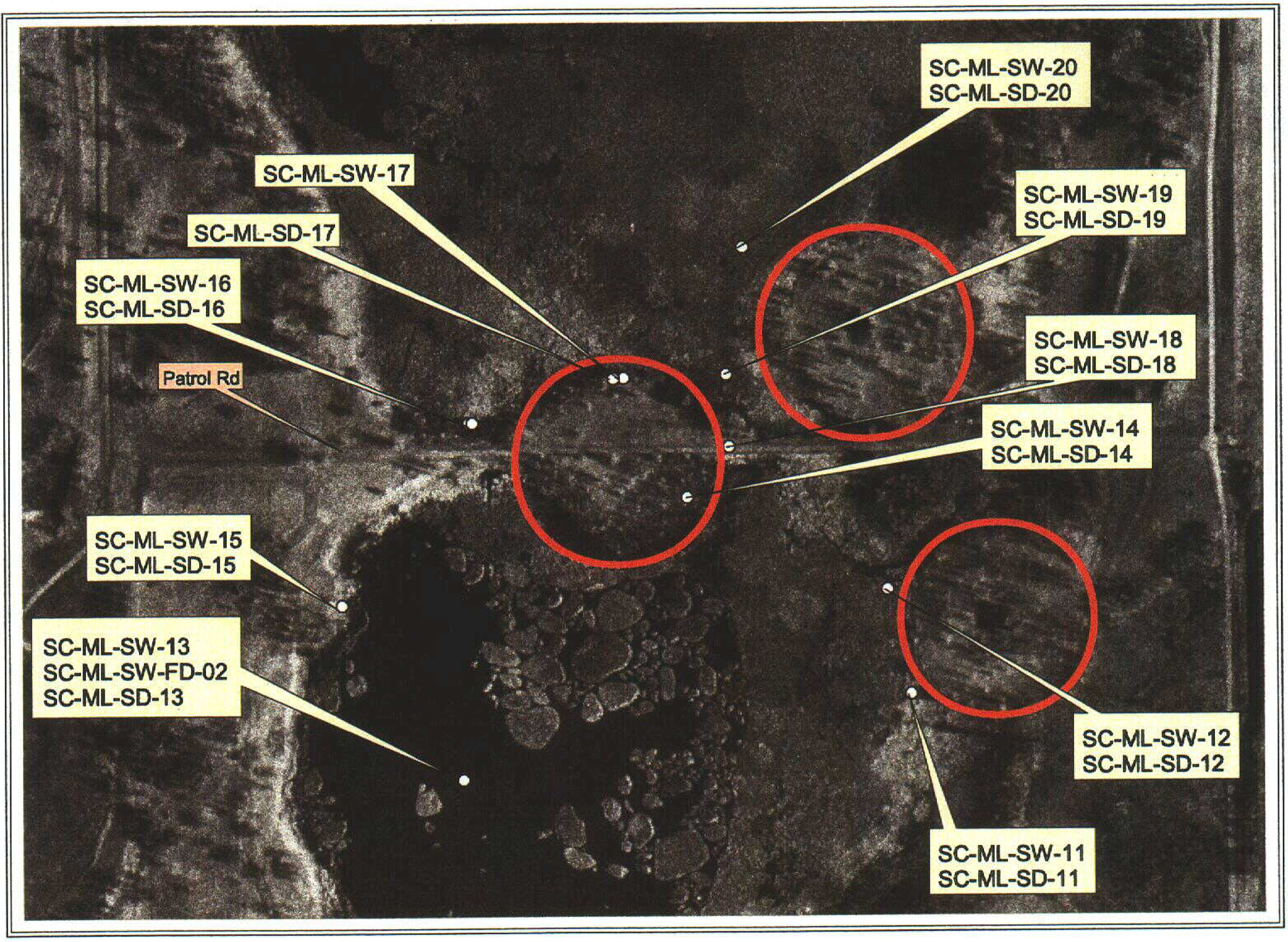
No flag	When the analyte value reported is detected and usable. The integrity of the analyte's identification, accuracy, and precision has been validated
"U"	When the material was analyzed for, but not detected above the level of the associated detection limit value.
"UB"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with.
"J"	When the associated value is an estimated quantity. Indicating there is cause to question accuracy or precision of the reported value. The value is still used in the risk assessment.
"UJ"	When the analyte was analyzed for, but not detected, above the associated value, however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"UBJ"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with; however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"R"	When the analyte value reported is unusable. The integrity of the analyte's identification, accuracy, precision, or sensitivity has raised significant question as to the reality of the information presented.

Other information regarding data validation can be found in Volume 2 (CD version) of the report.

Outlying data points

Figures D2 through D19 show a crude comparison of the distance from the center of the closest impact area to a sample point, versus analyte concentration at the sample point. This information was used to determine if the locations of points ML-13 and ML-15 were unrepresentative of the site (refer to section 4.1.1 of the ERA report)

Figure D1. Marsden Lake (North) Water and Sediment Sampling Locations



SC= Site characterization
ML= Marsden Lake
SW= Surface water
SD= Sediment
FD= Field duplicate
○ =Grenade Range Impact Areas

0.08 0 0.08 0.16 Miles

Table D1. Marsden Lake (North) Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	North*			West*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-ML-SW-11	24-Sep-99	4"	45	6	0.55	93	9	2.61
SC-ML-SW-12	26-Sep-99	2-4"	45	6	2.89	93	9	3.36
SC-ML-SW-13 SC-ML-SW-FD-02	22-Sep-99	<6"	45	5	58.65	93	9	16.64
SC-ML-SW-14	26-Sep-99	4"	45	6	4.96	93	9	9.65
SC-ML-SW-15	22-Sep-99	4"	45	6	2.55	93	9	20.43
SC-ML-SW-16	21-Sep-99	4"	45	6	6.63	93	9	16.35
SC-ML-SW-17	21-Sep-99	4"	45	6	7.62	93	9	11.63
SC-ML-SW-18	20-Sep-99	4 - 6"	45	6	6:08	93	9	8.34
SC-ML-SW-19	20-Sep-99	Na	45	6	7.65	93	9	8.42
SC-ML-SW-19	21-Sep-99	3" (hole cut in cattails)	45	6	7.7	93	9	8.42
SC-ML-SW-20	21-Sep-99	Na	45	6	10.57	93	9	7.91

Na: Not available

Table D2. Marsden Lake (North) Sediment Sampling GPS Coordinates

Sample Number	Date	Sample depth	North*			West*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-ML-SD-11	24-Sep-99	6"	45	6	0.55	93	9	2.61
SC-ML-SD-12	26-Sep-99	1'	45	6	2.89	93	9	3.36
SC-ML-SD-13	22-Sep-99	4-6"	45	5	58.65	93	9	16.64
SC-ML-SD-14	26-Sep-99	3'	45	6	4.96	93	9	9.65
SC-ML-SD-15	22-Sep-99	4-6"	45	6	2.55	93	9	20.43
SC-ML-SD-16	21-Sep-99	13"	45	6	6.63	93	9	16.35
SC-ML-SD-17	20-Sep-99	1'	45	6	7.61	93	9	11.96
SC-ML-SD-18	20-Sep-99	8"	45	6	6.08	93	9	8.34
SC-ML-SD-19	21-Sep-99	6" (hole cut in cattails)	45	6	7.7	93	9	8.42
SC-ML-SD-20	21-Sep-99	NA	45	6	10.57	93	9	7.91

NA: not available

*The GPS unit used in the study was a military-issue Precision Light Weight GPS Receiver (PLGR). The Department of Defense (DoD) intentionally places errors in the GPS navigation and timing signal, in a process called selective availability (Department of the Army 1993). Errors resulting from this process are unpredictable and can produce significant horizontal and elevation errors. Military GPS receivers must have crypto keys loaded to detect and nullify selective availability errors, in order to allow for more accurate position data (Department of the Army 1993). The PLGR was equipped with a crypto key, and was therefore not subject to the selective availability and spoofing errors that affect non-military GPS. (The only exception was during the second quarter). The PLGR operations manual (Department of the Army 1995) states that with crypto keys installed, the PLGR provides horizontal 2-D position accuracies better than the worst case error of 10 meters.

Note: References cited in the text of this appendix can be found in the reference section of the main Tier II Ecological Risk Assessment Report.

Table D3. Marsden Lake (North) Surface Water Quality and Metals Data

Parameter	SC-ML-SW-11	SC-ML-SW-12*	SC-ML-SW-14*	SC-ML-SW-16	SC-ML-SW-17	SC-ML-SW-18	SC-ML-SW-19	SC-ML-SW-20	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-FB-02	SC-ML-FD-XX-02
pH	6.46	6.23	6.77	6.62	6.92	6.61	6.91	6.51	6.23	6.92	10 / 10	na	na	na	na	na
DO (mg/L)	10.26	1.26	13.65	2.11	17.80	0.84	9.71	2.06	0.84	17.80	10 / 10	na	na	na	na	na
DO (%)	1.02	13.2	38.1	19.5	2.02	8.3	1.08	18	1.02	38.10	10 / 10	na	na	na	na	na
Conductivity (umhos/cm)	329.00	358.00	289.00	412.00	553.00	382.00	653.00	555.00	289.00	653.00	10 / 10	na	na	na	na	na
Temperature (°C)	14.46	15.65	17.24	11.68	9.76	15.61	9.22	9.34	9.22	17.24	10 / 10	na	na	na	na	na
Hardness (mg/L)	114	142	102	194	244	138	310	340	102.00	340.00	8 / 8	na	na	na	na	98.00
TKN (mg/L)	2.70	2.30	0.79	1.80	3.60	1.60	1.40	7.30	0.79	7.30	8 / 8	na	na	na	na	0.54
TOC (mg/L)	14.00	21.00	9.60	13.00	14.00	12.00	14.00	50.00	9.60	50.00	8 / 8	na	na	na	na	8.50
Total Phosphorus (mg/L)	0.41	0.29	0.05 U	0.38	0.56	0.17	0.39	0.27	<0.05	0.56	7 / 8	na	na	na	na	0.05 U
TSS (mg/L)	33.00	80.00	8.00	104.00	174.00	16.00 J	82.00	250.00	8.00	250.00	8 / 8	na	na	na	na	8.00
Metals (ug/L)																
Al	6.05	311.00	17.00	510.25	540.25	37.25	23.25	200.25	6.05	540.25	8 / 8	125.00	0.20	0.0560	8.85	4.85
Sb	0.071	0.065	0.11	0.050	0.088	0.051	0.038 U	0.21	0.038	0.21	8 / 8	31.00	0.01	0.0089	0.02	0.10
As	0.88	2.1	0.68	1.7	4.5	1.4	2.4	2.8	0.88	4.50	8 / 8	190.00	0.60	0.0440	0.04 U	0.67
Ba	43 J	62.02 J	40.02 J	92 J	200 J	63 J	160 J	130 J	40	200.00	8 / 8	3.80	0.01	0.0056	0.099	39.00 J
Be	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	<0.048	<0.048	0/ 8	5.10	0.10	0.0480	0.05 U	0.05 U
Cd	0.015 U	0.075	0.015 U	0.015 U	0.13	0.015 U	0.015 U	0.069	<0.015	0.13	3 / 8	0.66	0.01	0.0150	0.07	0.02 U
Cr	0.479	1.492 J	0.512 UB	1.749 J	2.349 J	1.039 J	1.749 J	1.449 J	0.479	1.75	7 / 8	117.00	0.03	0.0420	0.116 UB	0.749 J
Co	0.1040	1.2	0.075	0.5470	1.1070	0.2370	0.3970	1.0070	0.0750	1.20	8 / 8	5.00	0.01	0.0038	0.012 UB	1.007
Cu	0.071 U	2.617	0.237	1.2	4.1	0.25	0.58	1.6	0.07	4.10	8 / 8	6.40	0.01	0.0140	0.30	0.15
Pb	0.11	1.7	0.26	1.4	7.0	0.29	1.1	5.4	0.110	7.0	8 / 8	1.30	0.01	0.0049	1.30	0.08
Mg	6300 J	6700.49 J	7100.49 J	8900 J	19000 J	8100 J	13000 J	20000 J	6300.00	20000.00	8 / 8	82000.00	25.00	0.0500	1.60 U	6700.00 J
Mn	56	660	17	1000	1300	420	640	1100	17	1300.00	8 / 8	80.30	0.01	0.0100	0.13	19.00
Hg	0.0016	0.0062	0.0011	0.0029	0.013	0.00059	0.0028	0.011	0.00059	0.0130	8 / 8	0.00	0.00	0.0002	0.00022	0.000940
Ni	0.17	1.4	0.25	1.3	2.0	0.46	0.45	1.1	0.17	2.00	8 / 8	80.00	0.04	0.0320	0.18	0.41
Se	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	<0.39	0.39	8 / 8	5.00	0.30	0.3900	0.39 U	0.39 U
Ag	0.0322	0.0114 UB	0.0150 UB	0.0182 UB	0.0182 UB	0.0131 UB	0.0124 UB	0.0292 UB	<0.0043	0.0322	1 / 8	1.00	0.01	0.0043	0.0120 UB	0.0105 UB
Tl	0.0035 U	0.0083	0.0045	0.0086	0.021	0.0035 U	0.0035 U	0.010	<0.0035	0.0210	5 / 8	18.00	0.00	0.0035	0.0035 U	0.0035 U
Zn	0.500 UB	3.5	0.035	9.310	15.110	4.110	2.610	6.010	<0.035	15.11	7 / 8	59.00	0.04	0.0350	7.71	27.61
																Sample Location ML-13

*Samples ML-13 and ML-15 were combined with the Marsden Lake South data set (see section 4.1.2.2)

Table D4. Marsden Lake (North) Surface Water Pesticides and PCBs Data

Parameter (ug/L)	SC-ML-SW-11	SC-ML-SW-12*	SC-ML-SW-14*	SC-ML-SW-16	SC-ML-SW-17	SC-ML-SW-18	SC-ML-SW-19	SC-ML-SW-20	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-FB-02	SC-ML-FD-XX-02
4,4'-DDT	0.05 UJ	0.05 UJ	0.05 UJ	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.01	0.05	0.05	0.05 U	0.05 U
4,4'-DDD	0.05 U	0.05 U	0.05 UJ	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.01	0.05	0.05	0.05 U	0.05 U
4,4'-DDE	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.00	0.01	0.05	0.05 U	0.05 U
Aldrin	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	none	0.05	0.05	0.05 U	0.05 U
Alpha-BHC	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	2.40	0.05	0.05	0.05 U	0.05 U
Beta-BHC	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	2.40	0.05	0.05	0.05 U	0.05 U
Chlordane, Technical	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	2.40	0.05	0.05	0.05 U	0.05 U
Delta-BHC	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	2.40	0.05	0.05	0.05 U	0.05 U
Dieldrin	0.05 UJ	0.05 UJ	0.05 UJ	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.00	0.05	0.05	0.05 U	0.05 U
Endosulfan I	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.03	0.05	0.05	0.05 U	0.05 U
Endosulfan II	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.03	0.10	0.05	0.05 U	0.05 U
Endosulfan Sulfate	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	none	0.10	0.05	0.05 U	0.05 U
Endrin	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.02	0.04	0.05	0.05 U	0.05 U
Endrin Aldehyde	0.05 U	0.05 U	0.05 UJ	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	none	0.10	0.05	0.05 U	0.05 U
Heptachlor	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.00	0.05	0.05	0.05 U	0.05 U
Heptachlor Epoxide	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.00	0.05	0.05	0.05 U	0.05 U
Lindane	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	0.08	0.05	0.05	0.05 U	0.05 U
Methoxychlor	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 8	none	0.40	0.05	0.05 U	0.05 UJ
Toxaphene	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	<0.5	<0.5	0 / 8	0.00	0.50	0.50	0.50 U	0.5 U
Aroclor-1016	0.10 U	0.10 U	0.10 U	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	<0.1	<0.1	0 / 8	none	0.50	0.10	0.10 UJ	0.1 U
Aroclor-1221	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 8	0.27	0.50	0.10	0.10 U	0.1 U
Aroclor-1232	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 8	0.50	0.50	0.10	0.10 U	0.1 U
Aroclor-1242	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 8	0.06	0.50	0.10	0.10 U	0.1 U
Aroclor-1248	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 8	0.01	0.50	0.10	0.10 U	0.1 U
Aroclor-1254	0.10 U	0.10 U	0.10 U	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	<0.1	<0.1	0 / 8	0.02	0.50	0.10	0.10 U	0.1 UJ
Aroclor-1260	0.10 U	0.10 U	0.10 U	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	<0.1	<0.1	0 / 8	1.30	0.50	0.10	0.10 UJ	0.1 U
																Sample Location ML-13

*Samples ML-13 and ML-15 were combined with the Marsden Lake South data set (see section 4.1.2.2)

Table D5. Marsden Lake (North) Surface Water Semivolatile Organic Compound (SVOC) Data

Parameter (ug/L)	SC-ML-SW-11	SC-ML-SW-12*	SC-ML-SW-14*	SC-ML-SW-16	SC-ML-SW-17	SC-ML-SW-18	SC-ML-SW-19	SC-ML-SW-20	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-FB-02	SC-ML-FD-XX-02
Phenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	117.00	10.00	10.00	<10 U	<10 U
1,3-Dichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
1,2-Dichlorobenzene	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
4-Methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	72.20	10.00	10.00	<10 U	<10 U
Nitrobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
2,4-Dimethylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
2,4-Dichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
4-Chloroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	none	10.00	20.00	<20 U	<20 U
2-Methylnaphthalene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	2.08	10.00	10.00	<10 U	<10 U
2,4,5-Trichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	2.00	25.00	10.00	<10 U	<10 U
Dimethylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	0.00	10.00	10.00	<10 U	<10 U
3-Nitroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	none	10.00	20.00	<20 U	<20 U
4-Nitrophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	163.00	10.00	20.00	<20 U	<20 U
Diethylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	1.65	10.00	10.00	<10 U	<10 U
4-Nitroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	none	10.00	20.00	<20 U	<20 U
4-Bromophenyl-phenylether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	1.50	10.00	10.00	<10 U	<10 U
Phenanthrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	2.10	10.00	10.00	<10 U	<10 U
Fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	20.00	10.00	10.00	<10 U	<10 U
3,3'-Dichlorobenzidine	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	none	10.00	20.00	<20 U	<20 U
bis(2-Ethylhexyl)phthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	**8.30 J**	**44.00**	**8.30**	**44.00**	2 / 8	0.05	10.00	10.00	<10 U	<10 U
Benzo[k]fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Dibenz[a,h]anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
bis(2-Chloroethyl)ether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
1,4-Dichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	15.00	10.00	10.00	<10 U	<10 U
2-Methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	72.20	10.00	10.00	<10 U	<10 U
N-Nitroso-di-n-propylamine	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Isophorone	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Benzoic acid	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	41.60	10.00	20.00	<20 U	<20 U
1,2,4-Trichlorobenzene	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	110.00	10.00	10.00	<10 U	<10 U
Hexachlorobutadiene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Hexachlorocyclopentadiene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
2-Chloronaphthalene	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Acenaphthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	12.00	10.00	10.00	<10 U	<10 U
Acenaphthylene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	12.00	10.00	10.00	<10 U	<10 U
Dibenzofuran	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	20.40	10.00	10.00	<10 U	<10 U
4-Chlorophenyl-phenylether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
4,6-Dinitro-2-methylphenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	none	25.00	20.00	<20 U	<20 U
Hexachlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	0.00	10.00	10.00	<10 U	<10 U
Anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	0.03	10.00	10.00	<10 U	<10 U
Pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Benzo[a]anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	0.03	10.00	10.00	<10 U	<10 U
Di-n-octylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	30.00	10.00	10.00	<10 U	<10 U
Benzo[a]pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	0.01	10.00	10.00	<10 U	<10 U
Benzo[g,h,i]perylene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
2-Chlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Benzyl alcohol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	58.00	10.00	10.00	<10 U	<10 U
bis(2-chloroisopropyl)ether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Hexachloroethane	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10	<10	0 / 8	12.00	10.00	10.00	<10 UJ	<10 UJ
2-Nitrophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	163.00	10.00	10.00	<10 U	<10 U
bis(2-Chloroethoxy)methane	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Naphthalene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	81.00	10.00	10.00	<10 U	<10 U
4-Chloro-3-methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
2,4,6-Trichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	2.00	10.00	10.00	<10 U	<10 U
2-Nitroaniline	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	25.00	10.00	<10 U	<10 U
2,6-Dinitrotoluene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	50.00	10.00	10.00	<10 U	<10 U
2,4-Dinitrophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	none	25.00	20.00	<20 U	<20 U
2,4-Dinitrotoluene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	18.50	10.00	10.00	<10 U	<10 U
Fluorene	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	3.90	10.00	10.00	<10 U	<10 U
n-Nitrosodiphenylamine	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	24.50	10.00	10.00	<10 U	<10 U
Pentachlorophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 8	3.50	25.00	20.00	<20 U	<20 U
Di-n-butylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	0.10	10.00	10.00	<10 U	<10 U
Butylbenzylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	0.08	10.00	10.00	<10 U	<10 U
Chrysene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Benzo[b]fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
Indeno[1,2,3-cd]pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 8	none	10.00	10.00	<10 U	<10 U
																Sample Location ML-13

*Samples ML-13 and ML-15 were combined with the Marsden Lake South data set (see section 4.1.2.2)

Table D6. Marsden Lake (North) Sediment TOC and Metals Data

Parameter	SC-ML-SD-11	SC-ML-SD-12*	SC-ML-SD-14*	SC-ML-SD-16	SC-ML-SD-17	SC-ML-SD-18	SC-ML-SD-19	SC-ML-SD-20	Minimum	Maximum	DF	RBAL	Target DL	Achieved DL	SC-ML-SD-FB-02	SC-ML-SD-FD-XX-02
Total Organic Carbon (%)	8.90	15.00	18.00	1.80	1.70	0.57	17.00	16.00	0.57	18.00	na	na	na	na	na	16.00
GFAA Weight (g)	1.01	1.00	1.01	1.00	1.02	1.00	1.000	1.01	1.000	1.02	na	na	na	na	10.09	1.00
ICP Weight (g)	1.01	1.01	1.08	1.010	1.000	1.01	1.01	1.00	1.000	1.080	na	na	na	na	10.01	1.01
Percent Solids	19.90	6.20	8.30	61.80	47.40	71.20	16.20	12.0	6.20	71.20	na	na	na	na	na	6.40
Metals (mg/kg)																
Aluminum	6700.00	3200.00	5000.00	2200.00 J	3500.00 J	1700.00	6300.00 J	6100.00 J	1700.00	6700.00	8 / 8	none	40.00	40.00	4.0 UJ	7200 J
Antimony	0.49 UJ	0.50 UJ	0.50 UJ	0.70	48.00	0.50 U	0.50 U	0.50 U	0.49	48.00	2 / 8	none	40.00	0.50	0.050 U	0.80
Arsenic	5.90	8.10 J	5.60 J	1.20 J	1.90 J	1.0 U	8.90 J	6.70	1.00	8.90	7 / 8	6.00	1.50	1.0	0.10 U	12.00 J
Barium	59.00	70.00	60.00	19.00	50.00	12.00	99.00	100.00	12.00	100.00	8/ 8	none	4.00	4.0	0.40 U	130
Beryllium	0.99 U	0.99 U	0.93 U	0.99 U	1.0 U	0.99 U	0.99 U	1.0 U	0.93	1.00	0 / 8	none	4.00	1.0	0.10 U	0.99 U
Cadmium	0.49 U	1.40	0.51	0.50 U	0.49 U	0.50 U	0.50 J	0.50 U	0.49	1.40	3 / 8	0.60	0.50	0.50	0.050 U	1.3 J
Chromium	14.00 J	7.0	11.00	4.60 J	6.70 J	3.20 J	7.80 J	7.60 J	3.20	14.00	8/8	26.00	1.00	1.0	0.10 U	18.00 J
Cobalt	10.00 U	9.90 U	9.30 U	9.90 U	10.00 U	9.90 U	9.90 U	10.00 U	9.30	10.00	0 / 8	none	10.00	10.00	1.0 U	9.9 U
Copper	28.00	43.00 J	17.00 J	3.40	8.0	1.80	6.10	6.20	1.80	43.00	8 / 8	16.00	1.00	1	0.10 U	25.00
Lead	15.00 J	26.00	26.00	5.70 J	3.60 J	3.90 J	17.00 J	9.90 J	3.60	26.00	8 / 8	31.00	1.00	1	0.10 UJ	64.00 J
Magnesium	3300.00	1700.00	2300.00	700.00	1900.00	540.00	2000.00	2200.00	540.00	3300.00	8 /8	none	40.00	40.00	4.0 U	3100
Manganese	200.00 J	330.00	120.00	41.00 J	100.00 J	29.00 J	160.00 J	220.00 J	29.00	330.00	8 / 8	460.00	2.00	2.00	0.20 U	470 J
Mercury	0.10 U	0.14 UJ	0.10 J	0.10 U	0.10 U	0.10 U	0.12 J	0.10 J	0.10	0.14	3 / 8	0.20	0.10	0.10	0.01 U	0.16 J
Nickel	14.00	13.00	9.30 U	9.90 U	10.00 U	9.90 U	9.90 U	10.00 U	9.30	14.00	2 / 8	16.00	10.00	10.00	1.0 U	18
Selenium	0.99 U	1.0 U	0.99 U	1.0 U	0.98 U	1.0 U	1.20 J	1.0 J	0.98	1.20	2 / 8	none	40.00	1.0	0.10 U	1.0 U
Silver	0.49 U	0.50 U	0.50 U	0.50 U	0.49 U	0.50 U	0.50 U	0.50 U	0.49	0.50	0 / 8	1.00	0.50	0.50	0.050 U	0.50 U
Thallium	0.99 U	1.0 U	0.99 U	1.0 U	0.98 U	1.0 U	1.0 U	0.99 U	0.98	1.00	0 / 8	none	40.00	1.0	0.10 U	1.0 U
Zinc	64.00	86.00	50.00	15.00	21.00	18.00	50.00	50.00	15.00	86.00	8/ 8	120.00	5.00	5.00	0.50 U	70
																Sample Location ML-13

*Samples ML-13 and ML-15 were combined with the Marsden Lake South data set (see section 4.1.2.2)

Table D7. Marsden Lake (North) Sediment Pesticides and PCBs Data

Parameter (mg/kg)	SC-ML-SD-11	SC-ML-SD-12*	SC-ML-SD-14*	SC-ML-SD-16	SC-ML-SD-17	SC-ML-SD-18	SC-ML-SD-19	SC-ML-SD-20	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-SD-FB-02	SC-ML-SD-FD-XX-02
% SOLIDS	22.00	8.00	9.20	59.00	51.00	69.00	15.00	11.00	4.30	69.00	0 / 8	none	none	none	na	8.50
4,4'-DDT	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.1	0 / 8	0.01	0.01	0.08	0.05 U	0.08 U
4,4'-DDD	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.01	0.01	0.08	0.05 U	**0.095**
4,4'-DDE	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.01	0.01	0.08	0.05 U	0.08 U
Aldrin	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.00	0.01	0.08	0.05 U	0.08 U
Alpha-BHC	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.01	0.01	0.08	0.05 U	0.08 U
Beta-BHC	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.01	0.01	0.08	0.05 U	0.08 U
Chlordane, Technical	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.01	0.01	0.08	0.36 U	0.08 U
Delta-BHC	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.00	0.01	0.08	0.05 U	0.08 U
Dieldrin	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.00	0.01	0.08	0.05 U	0.08 U
Endosulfan I	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.05	0.01	0.08	0.05 U	0.08 U
Endosulfan II	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.05	0.01	0.08	0.05 U	0.08 U
Endosulfan Sulfate	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	none	0.01	0.08	0.05 U	0.08 U
Endrin	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.00	0.01	0.08	0.05 U	0.08 U
Endrin Aldehyde	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	none	0.01	0.08	0.05 U	0.08 U
Heptachlor	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.00	0.01	0.08	0.05 U	0.08 U
Heptachlor Epoxide	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.01	0.01	0.08	0.05 U	0.08 U
Lindane	0.08 U	0.08 U	0.08 U	0.01 U	0.02 U	0.01 U	0.06 U	0.08 U	<0.01	<0.08	0 / 8	0.00	0.01	0.08	0.05 U	0.08 U
Methoxychlor	0.08 U	0.08 U	0.08 U	0.01 UJ	0.02 UJ	0.01 UJ	0.06 UJ	0.08 UJ	<0.01	<0.08	0 / 8	0.02	0.01	0.08	0.05 U	0.08 UJ
Toxaphene	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	<0.17	<0.17	0 / 8	0.03	0.13	0.17	3.60 U	0.17 U
Aroclor-1016	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 8	0.01	1.00	0.33	0.10 UJ	0.33 U
Aroclor-1221	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 8	none	1.00	0.33	0.10 U	0.33 U
Aroclor-1232	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 8	none	1.00	0.33	0.10 U	0.33 U
Aroclor-1242	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 8	none	1.00	0.33	0.10 U	0.33 U
Aroclor-1248	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 8	0.03	1.00	0.33	0.10 U	0.33 U
Aroclor-1254	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 8	0.06	1.00	0.33	0.10 UJ	0.33 U
Aroclor-1260	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 8	0.01	1.00	0.33	0.10 UJ	0.33 U
																Sample Location ML-13

*Samples ML-13 and ML-15 were combined with the Marsden Lake South data set (see section 4.1.2.2)

Table D8. Marsden Lake (North) Sediment Semivolatile Organic Compounds (SVOCs) Data

Parameter (ug/kg)	SC-ML-SD-11	SC-ML-SD-12*	SC-ML-SD-14*	SC-ML-SD-16	SC-ML-SD-17	SC-ML-SD-18	SC-ML-SD-19	SC-ML-SD-20	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-SD-FB-02	SC-ML-SD-FD-XX-02
Phenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
1,3-Dichlorobenzene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	1700.00	330.00	1900.00	82.00 U	1900.00 U
1,2-Dichlorobenzene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	340.00	330.00	1900.00	82.00 U	1900.00 U
4-Methylphenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Nitrobenzene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
2,4-Dimethylphenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
2,4-Dichlorophenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
4-Chloroaniline	10000.00 UJ	19000.00 UJ	17000.00 UJ	4900.00 U	5700.00 U	4400.00 U	14000.00 UJ	14000.00 UJ	<4400	<19000	0 / 8	none	330.00	19000.00	160.00 U	19000.00 U
2-Methylnaphthalene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	70.00	330.00	1900.00	82.00 U	1900.00 U
2,4,5-Trichlorophenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Dimethylphthalate	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
3-Nitroaniline	2000.00 U	3700.00 U	3300.00 U	970.00 U	1100.00 U	870.00 U	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	800.00	3800.00	160.00 U	3800.00 U
4-Nitrophenol	2000.00 U	3700.00 U	3300.00 U	970.00 U	1100.00 U	870.00 U	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	330.00	3800.00	160.00 U	3800.00 U
Diethylphthalate	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	630.00	330.00	1900.00	82.00 U	1900.00 U
4-Nitroaniline	2000.00 U	3700.00 U	3300.00 U	970.00 U	1100.00 U	870.00 U	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	330.00	3800.00	160.00 U	3700.00 U
4-Bromophenyl-phenylether	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	1300.00	330.00	1900.00	82.00 U	1900.00 U
Phenanthrene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	560.00	330.00	1900.00	82.00 U	1900.00 U
Fluoranthene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	190.00	330.00	1900.00	82.00 U	1900.00 U
3,3'-Dichlorobenzidine	2000.00 U	3700.00 U	3300.00 U	970.00 U	1100.00 UJ	870.00 UJ	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	330.00	3800.00	160.00 U	3800.00 UJ
bis(2-Ethylhexyl)phthalate	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	25000.00	330.00	1900.00	**55.00 J**	1900.00 U
Benzo[k]fluoranthene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	240.00	330.00	1900.00	82.00 U	1900.00 U
Dibenz[a,h]anthracene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	60.00	330.00	1900.00	82.00 U	1900.00 U
bis(2-Chloroethyl)ether	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
1,4-Dichlorobenzene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	350.00	330.00	1900.00	82.00 U	1900.00 U
2-Methylphenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
N-Nitroso-di-n-propylamine	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Isophorone	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Benzoic acid	2000.00 U	3700.00 U	3300.00 U	970.00 U	1100.00 U	870.00 U	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	330.00	3800.00	160.00 U	3800.00 U
1,2,4-Trichlorobenzene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	9200.00	330.00	1900.00	82.00 U	1900.00 U
Hexachlorobutadiene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Hexachlorocyclopentadiene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 UJ	440.00 UJ	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 UJ
2-Chloronaphthalene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Acenaphthene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	620.00	330.00	1900.00	82.00 U	1900.00 U
Acenaphthylene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	620.00	330.00	1900.00	82.00 U	1900.00 U
Dibenzofuran	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	2000.00	330.00	1900.00	82.00 U	1900.00 U
4-Chlorophenyl-phenylether	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
4,6-Dinitro-2-methylphenol	2000.00 U	3700.00 U	3300.00 U	970.00 UJ	1100.00 UJ	870.00 UJ	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	330.00	3800.00	160.00 U	3800.00 UJ
Hexachlorobenzene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Anthracene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	220.00	330.00	1900.00	82.00 U	1900.00 U
Pyrene	1000.00 UJ	1900.00 UJ	1700.00 UJ	490.00 U	570.00 U	440.00 U	1400.00 UJ	1400.00 UJ	<440	<1900	0 / 8	490.00	330.00	1900.00	82.00 U	1900.00 U
Benzo[a]anthracene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	320.00	330.00	1900.00	82.00 U	1900.00 U
Di-n-octylphthalate	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Benzo[a]pyrene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	370.00	330.00	1900.00	82.00 U	1900.00 U
Benzo[g,h,i]perylene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	170.00	330.00	1900.00	82.00 U	1900.00 U
2-Chlorophenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Benzyl alcohol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
bis(2-chloroisopropyl)ether	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Hexachloroethane	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	1000.00	330.00	1900.00	82.00 UJ	1900.00 U
2-Nitrophenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
bis(2-Chloroethoxy)methane	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Naphthalene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	480.00	330.00	1900.00	82.00 U	1900.00 U
4-Chloro-3-methylphenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
2,4,6-Trichlorophenol	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
2-Nitroaniline	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	800.00	1900.00	82.00 U	1900.00 U
2,6-Dinitrotoluene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	21.00	330.00	1900.00	82.00 U	1900.00 U
2,4-Dinitrophenol	2000.00 U	3700.00 U	3300.00 U	970.00 UJ	1100.00 UJ	870.00 UJ	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	800.00	3800.00	160.00 U	3800.00 UJ
2,4-Dinitrotoluene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	42.90	330.00	1900.00	82.00 U	1900.00 U
Fluorene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	190.00	330.00	1900.00	82.00 U	1900.00 U
n-Nitrosodiphenylamine	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Pentachlorophenol	2000.00 U	3700.00 U	3300.00 U	970.00 U	1100.00 U	870.00 U	2700.00 U	2700.00 U	<870	<3700	0 / 8	none	800.00	3800.00	160.00 U	3800.00 U
Di-n-butylphthalate	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	11000.00	330.00	1900.00	82.00 U	1900.00 U
Butylbenzylphthalate	1000.00 UJ	1900.00 UJ	1700.00 UJ	490.00 UJ	570.00 UJ	440.00 UJ	1400.00 UJ	1400.00 UJ	<440	<1900	0 / 8	11000.00	330.00	1900.00	82.00 U	1900.00 UJ
Chrysene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	340.00	330.00	1900.00	82.00 U	1900.00 U
Benzo[b]fluoranthene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	none	330.00	1900.00	82.00 U	1900.00 U
Indeno[1,2,3-cd]pyrene	1000.00 U	1900.00 U	1700.00 U	490.00 U	570.00 U	440.00 U	1400.00 U	1400.00 U	<440	<1900	0 / 8	200.00	330.00	1900.00	82.00 U	1900.00 U
																Sample Location ML-13

*Samples ML-13 and ML-15 were combined with the Marsden Lake South data set (see section 4.1.2.2)

Figure D2. Aluminum concentration at MLN water sampling locations versus distance away from grenade impact area

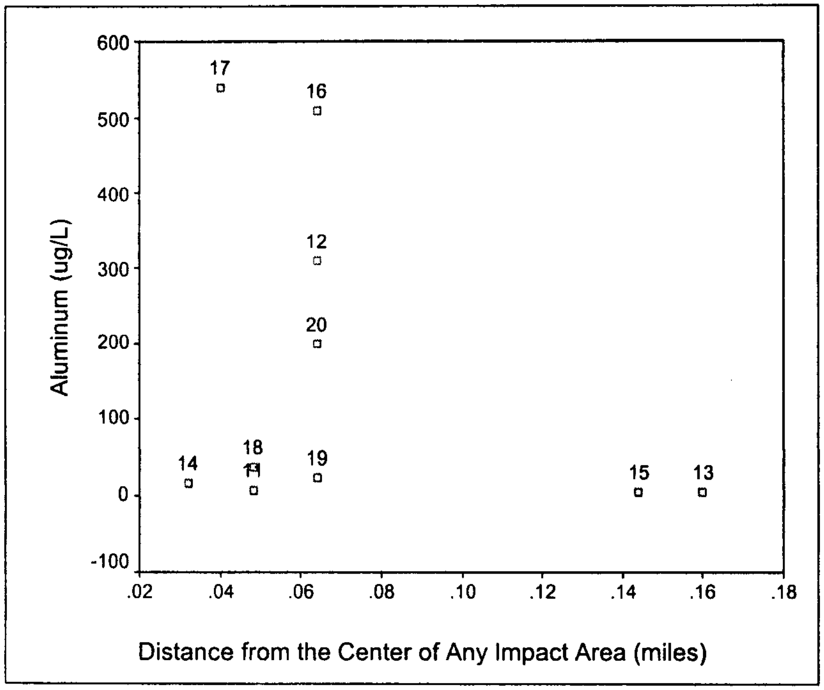
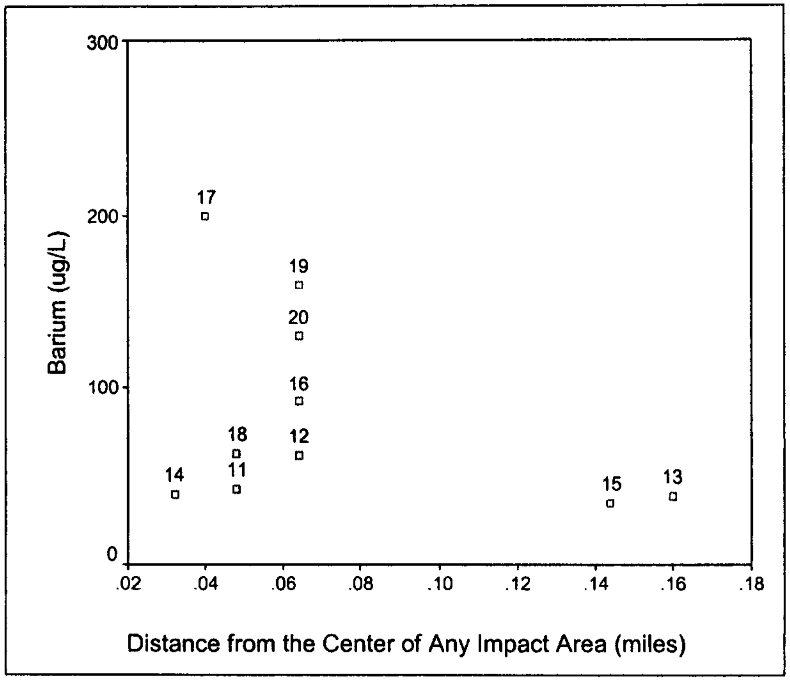


Figure D3. Barium concentration at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (Figure D1) and a ruler.

Figure D4. Antimony concentrations at MLN water sampling locations versus distance away from grenade impact area

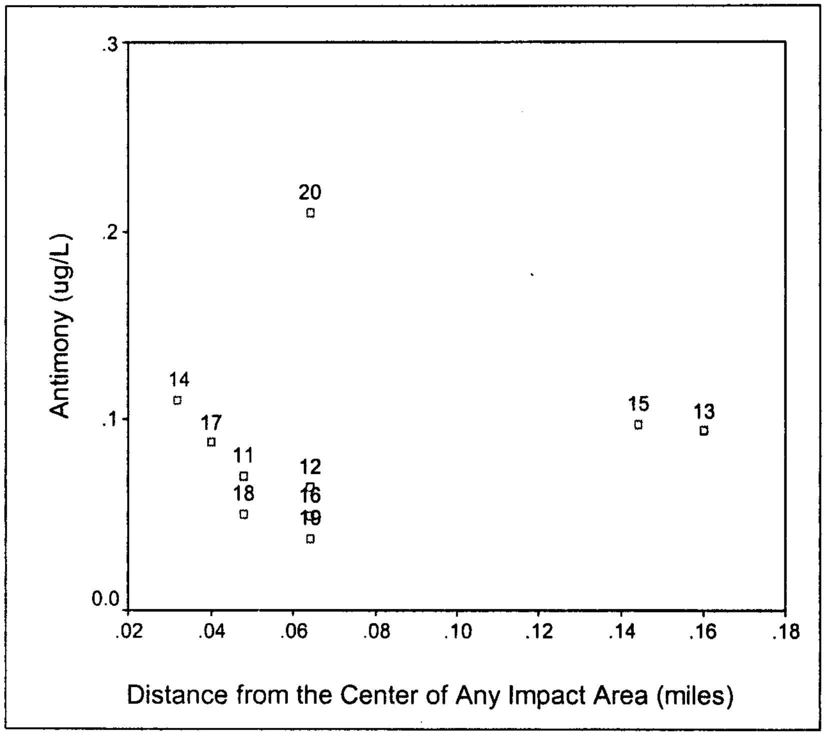
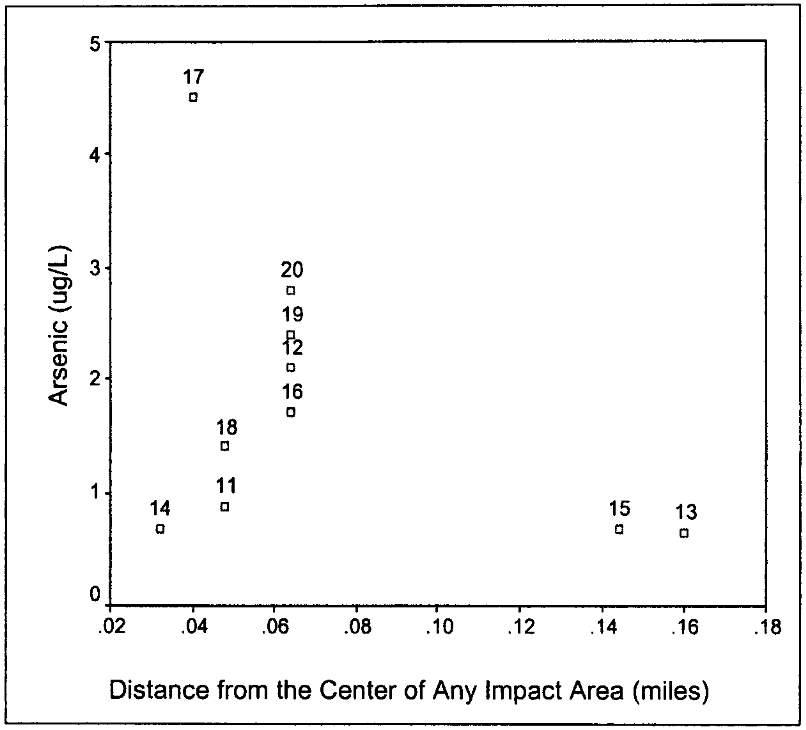


Figure D5. Arsenic concentration at MLN water sampling locations versus distance away from the grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (see Figure D1) and a ruler.

Figure D6. Beryllium concentrations at MLN water sampling locations versus distance away from grenade impact area

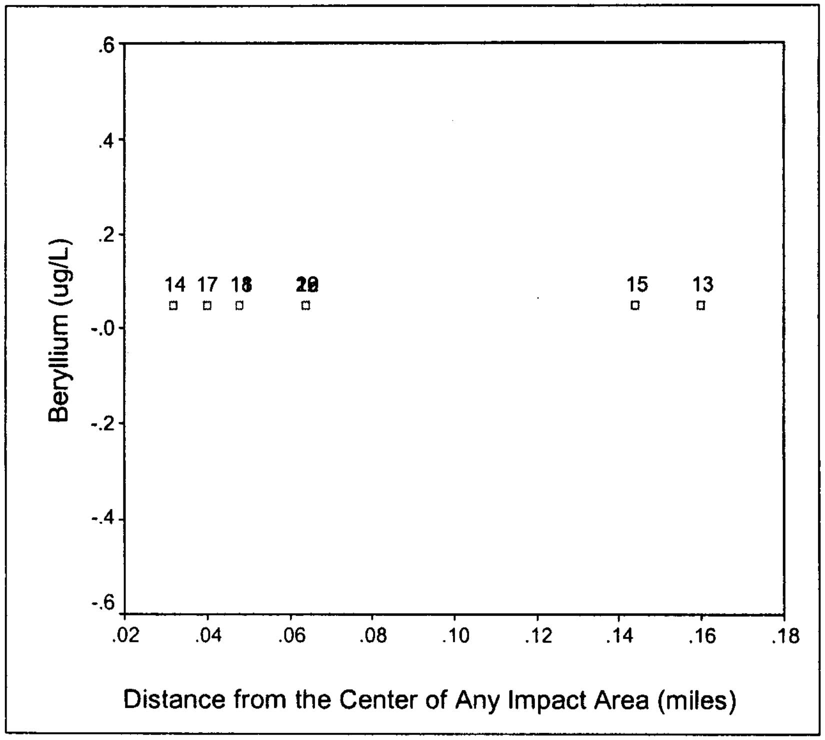
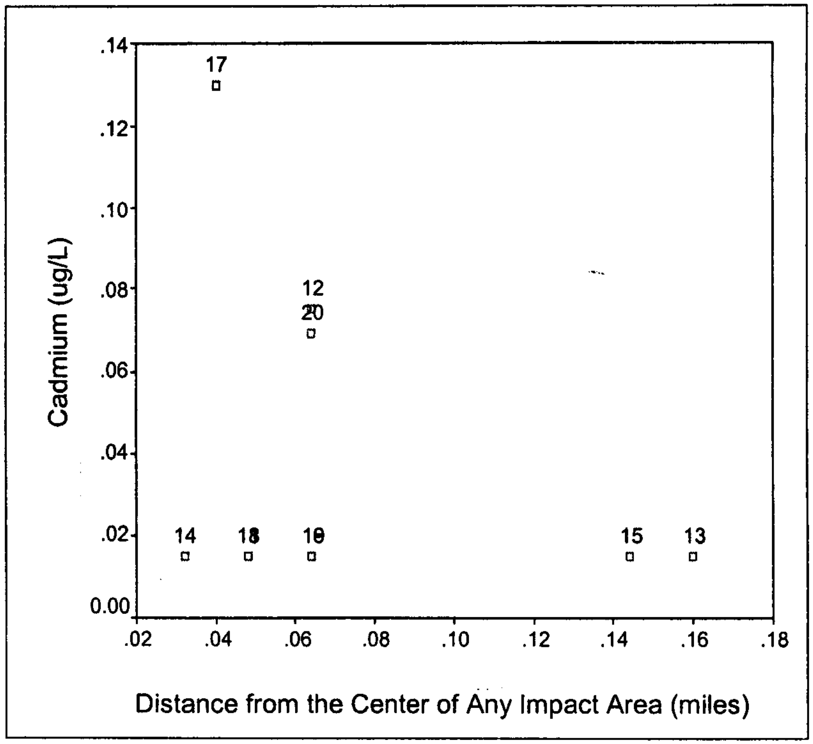


Figure D7. Cadmium concentrations at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (see Figure D1) and a ruler.

