# SUDBURY AREA RISK ASSESSMENT

# VOLUME III – CHAPTER 5.0
# AQUATIC PROBLEM FORMULATION

## EXECUTIVE SUMMARY

This report comprises the aquatic problem formulation for the Sudbury Soils Study ecological risk assessment (ERA). It contributes to an initial understanding of how metals from smelter particulate emissions may be affecting the aquatic and wetland environments in the City of Greater Sudbury and surrounding area. The problem formulation component of an ERA acts as an information gathering and interpretation stage that is conducted to plan and focus the approach of a detailed risk assessment on critical areas of concern. This phase of the ERA is where the initial study area is defined, chemicals of concern (COC) are selected, the ecological species, populations or communities of concern (*i.e.*, Valued Ecosystem Components or VECs) are identified, assessment endpoints are defined, and the initial conceptual model is developed. This aquatic problem formulation provides an introduction to what is known about the aquatic environments around Sudbury. It also identifies significant uncertainties and data gaps. The results of this problem formulation can then be used to guide a more detailed risk assessment of aquatic and or wetland ecosystems.

### Study Area

The study area for this aquatic problem formulation was defined largely by the boundaries of the Sudbury Soils Study and the availability of metals data for lakes in the City of Greater Sudbury. There are over 300 lakes within the boundaries of the City of Greater Sudbury. In general, these are lakes within the study area defined for the terrestrial ERA. In addition, a large number of lakes have been studied for impacts by acid rain by the Ministry of Environment and Laurentian University. Many of these lakes are located northeast and southwest of the terrestrial ERA study area. It is recommended that the study area for a comprehensive aquatic ERA be refined, in future, based on the information contained within this report, and according to the required scope of such a study.

### Chemicals of Concern

The COC selection process included the following four steps:

Step 1: Compile surface water and sediment data for 21 inorganic parameters and pH;

Step 2: Select appropriate environmental quality guidelines for each parameter;

Step 3: Apply screening criteria:

- Compare maximum concentrations to guidelines;
- Compare environmental levels to regional background levels;
- Characterize the distribution of any exceedences across the study area;
- Determine if the metal has been scientifically linked to smelting activities; and

Step 4: Prepare a summary of the rationale for the final list of COC.

- The recommended list of COC within surface water and sediments of Sudbury lakes includes:
    - Surface water: aluminium, cadmium, cobalt, copper, iron, lead, mercury, nickel and selenium.
    - Sediment: arsenic, cadmium, chromium, cobalt, copper, iron, lead, mercury, manganese, nickel, selenium, vanadium, and zinc.

**Valued Ecosystem Components and Assessment Endpoints**

Identification and selection of candidate VECs is a critical step to ensure that all relevant ecological groups within the Sudbury aquatic environment are adequately represented. The recommended VECs and assessment endpoints for the aquatic problem formulation are:

Fish Populations (white sucker, lake trout, common shiner, yellow perch, walleye)

- Presence, relative abundance, growth, development, and reproductive success
- Habitat suitability

Benthic Invertebate Community

- Community composition

Zooplankton (Pelagic Invertebrate) Community

- Community composition

Algal Community

- Community composition

Macrophyte Community

- Community composition
- Habitat suitability

Amphibian Community

- Community composition
- Presence, relative abundance, growth, development and reproductive success of particular species
- Habitat suitability for particular species

**Selection of Lakes for Potential Future Study**

Information is presented to assist in the selection of lakes for further study in the Sudbury area. Several factors should be considered before selecting lakes for further study, including: whether or not the lakes receive direct chemical inputs other than from atmoshpheric deposition from the smelters (*e.g*., industrial or municipal effluents, direct releases of mine effluents); the pH of the lake as well as whether or not the lakes have been artificially neutralized; distance from the smelters (to provide a range of metal concentrations for study); and whether or not the lakes are part of any monitoring programs (*e.g*., water and sediment chemistry, biological or toxicity studies).

A review of available information revealed that there are several lakes and rivers that have been impacted by sources other than particulate emissions from the Sudbury smelters. Therefore, these should be excluded from any further aquatic ERA of smelter airborne emissions.

Considering the availability of lake characterization data, distance from the smelters, and potential sources of metals, seven lakes are recommended for inclusion in an aquatic ERA focusing on the impacts of metals from smelter emissions: Clearwater, Hannah, Middle, McFarlane, Nelson, Ramsey, and Silver. It is recommended that the final selection of lakes for further study be completed in consultation with stakeholders (including researchers at Laurentian University and Freshwater Co-op Unit), and that a detailed description of the scope for the ERA be used to help guide this process.

It is also recommended that the marshes and wetlands of the Sudbury area are included in an aquatic risk assessment, as they serve an important role in the purification of surface waters, as well as providing unique habitat to numerous species of wildlife.

**Review of Aquatic Effects Data**

Many environmental studies across the Sudbury region have been conducted during the past few decades for a variety of purposes. Information relating to aquatic impacts from metals in lakes across the Sudbury region was compiled into a comprehensive summary of published information. This information was supplemented by non-published data and information provided by Laurentian University faculty. Compiling and reviewing this information was critical in order to understand the current state of knowledge regarding aquatic ecological effects, as well as to identify data gaps for potential further aquatic ERA studies.

Of the four species of fish selected as VECs (walleye, yellow perch, white sucker, and common shiner), Sudbury-specific studies that could be used to assess potential risks were only available for yellow perch. The results of these studies provided evidence that fish within Sudbury area lakes have been, and likely continue to be, adversely affected by the influence of smelter emissions on water and sediment quality. The extent of this impact varies from lake to lake, and species to species.

Zooplankton populations and communities began to be studied studied more than 20 years ago, when acid rain, $SO_2$ emissions and lake pH were of greater concern than today. Planktonic rotifers and crustacean zooplankton, such as daphnids, were studied relative to their sensitivity to acid pH, as well as to monitor their recovery with changes in pH and other factors (*e.g.*, improved water quality). This report summarizes the effects and "recovery" of plankton communities for lakes in Sudbury, with particular focus on Middle, Hannah and Nelson Lakes, three of the more intensively studied lakes.

No studies were found that evaluated the benthic macroinvertebrate community relative to sediment quality. Without these biological data, only a preliminary, qualitative assessment of potentially unacceptable risks can be made for this VEC. Based on the exceedances of sediment quality guidelines, risks cannot be ruled out for the benthic community in any of the 19 lakes for which sediment chemistry data are available. However, the biological monitoring studies conducted by researchers in the Sudbury area provide valuable information that can be integrated with metal exposure data in a future detailed aquatic ERA.

Little recent information on algal or macrophyte communities in Sudbury area lakes was found with which to evaluate these VECs. Macrophytes are being considered as part of a remediation strategy in Sudbury, for example, in Kelly Lake.

A current list of amphibian species present in the Sudbury region is presented in this report. However, the most extensive survey of amphibian presence in selected habitats was conducted in 1982 and 1983, making the data now over 20 years old. A detailed aquatic ERA may require additional sampling of water and possibly amphibians from wetlands and ponds of the Sudbury area to develop a stronger understanding of the potential impacts of metals on amphibians in the study area.

**Evaluation of Risks to Wildlife with an Aquatic-based Diet**

Risks were estimated for common loon, mallard and mink. The methods used are the same as those to estimate risks to other wildlife, as presented in Chapter 4 and associated appendices. No unacceptable risks were predicted for loon, mallard or mink exposed to arsenic, cadmium, cobalt, copper, lead or nickel in any portion of the study area. Unacceptable risks were predicted for mink, loon and/or mallard exposed to selenium in every portion of the study area. Unacceptable risks resulted predominantly from exposure to benthic invertebrates. There is considerable uncertainty surrounding these risk estimates. Selenium was not measured in benthic invertebrates, but was modelled based on selenium concentrations in sediment. Selenium was measured in single samples taken from only eight lakes in the study area. In addition, little information is available in the literature for uptake factors for selenium from sediment to benthic invertebrates. Therefore, there is low confidence in the results of the risk modelling for mink, loon and mallard from exposure to selenium.

There have been several observations from wildlife researchers that suggest mink, loon and mallards are common in the Sudbury area. In fact, populations of piscivorous and benthivorous birds and mammals may be increasing in size, possibly due to habitat improvements. Breeding success and abundance data have been collected for birds for the period from 2001 to 2005. The results were published in Cadman *et al.* (2008) Atlas of Breeding Birds of Ontario, 2001 – 2005. Results from this atlas could be reviewed in context of future aquatics studies or when planning regreening activities in the Sudbury area.

**Uncertainties and Data Gaps**

There are numerous uncertainties and data gaps that should be filled before a comprehensive aquatic ERA could be completed. It is recommended that any future aquatic ERA consider the following data gaps and methods to fill them:

- Comprehensive water chemistry, including but not limited to parameters such as pH and hardness, metal concentrations (for all COC and for metals not retained as COC, but for which data are limited [e.g., antimony, selenium]), for each lake selected for in-depth study;

- Comprehensive sediment chemistry, including but not limited to parameters such as organic carbon content, acid volatile sulfide and sediment texture, metal concentrations (for all COC, and for metals not retained as COC, but for which data are limited [e.g., antimony, magnesium, mercury]) for each lake selected for in-depth study;

- Sequential extraction analysis of sediments to evaluate bioavailability;

- Chemical and biological data for marshes and wetlands in the Sudbury area. Data from the 1980s suggest potential impacts on wetlands near the smelters that should be investigated;

- Biological or ecological data for the fish species identified as VECs (particularly common shiner, white sucker and walleye) in the lakes of interest;

- Benthic invertebrate community data for all lakes that may be part of the aquatic ERA; laboratory bioassays and measurements of uptake from sediment could be considered, using sediment from lakes of interest;

- Data are available for zooplankton communities in many lakes, particularly Middle and Hannah Lakes. Community metrics and laboratory bioassays should be considered for inclusion in the aquatic ERA for other lakes of interest;

- Little recent data are available for algal or macrophyte communities in the Sudbury area; surveys for lakes of interest, and laboratory algal bioassays could be considered to evaluate water quality on particular algal species; and

- Few data are available for amphibians; consideration may be given to conducting amphibian call surveys to evaluate populations in the Sudbury area.

It is anticipated that the list of studies required to complete the aquatic ERA may differ from this suggested list, based on future discussions between stakeholders.

The goals of any future ERA for aquatic life should determine the scope of the assessment. These will also assist with the delineation of the study area for the aquatic ERA. The numerous studies and long-term monitoring programs conducted by researchers in the Sudbury area will provide important ecological data that may be integrated into the detailed aquatic ERA. These studies have linked lake water pH to species abundance and community composition, dealing with fish, invertebrates, plants, and algae. The results from these programs can be used to help focus research efforts by illustrating the long-term trends in monitoring data, identifying which lakes have been significantly affected by acidification and/or metals, and which lakes may be disregarded from further research.

*This page left blank intentionally*

**SUDBURY AREA RISK ASSESSMENT**

**VOLUME III - CHAPTER 5.0**
**AQUATIC PROBLEM FORMULATION**

**Table of Contents**

**Page**

## Tables

## Figures

# 5.0 INTRODUCTION

The focus of the Sudbury Soils Study is the elevated level of metals in soil and their associated terrestrial ecological risks. The Technical Committee agreed that the emphasis of the ecological risk assessment (ERA) was the terrestrial environment. However, metals may enter and affect aquatic environments as well. Metals produced by the Sudbury smelters can enter the aquatic environment through direct deposition of airborne particulates, surface runoff, or migration of impacted groundwater. While groundwater in varying amounts is anticipated to be entering several watercourses (*e.g.*, Moose Creek, Junction Creek, Coniston Creek, Wanapitei River, *etc.*), the influence that this may have on surface water quality is difficult to determine (Smith, 2005). It was agreed early in the ERA that a detailed quantative aqutic risk assessment of the Sudbury environment would not be part of the Sudbury Soils Study. However, it was considered appropropriate to complete a problem formulation as the first phase of conducting a more detailed aquatic ERA. The contents and purpose of a problem formulation are described below in this chapter.

The deposition of smelter emissions can affect the quality of Sudbury aquatic environments both directly through deposition to surface water, and indirectly through deposition to watershed soils. If a pH of 6.0 is used to indicate the level below which significant damage will occur to most acid-sensitive components of the biological community, then over 7,000 lakes in a 17,000 km$^2$ area are predicted to have been affected by the Sudbury emissions (Keller and Gunn, 1994). Reduced smelter emissions beginning in the 1970s (see also Volume I) and the introduction of the 381 m 'superstack' at the Inco Copper Cliff site in 1972 have significantly reduced the deposition of metals to sediments in nearby lakes. A 15 to 40% reduction was observed in concentrations of Co, Zn and Ni in the sediments of some Sudbury lakes, which is consistent with changes in the local atmospheric fallout (Kramer, 1976; Chan and Lusis, 1985).

There has been an observed lag period between the occurrence of major emission reductions and improvements in surface water and sediment quality. This is a result of numerous lake- and watershed-specific physical and chemical properties. Of the 41 lakes included in a 1981 survey (Keller *et al.*, 1992a), the average hydraulic retention time was estimated to be approximately three years. This would indicate that the full effect of changes in emissions would take close to 10 years to be fully realized. Therefore, the smoothing of changing emission patterns during the mid-1980s is linked to the reductions in emissions made in the late 1970s (Keller *et al.*, 1992a).

Reductions in emissions since the 1970s have resulted in improved water quality and the recovery of phytoplankton, zooplankton, zoobenthos and fish communities, (Keller *et al.*, 2004). However, most lakes have not shown decreases in concentrations of metals such as Cu, Ni, Zn and Pb in water (Nriagu *et al*., 1998; Keller *et al*., 1992a). This is believed to be the result of the saturation of the Sudbury catchments with these metals that become mobilized from soils and glacial overburden through surface runoff, groundwater drainage, and windblown particles. This is supported by the observation that pH and metal concentrations are closely affected by rainfall (Keller *et al*., 1992a; McNicol and Mallory, 1994). Low levels of metals were reported during the dry years of 1986 and 1987 followed by significant increases in 1988 and 1989 with increased rainfall. It has been estimated that 75% of the Cu and 40% of the Ni found within Sudbury soils is geochemically mobile, with increased mobility resulting from acidic conditions (Dudka *et al*., 1995). The average contribution of water through terrestrial runoff to Sudbury lakes ranges from 60 to 90%, with 10 to 40% coming from direct precipitation to the lake surface. It is suggested that concentrations of metals stored in the catchments are high enough to sustain elevated water concentrations within the lakes for well over 1,000 years – the time it would take to reduce metal concentrations in soil from the 90$^{th}$ percentile values to the 50$^{th}$ percentile values, based on modeled soil decontamination rates (Nriagu *et al*., 1998). While improved emission control measures were an important step in the recovery process of Sudbury area lakes, the next crucial step may be the immobilization of metals within the watershed (Nriagu *et al*., 1998).

Completion of this aquatic problem formulation accomplishes the need to identify the priority issues and data gaps for any further aquatic ERA.

## 5.1 Goal and Objective

The main goal for the ERA is:

To characterize the current and future risks of chemicals of concern (COC) to terrestrial and aquatic ecosystem components from particulate emissions from Sudbury smelters. To provide information to support activities related to the recovery of regionally representative, self-sustaining ecosystems in areas of Sudbury affected by the COC.

This report addresses one of four objectives to meet this goal, namely:

> **Objective 4**
> Conduct a comprehensive problem formulation for the aquatic and wetland environments in the Sudbury area to facilitate a more detailed risk assessment in the aquatic/wetland ecosystems.

The problem formulation component of an ERA acts as an information gathering and interpretation stage which is conducted to plan and focus the approach of the risk assessment on critical areas of concern. This phase of the ERA is where the scope of the ERA is defined, COC are selected, the ecological species, populations or communities of concern (*i.e.*, valued ecosystem components or VECs) are identified, assessment endpoints are defined, and the conceptual model is developed. This aquatic problem formulation provides an introduction to what is known about the aquatic environments around Sudbury. It also identifies significant uncertainties and data gaps. The results of this problem formulation can then be used to guide a more detailed risk assessment of aquatic and/or wetland ecosystems, if needed.

## 5.2 Sources of Data and Information for this Report

Environmental studies across the Sudbury region have been conducted for a variety of purposes. These studies have revealed elevated concentrations of metals in sediments and surface waters, some of which originate from either atmospheric deposition, run-off from contaminated soil, or other sources (*e.g.*, dust from tailings, groundwater). Studies that relate specifically to aquatic VECs for this study were collected and summarized. The focus of the problem formulation is related to effects from acidification and metal contamination from airborne emissions (including subsequent runoff from soil contaminated *via* the airborne emissions). Direct effluent sources (*e.g.*, mining effluent, sewage effluent, other industrial sources, *etc*.) are not considered.

Information was collected from relevant published scientific documents, web-based sources, industry and government publications, as well as local collections (*e.g.*, the offices of the MOE, Vale Inco, Xstrata Nickel, City of Greater Sudbury, Sudbury Public Library, and local educational institutions such as the collections at the J.N. Desmarais Library and the Centre for Environmental Monitoring at Laurentian University).

In addition to the information contained within the various collections in the Sudbury area, a literature search was conducted using commercial ecological and toxicological databases (*e.g.*, Biosis, Enviroline, Agricola). The searches of relevant peer-reviewed journals, dissertations, government publications, and databases were conducted using selected keywords to obtain key results as they relate to studies conducted in the Sudbury area, for the aquatic environment. All papers or information used in the ERA were input into an electronic bibliographic database, which includes a fully-searchable catalogue.

Metal concentrations in water and sediment of Sudbury area lakes were obtained primarily from the SES (Sudbury Environmental Study) Extensive Monitoring program (Co-Op, unpublished a), the Urban Lake Water Recovery program (1990 to 2003) (Co-Op, 2004) and annual averages provided by researchers at Laurentian University (Co-Op, unpublished b). To supplement these data, the published scientific literature was reviewed to locate data for lakes not included in these programs. Please refer to section 5.5 for more information on lake water and sediment metal concentration data.

In summary, over 60 publications were reviewed for information related to direct toxicity and indirect effects, as well as monitoring and recovery of aquatic populations and communities in the Sudbury area. The SES and Urban Lakes monitoring programs were the primary source of water and sediment chemistry data. Keller *et al*. (2004) produced a summary on the recovery of acid and metal-damaged lakes near Sudbury as part of the Sudbury Soils Study and researchers at Laurentian University provided input through regular communication as well as original data, including whole body metal concentrations from forage-sized perch from 8 lakes (SARA, 2006): Ashigami, Crooked, Long, Massey [Lac St. Jean], McFarlane, Ramsey, Vermillion, and Whitson.

## 5.3 Study Area

The study area for the aquatic problem formulation began as the same area as the terrestrial ERA and 2001 soil survey. For the aquatic problem formulation, data on metal levels in lakes within this larger area were obtained. In addition, a large number of lakes have been studied related to impacts from acid rain; many of these lakes are located northeast and southwest of the terrestrial ERA study area. It is recommended that the study area for a comprehensive aquatic ERA be refined, in future, based on the information contained within this report, and according to the required scope of such a study.

Within the City of Greater Sudbury, there are 330 lakes over 10 ha in size, with only 227 given official names. There are also several hundred smaller lakes and ponds throughout the area. Water covers 12.1% of the city area, with wetlands covering another 4.2% (Pearson *et al*., 2002a). The majority of the water in the eastern part of the Sudbury area flows into the Wanapitei River, which is connected to the French River, and drains into Georgian Bay. The French River effectively dilutes the elevated nickel and copper concentrations from the Wanapitei River to regional background levels. The western area is drained by the Spanish River, which historically was more contaminated by industrial runoff (Fitchko, 1978). Reductions in emissions of sulphur dioxide and metals from the smelters over the past 30 years have had differing effects on each of the 25 separate watersheds in the Sudbury area. Some watersheds that were originally heavily impacted from air deposition are now more influenced by waterborne metals received from upstream sources, whereas other watersheds continue to have a greater contribution coming from air deposition rather than water sources. It has been estimated that 7,000 lakes within a 17,000 km$^2$ area surrounding the Greater Sudbury area were acidified to pH of 6.0 or less in the 1980s (Neary *et al*., 1990), a level at which sensitive aquatic organisms will be affected. Not all lakes located close to the smelters were acidified. The larger lakes that had a longer flushing time were able to resist acidification (Pearson *et al*., 2002a).

Generally, lakes within 100 km of the Sudbury smelters are considered to be within the range of deposition from air emissions. Lakes located to the northeast and southwest of Sudbury may be more significantly affected as a result of the prevailing wind directions (MacIsaac *et al*., 1987). Those lakes within 20 or 30 km of the smelters were found to be the most damaged, suffering from acidification as well as from potential toxic effects from trace metals such as copper and nickel. Elevated waterborne nickel and copper concentrations were found in lakes further than 50 km from Sudbury during the 1970s. Biological improvements have been noted since reductions in smelter emissions began in the 1970s. Several lakes have shown continuing reductions in metal concentrations into the 1980s, while others have either shown no clear patterns in reductions or even increases in concentrations. The presence of distinct patterns may be distorted as a result of two conditions during the 1980s. The first was a period of production cuts during 1982 to 1983 resulting in significantly reduced smelter emissions. The other is a two-year drought from 1986 to 1987 which prevented metals, collected in catchment soils, from entering lake basins through runoff. Large reductions in copper and nickel concentrations in lakes close to Sudbury followed the implementation of the Acid Rain Program in the 1990s (with the subsequent decrease in smelter emissions) (Keller *et al*., 1999, 2004).

Concentrations of copper and nickel exceeding the Provincial Water Quality Objectives (PWQOs) are now only found in lakes within approximately 30 km of Sudbury (Keller *et al*., 1999). However, metals in sediments continue to be a concern, with elevated levels still extending to 50 km from Sudbury (Semkin and Kramer, 1976; Conroy *et al*., 1978; Keller *et al*., 2004). Concentrations of copper and nickel can be found well above 1,000 µg/g in the sediments of lakes closest to the smelters (Keller *et al*., 1999). The PSQGs (Provincial Sediment Quality Guidelines) relate significant biological effects to the benthic community to concentrations above 110 µg/g for copper and 75 µg/g for nickel.

A recent study of core samples of four lakes within 15 km of Sudbury showed that only two of the lakes had declines in concentrations of copper and nickel in the uppermost (1 cm) sediment, indicating that the natural process of burying contaminated sediment may be slow (Borgmann *et al*., 1998).

Originally, there was a high degree of uncertainty whether lakes that suffered from acidification would be able to recover naturally or if artificial neutralization would be required. This was largely dependent on the acid-neutralizing capacity of the watershed soils. The resilience of the Sudbury aquatic systems became evident as the pH of more and more lakes increased, approaching neutrality (Figure 5-1). It appeared that the buffering capacity of many of the affected lakes was temporally overloaded but not exhausted (Keller and Gunn, 1994). It has been suggested that the effects of the smelters may have been worse if it were not for the natural buffering capacity of many of the lakes in the Sudbury area. A survey of the 33 largest lakes in the Sudbury area showed that only 11 had pH less than 6 in 1990 (Gunn and Keller, 1995), indicating that the natural alkalinity of these lakes was able to offset the acidification process.

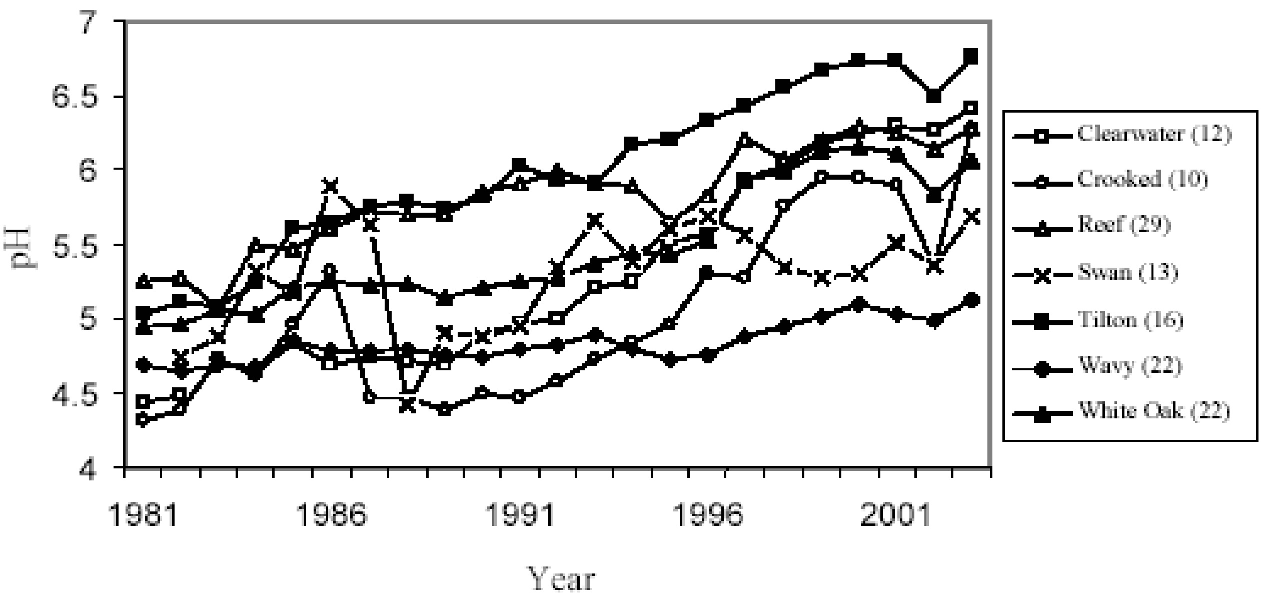



**Figure 5-1 Trends of Surface Water pH from 1981 to 2003 in Seven Lakes Found Within 30 km of Sudbury. Values in Brackets Represent the Distance (km) from Sudbury (Keller *et al.*, 2004)**

With reductions in sulfur dioxide emissions, levels of sulfate in surface water have shown substantial declines, along with increases in alkalinity and pH.