Figure D8. Chromium concentrations at MLN water sampling locations versus distance away from grenade impact area

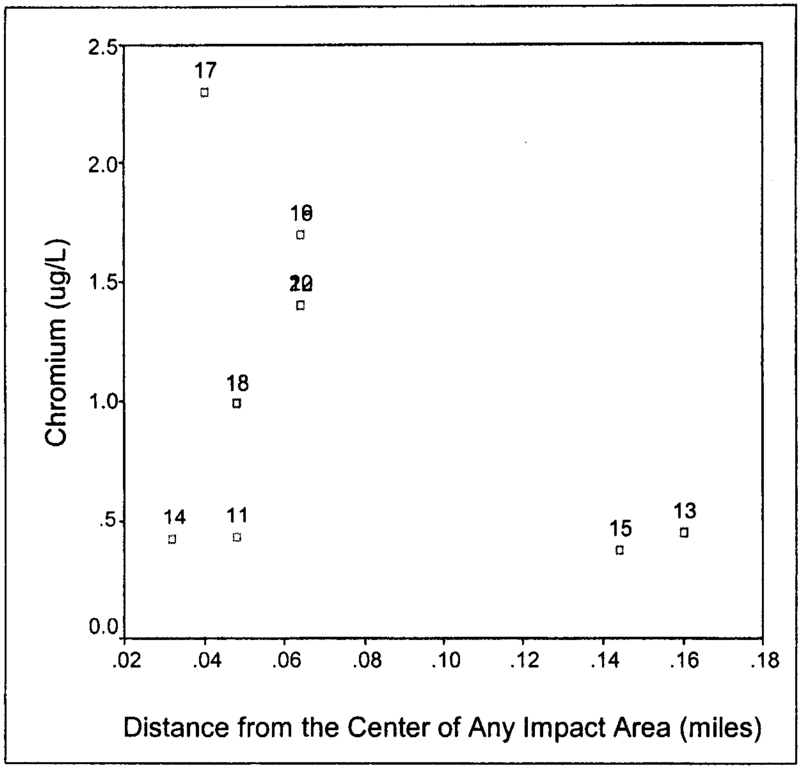
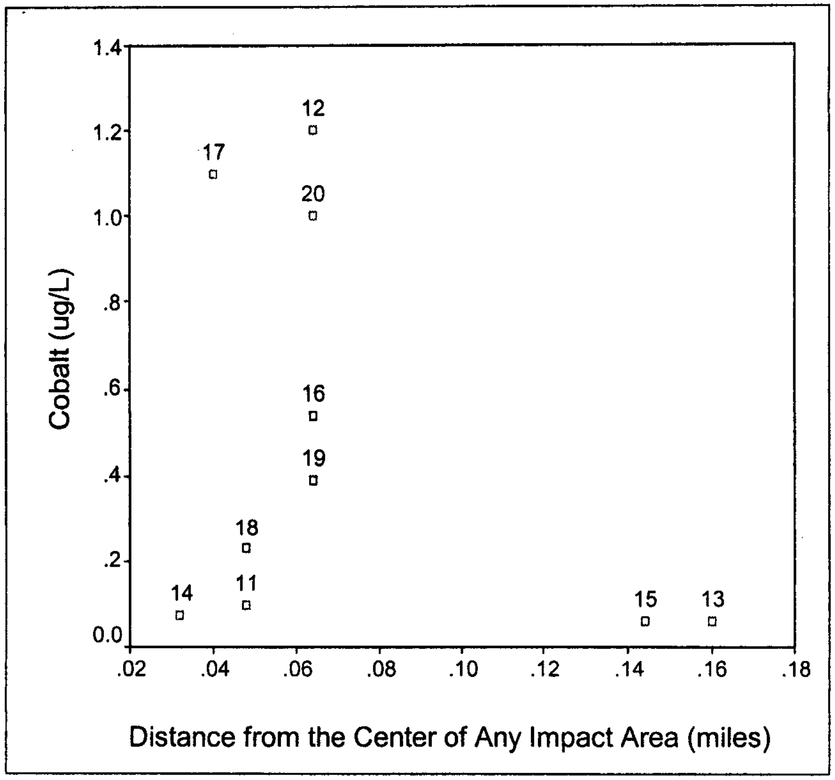


Figure D9. Cobalt concentrations at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (see Figure D1) and a ruler.

Figure D10. Copper concentrations at MLN water sampling locations versus distance away from grenade impact area

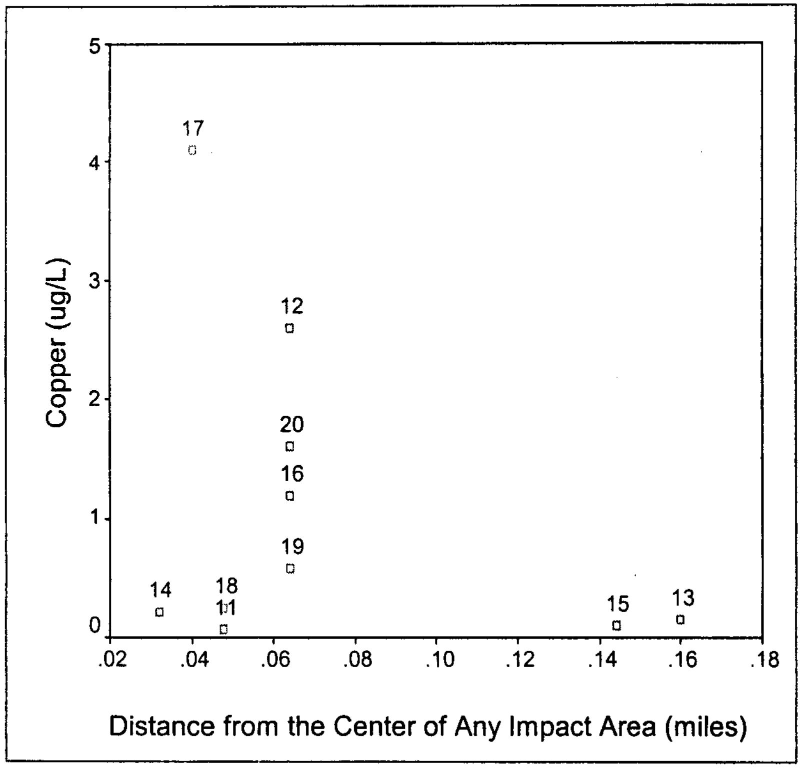
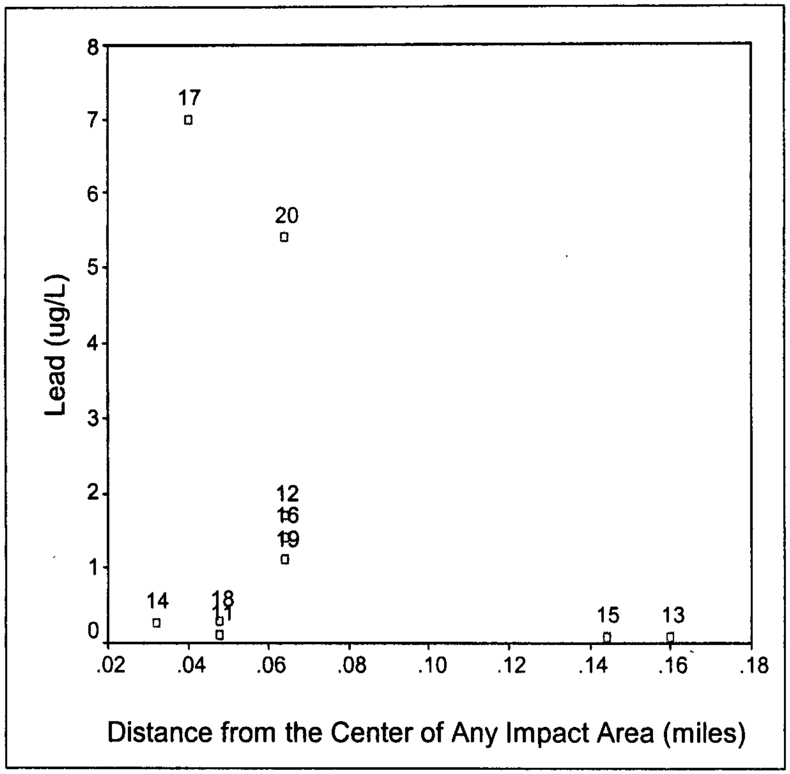


Figure D11. Lead concentrations at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (see Figure D1) and a ruler.

Figure 13. Magnesium concentrations at MLN water sampling locations versus distance away from grenade impact area

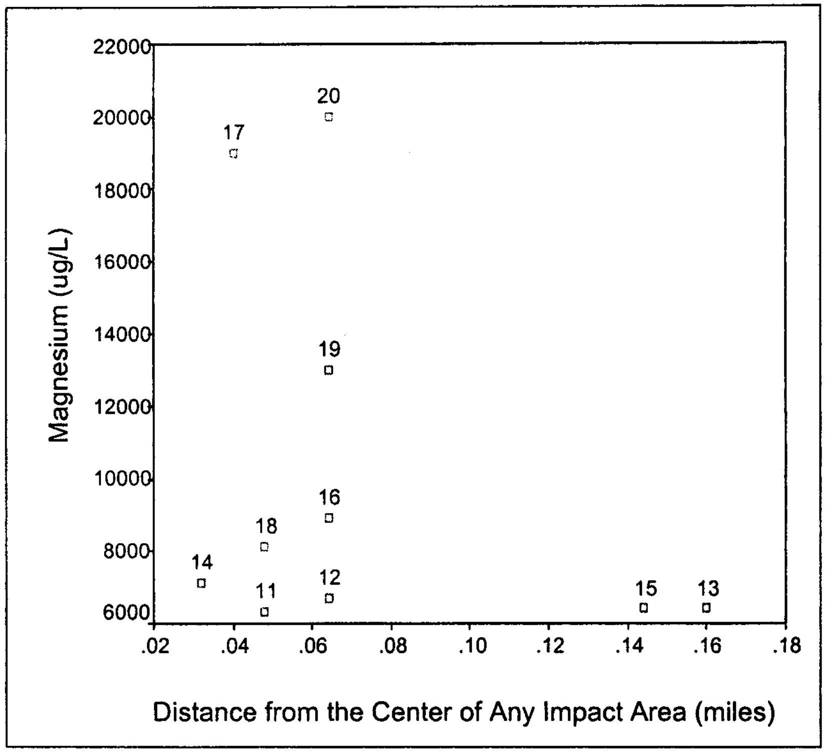
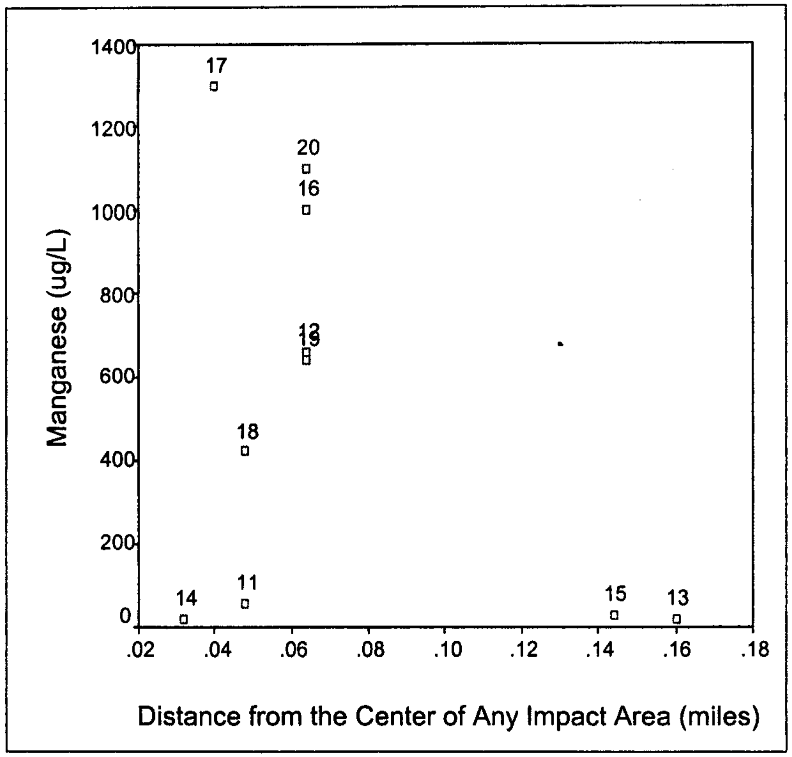


Figure D12. Manganese concentrations at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (see Figure D1) and a ruler.

Figure D14. Mercury concentrations at MLN water sampling locations versus distance away from grenade impact area

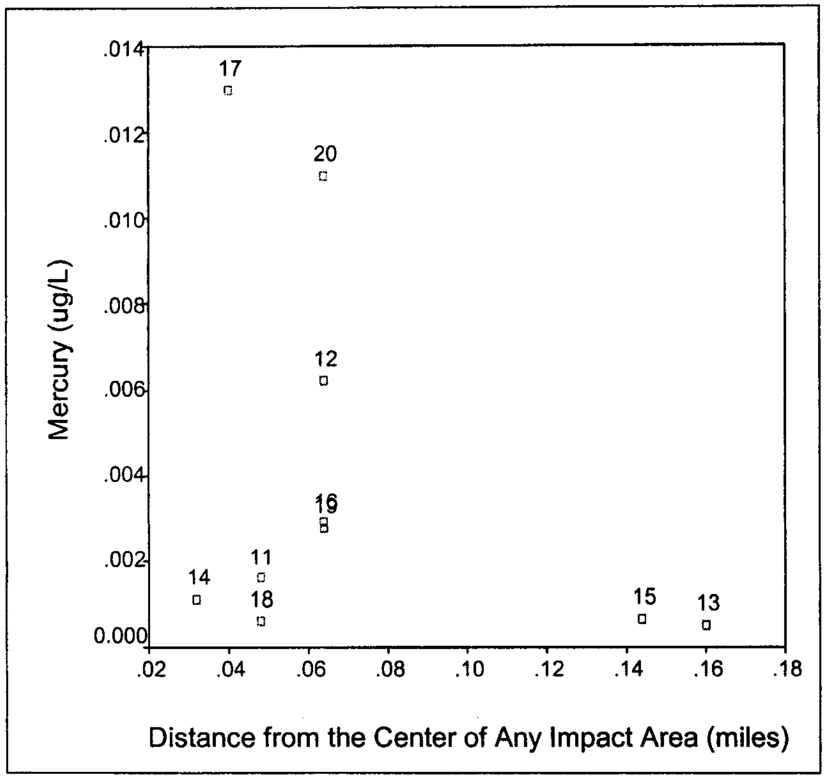
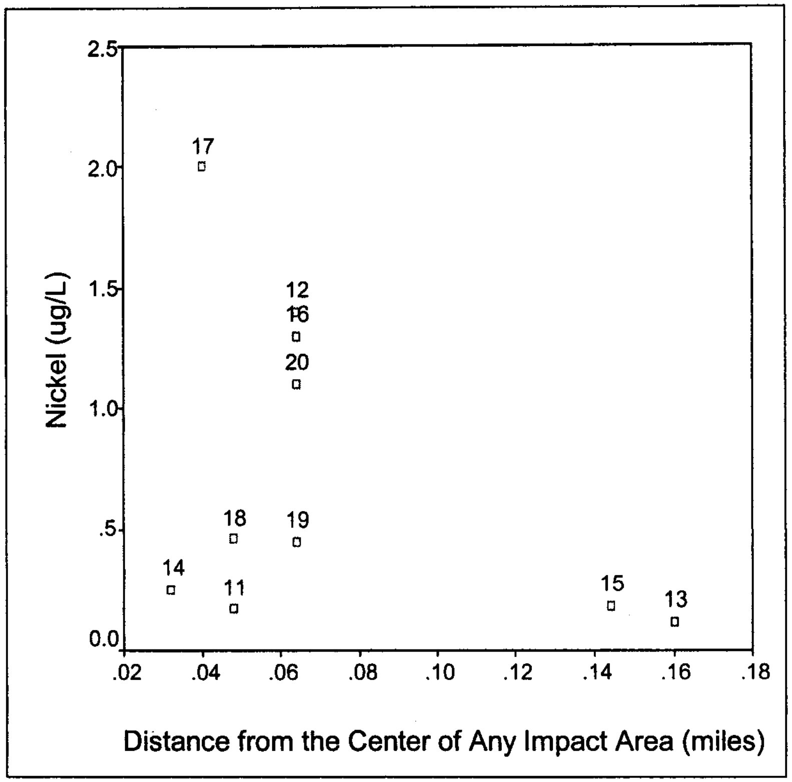


Figure D15. Nickel concentrations at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (see Figure D1) and a ruler.

Figure D16. Selenium concentrations at MLN water sampling locations versus distance away from grenade impact area

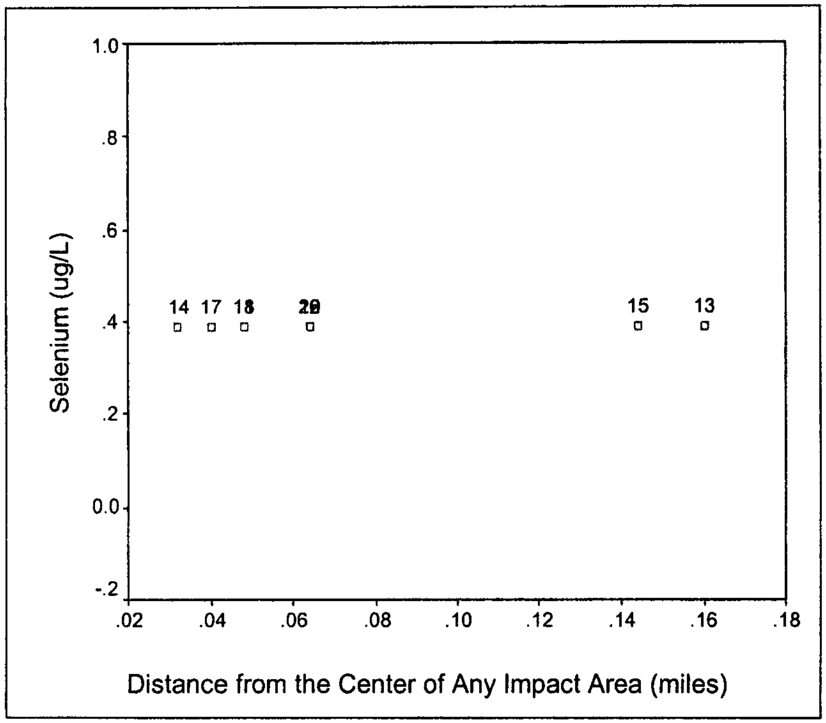
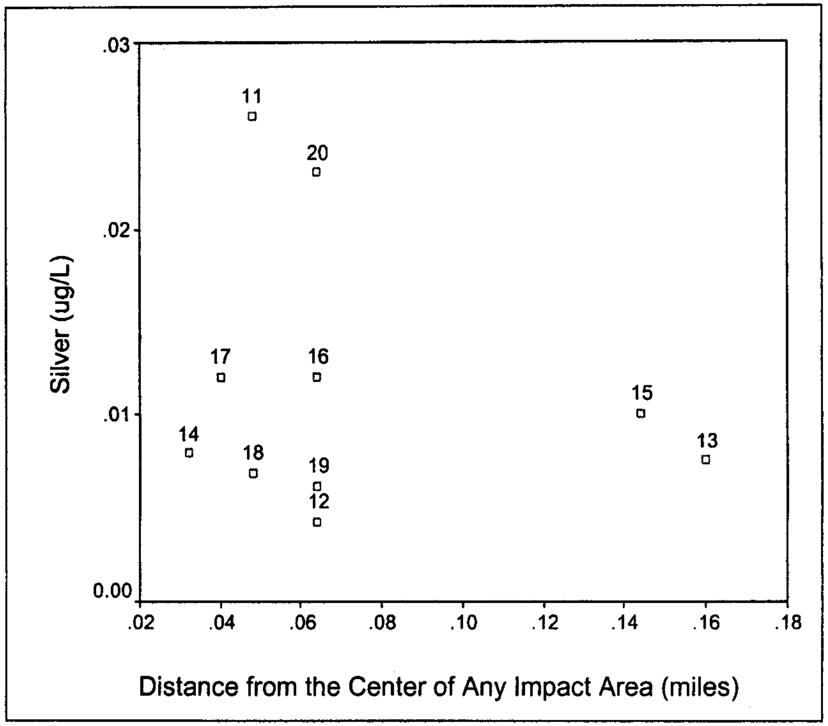


Figure D17. Silver concentrations at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (See Figure D1) and a ruler.

Figure 18. Thallium concentrations at MLN water sampling locations versus distance away from grenade impact area

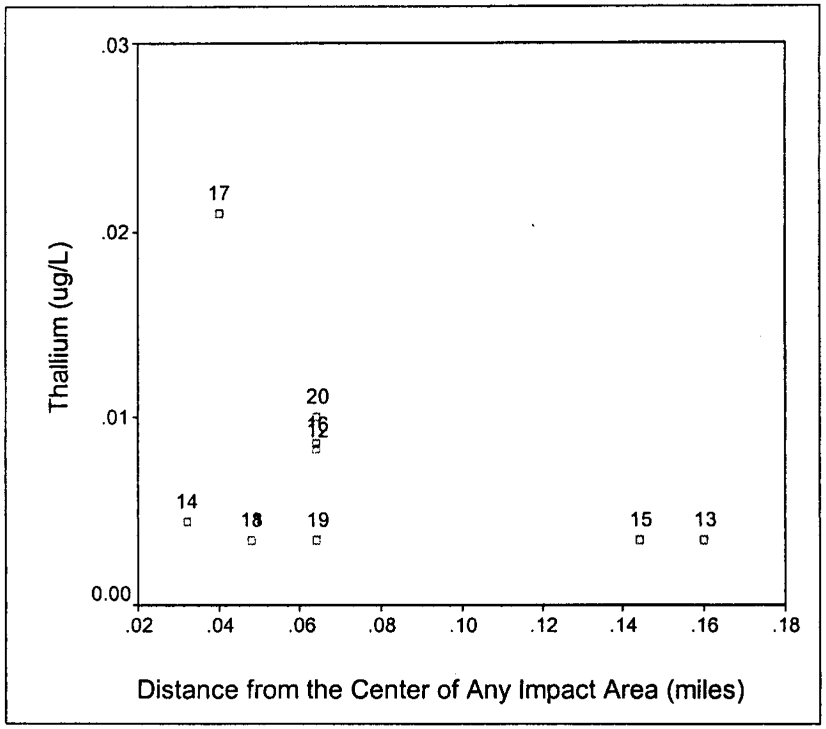
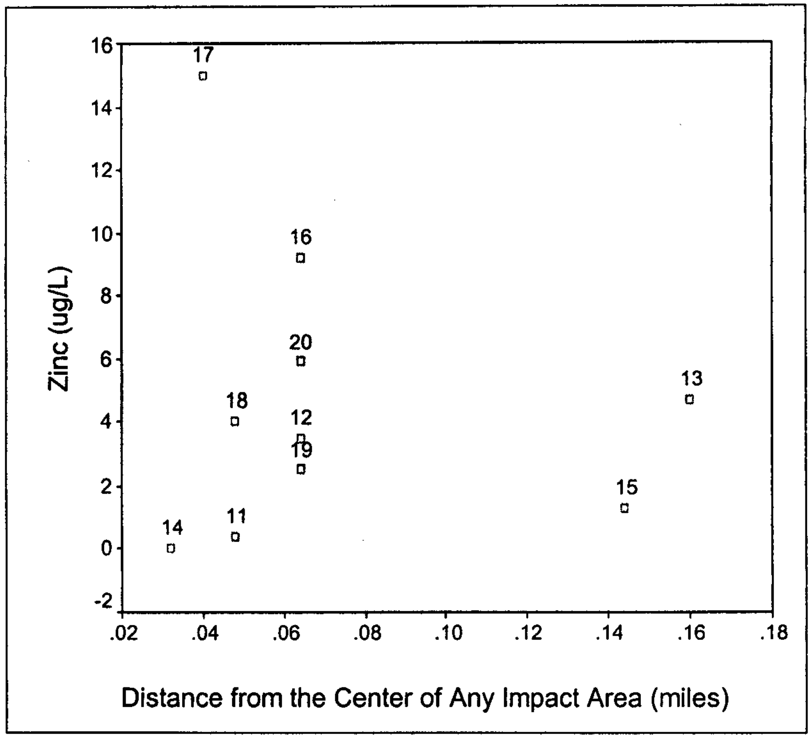


Figure D19. Zinc concentrations at MLN water sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (See Figure D1).
Distance measured is based on a physical measurement using the map scale bar (see Figure D1) and a ruler.

Figure D20. Aluminum concentrations at MLN sediment sampling locations versus distance away from grenade impact area

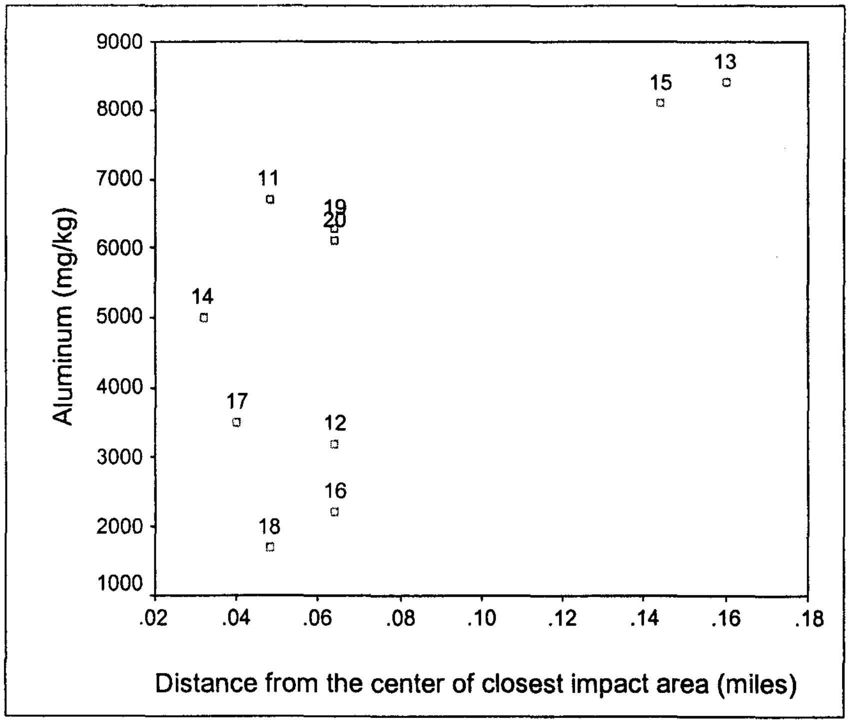
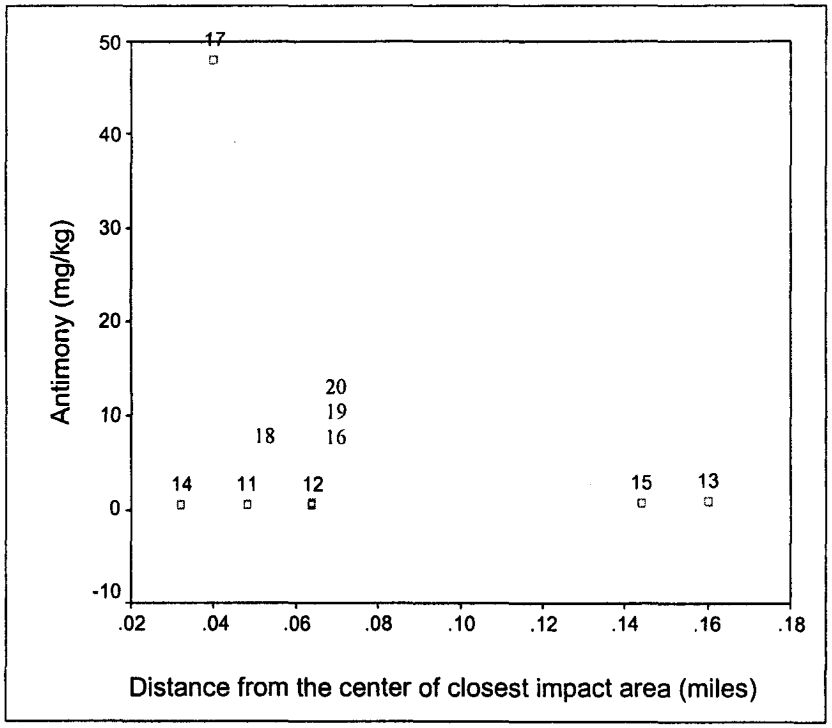


Figure D21. Antimony concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D22. Arsenic concentrations at MLN sediment sampling locations versus distance away from grenade impact area

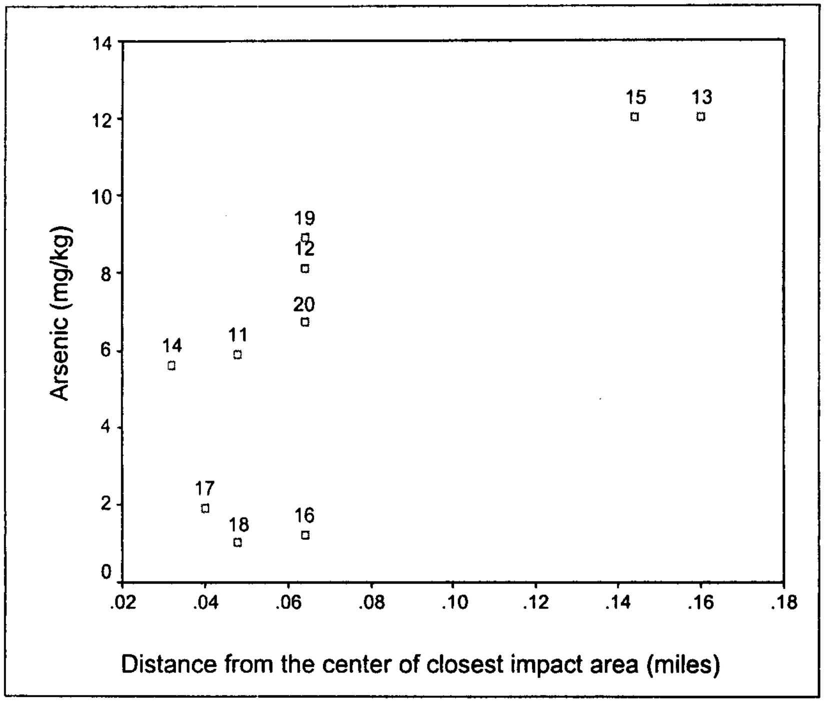
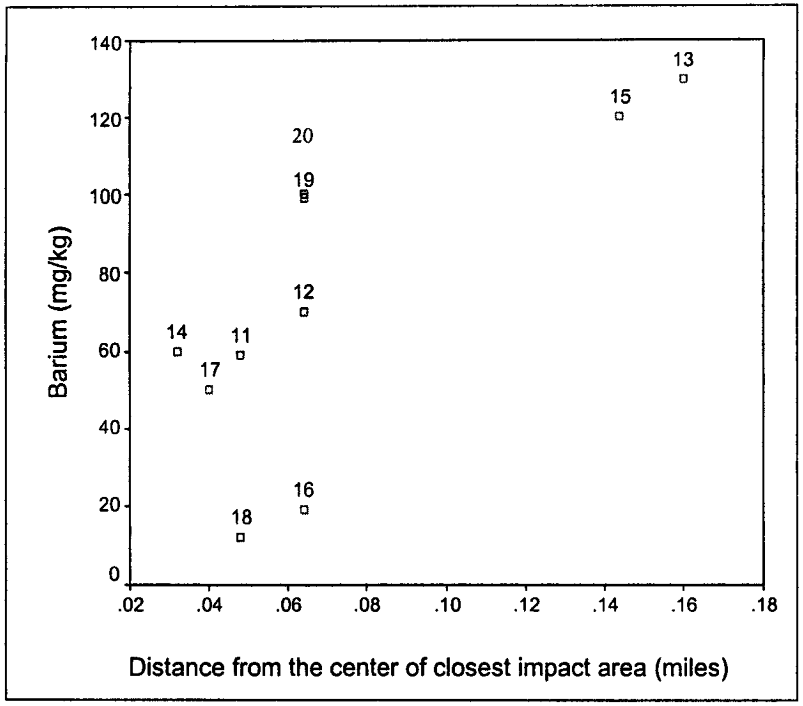


Figure D23. Barium concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D24. Beryllium concentrations at MLN sediment sampling locations versus distance away from grenade impact area

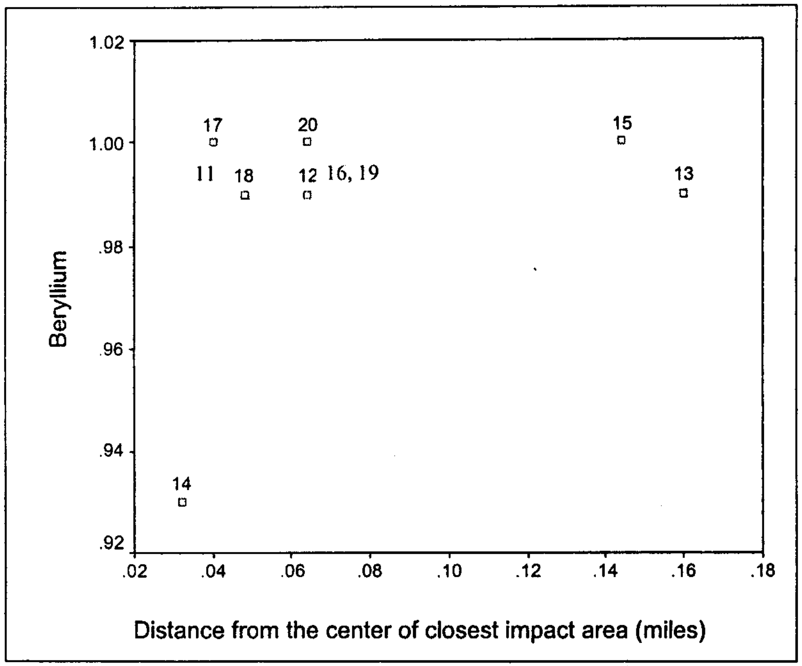
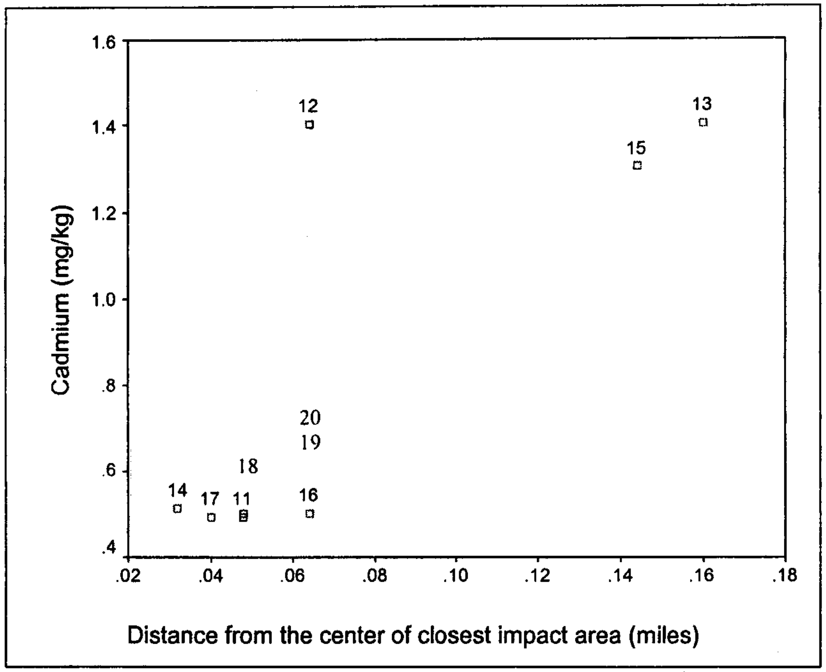


Figure D25. Cadmium concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D26. Chromium concentrations at MLN sediment sampling locations versus distance away from grenade impact area

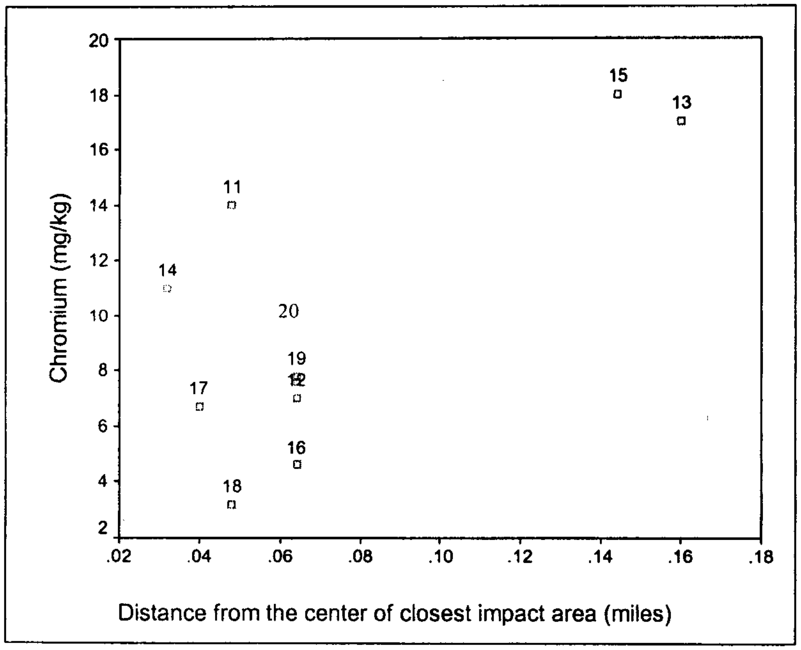
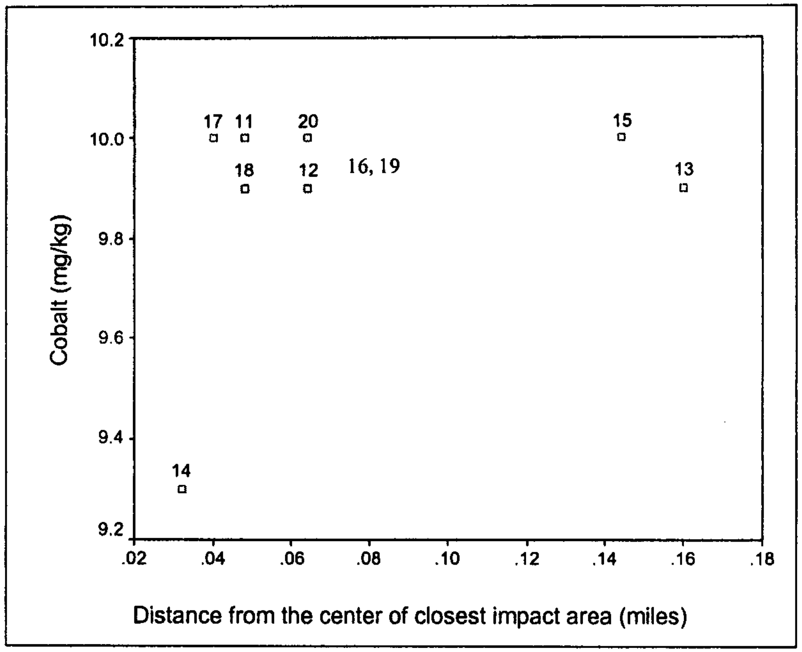


Figure D27. Cobalt concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D28. Copper concentrations at MLN sediment sampling locations versus distance away from grenade impact area

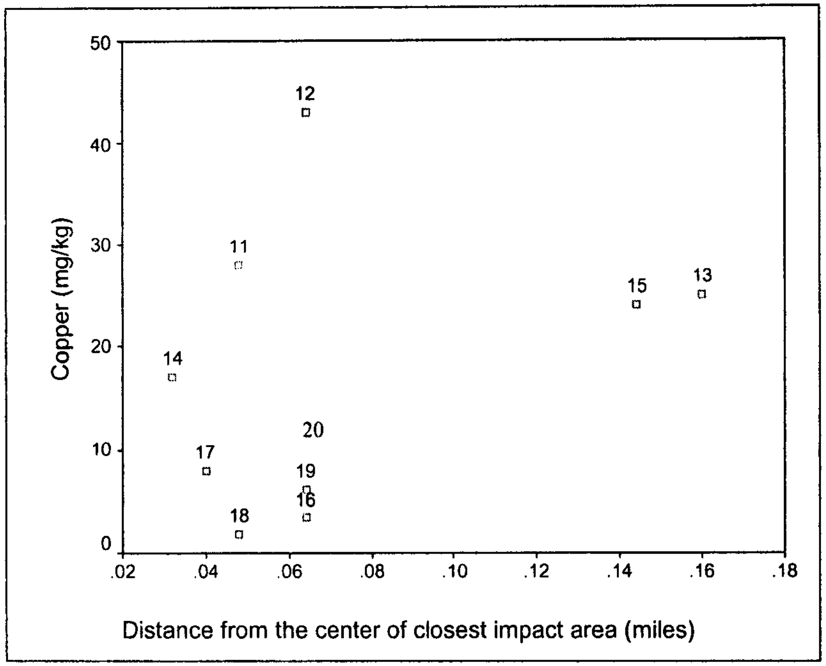
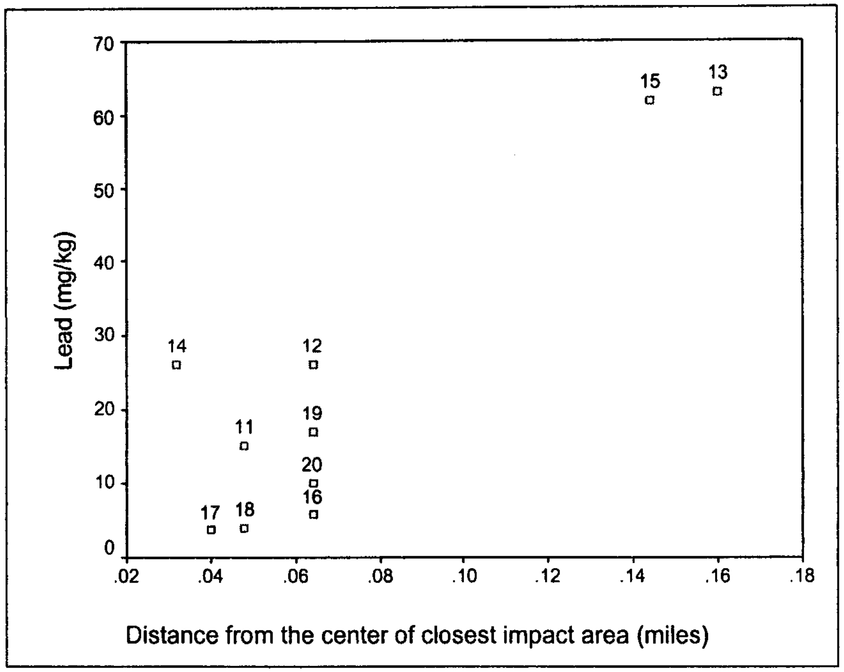


Figure D29. Lead concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D30. Magnesium concentrations at MLN sediment sampling locations versus distance away from grenade impact area

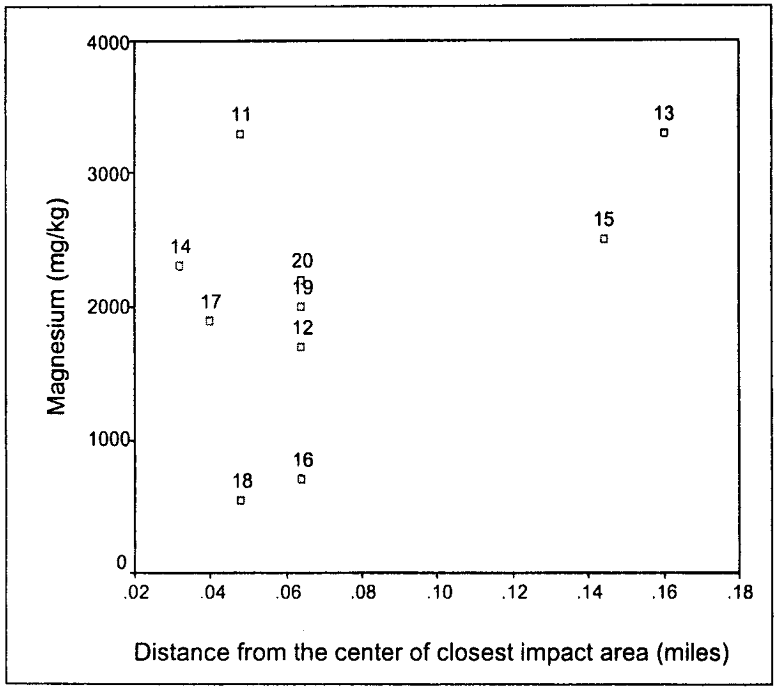
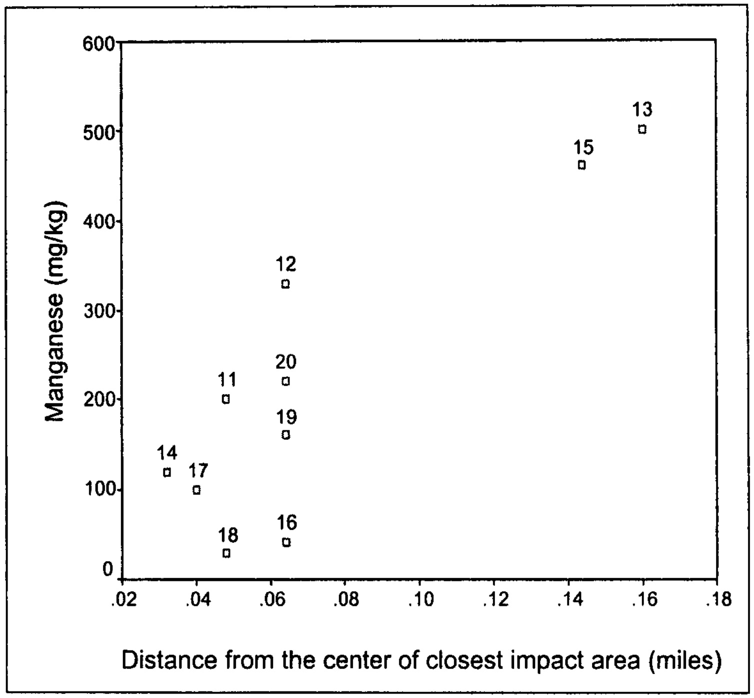


Figure D31. Manganese concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D32. Mercury concentrations at MLN sediment sampling locations versus distance away from grenade impact area