Elevated concentrations of copper and nickel have historically been documented in surface water and sediments in lakes extending to greater than 50 km from Sudbury (Semkin and Kramer, 1976; Conroy *et al*., 1978), and more recent sediment chemistry data from the 1990s illustrates a continuing relationship between concentrations of these chemicals and distance from Sudbury, with contamination out to approximately 50 km (Keller *et al*., 2004).

Therefore, for the purpose of the aquatic problem formulation, the initial recommended study area for evaluating effects of metals in water and sediment is within 50 km of the smelters. It is acknowledged that due to prevailing wind direction, a circular radius of 50 km from the City of Greater Sudbury should not be used as a distinct limit of the study area to be addressed in the aquatic risk assessment. Additional rationale for the inclusion or exclusion of lakes within or beyond this area should be evaluated accordingly.

## 5.4 Chemicals of Concern for the Aquatic Environment

Different metals may be identified as COC in surface water and sediment. This may be due to differences in the distribution of metals within a lake. For example, the seasonal temperature changes in the Sudbury area create conditions of thermal stratification in many of the local lakes, influencing the distribution of metals within the water column and sediment. In the months of May to October, stratified lakes, such as Kelly Lake, can often have low concentrations of dissolved and particulate copper in the upper water column (uppermost 4 m) and elevated levels of particulate-associated copper in bottom water. This is a result of dissolved copper adsorbing to dead algae as it sinks from the surface water to the sediment, acting as a scavenger as it falls. Summer algae blooms are frequent in lakes that have high nutrient loading, particularly those receiving sewage. Iron is another metal that will fall from the upper oxygenated water during the summer months and dissolve in the lower anoxic water (Pearson *et al.*, 2002b). Nickel is found primarily in dissolved form, rather than particulate. It is not removed from the water column as effectively as copper. Following spring stratification of a lake, concentrations of nickel in the bottom water decrease and remain lower than the concentration in the upper portion of the water column. Summer sedimentation of nickel does not occur as it does for copper. The distribution of nickel throughout the water column remains consistent throughout the year (Pearson *et al.*, 2002b). Although nickel may settle, complex, or diffuse onto the sediment surface from the lower water column, the high concentrations of nickel found in sediment in Sudbury are more likely a result of atmospheric deposition of smelter dust containing high concentrations of nickel (Pearson *et al.*, 2002b).

The approach for selection of COC in the aquatic environment is illustrated in Figure 5-2. Although this process will potentially identify separate COC for water and sediments, the final list of COC for the aquatic problem formulation includes all metals selected from either sediment or surface water. This may be particularly important for VECs that may experience exposure to both water and sediment (*e.g.*, benthic-feeding fish), since screening criteria for sediment are developed only for exposure to benthic invertebrates (not fish).

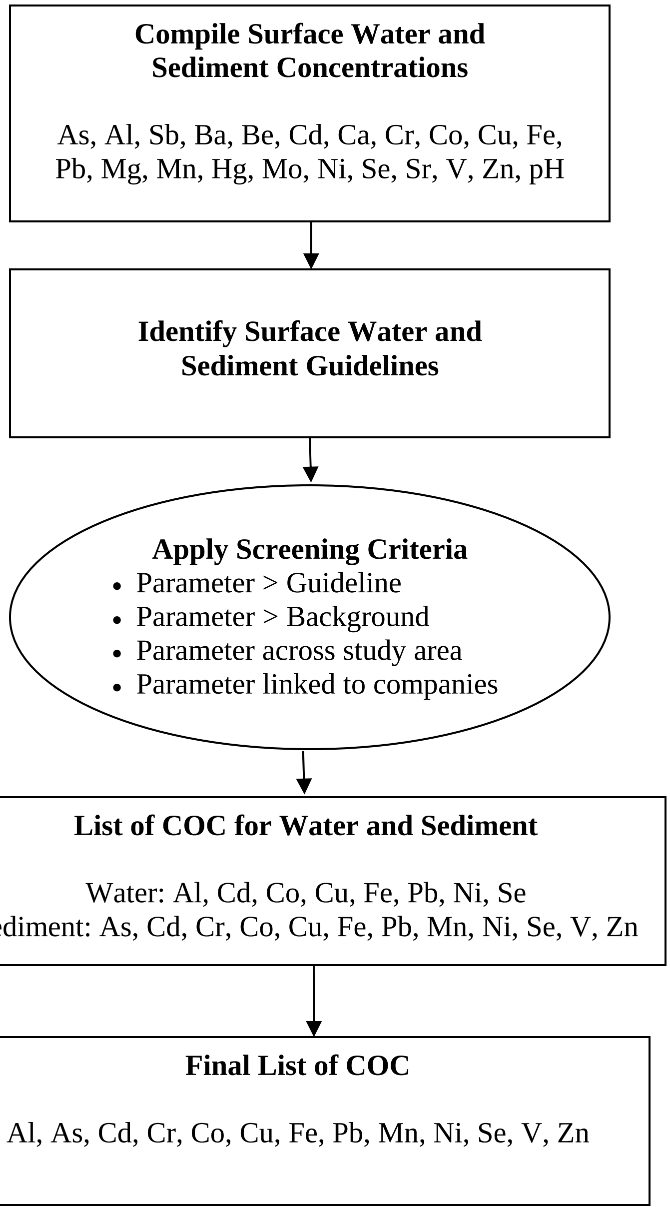



**Figure 5-2 Process for the Identification of COC in the Aquatic Environment**

The COC selection process includes the following four steps that are described in further detail in sections 5.5 and 5.6 for surface water and sediment, respecitively.

Step 1: Compile surface water and sediment data for 21 inorganic parameters and pH;

Step 2: Select appropriate environmental quality guidelines for each parameter;

Step 3: Apply screening criteria:

- Compare maximum concentrations to guidelines;
- Compare environmental levels to regional background levels;
- Characterize the distribution of any exceedences across the study area;
- Determine if the metal has been scientifically linked to smelting activities; and,

Step 4: Prepare a summary of the rationale for the final list of COC.

Since the basis of the current aquatic problem formulation is the Sudbury Soils Study, the elements initially considered for the screening process were those that were analyzed in soils collected throughout the Greater Sudbury area. The list of metals and metalloids (referred to collectively as metals) selected by the Technical Committee include:

Al, As, Sb, Ba, Be, Cd, Ca, Cr, Co, Cu, Fe, Pb, Mg, Mn, Mo, Ni, Se, Sr, V and Zn.

In addition to these 20 parameters, levels of mercury in surface water and sediment were also considered for the aquatic problem formulation. Mercury was added because there is general concern regarding mercury levels in fish, and there are ongoing monitoring and research programs addressing this chemical. As directed by the Technical Committee, pH is not considered a COC. Rather, surface water and sediment pH is evaluated as a modifying factor for metal toxicity (Section 5.8).

Concentrations measured in surface water or sediment were compared to relevant environmental quality criteria, guidelines, objectives, or other benchmarks (collectively referred to as "guidelines" hereafter). If exceedences occurred, the regional background concentrations were considered to determine if the exceedence was atypical of the area and likely a result of anthropogenic activities, or simply a result of the local geology and chemistry. It was also relevant to characterize the spatial distribution of the exceedence, to determine whether or not environmental concentrations exceeded guidelines throughout the study area or whether elevated levels were restricted to localized zones. Exceedences of a chemical-specific screening criterion in 5% or less of lakes for which sampling data were available for that chemical was not considered to be representative of conditions throughout the study area. Therefore, metals that were only found above screening criteria in 5% or less of the sampled lakes were not retained as COC. Finally, since the objective of the ERA is to characterize risks resulting from exposure to metals associated with local smelting activities, only those metals that are associated with airborne smelter particulate emissions were retained as COC.

## 5.5 Selection of COC in Surface Water

The four-step process identified in Figure 5-2 was followed to screen COC in surface water in Sections – 5.5.1 through 5.5.4

### 5.5.1 Compilation of Surface Water Data

To characterize the environmental distribution of the metals under consideration, data were compiled that describe the most recent concentrations in the surface water of numerous lakes throughout the Greater Sudbury area. Data obtained as part of the SES (Sudbury Environmental Study) Extensive Monitoring program (Co-Op, unpublished a), the Urban Lake Water Recovery program (1990 to 2003) (Co-Op, 2004) and annual averages provided by researchers at Laurentian University (Co-Op, unpublished b) formed the basis for much of the surface water characterization. The locations of these lakes are provided in Figure 5-3. The SES Extensive Monitoring lakes have been sampled once a year since the early 1980s to monitor acidification and levels of metals associated with smelter emissions. As can be seen in Figure 5-3, many of these lakes are located northeast of the study area boundary for the terrestrial and aquatic ERAs. With a total of 45 lakes included in this monitoring program, 11 are located within the City of Greater Sudbury (Co-Op, 2004). The Urban Lake Water Recovery study monitors changes in water chemistry collected from 31 lakes in the core area of the City of Greater Sudbury. Data from this study were collected by the Ministry of the Environment (MOE) in mid-summer 2003. Samples are non-volume-weighted, tygon tube composites taken from the epilimnion and metalimnion within a deep basin of each lake, and represent total metal concentrations in water (Co-Op, 2004).

To supplement these data, the published scientific literature was reviewed to locate data for lakes not included in these other programs. In total, metal concentrations for 85 lakes were considered during the COC screening process. The maximum concentration for each metal was selected and used for comparison to appropriate screening criteria.

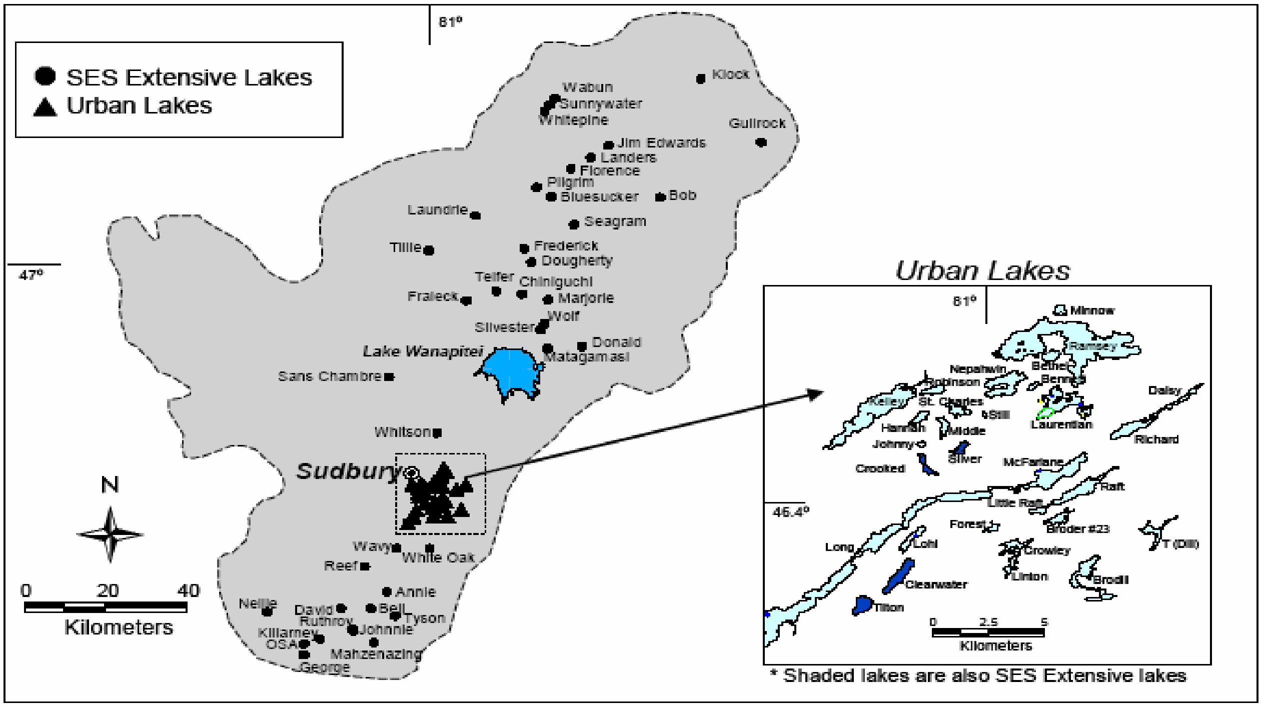



**Figure 5-3 Sudbury Lakes Included in the SES Extensive Monitoring Program and the Urban Lake Water Recovery Study (Keller *et al*., 2004)**

### 5.5.2 Identification of Surface Water Guidelines

The Provincial Water Quality Objectives (MOEE, 1994) and the Canadian Environmental Quality Guidelines (CCME, 2002) were used preferentially to screen COC (Table 5.1). These guidelines are set to maintain a level of water quality that is protective of all forms of aquatic life during all life cycles for chronic exposure durations (MOEE, 1994). When MOE or CCME guidelines were not available, guidelines recommended by other agencies were considered.

**Table 5.1 Screening Criteria for Chemicals of Concern in Surface Water (µg total metal/L water)**

| Parameter | MOE PWQO | CCME | U.S. EPA CCC[b] | RAIS/Other |
|---|---|---|---|---|
| Aluminium | **15 at pH 4.5 - 5.5<br>see note a for pH >5.5 – 6.5<br>75 at pH >6.5 – 9.0** | 5 to 100 | - | 70 |
| Antimony | **20** | - | - | 31 |
| Arsenic | **5** | 5.0 | 150[c] | 5 |
| Barium | - | - | - | **1,000[e]** |
| Beryllium | **11 at $CaCO_3$ <75 mg/L<br>1,100 at $CaCO_3$ >75 mg/L** | - | - | 0.66 |
| Cadmium | 0.1 at $CaCO_3$ <100 mg/L<br>0.5 at $CaCO_3$ >100 mg/L | **0.017** | 0.25[c] | 0.013 |
| Calcium | - | - | - | **116,000 [f]** |
| Chromium VI | **1** | 1.0 | 11 | 0.266 |
| Chromium III | **8.9** | 8.9 | 74[c] | 8.44 |
| Cobalt | **0.9** | - | - | 3.98 |
| Copper | **1 at $CaCO_3$ <20 mg/L<br>5 at $CaCO_3$ >20 mg/L** | 2 to 4 | 9.0[c] | 0.205 |
| Iron | **300** | 300 | 1,000 | 10 |
| Lead | **1 at $CaCO_3$ <30 mg/L<br>3 at $CaCO_3$ 30 - 80 mg/L<br>5 at $CaCO_3$ >80 mg/L** | 1 to 7 | 2.5[c] | 0.35 |
| Magnesium | - | - | - | **82,000 [f]** |
| Manganese | - | - | - | **638[e, g]** |
| Mercury | 0.2 | **0.1** | 0.77 | |
| Molybdenum | **40** | 73 | - | 73 |
| Nickel | **25** | 25 to 150 | 52[c] | 5 |
| Selenium | 100 | **1.0** | 5.0[d] | 1 |
| Strontium | - | - | - | **1,500** |
| Vanadium | **6** | - | - | 19 |
| Zinc | **20** | 30 | 120[c] | 20 |
| pH | 6.5 to 8.5 | 6.5 to 9.0 | - | - |

* Bolded values in grey scale were selected as screening guidelines for the current assessment.

[a] No condition should be permitted that would increase the acid soluble inorganic aluminium concentration in clay-free samples to more than 10% above natural background levels (MOEE, 1994).

[b] U.S. EPA Water Quality Criteria for metals are given as dissolved concentrations (U.S. EPA, 2002).

[c] Criterion is expressed as a function of hardness (mg/L) within the water column. The value given corresponds to a hardness of 100 mg/L. For waters with a different hardness, the criterion should be re-calculated (see U.S. EPA, 2002). This was not done since PWQOs were available for these metals.

[d] Criterion is expressed in total recoverable metal in the water column.

[e] BC Approved and Working Water Quality Guideline for Freshwater Aquatic Life (BC MELP, 1998a,b).

[f] Guideline is an LCV (Lowest Chronic Value) for Daphnids (Suter and Tsao, 1996).

[g] The guideline for manganese is based on water hardness (4.4 (hardness) + 605) where hardness was assumed = 7.5 mg/L as $CaCO_3$ for all lakes.

When both a PWQO and a CCME Guideline were available, to be conservative, the lower of the two values was selected as the screening guideline. For metals where neither were available, the most appropriate screening guideline available from the U.S. EPA, RAIS (Risk Assessment Information System, a database of screening guidelines from several U.S. and international jurisdictions), or other relevant sources was selected.

### 5.5.3 Application of Surface Water Screening Criteria

There are four screening criteria applied and discussed in this section:

- Compare maximum concentrations to guidelines;
- Compare environmental levels to regional background levels;
- Characterize the distribution of any exceedences across the study area; and
- Determine if the metal has been scientifically linked to smelting activities.

Data collected from 123 lakes were used to characterize regional background concentrations of COC within the Sudbury area (Table 5.2). Four of the reference lakes (Barlow, Big Marsh, Birch and Waubamac) are located within a relatively close range of the Copper Cliff smelter (39 to 50 km) but are not considered to be influenced by mining and smelting activities (Pyle *et al*., 2005). The remaining lakes are located at distances greater than 90 km from the City of Greater Sudbury, therefore, levels of COC in water may be considered representative of regional background conditions (Couture and Kumar, 2003; Couture and Rajotte, 2003; Chen *et al*., 2001; Borgmann *et al*., 2001b; Conroy *et al*., 1978).

Surface water concentrations obtained from the Pyle *et al*. (2005) study and presented in Table 5.2 represent the average of three samples taken at three depths (1.5 and 10 m). Concentrations from the Couture and Kumar (2003) study are the average of two samples taken at a depth of approximately 4 m. Values obtained from the Couture and Rajotte (2003) study represent average values taken from numerous studies. Details of the sampling conducted by Chen *et al*. (2001) were not provided.

Borgmann *et al*. (2001b) studied 12 lakes within 6 to 154 km of the smelter stacks at Copper Cliff as part of a study to investigate the impacts of atmospherically deposited metals on aquatic ecosystems. In order to specifically address the effects of metals, the survey only included lakes with surface waters with circumneutral pH (6.7 to 7.7). Four of these lakes (Tomiko, Restoule, Nosbonsing and Talon) were selected as reference lakes, as they were located considerable distances of 94, 107, 144, and 154 km, respectively, from the smelter. Water samples were obtained from two locations: 1m below the surface and 1 m from the bottom using a van Dorn sampler. Values taken from this study and presented in Table 5.2 represent the average of these samples (Borgmann *et al*., 2001b).

Conroy *et al*. (1978) conducted an extensive monitoring of 209 lakes in the Greater Sudbury area between 1974 and 1976. Water samples were collected in all 209 lakes at 1m from the surface and 1m from the bottom of the lake between 1974 and 1976. Surface water samples were collected by hand, while water samples obtained 1m from the bottom were sampled with a van Dorn bottle. To be consistent with Borgmann *et al*. (2001b), lakes located a minium of 90 km from Sudbury (116 lakes) were considered to be reference lakes and included within the current derivation of regional background concentrations. Values taken from this study and presented in Table 5.2 represent the average of one to seven samples, depending on the individual lake.

The Ontario MOE does not have guidelines for estimating background concentrations in water. However, the guidelines for estimating a site-specific background concentration in sediments recommend using the mean in the calculation. For the screening of COC in surface water, the regional background concentrations were not used as screening criteria but only to put certain exceedences into context.

Including these values in the screening process can indicate whether exceedences may be the result of the natural chemistry of the area. In cases where naturally-elevated levels of certain metals exist, consideration of these concentrations can show whether or not the screening guidelines selected are appropriate to indicate smelter emission impacts.

**Table 5.2a Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Anima Nipissing[f] | Anvil[f] | Aston[f] | Bain[f] | Banks[f] | Bark[f] | Barlow[a] | Barnet[f] | Bass[f] | Basswood[f] | Bernard[f] | Big Marsh[a] |
| Aluminium | - | - | - | - | - | - | 238.7 | - | - | - | - | 128.7 |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | 0.7 |
| Barium | - | - | - | - | - | - | 12.7 | - | - | - | - | 10.3 |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | - | - | - | - | - | - | - |
| Calcium | 5,000 | 3,000 | 5,000 | 4,000 | 3,000 | 5,000 | 2,899 | 5,000 | 3,000 | 4,000 | 4,000 | 2,183 |
| Chromium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | - | - | - | - | - | - | - |
| Copper | 4 | 6 | 7 | 5 | 3 | 11 | 4.3 | 8 | 3 | 2 | 5 | 3.7 |
| Iron | 10 | 55 | 37 | 52 | 30 | 86 | 389 | 30 | 58 | 5 | 22 | 421 |
| Lead | 2 | 2 | 2 | 2 | 2 | 2 | - | 3 | 2 | 2 | 2 | - |
| Magnesium | 1,000 | 1,000 | 2,000 | 2,000 | 1,000 | 1,000 | 1,314 | 1,000 | 1,000 | <1,000 | 1,000 | 1,155 |
| Manganese | - | - | - | - | - | - | 521 | - | - | - | - | 598 |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | <1 | 1 | 1 | 2 | 1 | 3 | 9.7 | 3 | 3 | <1.0 | 2 | 9 |
| Selenium | - | - | - | - | - | - | - | - | - | - | - | - |
| Strontium | - | - | - | - | - | - | 20 | - | - | - | - | 16.3 |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 4 | 19 | 2 | 2 | 6 | 6 | 10.3 | 5 | 7 | 2 | 3 | 7.3 |
| pH | 6.8 | 6.3 | 6.8 | 6.8 | 5.8 | 6.9 | 6.5 | 6.9 | 6.4 | 6.8 | 7 | 6 |

**Table 5.2b Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Bigwind[f] | Bigwood[f] | Birch[a] | Blackwater[f] | Bob[f] | Bragh[f] | Brule[f] | Buck[f] | Cavell[f] | Cecebe[f] | Chateau[f] | Chief[f] |
| Aluminium | - | - | 172.7 | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | - |
| Barium | - | - | 14.3 | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | - | - | - | - | - | - | - |
| Calcium | 3,000 | 4,000 | 2,129 | 5,000 | 3,000 | 5,000 | 3,000 | 3,000 | 5,000 | 4,000 | 5,000 | 4,000 |
| Chromium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | - | - | - | - | - | - | - |
| Copper | 3 | 18 | 2 | 4 | 3 | 2 | 2 | 2 | 6 | 58 | 3 | 3 |
| Iron | 17 | 118 | 714 | 144 | 92 | 61 | 69 | 133 | 109 | 175 | 29 | 49 |
| Lead | 2 | 4 | - | 2 | 2 | 2 | 3 | 2 | 2 | 3 | <3 | 2 |
| Magnesium | 1,000 | 1,000 | 830 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 2,000 | 1,000 |
| Manganese | - | - | 1,726 | - | - | - | - | - | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 2 | 8 | 7 | 2 | 6 | 2 | 2 | 2 | 3 | 3 | 2 | 3 |
| Selenium | - | - | - | - | - | - | - | - | | - | - | - |
| Strontium | - | - | 18.3 | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 6 | 16 | 7 | 2 | 12 | 3 | 6 | 7 | 4 | 9 | 2 | 15 |
| pH | 6.7 | 5.8 | 6.1 | 6.8 | 4.6 | 7.1 | 6.6 | 6.3 | 7.1 | 7 | 7.3 | 5.7 |

**Table 5.2c Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Cooke[f] | Diamond[f] | Eagle[f] | East Bull[f] | Fanny[f] | Flack[f] | Florence[f] | Foys[f] | Frables[f] | Geneva[c] | Gullrock[f] | Halfway[b] |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | - | - | - | - | - | - | 0.01 |
| Calcium | 5,000 | 4,000 | 3,000 | 3,000 | 4,000 | 4,000 | 4,000 | 4,000 | 3,000 | - | 6,000 | - |
| Chromium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | - | - | - | - | - | - | - |
| Copper | 2 | 13 | 6 | 29 | 10 | 19 | 6 | 5 | 4 | - | 4 | 3.4 |
| Iron | 120 | 27 | 45 | 27 | 67 | 18 | 41 | 30 | 53 | - | 28 | - |
| Lead | 2 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | - | 2 | - |
| Magnesium | 2,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 2,000 | 1,000 | - | 1,000 | - |
| Manganese | - | - | - | - | - | - | - | - | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | 0.0042 | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | <1 | 4 | 4 | 4 | 4 | 3 | 7 | 2 | 5 | - | 14 | - |
| Selenium | - | - | - | - | - | - | - | - | - | 0.096 | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | <1 | 11 | 7 | 11 | 16 | 9 | 9 | <2 | 7 | - | 67 | 36.7 |
| pH | 6.9 | 6 | 6.9 | 6.8 | 6.6 | 6.8 | 4.5 | 7.1 | 6.8 | - | 4.6 | - |