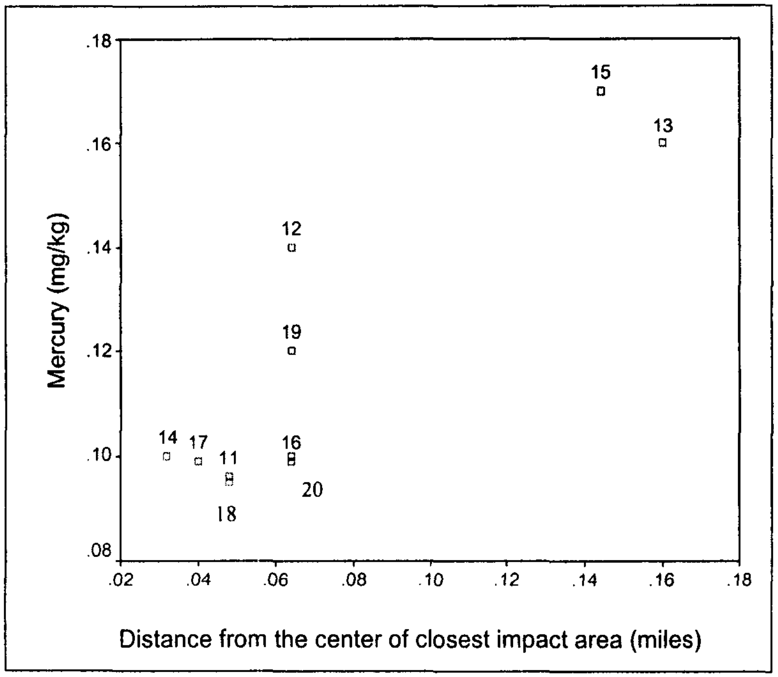
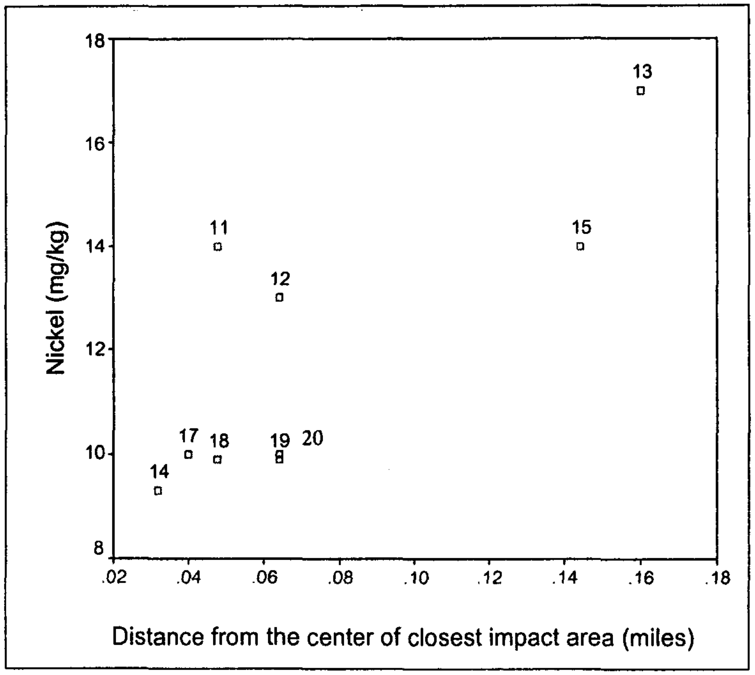


Figure D33. Nickel concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D34. Selenium concentrations at MLN sediment sampling locations versus distance away from grenade impact area

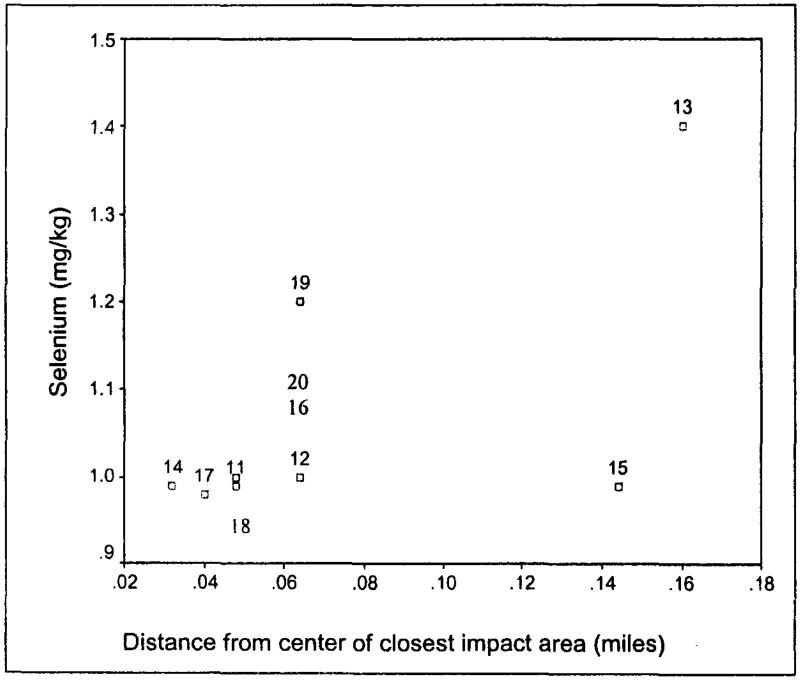
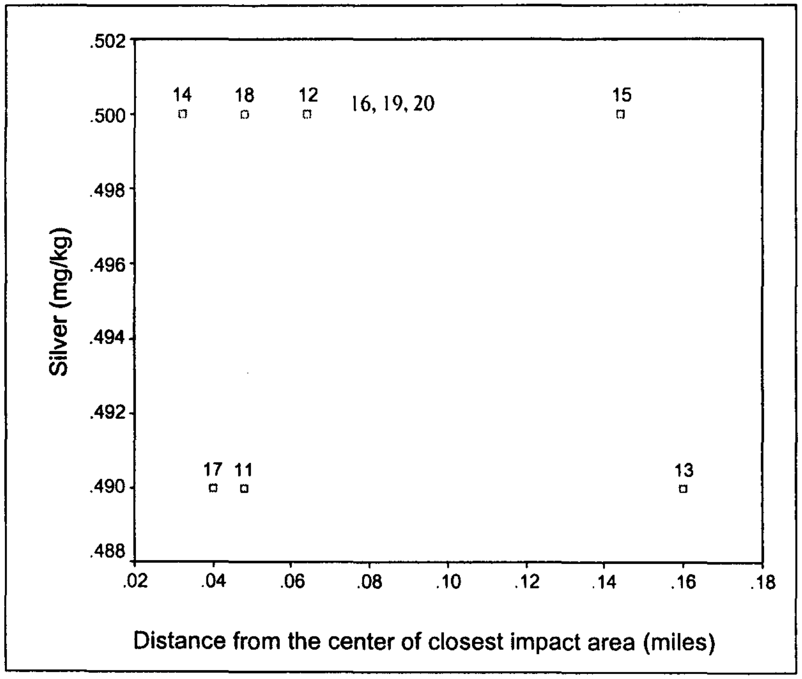


Figure D35. Silver concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

Figure D36. Thallium concentrations at MLN sediment sampling locations versus distance away from grenade impact area

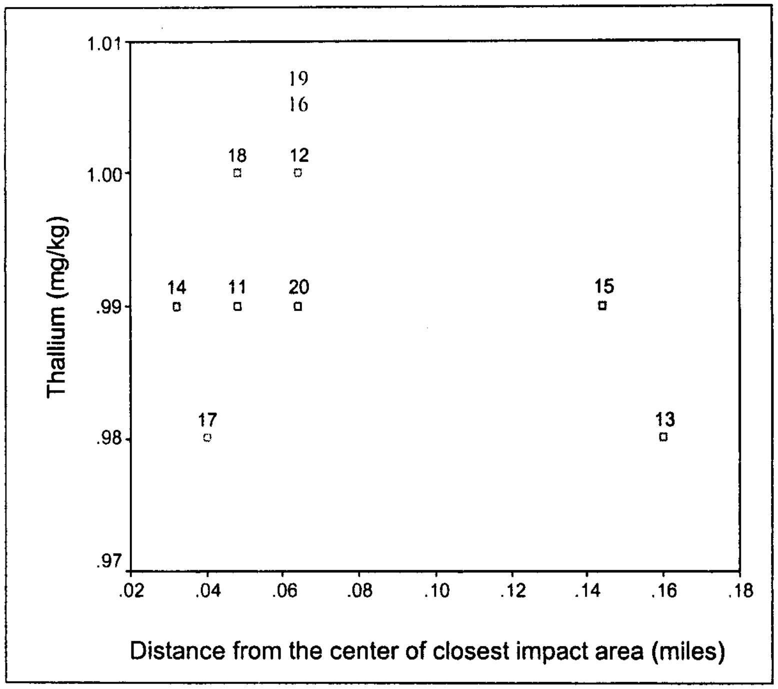
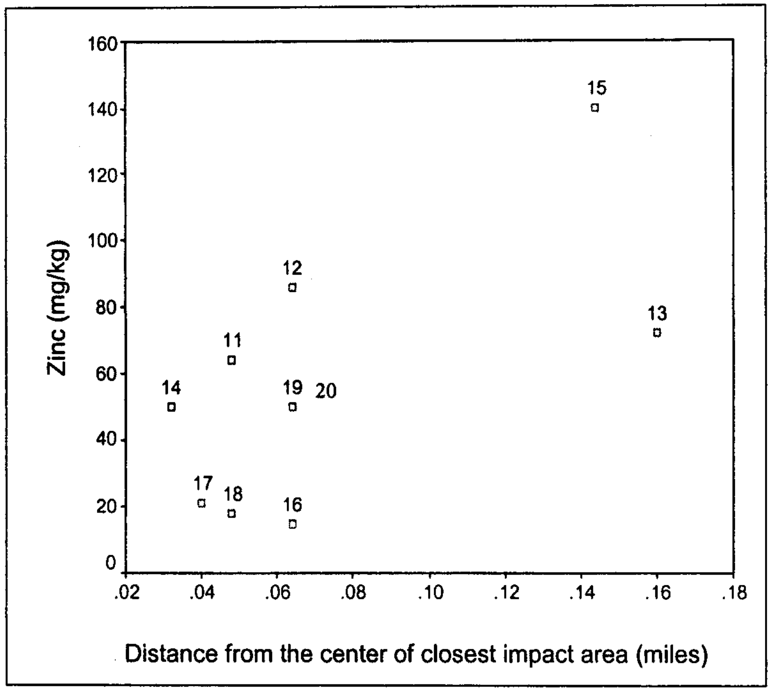


Figure D37. Zinc concentrations at MLN sediment sampling locations versus distance away from grenade impact area



Plotted numbers correspond to the sample location in Marsden Lake North (see figure D1).
Distance measured is based on a physical measurement using the map scale bar (see figure D1) and a ruler.

APPENDIX E

Marsden Lake (South) Sampling Locations and Analytical Measurement Results

Table of Contents

Marsden Lake (South) Data Presentation Summary

The map of Marsden Lake (South) (Fig.E1), shows the location where each media sample was taken. Global Positioning System (GPS) coordinates were used to plot the locations. These coordinates recorded in the field, during sampling, are reported in tables E1 and E2. The map image is from an orthophoto taken in 1997, obtained from Wenck Associates, Inc., Maple Plain, Minnesota.

The tables E3 through E8 summarize the data for Marsden Lake (South) surface water and sediment samples taken during the site characterization. The summaries include the field and analytical result for each analyte at each sample location. The minima and maxima are reported as well as the detection frequency (DF), risk based action level (RBAL), target detection limit and achieved detection limit. The RBALs and target detection limits are those that were defined in the QAPP. Summary data used for statistical analysis such as the central tendencies, standard deviations and confidence intervals are report in tables found in Appendix A.

The number of digits reported for each value in the following tables does not necessarily reflect the precision and accuracy of the measurement. The data tables present significant figures as they were originally reported by the analyzing laboratory. Significant figures reported for the analytes were based on the analytical detection limit values. However, some sample values have more significant figures than others to demonstrate that blank correction was discontinued, as was requested by USEPA comments on the draft version of this report (see Appendix R). Data quality is discussed in Appendix K.

The results are reported in the following manner to facilitate data review.

- If the detection limit is sample-specific (as it was for the pesticides in sediment), the value reported for the achieved detection limit for a site is the highest detection limit for that parameter for all samples (including the field duplicate) from that site.

- If a parameter was not detected in any samples from a site (e.g. Pond G, water, alpha-BHC) then the minimum and maximum values represent the lowest and highest achieved detection limits for those data.

- Any concentrations that were below the detection limit but still quantified are identified in Volume 2 of this report as estimated values (J-flagged). These values are reported and used uncensored in the following tables.

- If a parameter was undetected in at least one but not all samples from a site, then the minimum reflects the lowest detection limit of the undetected samples. If there was a J-flagged value lower than the highest undetected value, the J-flagged value was reported as the maximum because the value was actually detected but estimated.

- Field duplicates (FD) were used as quality control samples and were not included in the reported minima and maxima columns. However, if an analyte was detected in the FD and not in it's corresponding sample, then the FD detection was counted in the detection frequency column (only in the numerator).
- When laboratory replicate samples (R1, R2) were available, the values from the primary sample (R1) were reported. See Appendix K section K1.2.1 for explanation.

Definition of Data Qualifiers (Flags)

During the data validation process, all laboratory data were assigned appropriate data validation flags and reason codes. Validation flags used in the following tables are defined as follows:

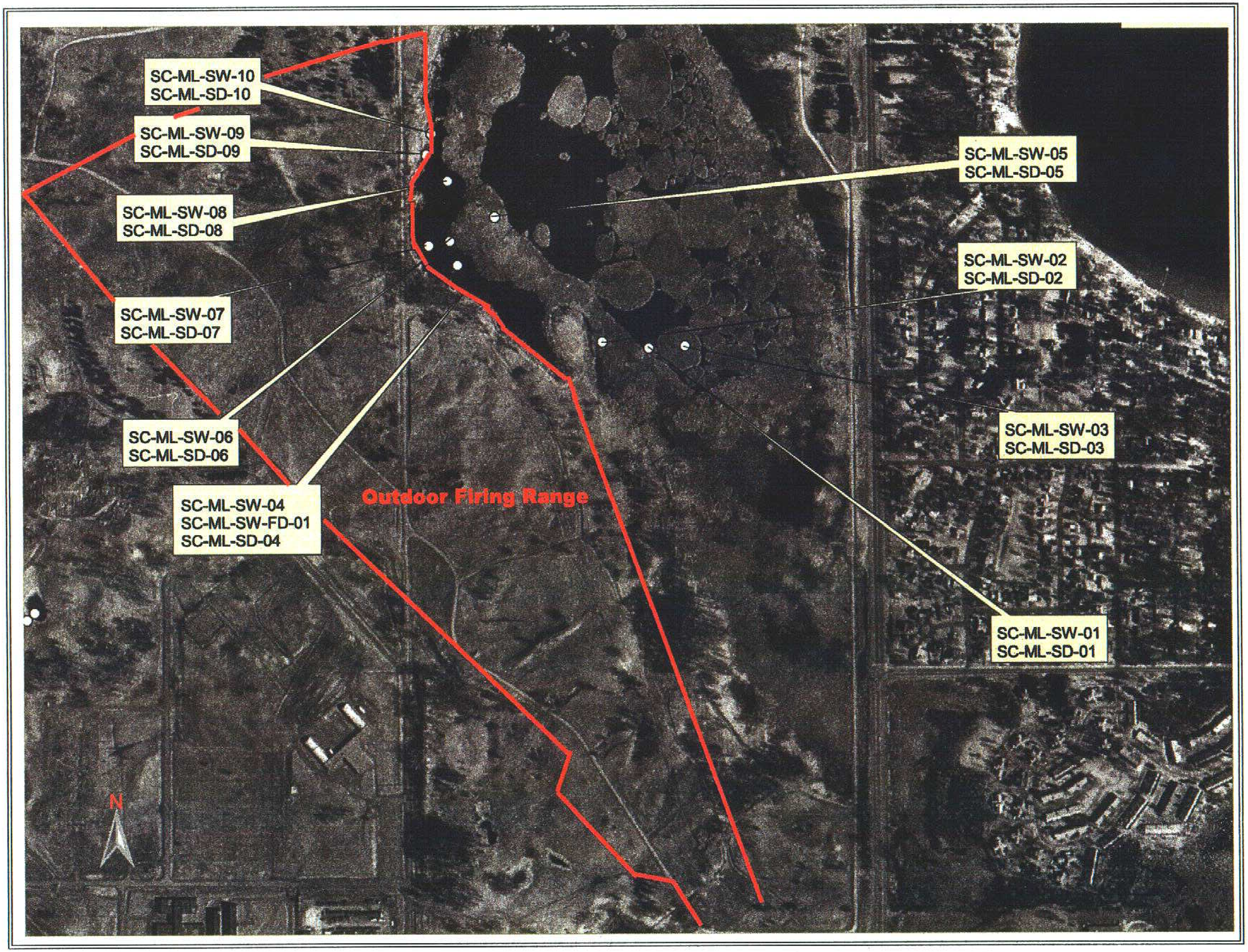
No flag	When the analyte value reported is detected and usable. The integrity of the analyte's identification, accuracy, and precision has been validated.
"U"	When the material was analyzed for, but not detected above the level of the associated detection limit value.
"UB"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with.
"J"	When the associated value is an estimated quantity. Indicating there is cause to question accuracy or precision of the reported value. The value is still used in the risk assessment.
"UJ"	When the analyte was analyzed for, but not detected, above the associated value, however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"UBJ"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with; however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"R"	When the analyte value reported is unusable. The integrity of the analyte's identification, accuracy, precision, or sensitivity has raised significant question as to the reality of the information presented.

Other information regarding data validation can be found in Volume 2 (CD ROM version) of the report.

Outlying data points

Table E9 shows the results of the Dixon test, to determine if there were outlying points from the Marsden Lake (South) sampling. Sample location ML-08 was the only location that was found to have outlying values for surface water metals. The results of the Dixon test were used to determine if the locations of points ML-01, ML-02, and ML-03 were unrepresentative of the site (refer to section 4.1.2 of the ERA report).

Figure E1. Marsden Lake (South) Water and Sediment Sampling Locations



SC= Site Characterization
ML= Marsden Lake
SW= Surface Water
SD= Sediment

0.06 0 0.06 0.12 Miles

Table E1. Marsden Lake (South) Surface Water Sampling GPS coordinates

Sample Number	Date	Sample depth	North*			West*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-ML-SW-01	22-Sep-99	4"	45	5	28.84	93	9	7.87
SC-ML-SW-02	22-Sep-99	4"	45	5	29.16	93	9	11.49
SC-ML-SW-03	22-Sep-99	4"	45	5	28.96	93	9	5.08
SC-ML-SW-04 SC-ML-SW-FD-01	23-Sep-99	4-8"	45	5	33.34	93	9	22.73
SC-ML-SW-05	23-Sep-99	4-8"	45	5	35.97	93	9	19.83
SC-ML-SW-06	23-Sep-99	4-8"	45	5	34.64	93	9	23.3
SC-ML-SW-07	23-Sep-99	4"	45	5	34.41	93	9	24.93
SC-ML-SW-08	24-Sep-99	4-6"	45	5	37.91	93	9	23.45
SC-ML-SW-09	23-Sep-99	4-6"	45	5	39.35	93	9	25.08
SC-ML-SW-10	24-Sep-99	4"	45	5	90.5	93	9	24.7

Table E2. Marsden Lake (South) Sediment Sampling GPS Coordinates

Sample Number	Date	Sample depth	North*			West*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-ML-SD-01	22-Sep-99	38"	45	5	28.84	93	9	7.87
SC-ML-SD-02	22-Sep-99	3'	45	5	29.16	93	9	11.49
SC-ML-SD-03	22-Sep-99	46"	45	5	28.96	93	9	5.08
SC-ML-SD-04	23-Sep-99	2' 6"	45	5	33.34	93	9	22.73
SC-ML-SD-05	23-Sep-99	48"	45	5	35.97	93	9	19.83
SC-ML-SD-06	23-Sep-99	24"	45	5	34.64	93	9	23.3
SC-ML-SD-07	23-Sep-99	3'	45	5	34.41	93	9	24.93
SC-ML-SD-08	24-Sep-99	32"	45	5	37.91	93	9	23.45
SC-ML-SD-09	23-Sep-99	2' 6"	45	5	39.35	93	9	25.08
SC-ML-SD-10	24-Sep-99	3'	45	5	90.5	93	9	24.7

*The GPS unit used in the study was a military-issue Precision Light Weight GPS Receiver (PLGR). The Department of Defense (DoD) intentionally places errors in the GPS navigation and timing signal, in a process called selective availability (Department of the Army 1993). Errors resulting from this process are unpredictable and can produce significant horizontal and elevation errors. Military GPS receivers must have crypto keys loaded to detect and nullify selective availability errors, in order to allow for more accurate position data (Department of the Army 1993). The PLGR was equipped with a crypto key, and was therefore not subject to the selective availability and spoofing errors that affect non-military GPS. (The only exception was during the second quarter). The PLGR operations manual (Department of the Army 1995) states that with crypto keys installed, the PLGR provides horizontal 2-D position accuracies better than the worst case error of 10 meters.

Note: References cited in the text of this appendix can be found in the reference section of the main Tier II Ecological Risk Assessment Report.

Table E3. Marsden Lake (South) Surface Water Quality and Metals Data

Parameter	SC-ML-SW-01	SC-ML-SW-02	SC-ML-SW-03	SC-ML-SW-04	SC-ML-SW-05	SC-ML-SW-06	SC-ML-SW-07	SC-ML-SW-08	SC-ML-SW-09	SC-ML-SW-10	SC-ML-SW-13*	SC-ML-SW-15*	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-FB-01	SC-ML-FD-XX-01
pH	7.32	7.33	7.36	7.31	7.43	7.36	7.01	6.85	6.80	6.75	7.51	7.64	6.75	7.64	12/12	na	na	na	na	na
DO (mg/L)	7.83	6.34	6.69	7.14	7.24	7.34	4.62	1.81	2.11	1.93	8.02	9.71	1.81	9.71	12/12	na	na	na	na	na
DO (%)	82.3	67.1	69.4	71.7	74.7	74.8	47.4	18	21.3	19	80.9	96.7	18.00	96.70	12/12	na	na	na	na	na
Conductivity (umhos/cm)	300.00	300.00	302.00	296.00	297.00	296.00	296.00	297.00	293.00	292.00	294.00	292.00	292.00	302.00	12/12	na	na	na	na	na
Temperature (°C)	17.54	18.39	17.13	15.35	16.61	16.25	16.47	14.94	16.20	14.52	15.75	15.19	14.52	18.39	12/12	na	na	na	na	na
Hardness (mg/L)	114	102	114	116	122	102	100	102	108	100	112	98	98.00	122.00	12/12	na	na	na	na	104.00
TKN (mg/L)	0.54	0.53	0.59	0.54	0.54	0.55	0.65	0.64	0.64	0.82	0.58	0.56	0.53	0.82	12/12	na	na	na	na	0.54
TOC (mg/L)	9.70	9.00	8.90	8.80	8.40	8.80	9.80	9.20	9.70	9.70	8.50	8.80	8.40	9.80	12/12	na	na	na	na	8.90
Total Phosphorus (mg/L)	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	na	na	na	na	0.05 U
TSS (mg/L)	7.00	1.00 U	8.00	8.00	7.00	2.00	6.00	4.00	4.00	9.00	3.00	8.00	1.00	9.00	11 / 12	na	na	na	na	4.00
Metals (ug/L)																				
Aluminum	2.55	4.65	3.25	5.05	4.65	6.95	6.65	1100.25	7.15	23.25	4.05	4.55	2.55	1100.25	12 / 12	125.00	0.20	0.0560	1.00 UB	5.35
Antimony	0.095	0.10	0.087	0.083	0.092	0.092	0.072	0.096	0.053	0.051	0.094	0.097	0.05	0.10	12 / 12	31.00	0.01	0.0089	0.01 U	0.10
Arsenic	0.75	0.79	0.79	0.76	0.72	0.83	0.94	3.6	0.86	0.99	0.65	0.68	0.72	3.60	12 / 12	190.00	0.60	0.0440	0.04 U	0.90
Barium	41 J	41 J	42 J	39 J	40 J	43 J	42 J	89 J	42 J	45 J	39 J	35 J	35.00	89.00	12/ 12	3.80	0.01	0.0056	0.13 J	42.00 J
Beryllium	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	<0.048	<0.048	0 / 12	5.10	0.10	0.0480	0.05 U	0.05 U
Cadmium	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.19	0.015 U	0.015 U	0.015 U	0.015 U	<0.015	0.19	1 / 12	0.66	0.01	0.0150	0.02 U	0.02 U
Chromium	0.469	0.629	0.569	0.609	0.519	0.529	0.529	3.549 J	0.609	0.589	0.499	0.419	0.419	3.549	12 / 12	117.00	0.03	0.0420	0.239 UB	0.579
Cobalt	0.0740	0.0790	0.0750	0.0730	0.0560	0.0680	0.1170	2.0070	0.1370	0.1570	0.0690	0.0670	0.0560	2.0070	12 / 12	5.00	0.01	0.0038	0.018 UB	0.0740
Copper	0.097	0.095	0.17	0.097	0.12	0.11	0.12	9.0	0.079 U	0.16	0.15	0.10	0.08	9.00	11 / 12	6.40	0.01	0.0140	0.12	0.098
Lead	0.060	0.068	0.073	0.076	0.059	0.11	0.16	6.8	0.12	0.25	0.078	0.078	0.06	6.80	12 / 12	1.30	0.01	0.0049	0.0049 U	0.08
Magnesium	6700 J	6600 J	6700 J	6100 J	6600 J	6400 J	6500 J	5500 J	6500 J	6300 J	6400 J	6400 J	5500.00	6700.00	12 / 12	82000.00	25.00	0.0500	15.00 J	6500.00 J
Manganese	30	28	48	48	20	50	50	370	56	49	19	27	19.00	370.00	12 / 12	80.30	0.01	0.0100	0.18	46.00
Mercury	0.00048	0.00069	0.00041	0.0013	0.00063	0.0016	0.0011	0.017	0.00043	0.0015	0.00050	0.00067	0.00043	0.02	12 / 12	0.00	0.00	0.0002	0.0002 U	0.00
Nickel	0.14	0.31	0.56	0.20	0.13	0.18	0.20	5.0	0.25	0.27	0.12	0.18	0.12	5.00	12 / 12	80.00	0.04	0.0320	0.460	0.10
Selenium	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	<0.39	<0.39	0 / 12	5.00	0.30	0.3900	0.39 U	0.45 J
Silver	0.0252 UB	0.0232 UB	0.0192 UB	0.0147 UB	0.0151 UB	0.0115 UB	0.0139 UB	0.0272 UB	0.0127 UB	0.0119 UB	0.0137 UB	0.0162 UB	0.0115	0.0272	0 / 12	1.00	0.01	0.0043	0.0182 UB	0.0144 UB
Thallium	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.040	0.0035 U	0.0035 U	0.0035 U	0.0035 U	<0.0035	0.04	1 / 12	18.00	0.00	0.0035	0.0035 U	0.00 U
Zinc	0.397 UB	16.110	15.110	0.430 UB	0.430 UB	0.450 UB	0.670	23.110	0.530 UB	0.720	4.810	1.410	0.397	23.110	7/ 12	59.00	0.04	0.0350	1.810	0.410 UB
																				Sample Location ML-04

*ML-13 and ML-15 were added to the Marsden Lake South data set from the Marsden Lake North data set (see section 4.1.2.2)

Table E4. Marsden Lake (South) Surface Water Pesticides and PCBs Data

Parameter (ug/L)	SC-ML-SW-01	SC-ML-SW-02	SC-ML-SW-03	SC-ML-SW-04	SC-ML-SW-05	SC-ML-SW-06	SC-ML-SW-07	SC-ML-SW-08	SC-ML-SW-09	SC-ML-SW-10	SC-ML-SW-13*	SC-ML-SW-15*	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-FB-01	SC-ML-FD-XX-01
4,4'-DDT	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 UJ	0.05 U	0.05 UJ	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.01	0.05	0.05	0.05 U	0.05 U
4,4'-DDD	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.01	0.05	0.05	0.05 U	0.05 U
4,4'-DDE	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.00	0.01	0.05	0.05 U	0.05 U
Aldrin	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	none	0.05	0.05	0.05 U	0.05 U
Alpha-BHC	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	2.40	0.05	0.05	0.05 U	0.05 U
Beta-BHC	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	2.40	0.05	0.05	0.05 U	0.05 U
Chlordane, Technical	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	2.40	0.05	0.05	0.05 U	0.05 U
Delta-BHC	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	2.40	0.05	0.05	0.05 U	0.05 U
Dieldrin	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 UJ	0.05 U	0.05 UJ	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.00	0.05	0.05	0.05 U	0.05 U
Endosulfan I	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.03	0.05	0.05	0.05 U	0.05 U
Endosulfan II	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.03	0.10	0.05	0.05 U	0.05 U
Endosulfan Sulfate	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	none	0.10	0.05	0.05 U	0.05 U
Endrin	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.02	0.04	0.05	0.05 U	0.05 U
Endrin Aldehyde	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	none	0.10	0.05	0.05 U	0.05 U
Heptachlor	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.00	0.05	0.05	0.05 U	0.05 U
Heptachlor Epoxide	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.00	0.05	0.05	0.05 U	0.05 U
Lindane	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 12	0.08	0.05	0.05	0.05 U	0.05 U
Methoxychlor	0.05 UJ	0.05 UJ	0.05 UJ	0.05 UJ	0.05 UJ	0.05 UJ	0.05 UJ	0.05 UJ	0.05 UJ	0.05 U	0.05 UJ	0.05 U	<0.05	<0.05	0 / 12	none	0.40	0.05	0.05 U	0.05 UJ
Toxaphene	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	<0.5	<0.5	0 / 12	0.00	0.50	0.50	0.5 U	0.5 U
Aroclor-1016	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 12	none	0.50	0.10	0.1 U	0.1 U
Aroclor-1221	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 12	0.27	0.50	0.10	0.1 U	0.1 U
Aroclor-1232	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 12	0.50	0.50	0.10	0.1 U	0.1 U
Aroclor-1242	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 12	0.06	0.50	0.10	0.1 U	0.1 U
Aroclor-1248	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 12	0.01	0.50	0.10	0.1 U	0.1 U
Aroclor-1254	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 U	0.10 UJ	0.10 U	<0.1	<0.1	0 / 12	0.02	0.50	0.10	0.1 U	0.1 UJ
Aroclor-1260	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 12	1.30	0.50	0.10	0.1 U	0.1 U
																				Sample Location ML-04

*ML-13 and ML-15 were added to the Marsden Lake South data set from the Marsden Lake North data set (see section 4.1.2.2)

Table E5. Marsden Lake (South) Surface Water Semivolatile Organic Compounds (SVOCs) Data

Parameter (ug/L)	SC-ML-SW-01	SC-ML-SW-02	SC-ML-SW-03	SC-ML-SW-04	SC-ML-SW-05	SC-ML-SW-06	SC-ML-SW-07	SC-ML-SW-08	SC-ML-SW-09	SC-ML-SW-10	SC-ML-SW-13	SC-ML-SW-15	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-FB-01	SC-ML-FD-XX-01
Phenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	117.00	10.00	10.00	<10 U	<10 U
1,3-Dichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
1,2-Dichlorobenzene	<10 U	<10 U	<10 U	<10 UJ	<10 U	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 UJ	<10 U
4-Methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	72.20	10.00	10.00	<10 U	<10 U
Nitrobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
2,4-Dimethylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
2,4-Dichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
4-Chloroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	none	10.00	20.00	<20 U	<20 U
2-Methylnaphthalene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	2.08	10.00	10.00	<10 U	<10 U
2,4,5-Trichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	2.00	25.00	10.00	<10 U	<10 U
Dimethylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.00	10.00	10.00	<10 U	<10 U
3-Nitroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	none	10.00	20.00	<20 U	<20 U
4-Nitrophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	163.00	10.00	20.00	<20 U	<20 U
Diethylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	1.65	10.00	10.00	<10 U	<10 U
4-Nitroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	none	10.00	20.00	<20 U	<20 U
4-Bromophenyl-phenylether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	1.50	10.00	10.00	<10 U	<10 U
Phenanthrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	2.10	10.00	10.00	<10 U	<10 U
Fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	20.00	10.00	10.00	<10 U	<10 U
3,3'-Dichlorobenzidine	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	none	10.00	20.00	<20 U	<20 U
bis(2-Ethylhexyl)phthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.05	10.00	10.00	<10 U	<10 U
Benzo[k]fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Dibenz[a,h]anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
bis(2-Chloroethyl)ether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
1,4-Dichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	15.00	10.00	10.00	<10 U	<10 U
2-Methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	72.20	10.00	10.00	<10 U	<10 U
N-Nitroso-di-n-propylamine	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Isophorone	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Benzoic acid	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	41.60	10.00	20.00	<20 U	<20 U
1,2,4-Trichlorobenzene	<10 U	<10 U	<10 U	<10 UJ	<10 U	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10	<10	0 / 12	110.00	10.00	10.00	<10 UJ	<10 U
Hexachlorobutadiene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Hexachlorocyclopentadiene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
2-Chloronaphthalene	<10 U	<10 U	<10 U	<10 UJ	<10 U	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 UJ	<10 U
Acenaphthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	12.00	10.00	10.00	<10 U	<10 U
Acenaphthylene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	12.00	10.00	10.00	<10 U	<10 U
Dibenzofuran	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	20.40	10.00	10.00	<10 U	<10 U
4-Chlorophenyl-phenylether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
4,6-Dinitro-2-methylphenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	none	25.00	20.00	<20 U	<20 U
Hexachlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.00	10.00	10.00	<10 U	<10 U
Anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.03	10.00	10.00	<10 U	<10 U
Pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Benzo[a]anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.03	10.00	10.00	<10 U	<10 U
Di-n-octylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	30.00	10.00	10.00	<10 U	<10 U
Benzo[a]pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.01	10.00	10.00	<10 U	<10 U
Benzo[g,h,i]perylene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
2-Chlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Benzyl alcohol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	58.00	10.00	10.00	<10 U	<10 U
bis(2-chloroisopropyl)ether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Hexachloroethane	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10	<10	0 / 12	12.00	10.00	10.00	<10 UJ	<10 UJ
2-Nitrophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	163.00	10.00	10.00	<10 U	<10 U
bis(2-Chloroethoxy)methane	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Naphthalene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	81.00	10.00	10.00	<10 U	<10 U
4-Chloro-3-methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
2,4,6-Trichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	2.00	10.00	10.00	<10 U	<10 U
2-Nitroaniline	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	25.00	10.00	<10 U	<10 U
2,6-Dinitrotoluene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	50.00	10.00	10.00	<10 U	<10 U
2,4-Dinitrophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	none	25.00	20.00	<20 U	<20 U
2,4-Dinitrotoluene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	18.50	10.00	10.00	<10 U	<10 U
Fluorene	<10 U	<10 U	<10 U	<10 UJ	<10 U	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 U	<10 U	<10	<10	0 / 12	3.90	10.00	10.00	<10 UJ	<10 U
n-Nitrosodiphenylamine	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	24.50	10.00	10.00	<10 U	<10 U
Pentachlorophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 12	3.50	25.00	20.00	<20 U	<20 U
Di-n-butylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.10	10.00	10.00	<10 U	<10 U
Butylbenzylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	0.08	10.00	10.00	<10 U	<10 U
Chrysene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Benzo[b]fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
Indeno[1,2,3-cd]pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 12	none	10.00	10.00	<10 U	<10 U
																				Sample Location ML-04

*ML-13 and ML-15 were added to the Marsden Lake South data set from the Marsden Lake North data set (see section 4.1.2.2)

Table E6. Marsden Lake (South) Sediment TOC and Metals Data

Parameter	SC-ML-SD-01	SC-ML-SD-02	SC-ML-SD-03	SC-ML-SD-04	SC-ML-SD-05	SC-ML-SD-06	SC-ML-SD-07	SC-ML-SD-08	SC-ML-SD-09	SC-ML-SD-10	SC-ML-SD-13*	SC-ML-SD-15*	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-SD-FB-01	SC-ML-SD-FD XX-01
Total Organic Carbon (%)	23.00	27.00	21.00	14.00	24.00	19.00	8.20	11.00	11.00	8.30	29.00	34.00	8.20	27.00	na	na	na	na	na	15.00
GFAA Weight (g)	1.010	1.00	1.000	1.000	1.000	1.00	1.000	1.000	1.01	1.01	1.020	1.01	1.000	1.010	na	na	na	na	10.02	1.00
ICP Weight (g)	1.00	1.010	1.03	1.01	1.03	1.01	1.01	1.00	1.00	1.000	1.01	1.00	1.000	1.03	na	na	na	na	10.01	1.000
Percent Solids	9.50	6.50	8.80	9.0	6.20	8.20	7.30	3.60	8.90	7.30	6.20	6.40	3.60	9.50	na	na	na	na	na	10.60
Metals (mg/kg)																				
Aluminum	12000.00 J	6400.00 J	11000.00 J	8400.00	3300.00	9400.00	14000.00	12000.00	15000.00	14000.00	8400.00 J	8100.00 J	3300.00	15000.00	12 / 12	none	40.00	40.00	4.0 U	9300.00
Antimony	0.59	1.0	0.50	0.50 UJ	0.70 J	0.50 UJ	0.60 J	0.50 UJ	0.89 J	0.50 J	0.98	0.80	0.50	1.0	9 / 12	none	40.00	0.50	0.050 UJ	0.60 J
Arsenic	9.50	7.30	11.00 J	6.80	8.30	6.70	8.0	9.80	10.00	8.60	12.00 J	12.00 J	6.70	12.00	12 / 12	6.00	1.50	1.00	0.10 U	6.40
Barium	190.00	120.00	140.00	130.00	110.00	130.00	120.00	130.00	150.00	140.00	130.00	120.00	110.00	190.00	12 / 12	none	4.00	4.00	0.40 U	130.00
Beryllium	1.0 U	0.99 U	0.97 U	0.99 U	0.97 U	0.99 U	0.99 U	1.0 U	1.0 U	1.0 U	0.99 U	1.0 U	0.97	1.00	0 / 12	none	4.00	1.00	0.10 U	1.0 U
Cadmium	1.60 J	0.81 J	1.10 J	0.50 J	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	1.40 J	1.30 J	0.50	1.60	6 / 12	0.60	0.50	0.50	0.050 UJ	0.50 J
Chromium	21.00 J	11.00 J	22.00 J	16.00 J	6.80 J	16.00 J	30.00 J	22.00 J	26.00 J	26.00 J	17.00 J	18.00 J	6.80	30.00	12 / 12	26.00	1.00	1.00	0.10 U	16.00 J
Cobalt	11.00	4.80	9.70 U	9.90 U	9.70 U	9.90 U	11.00	10.00 U	11.00	10.00 U	9.90 U	10.00 U	4.80	11.00	4 / 12	none	10.00	10.00	1.0 U	10.00 U
Copper	25.00	12.00	22.00	17.00	8.70	19.00	52.00	29.00	31.00	29.00	25.00	24.00	8.70	52.00	12 / 12	16.00	1.00	1.00	0.11	17.00
Lead	46.00 J	42.00 J	51.00 J	28.00 J	22.00 J	33.00 J	48.00 J	45.00 J	44.00 J	40.00 J	63.00 J	62.00 J	22.00	63.00	12 / 12	31.00	1.00	1.00	0.10 UJ	27.00 J
Magnesium	3200.00	2300.00	3300.00	2300.00	1700.00	2600.00	4300.00	3600.00	4300.00	4000.00	3300.00	2500.00	1700.00	4300.00	12 / 12	none	40.00	40.00	4.0 U	2500.00
Manganese	450.00 J	330.00 J	280.00 J	380.00 J	680.00 J	320.00 J	220.00 J	170.00 J	270.00 J	200.00 J	500.00 J	460.00 J	170.00	680.00	12 / 12	460.00	2.00	2.00	0.20 UJ	430.00 J
Mercury	0.18 J	0.15 J	0.16 J	0.12	0.13	0.14	0.11	0.17	0.18	0.15	0.16	0.17 J	0.11	0.18	12 / 12	0.20	0.10	0.10	0.01 U	0.10
Nickel	21.00	9.90 U	15.00 U	14.00	9.70 U	13.00	23.00	20.00	22.00	22.00	17.00	14.00	9.70	23.00	9 / 12	16.00	10.00	10.00	1.0 U	13.00
Selenium	1.0	1.0 U	1.0 J	1.20	1.0 U	1.0	1.0 U	1.0 U	0.99 U	1.20	1.40 J	0.99 U	0.99	1.40	5/ 12	none	40.00	1.00	0.10 U	1.40
Silver	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.50 U	0.49 U	0.50 U	0.49	0.50	0 / 12	1.00	0.50	0.50	0.050 U	0.50 U
Thallium	0.99 U	1.0 U	1.0 U	1.0 U	1.0 U	1.0 U	1.0 U	1.0 U	0.99 U	1.0 U	0.98 U	0.99 U	0.98	1.00	0 / 12	none	40.00	1.00	0.10 U	1.0 U
Zinc	100.00	55.00	87.00	66.00	55.00	53.00	88.00	80.00	80.00	73.00	72.00	140.00	53.00	140.00	12 / 12	120.00	5.00	5.00	0.50 U	63.00
																				Sample Location ML-04

*ML-13 and ML-15 were added to the Marsden Lake South data set from the Marsden Lake North data set (see section 4.1.2.2)

TOC and % Solids not provided by lab (note: FB was water sample)
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Table E7. Marsden Lake (South) Sediment Pesticides and PCBs Data

Parameter (mg/kg)	SC-ML-SD-01	SC-ML-SD-02	SC-ML-SD-03	SC-ML-SD-04	SC-ML-SD-05	SC-ML-SD-06	SC-ML-SD-07	SC-ML-SD-08	SC-ML-SD-09	SC-ML-SD-10	SC-ML-SD-13*	SC-ML-SD-15*	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-SD-FB 01	SC-ML-SD-FD-XX-01
% SOLIDS	9.00	7.00	7.90	12.00	6.90	10.00	12.00	6.70	11.00	13.00	4.30	4.40	7.00	13.00	0 / 12	none	none	none	na	13.00
4,4'-DDT	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.01	0.01	0.08	0.05 U	0.08 U
4,4'-DDD	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	**0.095****	0.08 U	<0.08	<0.08	1/12	0.01	0.01	0.08	0.05 U	0.08 U
4,4'-DDE	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.01	0.01	0.08	0.05 U	0.08 U
Aldrin	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.00	0.01	0.08	0.05 U	0.08 U
Alpha-BHC	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.01	0.01	0.08	0.05 U	0.08 U
Beta-BHC	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.01	0.01	0.08	0.05 U	0.08 U
Chlordane, Technical	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.01	0.01	0.08	0.05 U	0.08 U
Delta-BHC	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.00	0.01	0.08	0.05 U	0.08 U
Dieldrin	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.00	0.01	0.08	0.05 U	0.08 U
Endosulfan I	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.05	0.01	0.08	0.05 U	0.08 U
Endosulfan II	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.05	0.01	0.08	0.05 U	0.08 U
Endosulfan Sulfate	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	none	0.01	0.08	0.05 U	0.08 U
Endrin	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.00	0.01	0.08	0.05 U	0.08 U
Endrin Aldehyde	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	none	0.01	0.08	0.05 U	0.08 U
Heptachlor	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.00	0.01	0.08	0.05 U	0.08 U
Heptachlor Epoxide	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.01	0.01	0.08	0.05 U	0.08 U
Lindane	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	<0.08	<0.08	0 / 12	0.00	0.01	0.08	0.05 U	0.08 U
Methoxychlor	0.08 UJ	0.08 UJ	0.08 UJ	0.08 UJ	0.08 UJ	0.08 UJ	0.08 UJ	0.08 U	0.08 UJ	0.08 U	0.08 UJ	0.08 UJ	<0.08	<0.08	0 / 12	0.02	0.01	0.08	0.05 U	0.08 UJ
Toxaphene	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	<0.17	<0.17	0 / 12	0.03	0.13	0.17	0.50 U	0.17 U
Aroclor-1016	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 12	0.01	1.00	0.33	0.1 U	0.33 U
Aroclor-1221	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 12	none	1.00	0.33	0.1 U	0.33 U
Aroclor-1232	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 12	none	1.00	0.33	0.1 U	0.33 U
Aroclor-1242	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 12	none	1.00	0.33	0.1 U	0.33 U
Aroclor-1248	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 12	0.03	1.00	0.33	0.1 U	0.33 U
Aroclor-1254	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 12	0.06	1.00	0.33	0.1 U	0.33 U
Aroclor-1260	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 12	0.01	1.00	0.33	0.1 U	0.33 U
																				Sample Location ML-04

*ML-13 and ML-15 were added to the Marsden Lake South data set from the Marsden Lake North data set (see section 4.1.2.2)

**Value was detected in the Field duplicate for ML-13

Table E8. Marsden Lake (South) Sediment Semivolatile Organic Compounds (SVOCs) Data

Parameter (ug/kg)	SC-ML-SD-01	SC-ML-SD-02	SC-ML-SD-03	SC-ML-SD-04	SC-ML-SD-05	SC-ML-SD-06	SC-ML-SD-07	SC-ML-SD-08	SC-ML-SD-09	SC-ML-SD-10	SC-ML-SD-13*	SC-ML-SD-15*	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-ML-SD-FB 01	SC-ML-SD-FD-XX-01
Phenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
1,3-Dichlorobenzene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	1700.00	330.00	1900.00	82.00 U	1500 U
1,2-Dichlorobenzene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	340.00	330.00	1900.00	82.00 UJ	1500 U
4-Methylphenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Nitrobenzene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
2,4-Dimethylphenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
2,4-Dichlorophenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
4-Chloroaniline	16000.00 U	17000.00 U	16000.00 U	15000.00 UJ	19000.00 UJ	15000.00 UJ	16000.00 UJ	19000.00 UJ	15000.00 UJ	14000.00 UJ	17000.00 U	17000.00 U	<14000	<19000	0 / 12	none	330.00	19000.00	160.00 U	15000 UJ
2-Methylnaphthalene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	70.00	330.00	1900.00	82.00 U	1500 U
2,4,5-Trichlorophenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Dimethylphthalate	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
3-Nitroaniline	3200.00 U	3400.00 U	3300.00 U	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 U	3500.00 U	<2800	<3900	0 / 12	none	800.00	3900.00	160.00 U	3000 U
4-Nitrophenol	3200.00 U	3400.00 U	3300.00 U	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 U	3500.00 U	<2800	<3900	0 / 12	none	330.00	3900.00	160.00 U	3000 U
Diethylphthalate	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	630.00	330.00	1900.00	82.00 U	1500 U
4-Nitroaniline	3200.00 U	3400.00 U	3300.00 U	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 U	3500.00 U	<2800	<3900	0 / 12	none	330.00	3900.00	160.00 U	3000 U
4-Bromophenyl-phenylether	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	1300.00	330.00	1900.00	82.00 U	1500 U
Phenanthrene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	560.00	330.00	1900.00	82.00 U	1500 U
Fluoranthene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	190.00	330.00	1900.00	82.00 U	1500 U
3,3'-Dichlorobenzidine	3200.00 U	3400.00 UJ	3300.00 UJ	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 U	3500.00 U	<2800	<3900	0 / 12	none	330.00	3900.00	160.00 U	3000 U
bis(2-Ethylhexyl)phthalate	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	25000.00	330.00	1900.00	82.00 U	1500 U
Benzo[k]fluoranthene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	240.00	330.00	1900.00	82.00 U	1500 U
Dibenz[a,h]anthracene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	60.00	330.00	1900.00	82.00 U	1500 U
bis(2-Chloroethyl)ether	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
1,4-Dichlorobenzene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	350.00	330.00	1900.00	82.00 U	1500 U
2-Methylphenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
N-Nitroso-di-n-propylamine	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Isophorone	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Benzoic acid	3200.00 U	3400.00 U	3300.00 U	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 U	3500.00 U	<2800	<3900	0 / 12	none	330.00	3900.00	160.00 U	3000 U
1,2,4-Trichlorobenzene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	9200.00	330.00	1900.00	82.00 UJ	1500 U
Hexachlorobutadiene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Hexachlorocyclopentadiene	1600.00 U	1700.00 UJ	1600.00 UJ	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
2-Chloronaphthalene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 UJ	1500 U
Acenaphthene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	620.00	330.00	1900.00	82.00 U	1500 U
Acenaphthylene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	620.00	330.00	1900.00	82.00 U	1500 U
Dibenzofuran	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	2000.00	330.00	1900.00	82.00 U	1500 U
4-Chlorophenyl-phenylether	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
4,6-Dinitro-2-methylphenol	3200.00 UJ	3400.00 UJ	3300.00 UJ	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 UJ	3500.00 UJ	<2800	<3900	0 / 12	none	330.00	3900.00	160.00 U	3000 U
Hexachlorobenzene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Anthracene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	220.00	330.00	1900.00	82.00 U	1500 U
Pyrene	1600.00 U	1700.00 U	1600.00 U	1500.00 UJ	1900.00 UJ	1500.00 UJ	1600.00 UJ	1900.00 UJ	1500.00 UJ	1400.00 UJ	1700.00 U	1700.00 U	<1400	<1900	0 / 12	490.00	330.00	1900.00	82.00 U	1500 UJ
Benzo[a]anthracene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	320.00	330.00	1900.00	82.00 U	1500 U
Di-n-octylphthalate	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Benzo[a]pyrene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	370.00	330.00	1900.00	82.00 U	1500 U
Benzo[g,h,i]perylene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	170.00	330.00	1900.00	82.00 U	1500 U
2-Chlorophenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Benzyl alcohol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
bis(2-chloroisopropyl)ether	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Hexachloroethane	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	1000.00	330.00	1900.00	82.00 UJ	1500 U
2-Nitrophenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
bis(2-Chloroethoxy)methane	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Naphthalene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	480.00	330.00	1900.00	82.00 U	1500 U
4-Chloro-3-methylphenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
2,4,6-Trichlorophenol	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
2-Nitroaniline	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	800.00	1900.00	82.00 U	1500 U
2,6-Dinitrotoluene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	21.00	330.00	1900.00	82.00 U	1500 U
2,4-Dinitrophenol	3200.00 UJ	3400.00 UJ	3300.00 UJ	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 UJ	3500.00 UJ	<2800	<3900	0 / 12	none	800.00	3900.00	160.00 U	3000 U
2,4-Dinitrotoluene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	42.90	330.00	1900.00	82.00 U	1500 U
Fluorene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	190.00	330.00	1900.00	82.00 UJ	1500 U
n-Nitrosodiphenylamine	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Pentachlorophenol	3200.00 U	3400.00 U	3300.00 U	3100.00 U	3700.00 U	2900.00 U	3100.00 U	3900.00 U	3100.00 U	2800.00 U	3300.00 U	3500.00 U	<2800	<3900	0 / 12	none	800.00	3900.00	160.00 U	3000 U
Di-n-butylphthalate	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	11000.00	330.00	1900.00	82.00 U	1500 U
Butylbenzylphthalate	1600.00 UJ	1700.00 UJ	**870.00** J	1500.00 UJ	1900.00 UJ	1500.00 UJ	1600.00 UJ	1900.00 UJ	1500.00 UJ	1400.00 UJ	1700.00 UJ	1700.00 UJ	870.00	<1900	0 / 12	11000.00	330.00	1900.00	82.00 U	1500 UJ
Chrysene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	340.00	330.00	1900.00	82.00 U	1500 U
Benzo[b]fluoranthene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	none	330.00	1900.00	82.00 U	1500 U
Indeno[1,2,3-cd]pyrene	1600.00 U	1700.00 U	1600.00 U	1500.00 U	1900.00 U	1500.00 U	1600.00 U	1900.00 U	1500.00 U	1400.00 U	1700.00 U	1700.00 U	<1400	<1900	0 / 12	200.00	330.00	1900.00	82.00 U	1500 U

*ML-13 and ML-15 were added to the Marsden Lake South data set from the Marsden Lake North data set (see section 4.1.2.2)

Sample Location ML 04

Table E9. Marsden Lake (South) Dixon Test for outliers -- Surface water data

*	Al	Sb	As	Ba	Be	Cd	Cr	Co	Cu	Pb	Mg	Mn	Hg	Ni	Se	Ag	Tl	Zn
	2.300	0.051	0.720	39.000	0.048	0.015	0.420	0.049	0.079	0.060	5500.000	20.000	0.000	0.130	0.390	0.005	0.004	0.280
	3.000	0.053	0.750	40.000	0.048	0.015	0.470	0.061	0.095	0.060	6100.000	28.000	0.000	0.140	0.390	0.006	0.004	0.320
	4.400	0.072	0.760	41.000	0.048	0.015	0.480	0.066	0.097	0.070	6300.000	30.000	0.001	0.180	0.390	0.007	0.004	0.320
	4.400	0.083	0.790	41.000	0.048	0.015	0.480	0.067	0.097	0.070	6400.000	48.000	0.001	0.200	0.390	0.008	0.004	0.340
	4.800	0.087	0.790	42.000	0.048	0.015	0.520	0.068	0.110	0.080	6500.000	48.000	0.001	0.200	0.390	0.009	0.004	0.420
	6.400	0.092	0.830	42.000	0.048	0.015	0.540	0.072	0.120	0.110	6500.000	49.000	0.001	0.250	0.390	0.009	0.004	0.560
	6.700	0.092	0.860	42.000	0.048	0.015	0.560	0.110	0.120	0.120	6600.000	50.000	0.001	0.270	0.390	0.013	0.004	0.610
	6.900	0.095	0.940	43.000	0.048	0.015	0.560	0.130	0.160	0.160	6600.000	50.000	0.002	0.310	0.390	0.017	0.004	15.000
	23.000	0.096	0.990	45.000	0.048	0.015	0.580	0.150	0.170	0.250	6700.000	56.000	0.002	0.560	0.390	0.019	0.004	16.000
	1100.000	0.100	3.600	89.000	0.048	0.190	3.500	2.000	9.000	6.800	6700.000	370.000	0.017	5.000	0.390	0.021	0.040	23.000
Dixon outlier test																		
x(n)	**1100.000**	0.100	**3.600**	**89.000**	0.048	**0.190**	**3.500**	**2.000**	**9.000**	**6.800**	6700.000	**370.000**	**0.017**	**5.000**	0.390	0.021	**0.040**	23.000
x(n-1)	23.000	0.096	0.990	45.000	0.048	0.015	0.580	0.150	0.170	0.250	6700.000	56.000	0.002	0.560	0.390	0.019	0.004	16.000
x(2)	3.000	0.053	0.750	40.000	0.048	0.015	0.470	0.061	0.095	0.060	6100.000	28.000	0.000	0.140	0.390	0.006	0.004	0.320
C=	0.982	0.085	0.916	0.898	#DIV/0!	1.000	0.964	0.954	0.992	0.972	0.000	0.918	0.928	0.914	#DIV/0!	0.131	1.000	0.309
C(.05,1.0)	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477	0.477
	*****		*****	*****		****	****	****	*****	****		****	****	*****			****	

*Values for each analyte are arranged from the smallest to the largest value (ug/L). (Refer to Table E3 to determine which location each value is associated with).
X(n)= largest number in the series
X(n-1)= the second largest number in the series
X(2)= the next to smallest number in the series
C= the test statistic
C(0.05)= the critical table value[1]

If C exceeds the critical table value then X(n) is an outlier and should be further investigated[1]. (Bold values are outliers).