**Table 5.2d Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Hammond[f] | Horn[f] | Island[f] | Jerry[f] | Jim Edward[f] | Jumping Cariboo[f] | Kagawon [f] | Kasakanta[f] | Kindiogami[f] | Kirby[f] | Klock[f] | Kokoko[f] |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | - | - | - | - | - | - | - |
| Calcium | 11,000 | 2,000 | 2,000 | 3,000 | 3,000 | 7,000 | 36,000 | 10,000 | 5,000 | 3,000 | 3,000 | 9,000 |
| Chromium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | - | - | - | - | - | - | - |
| Copper | 19 | 3 | 18 | 4 | 5 | 9 | 16 | 4 | 2 | 2 | 2 | 2 |
| Iron | 31 | 54 | 92 | 24 | 38 | 23 | 17 | 88 | 16 | 44 | 42 | 11 |
| Lead | 3 | 2 | 3 | 4 | 3 | 6 | 6 | 2 | 2 | 2 | 2 | 2 |
| Magnesium | 3,000 | 2,000 | 1,000 | 1,000 | 1,000 | 2,000 | 14,000 | 2,000 | 1,000 | 1,000 | 1,000 | 2,000 |
| Manganese | - | - | - | - | - | - | - | - | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 4 | 2 | 15 | 4 | 6 | 4 | 4 | 1 | 1 | 2 | 3 | 1 |
| Selenium | - | - | - | - | - | - | - | - | - | - | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 15 | 10 | 14 | 13 | 13 | 13 | 9 | <1 | 2 | 2 | 8 | 1 |
| pH | 7.5 | 5.1 | 5.8 | 4.9 | 4.6 | 7.2 | 8.4 | 7.5 | 7.2 | 6.6 | 4.7 | 7.2 |

**Table 5.2e Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | La Muir[f] | Lac aux Sables[f] | Lady Evelyn[f] | Lady Sydney[f] | Laundric[f] | Leonard[f] | Lepha[f] | Lorraine[f] | Lost[f] | Louisa[f] | Low Water[f] | Madawanson[f] |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | - | - | - | - | - | - | - |
| Calcium | 3,000 | 4,000 | 4,000 | 4,000 | 4,000 | 3,000 | 3,000 | 8,000 | 3,000 | 3,000 | 4,000 | 3,000 |
| Chromium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | - | - | - | - | - | - | - |
| Copper | 11 | 12 | 6 | 3 | 6 | 3 | 3 | 8 | 3 | 3 | 8 | 2 |
| Iron | 36 | 38 | 35 | 26 | 114 | 69 | 23 | 31 | 270 | 15 | 153 | 29 |
| Lead | 3 | 3 | 3 | 2 | 4 | 2 | 2 | 3 | 2 | 3 | 3 | 2 |
| Magnesium | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 2,000 | 1,000 | 1,000 | 1,000 | 1,000 |
| Manganese | - | - | - | - | - | - | - | - | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 2 | 4 | 3 | <1 | 6 | 3 | 2 | 3 | 2 | 2 | 3 | 1 |
| Selenium | - | - | - | - | - | - | - | - | - | - | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 2 | 10 | 8 | 17 | 16 | 14 | 10 | 8 | 7 | 6 | 9 | 2 |
| pH | 6.9 | 6.4 | 6.5 | 5.9 | 4.7 | 5.8 | 6.2 | 7.2 | 5.9 | 6.5 | 6.2 | 6.7 |

**Table 5.2f Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Midlothian[f]** | **Morrison[f]** | **Mountain[f]** | **Mountain[f]** | **Mozhabong[f]** | **Naiscoot[f]** | **Nine Mile[f]** | **North Grace[f]** | **Nosbonsing[e]** | **Obakia[f]** | **Obushkong[f]** | **Opikinimika[f]** |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | 0.45 | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | - | - | - | 0.43 | - | - | - |
| Calcium | 8,000 | 3,000 | 10,000 | 9,000 | 3,000 | 3,000 | 3,000 | 3,000 | 6,000[f] | 6,000 | 6,000 | 12,000 |
| Chromium | - | - | - | - | - | - | - | - | 1.15 | - | - | - |
| Cobalt | - | - | - | - | - | - | - | - | 0.015 | - | - | - |
| Copper | 7 | 7 | 16 | 8 | 26 | 9 | 3 | 3 | 12.4[g] | 16 | 3 | 34 |
| Iron | 15 | 100 | 98 | 22 | 25 | 166 | 122 | 25 | 1,193[f] | 18 | 59 | 82 |
| Lead | 3 | 2 | 4 | 3 | 3 | 3 | 2 | 3 | 1.6[g] | 3 | 2 | 3 |
| Magnesium | 2,000 | 1,000 | 2,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 2,000[f] | 1,000 | 2,000 | 2,000 |
| Manganese | - | - | - | - | - | - | - | - | 680.5 | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 4 | 3 | 3 | 3 | 4 | 3 | 3 | 2 | 2.3[g] | 4 | 2 | 21 |
| Selenium | - | - | - | - | | - | - | - | 0.61 | - | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 11 | 3 | 9 | 21 | 15 | 11 | 2 | 8 | 8.3[g] | 13 | 2 | 8 |
| pH | 7.5 | 6.7 | 7.2 | 7.2 | 6.7 | 6.2 | 6.6 | 6.2 | 7.4[g] | 6.7 | 6.9 | 7.6 |

## Table 5.2g Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Proulx**[f] | **Rabbit**[f] | **Rawhide**[f] | **Red Cedar**[f] | **Red Pine**[f] | **Restoule**[e] | **Rib**[f] | **Rice**[f] | **Riggar**[f] | **Round**[f] | **Rumsay**[f] | **Schist**[f] |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | 0.53 | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | 0.088 | - | - | - | - | - | - |
| Calcium | 4,000 | 10,000 | 4,000 | 8,000 | 3,000 | 4,000[f] | 8,000 | 5,000 | 4,000 | 2,000 | 4,000 | 9,000 |
| Chromium | - | - | - | - | - | 1.26 | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | 0.028 | - | - | - | - | - | - |
| Copper | 2 | 7 | 1 | 24 | 5 | 5.1[g] | 13 | 7 | 2 | 14 | 3 | 7 |
| Iron | 80 | 52 | 8 | 107 | 25 | 101[f] | 29 | 58 | 137 | 193 | 106 | 47 |
| Lead | 3 | 4 | 2 | 9 | 2 | 1.7[g] | 3 | 1 | 3 | 4 | 2 | 2 |
| Magnesium | 2,000 | 2,000 | 1,000 | 2,000 | 1,000 | 1,000[f] | 2,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 |
| Manganese | - | - | - | - | - | 27.9 | - | - | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 2 | 4 | <2 | 4 | 2 | 2.5[g] | 3 | <1 | 2 | 14 | 2 | 3 |
| Selenium | - | - | - | - | - | 0.35 | - | - | - | - | - | |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 2 | 7 | <2 | 18 | 8 | 9.1[g] | 16 | 2 | 2 | 28 | 4 | 4 |
| pH | 6.9 | 7.2 | 6.8 | 7 | 6.3 | 6.6[g] | 7 | 7.2 | 7 | 4.9 | 6.8 | 7.2 |

**Table 5.2h Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (μg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Shack[f] | Shawanaga[f] | Shining Tree[f] | Shoofl[f] | Skeleton[f] | Smith[f] | Smoke[f] | Smoothwater[f] | Stull[f] | Sugar[f] | Sunnywater[f] | Talon[e] |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | 0.3 |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | - | - | - | - | - | - | 0.034 |
| Calcium | 6,000 | 4,000 | 13,000 | 32,000 | 4,000 | 3,000 | 3,000 | 4,000 | 4,000 | 4,000 | 2,000 | 6,000[f] |
| Chromium | - | - | - | - | - | - | - | - | - | - | - | 1.38 |
| Cobalt | - | - | - | - | - | - | - | - | - | - | - | 0.018 |
| Copper | 4 | 39 | 3 | 6 | 3 | 4 | 4 | 2 | 12 | 4 | 6 | 14.9[g] |
| Iron | 59 | 135 | 37 | 5 | 7 | 30 | 39 | 13 | 29 | 15 | 29 | 100[f] |
| Lead | 3 | 3 | 2 | 5 | 2 | 2 | 3 | 2 | 3 | 2 | 4 | 5.5[g] |
| Magnesium | 1,000 | 1,000 | 3,000 | 5,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000 | 1,000[f] |
| Manganese | - | - | - | - | - | - | - | - | - | - | - | 4.6 |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 3 | 4 | 2 | 3 | 2 | 2 | 2 | 1 | 3 | 1 | 4 | 2.2[g] |
| Selenium | - | - | - | - | - | - | - | - | - | - | - | 0.19 |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 3 | 16 | 2 | 11 | 5 | 10 | 3 | 9 | 13 | 5 | 20 | 16.5[g] |
| pH | 6.9 | 6.4 | 7.4 | 8.3 | 6.8 | 5.4 | 6.9 | 5.8 | 5.9 | 6.5 | 4.4 | 7.0[g] |

**Table 5.2i Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Tatachikapika[f]** | **Temegami[f]** | **Tenfish[f]** | **Timber[f]** | **Tim[f]** | **Tomiko[e]** | **Trethewey[f]** | **Trout [f]** | **Trout[f]** | **Valin[f]** | **Wabun[f]** | **Waonga[f]** |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | 0.56 | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | - | - | - | - | - | 0.073 | - | - | - | - | - | - |
| Calcium | 7,000 | 8,000 | 4,000 | 4,000 | 2,000 | 4,000[f] | 3,000 | 3,000 | 6,000 | 3,000 | 3,000 | 20,000 |
| Chromium | - | - | - | - | - | 1.66 | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | 0.058 | - | - | - | - | - | - |
| Copper | 7 | 13 | 7 | 8 | 2 | 6.3[g] | 3 | 10 | 16 | 19 | 5 | 2 |
| Iron | 84 | 14 | 16 | 73 | 92 | 123[f] | 30 | 47 | 28 | 105 | 40 | 11 |
| Lead | 3 | 3 | 3 | 4 | 3 | 1.6[g] | 2 | 3 | 4 | 5 | 2 | 1 |
| Magnesium | 1,000 | 2,000 | 1,000 | 1,000 | 1,000 | 1,000[f] | 1,000 | 1,000 | 2,000 | 1,000 | <1,000 | 5,000 |
| Manganese | - | - | - | - | - | 35.9 | - | - | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 3 | 3 | 3 | 4 | 2 | 3[g] | 1 | 12 | 3 | 4 | 3 | 2 |
| Selenium | - | - | - | - | - | 1.72 | - | - | - | - | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 12 | 12 | 5 | 20 | 6 | 18.4[g] | 7 | 13 | 66 | 16 | 11 | 2 |
| pH | 7 | 6.9 | 6.7 | 6.5 | 6.4 | 6.6[g] | 5.6 | 6 | 7.1 | 6.3 | 4.7 | 8.1 |

**Table 5.2j Surface Water Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg total metal/L water)**

| Parameter | Lake | | | |
|---|---|---|---|---|
| | Waubamac[a] | Welcome[f] | White Owl[f] | Mean[j] |
| Aluminium | 80 | - | - | **155.0** |
| Antimony | - | - | - | - |
| Arsenic | - | - | - | **0.51** |
| Barium | 10 | - | - | **11.8** |
| Beryllium | - | - | - | - |
| Cadmium | - | - | - | **0.13** |
| Calcium | 4,203 | 6,000 | 5,000 | **5632** |
| Chromium | - | - | - | **1.4** |
| Cobalt | - | - | - | **0.03** |
| Copper | 2 | 15 | 3 | **7.9** |
| Iron | 412 | 42 | 53 | **80.2** |
| Lead | - | 3 | 2 | **2.8** |
| Magnesium | 1,811 | 1,000 | 1,000 | **1526** |
| Manganese | 344 | - | - | **492.2** |
| Mercury | - | - | - | **0.0042** |
| Molybdenum | - | - | - | - |
| Nickel | 8 | 3 | 2 | **3.4** |
| Selenium | 0.6 | - | - | **0.59** |
| Strontium | 28.3 | - | - | **20.7** |
| Vanadium | - | - | - | - |
| Zinc | 5.7 | 10 | 3 | **9.7** |
| pH | 6.8 | 6.6 | 7 | - |

\- indicates data were not available

[a] Values are total concentrations collected in 2001 (Pyle *et al*., 2005).

[b] Values are assumed to be total concentrations collected from 1997 to 2001 (Couture and Kumar, 2003; Couture and Rajotte, 2003).

[c] Values are total concentrations collected in 1996-1997 (Chen *et al*., 2001).

[d] If only one value available, then it is presented also in this column.

[e] Values are the average of samples collected by van Dorn sampler at 1m below the surface and 1m above the bottom at each sampling site in 1998, and samples collected in water 1m off the bottom at each station in 1996 (Borgmann *et a*l., 1998, 2001b). Samples under the detection limit were exluded as the detection limits were not provided.

[f] Values are mean values determined by Conroy *et al.* (1978). Values are based on the average of one to seven samples, depending on the individual lake, sampled from 1974 to 1976 1m below the surface and 1m from the bottom by hand and with a van Dorn bottle, respectively. All of the water samples were under the detection limit (<1 µg/L) for cadmium.

[g] Value is the average of total metal concentrations in water 1m below the surface and 1m above the bottom obtained between 1974 to 1976 and 1996 to 1998 by Conroy *et al*., 1978 and Borgmann *et al*., 2001b, respectively.

[i] Several copper concentrations sampled in 1996 were anomalously high when compared to other stations from the same lake (Borgmann *et al*., 2001b). As the high levels were believed to result from contamination these values were excluded when calculating the mean copper concentration.

[j] To calculate the mean, half of the detection limit was used for those concentrations expressed as a less than value.

The average concentration of four metals (aluminium, cadmium, chromium, and copper) in the surface water of the reference lakes exceeded the corresponding screening guidelines. This may be indicative of naturally-elevated levels of these metals in the Sudbury area.

The maximum concentration for each metal, detected in any lake, was compared to the corresponding guideline and Sudbury background levels (Table 5.3). The complete screening for each of 76 individual lakes is presented in Table 5.4.

**Table 5.3 Screening of Maximum Concentrations of Metals in Surface Water (μg total metal/L water)**

| Parameter | Maximum Concentration | Screening Guideline | Sudbury Regional Background |
|---|---|---|---|
| **Aluminium** | **401** | **75** | 155 |
| Antimony | - | **20** | - |
| Arsenic | 3.3 | **5** | 0.51 |
| Barium | 101 | **1,000** | 12 |
| Beryllium | 0.08 | **11** | - |
| **Cadmium** | **0.99** | **0.017** | 0.13 |
| **Calcium** | **274,000** | **116,000** | 5,632 |
| Chromium | 0.85 | **1.0** | 1.4 |
| **Cobalt** | **14** | **0.9** | 0.03 |
| **Copper** | **48.4** | **1, 5** | 7.9 |
| **Iron** | **781** | **300** | 80 |
| **Lead** | **22.16** | **1, 3, 5** | 2.8 |
| Magnesium | 43,500 | **82,000** | 1,526 |
| **Manganese** | **1,020** | **638** | 492 |
| Mercury | 0.04 | **0.1** | 0.0042 |
| Molybdenum | 0.95 | **40** | - |
| **Nickel** | **338** | **25** | 3.4 |
| **Selenium** | **5.8** | **1.0** | 0.59 |
| Strontium | 425 | **1,500** | 21 |
| Vanadium | 1.00 | **6** | - |
| **Zinc** | **36** | **20** | 9.7 |
| | | | |

* Bolded values were selected as screening guidelines for the current assessment. Bolded values in grey scale exceeded the corresponding guidline.

\- Indicates no data were available

The maximum concentration of 11 metals (aluminium, cadmium, calcium, cobalt, copper, iron, lead, manganese, nickel, selenium and zinc) in surface water exceeded the screening guidelines (Table 5.3). Concentrations of antimony in surface water were not available, and therefore, screening for this metal could not be completed. Antimony was not retained as a COC for soils. Therefore, it is recommended that Sb not be considered a COC for the aquatic ERA; however, future sampling of surface water should include an analysis for Sb to allow for proper screening.

*This page left blank intentionally*

**Table 5.4 Screening of COC in Surface Water for 85 Lakes**

| Lake | Sampling Date | pH | Hardness $CaCO_3$ (mg/L) | Al (μg/L) | As (μg/L) | Sb (μg/L) | Ba (μg/L) | Be (μg/L) | Cd (μg/L) | Ca (mg/L) | Cr (μg/L) | Co (μg/L) | Cu (μg/L) | Fe (μg/L) | Pb (μg/L) | Mg (mg/L) | Mn (μg/L) | Hg (μg/L) | Mo (μg/L) | Ni (μg/L) | Se (μg/L) | Sr (μg/L) | V (μg/L) | Zn (μg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Screening Criterion** | | | | 75 | 5 | 20 | 1,000 | 11 | 0.017 | 116 | 1 | 0.9 | 1 | 300 | 1 | 82 | 638 | 0.1 | 40 | 25 | 1 | 1,500 | 6 | 20 |
| **SES Intensive Lakes** | | | | | | | | | | | | | | | | | | | | | | | | |
| Clearwater | 2002[a] | 6.1 | - | 26.2 | - | - | - | - | - | 4.25 | - | - | **10.15** | 49.15 | - | 1.07 | 41.32 | - | - | **77.85** | - | - | - | 13.23 |
| Daisy | 2002[a] | 6.433 | 11.1 | 36.1 | - | - | 17.05 | 0.009 | **0.12** | 2.46 | *0.5* | **1.28** | **12.44** | 135.13 | ***5.5*** | 1.21 | 31.18 | - | *0.75* | **90.28** | - | 13.2 | 0.11 | 9.14 |
| Hannah | 2002[a] | 7.325 | - | 16.1 | - | - | - | - | - | 10.25 | - | - | **27.3** | 81.25 | - | 3.42 | 38.13 | - | - | **139.50** | - | - | - | 4.84 |
| Lohi | 2002[a] | 6.411 | - | 20.2 | - | - | - | - | - | 3.76 | - | - | **11.68** | 99.78 | - | 1.21 | 34.35 | - | - | **65.75** | - | - | - | 10.21 |
| Middle | 2002[a] | 7.093 | - | 18.0 | - | - | - | - | - | 8.75 | - | - | **25.57** | 48.35 | - | 2.97 | 32.42 | - | - | **134.50** | - | - | - | 9.27 |
| Swan | 2002[a] | 5.444 | 11.4 | 32.2 | - | - | 14.717 | 0.019 | **0.34** | 3.34 | 0.06 | **2.84** | **11.48** | 123.71 | **22.16** | 0.65 | 59.83 | - | *0.75* | **94.72** | - | 15.37 | 0.10 | 11.53 |
| **SES Extensive Lakes** | | | | | | | | | | | | | | | | | | | | | | | | |
| Annie | 2002[b] | 6.29 | 7.4 | 29.4 | - | - | 8.46 | 0.011 | **0.26** | 1.96 | *0.5* | *0.75* | **2.53** | 10.1 | **4** | 0.69 | 20.6 | - | *0.75* | 5 | - | 10 | *0.5* | 3.05 |
| Bell | 2002[b] | 6.25 | 7.8 | 54.5 | - | - | 11.6 | 0.021 | **0.24** | 1.96 | 0.051 | 0.46 | **2.3** | - | ***5.5*** | 0.69 | 39.1 | - | *0.75* | 6.79 | - | 11.7 | *0.5* | 5.77 |
| Blue Sucker | 2002[b] | 5.46 | 6.2 | 66 | - | - | 40.2 | 0.028 | ***0.4*** | 1.52 | *0.5* | 0.71 | 0.52 | 14.3 | **1.68** | 0.51 | 67.4 | - | *0.75* | 2.48 | - | 12 | *0.5* | 2.76 |
| Bob | 2002[b] | 5.25 | 5 | **90.1** | - | - | 22.8 | 0.036 | **0.036** | 1.2 | 0.051 | 0.86 | 0.49 | 79.2 | **5.16** | 0.45 | 77.2 | - | *0.75* | 2.94 | - | 11.7 | *0.5* | 4.07 |
| Chiniguchi | 2002[b] | 5.48 | 7.2 | 57.7 | - | - | 41.6 | 0.035 | **0.22** | 1.9 | 0.25 | 0.57 | **1.37** | 13.2 | **2.49** | 0.59 | 72.4 | - | *0.75* | 6.77 | - | 13.3 | *0.5* | 3.86 |
| David | 2002[b] | 5.11 | 4.4 | 27.7 | - | - | 8.4 | 0.016 | **0.29** | 1.18 | *0.5* | 0.16 | 0.65 | 10.1 | ***5.5*** | 0.38 | 45.5 | - | *0.75* | 3.92 | - | 6.47 | *0.5* | 5.71 |
| Donald | 2002[b] | 5.28 | 7.6 | 40.2 | - | - | 11.4 | 0.026 | **0.26** | 2.12 | 0.02 | 0.67 | **1.29** | 0.95 | ***5.5*** | 0.5 | 82 | - | *0.75* | 8.96 | - | 13.6 | *0.5* | 4.71 |
| Dougherty | 2002[b] | 4.87 | 6.6 | **154** | - | - | 28.7 | 0.054 | ***0.4*** | 1.8 | *0.5* | **1.15** | **1.66** | 38 | ***5.5*** | 0.47 | 136 | - | *0.75* | 7.61 | - | 12.2 | 0.28 | 5.51 |
| Florence | 2002[b] | 5.23 | 6.2 | 69.1 | - | - | 54.3 | 0.031 | **0.29** | 1.5 | *0.5* | 0.55 | **1.21** | 9.97 | ***5.5*** | 0.48 | 64.7 | - | *0.75* | 2.12 | - | 10 | *0.5* | 3.44 |
| Fraleck | 2002[b] | 6.07 | 7.4 | 62 | - | - | 15.8 | 0.029 | **0.33** | 1.84 | *0.5* | 0.292 | **1.52** | 39.7 | **1.91** | 0.59 | 48.4 | - | *0.75* | 3.69 | - | 11.8 | *0.5* | 3.33 |
| Frederick | 2002[b] | 5.09 | 7 | **95.5** | - | - | 24.8 | 0.045 | ***0.4*** | 1.88 | *0.5* | 0.347 | **1.17** | 10.4 | **6.78** | 0.5 | 78.4 | - | *0.75* | 5.94 | - | 11.6 | *0.5* | 4.07 |
| George | 2002[b] | 6.11 | 6.8 | 25.5 | - | - | 15 | 0.015 | **0.29** | 1.72 | *0.5* | *0.75* | 0.609 | 10.2 | **4.46** | 0.62 | 60.4 | - | *0.75* | 2.74 | - | 13.6 | 0.23 | 6.32 |
| Gullrock | 2002[b] | 6.43 | 6.8 | 41.9 | - | - | 6.2 | 0.014 | **0.31** | 1.78 | *0.5* | *0.75* | **1.15** | 18.5 | 0.868 | 0.65 | 11.1 | - | *0.75* | 1.4 | - | 9.08 | 0.14 | 1.44 |
| Jim Edwards | 2002[b] | 5.15 | 4.8 | **89.4** | - | - | 101 | 0.038 | **0.19** | 1.16 | *0.5* | 0.837 | 0.41 | 6.84 | **2.38** | 0.4 | 53.2 | - | *0.75* | 1.9 | - | 11.1 | *0.5* | 3.5 |
| Johnnie | 2002[b] | 5.89 | 6.6 | 52.9 | - | - | 13.3 | 0.020 | **0.24** | 1.76 | 0.187 | *0.75* | **2.15** | 18.4 | ***5.5*** | 0.61 | 63.5 | - | *0.75* | 6.11 | - | 12.4 | 0.25 | 6.15 |
| Killarney | 2002[b] | 5.12 | 5.6 | 28.2 | - | - | 17.2 | 0.038 | **0.15** | 1.4 | 0.221 | **2.08** | 0.536 | 16.4 | **2.96** | 0.49 | 85.5 | - | *0.75* | 6.73 | - | 10.9 | *0.5* | 11.7 |
| Klock | 2002[b] | 5.73 | 5 | 35.7 | - | - | 50.1 | 0.033 | **0.22** | 1.28 | *0.5* | *0.75* | 0.476 | 4.28 | ***5.5*** | 0.46 | 36.4 | - | *0.75* | 0 | - | 11.3 | *0.5* | 1.32 |
| Landers | 2002[b] | 4.99 | 4.2 | **130** | - | - | 57.8 | 0.036 | ***0.4*** | 1.02 | 0.228 | **1.09** | 0.706 | 29.5 | **6.7** | 0.34 | 80.6 | - | *0.75* | 1.16 | - | 7.93 | 0.15 | 3.77 |
| Laundrie | 2002[b] | 5.46 | 6.6 | **119** | - | - | 14.5 | 0.040 | **0.35** | 1.74 | *0.5* | 0.836 | **1.23** | 98.4 | **2.03** | 0.5 | 82.5 | - | 0.029 | 2.93 | - | 12.5 | 0.64 | 5.71 |
| Mahzenazing | 2002[b] | 6.13 | 7.6 | 52.9 | - | - | 14.6 | 0.021 | **0.4** | 1.7 | 0.507 | *0.75* | **2.95** | 77.7 | **5.55** | 0.78 | 46.4 | - | *0.75* | 7.72 | - | 15.1 | 0.65 | 6.25 |
| Marjorie | 2002[b] | 4.61 | 4 | **81.4** | - | - | 33.2 | 0.050 | ***0.4*** | 0.9 | *0.5* | **2.13** | **1.41** | 45.2 | **1.06** | 0.34 | 116 | - | *0.75* | 12 | - | 9.32 | *0.5* | 9.47 |
| Matagamasi | 2002[b] | 5.83 | 8.4 | 56.9 | - | - | 34 | 0.028 | **0.09** | 2.1 | 0.097 | 0.133 | **2.25** | 18.1 | **6.7** | 0.65 | 58.5 | - | *0.75* | 10.8 | - | 12.7 | 0.94 | 5.12 |
| Nellie | 2002[b] | 4.66 | 5.2 | **401** | - | - | 22.2 | 0.084 | ***0.4*** | 1.22 | *0.5* | **3.41** | **1.72** | 38.6 | **4.93** | 0.38 | 161 | - | *0.75* | 9.75 | - | 9.47 | *0.5* | 18.5 |
| O.S.A. | 2002[b] | 4.91 | 6.8 | **105** | - | - | 17.8 | 0.045 | ***0.4*** | 1.72 | 0.335 | 0.675 | **1.04** | 8.34 | **2.63** | 0.54 | 92.9 | - | *0.75* | 5.85 | - | 11.3 | *0.5* | 14 |
| Pilgrim | 2002[b] | 5.6 | 5.8 | 29.1 | - | - | 31.5 | 0.003 | **0.22** | 1.54 | *0.5* | **1.31** | 0.916 | 5.08 | **4.84** | 0.5 | 42.7 | - | *0.75* | 2.54 | - | 11.9 | *0.5* | 3.33 |
| Reef | 2002[b] | 6.14 | 9.6 | 8.55 | - | - | 11.8 | 0 | **0.44** | 2.26 | 0.25 | 0.867 | **2.42** | 2.44 | ***5.5*** | 0.96 | 29 | - | *0.75* | 6.25 | - | 18.6 | 0.019 | 2.47 |
| Ruth Roy | 2002[b] | 4.83 | 4 | **91.2** | - | - | 12.6 | 0.026 | **0.58** | 0.84 | *0.5* | **1.54** | **1.57** | 43 | **5.84** | 0.31 | 60.3 | - | *0.75* | 13.5 | - | 8.11 | *0.5* | 14.9 |
| Sans Chambre | 2002[b] | 6.51 | 5.4 | 29.8 | - | - | 13.4 | 0.015 | ***0.4*** | 1.6 | 0.012 | 0.458 | **2.06** | 61.4 | **5.08** | 0.46 | 30.2 | - | *0.75* | 6.07 | - | 12 | *0.5* | 4.26 |
| Seagram | 2002[b] | 5.48 | 6.8 | **82.1** | - | - | 42.8 | 0.028 | **0.22** | 1.68 | *0.5* | 0.719 | 0.884 | 19.3 | ***5.5*** | 0.53 | 71.3 | - | *0.75* | 3.77 | - | 13 | 0.019 | 4.24 |
| Silver | 2002[b] | 5.57 | 12.8 | 26.6 | - | - | 24.5 | 0.013 | **0.55** | 7.22 | *0.5* | **9.21** | **31.5** | 108 | 0.538 | 2.72 | 114 | - | *0.75* | **164** | - | 42.3 | *0.5* | **25.4** |
| Silvester | 2002[b] | 4.91 | 6.6 | 18.9 | - | - | 27.2 | 0.0092 | 0.0016 | 1.64 | *0.5* | **1.04** | **1.15** | 3.2 | **5.09** | 0.51 | 54.6 | - | *0.75* | 5.74 | - | 8.44 | 0.3 | 3.03 |
| Sunny Water | 2002[b] | 4.77 | 5.4 | **258** | - | - | 79.3 | 0.061 | **0.54** | 1.52 | 0.095 | **2.18** | **1.02** | 23.9 | 0.6 | 0.4 | 0.23 | - | *0.75* | 3.01 | - | 13.8 | 0.59 | 9.43 |
| Telfer | 2002[b] | 5.15 | 6.4 | **105** | - | - | 19 | 0.04 | ***0.4*** | 1.78 | *0.5* | **1.19** | **1.35** | 30.4 | ***5.5*** | 0.46 | 85.1 | - | *0.75* | 8.88 | - | 14.7 | 0.042 | 5.51 |
| Tillie | 2002[b] | 5.09 | 7 | **162** | - | - | 14.5 | 0.052 | ***0.4*** | 1.6 | *0.5* | 0.58 | **1.46** | 97.6 | 0.835 | 0.51 | 93 | - | *0.75* | 2.59 | - | 11.1 | 0.29 | 3.29 |
| Tilton | 2002[b] | 6.5 | 32.6 | 25.6 | - | - | 14.9 | 0.0093 | **0.33** | 3.14 | *0.5* | **1.08** | **10.2** | 71.4 | ***5.5*** | 0.9 | 59.7 | - | *0.75* | **68.5** | - | 21.2 | 0.057 | 8.49 |
| Tyson | 2002[b] | 6.15 | 8 | 53.1 | - | - | 14.8 | 0.0096 | **0.85** | 1.76 | *0.5* | 0.351 | **2.27** | 68.6 | ***5.5*** | 0.81 | 36.3 | - | *0.75* | 8.97 | - | 16.2 | *0.5* | 5.39 |
| Wabun | 2002[b] | 5.1 | 4 | **152** | - | - | 44.3 | 0.042 | **0.47** | 1.1 | 0.325 | **1.44** | 0.433 | 32 | ***5.5*** | 0.34 | 106 | - | *0.75* | 1.28 | - | 11.1 | *0.5* | 3.72 |
| Wavy | 2002[b] | 4.99 | 5.8 | **155** | - | - | 17.8 | 0.07 | **0.019** | 1.46 | 0.091 | **2.33** | **8.55** | 105 | ***5.5*** | 0.53 | 91.1 | - | *0.75* | **53.9** | - | 12.4 | 0.361 | 8.92 |

**Table 5.4 Screening of COC in Surface Water for 85 Lakes**

| Lake | Sampling Date | pH | Hardness $CaCO_3$ (mg/L) | Al (µg/L) | As (µg/L) | Sb (µg/L) | Ba (µg/L) | Be (µg/L) | Cd (µg/L) | Ca (mg/L) | Cr (µg/L) | Co (µg/L) | Cu (µg/L) | Fe (µg/L) | Pb (µg/L) | Mg (mg/L) | Mn (µg/L) | Hg (µg/L) | Mo (µg/L) | Ni (µg/L) | Se (µg/L) | Sr (µg/L) | V (µg/L) | Zn (µg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Screening Criterion** | | | | 75 | 5 | 20 | 1,000 | 11 | 0.017 | 116 | 1 | 0.9 | 1 | 300 | 1 | 82 | 638 | 0.1 | 40 | 25 | 1 | 1,500 | 6 | 20 |
| White Oak | 2002[b] | 5.84 | 7.4 | 40.1 | - | - | 15.6 | 0.022 | **0.339** | 1.76 | *0.5* | **1.45** | **6.39** | 22.6 | **2.09** | 0.75 | 47.8 | - | *0.75* | **37.8** | - | 15.7 | 0.216 | 7.23 |
| White Pine | 2002[b] | 5.18 | 5.2 | **151** | - | - | 48.7 | 0.058 | ***0.4*** | 1.28 | *0.5* | 0.288 | 0.753 | 50 | ***5.5*** | 0.43 | 63.8 | - | *0.75* | 2.28 | - | 12.1 | *0.5* | 4.17 |
| Whitson | 2002[b] | 6.77 | 25.4 | 28.8 | - | - | 17.6 | 0.0036 | **0.949** | 5.72 | 0.10 | **1.02** | **21.2** | 94.8 | **2.15** | 1.88 | 28.5 | - | *0.75* | **147** | - | 30.7 | 0.561 | 11.2 |
| Whitson | 2001[d] | 7 | 25.2 | - | 0.7 | - | 21.3 | - | - | 6.6 | - | - | **19** | 53 | - | 2.1 | 21 | - | - | **155** | - | 36.3 | - | 17.7 |
| Whitson | 2000[e] | 6.62 | - | - | - | - | - | - | **0.13** | - | - | - | **15.5** | - | - | - | - | - | - | - | - | - | - | **35.7** |
| Whitson | 97-01[f] | - | - | - | - | - | - | - | **0.12** | - | - | - | **17.8** | - | - | - | - | - | - | - | - | - | - | - |
| Whitson | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.031 | - | - | 0.471 | - | - | - |
| Wolf | 2002[b] | 4.98 | 6.8 | 25.8 | - | - | 25.4 | 0.014 | **0.46** | 1.68 | 0.17 | **1.12** | 0.96 | 8.45 | **3.84** | 0.52 | 53 | - | *0.75* | 9.7 | - | 8.35 | 0.298 | 4.04 |
| **Urban Lakes** | | | | | | | | | | | | | | | | | | | | | | | | |
| Bennett | 2003[c] | 6.74 | - | 14 | *1* | - | 11.3 | *0.015* | ***0.5*** | 2.44 | *0.5* | *0.75* | **9** | **319** | ***5.5*** | 0.85 | 15 | - | *0.8* | 24 | *0.25* | 12 | *0.75* | 3 |
| Bethel | 2003[c] | 9 | - | 30 | *0.25* | - | 26.8 | *0.015* | ***0.5*** | 13.5 | *0.5* | *0.75* | **3** | 99 | ***5.5*** | 5.45 | 127 | - | *0.8* | 21 | *0.25* | 45.9 | *0.75* | 1 |
| Broder #23 | 2003[c] | 6.38 | - | 23 | *0.25* | - | 15.6 | *0.015* | ***0.5*** | 2.34 | *0.5* | *0.75* | **10** | 36 | ***5.5*** | 0.84 | 28 | - | *0.8* | **49** | *0.25* | 13.8 | *0.75* | 5 |
| Brodill | 2003[c] | 6.05 | - | 47 | *0.25* | - | 15.8 | 0.03 | ***0.5*** | 1.94 | *0.5* | *0.75* | **9** | 60 | ***5.5*** | 0.75 | 44 | - | *0.8* | **56** | *0.25* | 14.7 | *0.75* | 14 |
| Clearwater | 2003[c] | 6.33 | - | 16 | *0.25* | - | 17.6 | *0.015* | ***0.3*** | 4.3 | *0.5* | *0.75* | **10** | 15 | ***5.5*** | 1.09 | 26 | - | *0.4* | **70** | *0.25* | 21.7 | *0.45* | 11 |
| Clearwater | 2002[b] | 6.26 | 14.8 | 20.6 | - | - | 17.8 | 0.009 | **0.295** | 4.22 | *0.5* | **1.84** | **9.8** | 23.7 | **4.48** | 1.06 | 34.5 | - | *0.75* | **76.1** | - | 21.7 | 0.325 | 11.9 |
| Crooked | 2003[c] | 5.78 | - | **87** | *0.25* | - | 17.8 | 0.03 | ***0.4*** | 2.5 | *0.5* | **2.4** | **35** | **781** | ***5.5*** | 1.05 | 58 | - | *0.75* | **108** | *0.25* | 14.2 | *0.5* | 11 |
| Crooked | 2002[b] | 5.36 | 11 | **122** | - | - | 25.7 | 0.049 | **0.986** | 2.74 | *0.5* | **8.32** | **48.4** | **505** | **4.08** | 1.12 | 127 | - | *0.75* | **219** | - | 19.2 | 0.414 | **21.3** |
| Crowley | 2003[c] | 6.31 | - | 26 | *0.25* | - | 14.5 | *0.015* | ***0.3*** | 3.28 | *0.5* | *0.75* | **11** | 49 | ***5.5*** | 0.76 | 32 | - | *0.4* | **55** | *0.25* | 13.4 | *0.45* | 6 |
| Daisy | 2003[c] | 6.2 | - | 30 | *0.25* | - | 16 | *0.015* | ***0.3*** | 2.58 | *0.85* | *0.75* | **12** | 36 | ***5.5*** | 1.23 | 24 | - | *0.95* | **80** | *0.25* | 12.9 | *1* | 6 |
| Dill | 2003[c] | 6.61 | - | 48 | *0.25* | - | 11.4 | *0.015* | ***0.3*** | 2.84 | *0.5* | *0.75* | **10** | **331** | ***5.5*** | 1.29 | 32 | - | *0.4* | **49** | *0.25* | 16 | *0.45* | 5 |
| Forest | 2003[c] | 6.18 | - | 27 | *0.25* | - | 22 | *0.015* | ***0.3*** | 2.94 | *0.5* | *0.75* | **12** | 50 | ***5.5*** | 1 | 39 | - | *0.4* | **91** | *0.25* | 17.2 | *0.45* | 10 |
| Grant | 2003[c] | 7.21 | - | 7 | *0.5* | - | 29 | *0.015* | ***0.3*** | 15.7 | *0.5* | **1.9** | **5** | 161 | ***5.5*** | 4.82 | **1,020** | - | *0.4* | **53** | *0.25* | 53.5 | *0.45* | 5 |
| Hannah | 2003[c] | 7.25 | - | 13 | *0.5* | - | 21.9 | *0.015* | ***0.3*** | 10.6 | *0.5* | *0.75* | **22** | 114 | ***5.5*** | 3.57 | 70 | - | *0.6* | **111** | *0.25* | 57.3 | *0.75* | 3 |
| Hannah | 2001[d] | 7.6 | 46 | - | 2 | - | 23.7 | - | - | 11.6 | - | - | **25** | 71 | - | 4.1 | 251 | - | - | **181** | **2.3** | 64.7 | - | 10 |
| Hannah | 2000[e] | 7.72 | - | - | - | - | - | - | **0.13** | - | - | - | **21.9** | - | - | - | - | - | - | - | - | - | - | **36** |
| Hannah | 97-01[f] | - | - | - | - | - | - | - | **0.13** | - | - | - | **22.7** | - | - | - | - | - | - | - | - | - | - | - |
| Hannah | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.014 | - | - | 0.727 | - | - | - |
| Johnny | 2003[c] | 6.76 | - | 31 | *0.5* | - | 29.3 | *0.015* | ***0.5*** | 9.22 | *0.5* | *0.75* | **19** | **656** | ***5.5*** | 2.96 | 168 | - | *0.8* | **85** | *0.25* | 50.1 | *0.75* | 3 |
| Kelly | 2003[c] | 6.95 | - | 32 | *0.5* | - | 37.9 | 0.04 | ***0.3*** | **274** | *0.5* | **14** | **31** | 249 | ***5.5*** | 43.5 | 102 | - | *0.4* | **317** | *0.5* | 425 | *0.45* | 14 |
| Kelly | 2001[d] | 8.4 | 582.7 | - | 2.4 | - | 38.4 | - | - | **204** | - | - | **15** | **477** | - | 17.5 | 207 | - | - | **338.2** | **5.8** | 365 | - | 11.7 |
| Kelly | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **15** | - | - | - | - | - | - | - | - | - | - | - |
| Laurentian | 2003[c] | 6.53 | - | 38 | *0.25* | - | 11 | *0.015* | ***0.3*** | 3.44 | *0.5* | *0.75* | **14** | **585** | ***5.5*** | 1.35 | 30 | - | *0.4* | **37** | *0.25* | 17.7 | *0.45* | 2 |
| Laurentian | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.038 | - | - | 0.42 | - | - | - |
| Linton | 2003[c] | 6.16 | - | 34 | *0.25* | - | 14.4 | *0.015* | ***0.3*** | 2.08 | *0.5* | *0.75* | **10** | 50 | ***5.5*** | 0.71 | 25 | - | *0.4* | **59** | *0.25* | 12.6 | 1 | 7 |
| Little Raft | 2003[c] | 7.02 | - | 9 | *0.5* | - | 14.6 | *0.015* | ***0.5*** | 3.3 | *0.5* | *0.75* | **8** | 82 | ***5.5*** | 1.13 | 27 | - | *0.8* | **38** | *0.25* | 18.1 | *0.75* | 2 |
| Lohi | 2003[c] | 6.28 | - | 22 | *0.25* | - | 14.5 | *0.015* | ***0.3*** | 4.34 | *0.5* | *0.75* | **12** | 106 | ***5.5*** | 1.31 | 41 | - | *0.4* | **59** | *0.25* | 21.4 | *0.45* | 10 |
| Long | 2003[c] | 7.1 | - | 14 | *0.25* | - | 16.6 | *0.015* | ***0.3*** | 8.46 | *0.5* | *0.75* | **12** | 27 | ***5.5*** | 2.79 | 9 | - | *0.4* | **47** | *0.25* | 32.6 | *0.45* | 7 |
| Long | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.0026 | - | - | 0.095 | - | - | - |
| McFarlane | 2003[c] | 7.33 | - | 8 | *0.25* | - | 19.6 | *0.015* | ***0.3*** | 15.7 | *0.5* | *0.75* | **8** | 22 | ***5.5*** | 4.75 | 59 | - | *0.4* | **51** | *0.25* | 50.7 | *0.45* | 9 |
| Middle | 2003[c] | 6.91 | - | 13 | *0.25* | - | 22.6 | *0.015* | ***0.3*** | 11 | *0.5* | *0.75* | **24** | 26 | ***5.5*** | 3.21 | 20 | - | *0.4* | **114** | *0.25* | 50.4 | *0.45* | 11 |
| Minnow | 2003[c] | 8.79 | - | 25 | *0.5* | - | 25 | *0.015* | ***0.5*** | 19.4 | *0.5* | *0.75* | **5** | 155 | ***5.5*** | 4.65 | 29 | - | *0.8* | 22 | *0.25* | 74.4 | *0.75* | 1 |
| Nepahwin | 2003[c] | 7.4 | - | 10 | *0.25* | - | 19.3 | *0.01* | ***0.65*** | 19.1 | *0.5* | *0.75* | **11** | 19 | ***5.5*** | 6.23 | 36 | - | *0.75* | **45** | *0.25* | 67.4 | *0.5* | 4 |
| Raft | 2003[c] | 6.61 | - | 10 | *0.25* | - | 14 | *0.015* | ***0.5*** | 3.18 | *0.5* | *0.75* | **12** | 24 | ***5.5*** | 1.11 | 31 | - | *0.8* | **74** | *0.25* | 16.1 | *0.75* | 7 |

**Table 5.4 Screening of COC in Surface Water for 85 Lakes**

| Lake | Sampling Date | pH | Hardness $CaCO_3$ (mg/L) | Al (µg/L) | As (µg/L) | Sb (µg/L) | Ba (µg/L) | Be (µg/L) | Cd (µg/L) | Ca (mg/L) | Cr (µg/L) | Co (µg/L) | Cu (µg/L) | Fe (µg/L) | Pb (µg/L) | Mg (mg/L) | Mn (µg/L) | Hg (µg/L) | Mo (µg/L) | Ni (µg/L) | Se (µg/L) | Sr (µg/L) | V (µg/L) | Zn (µg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Screening Criterion** | | | | 75 | 5 | 20 | 1,000 | 11 | 0.017 | 116 | 1 | 0.9 | 1 | 300 | 1 | 82 | 638 | 0.1 | 40 | 25 | 1 | 1,500 | 6 | 20 |
| Ramsey | 2003[c] | 7.43 | - | 4 | *0.5* | - | 15.7 | *0.015* | ***0.5*** | 15.2 | *0.5* | *0.75* | **12** | 11 | ***5.5*** | 4.33 | 12 | - | *0.4* | **55** | *0.25* | 49.1 | *0.45* | 2 |
| Ramsey | 2001[d] | 8 | 52.4 | - | 1.3 | - | 14 | - | - | 14 | - | - | **9.7** | 36 | - | 4.2 | 72 | - | - | **52** | **2** | 48.7 | - | 5.7 |
| Ramsey | 2000[e] | 7.71 | - | - | - | - | - | - | **0.1** | - | - | - | **14.4** | - | - | - | - | - | - | - | - | - | - | 16.7 |
| Ramsey | 97-01[f] | - | - | - | - | - | - | - | **0.06** | - | - | - | **11.6** | - | - | - | - | - | - | - | - | - | - | - |
| Ramsey | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.021 | - | - | 0.611 | - | - | - |
| Richard | 2003[c] | 7.25 | - | 6 | *0.25* | - | 17.2 | *0.015* | ***0.3*** | 8.46 | *0.5* | *0.75* | **8** | 53 | ***5.5*** | 2.81 | 168 | - | *0.4* | **57** | *0.25* | 31.5 | *0.45* | 3 |
| Robinson | 2003[c] | 7.7 | - | 46 | *1* | - | 18.6 | *0.015* | ***0.5*** | 15.8 | *0.5* | *0.75* | **10** | 227 | ***5.5*** | 5.22 | 39 | - | *0.8* | **36** | *0.25* | 55.6 | *0.75* | 1 |
| Silver | 2003[c] | 6 | - | 14 | *0.25* | - | 22.4 | *0.01* | ***0.65*** | 7.34 | *0.5* | **4.9** | **17** | 90 | ***5.5*** | 2.74 | 88 | - | *0.75* | **105** | *0.25* | 40.6 | *0.5* | 18 |
| St. Charles | 2003[c] | 7.22 | - | 16 | *0.25* | - | 19.2 | *0.015* | ***0.5*** | 9.24 | *0.5* | *0.75* | **21** | 76 | ***5.5*** | 3.38 | 18 | - | *0.8* | **95** | *0.25* | 42.1 | *0.75* | 6 |
| Still | 2003[c] | 7.55 | - | **181** | *0.75* | - | 40.7 | 0.015 | ***0.5*** | 18 | *0.5* | *0.75* | **15** | **424** | ***5.5*** | 5.84 | 100 | - | *0.8* | **58** | *0.25* | 75 | *0.75* | 8 |
| Tilton | 2003[c] | 6.28 | - | 17 | *0.25* | - | 14.6 | *0.01* | ***0.4*** | 3.5 | *0.5* | *0.75* | **9** | 75 | ***5.5*** | 0.97 | 45 | - | *0.75* | **50** | *0.25* | 20.3 | *0.5* | 12 |
| **Additional Lakes** | | | | | | | | | | | | | | | | | | | | | | | | |
| Kusk | 2001[d] | 7.4 | 69.8 | 46 | 2 | - | 14.3 | - | - | 22 | - | - | **6** | 261 | - | 3.6 | 328 | - | - | **60** | **1.2** | 56.7 | - | 6 |
| Kusk | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **6** | - | - | - | - | - | - | - | - | - | - | - |
| Larder | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **10** | - | - | - | - | - | - | - | - | - | - | - |
| Larder | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.038 | - | - | 0.13 | - | - | - |
| McCharles | 2001[d] | 7.5 | 134.3 | - | 1.7 | - | 19.7 | - | - | 43.3 | - | - | **6** | 206 | - | 6.4 | 332 | - | - | **111.3** | 1 | 102 | - | 4.7 |
| McCharles | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **6** | - | - | - | - | - | - | - | - | - | - | - |
| Michiwakenda | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **5.1** | - | - | - | - | - | - | - | - | - | - | - |
| Michiwakenda | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.0034 | - | - | 0.088 | - | - | - |
| Mud | 2001[d] | 7.5 | 523.5 | 70.7 | 3.3 | - | 33.3 | - | - | **186** | - | - | **17** | **653** | - | 14.3 | 513 | - | - | **257.3** | **4.7** | 291 | - | 7.3 |
| Mud | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **17** | - | - | - | - | - | - | - | - | - | - | - |
| Nelson | 97-01[f] | - | - | - | - | - | - | - | **0.1** | - | - | - | **3** | - | - | - | - | - | - | - | - | - | - | - |
| Round | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **4.5** | - | - | - | - | - | - | - | - | - | - | - |
| Simon | 2001[d] | 7.4 | 380.3 | - | 2.3 | - | 33.3 | - | - | **128** | - | - | **9.7** | **422** | - | 14.8 | 522 | - | - | **318.7** | **5** | 271.3 | - | 9 |
| Simon | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **9.7** | - | - | - | - | - | - | - | - | - | - | - |
| Vermilion | 97-01[f] | - | - | - | - | - | - | - | - | - | - | - | **8.1** | - | - | - | - | - | - | - | - | - | - | - |
| Vermilion | 96-97[g] | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.0046 | - | - | 0.114 | - | - | - |
| **Maximum** | | | 582.7 | **401** | 3.3 | - | 101 | 0.08 | **0.99** | **274** | 0.85 | **14** | **48.4** | **781** | **22.16** | 43.5 | **1,020** | 0.04 | 0.95 | **338.2** | **5.8** | 425 | 1.0 | **36.0** |

**References**

[a] Co-Op. Unpublished a. Co-Operative Freshwater Ecology Unit. Unpublished data for SES Intensive Study Lakes (1984 to 2002). Provided by Bill Keller. Department of Biology, Laurentian University. Sudbury, Ontario, Canada.

[b] Co-Op. Unpublished b. Co-Operative Freshwater Ecology Unit. Unpublished data for SES Extensive Study Lakes (1981 to 2002). Provided by Bill Keller. Department of Biology, Laurentian University. Sudbury, Ontario, Canada. Note: Hardness values are for 2001.

[c] Keller, B., Heneberry, J., Gunn, J.M., Snucins, E., Morgan, G. and Leduc, J. 2004. Recovery of Acid and Metal Damaged Lakes Near Sudbury Ontario: Trends and Status. Cooperative Freshwater Ecology Unit. Department of Biology, Laurentian University. Sudbury, Ontario, Canada.

[d] Pyle, G.G., J.W. Rajotte, and P. Couture. 2005. Effects of Industrial Metals on Wild Fish Populations along a Metal Contamination Gradient. Ecotoxicology and Environmental Saftey. In Press. Note: Values are the mean of 3 samples.