[1] USEPA. 1998. Guidance for Data Quality Assessment: Practical Assessment for Data Analysis. Office of Environmental Information. Washington D.C. July 2000. EPA/600/R-96/084

Figure E1. Numerical comparison of Al concentrations to background

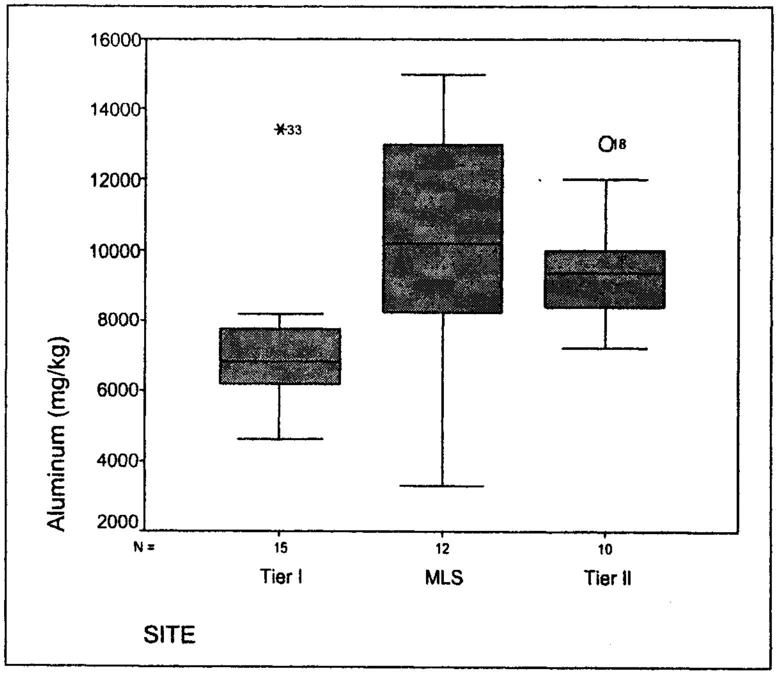


Figure E2. Numerical comparison of As concentrations to background

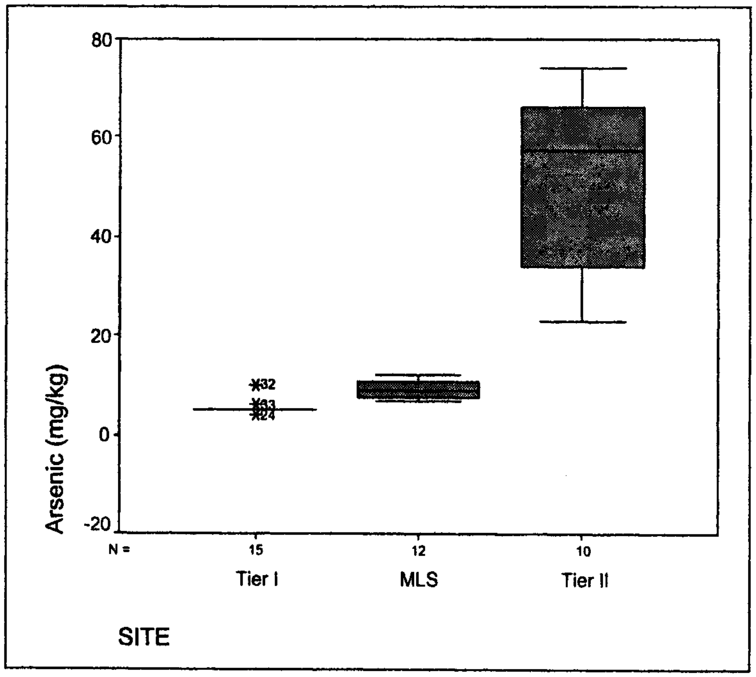


Figure E3. Numerical comparison of Cd concentrations to background

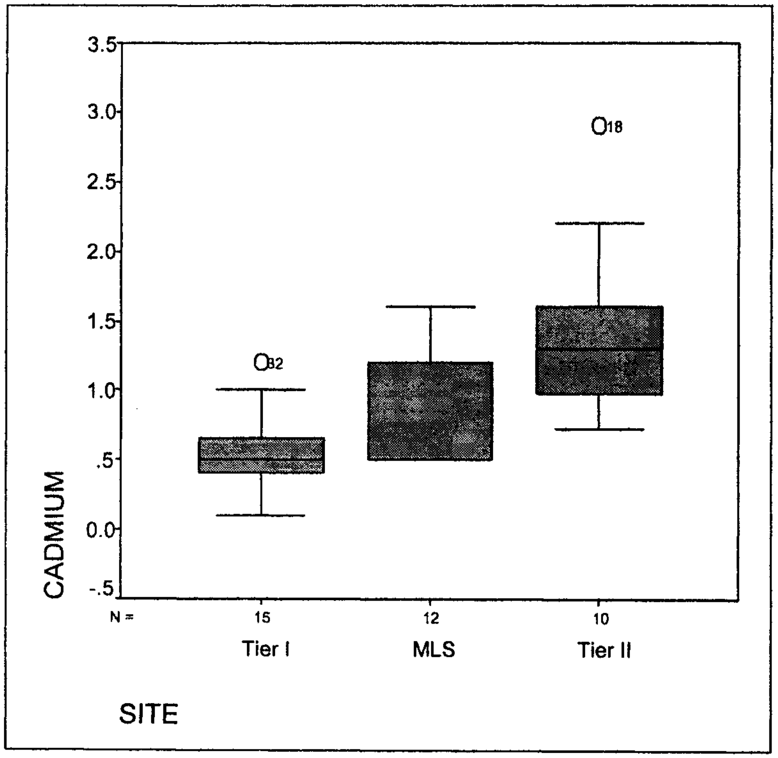


Figure E4. Numerical comparison of Cu concentrations to background

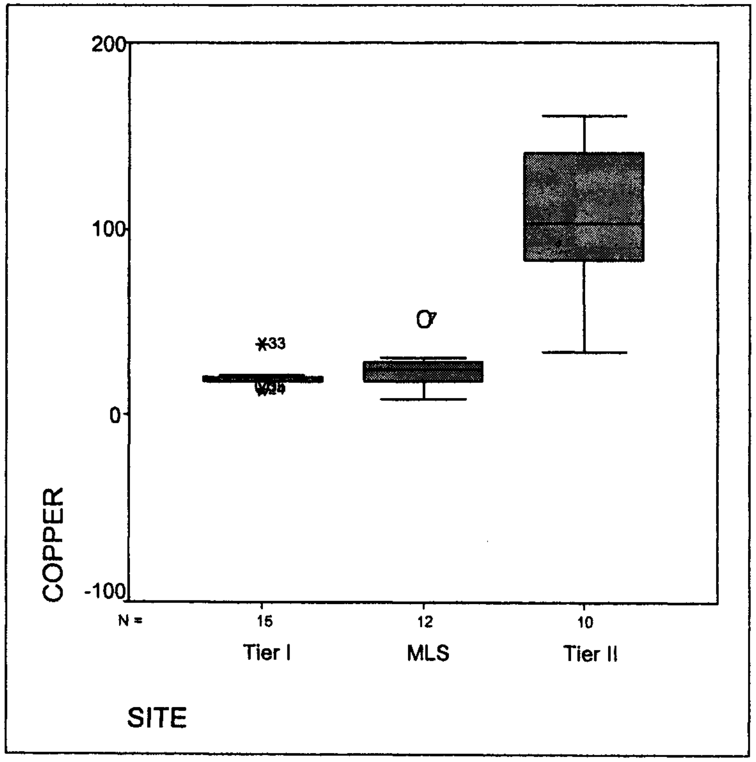
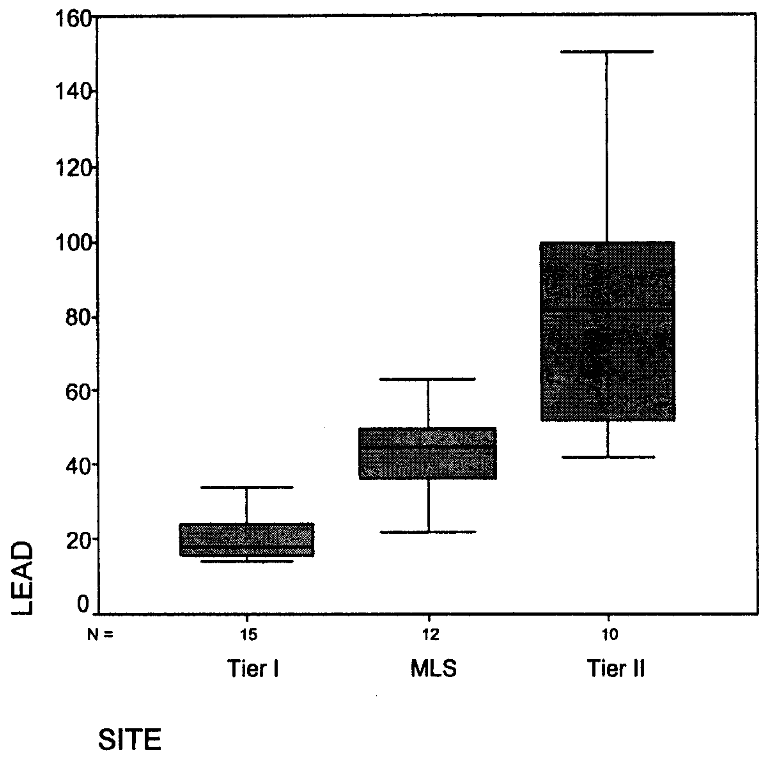


Figure E5. Numerical comparison of Pb concentrations to background



APPENDIX F

Pond G Sampling Locations and Analytical Measurement Results

Table of Contents

Pond G Field and Analytical Data

The map of Pond G (Fig.F1), shows the location where each media sample was taken. Global Positioning System (GPS) coordinates were used to plot the locations. These coordinates recorded in the field, during sampling, are reported in tables F1 and F2. The map image is from an orthophoto taken in 1997, obtained from Wenck Associates, Inc., Maple Plain, Minnesota.

The tables F3 through F8 summarize the data for Pond G surface water and sediment samples taken during the site characterization. The summaries include the field and analytical result for each analyte at each sample location. The minima and maxima are reported as well as the detection frequency (DF), risk based action level (RBAL), target detection limit and achieved detection limit. The RBALs and target detection limits are those that were defined in the QAPP. Summary data used for statistical analysis such as the central tendencies, standard deviations and confidence intervals are report in tables found in Appendix A.

The number of digits reported for each value in the following tables does not necessarily reflect the precision and accuracy of the measurement. The data tables present significant figures as they were originally reported by the analyzing laboratory. Significant figures reported for the analytes were based on the analytical detection limit values. However, some sample values have more significant figures than others to demonstrate that blank correction was discontinued, as was requested by USEPA comments on the draft version of this report (see Appendix R). Data quality is discussed in Appendix K.

The results are reported in the following manner to facilitate data review.

- If the detection limit is sample-specific (as it was for the pesticides in sediment), the value reported for the achieved detection limit for a site is the highest detection limit for that parameter for all samples (including the field duplicate) from that site.

- If a parameter was not detected in any samples from a site (e.g. Pond G, water, alpha-BHC) then the minimum and maximum values represent the lowest and highest achieved detection limits for those data.

- Any concentrations that were below the detection limit but still quantified are identified in Volume 2 of this report as estimated values (J-flagged). These values are reported and used uncensored in the following tables.

- If a parameter was undetected in at least one but not all samples from a site, then the minimum reflects the lowest detection limit of the undetected samples. If there was a J-flagged value lower than the highest undetected value, the J-flagged value was reported as the maximum because the value was actually detected but estimated.

- Field duplicates (FD) were used as quality control samples and were not included in the reported minima and maxima columns. However, if an analyte was detected in the FD and not in it's corresponding sample, then the FD detection was counted in the detection frequency column (only in the numerator).

- When laboratory replicate samples (R1, R2) were available, the values from the primary sample (R1) were reported. See Appendix K section K1.2.1 for explanation.

Definition of Data Qualifiers (Flags)

During the data validation process, all laboratory data were assigned appropriate data validation flags and reason codes. Validation flags used in the following tables are defined as follows:

No flag	When the analyte value reported is detected and usable. The integrity of the analyte's identification, accuracy, and precision has been validated
"U"	When the material was analyzed for, but not detected above the level of the associated detection limit value.
"UB"	When an analyte was detected in the method blank, then the analyte is considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with.
"J"	When the associated value is an estimated quantity. Indicating there is cause to question accuracy or precision of the reported value. The value is still used in the risk assessment.
"UJ"	When the analyte was analyzed for, but not detected, above the associated value, however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"UBJ"	When an analyte was detected in the method blank, then the analyte is considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with; however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"R"	When the analyte value reported is unusable. The integrity of the analyte's identification, accuracy, precision, or sensitivity has raised significant question as to the reality of the information presented.

Other information regarding data validation can be found in Volume 2 (CD ROM version) of the report.

Note: References cited in the text of this appendix can be found in the reference section of the main Tier II Ecological Risk Assessment Report.

Figure F1. Pond G Surface Water and Sediment Site Characterization

SC= Site Characterization
PG= Pond G
SW= Surface Water
SD= Sediment

Table F1. Pond G Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	North*			West*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-PG-SW-01	28-Sep-99	4"	45	5	14.43	93	9	55.52
SC-PG-SW-02	28-Sep-99	4"	45	5	14	93	9	56.12
SC-PG-SW-03 SC-PG-SW-FD	28-Sep-99	4"	45	5	15.21	93	9	56.78

Table F2. Pond G Sediment Sampling GPS Coordinates

Sample Number	Date	Sample depth	North*			West*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-PG-SD-01	28-Sep-99	12"	45	5	14.43	93	9	55.52
SC-PG-SD-02	28-Sep-99	12"	45	5	14	93	9	56.12
SC-PG-SD-03	28-Sep-99	12"	45	5	15.21	93	9	56.78

* The GPS unit used in the study was a military-issue Precision Light Weight GPS Receiver (PLGR). The Department of Defense (DoD) intentionally places errors in the GPS navigation and timing signal, in a process called selective availability (Department of the Army 1993). Errors resulting from this process are unpredictable and can produce significant horizontal and elevation errors. Military GPS receivers must have crypto keys loaded to detect and nullify selective availability errors, in order to allow for more accurate position data (Department of the Army 1993). The PLGR was equipped with a crypto key, and was therefore not subject to the selective availability and spoofing errors that affect non-military GPS. (The only exception was during the second quarter). The PLGR operations manual (Department of the Army 1995) states that with crypto keys installed, the PLGR provides horizontal 2-D position accuracies better than the worst-case error of 10 meters.

Table F3. Pond G Surface Water Quality and Metals Data

Parameter	SC-PG-SW-01	SC-PG-SW-02	SC-PG-SW-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-FB-01	SC-PG-FD-XX
pH	6.25	6.74	6.64	6.25	6.74	6.54	0.26	3 / 3	na	na	na	na	na
DO (mg/L)	3.76	6.11	2.51	2.51	6.11	4.13	1.83	3 / 3	na	na	na	na	na
DO %	35.1	57.5	23.2	23.2	57.5	38.6	17.4	3 / 3	na	na	na	na	na
Conductivity (umhos/cm)	112	111	114	111	114	112.33	1.53	3 / 3	na	na	na	na	na
Temperature (C)	12.03	12.47	12.26	12.03	12.47	12.25	0.22	3 / 3	na	na	na	na	na
Hardness (mg/L)	44	52	44	44	52	47	4.6	3 / 3	na	na	na	na	na
Metals (ug/L)													
Al	241	451	1601	241	1601	764	732	3 / 3	125	0.2	0.056	1.9 UB	810
Sb	0.51	0.49	1.1	0.49	1.1	0.70	0.35	3 / 3	31	0.008	0.0089	0.010 U	1.3
As	3.7	3.7	5.1	3.7	5.1	4.2	0.81	3 / 3	190	0.6	0.044	0.044 U	4.2
Ba	51.016 J	56.016 J	100.016 J	51.016	100.016	69	27	3 / 3	3.8	0.006	0.0056	0.126 J	74.016 J
Be	0.048 U	0.048 U	0.120	<0.048	0.12	0.07	0.04	1 / 3	5.1	0.1	0.048	0.048 U	0.053
Cd	0.027	0.041	0.140	0.027	0.14	0.07	0.06	3 / 3	0.66	0.01	0.015	0.015 U	0.061
Cr	0.912 J	1.292 J	3.492 J	0.912	3.492	1.89	1.4	3 / 3	117	0.03	0.042	0.134 UB	1.992 J
Co	0.866	1.206	2.406	0.866	2.406	1.5	0.8	3 / 3	5	0.01	0.0038	0.01 UB	1.706
Cu	1.5	2.0	5.6	1.5	5.6	3.0	2.2	3 / 3	6.4	0.01	0.014	0.15	2.9
Pb	2.4	3.3	7.9	2.4	7.9	4.5	3.0	3 / 3	1.3	0.008	0.0049	0.31	4.3
Mg	4300.49 J	4300.49 J	4600.49 J	4300.49	4600.49	4400.49	173	3 / 3	82000	25	0.05	1.0 UB	500.49 J
Mn	320	380	340	320	380	347	31	3 / 3	80.3	0.008	0.01	0.072 UB	550
Hg	0.0027 U	0.0046 U	0.017 U	<0.0027	<0.017	na	na	0 / 3	0.003	0.0002	0.00018	0.00029 U	0.0074 UB
Ni	2.7	3.2	5.8	2.7	5.8	3.9	1.7	3 / 3	80	0.04	0.032	0.077	4.2
Se	0.39 U	0.39 U	0.41	<0.39	0.41	0.40	0.01	1 / 3	5	0.3	0.39	0.39 U	0.39 U
Ag	0.0191 UB	0.016 UB	0.0261 UB	0.016	0.0261	na	na	3/3	1	0.005	0.0043	0.0114 UB	0.013 UB
Tl	0.0061	0.012	0.037	0.0061	0.037	0.02	0.02	3 / 3	18	0.002	0.0035	0.0035 U	0.015 UB
Zn	0.58	2.8	14	0.58	14	5.8	7.2	3 / 3	59	0.04	0.035	2.4	5.2
													Sample Location PG-03

Table F4. Pond G Surface Water Pesticides and PCBs Data

Parameter (ug/L)	SC-PG-SW-01	SC-PG-SW-02	SC-PG-SW-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-FB-01	SC-PG-FD-XX
4,4'-DDT	0.05 UJ	0.05 UJ	0.05 UJ	<0.05	<0.05	na	na	0 / 10	0.01	0.05	0.05	0.05 UJ	0.05 UJ
4,4'-DDD	0.05 UJ	0.05 UJ	0.05 UJ	<0.05	<0.05	na	na	0 / 10	0.01	0.05	0.05	0.05 UJ	0.05 UJ
4,4'-DDE	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	0.001	0.01	0.05	0.05 U	0.05 U
Aldrin	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	none	0.05	0.05	0.05 U	0.05 U
Alpha-BHC	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	2.4	0.05	0.05	0.05 U	0.05 U
Beta-BHC	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	2.4	0.05	0.05	0.05 U	0.05 U
Chlordane, Technical	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	2.4	0.05	0.05	0.05 U	0.05 U
Delta-BHC	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	2.4	0.05	0.05	0.05 U	0.05 U
Dieldrin	0.05 UJ	0.05 UJ	0.05 UJ	<0.05	<0.05	na	na	0 / 10	0.000026	0.05	0.05	0.05 UJ	0.05 UJ
Endosulfan I	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	0.031	0.05	0.05	0.05 U	0.05 U
Endosulfan II	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	0.031	0.1	0.05	0.05 U	0.05 U
Endosulfan Sulfate	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	none	0.1	0.05	0.05 U	0.05 U
Endrin	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	0.016	0.04	0.05	0.05 U	0.05 U
Endrin Aldehyde	0.05 UJ	0.05 UJ	0.05 UJ	<0.05	<0.05	na	na	0 / 10	none	0.1	0.05	0.05 UJ	0.05 UJ
Heptachlor	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	0.0038	0.05	0.05	0.05 U	0.05 U
Heptachlor Epoxide	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	0.0038	0.05	0.05	0.05 U	0.05 U
Lindane	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	0.08	0.05	0.05	0.05 U	0.05 U
Methoxychlor	0.05 U	0.05 U	0.05 U	<0.05	<0.05	na	na	0 / 10	none	0.4	0.05	0.05 U	0.05 U
Toxaphene	0.5 U	0.5 U	0.5 U	<0.5	<0.5	na	na	0 / 10	0.0013	0.5	0.5	0.5 U	0.5 U
Aroclor-1016	0.1 U	0.1 U	0.1 U	<0.1	<0.1	na	na	0 / 10	none	0.5	0.1	0.1 U	0.1 U
Aroclor-1221	0.1 U	0.1 U	0.1 U	<0.1	<0.1	na	na	0 / 10	0.27	0.5	0.1	0.1 U	0.1 U
Aroclor-1232	0.1 U	0.1 U	0.1 U	<0.1	<0.1	na	na	0 / 10	0.5	0.5	0.1	0.1 U	0.1 U
Aroclor-1242	0.1 U	0.1 U	0.1 U	<0.1	<0.1	na	na	0 / 10	0.06	0.5	0.1	0.1 U	0.1 U
Aroclor-1248	0.1 U	0.1 U	0.1 U	<0.1	<0.1	na	na	0 / 10	0.01	0.5	0.1	0.1 U	0.1 U
Aroclor-1254	0.1 U	0.1 U	0.1 U	<0.1	<0.1	na	na	0 / 10	0.02	0.5	0.1	0.1 U	0.1 U
Aroclor-1260	0.1 U	0.1 U	0.1 U	<0.1	<0.1	na	na	0 / 10	1.3	0.5	0.1	0.1 U	0.1 U

Sample Location PG-03

Table F5. Pond G Surface Water Semivolatile Organic Compounds (SVOCs)

Parameter (ug/L)	SC-PG-SW-01	SC-PG-SW-02	SC-PG-SW-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-FB-01	SC-PG-FD-XX
Phenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	117	10	10	10 U	10 U
1,3-Dichlorobenzene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
1,2-Dichlorobenzene	<10 UJ	<10 UJ	<10 UJ	<10	<10	na	na	0 / 3	none	10	10	10 UJ	10 UJ
4-Methylphenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	72.2	10	10	10 U	10 U
Nitrobenzene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
2,4-Dimethylphenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
2,4-Dichlorophenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
4-Chloroaniline	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	none	10	20	20 U	20 U
2-Methylnaphthalene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	2.08	10	10	10 U	10 U
2,4,5-Trichlorophenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	2	25	10	10 U	10 U
Dimethylphthalate	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	0.001	10	10	10 U	10 U
3-Nitroaniline	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	none	10	20	20 U	20 U
4-Nitrophenol	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	163	10	20	20 U	20 U
Diethylphthalate	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	1.65	10	10	10 U	10 U
4-Nitroaniline	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	none	10	20	20 U	20 U
4-Bromophenyl-phenylether	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	1.5	10	10	10 U	10 U
Phenanthrene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	2.1	10	10	10 U	10 U
Fluoranthene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	20	10	10	10 U	10 U
3,3'-Dichlorobenzidine	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	none	10	20	20 U	20 U
bis(2-Ethylhexyl)phthalate	<10 U	**8.3 J**	**9.3 J**	**8.3**	**9.3**	8.8	0.707	2 / 3	0.05	10	10	**10**	**10**
Benzo[k]fluoranthene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
bis(2-Chloroethyl)ether	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Dibenz[a,h]anthracene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
1,4-Dichlorobenzene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	15	10	10	10 U	10 U
2-Methylphenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	72.2	10	10	10 U	10 U
N-Nitroso-di-n-propylamine	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Isophorone	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Benzoic acid	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	41.6	10	20	20 U	20 U
1,2,4-Trichlorobenzene	<10 UJ	<10 UJ	<10 UJ	<10	<10	na	na	0 / 3	110	10	10	10 UJ	10 UJ
Hexachlorobutadiene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Hexachlorocyclopentadiene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U

Parameter (ug/L)	SC-PG-SW-01	SC-PG-SW-02	SC-PG-SW-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-FB-01	SC-PG-FD-XX
2-Chloronaphthalene	<10 UJ	<10 UJ	<10 UJ	<10	<10	na	na	0 / 3	none	10	10	10 UJ	10 UJ
Acenaphthene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	12	10	10	10 U	10 U
Acenaphthylene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	12	10	10	10 U	10 U
Dibenzofuran	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	20.4	10	10	10 U	10 U
4-Chlorophenyl-phenylether	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
4,6-Dinitro-2-methylphenol	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	none	25	20	20 U	20 U
Hexachlorobenzene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	0.00024	10	10	10 U	10 U
Anthracene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	0.029	10	10	10 U	10 U
Pyrene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Benzo[a]anthracene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	0.027	10	10	10 U	10 U
Di-n-octylphthalate	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	30	10	10	10 U	10 U
Benzo[a]pyrene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	0.014	10	10	10 U	10 U
Benzo[g,h,i]perylene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
2-Chlorophenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Benzyl alcohol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	58	10	10	10 U	10 U
bis(2-chloroisopropyl)ether	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Hexachloroethane	<10 UJ	<10 UJ	<10 UJ	<10	<10	na	na	0 / 3	12	10	10	10 UJ	10 UJ
2-Nitrophenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	163	10	10	10 U	10 U
bis(2-Chloroethoxy)methane	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Naphthalene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	81	10	10	10 U	10 U
4-Chloro-3-methylphenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
2,4,6-Trichlorophenol	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	2	10	10	10 U	10 U
2-Nitroaniline	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	25	10	10 U	10 U
2,6-Dinitrotoluene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	50	10	10	10 U	10 U
2,4-Dinitrophenol	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	none	25	20	20 U	20 U
2,4-Dinitrotoluene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	18.5	10	10	10 U	10 U
Fluorene	<10 UJ	<10 UJ	<10 UJ	<10	<10	na	na	0 / 3	3.9	10	10	10 UJ	10 UJ
n-Nitrosodiphenylamine	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	24.5	10	10	10 U	10 U
Pentachlorophenol	<20 U	<20 U	<20 U	<20	<20	na	na	0 / 3	3.5	25	20	20 U	20 U
Di-n-butylphthalate	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	0.1	10	10	10 U	10 U
Butylbenzylphthalate	<10 U	<10 U	<10 U	<10	<10	na	na	1 / 3	0.075	10	10	10 U	7.9 J
Chrysene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Benzo[b]fluoranthene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
Indeno[1,2,3-cd]pyrene	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U

Parameter (ug/L)	SC-PG-SW-01	SC-PG-SW-02	SC-PG-SW-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-FB-01	SC-PG-FD-XX
N-Nitrosodimethylamine	<10 U	<10 U	<10 U	<10	<10	na	na	0 / 3	none	10	10	10 U	10 U
													Sample Location PG-03

Table F6. Pond G Sediment TOC and Metals Data

Parameter	SC-PG-SD-01	SC-PG-SD-02	SC-PG-SD-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-SD-FB	SC-PG-SD-FD
GFAA Weight (g)	1.05	1.04	1.01	1.01	1.05	1.03	0.02	3 / 3	na	na	na	10.001	1.01
ICP Weight (g)	1.07	1.00	1.00	1.00	1.07	1.02	0.04	3 / 3	na	na	na	10.028	1.01
Percent Solids	13.9	44.2	26.1	13.9	44.2	28.1	15.2	3 / 3	na	na	na	na	21.3
Metals (mg/kg)													
Aluminum	13000	11000	18000	11000	18000	14000	3606	3 / 3	none	40	40	4.0 UJ	19000
Antimony	0.63 J	0.48 UJ	0.50 UJ	<0.48	0.63	0.54	0.08	1 / 3	none	40	0.5	0.050 U	0.51 J
Arsenic	3.5 J	4.3 J	7.6 J	3.5	7.6	5.1	2.2	3 / 3	6	1.5	1	0.10 U	7.3 J
Barium	140	130	190	130	190	153	32.1	3 / 3	none	4	4	0.40 U	190
Beryllium	0.93 U	1.0 U	1.0 U	<0.93	<1.0	na	na	0 / 3	none	4	1	0.10 U	0.99 U
Cadmium	0.74	0.73	0.81	0.73	0.81	0.76	0.04	3 / 3	0.6	0.5	0.5	0.050 U	0.74
Chromium	19	17	36	17	36	24	10	3 / 3	26	1	1	0.10 U	35
Cobalt	9.3 U	10 UJ	10 U	<9.3	<10	na	na	0 / 3	none	10	10	1.0 U	9.9 U
Copper	19 J	18	39 J	18	39	25	12	3 / 3	16	1	1	0.28 J	38 J
Lead	28	24	39	24	39	30	7.8	3 / 3	31	1	1	0.10 U	38
Magnesium	2800	2200	3600	2200	3600	2867	702	3 / 3	none	40	40	4.0 U	3800
Manganese	340	270	390	270	390	333	60	3 / 3	460	2	2	0.20 U	390
Mercury	0.14 J	0.044 J	0.13 J	0.044	0.14	0.10	0.05	3 / 3	0.2	0.1	0.1	0.0097 U	0.13 J
Nickel	14	12	22	12	22	16	5	3 / 3	16	10	10	1.0 U	22
Selenium	0.95 U	0.96 U	0.99 U	<0.95	<0.99	na	na	0 / 3	none	40	1	0.10 U	0.99 U
Silver	0.48 U	0.48 U	0.50 U	<0.48	<0.50	na	na	0 / 3	1	0.5	0.5	0.050 U	0.50 U
Thallium	0.95 U	0.96 U	0.99 U	<0.95	<0.99	na	na	0 / 3	none	40	1	0.10 U	0.99 U
Zinc	85	62	110	62	110	86	24	3 / 3	120	5	5	1.4	100
													Sample Location PG-03

Table F7. Pond G Sediment Pesticides and PCBs Data

Parameter (mg/kg)	SC-PG-SD-01		SC-PG-SD-02		SC-PG-SD-03		Min	Max	DF	RBAL	Target DL	Achieved DL	SC-PG-SD-FB		SC-PG-SD-FD	
% SOLIDS	40		52		30		30	52	na	na	na	na	na		28	
4,4'-DDT	0.04	U	**0.03**	**J**	0.03	U	<0.03	**0.03**	1 / 3	0.008	0.01	0.06	0.4	UJ	0.06	U
4,4'-DDD	0.068	U	**0.16**		0.039	U	<0.039	**0.16**	2 / 3	0.005	0.01	0.06	0.4	UJ	**0.097**	
4,4'-DDE	0.04	U	**0.053**		0.03	U	<0.03	**0.053**	1 / 3	0.008	0.01	0.06	0.4	UJ	0.06	U
Aldrin	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.002	0.01	0.06	0.4	UJ	0.06	U
Alpha-BHC	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.006	0.01	0.06	0.4	UJ	0.06	U
Beta-BHC	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.005	0.01	0.06	0.4	UJ	0.06	U
Chlordane, Technical	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.007	0.01	0.04	1	UJ	0.03	U
Delta-BHC	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.003	0.01	0.06	0.4	UJ	0.06	U
Dieldrin	0.04	U	**0.071**		0.03	U	<0.03	**0.071**	1 / 3	0.002	0.01	0.06	0.4	UJ	0.06	U
Endosulfan I	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.051	0.01	0.06	0.4	UJ	0.06	U
Endosulfan II	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.051	0.01	0.06	0.4	UJ	0.06	U
Endosulfan Sulfate	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	none	0.01	0.06	0.4	UJ	0.06	U
Endrin	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.003	0.01	0.06	0.4	UJ	0.06	U
Endrin Aldehyde	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	none	0.01	0.06	0.4	UJ	0.06	U
Heptachlor	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.0003	0.01	0.06	0.4	UJ	0.06	U
Heptachlor Epoxide	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.005	0.01	0.06	0.4	UJ	0.06	U
Lindane	0.04	U	0.03	U	0.03	U	<0.03	<0.04	0 / 3	0.003	0.01	0.06	0.4	UJ	0.06	U
Methoxychlor	0.04	U	0.03	UJ	0.03	U	<0.03	<0.04	0 / 3	0.019	0.01	0.06	0.4	UJ	0.06	UJ
Toxaphene	0.17	U	0.17	U	0.17	U	<0.17	<0.17	0 / 3	0.028	0.13	0.17	1.0	UJ	0.17	U
Aroclor-1016	0.33	U	0.33	U	0.33	U	<0.33	<0.33	0 / 3	0.007	1	0.33	0.7	UJ	0.33	U
Aroclor-1221	0.33	U	0.33	U	0.33	U	<0.33	<0.33	0 / 3	none	1	0.33	0.7	UJ	0.33	U
Aroclor-1232	0.33	U	0.33	U	0.33	U	<0.33	<0.33	0 / 3	none	1	0.33	0.7	UJ	0.33	U
Aroclor-1242	0.33	U	0.33	U	0.33	U	<0.33	<0.33	0 / 3	none	1	0.33	0.7	UJ	0.33	U
Aroclor-1248	0.33	U	0.33	U	0.33	U	<0.33	**0.39**	1 / 3	0.03	1	0.33	0.7	UJ	**0.39**	
Aroclor-1254	0.33	U	0.33	U	0.33	U	<0.33	<0.33	0 / 3	0.06	1	0.33	0.7	UJ	0.33	U
Aroclor-1260	0.33	U	0.33	U	0.33	U	<0.33	<0.33	0 / 3	0.005	1	0.33	0.7	UJ	0.33	U
															Sample Location PG-03	

Table F8. Pond G Sediment Semivolatile Organic Compounds (SVOCs) Data

Parameter (ug/kg)	SC-PG-SD-01	SC-PG-SD-02	SC-PG-SD-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-SD-FB	SC-PG-SD-FD
Phenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
1,3-Dichlorobenzene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	1.7	330	900	82 U	900 U
1,2-Dichlorobenzene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.34	330	900	82 UJ	900 U
4-Methylphenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	**550 J**
Nitrobenzene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
2,4-Dimethylphenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
2,4-Dichlorophenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
4-Chloroaniline	5800 UJ	6200 UJ	9000 UJ	<5800	<9000	na	na	0 / 3	none	330	9000	160 U	9000 UJ
2-Methylnaphthalene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.07	330	900	82 U	900 U
2,4,5-Trichlorophenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Dimethylphthalate	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
3-Nitroaniline	1200 U	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	800	1800	160 U	1800 U
4-Nitrophenol	1200 U	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	330	1800	160 U	1800 U
Diethylphthalate	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.63	330	900	82 U	900 U
4-Nitroaniline	1200 U	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	330	1800	160 U	1800 U
4-Bromophenyl-phenylether	580 U	620 U	900 U	<580	<900	na	na	0 / 3	1.3	330	900	82 U	900 U
Phenanthrene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.56	330	900	82 U	900 U
Fluoranthene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.19	330	900	82 U	900 U
3,3'-Dichlorobenzidine	1200 UJ	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	330	1800	160 U	1800 UJ
bis(2-Ethylhexyl)phthalate	580 U	620 U	900 U	<580	<900	na	na	0 / 3	25	330	900	82 U	900 U
Benzo[k]fluoranthene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.24	330	900	82 U	900 U
Dibenz[a,h]anthracene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.06	330	900	82 U	900 U
bis(2-Chloroethyl)ether	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
1,4-Dichlorobenzene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.35	330	900	82 U	900 U
2-Methylphenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
N-Nitroso-di-n-propylamine	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Isophorone	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Benzoic acid	1200 U	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	330	1800	160 U	1800 U
1,2,4-Trichlorobenzene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	9.2	330	900	82 UJ	900 U
Hexachlorobutadiene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U

Parameter (ug/kg)	SC-PG-SD-01	SC-PG-SD-02	SC-PG-SD-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-SD-FB	SC-PG-SD-FD
Hexachlorocyclopentadiene	580 UJ	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 UJ
2-Chloronaphthalene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 UJ	900 U
Acenaphthene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.62	330	900	82 U	900 U
Acenaphthylene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.62	330	900	82 U	900 U
Dibenzofuran	580 U	620 U	900 U	<580	<900	na	na	0 / 3	2	330	900	82 U	900 U
4-Chlorophenyl-phenylether	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
4,6-Dinitro-2-methylphenol	1200 U	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	330	1800	160 U	1800 U
Hexachlorobenzene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Anthracene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.22	330	900	82 U	900 U
Pyrene	580 UJ	620 UJ	900 UJ	<580	<900	na	na	0 / 3	0.49	330	900	82 U	900 UJ
Benzo[a]anthracene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.32	330	900	82 U	900 U
Di-n-octylphthalate	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Benzo[a]pyrene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.37	330	900	82 U	900 U
Benzo[g,h,i]perylene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.17	330	900	82 U	900 U
2-Chlorophenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Benzyl alcohol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
bis(2-chloroisopropyl)ether	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Hexachloroethane	580 U	620 U	900 U	<580	<900	na	na	0 / 3	1	330	900	82 UJ	900 U
2-Nitrophenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
bis(2-Chloroethoxy)methane	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Naphthalene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.48	330	900	82 U	900 U
4-Chloro-3-methylphenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
2,4,6-Trichlorophenol	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
2-Nitroaniline	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	800	900	82 U	900 U
2,6-Dinitrotoluene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.021	330	900	82 U	900 U
2,4-Dinitrophenol	1200 U	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	800	1800	160 U	1800 U
2,4-Dinitrotoluene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.0429	330	900	82 U	900 U
Fluorene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.19	330	900	82 UJ	900 U
n-Nitrosodiphenylamine	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
Pentachlorophenol	1200 U	1200 U	1800 U	<1200	<1800	na	na	0 / 3	none	800	1800	160 U	1800 U
Di-n-butylphthalate	580 U	620 U	900 U	<580	<900	na	na	0 / 3	11	330	900	82 U	900 U
Butylbenzylphthalate	580 UJ	620 UJ	900 UJ	<580	<900	na	na	0 / 3	11	330	900	82 U	900 UJ
Chrysene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.34	330	900	82 U	900 U
Benzo[b]fluoranthene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U

Parameter (ug/kg)	SC-PG-SD-01	SC-PG-SD-02	SC-PG-SD-03	Min	Max	Mean	SD	DF	RBAL	Target DL	Achieved DL	SC-PG-SD-FB	SC-PG-SD-FD
Indeno[1,2,3-cd]pyrene	580 U	620 U	900 U	<580	<900	na	na	0 / 3	0.2	330	900	82 U	900 U
N-Nitrosodimethylamine	580 U	620 U	900 U	<580	<900	na	na	0 / 3	none	330	900	82 U	900 U
													Sample Location PG-03

APPENDIX G

Rice Creek Sampling Locations and Analytical Measurement Results

Table of Contents

Rice Creek Data Presentation Summary

The map of Rice Creek (Fig.G1), shows the location where each water sample was taken. Global Positioning System (GPS) coordinates were used to plot the locations. The map image is from an orthophoto taken in 1997, obtained from Wenck Associates, Inc., Maple Plain, Minnesota. These coordinates recorded in the field, during sampling, are reported in tables G1 through G5. For mapping purposes, second quarter sampling locations were not mapped by their GPS coordinates. (However, the following GPS data tables do show the recorded coordinates). Second quarter samples were mapped with first quarter coordinates, with the exception of sampling location 03. During second quarter sampling there was a problem with the crypto key, causing samples to show imprecise coordinates. Based on landmarks, the field team leader noted in the field logbook (USACHPPM 2000) that sample locations for the second quarter were as close as possible to first quarter coordinates. Maps from the Work Plan (USACHPPM 1999) were also used to pinpoint the location to be sampled.

The tables G6 through G9 summarize the data for Rice Creek surface water samples taken during the quarterly water monitoring. The summaries include the field and analytical result for each analyte at each sample location. Summary data used for statistical analysis such as the central tendencies, standard deviations and confidence intervals are report in tables found in Appendix A.

The number of digits reported for each value in the following tables does not necessarily reflect the precision and accuracy of the measurement. The data tables present significant figures as they were originally reported by the analyzing laboratory. Significant figures reported for the analytes were based on the analytical detection limit values. However, some sample values have more significant figures than others to demonstrate that blank correction was discontinued, as was requested by USEPA comments on the draft version of this report (see Appendix R). Data quality is discussed in Appendix K.

The results are reported in the following manner to facilitate data review.

- If the detection limit is sample-specific (as it was for the pesticides in sediment), the value reported for the achieved detection limit for a site is the highest detection limit for that parameter for all samples (including the field duplicate) from that site.

- If a parameter was not detected in any samples from a site (e.g. Pond G, water, alpha-BHC) then the minimum and maximum values represent the lowest and highest achieved detection limits for those data.

- Any concentrations that were below the detection limit but still quantified are identified in Volume 2 of this report as estimated values (J-flagged). These values are reported and used uncensored in the following tables.

- If a parameter was undetected in at least one but not all samples from a site, then the minimum reflects the lowest detection limit of the undetected samples. If there was a

J-flagged value lower than the highest undetected value, the J-flagged value was reported as the maximum because the value was actually detected but estimated.