[e] Couture, P. and P.R. Kumar. 2003. Impairment of metabolic capacities in copper and cadmium contaminated wild yellow perch (*Perca flavescens*). Aquatic Toxicology. 64(1):107-120.

[f] Couture, P. and Rajotte, J.W. 2003. Morphometric and Metabolic Indicators of Metal Stress in Wild Yellow Perch (*Perca flavescens*) from Sudbury, Ontario: A Review. J. Environ. Monit. 5:216-221. Note: Values are the mean of concentrations measured in studies from 1997 to 2001.

[g] Chen, Y.W., Belzile, N. and Gunn, J.M. 2001. Antagonistic Effect of Selenium on Mercury Assimilation by Fish Populations Near Sudbury Metal Smelters. Limnol. Oceanogr. 46:1814-1818.

Note Values in italics represent 1/2 detection limit
Bolded values in grey scale indicate concentrations that exceed the screening criterion

*This page left blank intentionally*

Four measured concentrations of calcium (274,000, 204,000, 186,000, and 128,000 µg/L) exceeded the screening guideline (116,000 µg/L) in three of the sampled lakes. The two highest concentrations were from Kelly Lake, a lake which is not recommended for evaluation in this ERA due to impacts from sources other than smelter airborne particulate emissions (Section 5.6.2). All other measured concentrations were well below the guideline. Therefore, calcium does not meet the screening criterion of being elevated across the study area (3 of 80 sampled lakes = 3.75%), and it is recommended that Ca not be considered a COC.

The maximum concentration of manganese (1,020 µg/L) was measured in Grant Lake. Water hardness in Grant Lake was measured at 39 mg/L. This value was used to calculate the lake-specific screening guideline of 777 µg/L ((4.4 x 39) + 605). All other concentrations of manganese were well below the minimum lake-specific guideline (605 µg/L) calculated based on hardness. Therefore, Mn does not meet the screening criterion of being elevated across the study area (1 of 80 sampled lakes = 1.25%), and it is recommended that Mn not be considered a COC.

Zinc concentrations in four (21.3, 25.4, 35.7 and 36 µg/L) of 80 sampled lakes (or 5% of sampled lakes) exceeded the screening guideline (20 µg/L). One exceedance was for Whitson Lake, a lake which has received direct industrial effluent and is not recommended for evaluation in this ERA (Section 5.6.2). Therefore, zinc does not meet the screening criterion of being elevated across the study area, and it is recommended that Zn not be considered a COC.

Selenium concentrations in six (1.2, 2, 2.3, 4.7, 5 and 5.8 µg/L) of 39 sampled lakes (or 15%) exceeded the screening guideline (1 µg/L). One exceedence is for Kelly Lake, a lake which is not recommended for evaluation in this ERA (Section 5.6.2). Two other exceedences were for Hannah and Ramsey Lakes, lakes which were sampled in the Urban Lakes program where levels were all non-detect and below guidelines. In fact, the only exceedances were reported in a single primary literature source (Pyle *et al*., 2005). All other concentrations reported were below the guideline, and many were below the detection limit and guideline. However, since exceedences of selenium were found in more than 5% of lakes sampled, to be conservative it is recommended that Se be considered a COC in surface water.

Concentrations of aluminium in many of the SES Extensive Lakes exceeded the screening guideline (75 µg/L). The SES Extensive Lakes are not recommended for evaluation in this ERA because they are primarily affected by acidity and less so by metals. Since the majority of these lakes suffer from acidification, it is possible that the low pH has increased the mobility of aluminium, resulting in its release to the water column. Concentrations of aluminium in urban lakes, which have more neutral pH, were only found to exceed the screening guideline in two lakes. However, since exceedences of aluminium were found in 26% of those lakes sampled (20 of 77), to be conservative it is recommended that Al be considered a COC.

Iron concentrations exceeded the screening guideline (300 µg/L) in nine lakes, and background concentrations (80 µg/L) in 30 lakes (maximum concentration of 781 µg/L). Cadmium concentrations consistently exceed both the screening guideline and background concentrations, and cadmium has been associated with smelter emissions (SARA, 2005). Cobalt, copper, lead and nickel concentrations exceed guidelines in many lakes, although cobalt concentrations exceed the guideline only in a few lakes monitored as part of the Urban Lakes program. In addition, the detection limit for lead in the Urban Lakes program was 11 µg/L, well above the screening guideline of 1 µg/L, and therefore the non-detect concentrations reported for the urban lakes do not allow proper screening. However, cobalt, copper, lead and nickel are linked to smelter particulate emissions and are COC for soils. Therefore, Cd, Co, Cu, Fe, Pb and Ni are recommended as COC for surface water.

### 5.5.4 Final List of Recommended COC for Surface Water

The recommended final list of COC within the surface water of Sudbury lakes is:

- Aluminium
- Cadmium
- Cobalt
- Copper
- Iron
- Lead
- Nickel
- Selenium

In addition, it is noted that pH alone, or in combination with metals, may influence aquatic organisms. It is also noted (Table 5.4) that pH is a more significant issue for the SES Extensive Lakes (outside the study area) than for urban lakes (within the study area) where pH is generally >6. Therefore, pH is not a COC; however, it should be considered as a modifying factor in a detailed ERA (see also Section 5.8)

## 5.6 Selection of COC in Sediment

The selection of COC in sediment of Sudbury lakes was completed (Section 5.6.1 through 5.6.4) with a similar approach as was used for surface water, following the four-step process identified in Figure 5.2.

### 5.6.1 Compilation of Sediment Data

Government, academic and primary literature resources were reviewed and appropriate data were compiled. In cases where sediment samples were taken at multiple depths, those that were taken within the top 10 cm were selected over those taken at greater depths. The top 5 to 10 cm of sediment is generally regarded as the biologically active zone in which there is significant interaction with benthic invertebrate activities and the overlying water (Best, 2001). The primary source of sediment chemistry data was a study in which the top 2 cm of sediment from 11 Sudbury area lakes were sampled using an Eckman dredge in the mid-1990s (Co-Op, 2004). Three samples were taken from within a deep basin of each of the lakes. The MOE analyzed the samples and provided results on a dry weight basis. In addition to these results, data from numerous primary literature resources were also considered. The maximum concentration for each metal was selected and used for comparison to screening guidelines.

### 5.6.2 Identification of Sediment Guidelines

Several sources of screening guidelines were consulted (Table 5.5). The Provincial Sediment Quality Guidelines (PSQGs) (MOE, 1993) and the Canadian Environmental Quality Guidelines (CEQGs) (CCME, 2002) were used preferentially over other guidelines.

Three different guidelines are available within the PSQGs that provide a range of protection for aquatic life. The No Effect Level (NEL) protects from any toxic effects for fish and sediment-dwelling organisms. The Lowest Effect Level (LEL) is protective of the majority of sediment-dwelling organisms. Sediment at this level is considered to be clean to marginally polluted, with the majority of the benthic community capable of tolerating chemicals at this level. However, the MOE (1993) recognizes that local background concentrations of metals in sediments may exceed LELs, at which point the background concentration is recommended as a replacement for the LEL. The Severe Effect Level (SEL) represents the level at which a “pronounced disturbance of the sediment-dwelling community” is expected to occur (MOE, 1993). Both the LEL and SEL are presented in Table 5.5. The PSQGs were developed using data from the Great Lakes; inland lakes, particularly lakes on the Precambrian Shield, were not included in the development of these guidelines. As a result, in highly mineralized areas, even the SEL can be exceeded in undisturbed areas (Wren, 1996). However, these guidelines are still useful as a screening tool.

The CCME sediment guidelines also are available for more than a single level of protection. They are designed to be protective of all forms of aquatic life during all life cycles for chronic exposure periods. The Interim Sediment Quality Guideline (ISQG) represents the level at which adverse effects are rarely observed. The Probable Effect Level (PEL) represents the level at which adverse biological effects are usually or always observed (CCME, 2002). Both the ISQG and PEL are presented in Table 5.5. The lower of the MOE LEL and CCME ISQG was selected for screening metals in sediment, except when these were lower than the local background concentrations (Table 5.6). The background concentration was used as the screening guideline under these circumstances, provided there was sufficient data and confidence that the derived value was truly representative of background conditions. When no guideline was available from either MOE or CCME, the lowest of a number of additional screening benchmarks from the U.S. EPA Region 4, Oak Ridge National Laboratory, RAIS, and Thompson *et al*. (2005) was selected. The values taken from Thompson *et al*. (2005) were derived from a large dataset collected from areas affected by uranium mining and milling regions of Canada and are based on the methods used to derive PSQG LELs.

**Table 5.5 Screening Criteria for Chemicals of Concern in Sediments (µg/g d.w.)**

| COC | MOE[a] | | CCME[a] | | Oak Ridge SEC[c] | U.S. EPA Region 4[d] | RAIS | Thompson *et al.* |
|---|---|---|---|---|---|---|---|---|
| | LEL | SEL | ISQG | PEL | | | | |
| Aluminium | - | - | - | - | 58,030 | - | **58,000** | - |
| Antimony | - | - | - | - | - | 12 | **2** | - |
| Arsenic | 6 | 33 | **5.9** | 17 | 10.8 | 7.24 | 5.9 | 9.8 |
| Barium | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - |
| Cadmium | **0.6** | 10 | 0.6 | 3.5 | 0.58 | 1 | 0.59 | - |
| Calcium | - | - | - | - | - | - | - | - |
| Chromium | **26** | 110 | 37 | 90 | 36.3 | 52.3 | 26 | 47.6 |
| Cobalt | **50** [b] | - | - | - | - | - | 50 | - |
| Copper | **16** | 110 | 36 | 197 | 28 | 18.7 | 16 | 22.2 |
| Iron | **2%** | 4% | - | - | - | - | 20,000 (2%) | - |
| Lead | **31** | 250 | 35 | 91.3 | 34.2 | 30.2 | 30.2 | 36.7 |
| Magnesium | - | - | - | - | - | - | - | - |
| Manganese | **460** | 1100 | - | - | 631 | - | 460 | - |
| Mercury | 0.2 | 2 | **0.17** | 0.486 | - | 0.13 | 0.13 | - |
| Molybdenum | - | - | - | - | - | - | - | **13.8** |
| Nickel | **16** | 75 | - | - | 19.5 | 15.9 | 15.9 | 23.4 |
| Selenium | - | - | - | - | - | - | 5[e] | **1.9** |
| Strontium | - | - | - | - | - | - | - | - |
| Vanadium | **-** | **-** | - | - | - | - | - | **35.2** |
| Zinc | **120** | 820 | 123 | 315 | 94.2 | 124 | 120 | - |

* Bolded values in grey scale were selected for the current assessment.

[a] Lowest Effect Level (LEL), Severe Effect Level (SEL) (MOE, 1993); Interim Sediment Quality Guideline (ISQG), Probable Effects Level (PEL) (CCME, 2002).

[b] An MOE PSQG is not available for cobalt. This value is carried over from the Open Water Disposal Guidelines (MOE, 1993).

[c] Values are the lowest of the recommended Sediment Effect Concentrations (SECs) that meet the minimum requirements for recommendation (Jones *et al.*, 1997).

[d] U.S. EPA Region 4 Sediment Screening Values (U.S. EPA, 2001).

[e] Value was taken from the BC Working Guidelines for Sediments (Freshwater) (BC MELP, 1998).

### 5.6.3 Application of Screening Criteria

There are four screening criteria applied and discussed in this section:

- Compare maximum concentrations to guidelines;
- Compare environmental levels to regional background levels;
- Characterize the distribution of any exceedences across the study area; and
- Determine if the metal has been scientifically linked to smelting activities.

Concentrations from 50 reference lakes were used to characterize regional background concentrations of COC in sediments within the Sudbury area (Table 5.6). This provided some insight into the relative contribution of smelter emissions to the total levels in sediment. The mean concentration was used as the screening value. The Ontario MOE "Guidelines for the protection and management of aquatic sediment quality in Ontario" (MOE, 1993) recommend that site-specific background concentrations be calculated as "the mean of 5 surficial sediment samples (top 5 cm) taken from an area contiguous to the area under investigation, but unaffected by any current or historical point source inputs; or the mean of 5 samples taken by a sediment core from the pre-colonial sediment horizon". However, for the purposes of the current evaluation, background was considered to be the concentration of metals in sediments of lakes that have not been directly impacted by smelter emissions.

Sediment concentrations obtained from the Pyle *et al.* (2005) study and presented in Table 5.6 represent the average of three surface sediment samples (top 5 cm) taken at three depths (1, 5, and 10 m) using an Eckman grab sampler. Concentrations from the Couture and Kumar (2003) study are the average of two surface sediment samples taken at a depth of approximately 4 m, also collected using an Eckman grab sampler. Values obtained from the Audet and Couture (2003) study are the averages of nine surface samples (upper 5 cm) collected using an Eckman grab sampler at various water depths.

As explained for the calculation of background surface water concentrations, four of the reference lakes (Barlow, Big Marsh, Birch and Waubamac) are located within a relatively close range of the Copper Cliff smelter (39 to 50 km) but are not considered to be influenced by mining and smelting activities (Pyle *et al*., 2005). The remaining lakes are located at distances greater than 90 km of the City of Greater Sudbury, therefore, levels of COC in sediment may be considered representative of regional background conditions (Couture and Kumar, 2003; Couture and Rajotte, 2003; Audet and Couture, 2003; Keller *et al*., 2004; Borgmann *et al*., 2001b; Conroy *et al*., 1978).

Sediments were sampled from the four reference lakes (Tomiko, Restoule, Nosbonsing and Talon) in 1996 and 1998 in the Borgmann *et al*. (2001b) study using a ponar grab sampler. In order to specifically address the effects of metals, the survey only included lakes with surface waters with circumneutral pH (6.7 to 7.7). The concentration of measured parameters for each lake presented in Table 5.6 is the average of a single sample taken in 1996 and two samples taken in 1998 (Borgmann *et al*., 2001b). Values obtained from the Couture and Rajotte (2003) study represent average values taken from numerous studies.

Conroy *et al*. (1978) conducted extensive monitoring of 209 lakes in the greater Sudbury area between 1974 and 1976. Sediment samples were collected in 101 lakes using an Ekman dredge in 1974 and 1975. To be consistent with Borgmann *et al*. (2001b), lakes located a minium of 90 km from Sudbury (116 lakes) were considered to be reference lakes and included within the current derivation of regional background concentrations. The concentrations of measured parameters for each lake presented in Table 5.6 are the average of one to eight samples, depending on the individual lake.

**Table 5.6a Sediment Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg/g d.w.)**

| Parameter | Lake | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Bark [f] | Barlow[a] | Big Marsh[a] | Birch[a] | Cavell [f] | Cecebe [f] | Diamond [f] | Eagle [f] | East Bull | Fanny[f] | Flack | Florence | Geneva[c] |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - | 12,000 |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | 13.8 | 12 | 33.8 | - | - | - | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - | 62 |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - | 0.81 |
| Cadmium | 2 | 0.9 | 0.7 | 1.7 | 4 | 2.8 | 2 | <2 | 4.8 | 2.8 | 3.5 | 3 | 2.9 |
| Calcium | - | 3,644 | 3,593 | 5,881 | - | - | - | - | - | - | - | - | - |
| Chromium | - | 44 | 52.2 | 32.4 | - | - | - | - | - | - | - | - | 30 |
| Cobalt | - | 22.4 | 49.5 | 13.8 | - | - | - | - | - | - | - | - | 19 |
| Copper | 11 | 65.3 | 58.4 | 174.7 | 66 | 19 | 48 | <5 | 52 | 23 | 81 | 57 | 82 |
| Iron | 22,000 | 22,728 | 37,691 | 11,024 | 5,000 | 28,000 | 37,000 | 9,000 | 37,000 | 21,000 | 36,000 | 39,000 | 25,000 |
| Lead | 32 | 23.3 | 18.1 | 67.9 | 59 | 36 | 50 | 14 | 136 | 46 | 125 | 50 | 106 |
| Magnesium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Manganese | - | 460 | 469 | 153 | - | - | - | - | - | - | - | - | 450 |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - | 1.1 |
| Nickel | 9 | 122.1 | 95.6 | 210.8 | 35 | 22 | 41 | 6 | 24 | 35 | 22 | 42 | 100 |
| Selenium | - | 24.9 | 22.8 | 70.5 | - | - | - | - | - | - | - | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - | 24 |
| Vanadium | - | 40.7 | 58.7 | 47.6 | - | - | - | - | - | - | - | - | 41.3 |
| Zinc | 77 | 132 | 137.9 | 158.4 | 119 | 130 | 130 | 57 | 170 | 91 | 177 | 149 | 150 |
| pH | | - | - | - | | | | | | | | | 3.8 |

**Table 5.6b Sediment Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg/g d.w.)**

| Parameter | Lake | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Halfway[b] | Hammond[f] | Island[f] | Jim Edwards | Jumping Cariboo[f] | Kagawon[f] | Lac aux Sables[f] | Lady Evelyn[f] | Lorraine[f] | Low Water[f] | Marne[f] | Marten | McConnell |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | 1 | 2.8 | 3.8 | 3 | 12 | <2 | 1.7 | 4.5 | 5 | <2 | <2 | 3.9 | 1.5 |
| Calcium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Chromium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Cobalt | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Copper | 21.5 | 26 | 29 | 27 | 68 | 14 | 33 | 54 | 51 | 28 | 18 | 41 | 11 |
| Iron | - | 35,000 | 33,000 | 48,000 | 17,000 | 12,000 | 22,000 | 56,000 | 22,000 | 15,000 | 9,000 | 56,000 | 10,000 |
| Lead | - | 23 | 75 | 179 | 45 | 21 | 54 | 59 | 96 | 35 | 30 | 72 | 8 |
| Magnesium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Manganese | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 4.8 | 53 | 41 | 20 | 130 | 38 | 19 | 45 | 51 | 30 | 33 | 66 | 13 |
| Selenium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 81.9 | 63 | 200 | 85 | 95 | 35 | 113 | 202 | 188 | 100 | 52 | 201 | 38 |
| pH | - | | | | | | | | | | | | |

**Table 5.6c Sediment Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg/g d.w.)**

| Parameter | Lake | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Midlothian | Mozhabong[f] | Neiscoot[f] | Nosbonsing[e] | Obakia[f] | Opikinimika[f] | Rabbit[f] | Red Cedar[f] | Restoule[e] | Rib[f] | Round[f] | Schist[f] | Shoofl[f] |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Arsenic | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Barium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Cadmium | <2 | 1.7 | 3 | 1.9[g] | 2 | 1.1 | <2 | 3.3 | 2.49 | 5 | 4 | 3 | 1.3 |
| Calcium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Chromium | - | - | - | 90 | - | - | - | - | 51 | - | - | - | - |
| Cobalt | - | - | - | 19 | - | - | - | - | 26 | - | - | - | - |
| Copper | 44 | 23 | 47 | 32[g] | 24 | 14 | 10 | 63 | 33 | 94 | 36 | 64 | 23 |
| Iron | 10,000 | 35,000 | 35,000 | 45,700[g] | 8,000 | 31,000 | 19,000 | 45,000 | 51,300 | 40,000 | 25,000 | 6,000 | 5,000 |
| Lead | 11 | 46 | 153 | 34[g] | 19 | 19 | 7 | 81 | 43 | 83 | 67 | 13 | 27 |
| Magnesium | - | - | - | 9,440 | - | - | - | - | 4,900 | - | - | - | - |
| Manganese | - | - | - | 619 | - | - | - | - | 2,723 | - | - | - | - |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Nickel | 33 | 23 | 38 | 49[g] | 82 | 14 | 14 | 89 | 39 | 69 | 27 | 35 | 11 |
| Selenium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Vanadium | - | - | - | - | - | - | - | - | - | - | - | - | - |
| Zinc | 58 | 105 | 215 | 130[g] | 52 | 81 | 27 | 199 | 245 | 266 | 248 | 96 | 76 |
| pH | | | | | | | | | | | | | |

**Table 5.6d Sediment Concentrations in Sudbury Lakes that are Indicative of Regional Background Levels (µg/g d.w.)**

| Parameter | Lake | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Stull** | **Talon[e]** | **Tatachikapika[f]** | **Temegami[f]** | **Tenfish** | **Timber** | **Tomiko[e]** | **Trout[f]** | **Valin** | **Waubamac[a]** | **Welcome[f]** | **Mean[h]** |
| Aluminium | - | - | - | - | - | - | - | - | - | - | - | **12,000[d]** |
| Antimony | - | - | - | - | - | - | - | - | - | - | - | **-** |
| Arsenic | - | - | - | - | - | - | - | - | - | 20.6 | - | **20** |
| Barium | - | - | - | - | - | - | - | - | - | - | - | **62[d]** |
| Beryllium | - | - | - | - | - | - | - | - | - | - | - | **0.81[d]** |
| Cadmium | 3 | 1.91 | <2 | 2 | 2.6 | 3.8 | 4.5[g] | <2 | 3 | 2.4 | 2 | **2.6** |
| Calcium | - | - | - | - | - | - | - | - | - | 5,098 | - | **4,554** |
| Chromium | - | 94 | - | - | - | - | 73 | - | - | 53.3 | - | **58** |
| Cobalt | - | 23 | - | - | - | - | 26 | - | - | 28.5 | - | **25** |
| Copper | 40 | 32 | 9 | 46 | 55 | 27 | 31[g] | 5 | 14 | 118.1 | 65 | **42** |
| Iron | 48,000 | 69,000 | 9,000 | 33,000 | 14,000 | 21,000 | 116,750[g] | 4,000 | 5,000 | 23,715 | 28,000 | **28,427** |
| Lead | 62 | 35 | 10 | 33 | 46 | 30 | 40[g] | 10 | 19 | 56.2 | 30 | **50** |
| Magnesium | - | 11,560 | - | - | - | - | 6,360 | - | - | - | - | **8,065** |
| Manganese | - | 2,743 | - | - | - | - | 1,511 | - | - | 528 | - | **1,073** |
| Mercury | - | - | - | - | - | - | - | - | - | - | - | **-** |
| Molybdenum | - | - | - | - | - | - | - | - | - | - | - | **1.1[d]** |
| Nickel | 40 | 55 | 9 | 43 | 12 | 26 | 53[g] | <5 | 24 | 159.2 | 37 | **45.8** |
| Selenium | - | - | - | - | - | - | - | - | - | 40.5 | - | **39.7** |
| Strontium | - | - | - | - | - | - | - | - | - | - | - | **24[d]** |
| Vanadium | - | - | - | - | - | - | - | - | - | 52.5 | - | **48** |
| Zinc | 139 | 285 | 61 | 112 | 79 | 204 | 254[g] | 16 | 104 | 197.1 | 144 | **131** |
| pH | | | | | | | | | | - | | **3.8[d]** |

[a] Values are from 2001 (Pyle *et al*., 2005).
[b] Values are averages from samples taken from 1997-2001 (Couture and Kumar, 2003; Couture and Rajotte, 2003; Audet and Couture, 2003).
[c] Values are the average of three samples taken in 1993 (Keller *et al*., 2004).
[d] If only one value available, then it is presented also in this column.
[e] Values are the average of samples collected by ponar grab in 1996 and 1998 and surface grab samples collected in 1996 by Borgmann *et al*. (2001b). Ponar grab samples collected at a depth of 10 m, and at a depth of 23, 28, 14, and 40 m for Tomiko, Restoule, Nosbonsing and Talon lake, respectively. Values under the detection limit were included at a value of half the detection limit
[f] Values are mean values determined by Conroy *et al.* (1978). Values are based on the average of one to eight samples, depending on the individual lake, sampled in 1974 and 1975 with an Ekman dredge.
[g] Value is the average of sediment concentration from samples obtained in 1974 and 1996/98 (Conroy *et al*., 1978; Borgmann *et al*., 2001b).
[h] To calculate the mean half of the detection limit was used for those concentrations expressed as a less than value.

The average concentration of ten metals (arsenic, cadmium, chromium, copper, iron, lead, manganese, nickel, selenium and zinc) in the sediment of the reference lakes exceeded the corresponding screening criteria. This may be indicative of naturally-elevated levels of these metals in the Sudbury area, or may reflect the conservative nature of these screening guidelines. As stated previously, MOE (1993) acknowledged that local background concentrations of metals may exceed LELs (which are very close to the values of the CCME ISQGs). In this case, MOE (1993) recommends that local background levels be used as the screening value (Table 5.7). Therefore, the mean regional background concentrations were used as screening values for those metals listed above when there was sufficient data to adequately characterize background concentrations (*i.e.*, cadmium, chromium, copper, iron, lead, manganese, nickel, and zinc). The screening for each of 25 lakes is presented in Table 5.8.

**Table 5.7 Screening of Maximum Concentrations of Metals in Sediments (µg/g d.w.)**

| Parameter | Maximum Concentration | Guideline | Ontario Background[a] | Sudbury Background[b] |
|---|---|---|---|---|
| Aluminium | 29,000 | **58,000** | - | 12,000 |
| Antimony | - | **2** | - | - |
| **Arsenic** | **88** | **5.9** | 4.2 | 20[c] |
| Barium | 740 | - | - | 62 |
| Beryllium | 1.8 | - | - | 0.81 |
| **Cadmium** | **10** | 0.6 | 1.1 | **2.6** |
| Calcium | 7,142 | - | - | 4,554 |
| **Chromium** | **111** | 26 | 31 | **58** |
| **Cobalt** | **210** | **50** | - | 25 |
| **Copper** | **3,690** | 16 | 25 | **42** |
| **Iron** | **76,600** | 20,000 | 31,200 | **28,427** |
| **Lead** | **280** | 31 | 23 | **50** |
| Magnesium | 11,600 | - | - | 8,065[c] |
| **Manganese** | **69,000** | 460 | 400 | **1,073** |
| Mercury | - | **0.17** | 0.1 | - |
| Molybdenum | 34 | **13.8** | - | 1.1 |
| **Nickel** | **4,900** | 16 | 31 | **46** |
| **Selenium** | **127** | **1.9** | - | 40[c] |
| Strontium | 43 | - | - | 24 |
| Vanadium | 65 | **35.2** | - | 48 |
| **Zinc** | **680** | 120 | 65 | **131** |

* Bolded values were selected as screening values for the current assessment. Bolded values in grey scale exceeded the corresponding screening value.

[a] Background levels are based on analyses of Great Lakes pre-colonial sediment horizon (MOE, 1993) unless otherwise indicated; these values are presented for comparison purposes only and were not used in screening COC.

[b] See Table H.6

[c] Confidence in this value as an accurate representation of background concentrations is low, therefore, it was not selected as the final screening criterion.

The maximum concentration of 13 metals (arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, molybdenum, nickel, selenium, vanadium and zinc) in sediments exceeded the selected screening criteria (Table 5.7).

**Table 5.8 Screening of Chemicals of Concern in Sediment for 26 Lakes**

| Lake | Sampling Date | pH | Al (µg/g) | As (µg/g) | Sb (µg/g) | Ba (µg/g) | Be (µg/g) | Cd (µg/g) | Ca (µg/g) | Cr (µg/g) | Co (µg/g) | Cu (µg/g) | Fe (µg/g) | Pb (µg/g) | Mg (µg/g) | Mn (µg/g) | Hg (µg/g) | Mo (µg/g) | Ni (µg/g) | Se (µg/g) | Sr (µg/g) | V (µg/g) | Zn (µg/g) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Screening Criterion** | | | 58,000 | 5.9 | 2 | | | 2.6 | | 58 | 50 | 42 | 28,427 | 50 | | 1,073 | 0.17 | 13.8 | 46 | 1.9 | | 35.2 | 131 |
| Clearwater | 1993[a] | 4.10 | 18,000 | - | - | 78 | 0.71 | **7.7** | - | 53 | **80** | **1,900** | 21,000 | **150** | - | 130 | - | 2.5 | **2,100** | - | 20 | **39** | **330** |
| Clearwater | 1993[a] | 4.00 | 18,000 | - | - | 75 | 0.84 | **7.2** | - | 51 | **88** | **1,800** | 26,000 | **150** | - | 140 | - | 2.4 | **2,300** | - | 20 | **40** | **350** |
| Clearwater | 1993[a] | 4.00 | 18000 | - | - | 80 | 0.68 | **4.7** | - | 52 | **61** | **1,600** | 23,000 | **150** | - | 150 | - | 2.2 | **1,700** | - | 21 | **40** | **200** |
| Daisy | 1993[a] | 4.30 | 25,000 | - | - | 110 | 0.81 | 1.7 | - | **69** | 45 | **670** | **29,000** | **57** | - | 230 | - | 0.5 | **1,200** | - | 27 | **45** | 120 |
| Daisy | 1993[a] | 4.50 | 25,000 | - | - | 85 | 0.61 | 1.1 | - | **66** | 45 | **730** | **31,000** | **64** | - | 180 | - | 0.85 | **1,300** | - | 23 | **46** | 85 |
| Daisy | 1993[a] | 4.50 | 24,000 | - | - | 85 | 0.70 | 1.1 | - | **64** | 43 | **760** | **39,000** | **73** | - | 190 | - | 0.74 | **1,100** | - | 23 | **44** | 89 |
| Fairbank | 1993[a] | 5.00 | 13,000 | - | - | 740 | 0.62 | **5.8** | - | 33 | 23 | **280** | **46,000** | **150** | - | **69,000** | - | **34.0** | **350** | - | 43 | **41** | **270** |
| Fairbank | 1993[a] | 4.90 | 13,000 | - | - | 560 | 0.66 | **5.8** | - | 35 | 22 | **260** | **69,000** | **140** | - | **38,000** | - | **25.0** | **320** | - | 36 | **46** | **260** |
| Fairbank | 1993[a] | 5.30 | 15,000 | - | - | 590 | 0.69 | **5.2** | - | 40 | 22 | **250** | **45,000** | **150** | - | **34,000** | - | **20.0** | **310** | - | 39 | **49** | **260** |
| Hannah | 97-01[c] | - | - | - | - | - | - | 2.0 | - | - | - | **1,856** | - | - | - | - | - | - | - | - | - | - | - |
| Hannah | 2000[b] | - | - | - | - | - | - | 0.28 | - | - | - | **2,212** | - | - | - | - | - | - | - | - | - | - | **409** |
| Hannah | 2001[d] | - | - | **87.7** | - | - | - | **2.7** | 2,906 | **64.8** | 47 | **1,450** | **34,151** | **67.1** | - | 264 | - | - | **1,460** | **31.8** | - | **48.4** | **148** |
| Johnnie | 1993[a] | 4.10 | 29,000 | - | - | 52 | 1.80 | **5.1** | - | 47 | 41 | **140** | **42,000** | **130** | - | 560 | - | 2.0 | **210** | - | 25 | **62** | **320** |
| Johnnie | 1993[a] | 4.20 | 28,000 | - | - | 62 | 1.60 | **4.2** | - | 45 | 27 | **100** | **41,000** | **97** | - | 680 | - | 1.3 | **130** | - | 25 | **59** | **220** |
| Johnnie | 1993[a] | 4.20 | 27,000 | - | - | 63 | 1.70 | **5.0** | - | 45 | 27 | **180** | 28,000 | **150** | - | 320 | - | 1.8 | **210** | - | 24 | **56** | **300** |
| Kakakiwaganda | 1996[g] | - | - | - | - | - | - | 2.0 | - | **90** | 31 | **120** | **38,100** | 28 | 9,990 | **1,200** | - | - | 189 | - | - | - | **204** |
| Kakakiwaganda | 1996[g] | - | - | - | - | - | - | **2.63** | - | **81** | 32 | **238** | **42,500** | **134** | 8,890 | 885 | - | - | 263 | - | - | - | **302** |
| Kakakiwaganda | 1998[g] | - | - | - | - | - | - | 1.58 | - | **86** | 33 | **118** | **41,400** | 18 | 10,100 | **1,660** | - | - | 195 | - | - | - | **196** |
| Kakakiwaganda | 1998[g] | - | - | - | - | - | - | **2.74** | - | **82** | 27 | **59** | **56,500** | **59** | 9,880 | 885 | - | - | 77 | - | - | - | **174** |
| Kelly | 97-01[c] | - | - | - | - | - | - | **5.0** | - | - | - | **3,690** | - | - | - | - | - | - | - | - | - | - | - |
| Kelly | 2001[d] | - | - | **46.8** | - | - | - | **4.6** | 6,665 | 49.7 | **196.8** | **2,672** | **37,032** | **75.8** | - | 230 | - | - | **4,680** | **95.7** | - | 30.5 | **394** |
| Kusk | 97-01[c] | - | - | - | - | - | - | 2.0 | - | - | - | **438** | - | - | - | - | - | - | - | - | - | - | - |
| Kusk | 2001[d] | - | - | **36.1** | - | - | - | 1.5 | 4,403 | 46.3 | **80.4** | **349** | **29,562** | **124.3** | - | 380 | - | - | **865** | **40.2** | - | **45.8** | **289** |
| Long | 1993[a] | 4.80 | 20,000 | - | - | 120 | 1.10 | **4.6** | - | 57 | **90** | **1,300** | **31,000** | **110** | - | 510 | - | 0.5 | **1,400** | - | 32 | **45** | **290** |
| Long | 1993[a] | 4.70 | 19,000 | - | - | 120 | 1.10 | **4.8** | - | 55 | **90** | **1,300** | **30,000** | **110** | - | 470 | - | 0.5 | **1,400** | - | 30 | **43** | **290** |
| Long | 1993[a] | 4.70 | 19,000 | - | - | 110 | 1.10 | **4.4** | - | 55 | **88** | **1,200** | **30,000** | **110** | - | 460 | - | 0.5 | **1,400** | - | 30 | **44** | **290** |
| Lower Sturgeon | 1996[g] | - | - | - | - | - | - | 2.29 | - | **64** | 32 | **142** | **42,100** | **97** | 4,910 | 896 | - | - | **173** | - | - | - | **230** |
| Lower Sturgeon | 1996[g] | - | - | - | - | - | - | 2.07 | - | **79** | 36 | **94** | **36,600** | **93** | 9,010 | 906 | - | - | **115** | - | - | - | **212** |
| Lower Sturgeon | 1998[g] | - | - | - | - | - | - | 1.58 | - | **73** | 36 | **222** | **63,900** | **115** | 8,320 | **2,150** | - | - | **308** | - | - | - | **264** |
| Lower Sturgeon | 1998[g] | - | - | - | - | - | - | 1.52 | - | **73** | 38 | **43** | **56,900** | 2.5 | 9,140 | **2,100** | - | - | **48** | - | - | - | **194** |
| McCharles | 97-01[c] | - | - | - | - | - | - | 2.0 | - | - | - | **926** | - | - | - | - | - | - | - | - | - | - | - |
| McCharles | 2001[d] | - | - | **79.3** | - | - | - | **3.4** | 5,528 | 51.6 | **187.4** | **1,288** | **33,420** | **75.7** | - | 328 | - | - | **3,654** | **60.7** | - | **44.9** | **289** |
| McFarlane | 1993[a] | 4.70 | 16,000 | - | - | 37 | 0.66 | **6.4** | - | 50 | **110** | **1,200** | **32,000** | **110** | - | 910 | - | 0.74 | **2,200** | - | 30 | **40** | **420** |
| McFarlane | 1993[a] | 4.50 | 17,000 | - | - | 26 | 0.70 | **7.1** | - | 58 | **130** | **1,200** | **34,000** | **120** | - | **1,100** | - | 0.5 | **2,400** | - | 32 | **41** | **460** |
| McFarlane | 1993[a] | 4.50 | 17,000 | - | - | 33 | 0.68 | **6.9** | - | 51 | **120** | **1,200** | **33,000** | **110** | - | 970 | - | 0.5 | **2,200** | - | 31 | **41** | **450** |
| McFarlane | 1996[g] | - | - | - | - | - | - | **5.40** | - | **83** | **117** | **1,070** | **28,600** | **83** | 8,930 | 764 | - | - | **2,430** | - | - | - | **492** |

**Table 5.8 Screening of Chemicals of Concern in Sediment for 26 Lakes**

| Lake | Sampling Date | pH | Al (µg/g) | As (µg/g) | Sb (µg/g) | Ba (µg/g) | Be (µg/g) | Cd (µg/g) | Ca (µg/g) | Cr (µg/g) | Co (µg/g) | Cu (µg/g) | Fe (µg/g) | Pb (µg/g) | Mg (µg/g) | Mn (µg/g) | Hg (µg/g) | Mo (µg/g) | Ni (µg/g) | Se (µg/g) | Sr (µg/g) | V (µg/g) | Zn (µg/g) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Screening Criterion** | | | 58,000 | 5.9 | 2 | | | 2.6 | | 58 | 50 | 42 | 28,427 | 50 | | 1,073 | 0.17 | 13.8 | 46 | 1.9 | | 35.2 | 131 |
| McFarlane | 1996[g] | - | - | - | - | - | - | 8.82 | - | 78 | 142 | 1,810 | 37,500 | 150 | 7,370 | 3,400 | - | - | 2,780 | - | - | - | 554 |
| McFarlane | 1998[g] | - | - | - | - | - | - | 1.86 | - | 74 | 34 | 262 | 45,000 | 79 | 6,310 | 1,020 | - | - | 509 | - | - | - | 180 |
| McFarlane | 1998[g] | - | - | - | - | - | - | 9.03 | - | 70 | 137 | 1,570 | 62,500 | 150 | 7,930 | 6,900 | - | - | 2,700 | - | - | - | 451 |
| Minnow | 1984[f] | - | - | - | - | - | - | - | - | - | - | 1,130 | - | - | - | - | - | - | 2,500 | - | - | - | 545 |
| Mud | 97-01[c] | - | - | - | - | - | - | 2.0 | - | - | - | 1,038 | - | - | - | - | - | - | - | - | - | - | - |
| Mud | 2001[d] | - | - | 18.1 | - | - | - | 1.6 | 7,142 | 49.9 | 58.3 | 746.5 | 26,019 | 23.8 | - | 206 | - | - | 1,195 | 51.4 | - | 39.1 | 144 |
| Nepahwin | 1984[f] | - | - | - | - | - | - | - | - | - | - | 2,900 | - | - | - | - | - | - | 4,490 | - | - | - | 448 |
| Nepahwin | 1993[a] | 4.70 | 22,000 | - | - | 20 | 0.88 | 9.8 | - | 72 | 200 | 3,200 | 47,000 | 270 | - | 1,700 | - | 2.6 | 4,400 | - | 34 | 53 | 650 |
| Nepahwin | 1993[a] | 4.40 | 22,000 | - | - | 22 | 0.87 | 10.0 | - | 72 | 200 | 2,900 | 44,000 | 260 | - | 1,700 | - | 2.8 | 4,200 | - | 35 | 53 | 670 |
| Nepahwin | 1993[a] | 4.60 | 22,000 | - | - | 19 | 0.88 | 10.0 | - | 74 | 210 | 3,200 | 46,000 | 280 | - | 1,700 | - | 2.9 | 4,600 | - | 33 | 53 | 680 |
| Nelson | 1984[f] | - | - | - | - | - | - | - | - | - | - | 375 | - | - | - | - | - | - | 444 | - | - | - | 270 |
| Nepewassi | 1996[g] | - | - | - | - | - | - | 6.37 | - | 90 | 105 | 137 | 27,600 | 30 | 9,640 | 701 | - | - | 289 | - | - | - | 249 |
| Nepewassi | 1998[g] | - | - | - | - | - | - | 1.29 | - | 107 | 29 | 138 | 50,400 | 63 | 7,780 | 344 | - | - | 1,920 | - | - | - | 240 |
| Raft | 1996[g] | - | - | - | - | - | - | 0.42 | - | 44 | 18 | 135 | 18,600 | 2.5 | 5,470 | 324 | - | - | 178 | - | - | - | 47 |
| Raft | 1996[g] | - | - | - | - | - | - | 4.99 | - | 72 | 54 | 1,470 | 35,200 | 138 | 7,310 | 363 | - | - | 1,570 | - | - | - | 142 |
| Raft | 1998[g] | - | - | - | - | - | - | 1.35 | - | 79 | 17 | 35 | 76,600 | 10 | 11,600 | 702 | - | - | 53 | - | - | - | 146 |
| Raft | 1998[g] | - | - | - | - | - | - | 2.60 | - | 67 | 32 | 536 | 44,100 | 52 | 7,970 | 389 | - | - | 773 | - | - | - | 137 |
| Ramsey | 1993[a] | 4.40 | 19,000 | - | - | 69 | 0.75 | 7.3 | - | 62 | 160 | 2,900 | 43,000 | 240 | - | 430 | - | 1.5 | 4,100 | - | 32 | 52 | 400 |
| Ramsey | 1993[a] | 4.40 | 20,000 | - | - | 51 | 0.79 | 8.5 | - | 70 | 190 | 3,200 | 47,000 | 270 | - | 420 | - | 1.2 | 4,900 | - | 33 | 54 | 460 |
| Ramsey | 1993[a] | 4.50 | 21,000 | - | - | 140 | 0.81 | 6.4 | - | 76 | 160 | 2,700 | 44,000 | 220 | - | 420 | - | 1.0 | 3,900 | - | 38 | 57 | 360 |
| Ramsey | 1996[g] | - | - | - | - | - | - | 2.43 | - | 96 | 71 | 1,300 | 42,600 | 95 | 8,950 | 487 | - | - | 1,590 | - | - | - | 192 |
| Ramsey | 1998[g] | - | - | - | - | - | - | 4.72 | - | 83 | 94 | 1,840 | 58,300 | 143 | 9,690 | 508 | - | - | 2,130 | - | - | - | 258 |
| Ramsey | 97-01[c] | - | - | - | - | - | - | 3.0 | - | - | - | 1,575 | - | - | - | - | - | - | - | - | - | - | - |
| Ramsey | 2000[b] | - | - | - | - | - | - | 0.94 | - | - | - | 651 | - | - | - | - | - | - | - | - | - | - | 281 |
| Ramsey | 2001[d] | - | - | 67.9 | - | - | - | 2.8 | 2,672 | 36.3 | 58.8 | 1,606 | 26,187 | 67.8 | - | 292 | - | - | 1,890 | 33.4 | - | 33.2 | 180.5 |
| Richard | 1996[g] | - | - | - | - | - | - | 1.41 | - | 111 | 38 | 165 | 37,100 | 30 | 9,640 | 701 | - | - | 289 | - | - | - | 249 |
| Richard | 1998[g] | - | - | - | - | - | - | 38 | - | 67 | 74 | 1,120 | 41,600 | 63 | 7,780 | 344 | - | - | 1,920 | - | - | - | 240 |
| Simon | 97-01[c] | - | - | - | - | - | - | 7.0 | - | - | - | 2,412 | - | - | - | - | - | - | - | - | - | - | - |
| Simon | 2001[d] | - | - | 51.7 | - | - | - | 4.5 | 7,047 | 52.5 | 205 | 1,659 | 34,855 | 68.2 | - | 290 | - | - | 4,745 | 127 | - | 44.6 | 302 |
| Trout | 1996[g] | - | - | - | - | - | - | 3.14 | - | 51 | 33 | 127 | 39,500 | 79 | 4,300 | 1,940 | - | - | 220 | - | - | - | 232 |
| Trout | 1996[g] | - | - | - | - | - | - | 1.77 | - | 73 | 32 | 54 | 35,100 | 14 | 9,190 | 2,380 | - | - | 99 | - | - | - | 228 |
| Trout | 1998[g] | - | - | - | - | - | - | 2.85 | - | 53 | 31 | 156 | 38,800 | 92 | 5,110 | 1,400 | - | - | 287 | - | - | - | 248 |
| Trout | 1998[g] | - | - | - | - | - | - | 3.02 | - | 67 | 44 | 173 | 54,200 | 132 | 5,500 | 1,160 | - | - | 239 | - | - | - | 272 |
| Tyson | 1984[f] | - | - | - | - | - | - | - | - | - | - | 220 | - | - | - | - | - | - | 300 | - | - | - | 235 |
| Tyson | 1993[a] | 4.40 | 23,000 | - | - | 150 | 1.40 | 4.1 | - | 45 | 33 | 200 | 64,000 | 150 | - | 1,600 | - | 1.5 | 280 | - | 29 | 65 | 230 |
| Tyson | 1993[a] | 4.40 | 23,000 | - | - | 110 | 1.40 | 3.7 | - | 44 | 45 | 180 | 73,000 | 140 | - | 2,200 | - | 2.0 | 270 | - | 29 | 65 | 200 |

**Table 5.8 Screening of Chemicals of Concern in Sediment for 26 Lakes**

| Lake | Sampling Date | pH | Al (µg/g) | As (µg/g) | Sb (µg/g) | Ba (µg/g) | Be (µg/g) | Cd (µg/g) | Ca (µg/g) | Cr (µg/g) | Co (µg/g) | Cu (µg/g) | Fe (µg/g) | Pb (µg/g) | Mg (µg/g) | Mn (µg/g) | Hg (µg/g) | Mo (µg/g) | Ni (µg/g) | Se (µg/g) | Sr (µg/g) | V (µg/g) | Zn (µg/g) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Screening Criterion** | | | 58,000 | 5.9 | 2 | | | 2.6 | | 58 | 50 | 42 | 28,427 | 50 | | 1,073 | 0.17 | 13.8 | 46 | 1.9 | | 35.2 | 131 |
| Tyson | 1993[a] | 4.10 | 24,000 | - | - | 140 | 1.50 | **4.0** | - | 47 | 33 | **220** | **59,000** | **150** | - | 840 | - | 1.6 | **300** | - | 29 | **64** | **230** |
| Whitson | 1993[a] | 4.70 | 17,000 | - | - | 84 | 0.63 | **2.8** | - | 49 | 48 | **1,100** | **52,000** | **160** | - | 250 | - | 0.93 | **1,400** | - | 32 | **47** | 130 |
| Whitson | 1993[a] | 4.70 | 15,000 | - | - | 66 | 0.52 | 2.2 | - | 46 | **53** | **760** | **43,000** | **120** | - | 410 | - | 0.75 | **1,100** | - | 26 | **42** | 110 |
| Whitson | 1993[a] | 4.80 | 15,000 | - | - | 66 | 0.52 | 2.3 | - | 44 | **57** | **780** | **46,000** | **130** | - | 500 | - | 0.72 | **1,100** | - | 26 | **42** | 110 |
| Whitson | 97-01[c] | - | - | - | - | - | - | **6.0** | - | - | - | **1,470** | - | - | - | - | - | - | - | - | - | - | - |
| Whitson | 1999[e] | - | - | - | - | - | - | **7.3** | - | - | - | **1,140** | - | - | - | - | - | - | **1,518** | - | - | - | **215** |
| Whitson | 2000[b] | - | - | - | - | - | - | **4.6** | - | - | - | **1,627** | - | - | - | - | - | - | - | - | - | - | **586** |
| Whitson | 2001[d] | - | - | **77.5** | - | - | - | **3** | 3,387 | 36.2 | **66.9** | **1,916** | 28,408 | **204.7** | - | 307 | - | - | **1,303** | **38** | - | **42.3** | **177** |
| Whitewater | 1984[f] | - | - | - | - | - | - | - | - | - | - | **1,330** | - | - | - | - | - | - | **4,390** | - | - | - | **340** |
| **Maximum** | | | 29,000 | **88** | - | 740 | 2 | **10** | 7,142 | **111** | **210** | **3,690** | **76,600** | **280** | 11,600 | **69000** | - | **34** | **4,900** | **127** | 43 | **65** | **680** |

**References**

[a] Keller, B., Heneberry, J., Gunn, J.M., Snucins, E., Morgan, G. and Leduc, J. 2004. Recovery of Acid and Metal Damaged Lakes Near Sudbury Ontario: Trends and Status. Cooperative Freshwater Ecology Unit. Department of Biology, Laurentian University. Sudbury, Ontario, Canada.

[b] Couture, P. and P.R. Kumar. 2003. Impairment of metabolic capacities in copper and cadmium contaminated wild yellow perch (*Perca flavescens*). Aquatic Toxicology. 64(1):107-120.

[c] Couture, P. and Rajotte, J.W. 2003. Morphometric and Metabolic Indicators of Metal Stress in Wild Yellow Perch (*Perca flavescens*) from Sudbury, Ontario: A Review. J. Environ. Monit. 5: 216- 221. Note: Values are the mean of concentrations measured in studies from 1997 to 2001.

[d] Pyle, G.G., J.W. Rajotte and P. Couture. 2005. Effects of Industrial Metals on Wild Fish Populations along a Metal Contamination Gradient. Ecotoxicology and Environmental Saftey. In Press. Note: Values are the mean of 3 samples.

[e] Audet, D. and Couture, P. 2003. Seasonal Variations in Tissue Metabolic Capacities of Yellow Perch (*Perca flavescens*) from Clean and Metal-Contaminated Environments. Can. J. Fish. Aquat. Sci. 60: 269- 278. Note: Values are pooled for spring, summer and fall samples.

[f] Bradley, R.W. and Morris, J.R. 1986. Heavy Metals in Fish from a Series of Metal-Contaminated Lakes Near Sudbury, Ontario. Water, Air, and Soil Pollution. 27: 341-354.

[g] Bormann, U., T.B. Reynoldson, F. Rosa, and W.P. Norwood. 2001b. Final Report on the Effects of Atmospheric Deposition of Metals from the Sudbury Smelters on Aquatic Ecosystems. NWRI Contribution No. 01-023. Note: When multiple samples were available for a given year and similar sample location, the maximum value was selected.

Note: Bolded values in grey scale indicate concentrations that exceed the screening criterion

*This page left blank intentionally*

Concentrations of mercury in sediments from the study area were not available in the primary literature (Tables 5.7 and 5.8). However, given general public and scientific concerns regarding mercury in fish tissue and subsequent effects on consumers, it seems prudent that future studies examine mercury concentrations in lake sediments, and possibly fish tissue, and apply screening criteria as outlined in this report to determine if mercury qualifies as a COC for future studies.

Screening criteria were not available for five metals (barium, beryllium, calcium, magnesium, and strontium). None of these metals have been associated with smelter emissions (SARA, 2004), and none were retained as COC in soil or water. Due to the small number of samples that were identified for the purpose of characterizing regional background concentrations (n=1 for barium, beryllium, and strontium; n= 4 for calcium and magnesium), confidence in these values was considered to be low, therefore, screening against regional background levels was not considered to be relevant. It is noted (Table 5.7) that barium, manganese and molybdenum concentrations are approximately 10-times higher in the sediments of Fairbank Lake than other lakes. The significance of this is unknown, as this lake is located approximately 55 km west of downtown Sudbury and water quality data for this lake are not available. Thus, these elevated metals levels are unlikely to be related to the smelters. Therefore, since there is no available toxicological information to indicate that levels of these chemicals in the sediment of Sudbury lakes have the potential to adversely affect aquatic organisms, it is recommended that barium, beryllium, calcium, magnesium, and strontium not be retained as COC in sediment.