- Field duplicates (FD) were used as quality control samples and were not included in the reported minima and maxima columns. However, if an analyte was detected in the FD and not in it's corresponding sample, then the FD detection was counted in the detection frequency column (only in the numerator).

- When laboratory replicate samples (R1, R2) were available, the values from the primary sample (R1) were reported. See Appendix K section K1.2.1 for explanation.

Definition of Data Qualifiers (Flags)

During the data validation process, all laboratory data were assigned appropriate data validation flags and reason codes. Validation flags used in the following tables are defined as follows:

No flag	When the analyte value reported is detected and usable. The integrity of the analyte's identification, accuracy, and precision has been validated
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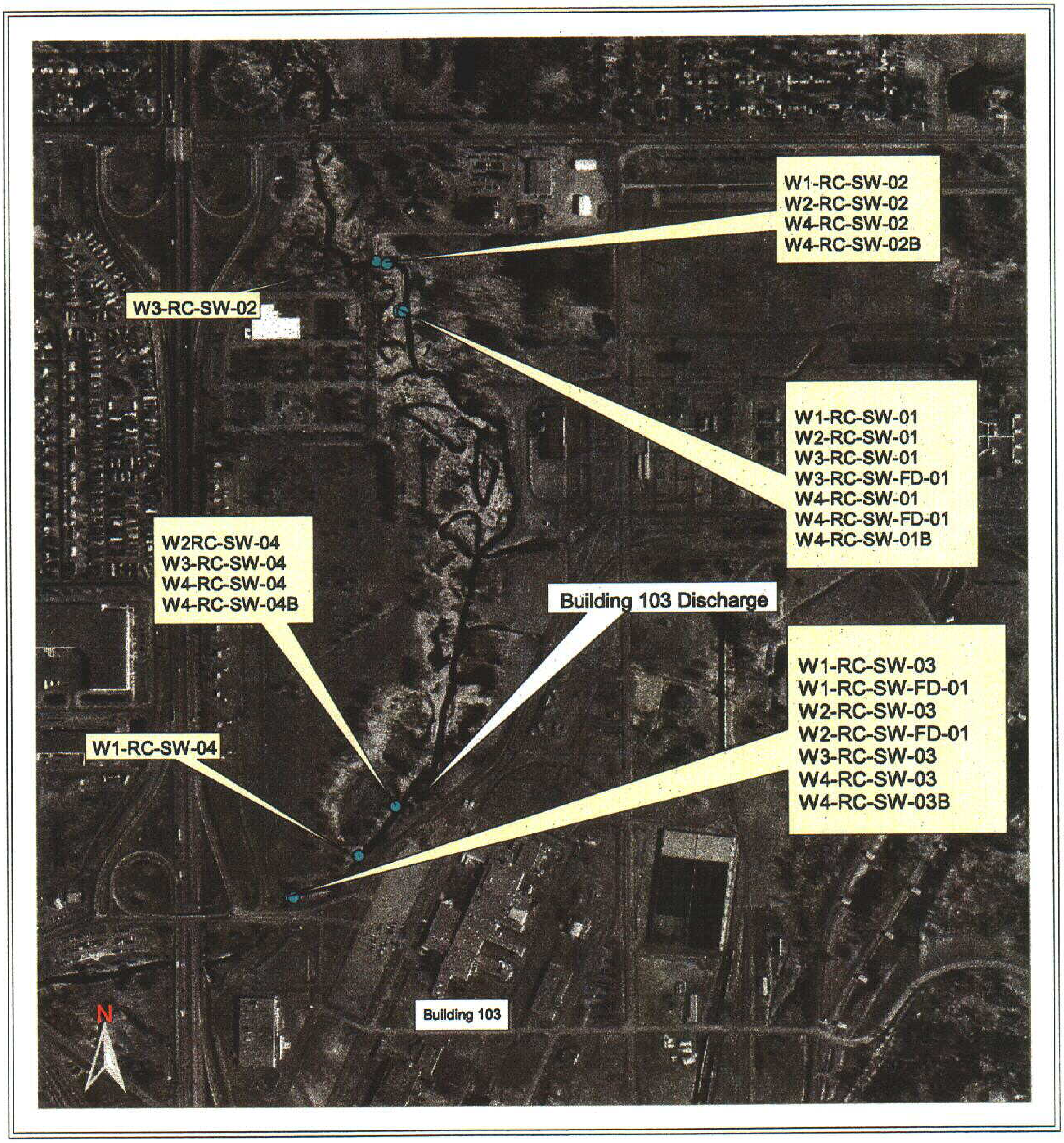
Other information regarding data validation can be found in Volume 2 (CD ROM version) of the report.

Note: References cited in the text of this appendix can be found in the reference section of the main Tier II Ecological Risk Assessment Report.

Figure G1. Rice Creek Water Monitoring Sample Locations



Wx= sampling quarter
RC= Rice Creek
SW=Surface Water
FD= Field duplicate
0#B = second set of samples taken during qtr 4

100 0 100 200 Meters

Table G1. Rice Creek First Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W1-RC-SW-01	26-Sep-99	4"	45	6	19.02	93	10	59.16
W1-RC-SW-02	26-Sep-99	4"	45	6	21.76	93	11	0.19
W1-RC-SW-03 W1-RC-SW-FD-01	26-Sep-99	4"	45	5	44.86	93	11	8.22
W1-RC-SW-04	26-Sep-99	4"	45	5	47.07	93	11	2.84

Table G2. Rice Creek Second Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W2-RC-SW-01	21-Jan-00	6"	45	6	19.8	93	10	58.27
W2-RC-SW-02	21-Jan-00	5"	45	6	22.46	93	11	1.18
W2-RC-SW-03 W2-RC-SW-FD-01	21-Jan-00	7"	45	5	45.31	93	11	10.18
W2-RC-SW-04	21-Jan-00	10"	45	5	50.77	93	10	59.06

Table G3. Rice Creek Third Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W3-RC-SW-01 W3-RC-SW-FD-01	27-Apr-00	6-10"	45	6	18.91	93	10	59.38
W3-RC-SW-02	27-Apr-00	6-10"	45	6	21.82	93	11	1.27
W3-RC-SW-03	27-Apr-00	6-10"	45	5	44.7	93	11	8.48
W3-RC-SW-04	27-Apr-00	6-10"	45	5	50.12	93	10	59.9

Table G4. Rice Creek Fourth Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W4-RC-SW-01 W4-RC-SW-FD-01	21-Jun-00	4-6”	45	6	18.87	93	10	59.08
W4-RC-SW-02	21-Jun-00	4-6”	45	6	21.72	93	11	0.29
W4-RC-SW-03	21-Jun-00	6-8”	45	5	44.79	93	11	8.22
W4-RC-SW-04	21-Jun-00	4-6”	45	5	50.01	93	10	59.79

Table G5. Rice Creek Additional Fourth Quarter Surface Water Sample GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W4-RC-SW-01B	23-Jun-00	4-6"	45	6	18.87	93	10	59.08
W4-RC-SW-02B	23-Jun-00	4-6"	45	6	21.68	93	11	0.4
W4-RC-SW-03B	23-Jun-00	6-8"	45	5	44.66	93	11	8.18
W4-RC-SW-04B	23-Jun-00	4-6"	45	5	49.94	93	10	59.81

* The GPS unit used in the study was a military-issue Precision Light Weight GPS Receiver (PLGR). The Department of Defense (DoD) intentionally places errors in the GPS navigation and timing signal, in a process called selective availability (Department of the Army 1993). Errors resulting from this process are unpredictable and can produce significant horizontal and elevation errors. Military GPS receivers must have crypto keys loaded to detect and nullify selective availability errors, in order to allow for more accurate position data (Department of the Army 1993). The PLGR was equipped with a crypto key, and was therefore not subject to the selective availability and spoofing errors that affect non-military GPS. (The only exception was during the second quarter). The PLGR operations manual (Department of the Army 1995) states that with crypto keys installed, the PLGR provides horizontal 2-D position accuracies better than the worst-case error of 10 meters.

Table G6. Rice Creek First Quarter Water Quality and Metals Data

Parameter	W1-RC-SW-01	W1-RC-SW-02	W1-RC-SW-03	W1-RC-SW-04	Detection limit
pH	8.49	8.54	8.59	8.62	na
DO (mg/L)	8.62	8.74	9.43	9.5	na
DO (%)	91.6	93.6	101.5	101.9	na
Conductivity (µmhos/L)	376	371	378	378	na
Temperature (°C)	18.33	18.52	18.90	18.70	na
Hardness (mg/L)	156	156	150	152	2.00
TKN (mg/L)	3.9	3.6	3.3	3.8	0.50
TOC (mg/L)	21	21	21	21	1.00
Total Phosphorus (mg/L)	0.24	0.27	0.24	0.26	0.01
TSS (mg/L)	78	76	66	64	1.00
Metals (µg/L)					
Aluminum	73.25	74.25	73.25	78.25	0.056
Barium	68 J	68 J	69 J	72 J	0.0056
Cadmium	0.021	0.025	0.022	0.025	0.015
Copper	1.0	1.0	1.5	1.1	0.014
Mercury	0.042	0.0035	0.0038	0.0033	0.00018
Silver	0.0282 UB	0.0212 UB	0.0172 UB	0.0172 UB	0.0043

QC Samples

Parameter	W1-RC-FB-01	W1-RC-SW-FD-XX
pH	na	na
DO (mg/L)	na	na
DO (%)	na	na
Conductivity (µmhos/L)	na	na
Temperature (°C)	na	na
Hardness (mg/L)	na	160
TKN (mg/L)	na	3.5
TOC (mg/L)	na	21
Total Phosphorus (mg/L)	na	0.23
TSS (mg/L)	na	66
Metals (µg/L)		
Aluminum	1.30 UB	89
Barium	0.22 UB	70.16 J
Cadmium	0.015 U	0.015 U
Copper	0.19	1.2
Mercury	0.00022	0.0035
Silver	0.0114 UB	0.013 UB
		W1-RC-SW-03

Table G7. Rice Creek Second Quarter Surface Water Quality and Metals Data

Parameter	W2-RC-SW-01	W2-RC-SW-02	W2-RC-SW-03	W2-RC-SW-04	Detection limit
pH	7.23	7.25	7.23	7.25	na
DO (mg/L)	7.23	8.91	12.07	11.21	na
DO (%)	49.2	58.8	82.8	74.9	na
Conductivity (μmhos/L)	667	661	673	650	na
Temperature (°C)	0.07	0.086	0.05	0.05	na
Hardness (mg/L)	268	272	264	272	2.00
TKN (mg/L)	1.8	2.1	2.1	1.9	0.50
TOC (mg/L)	20	20	19	19	1.00
Total Phosphorus (mg/L)	0.06	0.06	0.06	0.06	0.01
TSS (mg/L)	2	4	5	5	1.00
Metals (μg/L)					
Aluminum	5.22	4.92	4.52	4.32	0.056
Barium	83.011	81.011	80.011	86.011	0.0056
Cadmium	<0.015 U	<0.015 U	<0.015 U	<0.015 U	0.015
Copper	0.86	0.90	0.84	0.80	0.014
Mercury	0.0009	0.00057 UB	0.00059 UB	0.00078	0.000039
Silver	0.0143 UBJ	0.0143 UBJ	0.0143 UBJ	0.0107 UBJ	0.0043

QC Samples Parameter	W2-RC-FB-01	W2-RC-FD-00
pH	na	na
DO (mg/L)	na	na
DO (%)	na	na
Conductivity (μmhos/L)	na	na
Temperature (°C)	na	na
Hardness (mg/L)	na	290
TKN (mg/L)	na	1.8
TOC (mg/L)	na	18
Total Phosphorus (mg/L)	na	0.06
TSS (mg/L)	na	4
Metals (μg/L)		
Aluminum	1.62 UB	4.82
Barium	0.0183 UBJ	82.011
Cadmium	0.015 U	0.015 U
Copper	0.024	0.88
Mercury	0.00047 UB	0.00055 UB
Silver	0.0189 UBJ	0.0143 UBJ
		W2-RC-SW-03

Table G8. Rice Creek Third Quarter Water Quality and Metals Data

Parameter	W3-RC-SW-01	W3-RC-SW-02	W3-RC-SW-03	W3-RC-SW-04	Detection limit
pH	8.7	8.75	8.73	8.76	na
DO (%)	117	120.2	122.1	122.2	na
DO (mg/L)	10.7	10.98	11.07	11.08	na
Conductivity (umhos/L)	397	395	401	399	na
Temperature (C)	19.54	19.7	20.09	20.11	na
Hardness (mg/L)	168	166	170	168	2.00
TKN (mg/L)	3.2	2.8	3.1	3.4	0.50
TOC (mg/L)	16	16	16	16	0.20
Total Phosphorus (mg/L)	0.2	0.21	0.18	0.24	0.01
TSS (mg/L)	83	82	94	88	1.00
Metals (ug/L)					
Aluminum	180	150	240	240	0.0099
Barium	92 J	91 J	96 J	94 J	0.00088
Cadmium	0.03495	0.03495	0.04095	0.04695	0.00066
Copper	1.3	1.2	1.6	1.7	0.0043
Mercury	0.00468	0.00488	0.00508	0.00528	0.000063
Silver	0.01858 UBJ	0.01858 UBJ	0.01858 UBJ	0.01858 UBJ	0.00058

QC samples

Parameter	W3-RC-FB-01	W3-RC-FD-01
pH	na	8.7
DO (%)	na	116.9
DO (mg/L)	na	10.69
Conductivity (umhos/L)	na	397
Temperature (C)	na	19.55
Hardness (mg/L)	na	188
TKN (mg/L)	na	3
TOC (mg/L)	na	17
Total Phosphorus (mg/L)	na	0.22
TSS (mg/L)	na	93
Metals (ug/L)		
Aluminum	2.2	180
Barium	0.014 J	93 J
Cadmium	0.00161 UB	0.03695
Copper	0.010	1.3
Mercury	0.000318 UB	0.00438
Silver	0.01858 UBJ	0.01858 UBJ

Table G9. Rice Creek Fourth Quarter Surface Water Quality and Metals Data

Parameter	W4-RC-SW-01	W4-RC-SW-01B	W4-RC-SW-02	W4-RC-SW-02B	W4-RC-SW-03	W4-RC-SW-03B	W4-RC-SW-04	W4-RC-SW-04B	Detection limit
pH	7.99	7.72	8.03	7.78	8.03	7.74	8.06	7.8	na
DO (%)	86.8	69.7	88.6	69.8	91.2	74.6	91.8	76	na
DO (mg/L)	7.83	6.22	8.03	6.23	8.22	6.65	8.3	6.77	na
Conductivity (umhos/L)	385	379	383	378	392	386	390	383	na
Temperature (C)	20.3	20.89	20.1	20.88	20.33	20.87	20.17	20.99	na
Hardness (mg/L)	148	150	168	150	154	148	150	150	2.00
TKN (mg/L)	2.2	2.4	1.8	2.1	2.1	2.2	2.1	2.5	0.50
TOC (mg/L)	17	17	17	17	17	16	17	17	0.20
Total Phosphorus (mg/L)	0.17	0.18	0.17	0.18	0.17	0.19	0.18	0.18	0.01
TSS (mg/L)	60	48	58	56	52	49	47	50	1.00
Metals (ug/L)									
Aluminum	180.49	180.49	130.49	200.49	150.49	190.49	120.49	170.49	0.0099
Barium	74	73	71	73	73	74	70	74	0.00088
Cadmium	0.029	0.035	0.029	0.033	0.026	0.029	0.024	0.030	0.00066
Copper	1.3	1.6	1.3	1.5	1.3	1.6	1.2	1.6	0.0043
Mercury	0.0041	0.0046	0.0038	0.0045	0.0038	0.0050	0.0036	0.0040	0.000039
Silver	0.00348 UBJ	0.00328 UBJ	0.0034 UBJ	0.0055 UBJ	0.00363 UBJ	0.0057 UBJ	0.00331 UBJ	0.0047 UBJ	0.004

QC Samples

Parameter	W4-RC-FB-01	W4-RC-FD-01
pH	na	8
DO (%)	na	86.7
DO (mg/L)	na	7.83
Conductivity (umhos/L)	na	386
Temperature (C)	na	20.3
Hardness (mg/L)	na	174
TKN (mg/L)	na	2.7
TOC (mg/L)	na	17
Total Phosphorus (mg/L)	na	0.19
TSS (mg/L)	na	64
Metals (ug/L)		
Aluminum	0.98 UB	170.49
Barium	<0.00088 U	73
Cadmium	0.0017	0.031
Copper	<0.0043 U	1.3
Mercury	0.00021	0.0049
Silver	0.0035 UBJ	0.0041 UBJ

APPENDIX H

Sunfish Lake Sampling Locations and Analytical Measurement Results

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Sunfish Lake Data Presentation Summary

The map of Sunfish Lake (Fig.H1), shows the location where each media sample was taken. The map image is from an orthophoto taken in 1997, obtained from Wenck Associates, Inc., Maple Plain, Minnesota. Global Positioning System (GPS) coordinates were used to plot the locations. These coordinates recorded in the field, during sampling, are reported in tables H1 and H5. For mapping purposes, second quarter sampling locations were not mapped by their GPS coordinates. (However, the following GPS data tables do show the recorded coordinates). Second quarter samples were mapped with first quarter coordinates. During second quarter sampling there was a problem with the crypto key, causing all samples to show imprecise coordinates. Based on landmarks, the field team leader noted in the field log book (USACHPPM 2000) that sample locations for the second quarter were as close as possible to first quarter coordinates. Maps from the Work Plan (USACHPPM 1999) were also used to pinpoint the location to be sampled.

The tables H6 through H9 summarize the data for Sunfish Lake surface water and sediment samples taken during the water monitoring activities. The summaries include the field and analytical result for each analyte at each sample location. Summary data used for statistical analysis such as the central tendencies, standard deviations and confidence intervals are report in tables found in Appendix A.

The number of digits reported for each value in the following tables does not necessarily reflect the precision and accuracy of the measurement. The data tables present significant figures as they were originally reported by the analyzing laboratory. Significant figures reported for the analytes were based on the analytical detection limit values. However, some sample values have more significant figures than others to demonstrate that blank correction was discontinued, as was requested by USEPA comments on the draft version of this report (see Appendix R). Data quality is discussed in Appendix K.

The results are reported in the following manner to facilitate data review.

- If the detection limit is sample-specific (as it was for the pesticides in sediment), the value reported for the achieved detection limit for a site is the highest detection limit for that parameter for all samples (including the field duplicate) from that site.

- If a parameter was not detected in any samples from a site (e.g. Pond G, water, alpha-BHC) then the minimum and maximum values represent the lowest and highest achieved detection limits for those data.

- Any concentrations that were below the detection limit but still quantified are identified in Volume 2 of this report as estimated values (J-flagged). These values are reported and used uncensored in the following tables.

- If a parameter was undetected in at least one but not all samples from a site, then the minimum reflects the lowest detection limit of the undetected samples. If there was a

J-flagged value lower than the highest undetected value, the J-flagged value was reported as the maximum because the value was actually detected but estimated.

- Field duplicates (FD) were used as quality control samples and were not included in the reported minima and maxima columns. However, if an analyte was detected in the FD and not in it's corresponding sample, then the FD detection was counted in the detection frequency column (only in the numerator).

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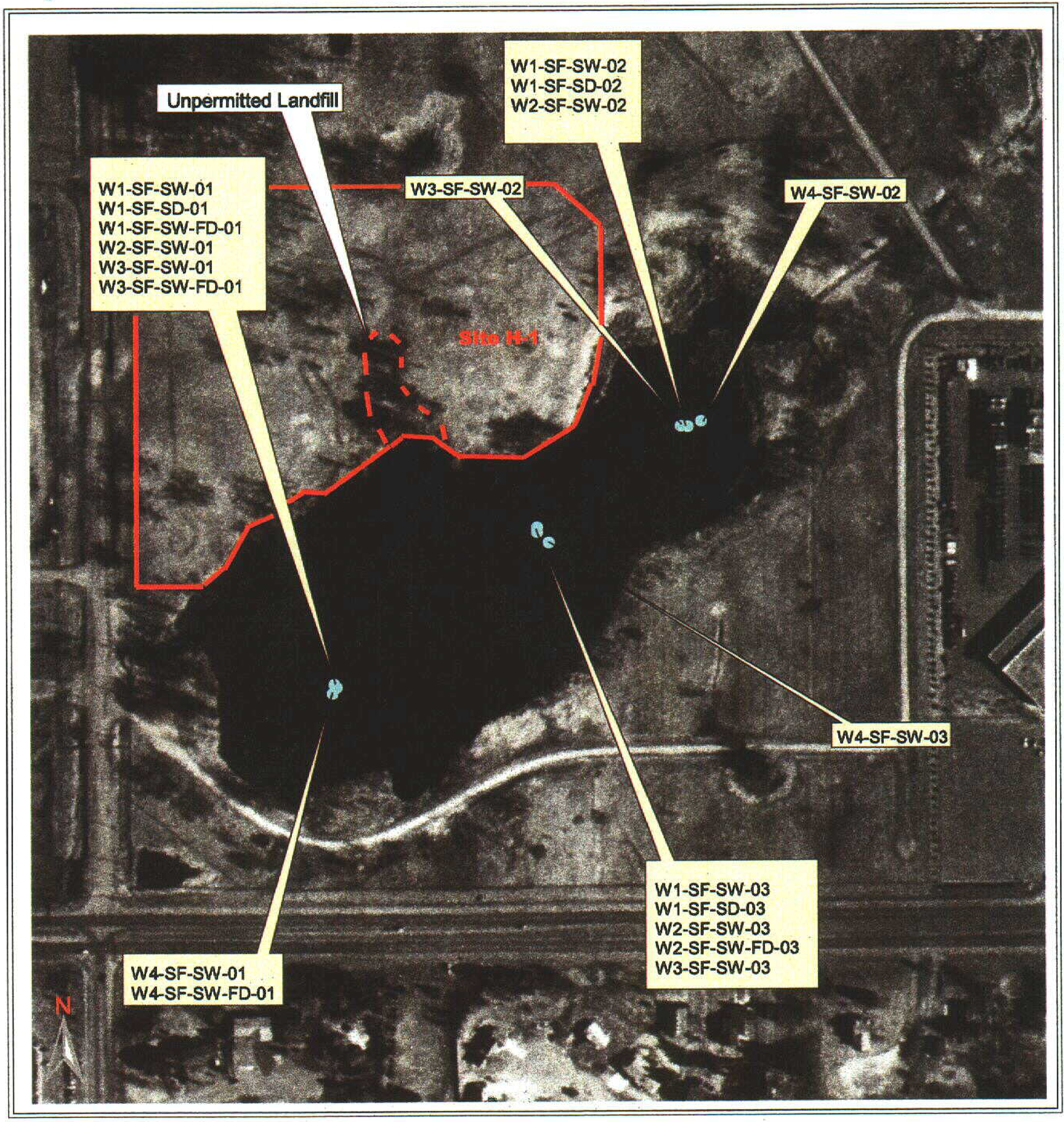
"J"	When the associated value is an estimated quantity. Indicating there is cause to question accuracy or precision of the reported value. The value is still used in the risk assessment.
"UJ"	When the analyte was analyzed for, but not detected, above the associated value, however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"UBJ"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with; however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.

Other information regarding data validation can be found in Volume 2 (CD ROM version) of the report.

Note: References cited in the text of this appendix can be found in the reference section of the main Tier II Ecological Risk Assessment Report.

Figure H1. Sunfish Lake Surface Water and Sediment Sample Locations



Wx=quarter
SF= Sunfish Lake
SW=Surface Water
SD=Sediment
FD=Field Duplicate

0.03 0 0.03 0.06 Miles

Table H1. Sunfish Lake First Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W1-SF-SW-01 W1-SF-SW-FD-01	29-Sep-99	4-8"	45	4	50.09	93	9	20.39
W1-SF-SW-02	29-Sep-99	4-8"	45	4	54.86	93	9	11.21
W1-SF-SW-03	29-Sep-99	4-8"	45	4	52.9	93	9	15.14

Table H2. Sunfish Lake Second Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W2-SF-SW-01	20-Jan-00	10"	45	4	50.46	93	9	23.59
W2-SF-SW-02	20-Jan-00	1.5'	45	4	54.66	93	9	10.43
W2-SF-SW-03 W2-SF-SW-FD-01	20-Jan-00	9"	45	4	52.36	93	9	11.97

Table H3. Sunfish Lake Third Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W3-SF-SW-01 W3-SF-SW-FD-01	26-Apr-00	6-10"	45	4	50.03	93	9	20.33
W3-SF-SW-02	26-Apr-00	6-10"	45	4	54.87	93	9	11.4
W3-SF-SW-03	26-Apr-00	6-10"	45	4	52.98	93	9	15.13

Table H4. Sunfish Lake Fourth Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W4-SF-SW-01 W4-SF-SW-FD-01	20-Jun-00	2-6"	45	4	49.91	93	9	20.44
W4-SF-SW-02	20-Jun-00	2-6"	45	4	54.96	93	9	10.87
W4-SF-SW-03	20-Jun-00	2-6"	45	4	52.7	93	9	14.83

Table H5. Sunfish Lake Sediment Bioassay Sampling GPS Coordinates (1st Quarter)

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
WI-SF-SD-01	29-Sep-99	6'	45	4	50.09	93	9	20.39
WI-SF-SD-02	29-Sep-99	5'	45	4	54.86	93	9	11.21
WI-SF-SD-03	29-Sep-99	5' 9"	45	4	52.9	93	9	15.14

*The GPS unit used in the study was a military-issue Precision Light Weight GPS Receiver (PLGR). The Department of Defense (DoD) intentionally places errors in the GPS navigation and timing signal, in a process called selective availability (Department of the Army 1993). Errors resulting from this process are unpredictable and can produce significant horizontal and elevation errors. Military GPS receivers must have crypto keys loaded to detect and nullify selective availability errors, in order to allow for more accurate position data (Department of the Army 1993). The PLGR was equipped with a crypto key, and was therefore not subject to the selective availability and spoofing errors that affect non-military GPS. (The only exception was during the second quarter). The PLGR operations manual (Department of the Army 1995) states that with crypto keys installed, the PLGR provides horizontal 2-D position accuracies better than the worst case error of 10 meters.

Table H6 Sunfish Lake First Quarter Surface Water Quality and Metals Data

Parameter	W1-SF-SW-01	W1-SF-SW-02	W1-SF-SW-03	Detection limit
pH	7.23	7.3	7.44	na
DO (mg/L)	5.25	61.14	7.24	na
DO (%)	52.4	6.2	72.7	na
Conductivity (μmhos/L)	229	229	229	na
Temperature (°C)	15.40	14.67	15.54	na
Hardness	56	64	58	2.00
TKN (mg/L)	1.9	0.73	1.2	0.50
TOC (mg/L)	7.5	7.2	7.4	1.00
Total Phosphorus (mg/L)	0.18	0.05 U	0.09	0.01
TSS (mg/L)	13	1 U	4	1.00
Metals (ug/L)				
Al	17	4.2 UB	8.6	0.056
Ba	33.016 J	30.016 J	32.016 J	0.0056
Cd	0.015 U	0.015 U	0.015 U	0.015
Cu	0.827	0.547	0.697	0.014
Hg	0.0011 U	0.00067 U	0.0067 U	0.00018
Ag	0.0114 UB	0.0114 UB	0.0114 UB	0.0043
Zn	0.035 U	2.8	0.035 U	0.035

QC Samples

Parameter	W1-SF-FB-01	W1-SF-SW-FD-XX
pH	na	na
DO (%)	na	na
DO (mg/L)	na	na
Conductivity (umhos/L)	na	na
Temperature (C)	na	na
Hardness (mg/L)	na	90
TKN (mg/L)	na	1.9
TOC (mg/L)	na	8
Total Phosphorus (mg/L)	na	0.19
TSS (mg/L)	na	13
Metals (ug/L)		
Aluminum	2.4 UB	16
Barium	0.062 UB	32.016 J
Cadmium	0.019	0.015 U
Copper	0.157	0.847
Mercury	0.00018 U	0.0013 U
Silver	0.0114 UB	0.0114 UB
Zinc	3.0	0.035 U
		W1-SF-SW-01

Table H7 Sunfish Lake Second Quarter Water Quality and Metals Data

Parameter	W2-SF-SW-01	W2-SF-SW-02	W2-SF-SW-03	Detection limit
pH	7.58	7.11	7.57	na
DO (mg/L)	6.52	1.53	5.88	na
DO (%)	47.7	11.1	42.8	na
Conductivity (µmhos/L)	311	314	311	na
Temperature (°C)	2.25	2.61	3.03	na
Hardness (mg/L)	80	88	90	2.00
TKN (mg/L)	0.5	0.75	0.52	0.50
TOC (mg/L)	6.1	6.7	6.4	1.00
Total Phosphorus (mg/L)	0.02	0.06	0.03	0.01
TSS (mg/L)	1 U	2	1 U	1.00
Metals (µg/L)				
Al	11.72	12.72	11.72	0.056
Ba	48.011	53.011	47.011	0.0056
Cd	0.020	0.015 U	0.018	0.015
Cu	1.5	1.0	1.4	0.014
Hg	0.000643 UB	0.001203	0.001203	0.000039
Ag	0.0143 UBJ	0.0143 UBJ	0.0143 UBJ	0.0043
Zn	2.337	1.437	2.337	0.035

QC Samples

Parameter	W2-SF-FB-01	W2-SF-SW-03 W2-SF-FD-00
pH	na	na
DO (%)	na	na
DO (mg/L)	na	na
Conductivity (umhos/L)	na	na
Temperature (C)	na	na
Hardness (mg/L)	na	84
TKN (mg/L)	na	0.59
TOC (mg/L)	na	6.2
Total Phosphorus (mg/L)	na	0.02
TSS (mg/L)	na	1 U
Metals (ug/L)		
Aluminum	0.776 UB	7.42
Barium	0.032 UB	46.011
Cadmium	0.015 U	0.020
Copper	0.30	1.4
Mercury	0.000823	0.000493 UB
Silver	0.022 UBJ	0.0143 UBJ
Zinc	0.104 UB	2.037

Table H8. Sunfish Lake Third Quarter Water Quality and Metals Data

Parameter	W3-SF-SW-01	W3-SF-SW-02	W3-SF-SW-03	Detection limit
pH	8.15	8.00	8.35	na
DO (%)	95.7	96.1	98.7	na
DO (mg/L)	9.25	9.35	9.53	na
Conductivity (umhos/L)	278	279	276	na
Temperature (C)	16.52	16.52	16.87	na
Hardness (mg/L)	68	74	72	2.00
TKN (mg/L)	0.5 U	0.5 U	0.5 U	0.50
TOC (mg/L)	5.4	5.6	5.6	0.20
Total Phosphorus (mg/L)	0.03	0.03	0.03	0.01
TSS (mg/L)	1	2	1 U	1.00
Metals (ug/L)				
Aluminum	71	65	54	0.0099
Barium	38 J	39 J	38 J	0.00088
Cadmium	0.01695	0.01695	0.01795	0.00066
Copper	1.8	1.8	1.8	0.0043
Mercury	0.00118	0.00158	0.00108	0.000063
Silver	0.0186 UBJ	0.0186 UBJ	0.0186 UBJ	0.00058
Zinc	1.918	1.718	1.718	0.0044

QC Samples

Parameter	W3-SF-FB-01	W3-SF-FD-01
pH	na	8.16
DO (%)	na	95.5
DO (mg/L)	na	9.24
Conductivity (umhos/L)	na	279
Temperature (C)	na	16.52
Hardness (mg/L)	na	62
TKN (mg/L)	na	0.5 U
TOC (mg/L)	na	5.5
Total Phosphorus (mg/L)	na	0.03
TSS (mg/L)	na	2
Metals (ug/L)		
Aluminum	0.69	60
Barium	0.080 J	38 J
Cadmium	0.00161 UB	0.01495
Copper	0.0063	1.8
Mercury	0.000418	0.00148
Silver	0.0186 UBJ	0.0186 UBJ
Zinc	0.09 UB	1.818
		W3-SF-SW-01

Table H9.Sunfish Lake Fourth Quarter Water Quality and Metals Data

Parameter	W4-SF-SW-01	W4-SF-SW-02	W4-SF-SW-03	Detection limit
pH	10.28	10.05	10.2	na
DO (%)	135.4	132.2	131.8	na
DO (mg/L)	11.5	11.23	11.09	na
Conductivity (umhos/L)	257	252	258	na
Temperature (C)	23.39	23.41	23.9	na
Hardness (mg/L)	52	56	52	2.00
TKN (mg/L)	0.5 U	0.5 U	0.63	0.50
TOC (mg/L)	7.8	7.7	7.8	0.20
Total Phosphorus (mg/L)	0.08	0.1	0.09	0.01
TSS (mg/L)	2	2	2	1.00
Metals (ug/L)				
Aluminum	25.49	35.49	21.49	0.0099
Barium	15	17	16	0.00088
Cadmium	0.014	0.0089	0.0087	0.00066
Copper	1.4	1.4	1.4	0.0043
Mercury	0.0015	0.0020	0.0016	0.000039
Silver	0.00328 UBJ	0.0039 UBJ	0.00328 UBJ	0.00058
Zinc	0.612	0.692	0.612	0.004

QC Samples

W4-SF-SW-01

Parameter	W4-SF-FB-01	W4-SF-FD-01
pH	na	10.28
DO (%)	na	134.7
DO (mg/L)	na	11.48
Conductivity (umhos/L)	na	258
Temperature (C)	na	23.39
Hardness (mg/L)	na	52
TKN (mg/L)	na	0.63
TOC (mg/L)	na	7.7
Total Phosphorus (mg/L)	na	0.08
TSS (mg/L)	na	2
Metals (ug/L)		
Aluminum	1.99 UB	29.49
Barium	<0.00088 U	15
Cadmium	<0.00066 U	0.013
Copper	<0.0043 U	1.4
Mercury	0.00026 ~~B~~	0.0016
Silver	0.00328 UBJ	0.00328 UBJ
Zinc	0.056 UB	0.652

Sunfish Lake Sediment

In Tier I, wildlife exposures to aluminum, chromium, and vanadium in sediment were identified as potentially unacceptable. Also, the Tier I risk assessment results recommended that a bioavailability study be performed and concurrent benthic community data be collected in order to evaluate the remaining Tier I sediment COCs (chromium, copper, lead, and zinc).

The following tables show the results of the sediment sampling performed by Montgomery Watson (1994), that were used in Tier I assessment. Results of the Tier II bioavailability study and benthic survey are also presented (USACHPPM 1998). A map is also included to show sampling locations used during Tier I sediment sampling, bioavailability sampling and benthic community surveys

Table H10. Tier I Analytical Data set from Operable Unit 2 Feasibility Study Sampling Effort (1992-1993) for Sunfish Lake Sediment (0-1 ft. depth)

Substance (mg/kg)	SFL01	SFL02	SFL03	SFL04.a	SFL04.b	SFL05	SFL06	SFL07	SFL08	SFL09.a	SFL09.b	SFL10
aluminum	18000	13100	14700	13600	13500	13000	12800	13200	7780	10100	11800	9760
antimony	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9
arsenic	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7
barium	265	178	191	155	150	172	168	173	143	217	218	170
beryllium	<0.25	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7
calcium	9640	7400	9490	8130	8460	6710	5820	683	8230	3030	4000	3630
cadmium	<0.427	<0.427	<0.427	<0.427	<0.427	<0.427	<0.427	<0.427	<0.427	1.5	1.96	<0.427
chromium	229	188	183	97.7	93.3	264	250	183	38.2	42.4	43.2	34.9
cobalt	27	<2.5	19.4	17.1	17.9	21.1	22	<2.5	<2.5	15.8	16.8	11.8
copper	92.1	85.9	79.7	67.9	65.7	126	90.3	105	38.8	38	47.6	28.7
cyanide	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22
iron	24000	21000	22000	21000	21000	21000	17000	18000	9700	18000	23000	14000
lead	192	128	133	105	110	202	134	144	82.2	73	82.3	102
magnesium	5490	4780	5320	5940	6220	4390	3610	3860	2700	2790	3570	2920
manganese	380	371	365	342	361	370	296	319	406	435	600	454
mercury	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087
nickel	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5
potassium	2520	1770	2200	1870	1760	1660	1880	1940	1620	1180	1530	1220
silver	<0.5	<0.5	<0.5	2.69	<0.5	<0.5	<0.5	4.82	<0.5	<0.5	<0.5	<0.5
sodium	471	461	425	353	361	455	371	413	293	177	207	169
thallium	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5
vanadium	61.5	44.3	54.4	50	50.9	48	47.7	44.2	24.3	45.3	53.7	36
zinc	329	286	343	240	229	334	339	279	113	401	501.1	230
acetone	<0.045	<0.045	0.32	0.32	<0.045	<0.045	<0.045	<0.045	<0.045	<0.045	<0.045	0.17
methylethyl ketone	<0.005	<0.005	0.04	0.05	<0.005	0.03	<0.005	0.02	0.03	0.02	0.01	0.04
SVOCs	—	—	—	—	—	—	—	—	—	—	—	—
herbicides	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
TPH	—	—	—	—	—	—	—	—	—	—	—	—
explosives	—	—	—	—	—	—	—	—	—	—	—	—
dioxins/furans	—	—	—	—	—	—	—	—	—	—	—	—
UNK069	—	—	—	—	—	—	—	—	400	—	—	—
UNK091	—	—	0.04	—	—	—	—	—	—	—	—	—
UNK092	—	—	—	—	—	—	—	—	—	—	—	0.05

VOCs-volatile Organic Compounds SVOCs – Semivolatile Organic Compounds TPH – total petroleum hydrocarbons

Table H11. Chemical Characteristics associated with surficial sediments from Round Lake, Sunfish Lake, and Snail Lake

	Overlying Water[a]		Porewater									
location	temp. (°C)	DO (ppm)	pH	TOC (mg/g)	CEC (meq/g)	AVS	SEM_{Cd}	SEM_{Cu}	SEM_{Ni}	SEM_{Pb}	SEM_{Zn}	SEM/AVS[b] ratio
RL0501	18.3	4	6.8	230	0.076	28	0.277	<0.01	0.518	0.858	23.2	0.89
RL0501d	--	--	--	--	--	20	0.05	<0.01	0.5	0.3	24	1.24
RL0701	19	3.75	6.6	77	0.049	20	0.01	0.03	0.07	0.06	1.2	0.068
RL0901	19	4.5	6.8	200	0.078	4.1	<0.01	<0.01	0.02	0.03	0.33	0.093
RLXX01	19.5	6.1	7.3	69	0.018	2.1	<0.01	<0.01	0.01	0.01	0.14	0.076
SFL0301	21	3.4	7.2	64	0.052	2.6	<0.01	<0.01	0.01	0.03	0.18	0.085
SFL0401	18.5	5.1	6.88	53	0.047	5.7	<0.01	0.02	0.06	0.09	0.66	0.146
SFL0701	18.5	1.5	6.6	43	0.014	2.3	<0.01	<0.01	<0.01	0.02	0.09	0.048
SFLXX01	18	3.5	6.4	120	0.061	13	<0.01	<0.01	0.04	0.36	2.2	0.2
SFLXX01d	--	--	--	--	--	10	<0.01	<0.01	0.04	0.29	1.7	0.203
SFLSD1	21	5.2	--	--	--	20	<0.01	0.08	0.09	0.19	1.2	0.078
SFLSD2	19	4.7	--	--	--	<1	<0.01	0.029	0.017	0.04	0.225	>0.311
SNL111	19.5	4.25	7.1	120	0.048	21	<0.01	0.04	0.06	0.16	0.61	0.041
SNL111d	--	--	--	--	--	18	<0.01	0.05	0.07	0.16	0.48	0.042
SNL121	21	5.6	7	36	0.00074	<1	<0.01	<0.01	<0.01	<0.01	0.03	>0.03

Sites are identified by initials: Round Lake (RL), Sunfish Lake (SFL), and Snail Lake (SNL).
d- field duplicate.
SE-Cd --denotes the simultaneously extracted cadmium
[a] Taken at the sediment-water interface (approximately 8 cm form the sediment surface).
[b] This value is derived with ΣSEM, including Cd, Cu, Hg, Ni, Pb, and Zn.
Table taken from the Bioavailability of Sediment-Metals in Round and Sunfish Lakes (USACHPPM 1998)

Table H12. Occurances of Species and Individuals During Benthic Community Diversity Evaluation (USACHPPM 1998)

	Snail Lake		Sunfish Lake				Round Lake			
Taxonomy	SNL111	SNL121	SFL0301	SFL0401	SFL0701	SFLXX01	RL0501	RL0701	RL0901	RLXX01
ARACHNIDA										
Lebertia sp.	0	0	0	1	0	0	0	0	0	0
BIVALVIA										
Corbicula fluminae	2	0	0	0	0	0	0	0	0	0
Sphaerium striatum	6	0	16	26	1	0	0	1	1	1
CRUSTACEA										
Crangonyx gracilis	5	43	1	0	2	24	67	3	25	3
INSECTA										
Ablebesmia sp.	0	0	0	0	4	0	0	0	0	0
Baetis sp.	0	0	0	0	1	0	0	0	0	0
Caenis sp.	0	11	0	0	0	0	0	0	0	0
Chaoborus punctipennis	0	0	35	39	8	29	12	27	1	47
Chironomus riparious	0	0	2	0	0	0	0	0	0	0
Chrysops sp.	1	0	0	0	0	0	0	0	0	0
Corixid sp.	0	0	0	0	0	0	0	0	0	1
Culicoides sp.	0	0	0	0	1	0	0	0	1	0
Glyptotendipes lobiferus	0	0	0	51	3	1	24	4	0	0
Ischnura sp.	0	0	0	0	0	2	0	0	0	0
Paratendipes sp.	0	4	1	2	0	8	3	4	0	0
Plea striolata	0	0	0	0	0	0	0	0	1	0
Polypedilius fallax	0	7	5	16	2	52	3	2	0	19
Procladius sublettei	2	1	3	8	1	1	0	1	0	0
Tetragoneuria cynosura	0	1	0	0	1	1	0	0	0	0
Tricoptera sp.	0	0	0	0	0	0	1	0	0	0
Ceraclea sp.	0	1	0	0	0	0	0	0	0	0
GASTROPODA										
Birgella subglobosa	11	3	0	0	0	0	0	0	0	0
Ferrissia walkeri	1	0	0	0	0	0	0	0	0	0
Gyralus deflatus	0	0	0	3	2	0	0	0	1	2
Physella gyrina	0	0	6	2	18	12	0	4	19	7
Planorbella armigera	2	1	0	0	0	0	0	0	0	0
Valvata tricarinata	0	1	0	0	0	0	0	0	0	0
HIRUDINEA										
Erpobdella punctata	0	1	0	0	0	0	0	0	0	0
Helobdella papillata	4	1	0	0	0	0	0	1	2	0
OLIGOCHAETA										
Amphichaeta americanus	14	12	13	6	6	0	4	0	0	0
Aulodrilus pluriseta	89	13	0	1	4	4	3	5	0	4
Lumberculus variegatus	0	1	0	0	0	0	0	0	0	0
INDIVIDUALS (N)	137	101	82	155	53	134	117	52	51	84
SPECIES (S)	11	15	9	11	13	10	8	10	8	8

Figure H2. Sample Locations from Previous Studies at Sunfish Lake

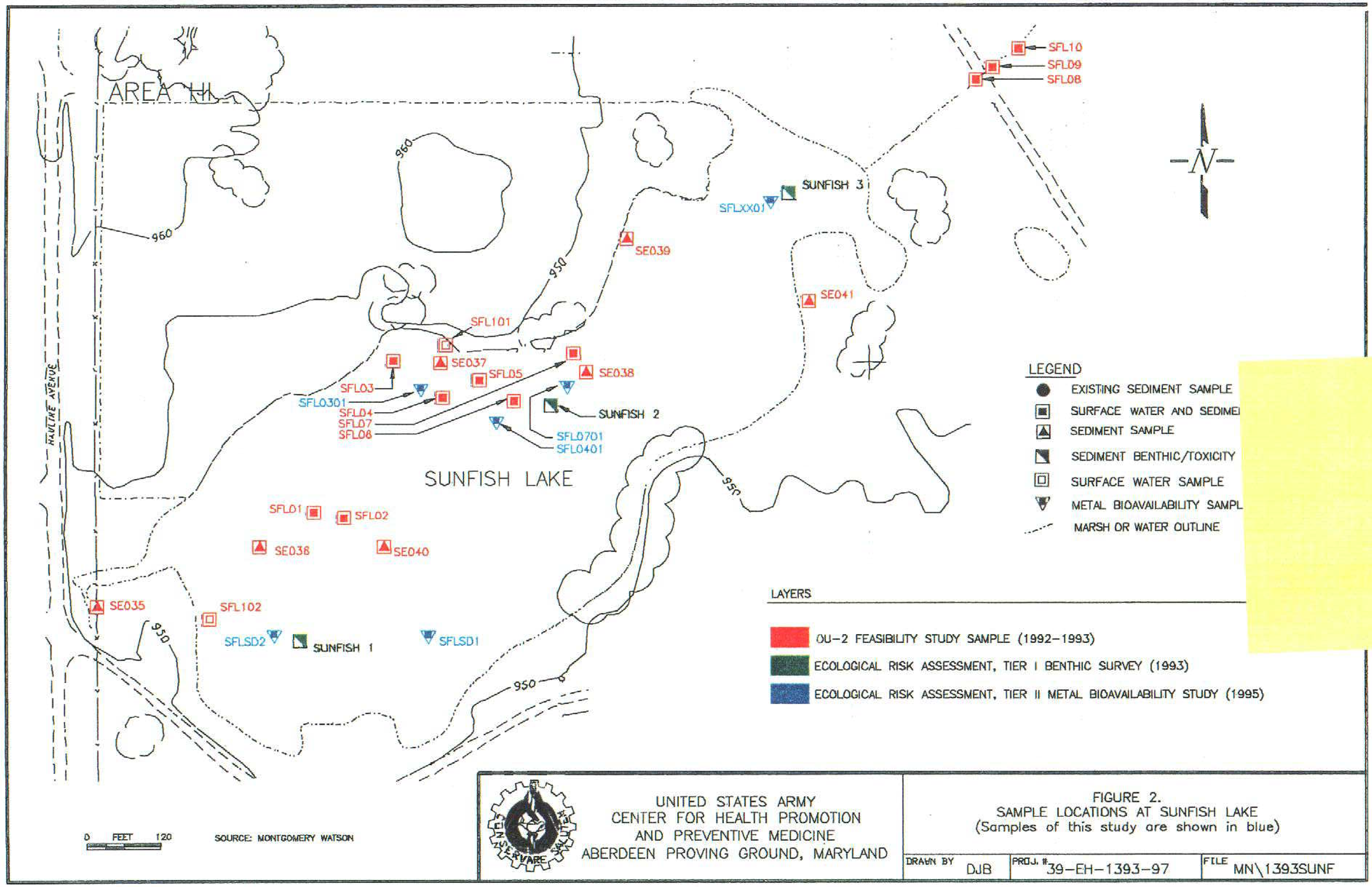


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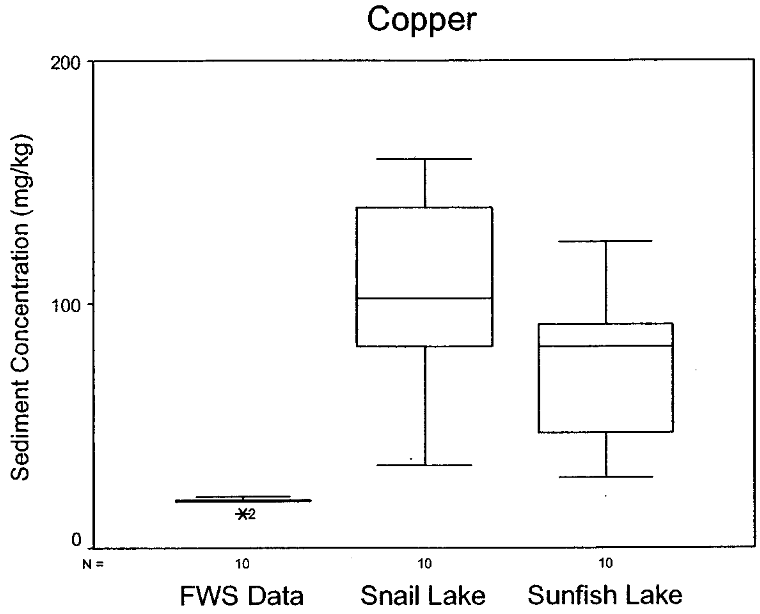
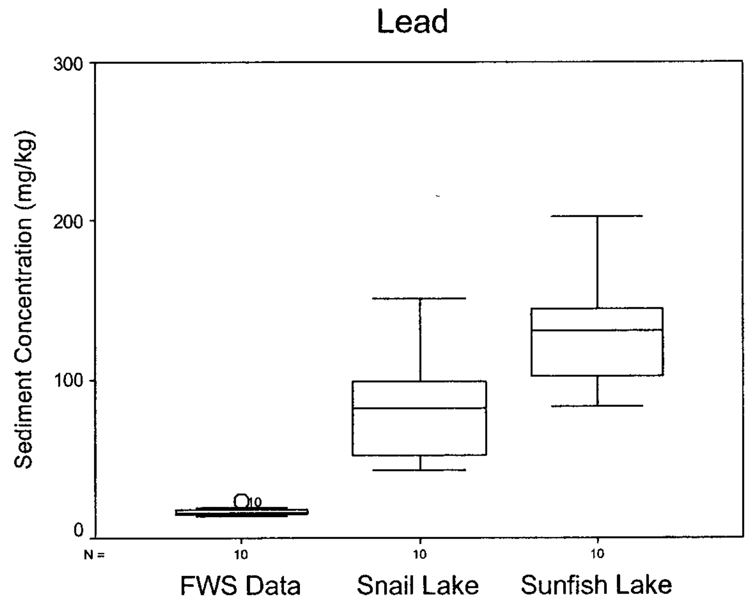
Figure H-3. Sediment Concentration Boxplots for Copper and Lead Related to Boxplots for Background Datasets





Box plots are summary plots of the data based on the median, quartiles, and extreme values. The box represents the interquartile range (25th – 75th percentiles), which contains 50 percent of the values. The whiskers are lines that extend from the box to the highest and lowest values, excluding outliers. A line across the box indicates the median.