Of the ten lakes for which concentrations of molybdenum in sediment are available, only Fairbank Lake contained concentrations (34, 25, and 20 µg/g) in excess of the screening criterion (13.8 µg/g). Since only a single measurement of molybdenum was available for the derivation of background concentrations in sediments (1.1 µg/g), use of this parameter as a screening criterion was not considered to be appropriate. Concentrations of molybdenum presented in Table 5.8 (maximum concentration of 34 µg/g) are within the observed environmental range (2 to 400 µg/g) identified in other studies (Chappell, 1975; Webb *et al.*, 1968). In the aquatic environment, molybdenum is influenced by pH, and will remain in solution at pH>5 (LeGrande and Runnells, 1975). Since lakes to be considered in the aquatic ERA are those that have not been heavily acidified, it is likely that molybdenum will primarily remain within the water column. Since concentrations of molybdenum in sediment were only found in excess of the screening criterion in a single lake, and concentrations in surface water did not exceed the screening criterion, it is not anticipated that it is found at levels in the aquatic environment that have the potential to cause significant adverse effects. Therefore, it is not recommended that molybdenum be retained as a COC in sediment.

### 5.6.4 Final List of Recommended COC for Sediment

Metals found above screening guidelines and background concentrations, and which are known to be present in airborne particulate emissions from the Sudbury smelters, are recommended as COC. Therefore, the recommended final list of COC within the sediments of Sudbury lakes includes:

- Arsenic
- Cadmium
- Chromium
- Cobalt
- Copper
- Iron
- Lead
- Mercury
- Manganese
- Nickel
- Selenium
- Vanadium
- Zinc

## 5.7 Summary of COC in Surface Water and Sediment

The recommended final list of COC in surface water and sediment includes:

| Surface Water | Sediment |
|---|---|
| Aluminium | - |
| - | Arsenic |
| Cadmium | Cadmium |
| - | Chromium |
| Cobalt | Cobalt |
| Copper | Copper |
| Iron | Iron |
| Lead | Lead |
| - | Mercury |
| - | Manganese |
| Nickel | Nickel |
| Selenium | Selenium |
| - | Vanadium |
| - | Zinc |

## 5.8 Acidity and Alkalinity as Modifying Factors

The available literature on acid precipitation and its effects on aquatic ecosystems is voluminous. After Scandinavia, the effects of acid precipitation were first observed in the Killarney lakes south of Sudbury in the late 1960s and early 1970s. This became the subject of focused research by government and academia for the next two decades. This report makes no attempt to review or synthesize what is known about acid precipitation and the monitoring work that has taken place around Sudbury. Rather, the subject of lake acidity, alkalinity and pH as factors modifying the toxicity of metals to aquatic biota is introduced.

The pH of surface waters can affect the health and survival of aquatic organisms through both direct toxicity mechanisms and through the modification of the toxic effects of other compounds. A pH of 6.0 is commonly used to indicate the level below which effects will begin to occur to most acid-sensitive components of the biological community including aquatic insects, crustaceans, and small fish (Keller and Gunn, 1994). Lakes which maintain a pH at or above this, but which have diminished biological communities, could be used to demonstrate the effects of metals. Indirectly, surface water pH can impact aquatic life by altering the toxicity of metals. This can occur by either making organisms more susceptible to the toxic effects of metals *via* increased bioavailability or by increasing their concentrations within water as a result of increased mobilization from soils and sediments.

Levels of aluminium in lakes within Killarney Provincial Park (outside the study area) were found to increase with decreasing pH (Dixit *et al*., 1992). This is believed to be a result of increased mobilization of aluminium from watershed soils and lake sediments. This trend of decreasing pH and increasing concentrations of aluminium in Lumsden Lake during the period of 1960 to 1980 resulted in a reduction in the abundance of the acid- and metal-sensitive diatom species *Asterionella ralfsii v. Americana*. Following increases in pH beginning around 1980, aluminium concentrations started to decline and populations of *Asterionella ralfsii v. Americana* begin to increase. The disappearance of lake trout and the extirpation of the slimy sculpin, white sucker, lake herring and trout-perch were also attributed to the acidification of Lumsden Lake and the subsequent increase in aluminium concentrations (Harvey, 1975). Similar trends were observed in Acid Lake within Killarney Provincial Park. Increasing levels of aluminium beginning in 1960 corresponded with decreasing pH. Acid- and metal-sensitive diatom species started to decline at this time, and sampling in 1972 and 1973 found no occurrences of fish populations (Harvey, 1975). Fish were able to survive in Acid Lake prior to 1960 despite the lake having an estimated pH of 5.0 to 5.25. It was not until levels of aluminium increased that the disappearance of fish in Acid Lake occurred (Harvey, 1975).

Surface water pH can also influence the impact of aluminium on aquatic organisms by modifying its direct toxicity. The toxicity of aluminium to the leopard frog has been observed to increase with increasing water pH, whereas aluminium toxicity is inversely related to water pH for the American toad (Freda, 1991). In surface waters with lethal pHs, aluminium has been observed to reduce mortality in spotted salamander and leopard frog embryos (Clark and LaZerte, 1987; Freda and McDonald, 1990).

As a result of increased mobilization, Sudbury lakes that suffer from acidification may have lower concentrations of metals within sediments than lakes with near neutral pH. This was determined through the collection of sediment cores during the summer of 1985 from the acid stressed Silver Lake and the non-acid stressed Lake McFarlane. Both lakes showed a sharp increase in concentrations of Ni, Pb, Co, and Zn in sediments starting at a depth of 13 to14 cm. Assuming that this corresponds with the start of the mining and smelting operations around 1883, the sedimentation rate is estimated to be 1.4 mm *per* year. Although the two lakes have similar slopes of Ni profiles during this period of sharp increase, the total amount stored is greater in the sediments of Lake McFarlane compared to Silver Lake (Nriagu and Rao, 1987). Previous studies have shown that pH values lower than 4.5 reduce the ability of sediment particles to retain metals (Nriagu and Gaillard, 1984) and destabilize precipitated metal compounds within the sediment (Arafat and Nriagu, 1986).

While decreasing pH is generally considered to increase metal toxicity and uptake by increasing solubility, alkalinity does the opposite. Alkalinity in natural water-bodies is generally associated with hardness, which is the total amount of the divalent cations Mg and Ca. Both Mg and Ca have been shown to compete directly with metals for gill binding sites (Playle, 1998). These bicarbonate ions, as well as dissolved organic carbon (DOC), reduce the bioavailability of metals by complexing them (Lauren and McDonald, 1986; Playle *et al*., 1993; Playle, 1998).

For further information on the effects of acid precipitation, mechanisms and recovery of lakes in the Sudbury region, the interested reader is referred to Volume 32(3) of the journal Ambio (April 2003), which is a special issue devoted to Biological Recovery from Acidification: Northern Lakes Recovery Study.

## 5.9 Selection of Valued Ecosystem Components

A VEC is an ecological species, population or community that is important to people, has economic and/or social value, is ecologically significant and can be evaluated in the risk assessment. Identification and selection of candidate VECs is a critical step to ensure that all relevant ecological groups within the Sudbury aquatic environment are adequately represented. Four general groups of VECs were selected for the current assessment: fish, aquatic invertebrates, aquatic plants, and amphibians. Wildlife VECs with a significant portion of their diet arising from the aquatic environment (*i.e.*, common loon, mallard, mink) were also selected following the process detailed in Chapter 2 of the main ERA report; refer to Chapter 2 and associated appendices for further details. These wildlife VECs are discussed further in Section 5.15. The general process for aquatic VEC selection is illustrated in Figure 5-4.

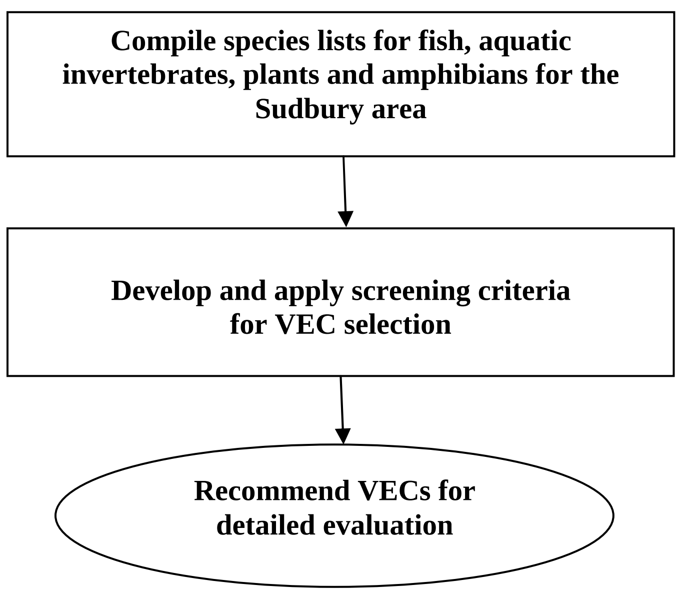



**Figure 5-4 Process for the Identification of VECs in the Aquatic Environment**

Primary literature, academic resources, government reports, and other information were all considered in the selection of VECs. The selection of VECs for each of the four major ecological groups is detailed in Section 5.9.1.

### 5.9.1 VEC Selection

The VEC selection process is presented in Sections 5.9.2 through 5.9.5 for fish, invertebrates, plants and amphibians, respectively. A summary of recommended VECs is provided in Section 5.9.6.

### 5.9.2 Fish

The Co-Operative Freshwater Ecology Unit of Laurentian University conducted a survey between 2000 and 2004 to collect information on the number of fish species present in 25 Sudbury area lakes (Co-Op, 2004). This survey identified 36 fish species that occurred in the Sudbury area. This survey was used as an initial step in the selection of fish species as potential VECs (Table 5.9).

**Table 5.9 Fish Species and the Percentage of Sudbury Area Lakes in which they are Found**

| Species | Presence in Sudbury Lakes (%) |
|---|---|
| Blacknose dace *(Rhinichthys atratulus)* | 4 |
| Blacknose Shiner *(Notropis heterolepis)* | 8 |
| Bluegill *(Lepomis macrochirus)* | 2 |
| Bluntnose minnow *(Pimephales notatus)* | 12 |
| Brown bullhead (*Ictalurus nebulosus)* | 76 |
| Burbot *(Lota lota)* | 12 |
| Cisco/ lake herring *(Coregonus artedii)* | 24 |
| Common shiner (*Notropis cornutus*) | 24 |
| Creek chub *(Semotilus atromaculatus)* | 16 |
| Emerald Shiner *(Notropis antherinoides)* | 4 |
| Fathead minnow *(Pimephales promelas)* | 12 |
| Finescale dace *(Phoxinus heogaeus)* | 4 |
| Golden shiner *(Notemigonus crysoleucas)* | 16 |
| Iowa darter *(Etheostoma exile)* | 20 |
| Lake chub *(Couesius plumbeus)* | 8 |
| Lake trout *(Salvelinus namaycush)* | 24 |
| Lake whitefish *(Coregonus clupeaformis)* | 12 |
| Largemouth bass *(Micropterus salmides)* | 8 |
| Logperch *(Percina caprodes)* | 24 |
| Mottled sculpin (Cottus bairdi) | 4 |
| Ninespine stickleback *(Pungitius pungitius)* | 4 |
| Northern pike (*Esox lucieus)* | 64 |
| Northern redbelly dace *(Chrosomus eos)* | 4 |
| Pearl dace *(Semotilus margarita)* | 8 |
| Pumkinseed *(Lepomis gibbosus)* | 64 |
| Rainbow smelt (Osmerus mordax) | 12 |
| Rock bass *(Ambloplites rupestris)* | 36 |
| Slimy sculpin *(Cottus cognatus)* | 8 |
| Smallmouth bass *(Micropterus dolomieui)* | 76 |
| Splake *(Salvelinus namaycush/Salvelinus fontinalis)* | 4 |
| Spoonhead sculpin *(Cottus ricei)* | 4 |
| Spottail shiner *(Notropis hudsonius)* | 24 |

**Table 5.9 Fish Species and the Percentage of Sudbury Area Lakes in which they are Found**

| Species | Presence in Sudbury Lakes (%) |
|---|---|
| Trout-perch *(Percopsis omiscomaycus)* | 8 |
| Walleye *(Stizostedion vitreum)* | 64 |
| White sucker *(Catostomus commersoni)* | 68 |
| Yellow perch *(Perca flavescens)* | 84 |

To reduce the candidate species list to the final list of selected fish VECs, a number of criteria were considered to ensure that the most appropriate species were selected and that all major ecological groups were represented. The approach to developing the VEC selection criteria was a Sudbury-specific approach modified from the approach outlined by Suter (1993), Becker *et al.* (1998), CCME (1996; 1997), U.S. EPA (1998) and MOE (2005), and included:

- Sensitive species (including species at risk);
- Potential for high exposure to the COC (*e.g.*, benthic fish in close association with contaminated sediment);
- Ecological significance (this reflects the species' role in the ecosystem as an important predator or prey species; non-native or pest species were considered to have low ecological significance);
- Identified by a stakeholder as being important;
- A connection to the human health risk assessment (*e.g.*, sportfish);
- Information existed on populations in the area;
- Toxicity data for COC were available for closely-related species; and
- Represented a major feeding guild (and trophic level).

These criteria were used for screening candidate fish species for consideration as VECs (Table 5.10). None of the fish species identified in Sudbury lakes were classified as species at risk in Ontario (MNR, 2004). For the purpose of this assessment, species that were found in less than 25% of those lakes surveyed were eliminated from the list of potential VECs. This was done because field surveys may be required to assess effects on candidate VECs. The criteria "connection to human health" and "identified by a stakeholders" were combined and referred to as "societal importance" in the screening (Table 5.10). Fish species that were identified by the public at the "Have Your Say" workshops (SARA, 2003) included northern pike, whitefish, walleye, catfish, trout and minnows. To ensure that each feeding guild or trophic level was represented by the selection of VECs, feeding and behavioural traits were also included in the evaluation. Finally, those species that have been the focus of Sudbury-specific studies or general toxicity assays involving the current COC, were given priority over those species that have limited information available describing their status in Sudbury lakes and their sensitivity to metal exposure.

The species meeting the greatest number of criteria were selected as VECs (Table 5.10). However, the VEC selection process also ensured that representatives of various habitats and feeding guilds were selected. Therefore, while there was a tie in the number of species that met two criteria, the white sucker was chosen because no other benthic species was identified. In addition, the common shiner was selected as a forage fish.

Relative sensitivity of fish species to metals is difficult to use as a screening criterion. There are insufficient chronic toxicity data to make this comparison (Brix *et al*., 2005). Relative sensitivity often varies with metal, and can vary with water parameters such as hardness. Also, salmonids are often over-represented in the datasets (Brix *et al*., 2005). While recognizing these shortcomings, the data do suggest that salmonids may be more sensitive than many species to metals.

These criteria were applied to determine which species would be recommended as the representative VEC for a particular trophic level and feeding guild. The fish species recommended as VECs for the current assessment, and the rationale for their selection, are as follows:

Common shiner (*Notropis cornutus*)

The common shiner was selected to represent species at the secondary trophic level that serves as an important link between primary producers and piscivorous species. Shiners, darters, chubs, and dace minnows all serve as important forage species to piscivorous fish, such as yellow perch, as well as top predators, such as northern pike and walleye. Although the common shiner was not found in more than 25% of those Sudbury lakes surveyed, it was the most common of all forage fish and was present in 24% of the surveyed lakes. The common shiner is a member of the minnow family (Cyprinidae). Shiners are commonly herbivorous and planktivorous but are also known to feed on small insects at the water's surface. They are typically 6 to 10 cm long but may reach lengths of 20 cm (Kraft *et al*., 2003).

Lake Trout (*Salvelinus namaycush*)

Lake trout are the largest of the trout. Two distinguishing features of lake trout are its tail, which is deeply forked, and its colour which generally is dappled. The body, head and fins of the lake trout are also covered with light coloured spots, which vary with the habitat from grey to brown or green. The latin name of lake trout indicates that it is a member of the char *(Salvelinus)* group, and that it inhabits deep water *(namaycush)*. Lake trout are one of the largest freshwater fish, and it was selected to represent pelagic species at the top trophic level. In lakes that do not have pelagic forage, lake trout could adapt to a planktivorous lifestyle. Where pelagic lake trout tend to be rare, planktivorous populations of lake trout are highly abundant, but mature slowly and are smaller in size. In lakes that do contain deep water forage, lake trout can adapt to become piscivorous. Piscivorous lake trout can grow much more quickly, mature at a larger size but are less abundant. Lake trout in Ontario can be found in Lake Ontario, Lake Huron, Lake Superior and across the deep, cold lakes of the Canadian Shield. The flesh of the lake trout range in colour from pale to deep pink, and have a delicate taste of superb eating quality. Hence, it is eagerly sought by commercial, sport, and subsistence fishermen (Fisheries and Oceans Canada, 2006).

**Table 5.10 Ranking of Fish Species for Selection of Representative VECs for the ERA**

| Ecological receptors | Selection Criteria | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Status | | Societal importance | Exposure Evaluation | | Effect Information | | |
| **Fish** | VTE[b] | Ecological Significance[c] | Recreational/ Sportfish | Habitat Zone[d] | Feeding Guil [e] | Information on Local Populations | Toxicity Data Available for COC | Total No. of "YES" |
| Brown bullhead (*Ictalurus nebulosus)* | NO | NO | NO | B | B | NO | NO | 0 |
| Common shiner (*Notropis cornutus*)[a] | NO | YES | NO | P | P,I | NO | YES | 2 |
| Lake trout (*Salvelinus namaycush*)[a] | NO | YES | YES | P | TP | YES | YES | 4 |
| Northern pike (*Esox lucieus)* | NO | YES | YES | P | TP | NO | NO | 2 |
| Pumkinseed (*Lepomis gibbosus)* | NO | NO | NO | P | P,I | NO | YES | 1 |
| Rock bass (*Ambloplites rupestris)* | NO | NO | NO | P | P,I | NO | NO | 0 |
| Smallmouth bass (*Micropterus dolomieui*) | NO | YES | YES | P | TP,I | NO | NO | 2 |
| Walleye (*Stizostedion vitreum)* | NO | YES | YES | P/B | TP,B | YES | NO | 3 |
| White sucker (*Catostomus commersoni)* | NO | YES | NO | B | B | NO | YES | 2 |
| Yellow perch (*Perca flavescens)* | NO | YES | YES | P | P,I | YES | YES | 4 |

**Notes:**

[a] The common shiner and lake trout were not found in more than 25% of Sudbury Lakes but were selected to represent lower trophic level (forage) fish, and salmonids, respectively.

[b] VTE indicates Vulnerable, Threatened or Endangered Species.

[c] Ecological Significance reflects role in ecosystem as important predator or prey species. Non-native species or pest species are not considered to have ecological significance (related to VEC selection).

[d] Habitat zone refers to whether the species lives in open water (P – Pelagic) or along the bottom of the lake (B – Benthic).

[e] Feeding Guild as adults indicated by TP (Top piscivore, includes fish at or near the top of the food chain), P (Piscivore), I (Insectivore, includes fish that consume adult and larval insects), B (benthivore, includes species that consume a variety of food types found on along the lake bottom).

White sucker (*Catostomus commersoni)*

The white sucker was selected to represent benthic species at the secondary trophic level. It is a member of the family Catostomidae and is typically 25 to 50 cm long and weighs 0.5 to 1 kg (Kraft *et al*., 2003). They are opportunistic feeders, scouring lake bottoms and acquiring food such as worms, snails, insect larvae, and anything that may be taken in using non-selective suction. Although they are often regarded as pest species by most anglers, the white sucker and other benthic fish may experience a higher level of exposure to COC as a result of their close association with contaminated sediments. White sucker also serve as an important food source to top predators such as northern pike and walleye. The white sucker was chosen over other benthic species such as the brown bullhead or carp as a result of a large database of toxicity information available for this species. White suckers have been used extensively in Canada to monitor the effects of acid precipitation. In addition, they are a common species used for Environmental Effects Monitoring (EEM) under the *Fisheries Act* in the mining and pulp and paper sectors. As such, there is a considerable body of information available on biological metrics such as growth rates, age at maturation, fecundity, liver somatic index (LSI) and gonadal somatic index (GSI) (Beamish and Harvey, 1972; Environment Canada, 1997).

Yellow perch (*Perca flavescens)*

The yellow perch was selected to represent piscivorous fish. Perch also feed on insects, crustaceans and snails as adults. The primary food source for juveniles is zooplankton, and the effects on juvenile perch populations around smelters have been studied (Sherwood *et al*., 2000; 2002). Perch are a member of the family Percidae and are generally 15 to 25 cm long, weighing 100 to 350 g (Kraft *et al.*, 2003). They are the most common species of fish found in the Sudbury area, largely as a result of their ability to adapt to a broad range of environmental conditions, including low pH and elevated levels of metals. They are an important recreational fish due to their abundance and desirable taste. Perch also serve as an important food item to top piscivores such as northern pike. Perch have been the focus of numerous Sudbury-based studies, examining responses to low pH, accumulation of metals into various tissues, population dynamics, and metabolic and physical indications of toxicity.

Walleye (*Stizostedion vitreum)*

Walleye were chosen to represent top piscivores and important sportfish in Sudbury lakes. They are generally regarded as the most sought after sportfish in Sudbury and populations have been established in numerous lakes as a result of stocking efforts through the Community Fisheries Involvement Program (CFIP) (Keller *et al*., 2004). Walleye represent an important connection between the aquatic environment and human health, being a preferred choice for consumption. They are also a member of the family Percidae, but are much larger than their relative the yellow perch, reaching weights approaching 9 kg (Kraft *et al*., 2003). Like other top piscivores, such as northern pike and lake trout, walleye consume large numbers of forage fish and can reach large sizes. Chemicals that have the ability to bioaccumulate up the food chain will have the potential to reach significant levels in these top predators.

### 5.9.3 Invertebrates

Two separate groups of invertebrates are recommended as VECs: benthic invertebrates and zooplankton. Both are important food sources for numerous species of fish at varying life stages, but as a result of differences in physiology and behaviour, they receive different exposures to metals in water and sediment.

**Benthic Invertebrates**

Benthic invertebrates are relatively immobile organisms and will often remain within a small area of sediment for prolonged periods of time. Since they live and feed in close association with the sediment, their primary route of exposure to COC will typically be to COC bound to deposited organic material or found within sediment pore water. COC that have high affinity for organic material will accumulate within sediments and have the potential to bioaccumulate, beginning with benthic invertebrates. As a result, when selecting species as VECs, recognition of the potential for biological uptake and transfer to upper trophic levels was considered.

Benthic invertebrates not only serve an important ecological role as a food source for fish, but also as facilitators of energy transfer. Many species graze on large pieces of organic material such as leaf litter and macrophytes, making these materials more available for nutrient cycling (Heneberry, 1997). Despite dramatic improvements in sediment and water quality, many Sudbury lakes still have low benthic community diversity and low overall population sizes of grazers. Populations of molluscs, amphipods, mayflies and crayfish have been found to be uncharacteristically low in many Sudbury lakes (Gunn and Keller, 1995; Heneberry, 1997; Reasbeck, 1997; Borgmann *et al*., 1998), which may account for the extensive growth of filamentous algae observed in Middle and Hannah lakes (Heneberry, 1997). By reducing the rate of turnover of organic nutrients in benthic environments, a reduction in the availability of food resources to all trophic levels may occur. As a result, lakes may not be able to support large fish populations or individual fish may experience stunted growth (Iles, 2003). Species of benthic invertebrates identified within Sudbury area lakes are listed in Table 5.11.

**Table 5.11 Benthic Invertebrates Identified in Sudbury Lakes (Grzela and Wright, 2003)**

| Order | Family | Genus | Common Name |
|---|---|---|---|
| Ephemeroptera | Ephemerellidae | *Drunella* | Body-Builder Mayfly |
| | Isonychidae | *Isonychidae* | Minnow Mayfly |
| | Heptageniidae | *Epeorus* | Two-Tailed Flathead Mayfly |
| | | *Stenonema* | Flat-Head Mayfly |
| Plecoptera | Perlidae | - | Common Stonefly |
| | Pteronarcyidae | *Pteronarcys* | Giant Stonefly |
| | Peltoperlidae | - | Roach-like Stonefly |
| Trichoptera | Philopotamidae | *Chimarra* | Fingernet Caddisfly |
| | | | Mid-Size Plant Casebuilder |
| | Rhyacophilidae | *Rhyacophila* | Michelin-Man Caddisfly |
| | Brachycentridae | *Brachycentrus* | Mid-Size Plant Casebuilder |
| | Glossosomatidae | *Glossosoma* | Saddle Casemaker |
| | Limnephilidae | *Apatania* | Cornucopia Casebuilder |
| | Hydropsychidae | - | Common Net-Spinner |
| Coleoptera | Elmidae | *Stenelmis* | Riffle Beetle |
| | Psephenidae | *Psephenus* | Water Penny Beetle |
| | Dytiscidae | - | Water Beetle |
| Diptera | Tipulidae | *Antocha* | Crane Fly |
| | Leptidae | - | Snipe Fly |
| | Chironomidae | - | Midge (bloodworm) |
| | Simulidae | - | Black Fly |
| | Culicidae | - | Mosquito |
| Decapoda | Cambaridae | *Orconectes* | Crayfish |
| | - | - | Freshwater Shrimp |
| Megaloptera | Corydalidae | *Nigronia* | Fishfly |
| | | *Corydalus* | Dobsonfly |
| | - | - | Alderfly |
| Odonata | - | - | Dragonfly |
| | - | - | Damselfly |
| Unionoida | - | - | Freshwater Clam |
| Isopoda | - | *Caecidotea* | Isopod |
| Hirudinea | - | - | Leech |
| Oligochaeta | - | - | Aquatic Worm |
| Amphipoda | - | - | - |

Benthic invertebrate communities have been used extensively as indicators of habitat quality in both lotic and lentic ecosystems. This has led to standard methodologies for the collection, identification and evaluation of benthic samples that permit meaningful comparisons of invertebrate communities between sites. In addition, a number of sensitive biological metrics (*e.g.*, diversity indices, proportion of pollution tolerant species, *etc.*) have been developed that allow for a quantitative assessment of the benthic community. Data obtained from a well-designed field survey also can be statistically compared to various parameters such as sediment-metal levels.