APPENDIX I

Round Lake Sampling Locations and Analytical Measurement Results

Table of Contents

Attachment: U.S. Fish and Wildlife Service Round Lake Management Plan

Round Lake Data Presentation Summary

The map of Round Lake (Fig.I1), shows the location where each media sample was taken. The map image is from an orthophoto taken during 1997, obtained from Wenck Associates, Inc., Maple Plains, Minnesota. Global Positioning System (GPS) coordinates were used to plot the locations. These coordinates recorded in the field, during sampling, are reported in tables I1 and I5. For mapping purposes, second quarter sampling locations were not mapped by their GPS coordinates. (However, the following GPS data tables do show the recorded coordinates). Second quarter samples were mapped with first quarter coordinates. During second quarter sampling there was a problem with the crypto key, causing all samples to show imprecise coordinates. Based on landmarks, the field team leader noted in the field logbook (USACHPPM 2000) that sample locations for the second quarter were as close as possible to first quarter coordinates. Maps from the Work Plan (USACHPPM 1999) were also used to pinpoint the location to be sampled.

The tables I6 through I9 summarize the data for Round Lake surface water and sediment samples taken during the water monitoring activities. The summaries include the field and analytical result for each analyte at each sample location. Summary data used for statistical analysis such as the central tendencies, standard deviations and confidence intervals are report in tables found in Appendix A.

The number of digits reported for each value in the following tables does not necessarily reflect the precision and accuracy of the measurement. The data tables present significant figures as they were originally reported by the analyzing laboratory. Significant figures reported for the analytes were based on the analytical detection limit values. However, some sample values have more significant figures than others to demonstrate that blank correction was discontinued, as was requested by USEPA comments on the draft version of this report (see Appendix R). Data quality is discussed in Appendix K.

The results are reported in the following manner to facilitate data review.

- If the detection limit is sample-specific (as it was for the pesticides in sediment), the value reported for the achieved detection limit for a site is the highest detection limit for that parameter for all samples (including the field duplicate) from that site.

- If a parameter was not detected in any samples from a site (e.g. Pond G, water, alpha-BHC) then the minimum and maximum values represent the lowest and highest achieved detection limits for those data.

- Any concentrations that were below the detection limit but still quantified are identified in Volume 2 of this report as estimated values (J-flagged). These values are reported and used uncensored in the following tables.

- If a parameter was undetected in at least one but not all samples from a site, then the minimum reflects the lowest detection limit of the undetected samples. If there was a

J-flagged value lower than the highest undetected value, the J-flagged value was reported as the maximum because the value was actually detected but estimated.

- Field duplicates (FD) were used as quality control samples and were not included in the reported minima and maxima columns. However, if an analyte was detected in the FD and not in it's corresponding sample, then the FD detection was counted in the detection frequency column (only in the numerator).

- When laboratory replicate samples (R1, R2) were available, the values from the primary sample (R1) were reported. See Appendix K section K1.2.1 for explanation.

Definition of Data Qualifiers (Flags)

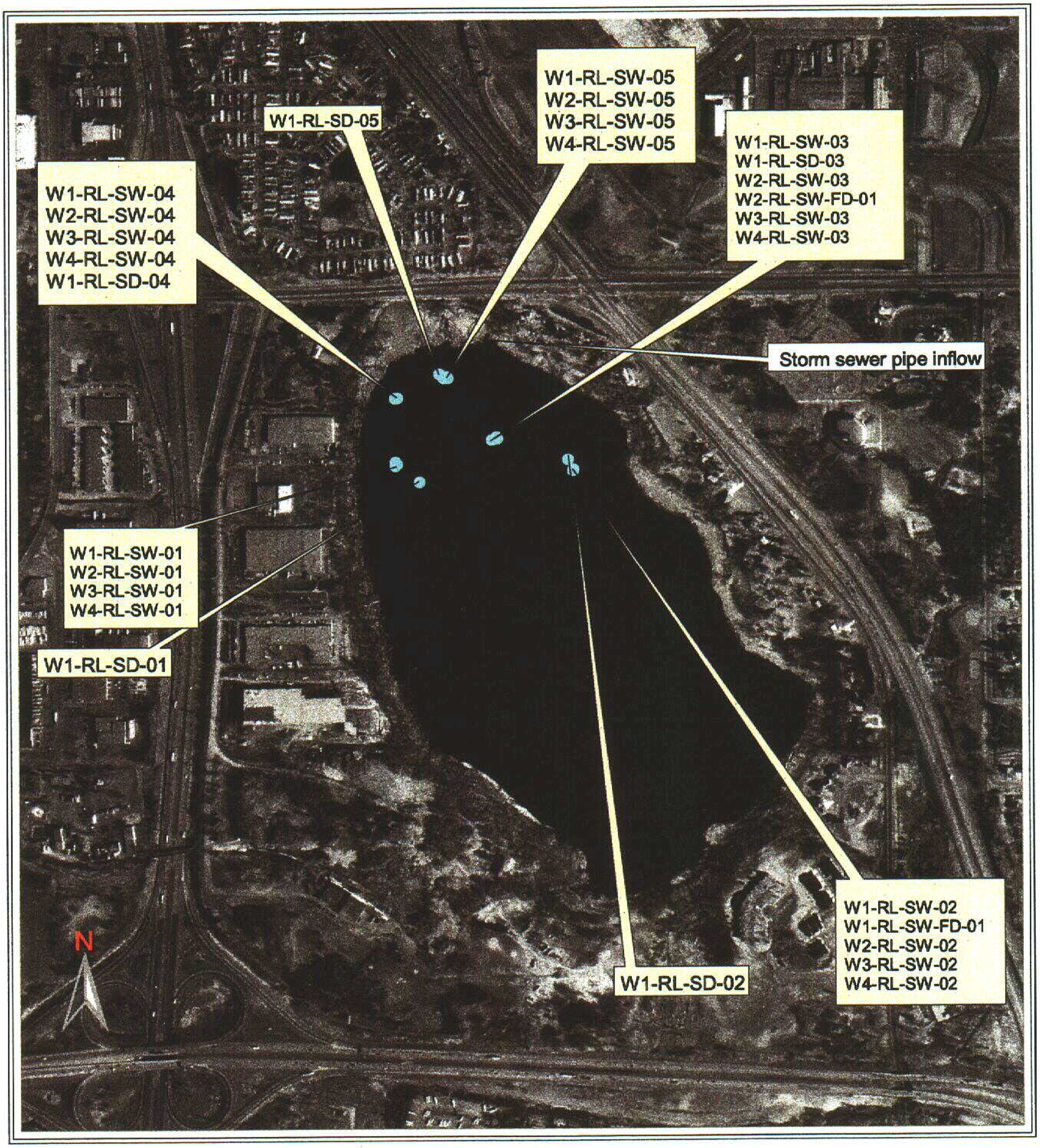
During the data validation process, all laboratory data were assigned appropriate data validation flags and reason codes. Validation flags used in the following tables are defined as follows:

No flag	When the analyte value reported is detected and usable. The integrity of the analyte's identification, accuracy, and precision has been validated
"U"	When the material was analyzed for, but not detected above the level of the associated detection limit value.
"UB"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with.
"J"	When the associated value is an estimated quantity. Indicating there is cause to question accuracy or precision of the reported value. The value is still used in the risk assessment.
"UJ"	When the analyte was analyzed for, but not detected, above the associated value, however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"UBJ"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with; however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.
"R"	When the analyte value reported is unusable. The integrity of the analyte's identification, accuracy, precision, or sensitivity has raised significant question as to the reality of the information presented.

Other information regarding data validation can be found in Volume 2 (CD ROM version) of the report.

Note: References cited in the text of this appendix can be found in the reference section of the main Tier II Ecological Risk Assessment Report.

Figure I1. Round Lake Water Characterization and Sediment Bioassay Sampling Locations



Wx= quarter
RL= Round Lake
SW=Surface Water
SD= Sediment

200 0 200 Meters

Table I1. Round Lake First Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W1-RL-SW-01	27-Sep-99	4-8"	45	4	35.11	93	10	49.96
W1-RL-SW-02 W1-RL-SW-FD-01	27-Sep-99	4"	45	4	34.83	93	10	36.03
W1-RL-SW-03	27-Sep-99	4"	45	4	36.57	93	10	42.06
W1-RL-SW-04	27-Sep-99	4"	45	4	38.89	93	10	50.12
W1-RL-SW-05	27-Sep-99	4"	45	4	40.09	93	10	46.1

Table I2. Round Lake Second Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W2-RL-SW-01	19-Jan-00	7"	45	4	35.35	93	10	48.98
W2-RL-SW-02	19-Jan-00	8"	45	4	35.86	93	10	41.69
W2-RL-SW-03 W2-RL-SW-FD-01	19-Jan-00	6"	45	4	35.93	93	10	44.78
W2-RL-SW-04	19-Jan-00	8"	45	4	38.83	93	10	54.69
W2-RL-SW-05	19-Jan-00	?	45	4	33.96	93	10	41.69

Table I3. Round Lake Third Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W3-RL-SW-01 W3-RL-SW-FD-01	26-Apr-00	6-10”	45	4	35	93	10	50.09
W3-RL-SW-02	26-Apr-00	6-10”	45	4	34.74	93	10	36.07
W3-RL-SW-03	26-Apr-00	6-10”	45	4	36.56	93	10	42.16
W3-RL-SW-04	26-Apr-00	6-10”	45	4	38.85	93	10	50.14
W3-RL-SW-05	26-Apr-00	6-10”	45	4	40	93	10	46.27

Table I4. Round Lake Fourth Quarter Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N[*]			W[*]		
			degrees	minutes	seconds	degrees	minutes	seconds
W4-RL-SW-01 W4-RL-SW-FD-01	23-Jun-00	4-6"	45	4	35.23	93	10	50.12
W4-RL-SW-02	23-Jun-00	4-6"	45	4	34.81	93	10	36.03
W4-RL-SW-03	23-Jun-00	4-6"	45	4	36.66	93	10	42.12
W4-RL-SW-04	23-Jun-00	4-6"	45	4	38.86	93	10	50.12
W4-RL-SW-05	23-Jun-00	4-6"	45	4	40	93	10	46

Table I5. Round Lake Sediment Bioassay Sampling GPS Coordinates (1st Quarter)

Sample Number	Date	Sample depth	N[*]			W[*]		
			degrees	minutes	seconds	degrees	minutes	seconds
W1-RL-SD-01	18-Sep-99	6"	45	4	34.09	93	10	48.22
W1-RL-SD-02	18-Sep-99	6'	45	4	35.38	93	10	36.47
W1-RL-SD-03	18-Sep-99	6'	45	4	36.49	93	10	42.49
W1-RL-SD-04	18-Sep-99	5'6"	45	4	38.87	93	10	49.95
W1-RL-SD-05	18-Sep-99	Na	45	4	40.25	93	10	46.65

[*] The GPS unit used in the study was a military-issue Precision Light Weight GPS Receiver (PLGR). The Department of Defense (DoD) intentionally places errors in the GPS navigation and timing signal, in a process called selective availability (Department of the Army 1993). Errors resulting from this process are unpredictable and can produce significant horizontal and elevation errors. Military GPS receivers must have crypto keys loaded to detect and nullify selective availability errors, in order to allow for more accurate position data (Department of the Army 1993). The PLGR was equipped with a crypto key, and was therefore not subject to the selective availability and spoofing errors that affect non-military GPS. (The only exception was during the second quarter). The PLGR operations manual (Department of the Army 1995) states that with crypto keys installed, the PLGR provides horizontal 2-D position accuracies better than the worst-case error of 10 meters.

Table I6. Round Lake First Quarter Surface Water Quality and Metals Data

Parameter	W1-RL-SW-01	W1-RL-SW-02	W1-RL-SW-03	W1-RL-SW-04	W1-RL-SW-05	Detection limit
pH	8.63	8.58	8.49	8.72	8.03	na
DO (mg/L)	8.63	8.64	8.41	9.32	6.18	na
DO (%)	90.3	88.4	86.4	95.6	65.7	na
Conductivity (μmhos/L)	476	476	474	476	441	na
Temperature (°C)	16.47	16.44	16.62	16.57	16.88	na
Hardness (mg/L)	144	120	138	120	120	2.00
TKN (mg/L)	1.2	1.3	1.2	1.2	1.2	0.50
TOC (mg/L)	10	13	14	15	12	1.00
Total Phosphorus (mg/L)	0.07	0.07	0.07	0.13	0.08	0.01
TSS (mg/L)	20	19	30	26	26	1.00
Metals (μg/L)						
Barium	62.016 J	62.016 J	62.016 J	62.016 J	55.016 J	0.0056
Cadmium	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015
Copper	0.51	0.63	0.61	0.55	1.0	0.014
Mercury	0.00093	0.00078	0.00069	0.00078	0.00099	0.00018
Silver	0.0114 UB	0.0114 UB	0.0114 UB	0.0114 UB	0.0114 UB	0.0043
Zinc	0.035 U	0.035 U	0.035 U	0.035 U	0.035 U	0.035

QC Samples

Parameter	W1-RL-SW-FB-01	W1-RL-SW-FD-XX
pH	na	na
DO (%)	na	na
DO (mg/L)	na	na
Conductivity (umhos/L)	na	na
Temperature (C)	na	na
Hardness (mg/L)	na	126
TKN (mg/L)	na	1.1
TOC (mg/L)	na	11
Total Phosphorus (mg/L)	na	0.09
TSS (mg/L)	na	22
Metals (ug/L)		
Barium	0.035 UB	63.016 J
Cadmium	0.015 U	0.015 U
Copper	0.16	0.50
Mercury	0.00018 U	0.00073 U
Silver	0.0114 UB	0.0114 UB
Zinc	2.4	0.035 U
		W1-RL-SW-02

Table I7. Round Lake Second Quarter Surface Water Quality and Metals Data

Parameter	W2-RL-SW-01	W2-RL-SW-02	W2-RL-SW-03	W2-RL-SW-04	W2-RL-SW-05	Detection limit
pH	7.87	7.88	7.82	7.32	8.9	na
DO (mg/L)	12.40	10.01	9.66	8.59	10.39	na
DO (%)	87.5	71.5	68.7	61.1	73.8	na
Conductivity (μmhos/L)	669	639	637	648	653	na
Temperature (°C)	1.60	1.7	2.03	1.57	1.39	na
Hardness (mg/L)	194	180	188	178	190	2.00
TKN (mg/L)	1.4	1.4	1.4	1.2	1.5	0.50
TOC (mg/L)	11	11	11	11	11	1.00
Total Phosphorus (mg/L)	0.06	0.04	0.05	0.07	0.06	0.01
TSS (mg/L)	8	8	4	8	8	1.00
Metals (μg/L)						
Barium	89.011	88.011	88.011	85.011	91.011	0.0056
Cadmium	0.020	0.015 U	0.021	0.054	0.026	0.015
Copper	1.2	1.1	1.1	1.6	1.1	0.014
Mercury	0.00058 UB	0.00085	0.00048 UB	0.00078	0.00061 UB	0.000039
Silver	0.0143 UBJ	0.0143 UBJ	0.0143 UB	0.0143 UBJ	0.0143 UBJ	0.0043
Zinc	2.437	2.737	2.737	4.037	3.237	0.035

QC Samples

Parameter	W2-RL-FB-01	W2-RL-FD-01
pH	na	na
DO (%)	na	na
DO (mg/L)	na	na
Conductivity (umhos/L)	na	na
Temperature (C)	na	na
Hardness (mg/L)	na	190
TKN (mg/L)	na	1.1
TOC (mg/L)	na	12
Total Phosphorus (mg/L)	na	0.05
TSS (mg/L)	na	6
Metals (ug/L)		
Barium	0.054 UB	89.011
Cadmium	0.015 U	0.016
Copper	0.015	1.0
Mercury	0.00038 UB	0.00081
Silver	0.0143 UBJ	0.0143 UBJ
Zinc	0.13 UB	2.737
		W2-RL-SW-03

na: not available

Table I8. Round Lake Third Quarter Surface Water Quality and Metals Data

Parameter	W3-RL-SW-01	W3-RL-SW-02	W3-RL-SW-03	W3-RL-SW-04	W3-RL-SW-05	Detection limit
pH	8.2	8.16	8.19	8.21	8.24	na
DO (%)	103.4	98.2	101.6	102.1	107.4	na
DO (mg/L)	10.28	9.67	9.96	10.06	10.51	na
Conductivity (umhos/L)	584	587	587	585	586	na
Temperature (C)	15.55	15.98	NA	15.88	16.17	na
Hardness (mg/L)	168	170	170	170	172	2.00
TKN (mg/L)	0.82	0.8	0.82	3.2	0.87	0.50
TOC (mg/L)	8	8.4	8.3	8.2	8.1	0.20
Total Phosphorus (mg/L)	0.03	0.04	0.03	0.04	0.04	0.01
TSS (mg/L)	4	4	4	3	6	1.00
Metals (ug/L)						
Barium	82 J	86 J	85 J	87 J	87 J	0.00088
Cadmium	0.01795	0.01995	0.02095	0.02195	0.02595	0.00066
Copper	1.2	1.3	1.3	1.3	1.4	0.00430
Mercury	0.000628	0.000888	0.000818	0.001018	0.000838	0.000063
Silver	0.01858 UBJ	0.01858 UBJ	UB 0.01858 J	UB 0.01858 J	0.01858 UBJ	0.00058
Zinc	1.418 J	1.818 J	1.718 J	1.718 J	1.818 J	0.00440

QC Samples

Parameter	W3-RL-FB-01	W3-RL-FD-01
pH	na	8.2
DO (%)	na	103.3
DO (mg/L)	na	10.28
Conductivity (umhos/L)	na	584
Temperature (C)	na	15.56
Hardness (mg/L)	na	174
TKN (mg/L)	na	0.75
TOC (mg/L)	na	8.8
Total Phosphorus (mg/L)	na	0.03
TSS (mg/L)	na	3
Metals (ug/L)		
Barium	0.066 J	81 J
Cadmium	0.00161 UB	0.01695
Copper	<0.0043 U	1.2
Mercury	0.000258 UB	0.000698
Silver	0.01858 UBJ	0.01858 UBJ
Zinc	0.118 UB	1.418 J
		W3-RL-SW-01

Table I9. Round Lake Fourth Quarter Surface Water and Metals Data

Parameter	W4-RL-SW-01	W4-RL-SW-02	W4-RL-SW-03	W4-RL-SW-04	W4-RL-SW-05	Detection limit
pH	8.46	8.55	8.55	8.44	8.53	na
DO (%)	108.9	115.1	117.8	107.9	115.5	na
DO (mg/L)	9.61	10.04	10.26	9.42	10.02	na
Conductivity (umhos/L)	524	522	521	524	521	na
Temperature (C)	21.46	21.86	22.16	21.97	22.2	na
Hardness (mg/L)	142	142	138	138	136	2.00
TKN (mg/L)	0.65	0.64	0.62	0.5 U	0.69	0.50
TOC (mg/L)	13	17	21	19	20	0.20
Total Phosphorus (mg/L)	0.04	0.04	0.04	0.04	0.04	0.01
TSS (mg/L)	6	8	6	7	LOST	1.00
Metals (ug/L)						
Barium	70	70	69	71	70	0.00088
Cadmium	0.012	0.011	0.0089	0.013	0.012	0.00066
Copper	0.75	0.77	0.76	0.86	0.78	0.0043
Mercury	0.0010	0.00076	0.00084	0.00080	0.0011	0.000039
Silver	0.00328 UBJ	0.00328 UBJ	0.00328 UBJ	0.00328 UBJ	0.00328 UBJ	0.00058
Zinc	0.712	0.612	0.472	0.812	0.992	0.004

QC Samples

Parameter	W4-RL-FB-01	W4-RL-FD-01
pH	na	8.46
DO (%)	na	108.8
DO (mg/L)	na	9.6
Conductivity (umhos/L)	na	524
Temperature (C)	na	21.47
Hardness (mg/L)	na	140
TKN (mg/L)	na	0.7
TOC (mg/L)	na	20
Total Phosphorus (mg/L)	na	0.04
TSS (mg/L)	na	6
Metals (ug/L)		
Barium	<0.00088 U	70
Cadmium	<0.00066 U	0.010
Copper	<0.0043 U	0.74
Mercury	0.00071	0.0015
Silver	0.00328 UBJ	0.00328 UBJ
Zinc	0.056 UB	2.852
		W4-RL-SW-01

Round Lake Sediment

In Tier I, wildlife exposures to aluminum, cadmium, and vanadium in sediment were identified for further consideration. Also, the Tier I risk assessment results recommended that a sediment-metal bioavailability study be performed at the northern end of Round Lake, sediment toxicity tests be performed at the southern end of the lake, and concurrent benthic community data be collected in order to evaluate the remaining Tier I sediment COCs (aluminum, cadmium, chromium, copper, silver, vanadium, and zinc).

The following tables show the results of the sediment sampling performed by Montgomery Watson (1994), that were used in Tier I assessment. Results of the Tier II bioavailability study and benthic survey are also presented (USACHPPM 1998). A map is included to show Tier I sediment sampling locations, and sampling locations for the bioavailability study and benthic community surveys.

Table I10. Tier I Analytical Data set from Operable Unit 2 Feasibility Study Sampling Effort (1992-1993) for Round Lake Sediment (0-1 ft. depth)

Substance (mg/kg)	RL02	RL05	RL06	RL07	RL08	RL09	RL10	RL11	RL12	RL13	RL14	RL15	RL16	RL17.A	RL17.B	RL18	RL19	RL20
aluminum	5820	8850	13100	11500	2810	12000	7350	6920	10500	10000	15400	1260	7010	5290	8300	12900	12600	1850
antimony	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9	<82.9
arsenic	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7	<12.7
barium	154	121	196	182	51.6	213	157	138	181	170	214	11.6	138	120	171	177	156	15.2
beryllium	<0.25	<0.25	<0.25	<0.25	<0.25	0.74	<0.25	<0.25	<0.25	<0.25	<0.25	<0.25	<0.25	<0.25	<0.25	<0.25	<0.25	<0.25
calcium	8880	9400	23300	22600	5460	56000	9060	13000	11700	53000	18500	1810	15200	12300	16700	15900	16400	3680
cadmium	<0.427	2.73	<0.427	3.81	<0.427	7.53	<0.427	<0.427	<0.427	<0.427	5.08	<0.427	<0.427	<0.427	<0.427	11.2	<0.427	<0.427
chromium	21	56.8	64.9	96.5	20.9	110	15.2	68.2	72.7	39.6	55.2	3.96	<0.974	11.8	19.7	172	142	4.23
cobalt	18	13.3	19.7	<2.5	<2.5	19.1	<2.5	<2.5	<2.5	14.1	19.5	<2.5	<2.5	<2.5	<2.5	<2.5	<2.5	<2.5
copper	43.2	521	608	905	139	1250	20.9	350	399	196	168	<3.38	<3.38	<3.38	51	365	320	<3.38
cyanide	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22	<1.22
iron	13000	16000.1	23000	19000	4500	26000	19000	12000	16000	16000	23000	2600	11300	10000	14000	20000	18200	3000
lead	<10.0	9.7	<10.0	<10.0	<10.0	113	<10.0	<10.0	<10.0	<10.0	<10.0	<10.0	<10.0	<10.0	<10.0	299	257	<10.0
magnesium	2880	8280	8470	6200	1340	9990	3650	3440	4340	7270	5799	906	4260	3370	4360	6100	5300	786
manganese	295	367	561	319	90.5	750	335	325	435	351	301	62	372	303	517	525	377	51.5
mercury	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087	<0.087
molybdenum	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0
nickel	<7.5	<7.5	<7.5	<7.5	<7.5	32.2	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5	<7.5
potassium	859	1330	1930	1890	445	1520	1170	<142.0	1750	1720	2610	<142.0	<142.0	<142.0	1820	<142.0	<142.0	<142.0
selenium	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4	<12.4
silver	<0.5	<0.5	2.45	86.8	<0.5	1.07	58.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
sodium	237	302	589	522	165	300	361	628	706	445	5790	<50.0	1190	570	774	1320	1210	124
thallium	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5	<12.5
vanadium	30.3	37.6	49.9	46.3	11.6	37.8	36.1	28.3	46	39.5	63	5.02	40.6	31	46.7	<2.0	<2.0	7.04
zinc	82.5	414	399	639	125	892	90	341	453	263	270	10.3	122	78.8	98.3	860	772	21.4
VOCs	Analytes not analyzed																	
SVOCs																		
herbicides																		
TPH																		
explosives																		
dioxins/furans																		
h. epoxide	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
PCB 1248	<0.040	<0.040	<0.040	<0.040	<0.040	1.280 R	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040
PCB 1254	<0.040	<0.040	<0.040	<0.040	<0.040	0.240 R	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040	<0.040
α-endosulfan	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
p,p-DDD	<0.011	<0.011	<0.011	<0.011	<0.011	0.4	<0.011	<0.011	<0.011	<0.011	<0.011	<0.011	<0.011	<0.011	<0.011	<0.011	<0.011	<0.011
p,p-DDE	<0.014	<0.014	<0.014	<0.014	<0.014	0.15	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014
p,p-DDT	<0.010	<0.010	<0.010	<0.010	<0.010	0.33	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010

VOCs-volatile Organic Compounds SVOCs – Semivolatile Organic Compounds TPH – total petroleum hydrocarbons

Table I11. Chemical Characteristics associated with surficial sediments from Round Lake, Sunfish Lake, and Snail Lake

location	Overlying Water[a] temp. (°C)	Overlying Water[a] DO (ppm)	Porewater pH	TOC (mg/g)	CEC (meq/g)	AVS	SEM_{Cd}	SEM_{Cu}	SEM_{Ni}	SEM_{Pb}	SEM_{Zn}	SEM/AVS[b] ratio
RL0501	18.3	4	6.8	230	0.076	28	0.277	<0.01	0.518	0.858	23.2	0.89
RL0501d	--	--	--	--	--	20	0.05	<0.01	0.5	0.3	24	1.24
RL0701	19	3.75	6.6	77	0.049	20	0.01	0.03	0.07	0.06	1.2	0.068
RL0901	19	4.5	6.8	200	0.078	4.1	<0.01	<0.01	0.02	0.03	0.33	0.093
RLXX01	19.5	6.1	7.3	69	0.018	2.1	<0.01	<0.01	0.01	0.01	0.14	0.076
SFL0301	21	3.4	7.2	64	0.052	2.6	<0.01	<0.01	0.01	0.03	0.18	0.085
SFL0401	18.5	5.1	6.88	53	0.047	5.7	<0.01	0.02	0.06	0.09	0.66	0.146
SFL0701	18.5	1.5	6.6	43	0.014	2.3	<0.01	<0.01	<0.01	0.02	0.09	0.048
SFLXX01	18	3.5	6.4	120	0.061	13	<0.01	<0.01	0.04	0.36	2.2	0.2
SFLXX01d	--	--	--	--	--	10	<0.01	<0.01	0.04	0.29	1.7	0.203
SFLSD1	21	5.2	--	--	--	20	<0.01	0.08	0.09	0.19	1.2	0.078
SFLSD2	19	4.7	--	--	--	<1	<0.01	0.029	0.017	0.04	0.225	>0.311
SNL111	19.5	4.25	7.1	120	0.048	21	<0.01	0.04	0.06	0.16	0.61	0.041
SNL111d	--	--	--	--	--	18	<0.01	0.05	0.07	0.16	0.48	0.042
SNL121	21	5.6	7	36	0.00074	<1	<0.01	<0.01	<0.01	<0.01	0.03	>0.03

Sites are identified by initials: Round Lake (RL), Sunfish Lake (SFL), and Snail Lake (SNL).
d- field duplicate.
SE-Cd –denotes the simultaneously extracted cadmium
[a] Taken at the sediment-water interface (approximately 8 cm form the sediment surface).
[b] This value is derived with ΣSEM, including Cd, Cu, Hg, Ni, Pb, and Zn.
Table taken from the Bioavailability of Sediment-Metals in Round and Sunfish Lakes (USACHPPM 1998)

Table I12. Occurances of Species and Individuals During Benthic Community Diversity Evaluation (USACHPPM 1998)

	Snail Lake		Sunfish Lake				Round Lake			
Taxonomy	SNL111	SNL121	SFL0301	SFL0401	SFL0701	SFLXX01	RL0501	RL0701	RL0901	RLXX01
ARACHNIDA										
Lebertia sp.	0	0	0	1	0	0	0	0	0	0
BIVALVIA										
Corbicula fluminae	2	0	0	0	0	0	0	0	0	0
Sphaerium striatum	6	0	16	26	1	0	0	1	1	1
CRUSTACEA										
Crangonyx gracilis	5	43	1	0	2	24	67	3	25	3
INSECTA										
Ablebesmia sp.	0	0	0	0	4	0	0	0	0	0
Baetis sp.	0	0	0	0	1	0	0	0	0	0
Caenis sp.	0	11	0	0	0	0	0	0	0	0
Chaoborus punctipennis	0	0	35	39	8	29	12	27	1	47
Chironomus riparious	0	0	2	0	0	0	0	0	0	0
Chrysops sp.	1	0	0	0	0	0	0	0	0	0
Corixid sp.	0	0	0	0	0	0	0	0	0	1
Culicoides sp.	0	0	0	0	1	0	0	0	1	0
Glyptotendipes lobiferus	0	0	0	51	3	1	24	4	0	0
Ischnura sp.	0	0	0	0	0	2	0	0	0	0
Paratendipes sp.	0	4	1	2	0	8	3	4	0	0
Plea striolata	0	0	0	0	0	0	0	0	1	0
Polypedilius fallax	0	7	5	16	2	.52	3	2	0	19
Procladius sublettei	2	1	3	8	1	1	0	1	0	0
Tetragoneuria cynosura	0	1	0	0	1	1	0	0	0	0
Tricoptera sp.	0	0	0	0	0	0	1	0	0	0
Ceraclea sp.	0	1	0	0	0	0	0	0	0	0
GASTROPODA										
Birgella subglobosa	11	3	0	0	0	0	0	0	0	0
Ferrissia walkeri	1	0	0	0	0	0	0	0	0	0
Gyralus deflatus	0	0	0	3	2	0	0	0	1	2
Physella gyrina	0	0	6	2	18	12	0	4	19	7
Planorbella armigera	2	1	0	0	0	0	0	0	0	0
Valvata tricarinata	0	1	0	0	0	0	0	0	0	0
HIRUDINEA										
Erpobdella punctata	0	1	0	0	0	0	0	0	0	0
Helobdella papillata	4	1	0	0	0	0	0	1	2	0
OLIGOCHAETA										
Amphichaeta americanus	14	12	13	6	6	0	4	0	0	0
Aulodrilus pluriseta	89	13	0	1	4	4	3	5	0	4
Lumberculus variegatus	0	1	0	0	0	0	0	0	0	0
INDIVIDUALS (N)	137	101	82	155	53	134	117	52	51	84
SPECIES (S)	11	15	9	11	13	10	8	10	8	8

Figure I2. Sample Locations from Previous Studies at Round Lake

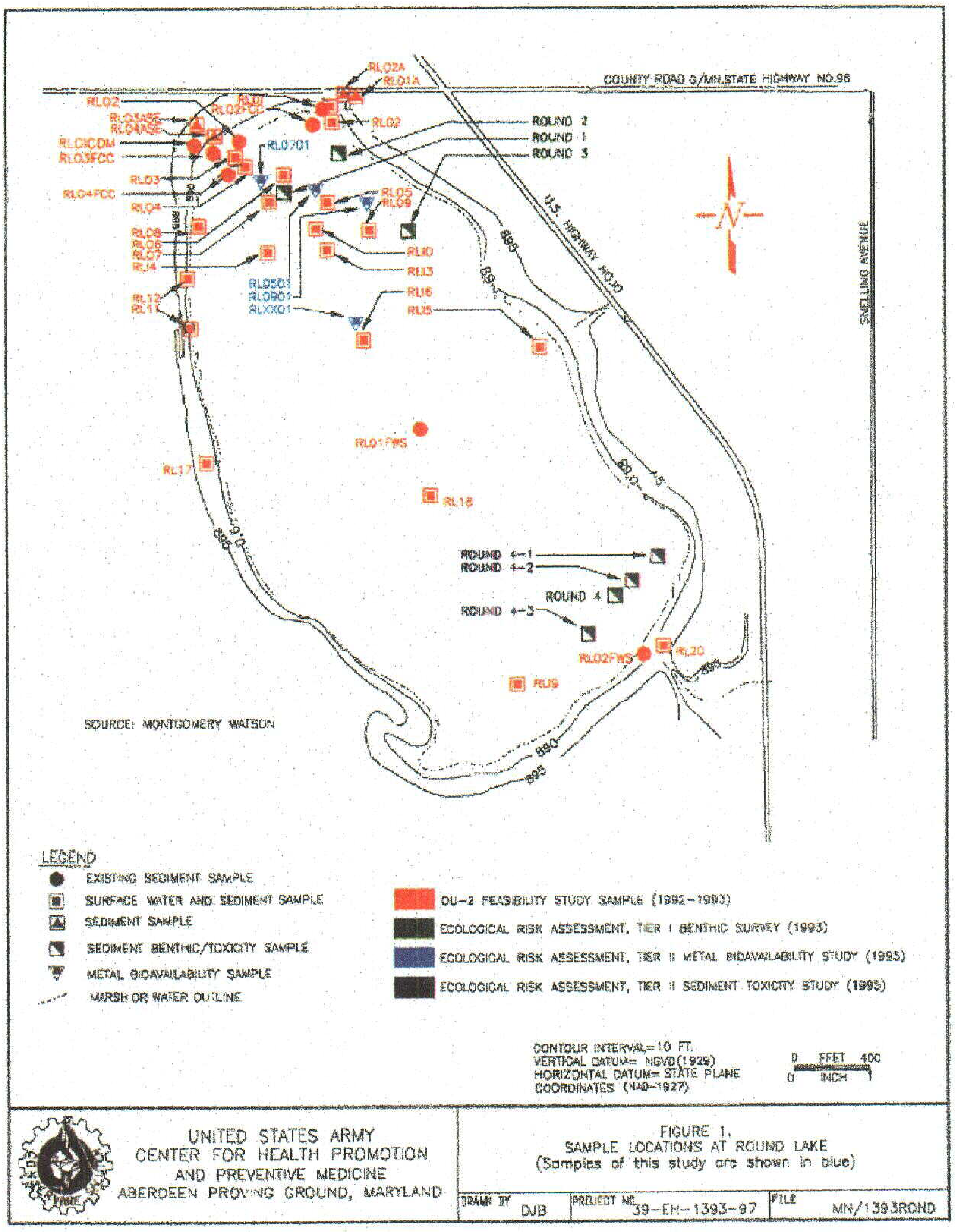
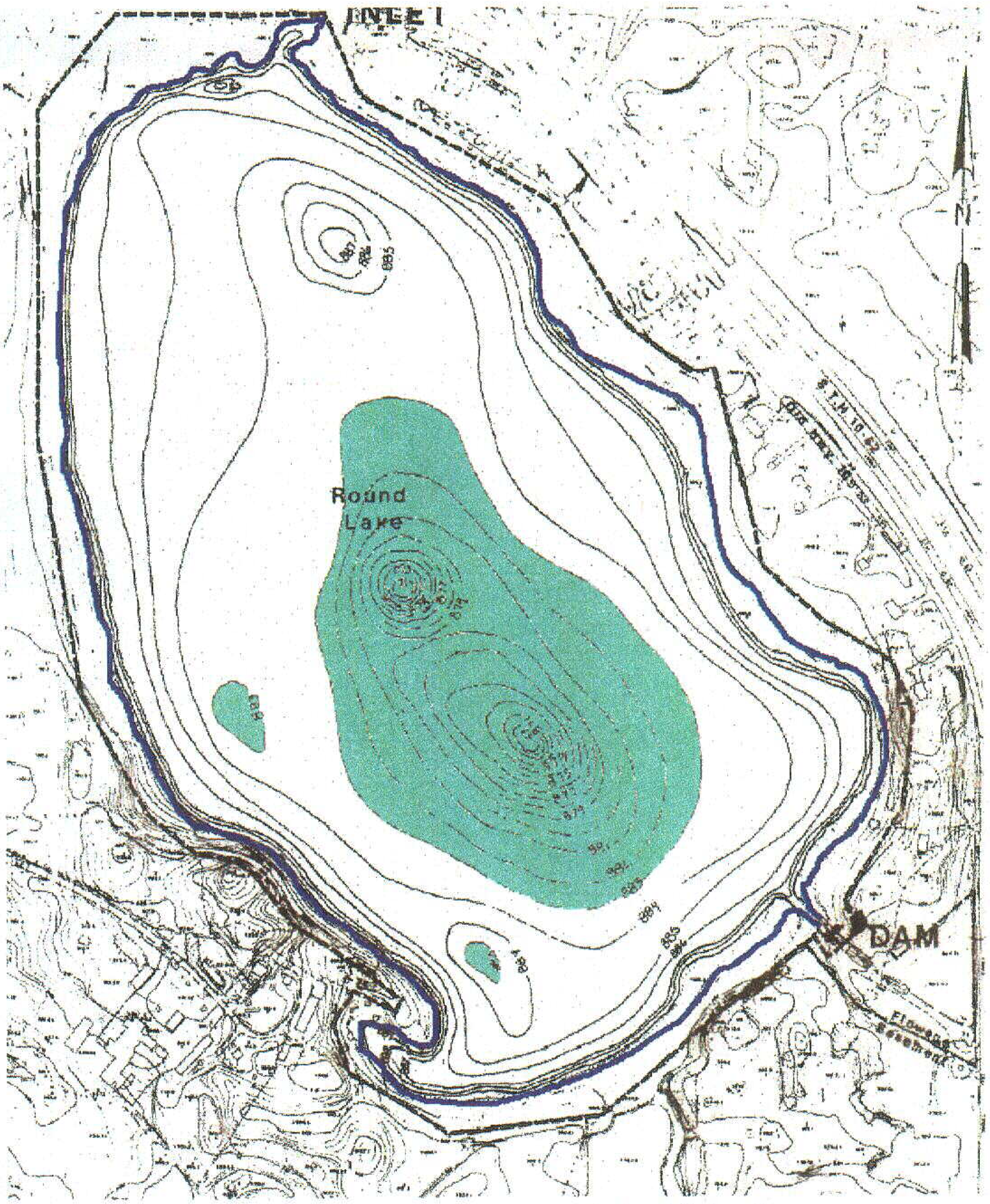


Figure I3. Round Lake Water Level as a Result of Fish and Wildlife Management Plan*



Light blue indicates Minimum Pool Elevation of 883 msl after full drawdown

*Source: Schreiner, Terry. Letter to Matt McAtee. 12 January 2000.

Table I13. Aluminum Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Aluminum Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	9420.0	7220.0	1910.0	4040.0
RL02	5820.0	4160.0	-	-	6460.0	7340.0
RL03	-	16800.0	7330.0	-	1660.0	1550.0
RL04	-	1950.0	6320.0	-	8710.0	6840.0
RL05	8850.0	9280.0	-	-	-	-
RL06	13100.0	9080.0	-	-	-	-
RL07	11500.0	14200	14200.0	-	2810.0	-
RL08	0.01	-	-	-	-	-
RL09	12000.0	8400.0	-	-	-	-
RL10	7350.0	15800.0	-	-	-	-
RL11	6920.0	7430.0	-	-	-	-
RL12	10500.0	6910.0	-	-	-	-
RL13	10000.0	7740.0	-	-	-	-
RL14	15400.0	5770.0	-	-	-	-
RL15	1260.0	-	-	-	-	-
RL16	7010.0	-	-	-	-	-
RL17	5290.0	-	-	-	-	-
RL17 DUP	8300.0	-	-	-	-	-
RL18	12900.0	-	-	-	-	-
RL19	12600.0	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	1850.0	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I14. Antimony Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Antimony Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<82.9	<82.9	<82.9	<82.9
RL02	<82.9	<82.9	-	-	<82.9	<82.9
RL03	-	<82.9	<82.9	-	<82.9	<82.9
RL04	-	<82.9	<82.9	-	<82.9	<82.9
RL05	<82.9	<82.9	-	-	-	-
RL06	<82.9	<82.9	-	-	-	-
RL07	<82.9	<82.9	-	-	-	-
RL08	<82.9	<82.9	-	-	-	-
RL09	<82.9	<82.9	-	-	-	-
RL10	<82.9	<82.9	-	-	-	-
RL11	<82.9	<82.9	-	-	-	-
RL12	<82.9	<82.9	-	-	-	-
RL13	<82.9	<82.9	-	-	-	-
RL14	<82.9	<82.9	-	-	-	-
RL15	<82.9	-	-	-	-	-
RL16	<82.9	-	-	-	-	-
RL17	<82.9	-	-	-	-	-
RL17 DUP	<82.9	-	-	-	-	-
RL18	<82.9	-	-	-	-	-
RL19	<82.9	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	<82.9	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table A15. Arsenic Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Arsenic Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<12.7	<12.7	<12.7	<12.7
RL02	<12.7	<12.7	-	-	<12.7	<12.7
RL03	-	<12.7	<12.7	-	<12.7	<12.7
RL04	-	<12.7	<12.7	<12.7	<12.7	<12.7
RL05	<12.7	<12.7	-	-	-	-
RL06	<12.7	<12.7	-	-	-	-
RL07	<12.7	<12.7	-	-	-	-
RL08	<12.7	<12.7	-	-	-	-
RL09	<12.7	<12.7	-	-	-	-
RL10	<12.7	<12.7	-	-	-	-
RL11	<12.7	<12.7	-	-	-	-
RL12	<12.7	<12.7	-	-	-	-
RL13	<12.7	<12.7	-	-	-	-
RL14	<12.7	<12.7	-	-	-	-
RL15	<12.7	-	-	-	-	-
RL16	<12.7	-	-	-	-	-
RL17	<12.7	-	-	-	-	-
RL17 DUP	<12.7	-	-	-	-	-
RL18	<12.7	-	-	-	-	-
RL19	<12.7	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	<12.7	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table A16. Barium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Barium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	179.0	154	24.5	58.5
RL02	154.0	127.0	-	-	126.0	132.0
RL03	-	376.0	161	-	24.4	26.1
RL04	-	38.0	148.0	-	175.0	134.0
RL05	121.0	153.0	-	-	-	-
RL06	196.0	155.0	-	-	-	-
RL07	182.0	246.0	-	-	-	-
RL08	51.6	163.0	-	-	-	-
RL09	213.0	161.0	-	-	-	-
RL10	157.0	231.0	-	-	-	-
RL11	138.0	150.0	-	-	-	-
RL12	181.0	148.0	-	-	-	-
RL13	170.0	167.0	-	-	-	-
RL14	214.0	145.0	-	-	-	-
RL15	11.6	-	-	-	-	-
RL16	138.0	-	-	-	-	-
RL17	120.0	-	-	-	-	-
RL17 DUP	171.0	-	-	-	-	-
RL18	177.0	-	-	-	-	-
RL19	156.0	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	15.2	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I17. Beryllium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Beryllium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<0.25	<0.25	<0.25	<0.25
RL02	<0.25	<0.25	-	-	<0.25	<0.25
RL03	-	<0.25	<0.25	-	<0.25	<0.25
RL04	-	<0.25	<0.25	<0.25	<0.25	<0.25
RL05	<0.25	<0.25	-	-	-	-
RL06	<0.25	<0.25	-	-	-	-
RL07	<0.25	<0.25	-	-	-	-
RL08	<0.25	<0.25	-	-	-	-
RL09	<0.25	<0.25	-	-	-	-
RL10	<0.25	<0.25	-	-	-	-
RL11	<0.25	<0.25	-	-	-	-
RL12	<0.25	<0.25	-	-	-	-
RL13	<0.25	<0.25	-	-	-	-
RL14	<0.25	<0.25	-	-	-	-
RL15	<0.25	-	-	-	-	-
RL16	<0.25	-	-	-	-	-
RL17	<0.25	-	-	-	-	-
RL17 DUP	<0.25	-	-	-	-	-
RL18	<0.25	-	-	-	-	-
RL19	<0.25	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	<0.25	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I18. Cadmium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Cadmium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<0.427	<0.427	<0.427	<0.427
RL02	<0.427	<0.427	-	-	3.11	<0.427
RL03	-	<0.427	3.6	-	<0.427	<0.427
RL04	-	0.854	<0.427	-	<0.427	<0.427
RL05	2.73	<0.427	-	-	-	-
RL06	<0.427	<0.427	-	-	-	-
RL07	3.81	11.3	-	-	-	-
RL08	<0.427	<0.427	-	-	-	-
RL09	7.53	<0.427	-	-	-	-
RL10	<0.427	5.75	-	-	-	-
RL11	<0.427	<0.427	-	-	-	-
RL12	<0.427	<0.427	-	-	-	-
RL13	<0.427	<0.427	-	-	-	-
RL14	5.080	<0.427	-	-	-	-
RL15	<0.427	-	-	-	-	-
RL16	<0.427	-	-	-	-	-
RL17	<0.427	-	-	-	-	-
RL17 DUP	<0.427	-	-	-	-	-
RL18	11.2	-	-	-	-	-
RL19	<0.427	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	<0.427	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I19. Calcium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Calcium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	10000.0	8810.0	4500.0	3040.0
RL02	8880.0	7960.0	-	-	23000.0	18000.0
RL03	-	22700.0	11100.0	-	2470.0	2020.0
RL04	-	2870.0	8900.0	-	8270.0	12700.0
RL05	9400.0	24000.0	-	-	-	-
RL06	23300.0	10900.0	-	-	-	-
RL07	22600.0	12400.0	-	-	-	-
RL08	5460.0	12500.0	-	-	-	-
RL09	56000.0	13600.0	-	-	-	-
RL10	9060.0	46000.0	-	-	-	-
RL11	13000.0	14400.0	-	-	-	-
RL12	11700.0	11100.0	-	-	-	-
RL13	53000.0	18400.0	-	-	-	-
RL14	18500.0	12400.0	-	-	-	-
RL15	1810.0	-	-	-	-	-
RL16	15200.0	-	-	-	-	-
RL17	12300.0	-	-	-	-	-
RL17 DUP	16700.0	-	-	-	-	-
RL18	15900.0	-	-	-	-	-
RL19	16400.0	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	3680.0	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I20. Chromium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Chromium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	25.4	17.0	7.16	13.1
RL02	21.0	13.1	-	-	21.5	23.2
RL03	-	54.1	44.2	-	6.36	5.95
RL04	-	22.1	16.9	-	22.6	17.8
RL05	56.8	84.1	-	-	-	-
RL06	64.9	32.6	-	-	-	-
RL07	96.5	35.9	-	-	-	-
RL08	20.9	18.8	-	-	-	-
RL09	110.0	20.0	-	-	-	-
RL10	15.2	89.6	-	-	-	-
RL11	68.2	17.4	-	-	-	-
RL12	72.7	19.0	-	-	-	-
RL13	39.6	17.1	-	-	-	-
RL14	55.2	14.6	-	-	-	-
RL15	3.96	-	-	-	-	-
RL16	<0.974	-	-	-	-	-
RL17	11.8	-	-	-	-	-
RL17 DUP	19.7	-	-	-	-	-
RL18	172.0	-	-	-	-	-
RL19	142.0	-	-	-	-	-
RL19 DUP	NA	-	-	-	-	-
RL20	4.23	-	-	-	-	-

The “—“ denotes that no data exists for that depth. The “R” denotes laboratory-rejected data.

Table I21. Cobalt Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Cobalt Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	21.3	19.7	<2.5	5.27
RL02	18	16.7	-	-	9.54	11.4
RL03	-	41.3	15.6	-	<2.5	<2.5
RL04	-	3.87	19.3	-	11.4	8.55
RL05	13.3	15.1	-	-	-	-
RL06	19.7	16.4	-	-	-	-
RL07	<2.5	<2.5	-	-	-	-
RL08	<2.5	<2.5	-	-	-	-
RL09	19.1	15.4	-	-	-	-
RL10	<2.5	22.0	-	-	-	-
RL11	<2.5	<2.5	-	-	-	-
RL12	<2.5	<2.5	-	-	-	-
RL13	14.1	<2.5	-	-	-	-
RL14	19.5	<2.5	-	-	-	-
RL15	<2.5	-	-	-	-	-
RL16	<2.5	-	-	-	-	-
RL17	<2.5	-	-	-	-	-
RL17 DUP	<2.5	-	-	-	-	-
RL18	<2.5	-	-	-	-	-
RL19	<2.5	-	-	-	-	-
RL20	<2.5	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I22. Copper Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	44.5	<3.38	8.9	19.8
RL02	43.2	18.3	-	-	40.4	36.2
RL03	-	162.0	398.0	-	<3.38	<3.38
RL04	-	185	22.6	-	18.1	14.9
RL05	521	1140	-	-	-	-
RL06	608	209	-	-	-	-
RL07	905	37.2	-	-	-	-
RL08	139	<3.38	-	-	-	-
RL09	1250	29.7	-	-	-	-
RL10	20.9	937	-	-	-	-
RL11	350	<3.38	-	-	-	-
RL12	399	<3.38	-	-	-	-
RL13	196	<3.38	-	-	-	-
RL14	168	<3.38				
RL15	<3.38	-	-	-	-	-
RL16	<3.38	-	-	-	-	-
RL17	<3.38	-	-	-	-	-
RL17 DUP	51	-	-	-	-	-
RL18	365	-	-	-	-	-
RL19	320	-	-	-	-	-
RL20	<3.38	-	-	-	-	-

The "--" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I23. Cyanide Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<1.22	<1.22	<1.22	<1.22
RL02	<1.22	<1.22	-	-	<1.22	<1.22
RL03	-	<1.22	<1.22	-	<1.22	<1.22
RL04	-	<1.22	<1.22	-	<1.22	<1.22
RL05	<1.22	<1.22	-	-	-	-
RL06	<1.22	<1.22	-	-	-	-
RL07	<1.22	<1.22	-	-	-	-
RL08	<1.22	<1.22	-	-	-	-
RL09	<1.22	<1.22	-	-	-	-
RL10	<1.22	<1.22	-	-	-	-
RL11	<1.22	<1.22	-	-	-	-
RL12	<1.22	<1.22	-	-	-	-
RL13	<1.22	<1.22	-	-	-	-
RL14	<1.22	<1.22	-	-	-	-
RL15	<1.22	-	-	-	-	-
RL16	<1.22	-	-	-	-	-
RL17	<1.22	-	-	-	-	-
RL17 DUP	<1.22	-	-	-	-	-
RL18	<1.22	-	-	-	-	-
RL19	<1.22	-	-	-	-	-
RL20	<1.22	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I24. Iron Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	19000	16000	2900	4600
RL02	13000	9700	-	-	11000	13000
RL03	-	31000	12000	-	2800	2700
RL04	-	3300	12000	-	14000	13000
RL05	16000	19000	-	-	-	-
RL06	23000	15000	-	-	-	-
RL07	19000	23000	-	-	-	-
RL08	4500	13000	-	-	-	-
RL09	26000	16000	-	-	-	-
RL10	19000	28000	-	-	-	-
RL11	12000	14000	-	-	-	-
RL12	16000	13000	-	-	-	-
RL13	16000	17000	-	-	-	-
RL14	23000	13000	-	-	-	-
RL15	2600	-	-	-	-	-
RL16	11300	-	-	-	-	-
RL17	10000	-	-	-	-	-
RL17 DUP	14000	-	-	-	-	-
RL18	20000	-	-	-	-	-
RL19	18200	-	-	-	-	-
RL19 DUP	Na	Na	Na	Na	Na	Na
RL20	3000	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I25. Lead Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<10	<10	<10	<10
RL02	<10	<10	-	-	45.4	45.1
RL03	<10	<10	-	-	<10	<10
RL04	-	13.8	<10	-	<10	<10
RL05	89.7	31.4	-	-	-	-
RL06	<10	<10	-	-	-	-
RL07	<10	<10	-	-	-	-
RL08	<10	<10	-	-	-	-
RL09	113	<10	-	-	-	-
RL10	<10	67.6	-	-	-	-
RL11	<10	<10	-	-	-	-
RL12	<10	<10	-	-	-	-
RL13	<10	<10	-	-	-	-
RL14	<10	<10	-	-	-	-
RL15	<10	-	-	-	-	-
RL16	<10	-	-	-	-	-
RL17	<10	-	-	-	-	-
RL17 DUP	<10	-	-	-	-	-
RL18	299	-	-	-	-	-
RL19	257	-	-	-	-	-
RL19 DUP	Na	Na	Na	Na	Na	Na
RL20	<10	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I26. Magnesium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	4120	3150	2720	2060
RL02	2880	2340	-	-	7600	6670
RL03	-	8380	4820	-	1270	1090
RL04	-	1120	3250	-	3470	5190
RL05	8280	9450	-	-	-	-
RL06	8470	4250	-	-	-	-
RL07	6200	5310	-	-	-	-
RL08	1340	3580	-	-	-	-
RL09	9990	6260	-	-	-	-
RL10	3650	11900	-	-	-	-
RL11	3440	4030	-	-	-	-
RL12	4340	3520	-	-	-	-
RL13	7270	4410	-	-	-	-
RL14	5790	2910	-	-	-	-
RL15	906	-	-	-	-	-
RL16	4260	-	-	-	-	-
RL17	3370	-	-	-	-	-
RL17 DUP	4360	-	-	-	-	-
RL18	6100	-	-	-	-	-
RL19	5300	-	-	-	-	-
RL20	786	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I27. Manganese Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	295	344	71.4	75
RL02	295	247	-	-	285	343
RL03	-	1040	315	-	65.3	60.9
RL04	-	83.4	371.0	-	298	370
RL05	367	710	-	-	-	-
RL06	561	269	-	-	-	-
RL07	319	519	-	-	-	-
RL08	90.5	440	-	-	-	-
RL09	750	479	-	-	-	-
RL10	335	534	-	-	-	-
RL11	325	394	-	-	-	-
RL12	435	392	-	-	-	-
RL13	351	368	-	-	-	-
RL14	301	347	-	-	-	-
RL15	62	-	-	-	-	-
RL16	372	-	-	-	-	-
RL17	303	-	-	-	-	-
RL17 DUP	517	-	-	-	-	-
RL18	525	-	-	-	-	-
RL19	377	-	-	-	-	-
RL20	51.5	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I28. Mercury Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<0.087	<0.087	<0.087	<0.087
RL02	<0.087	<0.087	-	-	<0.087	<0.087
RL03	-	<0.087	<0.087	-	<0.087	<0.087
RL04	-	<0.087	<0.087	-	<0.087	<0.087
RL05	<0.087	<0.087	-	-	-	-
RL06	<0.087	<0.087	-	-	-	-
RL07	<0.087	<0.087	-	-	-	-
RL08	<0.087	<0.087	-	-	-	-
RL09	<0.087	<0.087	-	-	-	-
RL10	<0.087	<0.087	-	-	-	-
RL11	<0.087	<0.087	-	-	-	-
RL12	<0.087	<0.087	-	-	-	-
RL13	<0.087	<0.087	-	-	-	-
RL14	<0.087	<0.087	-	-	-	-
RL15	<0.087	-	-	-	-	-
RL16	<0.087	-	-	-	-	-
RL17	<0.087	-	-	-	-	-
RL17 DUP	<0.087	-	-	-	-	-
RL18	<0.087	-	-	-	-	-
RL19	<0.087	-	-	-	-	-
RL20	<0.087	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I29. Molybdenum Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<4.0	<4.0	<4.0	<4.0
RL02	<4.0	<4.0	-	-	<4.0	<4.0
RL03	-	<4.0	<4.0	-	<4.0	<4.0
RL04	-	<4.0	<4.0	-	<4.0	<4.0
RL05	<4.0	<4.0	-	-	-	-
RL06	<4.0	<4.0	-	-	-	-
RL07	<4.0	<4.0	-	-	-	-
RL08	<4.0	<4.0	-	-	-	-
RL09	<4.0	<4.0	-	-	-	-
RL10	<4.0	<4.0	-	-	-	-
RL11	<4.0	<4.0	-	-	-	-
RL12	<4.0	<4.0	-	-	-	-
RL13	<4.0	<4.0	-	-	-	-
RL14	<4.0	<4.0	-	-	-	-
RL15	<4.0	-	-	-	-	-
RL16	<4.0	-	-	-	-	-
RL17	<4.0	-	-	-	-	-
RL17 DUP	<4.0	-	-	-	-	-
RL18	<4.0	-	-	-	-	-
RL19	<4.0	-	-	-	-	-
RL20	<4.0	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I30. Nickel Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<7.5	<7.5	<7.5	20.4
RL02	<7.5	<7.5	-	-	21	21.5
RL03	-	<7.5	<7.5	-	<7.5	<7.5
RL04	-	<7.5	<7.5	-	24.5	<7.5
RL05	<7.5	24.7	-	-	-	-
RL06	<7.5	<7.5	-	-	-	-
RL07	<7.5	<7.5	-	-	-	-
RL08	<7.5	<7.5	-	-	-	-
RL09	32.2	<7.5	-	-	-	-
RL10	<7.5	32.1	-	-	-	-
RL11	<7.5	<7.5	-	-	-	-
RL12	<7.5	<7.5	-	-	-	-
RL13	<7.5	<7.5	-	-	-	-
RL14	<7.5	<7.5	-	-	-	-
RL15	<7.5	-	-	-	-	-
RL16	<7.5	-	-	-	-	-
RL17	<7.5	-	-	-	-	-
RL17 DUP	<7.5	-	-	-	-	-
RL18	<7.5	-	-	-	-	-
RL19	<7.5	-	-	-	-	-
RL20	<7.5	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I31. Potassium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	1080	1010	242	393
RL02	859	<142	-	-	835	1050
RL03	-	<142	873	-	265	230
RL04	-	241	<142	-	1180	1000
RL05	1330	1270	-	-	-	-
RL06	1930	1290	-	-	-	-
RL07	1890	2480	-	-	-	-
RL08	445	<142	-	-	-	-
RL09	1520	1330	-	-	-	-
RL10	1170	2440	-	-	-	-
RL11	<142	<142	-	-	-	-
RL12	1750	<142	-	-	-	-
RL13	1720	<142	-	-	-	-
RL14	2610	<142	-	-	-	-
RL15	<142	-	-	-	-	-
RL16	<142	-	-	-	-	-
RL17	<142	-	-	-	-	-
RL17 DUP	1820	-	-	-	-	-
RL18	<142	-	-	-	-	-
RL19	<142	-	-	-	-	-
RL20	<142	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I32. Selenium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	-	-	<12.4	<12.4	<12.4	<12.4
RL02	<12.4	<12.4	-	-	<12.4	<12.4
RL03	-	<12.4	<12.4	-	<12.4	<12.4
RL04	-	<12.4	<12.4	-	<12.4	<12.4
RL05	<12.4	<12.4	-	-	-	-
RL06	<12.4	<12.4	-	-	-	-
RL07	<12.4	<12.4	-	-	-	-
RL08	<12.4	<12.4	-	-	-	-
RL09	<12.4	<12.4	-	-	-	-
RL10	<12.4	<12.4	-	-	-	-
RL11	<12.4	<12.4	-	-	-	-
RL12	<12.4	<12.4	-	-	-	-
RL13	<12.4	<12.4	-	-	-	-
RL14	<12.4	<12.4	-	-	-	-
RL15	<12.4	-	-	-	-	-
RL16	<12.4	-	-	-	-	-
RL17	<12.4	-	-	-	-	-
RL17 DUP	<12.4	-	-	-	-	-
RL18	<12.4	-	-	-	-	-
RL19	<12.4	-	-	-	-	-
RL20	<12.4	-	-	-	-	-

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I33. Silver Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Silver Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	—	—	< 0.5	< 0.5	< 0.5	< 0.5
RL02	< 0.5	< 0.5	—	—	< 0.5	< 0.5
RL03	—	< 0.5	< 0.5	—	< 0.5	< 0.5
RL04	—	< 0.5	< 0.5	—	< 0.5	< 0.5
RL05	< 0.5	< 0.5	—	—	—	—
RL06	2.45	< 0.5	—	—	—	—
RL07	86.8	< 0.5	—	—	—	—
RL08	< 0.5	< 0.5	—	—	—	—
RL09	1.07	< 0.5	—	—	—	—
RL10	58.5	< 0.5	—	—	—	—
RL11	< 0.5	38.0	—	—	—	—
RL12	< 0.5	< 0.5	—	—	—	—
RL13	< 0.5	< 0.5	—	—	—	—
RL14	< 0.5	< 0.5	—	—	—	—
RL15	< 0.5	—	—	—	—	—
RL16	< 0.5	—	—	—	—	—
RL17	< 0.5	—	—	—	—	—
RL17 DUP	< 0.5	—	—	—	—	—
RL18	< 0.5	—	—	—	—	—
RL19	< 0.5	—	—	—	—	—
RL20	< 0.5	—	—	—	—	—

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I34. Sodium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Sodium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	—	—	386	283	191	395
RL02	237	< 50	—	—	588	671
RL03	—	1540	510	—	266	272
RL04	—	109	585	—	621	562
RL05	302	222	—	—	—	—
RL06	589	505	—	—	—	—
RL07	522	767	—	—	—	—
RL08	165	874	—	—	—	—
RL09	300	265	—	—	—	—
RL10	361	493	—	—	—	—
RL11	628	813	—	—	—	—
RL12	706	993	—	—	—	—
RL13	445	859	—	—	—	—
RL14	555	652	—	—	—	—
RL15	< 50	—	—	—	—	—
RL16	1190	—	—	—	—	—
RL17	570	—	—	—	—	—
RL17 DUP	774	—	—	—	—	—
RL18	1320	—	—	—	—	—
RL19	1210	—	—	—	—	—
RL20	124	—	—	—	—	—

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I35. Thallium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Thallium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	—	—	< 12.5	< 12.5	< 12.5	< 12.5
RL02	< 12.5	< 12.5	—	—	< 12.5	< 12.5
RL03	—	< 12.5	< 12.5	—	< 12.5	< 12.5
RL04	—	< 12.5	< 12.5	—	< 12.5	< 12.5
RL05	< 12.5	< 12.5	—	—	—	—
RL06	< 12.5	< 12.5	—	—	—	—
RL07	< 12.5	< 12.5	—	—	—	—
RL08	< 12.5	< 12.5	—	—	—	—
RL09	< 12.5	< 12.5	—	—	—	—
RL10	< 12.5	< 12.5	—	—	—	—
RL11	< 12.5	< 12.5	—	—	—	—
RL12	< 12.5	< 12.5	—	—	—	—
RL13	< 12.5	< 12.5	—	—	—	—
RL14	< 12.5	< 12.5	—	—	—	—
RL15	< 12.5	—	—	—	—	—
RL16	< 12.5	—	—	—	—	—
RL17	< 12.5	—	—	—	—	—
RL17 DUP	< 12.5	—	—	—	—	—
RL18	< 12.5	—	—	—	—	—
RL19	< 12.5	—	—	—	—	—
RL20	< 12.5	—	—	—	—	—

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I36. Vanadium Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Vanadium Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	—	—	39.4	34.9	10.7	15.6
RL02	30.3	23.0	—	—	27.6	28.6
RL03	—	81.7	31.8	—	7.81	8.64
RL04	—	8.4	33.9	—	35.2	37.4
RL05	37.6	36.4	—	—	—	—
RL06	49.9	35.3	—	—	—	—
RL07	46.3	54.7	—	—	—	—
RL08	11.6	32.5	—	—	—	—
RL09	37.8	36.3	—	—	—	—
RL10	36.1	61.1	—	—	—	—
RL11	28.3	46.4	—	—	—	—
RL12	46.0	37.7	—	—	—	—
RL13	39.5	44.6	—	—	—	—
RL14	63.0	35.5	—	—	—	—
RL15	5.02	—	—	—	—	—
RL16	40.6	—	—	—	—	—
RL17	31.0	—	—	—	—	—
RL17 DUP	46.7	—	—	—	—	—
RL18	< 2.0	—	—	—	—	—
RL19	< 2.0	—	—	—	—	—
RL20	7.04	—	—	—	—	—

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I37. Zinc Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Zinc Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	—	—	99.0	67.7	32.3	30.3
RL02	82.5	53.0	—	—	183.0	186.0
RL03	—	245.0	28.1	—	16.3	16.8
RL04	—	106.0	62.0	—	75.2	74.1
RL05	414.0	572.0	—	—	—	—
RL06	399.0	183.0	—	—	—	—
RL07	639.0	144.0	—	—	—	—
RL08	125.0	99.6	—	—	—	—
RL09	892.0	83.2	—	—	—	—
RL10	90.0	626.0	—	—	—	—
RL11	341.0	104.0	—	—	—	—
RL12	453.0	84.3	—	—	—	—
RL13	263.0	148.0	—	—	—	—
RL14	270.0	62.1	—	—	—	—
RL15	10.3	—	—	—	—	—
RL16	122.0	—	—	—	—	—
RL17	78.8	—	—	—	—	—
RL17 DUP	98.3	—	—	—	—	—
RL18	860.0	—	—	—	—	—
RL19	772.0	—	—	—	—	—
RL20	21.4	—	—	—	—	—

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.

Table I38. Heptachlor Epoxide and Endosulfan Concentrations in Measured Sediment Depth Intervals of Round Lake

Sample Location	Pesticide Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	—	—	< 0.004 < 0.005	< 0.004 < 0.005	< 0.004 < 0.005	< 0.004 < 0.005
RL02	< 0.004 < 0.005	< 0.004 < 0.005	—	—	< 0.004 **0.038**	< 0.004 < 0.005
RL03	—	< 0.004 < 0.005	< 0.004 < 0.005	—	< 0.004 < 0.005	< 0.004 < 0.005
RL04	—	< 0.004 < 0.005	< 0.004 < 0.005	—	< 0.004 < 0.005	< 0.004 < 0.005
RL05	**0.016** < 0.005	< 0.004 < 0.005	—	—	—	—
RL06	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL07	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL08	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL09	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL10	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL11	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL12	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL13	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL14	< 0.004 < 0.005	< 0.004 < 0.005	—	—	—	—
RL15	< 0.004 < 0.005	—	—	—	—	—
RL16	< 0.004 < 0.005	—	—	—	—	—
RL17	< 0.004 < 0.005	—	—	—	—	—
RL17 DUP	< 0.004 < 0.005	—	—	—	—	—
RL18	< 0.004 < 0.005	—	—	—	—	—
RL19	< 0.004 < 0.005	—	—	—	—	—
RL20	< 0.004 < 0.005	—	—	—	—	—

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.
The top value is Heptachlor epoxide and the bottom value is Endosulfan, as alpha-Endosulfin / Endosulfin I.

Table I39. Concentrations of DDT & Metabolites in Measured Sediment Depth Intervals of Round Lake

Sample Location	DDT & Metabolite Concentrations (mg/kg) and Sediment Depth Intervals (ft)					
	0-1	1-2	2-3	3-4	4-5	5-6
RL01	—	—	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010
RL02	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	**0.043** **0.028** < 0.010	< 0.011 < 0.014 < 0.010
RL03	—	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010
RL04	—	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010
RL05	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL06	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL07	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL08	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL09	**0.403** **0.152** **0.328**	< 0.011 < 0.014 < 0.010	—	—	—	—
RL10	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL11	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL12	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL13	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL14	< 0.011 < 0.014 < 0.010	< 0.011 < 0.014 < 0.010	—	—	—	—
RL15	< 0.011 < 0.014 < 0.010	—	—	—	—	—
RL16	< 0.011 < 0.014 < 0.010	—	—	—	—	—
RL17	< 0.011 < 0.014 < 0.010	—	—	—	—	—
RL17 DUP	< 0.011 < 0.014 < 0.010	—	—	—	—	—
RL18	< 0.011 < 0.014 < 0.010	—	—	—	—	—
RL19	< 0.011 < 0.014 < 0.010	—	—	—	—	—
RL20	< 0.011 < 0.014 < 0.010	—	—	—	—	—

The "—" denotes that no data exists for that depth. The "R" denotes laboratory-rejected data.
Values from top to bottom are: DDT, DDE, and DDD.

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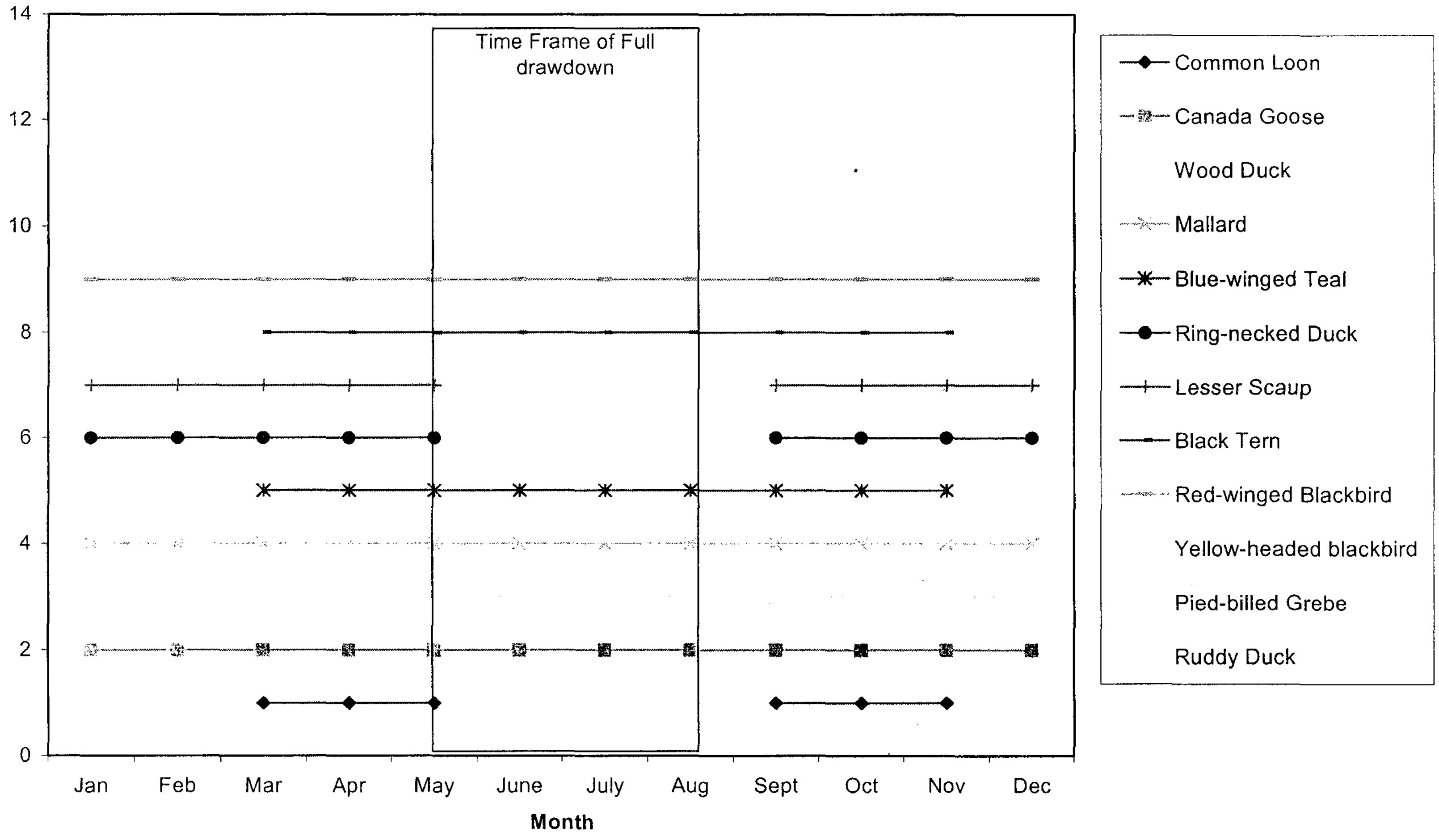
Figure I4. Avians Currently Present at Round Lake (Minnesota Valley National Wildlife Refuge)



United States Department of the Interior

FISH AND WILDLIFE SERVICE
Minnesota Valley National Wildlife Refuge
3815 East 80th Street
Bloomington, Minnesota 55425-1600

January 12, 2000

Mr. Matthew J. McAtee
U.S. Army Center For Health Promotion and Preventive Medicine
ATTN: MSHB-DC-EHR (40)
5158 Blackhawk Road
Aberdeen Proving Ground, Maryland 21010-5422

Dear Matthew:

Please find the water management plan for Round Lake enclosed. Implementation of the plan requires a water control structure capable of maintaining water levels from 883 msl to 889 msl. The current structure in place now has a control range of 887msl to 889 msl. A new structure would have to be installed to reach our ultimate management goals. mean sea level

Please contact me at 612-858-0705, if you have any questions about the plan. Contact Dave Warburton at the Twin Cities Ecological Services Office (612-725-3548) to discuss the ecological risk associated with the proposal.

Sincerely,

Terry Schreiner

Terry Schreiner
Refuge Operations Specialist

cc: Dave Warburton
Twin Cities Ecological Services Office
4101 East 80th Street
Bloomington, Mn. 55425

WATER LEVEL MANAGEMENT PLAN

ROUND LAKE UNIT

Background

A general water management proposal was developed for the Round Lake Unit of the Refuge in 1982. Several alternatives were proposed. The preferred alternative is the Optimum Wildlife Alternative. As stated in the plan, this alternative would provide the most immediate benefits to wildlife. Since the vast majority of the land base within the unit is comprised of the lake itself, the management of the water levels and the vegetation is the primary goal. The lake has been held at normal pool for several decades resulting in a dramatic decrease in emergent vegetation and to a lesser degree submergent vegetation. This decrease has reduced the amount of habitat available to wildlife, especially waterfowl. The following is a detailed water management plan for the Round Lake.

Goal

To maximize the and diversity of vegetation types of emergent and submergent aquatic vegetation across the lake bottom to provide habitat for the widest array of wildlife possible. Ideally, a mosaic of submergent and emergent vegetation interspersed across the lake bottom is desired creating what is called a hemi-marsh conditions. This condition maximizes the amount of food and cover in close proximity to each other. It is desirable to maintain this condition for as long as possible.

Management Technique

The drawdown technique to be employed is a gradual full drawdown. The intent is to begin the drawdown at the beginning of the growing season reaching the full drawdown stage by the end of June. This allows aquatic plants to grow in concentric bands around the lake shore. This technique keeps the soil moist during the critical early growth stages and provides plant diversity by utilizing the phenology of the various plants. The following water management regime will be followed to the extent possible.

Maximum Pool Elevation - 889.0 msl
Normal Pool elevation - 888.5 msl
Minimum pool elevation - 883.0 msl

Year 1
Conduct a full drawdown [redacted] Lower water approximately 1 foot every 8-10 days until 883.5 msl is reached. Begin raising water level around August 15. Raise water 1 foot every 12- 14 days until reach normal pool at 888.5.

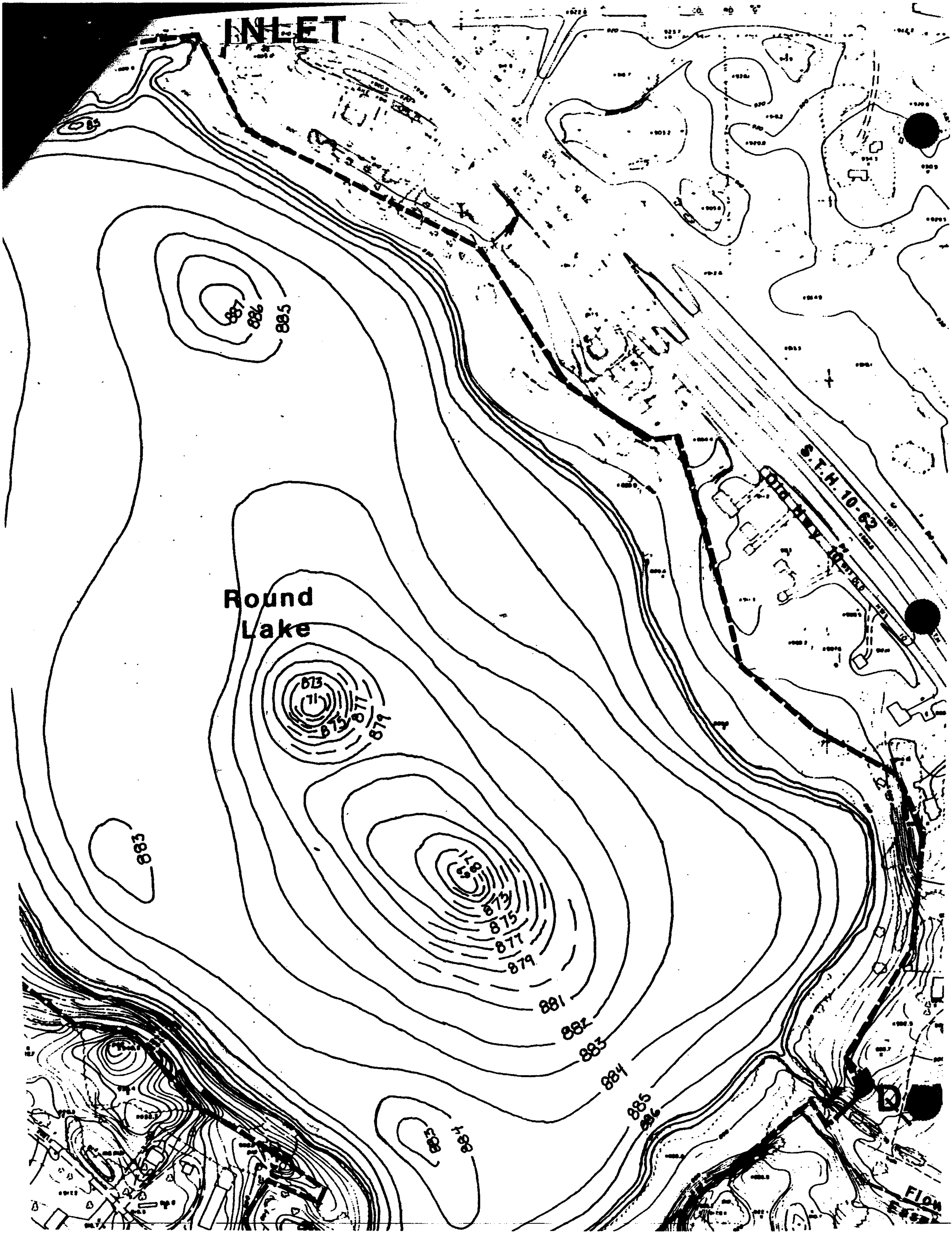
Year 2
Maintain normal pool elevation of 888.5 msl

Year 3
Maintain pool at normal elevation of 888.5 msl

Year 4
Conduct a partial drawdown. Begin drawdown around May 20. Lower water level approximately one foot every 8-10 days to the 886 msl. Raise water level .5 foot every 10 days beginning around July 15 until normal pool elevation of 88.5 msl is reached.

Year 5
Maintain normal pool elevation of 888.5 msl until Sept. 1.; then begin lowering water level 1 foot to 887.5 msl and hold. Raise to normal pool elevation of 888.5 msl beginning November 10.

Year 6
Maintain normal pool elevation of 888.5 msl until Sept. 1; then begin lowering water level 1 foot to 887.5 msl and hold. Raise to normal pool elevation of 888.5 msl beginning Nov. 10.


INLET
Round
Lake
887
886
885
883
873
871
875
877
879
881
882
883
884
885
886
883
884
S.T.H. 10-62
Old Hwy 10
D
Flow

MINNESOTA VALLEY NATIONAL WILDLIFE REFUGE

ROUND LAKE

UNIT MANAGEMENT PLAN UPDATE

1998

(Amendment to 1982 Original Plan)

Background

This document is intended to serve as an update to the original management plan for Round Lake Unit of Minnesota Valley National Wildlife Refuge managed by the U.S. Fish and wildlife Service (Service) (Citation). Although the Service recognizes that several conditions in the Round Lake area have changed since 1982, only Section II.F., Proposed Alternative and portions of Section III, Affected Area, of the original management plan area amended by this document.

Habitat - Present Condtions

As detailed in the section III, the Round Lake Unit is a relatively small area (152 acres) of which 120 acres consists of an permanently flooded palustrine emergent wetland. A narrow margin of upland surrounds the lake acting as a buffer between residential and light manufacturing development. This buffer is comprised of severely degraded oak savanna on the west and south shorelines and combination of early successional native and exotic grassland and upland forest, very minimal habitat management is planned.

The quantity and diversity of submergent and emergent aquatic vegetation as been decreasing in recent years due to stable high water. Natural changes in lake elevations through climatic variations have not occurred in recent years due to an extended climatic wet cycle, uncontrolled plugging of the water control structure by beaver, and increased runoff. This decline in habitat value has resulted in a steady decrease in wildlife utilization, especially waterfowl, water birds, and shorebirds. Providing habitat for these trust species is the primary objective of the Round Lake Unit. Minnesota Valley National Wildlife Refuge is part of a national system of refuges providing resting and feeding areas for these animals on all the major flyways in the nation. Therefore, to realize it's full potential for wildlife, Round Lake continues to be in dire need of lowered water levels.

The upland area consists of approximately 32 acres of degraded native plant communities which is arranged as a narrow band around Round Lake.

Preferred Management

The Service now recommends the implementation of the 1982 plan, Alternative Number 2, (Optimum Wildlife Alternative) as the preferred management alternative for Round Lake. As generally detailed in the plan, this alternative would provide the most benefits to wildlife in the shortest period of time with the least effort. Water level management capability is maximized (5.5 feet) between the 888.5 feet M.S.L. and 883.0 feet M.S.L. elevations with this alternative. Because water levels have been above normal for such a long period, a full draw down would provide the most benefit.. This approach would expose the largest area of lake bottom to sunlight and the air which would consolidate the sediments that have accumulated over the years. Once consolidated, emergent plants could germinate and take root. Water levels could then be brought up with the emergent growth to normal pool level. Increase submergent aquatic growth would be expected due to increased sunlight reaching bottom soils and consolidation stimulating and supporting root development. The normal pool level could then be maintained for several years until the emergents become sparse again. Normally, a full draw down is not necessary to stimulate emergent growth during the second cycle because enough remnant root system remains viable to spread over the area with water only lowered to point were enough sunlight reaches the bottom to stimulate the new growth; called a partial draw down. Therefore, a full draw down may be needed only once every 7-10 years and a partial draw down once every 3-5 years.

A less effective alternative would be Alternative Number 6. This approach employs a partial draw down which would increase the coverage of submergent vegetation, particularly where the bottom is the most consolidated. This alternative would not allow broad consolidation of bottom soils. Increases in the coverage of emergent vegetation would most likely be very small. The wetland conditions would improve over the present but only slightly. Under both alternatives, staff will be working closely with neighbors to ensure everyone understands what is being done and why.

Because upland area around Round Lake consists of a narrow band of severely degraded plant communities, very little management is planned. Larger tree limbs that fall on private property will be removed. Prescribed burning may be used, but probably only on rare occasions. No selective cutting of forest is planned.

Public Use

The Service intends to grant the City of Arden Hills a right-of-way 500 feet long and 15 feet wide along the west central portion of the lake. A 10 foot bituminous trail will be constructed in this right-of-way and serve as connector to city and regional trail systems. Once all connections are complete, interpretive signage will be designed and installed along the portion of trail along Round Lake. Public use of the Round Lake Unit will not be permitted by the Service except along this trail. Although the Service intends to continue to prohibit fishing in Round Lake, unauthorized fishing may be expected to occur.

Citations

A Proposal For Management of the Round Lake Unit, Minnesota Valley National Wildlife Refuge., August 1982.

Prepared by: Terry M Schreiner Date: 3/2/98
Refuge Operations Specialist

Reviewed by: Richard P. Schultz Date: 3/2/98
Refuge Manager

APPENDIX J

Snail Lake Sampling Locations and Analytical Measurement Results

Table of Contents

Snail Lake Data Presentation Summary

The map of Snail Lake (Fig.J1), shows the location where each media sample was taken. The map is an orthophoto taken in 1997, obtained from Wenck Associates Inc., Maple Plains, Minnesota. Sites that were sampled during water monitoring activities have green labels. Sites 1, 6, and 9 from the site characterization were the locations used during the quarterly water monitoring activities. Global Positioning System (GPS) coordinates were used to plot the locations. These coordinates recorded in the field, during sampling, are reported in tables J1 and J7. For mapping purposes of water monitoring data, second quarter sampling locations were not mapped by their GPS coordinates. (However, the following GPS data tables (J3 through J7) do show the recorded coordinates). Second quarter samples were mapped with first quarter coordinates. During second quarter sampling there was a problem with the crypto key, causing all samples to show imprecise coordinates. Based on landmarks, the field team leader noted in the field logbook (USACHPPM 2000) that sample locations for the second quarter were as close as possible to first quarter coordinates. Maps from the Work Plan (USACHPPM 1999) were also used to pinpoint the location to be sampled.

The tables J8 through J13 summarize the data for Snail Lake surface water and sediment samples taken during the site characterization. Tables J14 through J17 summarize the data collected during quarterly water monitoring activities. The summaries include the field and analytical result for each analyte at each sample location. The minima and maxima are reported as well as the detection frequency (DF), risk based action level (RBAL), target detection limit and achieved detection limit. The RBALs and target detection limits are those that were defined in the QAPP. Summary data used for statistical analysis such as the central tendencies, standard deviations and confidence intervals are reported in Appendix A.

The number of digits reported for each value in the following tables does not necessarily reflect the precision and accuracy of the measurement. The data tables present significant figures as they were originally reported by the analyzing laboratory. Significant figures reported for the analytes were based on the analytical detection limit values. However, some sample values have more significant figures than others to demonstrate that blank correction was discontinued, as was requested by USEPA comments on the draft version of this report (see Appendix R). Data quality is discussed in Appendix K. The results are reported in the following manner to facilitate data review.

- If the detection limit is sample-specific (as it was for the pesticides in sediment), the value reported for the achieved detection limit for a site is the highest detection limit for that parameter for all samples (including the field duplicate) from that site.

- If a parameter was not detected in any samples from a site (e.g. Pond G, water, alpha-BHC) then the minimum and maximum values represent the lowest and highest achieved detection limits for those data.

- Any concentrations that were below the detection limit but still quantified are identified in Volume 2 of this report as estimated values (J-flagged). These values are reported and used uncensored in the following tables.

- If a parameter was undetected in at least one but not all samples from a site, then the minimum reflects the lowest detection limit of the undetected samples. If there was a J-flagged value lower than the highest undetected value, the J-flagged value was reported as the maximum because the value was actually detected but estimated.

- Field duplicates (FD) were used as quality control samples and were not included in the reported minima and maxima columns. However, if an analyte was detected in the FD and not in it's corresponding sample, then the FD detection was counted in the detection frequency column (only in the numerator).

- When laboratory replicate samples (R1, R2) were available, the values from the primary sample (R1) were reported. See Appendix K (Section K1.2.1) for an explanation.

Definition of Data Qualifiers (Flags)

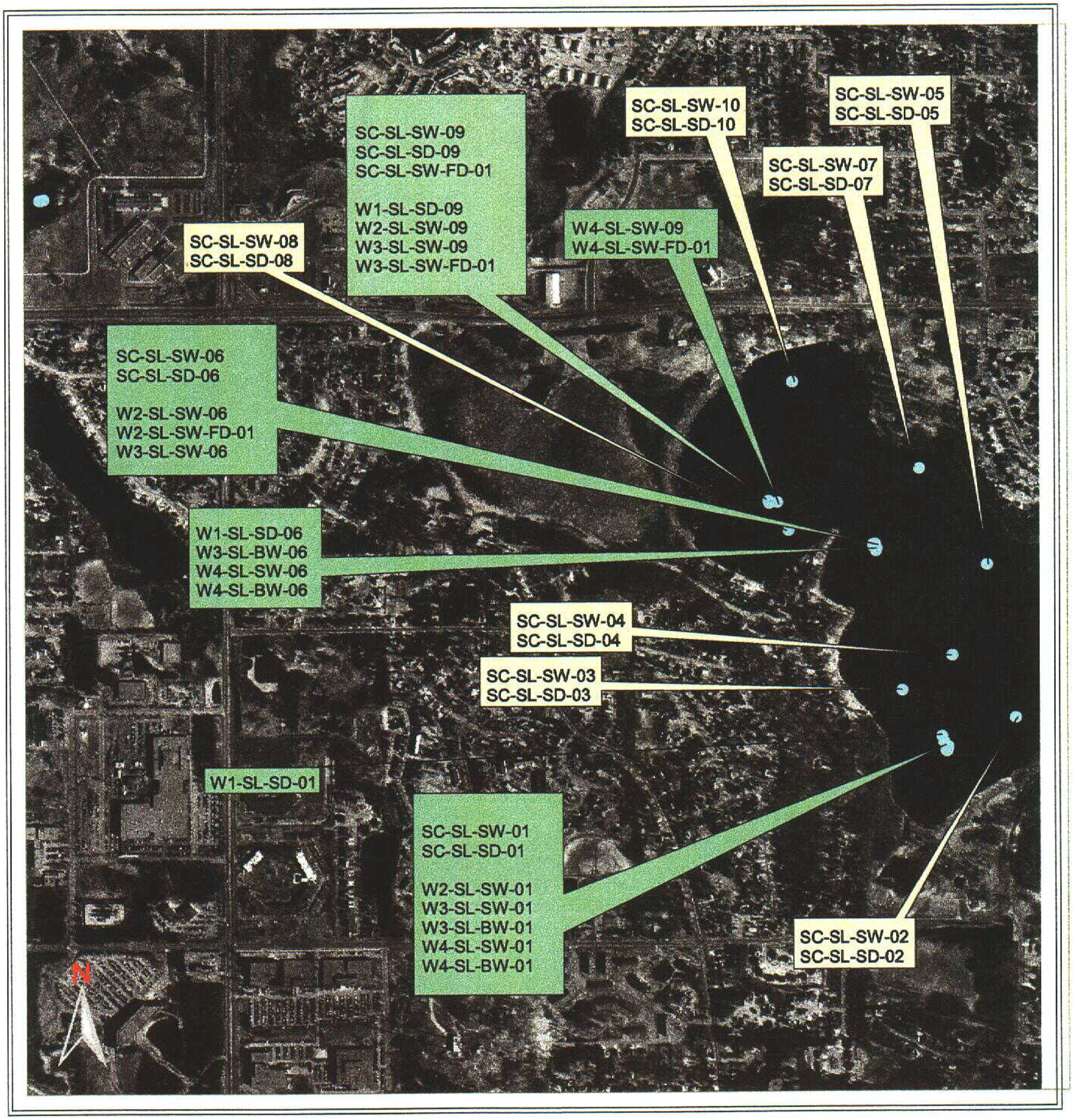
During the data validation process, all laboratory data were assigned appropriate data validation flags and reason codes. Additional information regarding data validation can be found in Volume 2 of the Tier II ERA Report (on CD ROM) of the report. Validation flags used in the following tables are defined as follows:

No flag	When the analyte value reported is detected and usable. The integrity of the analyte's identification, accuracy, and precision has been validated
"U"	When the material was analyzed for, but not detected above the level of the associated detection limit_value.
"UB"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with.
"J"	When the associated value is an estimated quantity. Indicating there is cause to question accuracy or precision of the reported value. The value is still used in the risk assessment.
"UJ"	When the analyte was analyzed for, but not detected, above the associated value, however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.

"UBJ"	When an analyte was detected in the method blank, then the analyte was considered non detected in the sample if the value was less than 5 times the highest method blank concentration associated with the particular batch the sample was analyzed with; however, the reported value is an estimate and demonstrates a decreased confidence in its accuracy or precision.

Note: Reference cited in the text of this appendix can be found in the reference section in the body of the Tier II Ecological Risk Assessment Report.

Figure J1. Snail Lake Surface Water and Sediment Sampling Locations



SC= Site Characterization
Wx= Water Monitoring
SL= Snail Lake
SW=Surface Water
SD=Sediment
BW=Bottom Water

0.1 0 0.1 0.2 Miles

Table J1. Snail Lake Site Characterization Surface Water Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-SL-SW-01	14-Sep-99	1'-22'	45	4	9.7	93	7	26.07
SC-SL-SW-01	15-Sep-99	4-6"	45	4	9.19	93	7	25.58
SC-SL-SW-02	15-Sep-99	4-6"	45	4	11.7	93	7	17.74
SC-SL-SW-03	15-Sep-99	4-6"	45	4	13.98	93	7	30.91
SC-SL-SW-04	15-Sep-99	Na	45	4	16.85	93	7	25.09
SC-SL-SW-05	17-Sep-99	5'	45	4	24.36	93	7	21.06
SC-SL-SW-06	16-Sep-99	22'	45	4	26.12	93	7	34.09
SC-SL-SW-07	17-Sep-99	6"	45	4	32.4	93	7	28.91
SC-SL-SW-08	17-Sep-99	6"	45	4	27.25	93	7	44.09
SC-SL-SW-09 SC-SL-SW-FD-01	16-Sep-99	6'	45	4	29.61	93	7	45.41
SC-SL-SW-10	17-Sep-99	6"	45	4	39.61	93	7	43.66

Table J2. Snail Lake Site Characterization Sediment Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-SL-SD-01	15-Sep-99	0-4"	45	4	9.19	93	7	25.58
SC-SL-SD-02	15-Sep-99	0-4"	45	4	11.7	93	7	17.74
SC-SL-SD-03	15-Sep-99	NA	45	4	13.98	93	7	30.91
SC-SL-SD-04	15-Sep-99	Na	45	4	16.85	93	7	25.09
SC-SL-SD-05	17-Sep-99	Na	45	4	24.36	93	7	21.06
SC-SL-SD-06	16-Sep-99	Na	45	4	26.12	93	7	34.09
SC-SL-SD-07	17-Sep-99	Na	45	4	32.4	93	7	28.91
SC-SL-SD-08	17-Sep-99	6'	45	4	27.25	93	7	44.09
SC-SL-SD-09	16-Sep-99	6'	45	4	29.61	93	7	45.41
SC-SL-SD-10	17-Sep-99	4'	45	4	39.61	93	7	43.66

Na: Not available

Table J3. Snail Lake First Quarter Water Monitoring Sample GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
SC-SL-SW-01	14-Sep-99	1'-22'	45	4	9.7	93	7	26.07
SC-SL-SW-01	15-Sep-99	4-6"	45	4	9.19	93	7	25.58
SC-SL-SW-06	16-Sep-99	22'	45	4	26.12	93	7	34.09
SC-SL-SW-09 SC-SL-SW-FD-01	16-Sep-99	6'	45	4	29.61	93	7	45.41

Table J4. Snail Lake Second Quarter Water Monitoring Sample GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W2-SL-SW-01	18-Jan-00	6"	45	4	7.84	93	7	26.92
W2-SL-SW-06	18-Jan-00	6"	45	4	23.69	93	7	32.13
W2-SL-SW-09 W2-SL-SW-FD-01	18-Jan-00	6"	45	4	30.1	93	7	44.24

Table J5. Snail Lake Third Quarter Water Monitoring Sample GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W3-SL-SW-01	25-Apr-00	10"	45	4	9.31	93	7	25.57
W3-SL-SW-06	25-Apr-00	Na	45	4	26.1	93	7	34.29
W3-SL-SW-09 W3-SL-SW-FD-01	25-Apr-00	Na	45	4	29.53	93	7	46.39

Bottom Water

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W3-SL-SW-BW-01	28-Apr-00	15'	45	4	8.92	93	7	25.73
W3-SL-SW-BW-06	28-Apr-00	15'	45	4	25.52	93	7	33.93
W3-SL-SW-BW-09	NA	NA	NA	NA	NA	NA	NA	NA

Table J6. Snail Lake Fourth Quarter Water Monitoring Sample GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W4-SL-SW-01	22-Jun-00	4-8"	45	4	9.05	93	7	25.74
W4-SL-SW-06	22-Jun-00	4-8"	45	4	25.71	93	7	33.82
W4-SL-SW-09 W4-SL-SW-FD-01	22-Jun-00	4-8"	45	4	29.74	93	7	46.51

Bottom Water

Sample Number	Date	Sample depth	N			W		
			degrees	minutes	seconds	degrees	minutes	seconds
W4-SL-SW-BW-01	22-Jun-00	18.1'	45	4	8.99	93	7	25.84
W4-SL-SW-BW-06	22-Jun-00	17.9'	45	4	25.71	93	7	33.82
W4-SL-SW-BW-09	NA	NA	NA	NA	NA	NA	NA	NA

Table J7. Snail Lake Sediment Bioassay Sampling GPS Coordinates

Sample Number	Date	Sample depth	N*			W*		
			degrees	minutes	seconds	degrees	minutes	seconds
W1-SL-SD-01	16-Sep-99	22'	45	4	10.17	93	7	26.25
W1-SL-SD-06	16-Sep-99	22'	45	4	25.86	93	7	34.25
W1-SL-SD-09	16-Sep-99	6'	45	4	29.61	93	7	45.41

* The GPS unit used in the study was a military-issue Precision Light Weight GPS Receiver (PLGR). The Department of Defense (DoD) intentionally places errors in the GPS navigation and timing signal, in a process called selective availability (Department of the Army 1993). Errors resulting from this process are unpredictable and can produce significant horizontal and elevation errors. Military GPS receivers must have crypto keys loaded to detect and nullify selective availability errors, in order to allow for more accurate position data (Department of the Army 1993). The PLGR was equipped with a crypto key, and was therefore not subject to the selective availability and spoofing errors that affect non-military GPS. (The only exception was during the second quarter). The PLGR operations manual (Department of the Army 1995) states that with crypto keys installed, the PLGR provides horizontal 2-D position accuracies better than the worst-case error of 10 meters.

Table J8. Snail Lake Surface Water Quality and Metals Data

Parameter	SC-SL-SW-01	SC-SL-SW-02	SC-SL-SW-03	SC-SL-SW-04	SC-SL-SW-05	SC-SL-SW-06	SC-SL-SW-07	SC-SL-SW-08	SC-SL-SW-09	SC-SL-SW-10	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-SL-FB-01	SC-SL-FD-XX
pH	8.57	8.58	8.69	8.73	8.65	8.65	8.74	8.56	8.49	8.94	8.49	8.94	10 / 10	na	na	na	na	na
DO (mg/L)	8.88	8.97	9.43	9.84	9.44	9.35	10.00	9.86	9.69	11.39	8.88	11.39	10 / 10	na	na	na	na	na
DO %	91.3	92.9	98.1	102.7	98.5	98.4	104.7	103.5	102	120.5	91.30	120.50	10 / 10	na	na	na	na	na
Conductivity (umhos/L)	299.00	300.00	298.00	298.00	304.00	299.00	301.00	300.00	299.00	301.00	298.00	304.00	10 / 10	na	na	na	na	na
Temperature (C)	16.64	17.05	17.23	no value	17.36	no value	17.64	17.67	no value	17.90	16.64	17.90	7 / 10	na	na	na	na	na
Hardness (mg/L)	102.00	102.00	100.00	104.00	104.00	98.00	100.00	102.00	96.00	104.00	96.00	104.00	10 / 10	na	na	na	na	102.00
TKN (mg/L)	0.81	0.82	0.92	0.82	0.79	0.81	0.71	0.76	0.77	0.74	0.71	0.92	10 / 10	na	na	na	na	0.72
TOC (mg/L)	8.00	12.00	8.90	8.10	11.00	7.80	8.80	8.60	9.00	8.00	7.80	12.00	10 / 10	na	na	na	na	8.30
Total Phosphorus (mg/L)	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05	0.05	10 / 10	na	na	na	na	0.05 U
TSS (mg/L)	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	<1.0	<1.0	10 / 10	na	na	na	na	1.00 U
Metals (ug/L)																		
Aluminum	6.25	6.85	3.35	6.95	6.35	6.25	5.50	6.85	6.05	6.35	3.35	6.95	10 / 10	125.00	0.20	0.0560	0.44 UB	6.15
Antimony	0.26	0.26	0.27	0.26	0.25	0.26	0.26	0.23	0.24	0.24	0.23	0.27	10 / 10	31.00	0.01	0.0089	0.01 U	0.24
Arsenic	5.6	5.6	5.9	5.5	5.8	5.7	5.6	5.6	5.6	5.4	5.40	5.90	10 / 10	190.00	0.60	0.0440	0.04 U	5.40
Barium	42 J	42 J	30 J	41 J	42 J	41 J	40 J	40 J	41 J	40 J	30.00	42.00	10 / 10	3.80	0.01	0.0056	0.85 J	40.00 J
Beryllium	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	0.048 U	<0.048	<0.048	0 / 10	5.10	0.10	0.0480	0.05 U	0.05 U
Cadmium	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	<0.015	<0.015	0 / 10	0.66	0.01	0.0150	0.02	0.02 U
Chromium	0.549	0.549	0.679	0.499	0.619	0.569	0.489	0.449	0.559	0.529	0.449	0.679	10 / 10	117.00	0.03	0.0420	0.108 UB	0.559
Cobalt	0.0500	0.0540	0.0530	0.0590	0.0560	0.0540	0.0530	0.0560	0.0570	0.0590	0.0500	0.0590	10 / 10	5.00	0.01	0.0038	0.0120 UB	0.0540
Copper	2.1	2.3	2.0	2.1	2.2	2.2	2.1	2.1	2.2	2.3	2.0	2.30	10 / 10	6.40	0.01	0.0140	0.50	2.10
Lead	0.086	0.082	0.051	0.088	0.092	0.086	0.087	0.074	0.086	0.071	0.05	0.09	10 / 10	1.30	0.01	0.0049	0.22	0.070
Magnesium	8800 J	9500 J	9800 J	9000 J	8900 J	9000 J	8800 J	8900 J	9000 J	8900 J	8800.00	9800.00	10 / 10	82000.00	25.00	0.0500	1.0 B	8800.00
Manganese	26	31	18	25	23	18	19	15	14	16	14.00	31.00	10 / 10	80.30	0.01	0.0100	0.05 B	13.00
Mercury	0.00061	0.00051	0.00058	0.00050	0.00069	0.00075	0.00051	0.00051	0.00048	0.00058	0.00	0.00	10 / 10	0.00	0.00	0.0002	0.00	0.00
Nickel	0.27	0.28	0.25	0.20	0.21	0.21	0.29	0.27	0.22	0.20	0.20	0.29	10 / 10	80.00	0.04	0.0320	0.51	0.25
Selenium	0.51 J	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	0.39 U	<0.39	0.51	9 / 10	5.00	0.30	0.3900	0.39 U	0.39 U
Silver	0.0202 UB	0.0105 UB	0.0242 UB	0.0242 UB	0.0105 UB	0.0157 UB	0.0117 UB	0.0141 UB	0.0162 UB	0.0152 UB	0.0105	0.0242	8 / 10	1.00	0.01	0.0043	0.0105 UB	0.0138 UB
Thallium	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	0.0035 U	<0.0035	<0.0035	0 / 10	18.00	0.00	0.0035	0.00 U	0.00 U
Zinc	0.67	0.79	0.55 UB	0.70	0.68	0.60	0.62	0.63	0.81	0.69	0.55	0.81	8 / 10	59.00	0.04	0.0350	10.110	0.650
																		Sample location SL-09

Table J9. Snail Lake Surface Water Pesticides and PCBs Data

Parameter (ug/L)	SC-SL-SW-01	SC-SL-SW-02	SC-SL-SW-03	SC-SL-SW-04	SC-SL-SW-05	SC-SL-SW-06	SC-SL-SW-07	SC-SL-SW-08	SC-SL-SW-09	SC-SL-SW-10	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-SL-FB-01	SC-SL-FD-XX
4,4'-DDD	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.01	0.05	0.01	0.01 R	0.01 U
4,4'-DDE	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.01	0.05	0.01	0.01 R	0.01 U
4,4'-DDT	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.00	0.01	0.01	0.01 R	0.01 U
Aldrin	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	none	0.05	0.01	0.01 R	0.01 U
Alpha-BHC	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	2.40	0.05	0.01	0.01 R	0.01 U
Beta-BHC	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	2.40	0.05	0.01	0.01 R	0.01 U
Chlordane (technical)	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	<0.05	<0.05	0 / 10	2.40	0.05	0.05	0.05 R	0.05 U
Delta-BHC	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	2.40	0.05	0.01	0.01 R	0.01 U
Dieldrin	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.00	0.05	0.01	0.01 R	0.01 U
Endosulfan I	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.03	0.05	0.01	0.01 R	0.01 U
Endosulfan II	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.03	0.10	0.01	0.01 R	0.01 U
Endosulfan Sulfate	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	none	0.10	0.01	0.01 R	0.01 U
Endrin	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.02	0.04	0.01	0.01 R	0.01 U
Endrin Aldehyde	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	none	0.10	0.01	0.01 R	0.01 U
Heptachlor	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.00	0.05	0.01	0.01 R	0.01 U
Heptachlor Epoxide	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.00	0.05	0.01	0.01 R	0.01 U
Lindane	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	0.08	0.05	0.01	0.01 R	0.01 U
Methoxychlor	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	0.01 U	<0.01	<0.01	0 / 10	none	0.40	0.01	0.01 R	0.01 U
Toxaphene	0.50 UJ	0.50 UJ	0.50 UJ	0.50 UJ	0.50 UJ	0.50 UJ	0.50 UJ	0.50 UJ	0.50 UJ	0.50 UJ	<0.5	<0.5	0 / 10	0.00	0.50	0.50	0.5 R	0.5 UJ
Aroclor-1016	0.10 U	0.10 U	0.10 U	0.10 U	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	<0.1	<0.1	0 / 10	none	0.50	0.10	0.1 R	0.1 UJ
Aroclor-1221	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 10	0.27	0.50	0.10	0.1 R	0.1 U
Aroclor-1232	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 10	0.50	0.50	0.10	0.1 R	0.1 U
Aroclor-1242	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 10	0.06	0.50	0.10	0.1 R	0.1 U
Aroclor-1248	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 10	0.01	0.50	0.10	0.1 R	0.1 U
Aroclor-1254	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	0.10 U	<0.1	<0.1	0 / 10	0.02	0.50	0.10	0.1 R	0.1 U
Aroclor-1260	0.10 U	0.10 U	0.10 U	0.10 U	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	0.10 UJ	<0.1	<0.1	0 / 10	1.30	0.50	0.10	0.10 R	0.10 UJ
																		Sample location SL-09

Table J10. Snail Lake Surface Water SVOCs Data

Parameter (ug/L)	SC-SL-SW-01	SC-SL-SW-02	SC-SL-SW-03	SC-SL-SW-04	SC-SL-SW-05	SC-SL-SW-06	SC-SL-SW-07	SC-SL-SW-08	SC-SL-SW-09	SC-SL-SW-10	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-SL-FB- 01 S	SC-SL-FD- XX
Phenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	117.00	10.00	10.00	<10 U	<10 U
1,3-Dichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
1,2-Dichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
4-Methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	72.20	10.00	10.00	<10 U	<10 U
Nitrobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
2,4-Dimethylphenol	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10	<10	0 / 10	none	10.00	10.00	<10 UJ	<10 UJ
2,4-Dichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
4-Chloroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	none	10.00	20.00	<20 U	<20 U
2-Methylnaphthalene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	2.08	10.00	10.00	<10 U	<10 U
2,4,5-Trichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	2.00	25.00	10.00	<10 U	<10 U
Dimethylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.00	10.00	10.00	<10 U	<10 U
3-Nitroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	none	10.00	20.00	<20 U	<20 U
4-Nitrophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	163.00	10.00	20.00	<20 U	<20 U
Diethylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	1.65	10.00	10.00	<10 U	<10 U
4-Nitroaniline	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	none	10.00	20.00	<20 U	<20 U
4-Bromophenyl-phenylether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	1.50	10.00	10.00	<10 U	<10 U
Phenanthrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	2.10	10.00	10.00	<10 U	<10 U
Fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	20.00	10.00	10.00	<10 U	<10 U
3,3'-Dichlorobenzidine	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	none	10.00	20.00	<20 U	<20 U
bis(2-Ethylhexyl)phthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.05	10.00	10.00	<10 U	<10 U
Benzo[k]fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
bis(2-Chloroethyl)ether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Dibenz[a,h]anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
1,4-Dichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	15.00	10.00	10.00	<10 U	<10 U
2-Methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	72.20	10.00	10.00	<10 U	<10 U
N-Nitroso-di-n-propylamine	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Isophorone	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Benzoic acid	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	41.60	10.00	20.00	<20 U	<20 U
1,2,4-Trichlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	110.00	10.00	10.00	<10 U	<10 U
Hexachlorobutadiene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Hexachlorocyclopentadiene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
2-Chloronaphthalene	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10	<10	0 / 10	none	10.00	10.00	<10 UJ	<10 UJ
Acenaphthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	12.00	10.00	10.00	<10 U	<10 U
Acenaphthylene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	12.00	10.00	10.00	<10 U	<10 U
Dibenzofuran	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	20.40	10.00	10.00	<10 U	<10 U
4-Chlorophenyl-phenylether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
4,6-Dinitro-2-methylphenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	none	25.00	20.00	<20 U	<20 U
Hexachlorobenzene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.00	10.00	10.00	<10 U	<10 U
Anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.03	10.00	10.00	<10 U	<10 U
Pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Benzo[a]anthracene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.03	10.00	10.00	<10 U	<10 U
Di-n-octylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	30.00	10.00	10.00	<10 U	<10 U
Benzo[a]pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.01	10.00	10.00	<10 U	<10 U
Benzo[g,h,i]perylene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
2-Chlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Benzyl alcohol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	58.00	10.00	10.00	<10 U	<10 U
bis(2-chloroisopropyl)ether	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Hexachloroethane	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10 UJ	<10	<10	0 / 10	12.00	10.00	10.00	<10 UJ	<10 UJ
2-Nitrophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	163.00	10.00	10.00	<10 U	<10 U
bis(2-Chloroethoxy)methane	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Naphthalene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	81.00	10.00	10.00	<10 U	<10 U
4-Chloro-3-methylphenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
2,4,6-Trichlorophenol	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	2.00	10.00	10.00	<10 U	<10 U
2-Nitroaniline	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	25.00	10.00	<10 U	<10 U
2,6-Dinitrotoluene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	50.00	10.00	10.00	<10 U	<10 U
2,4-Dinitrophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	none	25.00	20.00	<20 U	<20 U
2,4-Dinitrotoluene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	18.50	10.00	10.00	<10 U	<10 U
Fluorene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	3.90	10.00	10.00	<10 U	<10 U
n-Nitrosodiphenylamine	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	24.50	10.00	10.00	<10 U	<10 U
Pentachlorophenol	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20 U	<20	<20	0 / 10	3.50	25.00	20.00	<20 U	<20 U
Di-n-butylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.10	10.00	10.00	<10 U	<10 U
Butylbenzylphthalate	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	0.08	10.00	10.00	<10 U	<10 U
Chrysene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Benzo[b]fluoranthene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
Indeno[1,2,3-cd]pyrene	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
N-Nitrosodimethylamine	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10 U	<10	<10	0 / 10	none	10.00	10.00	<10 U	<10 U
																		Sample location SL-09

Table J11. Snail Lake Sediment TOC and Metals Data

Parameter	SC-SL-SD-01	SC-SL-SD-02	SC-SL-SD-03	SC-SL-SD-04	SC-SL-SD-05	SC-SL-SD-06	SC-SL-SD-07	SC-SL-SD-08	SC-SL-SD-09	SC-SL-SD-10	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-SL-FB-01 SED	SC-SL-FD-XX SED
TOC %	0.93	1.10	1.70	5.10	5.50	1.00	3.20	0.82	1.10	3.00	0.82	5.50	na	na	na	na	na	1.10
GFAA Weight (g)	1.04	1.01	1.03	1.010	1.01	1.01	1.03	1.04	1.00	1.020	1.002	1.044	na	na	na	na	10.018	1.01
ICP Weight (g)	0.66	1.01	1.01	1.000	1.000	1.00	1.01	1.000	1.01	1.01	0.659	1.008	na	na	na	na	10.000	1.000
Percent Solids	4.60	5.20	5.0	5.80	7.0	5.40	8.60	5.40	6.40	9.50	4.6	9.5	na	na	na	na	na	6.80
Metals (mg/kg)																		
Aluminum	12000.00 J	10000.00 J	7800.00 J	9400.00 J	9000.00 J	13000.00 J	7200.00 J	9300.00 J	9600.00 J	8400.00 J	7200.00	13000.00	10 / 10	none	40.00	40.00	4.0 UJ	8600.00 J
Antimony	0.80 J	1.1 J	0.82 J	1.1 J	1.2 J	1.1 J	1.4 J	1.1 J	1.4 J	0.77 J	0.77	1.40	10 / 10	none	40.00	0.50	0.050 UJ	1.10 J
Arsenic	66.00	49.00	23.00	57.00	34.00	61.00	58.00	74.00	70.00	32.00	23.00	74.00	10 / 10	6.00	1.50	1.00	0.10 U	69.00
Barium	140.00 J	110.00 J	130.00 J	140.00 J	140.00 J	130.00 J	210.00 J	120.00 J	160.00 J	180.00 J	110.00	210.00	10 / 10	none	4.00	4.00	0.40 UJ	150.00 J
Beryllium	1.50 U	0.99 U	0.99 U	1.0 U	1.0 U	1.0 U	0.99 U	1.0 U	0.99 U	1.0 U	<0.99	<1.5	0 / 10	none	4.00	1.00	0.10 U	1.0 U
Cadmium	2.20 J	1.00 J	0.72 J	1.60 J	0.98 J	2.90 J	1.30 J	1.30 J	1.50 J	0.85 J	0.72	2.90	10 / 10	0.60	0.50	0.50	0.050 UJ	1.50 J
Chromium	28.00 J	29.00 J	19.00 J	19.00 J	14.00 J	31.00 J	13.00 J	21.00 J	24.00 J	15.00 J	13.00	31.00	10 / 10	26.00	1.00	1.00	0.10 U	20.00 J
Cobalt	15.00 U	9.90 U	9.90 U	10.00 U	10.00 U	10.00 U	9.90 U	10.00 U	9.90 U	10.00 U	<9.9	<15	0 / 10	none	10.00	15.00	1.0 U	10.00 U
Copper	83.00 J	34.00 J	67.00 J	95.00 J	91.00 J	140.00 J	110.00 J	160.00 J	130.00 J	160.00 J	34.00	160.00	10 / 10	16.00	1.00	1.00	**0.10 R**	120.00 J
Lead	140.00 J	66.00 J	46.00 J	99.00 J	52.00 J	150.00 J	77.00 J	86.00 J	92.00 J	42.00 J	42.00	150.00	10 / 10	31.00	1.00	1.00	0.10 UJ	84.00 J
Magnesium	5000.00	3800.00	3300.00	4000.00	3900.00	4400.00	5200.00	3800.00	4200.00	4400.00	3300.00	5200.00	10 / 10	none	40.00	40.00	4.0 U	4100.00
Manganese	240.00	150.00	240.00	230.00	230.00	340.00	520.00	230.00	350.00	470.00	150.00	520.00	10 / 10	460.00	2.00	2.00	0.20 U	350.00
Mercury	0.22 UJ	0.12 J	0.10	0.16 J	0.13	0.27 J	0.14	0.14	0.15 J	0.13	0.10	0.27	10 / 10	0.20	0.10	0.10	0.0098 UJ	0.14 J
Nickel	22.00 J	18.00	14.00 J	17.00	17.00	22.00	13.00	18.00	17.00	13.00	13.00	22.00	10 / 10	16.00	10.00	10.00	1.0 U	17.00
Selenium	2.3 J	2.20 J	1.3 J	1.2 J	0.99 U	2.7 J	1.0 J	1.2 J	1.0 U	1.1 J	<0.99	2.70	8 / 10	none	40.00	1.00	0.10 U	1.20 J
Silver	0.48 U	0.49 U	0.49 U	0.50 U	0.49 U	0.50 U	0.49 U	0.48 U	0.5 U	0.49 U	<0.48	<0.50	0 / 10	1.00	0.50	0.50	0.050 U	0.50 U
Thallium	0.96 UJ	0.99 UJ	0.97 UJ	0.99 UJ	0.99 UJ	1.0 UJ	0.97 UJ	0.96 U	1.0 UJ	0.98 UJ	<0.96	<1.0	0 / 10	none	40.00	1.00	0.10 UJ	0.99 UJ
Zinc	150.00	110.00	82.00	120.00	120.00 J	160.00	110.00 J	100.00 J	120.00	95.00 J	82.00	160.00	10 / 10	120.00	5.00	5.00	0.50 U	120.00
																		Sample location SL-09

Table J12. Snail Lake Sediment Pesticides and PCBs Data

Parameter (mg/kg)	SC-SL-SD-01	SC-SL-SD-02	SC-SL-SD-03	SC-SL-SD-04	SC-SL-SD-05	SC-SL-SD-06	SC-SL-SD-07	SC-SL-SD-08	SC-SL-SD-09	SC-SL-SD-10	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-SL-FD-XX-S	SC-SL-FB-SED 01
%SOLIDS	5.7	4.5	3.7	3.90	5.50	5.00	8.10	5.70	4.80	7.70	3.70	8.10	10 / 10	none	none	none	0.1	1.00 U
4,4'-DDD	0.02 UJ	0.04 UJ	0.05 UJ	0.04 UJ	0.03 UJ	0.03 UJ	0.02 UJ	0.03 UJ	0.03 UJ	0.02 UJ	<0.02	<0.05	0 / 10	0.01	0.01	0.05	0.03 UJ	1.00 U
4,4'-DDE	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.01	0.01	0.05	0.03 U	1.00 U
4,4'-DDT	**0.083**	0.04 U	0.05 U	0.04 UJ	0.03 UJ	0.03 U	0.02 UJ	0.03 UJ	0.03 U	0.02 UJ	<0.02	**0.08**	1 / 10	0.01	0.01	0.05	0.03 U	1.00 U
Aldrin	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.00	0.01	0.05	0.03 U	1.00 U
Alpha-BHC	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.01	0.01	0.05	0.03 U	1.00 U
Beta-BHC	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.01	0.01	0.05	0.03 U	1.00 U
Chlordane (technical)	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.08 U	0.02 U	<0.02	<0.08	0 / 10	0.01	0.01	0.08	0.08 U	1.00 U
Delta-BHC	0.02 UJ	0.04 UJ	0.05 UJ	0.04 UJ	0.03 UJ	0.03 UJ	0.02 UJ	0.03 UJ	**0.03 J**	0.02 UJ	<0.02	**0.03**	1 / 10	0.00	0.01	0.05	0.03 UJ	1.00 U
Dieldrin	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.00	0.01	0.05	0.03 U	1.00 U
Endosulfan I	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.05	0.01	0.05	0.03 U	1.00 U
Endosulfan II	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.05	0.01	0.05	0.03 U	1.00 U
Endosulfan Sulfate	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	none	0.01	0.05	0.03 U	1.00 U
Endrin	0.02 UJ	0.04 UJ	0.05 UJ	0.04 UJ	0.03 UJ	0.03 UJ	0.02 UJ	0.03 UJ	0.03 UJ	0.02 UJ	<0.02	<0.05	0 / 10	0.00	0.01	0.05	0.03 UJ	1.00 U
Endrin Aldehyde	0.02 UJ	0.04 U	0.05 U	0.04 UJ	0.03 UJ	0.03 U	0.02 UJ	0.03 UJ	0.03 U	0.02 UJ	<0.02	<0.05	0 / 10	none	0.01	0.05	0.03 U	1.00 U
Heptachlor	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	**0.04**	0.02 U	<0.02	**0.04**	1 / 10	0.00	0.01	0.05	**0.031**	1.00 U
Heptachlor Epoxide	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	0.02 U	0.03 U	0.03 U	0.02 U	<0.02	<0.05	0 / 10	0.01	0.01	0.05	0.03 U	1.00 U
Lindane	0.02 U	0.04 U	0.05 U	0.04 UJ	0.03 U	0.03 U	**0.05**	0.03 U	**0.04**	0.02 U	<0.02	**0.05**	2 / 10	0.00	0.01	0.05	0.03 U	1.00 U
Methoxychlor	0.02 U	0.04 UJ	0.05 UJ	0.04 UJ	0.03 U	0.03 UJ	0.02 U	0.03 U	0.03 UJ	0.02 U	<0.02	<0.05	0 / 10	0.02	0.01	0.05	0.03 UJ	1.00 U
Toxaphene	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	0.17 U	<0.17	<0.17	0 / 10	0.03	0.13	0.17	0.17 U	1.00 UJ
Aroclor-1016	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 10	0.01	1.00	0.33	0.33 U	1.00 UJ
Aroclor-1221	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 10	none	1.00	0.33	0.33 U	1.00 U
Aroclor-1232	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 10	none	1.00	0.33	0.33 U	1.00 U
Aroclor-1242	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 10	none	1.00	0.33	0.33 U	1.00 U
Aroclor-1248	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 10	0.03	1.00	0.33	0.33 U	1.00 U
Aroclor-1254	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 10	0.06	1.00	0.33	0.33 U	1.00 U
Aroclor-1260	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	0.33 U	<0.33	<0.33	0 / 10	0.01	1.00	0.33	0.33 U	1.00 UJ
																	Sample Location SL-09	

Table J13. Snail Lake Sediment SVOCs Data

Parameter (ug/kg)	SC-SL-SD-01	SC-SL-SD-02	SC-SL-SD-03	SC-SL-SD-04	SC-SL-SD-05	SC-SL-SD-06	SC-SL-SD-07	SC-SL-SD-08	SC-SL-SD-09	SC-SL-SD-10	Min	Max	DF	RBAL	Target DL	Achieved DL	SC-SL-FD-XX-S	SC-SL-SD-FB-01
Phenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
1,3-Dichlorobenzene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	1700.00	330.00	2400.00	1700 U	77.00 U
1,2-Dichlorobenzene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	340.00	330.00	2400.00	1700 U	77.00 U
4-Methylphenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Nitrobenzene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	13000.00	1700 U	77.00 U
2,4-Dimethylphenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 UJ
2,4-Dichlorophenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
4-Chloroaniline	18000 U	20000 U	20000 U	20000 U	18000 U	24000 U	17000 U	20000 U	18000 U	16000 U	<16000	<24000	0 / 10	none	330.00	24000.00	17000 U	150.00 U
2-Methylnaphthalene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	70.00	330.00	2400.00	1700 U	77.00 U
2,4,5-Trichlorophenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Dimethylphthalate	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
3-Nitroaniline	3600 U	4000 U	4000 U	4000 U	3600 U	4700 U	3400 U	4000 U	3700 U	3300 U	<3300	<4000	0 / 10	none	800.00	4700.00	3500 U	150.00 U
4-Nitrophenol	3600 U	4000 U	4000 U	4000 U	3600 U	4700 U	3400 U	4000 U	3700 U	3300 U	<3300	<4000	0 / 10	none	330.00	4700.00	3500 U	150.00 U
Diethylphthalate	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	630.00	330.00	2400.00	1700 U	77.00 U
4-Nitroaniline	3600 U	4000 U	4000 U	4000 U	3600 U	4700 U	3400 U	4000 U	3700 U	3300 U	<3300	<4000	0 / 10	none	330.00	5200.00	3500 U	150.00 U
4-Bromophenyl-phenylether	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	1300.00	330.00	2400.00	1700 U	77.00 U
Phenanthrene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	560.00	330.00	2400.00	1700 U	77.00 U
Fluoranthene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	190.00	330.00	2400.00	1700 U	77.00 U
3,3'-Dichlorobenzidine	3600 UJ	4000 U	4000 U	4000 U	3600 U	4700 UJ	3400 U	4000 U	3700 U	3300 U	<3300	<4000	0 / 10	none	330.00	4700.00	3500 UJ	150.00 U
bis(2-Ethylhexyl)phthalate	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	25000.00	330.00	2400.00	1700 U	77.00 U
Benzo[k]fluoranthene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	240.00	330.00	2400.00	1700 U	77.00 U
Dibenz[a,h]anthracene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	60.00	330.00	2400.00	1700 U	77.00 U
bis(2-Chloroethyl)ether	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
1,4-Dichlorobenzene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	350.00	330.00	2400.00	1700 U	77.00 U
2-Methylphenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
N-Nitroso-di-n-propylamine	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Isophorone	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Benzoic acid	3600 U	4000 U	4000 U	4000 U	3600 U	4700 U	3400 U	4000 U	3700 U	3300 U	<3300	<4000	0 / 10	none	330.00	4700.00	3500 U	150.00 U
1,2,4-Trichlorobenzene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	9200.00	330.00	2400.00	1700 U	77.00 U
Hexachlorobutadiene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Hexachlorocyclopentadiene	1800 UJ	2000 U	2000 U	2000 U	1800 U	2400 UJ	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 UJ	77.00 U
2-Chloronaphthalene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 UJ
Acenaphthene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	620.00	330.00	2400.00	1700 U	77.00 U
Acenaphthylene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	620.00	330.00	2400.00	1700 U	77.00 U
Dibenzofuran	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	2000.00	330.00	2400.00	1700 U	77.00 U
4-Chlorophenyl-phenylether	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
4,6-Dinitro-2-methylphenol	3600 UJ	4000 UJ	4000 UJ	4000 UJ	3600 UJ	4700 UJ	3400 UJ	4000 UJ	3700 UJ	3300 UJ	<3300	<4000	0 / 10	none	330.00	4700.00	3500 UJ	150.00 U
Hexachlorobenzene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Anthracene	1800.00 U	2000.00 U	2000 U	2000 U	1800.00 U	2400.00 U	1700.00 U	2000.00 U	1800 U	1600 U	<1600	<2400	0 / 10	220.00	330.00	2400.00	1700.00 U	77.00 U
Pyrene	1800.00 U	2000.00 U	2000 U	2000 U	1800.00 U	2400.00 U	1700.00 U	2000.00 U	1800 U	1600 U	<1600	<2400	0 / 10	490.00	330.00	2400.00	1700.00 U	77.00 U
Benzo[a]anthracene	1800.00 U	2000.00 U	2000 U	2000 U	1800.00 U	2400.00 U	1700.00 U	2000.00 U	1800 U	1600 U	<1600	<2400	0 / 10	320.00	330.00	2400.00	1700.00 U	77.00 U
Di-n-octylphthalate	1800.00 U	2000.00 U	2000 U	2000 U	1800.00 U	2400.00 U	1700.00 U	2000.00 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700.00 U	77.00 U
Benzo[a]pyrene	1800.00 U	2000.00 U	2000 U	2000 U	1800.00 U	2400.00 U	1700.00 U	2000.00 U	1800 U	1600 U	<1600	<2400	0 / 10	370.00	330.00	2400.00	1700.00 U	77.00 U
Benzo[g,h,i]perylene	1800.00 U	2000.00 U	2000 U	2000 U	1800.00 U	2400.00 U	1700.00 U	2000.00 U	1800 U	1600 U	<1600	<2400	0 / 10	170.00	330.00	2400.00	1700.00 U	77.00 U
2-Chlorophenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Benzyl alcohol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
bis(2-chloroisopropyl)ether	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Hexachloroethane	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	1000.00	330.00	2400.00	1700 U	77.00 UJ
2-Nitrophenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
bis(2-Chloroethoxy)methane	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Naphthalene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	480.00	330.00	2400.00	1700 U	77.00 U
4-Chloro-3-methylphenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
2,4,6-Trichlorophenol	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
2-Nitroaniline	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	800.00	2400.00	1700 U	77.00 U
2,6-Dinitrotoluene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	21.00	330.00	2400.00	1700 U	77.00 U
2,4-Dinitrophenol	3600 UJ	4000 UJ	4000 UJ	4000 UJ	3600 UJ	4700 UJ	3400 UJ	4000 UJ	3700 UJ	3300 UJ	<3300	<4000	0 / 10	none	800.00	4700.00	3500 UJ	150.00 U
2,4-Dinitrotoluene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	42.90	330.00	2400.00	1700 U	77.00 U
Fluorene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	190.00	330.00	2400.00	1700 U	77.00 U
n-Nitrosodiphenylamine	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Pentachlorophenol	3600 U	4000 U	4000 U	4000 U	3600 U	4700 U	3400 U	4000 U	3700 U	3300 U	<3300	<4000	0 / 10	none	800.00	4700.00	3500 U	150.00 U
Di-n-butylphthalate	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	11000.00	330.00	2400.00	1700 U	77.00 U
Butylbenzylphthalate	1800 UJ	2000 UJ	2000 UJ	2000 UJ	1800 UJ	2400 UJ	1700 UJ	2000 UJ	1800 UJ	1600 UJ	<1600	<2400	0 / 10	11000.00	330.00	2400.00	1700 UJ	77.00 U
Chrysene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	340.00	330.00	2400.00	1700 U	77.00 U
Benzo[b]fluoranthene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	330.00	2400.00	1700 U	77.00 U
Indeno[1,2,3-cd]pyrene	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	200.00	330.00	2400.00	1700 U	77.00 U
N-Nitrosodimethylamine	1800 U	2000 U	2000 U	2000 U	1800 U	2400 U	1700 U	2000 U	1800 U	1600 U	<1600	<2400	0 / 10	none	none	2400.00	1700 U	77.00 U
																	Sample Location SL-09	

Table J14. Snail Lake First Quarter Water Quality and Metals Data

Parameter	SC-SL-SW-01	SC-SL-SW-02	SC-SL-SW-03	SC-SL-SW-04	SC-SL-SW-05	SC-SL-SW-06	SC-SL-SW-07	SC-SL-SW-08	SC-SL-SW-09	SC-SL-SW-10	Detection limit
pH	8.57	8.58	8.69	8.73	8.65	8.65	8.74	8.56	8.49	8.94	na
DO (mg/L)	8.88	8.97	9.43	9.84	9.44	9.35	10.00	9.86	9.69	11.39	na
DO %	91.3	92.9	98.1	102.7	98.5	98.4	104.7	103.5	102	120.5	na
Conductivity (umhos/L)	299.00	300.00	298.00	298.00	304.00	299.00	301.00	300.00	299.00	301.00	na
Temperature (C)	16.64	17.05	17.23	no value	17.36	no value	17.64	17.67	no value	17.90	na
Hardness (mg/L)	102.00	102.00	100.00	104.00	104.00	98.00	100.00	102.00	96.00	104.00	2.00
TKN (mg/L)	0.81	0.82	0.92	0.82	0.79	0.81	0.71	0.76	0.77	0.74	0.50
TOC (mg/L)	8.00	12.00	8.90	8.10	11.00	7.80	8.80	8.60	9.00	8.00	1.00
Total Phosphorus (mg/L)	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.05 U	0.01
TSS (mg/L)	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00 U	1.00
Metals (ug/L)											
Aluminum	6.25	6.85	3.35	6.95	6.35	6.25	5.50	6.85	6.05	6.35	0.06
Barium	42	42	30	41	42	41	40	40	41	40	0.0056
Cadmium	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.015 U	0.0150
Copper	2.1	2.3	2.0	2.1	2.2	2.2	2.1	2.1	2.2	2.3	0.0140
Mercury	0.00061	0.00051	0.00058	0.00050	0.00069	0.00075	0.00051	0.00051	0.00048	0.00058	0.00018
Silver	0.0202 UB	0.0105 UB	0.0242 UB	0.0242 UB	0.0105 UB	0.0157 UB	0.0117 UB	0.0141 UB	0.0162 UB	0.0152 UB	0.0043
Zinc	0.670	0.79	0.55 UB	0.70	0.68	0.60	0.620	0.630	0.810	0.69	0.0350

▨ These are the sampling locations that were repeated in the 2nd, 3rd, and 4th quarters.

QC Samples

Parameter	SC-SL-FB-01	SC-SL-FD- XX
pH	na	8.49
DO (%)	na	9.69
DO (mg/L)	na	102
Conductivity (umhos/L)	na	299.00
Temperature (C)	na	no value
Hardness (mg/L)	na	102.00
TKN (mg/L)	na	0.72
TOC (mg/L)	na	8.30
Total Phosphorus (mg/L)	na	0.05 U
TSS (mg/L)	na	1.00 U
Metals (ug/L)		
Aluminum	0.44 UB	6.15
Barium	0.85	40.00
Cadmium	0.02	0.02 U
Copper	0.50	2.10
Mercury	0.00	0.00
Silver	0.0105 UB	0.0138 UB
Zinc	10.11	0.65
		SC-SL-SW-09

Table J15. Snail Lake Second Quarter Surface Water Quality and Metals Data

Parameter	W2-SL-SW-01	W2-SL-SW-06	W2-SL-SW-09	Detection limit
pH	7.74	8.02	8.05	na
DO (mg/L)	12.86	11.88	11.11	na
DO (%)	not available	83.4	81.6	na
Conductivity (µmhos/L)	385	399	387	na
Temperature (°C)	0.95	1.10	2.30	na
Hardness (mg/L)	146	146	140	2.00
TKN (mg/L)	1.00	0.92	0.74	0.50
TOC (mg/L)	9.2	9.2	11.0	1.00
Total Phosphorus (mg/L)	0.04	0.03	0.02	0.01
TSS (mg/L)	3	2	1 U	1.00
Metals (µg/L)				
Aluminum	4.120	4.020	4.520	0.056
Barium	55.01	58.01	55.01	0.0056
Cadmium	0.027	0.015 U	0.019	0.015
Copper	2.4	2.5	2.5	0.014
Mercury	0.000783 J	0.001300 J	0.000630 J	0.000039
Silver	0.0143 UBJ	0.0143 UBJ	0.0143 UBJ	0.0043
Zinc	1.437	1.337	1.237	0.035

QC Samples

Parameter	W2-SL-FB-01	W2-SL-FD-00
pH	na	na
DO (%)	na	na
DO (mg/L)	na	na
Conductivity (umhos/L)	na	na
Temperature (C)	na	na
Hardness (mg/L)	na	146
TKN (mg/L)	na	0.68
TOC (mg/L)	na	8.8
Total Phosphorus (mg/L)	na	0.03
TSS (mg/L)	na	1
Metals (ug/L)		
Aluminum	0.785 UBJ	4.52
Barium	0.0159 UB	57
Cadmium	0.015 U	0.023
Copper	0.028	2.4
Mercury	0.000230 UBJ	0.000610 J
Silver	0.0143 UBJ	0.0143 UBJ
Zinc	0.9270	1.637
		W2-SL-SW-09

na=not available

Table J16. Snail Lake Third Quarter Water Quality and Metals Data

Parameter	W3-SL-SW-01	W3-SL-SW-06	W3-SL-SW-09	Detection limit
pH	8.45	8.53	8.50	na
DO (%)	108.90	112.4	113.6	na
DO (mg/L)	11.37	11.47	11.28	na
Conductivity (umhos/L)	370.00	372.00	373.00	na
Temperature (C)	13.38	14.23	15.66	na
Hardness	134	138	136	2.00
TKN (mg/L)	0.52	0.5 U	0.63	0.50
TOC (mg/L)	8.3	8.5	8.5	0.20
Total Phosphorus (mg/L)	0.02	0.02	0.03	0.01
TSS (mg/L)	1.00 U	2.00	1	1.00
Metals (ug/L)				
Aluminum	8.2	7.2	7.2	0.01
Barium	48 J	51 J	52 J	0.00088
Cadmium	0.01195	0.01095	0.01195	0.000660
Copper	2.3	2.1	2.1	0.0043
Mercury	0.00075	0.00082	0.00105	0.000063
Silver	0.01858 UBJ	0.01858 UBJ	0.01858 UBJ	0.00058
Zinc	1.1180 J	1.0180 J	1.1180 J	0.0044

Bottom Water

Parameter	W3-SL-BW-01	W3-SL-BW-06
pH	8.30	8.33
DO (%)	95.8	100.4
DO (mg/L)	11.57	11.72
Conductivity (umhos/L)	371.00	374.00
Temperature (C)	7.00	7.55
Hardness	148	136
TKN (mg/L)	0.82	0.68
TOC (mg/L)	8.3	7.6
Total Phosphorus (mg/L)	0.02	0.02
TSS (mg/L)	4	4
Metals (ug/L)		
Aluminum	7.3	5.9
Barium	56 J	57 J
Cadmium	0.00915	0.00734 UB
Copper	2.1	2.0
Mercury	0.000790	0.000780
Silver	0.01858 UBJ	0.01858 UBJ
Zinc	1.518 J	1.218 J

QC Samples

Parameter	W3-SL-FD-00	W3-SL-FD-02	W3-SL-FB-01	W3-SL-BB-01	W3-SL-EB-01
pH	8.47	8.30	na	na	na
DO (%)	108.9	95.5	na	na	na
DO (mg/L)	11.36	11.55	na	na	na
Conductivity (umhos/L)	370.00	371.00	na	na	na
Temperature (C)	13.37	6.99	na	na	na
Hardness (m/L)	140	134	na	na	na
TKN (mg/L)	0.55	0.76	na	na	na
TOC (mg/L)	8.1	8.4	na	na	na
Total Phosphorus (mg/L)	0.01	0.03	na	na	na
TSS (mg/L)	1	2	na	na	na
Metals (ug/L)					
Aluminum	7.2	6.7	0.085	1.0	<0.099 U
Barium	47 J	57 J	0.0026 J	0.023 J	0.0071 J
Cadmium	0.01035	0.00885 UB	0.00161 UB	0.00161 UB	0.00161 UB
Copper	2.3	2.1	0.013	0.0071	0.0086
Mercury	0.000690	0.000790 UB	0.000230 UB	0.000410	0.000380 UB
Silver	0.01858 UBJ	0.01858 UBJ	0.01858 UBJ	0.01858 UBJ	0.01858 UBJ
Zinc	1.118 J	1.218 J	0.022 UBJ	0.105 UBJ	2.318 J
na=not available	W3-SL-SW-01	W3-SL-BW-01			

Table J17. Snail Lake Fourth Quarter Water Quality and Metals Data

Parameter	W4-SL-SW-01	W4-SL-SW-06	W4-SL-SW-09	Detection limit
pH	8.59	8.42	8.48	na
DO (%)	92.00	103.00	100.00	na
DO (mg/L)	8.32	9.18	8.84	na
Conductivity (umhos/L)	320.00	329.00	not available	na
Temperature (C)	20.16	20.96	21.40	na
Hardness (mg/L)	108	112	114	2.00
TKN (mg/L)	0.5 U	0.5 U	0.52	0.50
TOC (mg/L)	9.4	9.4	9.4	0.20
Total Phosphorus (mg/L)	0.02	0.02	0.02	0.01
TSS (mg/L)	1	1	1	1.00
Metals (ug/L)				
Aluminum	8.09	7.09	5.29	0.009900
Barium	38	40	41	0.000880
Cadmium	0.0053	0.0030	0.0068	0.000660
Copper	11	11	11	0.004300
Mercury	0.0014	0.00072	0.00093	0.000039
Silver	0.00328 UBJ	0.00328 UBJ	0.00328 UBJ	0.000580
Zinc	0.862	0.482	1.452	0.004000

Bottom water

Parameter	W4-SL-BW-01	W4-SL-BW-06
pH	7.69	7.50
DO (%)	32.20	28.40
DO (mg/L)	3.51	2.83
Conductivity (umhos/L)	380.00	365.00
Temperature (C)	12.75	13.85
Hardness (mg/L)	140	142
TKN (mg/L)	0.76	0.64
TOC (mg/L)	9	8.9
Total Phosphorus (mg/L)	0.02	0.02
TSS (mg/L)	3	2
Metals (ug/L)		
Aluminum	5.99	7.59
Barium	53	48
Cadmium	0.0051	0.00507
Copper	2.1	3.4
Mercury	0.00062	0.00065
Silver	0.00328 UBJ	0.00328 UBJ
Zinc	2.052	0.672

QC Samples

Parameter	W4-SL-EB-01	W4-SL-FD-01	W4-SL-FD-02	W4-SL-FB-01
pH	na	8.59	7.69	na
DO (%)	na	92	32.5	na
DO (mg/L)	na	8.32	3.56	na
Conductivity (umhos/L)	na	319.00	379.00	na
Temperature (C)	na	20.16	12.75	na
Hardness (mg/L)	na	110	100	na
TKN (mg/L)	na	0.5 U	0.58	na
TOC (mg/L)	na	9.2	9.1	na
Total Phosphorus (mg/L)	na	0.01	0.02	na
TSS (mg/L)	na	1	3	na
Metals (ug/L)				
Aluminum	1.10 UB	7.49	7.39	0.97 UB
Barium	<0.00088 U	38	53	0.0061
Cadmium	<0.00066 U	0.0049	0.0048	<0.00066 U
Copper	<0.0043 U	11	2.1	<0.0043
Mercury	0.00054	0.00079	0.00064	0.00017
Silver	0.00328 UBJ	0.00328 UBJ	0.00328 UBJ	0.00328 UBJ
Zinc	0.952	0.662	0.552	0.109 UB
na= not available	W4-SL-BW-01	W4-SL-SW-01	W4-SL-BW-01	W4-SL-SW-01