Potential risks to benthic invertebrates will be assessed on a community level. Data for two species (*Hyalella azteca* and *Chironomus* sp.) were plentiful, and could be used as part of the assessment for this VEC. A short description of these organisms is provided below.

<u>*Hyalella azteca*</u>

*Hyalella azteca* is the most widely distributed amphipod in North America (Bousfield, 1958) and is found in lakes throughout the Sudbury area. They are a member of the family Crustacea and can reach sizes of 0.5 cm. They are considered to be scavengers, feeding on both plant and animal materials deposited on sediments, serving an important role in nutrient cycling. They are also an important food item for fish, predacious invertebrates and amphibians. This species has been included in a number of Sudbury-based studies and surveys. For example, the toxicity of Sudbury sediments to *Hyalella* has been attributed to elevated levels of nickel (Borgmann *et al*., 2001a). *Hyalella* are also one of the most commonly used benthic species for bioassays (Borgmann and Munawar, 1989), and have been used to predict the bioaccumulation of metals from sediments (Borgmann and Norwood, 1997; MacLean *et al*., 1996). Therefore, there are significant amounts of toxicity data available for this species.

<u>*Chironomus sp.*</u>

Chironomids (midges) are widely distributed throughout North American lakes and rivers. Larval midges are opportunistic feeders, feeding on diatoms, detritus, and other small plant and animal species; adults do not feed. They serve as an important food source to yellow perch in metal-contaminated lakes (Sherwood *et al*., 2000). This factor, coupled with their ability to accumulate metals from sediment (Warren *et al*., 1998), illustrates the potential influence that these organisms may have on local perch populations. A Sudbury-based study demonstrated that concentrations of copper and cadmium in perch livers were largely a result of accumulation through dietary uptake (Couture and Kumar, 2003). Chironomids also are a common species used in toxicity studies, and have been used in numerous bioassays describing the toxicity of many of the current COC.

**Zooplankton**

Selection of zooplankton species as VECs began with an initial evaluation of the distribution of species throughout the Sudbury area. The Co-Operative Freshwater Ecology Unit of Laurentian University surveyed 32 Sudbury lakes in 1990 and again in 2003 to collect information on the number of zooplankton species present. A total of 28 species were identified (Table 5.12).

**Table 5.12 Zooplankton Species and the Percentage of Sudbury Area Lakes in which they are found (2003 Survey; Co-Op, 2004)**

| Species | Presence in Sudbury Lakes (%) |
|---|---|
| *Acanthocyclops vernalis* | 0 |
| *Alona sp.* | 9 |
| *Bosmina sp.* | 97 |
| *Ceriodaphnia sp.* | 25 |
| *Chydorus sphaericus* | 34 |
| *Cyclops scutifer* | 12 |
| *Daphnia ambigua* | 9 |
| *Daphnia pulex* | 6 |
| *Daphnia retrocurva* | 25 |
| *Daphnia galeata mendotae* | 44 |
| *Daphnia sp.* | 0 |
| *Diacyclops bicuspidatus thomasi* | 44 |
| *Diaphanosoma birgei* | 94 |
| *Epischura lacustris* | 12 |
| *Eubosmina longispina* | 3 |
| *Eucyclops agilis* | 6 |
| *Eurycercus lamellatus* | 3 |
| *Holopedium glacialis* | 47 |
| *Leptodiatomus minutus* | 75 |
| *Leptodora kindtii* | 3 |
| *Macrocyclops albidus* | 3 |
| *Mesocyclops edax* | 50 |
| *Orthocyclops modestus* | 19 |
| *Polyphemus pediculus* | 0 |
| *Sida crystallina* | 9 |
| *Skistodiaptomus oregonensis* | 50 |
| *Tropocyclops extensus* | 62 |
| *Calanoid copepodid* | 94 |
| *Calanoid nauplius* | 97 |
| *Cyclopoid copepodid* | 94 |
| *Cyclopoid nauplius* | 91 |

The composition of zooplankton communities in lakes within the Sudbury area remains different than what would be expected in natural lakes despite significant improvements in water and sediment quality (Keller *et al*., 2004). Elevated levels of metals within the water column are considered to be the likely cause of the slow recovery of populations of cladoceran zooplankton in Sudbury lakes such as Middle Lake (Yan *et al*., 2004). However, this decline in recovery can also be caused by the unusual fish predation from the large yellow perch population (Yan *et al*., 2004). Copepod populations have shown a more substantial recovery and are now more typical of lakes of this region; this group is considered to be more tolerant of nickel contamination than cladocerans (Yan *et al*., 2004). Other zooplankton that have been observed to recolonize Sudbury lakes include the crustacean *Holopedium* (Keller and Yan, 1991) and *Daphnia mendotae* (Yan *et al*. 1996a,b).

It is recommended that the composition of the entire zooplankton community be evaluated in the aquatic ERA. The number of species and their population abundance within a lake's zooplankton community can often be a good indicator of environmental contamination. The absence of acid or metal sensitive species, or the dominance by a single or small number of tolerant species, may be indicative of inhospitable current or recent sediment and water quality. However, the absence of certain species may not necessarily be the result of poor water quality parameters, but rather the presence of a strong population of predators. Great care must be taken in the analysis of community composition as a measurable parameter representing water quality. Consideration of the presence of both predator and prey populations must be evaluated, as well as the strength of dispersal mechanisms of zooplankton species and the geographical limitations of colonization.

In addition, *Daphnia galeata mendotae* is considered to be a good indicator of the recovery of zooplankton from acidification (Keller *et al*., 1990) and therefore, particular attention should be paid to this species in the assessment. They are considered to be sensitive to metals, in particular, copper. Levels as low as 5 to 10 µg/L have been found to reduce longevity (Ingersoll and Winner, 1982; Koivisto *et al*., 1992), and a 16% reduction in reproductive success has been observed at 22 µg/L (Biesinger and Christensen, 1972). *Daphnia galeata mendotae* is a relatively large species of zooplankton that is abundant in lakes throughout central and eastern North America (Brooks, 1957; Carter *et al*., 1980; Keller and Conlon, 1995). Due to its large size, wide distribution and abundance, this species represents a larger proportion of the total biomass of crustacean zooplankton on the southern Canadian Shield than any other related species (Yan *et al*., 1988). Populations in many Sudbury lakes were dramatically affected as a result of its sensitivity to low pH, with ionoregulatory failure occurring in lakes with pH below 6 (Havens, 1992). It is a very well studied species in both Sudbury lakes and lakes that can be used as a reference for typical population dynamics (Yan *et al*., 1996c).

### 5.9.4 Plants

Two separate groups of aquatic plants were recommended as VECs: macrophytes and algae. Macrophytes are visible to the naked eye, whereas algae are small, often unicellular species. As with invertebrates, potential risks to aquatic plants will be assessed on a community level.

**Macrophytes**

Aquatic macrophytes can include emersed species (*i.e*., those that are rooted within the sediment and protrude up past the water's surface), submersed species (*i.e*., those that are rooted within the sediment but do not protrude past the water's surface), and floating species (*i.e*., those that are not rooted within the sediment and float within the water column or at the surface). Each of these types of macrophytes serves numerous roles in the dynamics of the aquatic ecosystem. They can provide habitat for spawning, nesting and feeding; they are a food source for herbivorous fish as well as for invertebrates and aquatic mammals (*e.g*., muskrats); some species can purify waters, effectively removing contaminants and storing them within their tissues; and they play an important role in nutrient cycling and the oxygenation of water. The characteristics of a lake macrophyte community can be a good indicator of lake quality in regard to levels of contamination and nutrient loading (*i.e*., eutrophic, mesotrophic, or oligotrophic status). A list of macrophytes observed in Sudbury area lakes (Sudbury, 2006) is provided in Table 5.13.

**Table 5.13 Aquatic Macrophytes Found in Lakes Throughout the Sudbury Area**

| Scientific Name | Common Name | Number of Lakes |
|---|---|---|
| *Acorus calamus* | Sweet Flag Calamus | 1 |
| *Alisma subcordatum* | Water Plantain | 1 |
| *Armoracia aquatica* | Lake Cress | 1 |
| *Aster lanceolatus* | Lance Leaved Aster | 1 |
| *Calamagrostis canadensis* | Canada Blue Joint | 1 |
| *Calamagrostis stricta* | Northern Reed Grass | 1 |
| *Callitriche hermaphroditica* | Submergent Water Starwort | 1 |
| *Carex lacustris* | Lakebank Sedge | 4 |
| *Carex lasiocarpa* | Wire Sedge | 2 |
| *Carex livida* | Livid Sedge | 1 |
| *Carex rostrata* | Beaked Sedge | 7 |
| *Carex scoparia* | Pointed Broom Sedge | 1 |
| *Ceratophyllum demersum* | Coontail | 1 |
| *Chamadaphne calyculata* | Leatherleaf | 5 |
| *Dulichium arundinaceum* | Three-way Sedge | 5 |
| *Eleocharis acicularis* | Needle Spike Rush | 9 |
| *Elodea canadensis* | Common Waterweed | 1 |
| *Eriocaulon septangulare* | Pipewort | 11 |
| *Fontinalis hypnoides* | Common Water Moss | 1 |
| *Genus Potamogeton* | Whitestem Pondweed | 1 |
| *Glyceria striata* | Fowl Mana Grass | 1 |
| *Halurus flosculosus* | Mermaids Hair | 3 |
| *Heteranthera dubia* | Water Stargrass | 1 |
| *Iris versicolor* | Northern Blue Flag | 2 |
| *Isoetes* spp. | Quillwort | 2 |
| *Juncus canadensis* | Canada Rush | 1 |
| *Juncus effusus* | Soft Rush | 2 |
| *Juncus militaris* | Bayonet Rush, Jointed Bog Rush | 1 |

**Table 5.13 Aquatic Macrophytes Found in Lakes Throughout the Sudbury Area**

| Scientific Name | Common Name | Number of Lakes |
|---|---|---|
| *Juncus pelocarpus* | Brown Fruited Rush | 4 |
| *Leersia hexandra* | Rice Cut Grass | 3 |
| *Lemna minor* | Duckweed | 4 |
| *Lobelia dortmanna* | Water Lobelia | 1 |
| *Lycopus spp* | Bugleweed | 2 |
| *Lythrum salicaria* | Purple Loosestrife | 2 |
| *Megalodonta beckii* | Water Marigold | 2 |
| *Muhlenbergia glomerata* | Marsh Timothy | 2 |
| *Myrica gale* | Sweet Gale | 2 |
| *Myriophyllum alternifolium* | Alternate Leaved Milfoil | 1 |
| *Myriophyllum sibiricum* | Northern Milfoil | 3 |
| *Myriophyllum spicatum* | Eurasian Milfoil | 5 |
| *Myriophyllum tenellum* | Slender Water Milfoil | 2 |
| *Nuphar variegate* | Yellow Pond Lily | 9 |
| *Nymphaeaceae* | Lily Pads | 3 |
| *Nymphaeaceae* | White Water Lily | 11 |
| *Phragmites australis* | Common Reed | 5 |
| *Plantago maritime* | Shore Plantain | 5 |
| *Polygonum amphibium* | Water Smartweed | 4 |
| *Pontederia chordata* | Pickerel Weed | 10 |
| *Potamogeton amplifolius* | Large Leaved Pondweed | 3 |
| *Potamogeton natans* | Floating-Leaved Pondweed | 1 |
| *Potamogeton pectinatus* | Sago Pondweed | 1 |
| *Potamogeton pusillus* | Slender Pondweed | 5 |
| *Potamogeton richardsonii* | Richardson's Pondweed | 9 |
| *Potamogeton zosteriformis* | Flat-Stemmed Pondweed | 2 |
| *Sagitteria* spp. | Arrowhead | 11 |
| *Scirpus acutus* | Hardstem Bulrush | 10 |
| *Scirpus americanus* | Common Three Square Sedge | 2 |
| *Scirpus cyperinus* | Woolgrass | 1 |
| *Sparganium emersum* | Broad Leaved Burreed | 1 |
| *Sparganium eurycarpum* | Large-fruited Burreed | 4 |
| *Sparganium fluctuans* | Floating-leaved Burreed | 11 |
| *Typha angustifolia* | Narrow Leaf Cattail | 2 |
| *Typha latifolia* | Common Cattail | 16 |
| *Urticularia intermedia* | Flat-Leaved Bladderwort | 1 |
| *Utricularia resupinata* | Lavender Bladderwort | 1 |
| *Vallisneria americana* | Wild Celery (Tape Grass) | 5 |

## Algae

Although many species of algae exist as single-celled life forms, they can produce large filamentous growths and dense layers over surfaces in the aquatic environment. There are also numerous species that are multi-cellular and are easily visible to the naked eye. Like macrophytes, they are photosynthetic, providing oxygen for other organisms, as well as forming the basis for the production of organic matter. They can also become a large consumer of oxygen when algal growths become abundant in summer months and mass numbers of dying algae begin to decompose. Algae serve as an important food source to invertebrates and fish, particularly during early life stages. Since many species of algae show a high tolerance for some metals, they can often accumulate significant levels before adverse effects are observed (Hutchinson and Stokes, 1973). As a result, the consumption of large amounts of algae by more metal-sensitive organisms, such as fish and invertebrates, can potentially result in toxic effects in higher trophic level species.

Studies have examined the effects of low pH and metals on populations of algae in Sudbury area lakes (Stokes *et al*., 1973; Hutchinson and Stokes, 1973). In particular, the toxicity of copper and nickel, and the influence of pH, has been well documented for *Chlorella vulgaris* and *Scenedesmus acutiformis. C. vulgaris* is used commonly in laboratory bioassays and there is a significant amount of toxicity and bioaccumulation information available for this species. Similar to other VECs, community composition was selected as a measurable parameter for the assessment of algae. The presence or abundance of particular species may be affected by various stressors. For example, the presence of a community dominated by *Zygogonium* may be indicative of a lake that is suffering from acidification (Keller and Gunn, 1994). Through comparisons of the planktonic and benthic algal community composition in Sudbury lakes and non-effected reference lakes, the overall status of the community can be assessed.

### 5.9.5 Amphibians

There are 15 amphibian species that exist in the Sudbury region (ROM, no date):

- American Toad (*Bufo americanus*) – These toads are usually found in deciduous or coniferous woodlands, but also may occur in clearings, fields, and urban areas. In the spring, eggs are laid in large or small bodies of water. Tadpoles congregate in shallow water and transform after about two months.
- Blue-spotted Salamander (*Ambystoma laterale*) - Adults live year-round in leaf litter on the forest floor, and enter woodland ponds only to breed.
- Bullfrog (*Lithobates catesbeiana*[1]) - Bullfrogs require permanent bodies of water because they have a long tadpole stage (tadpoles overwinter). They seldom move far from water.

[1] *Lithobates* was formerly *Rana*

- Eastern Newt (*Notophthalmus viridescens*) - Adults are almost totally aquatic, although they can become terrestrial if their habitat dries up during the summer. The juvenile stage prefers moist woodland habitat.
- Eastern Red-backed Salamander (*Plethodon cinereus*) - Redback salamanders inhabit moist areas on the forest floor. They do not enter water at any stage of their reproductive cycle, but instead lay their eggs on land, usually under or in rotting logs, and the entire larval stage takes place inside the egg.
- Four-toed salamander (*Hemidactylium scutatum*) - Adults prefer moist habitat and are usually found in sphagnum bogs. Eggs are laid in the moss above open pools of water. Larvae drop into the water once they hatch.
- Gray Treefrog (*Hyla versicolor*) - Gray treefrogs congregate in woodland ponds to breed but otherwise spend most of their time in trees or shrubs.
- Green Frog (*Lithobates clamitans*) - This species requires permanent water bodies because the tadpoles spend at least one winter in the water before transforming. Adults are seldom found far from water.
- Northern Leopard Frog (*Lithobates pipiens*) - This species breeds in early spring in ponds or larger bodies of water. Adults also overwinter at the bottom of ponds. However, the summer habitat for this frog is open fields or meadows.
- Mink Frog (*Lithobates septentrionalis*) - Mink frogs breed in early summer in permanent bodies of water. They prefer ponds with lily pads. This species remains at the tadpole stage for one winter before transforming.
- Mudpuppy (*Necturus maculosus*) - Mudpuppies live on the bottom of larger bodies of water. They have a long larval period, reaching maturity at five years of age.
- Spring Peeper (*Pseudacris crucifer*) - Spring peepers congregate at shallow, often temporary, ponds in woodlands. They also may breed in more open ponds and the shallow margins of larger bodies of water.
- Northern Two-lined Salamander (*Eurycea bislineata*) - Adults usually inhabit woodland streams, but they often forage in the leaf litter of the forest floor during the summer. Eggs are laid in streambeds. Larvae may metamorphose to adults after a few months, or overwinter and transform the next summer.

- Wood Frog (*Lithobates sylvatica*) – This species may be the most widely distributed amphibian species in Ontario. It breeds in early spring. The larval period is short, which allows this species to breed in temporary ponds. After the breeding season, wood frogs spend the summer on the forest floor.
- Spotted Salamander (*Ambystoma maculatum*) – This species breeds in early spring in woodland ponds. The larval stage is short and the juveniles lose their gills and leave the pond in midsummer. With the exception of the breeding season, these salamanders live buried in the soil or leaf litter in woodlands.

Amphibians may be key biological indicators of the status of wetlands and may be critical for evaluating wetland communities presumably affected by chemical stressors (Adamus and Brandt, 1990; Hebert *et al*., 1993). Amphibians may be adversely affected by metals either through habitat alteration or through direct exposure (Beiswenger, 1988).

The use of amphibians as indicators of environmental contamination is complicated because amphibian life cycles typically include both an aquatic and a terrestrial phase. Their semi-permeable skin makes them vulnerable to direct contact with many types of contaminated media. Their diets are generally herbivorous as tadpoles and insectivorous as adults (Lee and Stuebing, 1990), making them susceptible to chemicals that may accumulate in both plants and animals. Amphibians are particularly susceptible to environmental contamination during the sensitive embryonic and larval stages and during metamorphosis (Bonin *et al*., 1997).

Predicting the toxicity of impacted surface waters to amphibians during reproduction can often be a difficult task as a result of complex interactions between metals, pH and organic materials, as well as species-specific responses (Freda and McDonald, 1993). For example, due to the solubility of aluminium, its concentration in water increases with decreasing pH. Its toxicity to the leopard frog (*Rana pipiens*) increases with increasing pH, but its toxicity to the American toad (*Bufo americanus*) decreases with increasing pH (Freda, 1991). For other species, such as the spotted salamander (*Ambystoma maculatum*) and the leopard frog, aluminium can reduce the lethality of low pH (Clark and LaZerte, 1987; Freda and McDonald, 1990). Adding further to the complexity of these relationships, the toxicity of aluminium can be reduced as a result of interactions with dissolved organic matter through a reduction in bioavailability (Freda *et al*., 1990).

It has been estimated that one half of all frog species and one third of salamander species reproduce in small pools of water that are created from rainfall and snowmelt in early spring (Pough and Wilson, 1977). As a result, the chemical characteristics of these ephemeral pools are heavily influenced by the properties of precipitation. Acidic rainfall can cause the pH of these pools to be much lower than the pH of surrounding lakes and ponds, often creating conditions that are lethal to amphibian embryos and tadpoles (Pough, 1976; Beebee and Griffin, 1977; Pough and Wilson, 1977; Dunson and Connell, 1982; Pierce *et al*., 1984; Dale *et al*., 1985; Freda and Dunson, 1986; Gascon and Planas, 1986; Freda *et al*., 1991). While some amphibians may be able to tolerate acidic conditions, species such as the leopard frog (*Rana pipiens*) and the spotted salamander (*Ambystoma maculatum*) are known to be particularly sensitive to low pH.

Several metals, such as copper, zinc, iron, mercury, arsenic, cobalt, lead and chromium, have also been found to be toxic to amphibians (Lande and Guttman, 1973; Porter and Hakanson, 1976; Khangarot *et al*., 1985). Based on studies on fish, it can also be suggested that nickel, cadmium, and manganese would be expected to have adverse effects on amphibian populations as well (Glooschenko *et al*., 1992). Metals in lakes and breeding pools can reach elevated levels through atmospheric deposition, runoff, as well as through increased mobility from soils resulting from acidification (Kelso *et al*., 1982).

Although amphibians may be exposed to chemicals through contact with soil, sediment and surface water, and through consumption of food items, toxicity values are generally only available for direct contact of with surface waters during the egg and larval stages. Therefore, only risks associated with exposure to COC in surface water can be evaluated in the current evaluation. While risks to amphibians will be assessed at the community level, species that may be of particular interest include:

- Spotted Salamander (*Ambystoma maculatum*) since it is generally expected to occur in the Sudbury area but its occurrence appears to be limited;
- Green Frog *(Lithobates clamitans)* since a negative relationship was observed between distribution and aluminium concentrations;
- American Toad (*Bufo americanus*) since its absence in areas of Sudbury has been associated with elevated levels of nickel (Glooschenko *et al*., 1992)[2] and,

[2] Dr. Lesbarreres (2006; Biology Department, Laurentian University) reports that American toads and Leopard frogs are common in at least two areas he explored so far in Sudbury in 2005: the Ponderosa wetland (inbetween Junction Creek and Notre Dame Avenue) and the Lake Laurentian Conservation area behind the university.

- Northern Leopard Frog *(Lithobates pipiens)* since its absence in areas of Sudbury has been associated in the past with elevated levels of zinc (Glooschenko *et al*., 1992)[2].

### 5.9.6 Summary of Recommended VECs

The recommended final list of VECs for the aquatic ERA includes:

- Fish populations (common shiner, white sucker, lake trout, yellow perch and walleye);
- The benthic invertebrate community;
- The zooplankton community;
- The macrophyte community;
- The algal community; and
- The amphibian community.

### 5.9.7 Assessment Endpoints

An assessment endpoint is defined as a characteristic of a species, population or community that is to be evaluated and protected through the use of ERA. Assessment endpoints are identified for each VEC. The recommended VECs and assessment endpoints for the aquatic problem formulation are:

- Fish Populations (white sucker, lake trout, common shiner, yellow perch, walleye)
  - Presence, relative abundance, growth, development, and reproductive success
  - Habitat suitability
- Benthic Invertebrate Community
  - Community composition
- Zooplankton (Pelagic Invertebrate) Community
  - Community composition
- Algal Community
  - Community composition
- Macrophyte Community
  - Community composition
  - Habitat suitability
- Amphibian Community
  - Community composition

- o Presence, relative abundance, growth, development and reproductive success of particular species
- o Habitat suitability for particular species

The assessment endpoints can be evaluated using various types of data and approaches, including:

1) Comparisons of Sudbury-specific metal concentrations in water, sediment and tissue with general toxicity data from the published literature;
2) Site-specific toxicity test data; and,
3) Field population and community data.

For the aquatic problem formulation, all toxicity test and field data are from the published literature; no new studies were conducted as part of this study. Some fish tissues were collected for metal body burden analysis (SARA, 2006). These data are summarized and integrated in Section 5.14 of this report.

## 5.10 Conceptual Model

A conceptual model is a written description and a visual representation of the relationships between VECs and the COC to which they may be exposed. Conceptual models can serve three purposes: 1) clarification of assumptions concerning the situation being assessed; 2) communication tool for conveying those assumptions; and 3) providing a basis for organization and completion of the risk assessment (Suter, 1999). Conceptual models are powerful learning and communication tools when initiating an ERA, and they are easily modified as the ERA progresses and data gaps are filled. A preliminary conceptual model diagram showing direct and indirect toxicity linkages for aquatic organisms is provided in Figure 5-5. Many of the details associated with birds and mammals with a link to the aquatic environment are provided in Chapter 4 and associated appendices, as well as section 5.15 Risks to humans from consuming fish are addressed in Volume II (Human Health Risk Assessment).

Direct effects include adverse effects on survival, growth or reproduction of a species. Indirect effects include adverse effects on a species due to effects on other ecosystem components, such as a reduction in food abundance, a reduction in competition from another species (Preston, 2002), or a change in or loss of habitat. Indirect effects may have a positive or negative influence on a VEC (Chapman et al., 2003).

Aquatic organisms may be exposed directly to metals via ingestion, dermal contact and uptake from water at the gill surface. Metals may be ingested by consumption of food and water, and by incidental ingestion of sediment. Dermal exposure occurs when chemicals are absorbed through the skin from water or sediment. Food and water ingestion rates are generally not available for aquatic receptors, therefore, exposure to COC are primarily assessed based on metal concentrations in surface water and sediment. Risks to aquatic plants that may be rooted within sediments or floating within the water column also are assessed based on concentrations in surface water or sediment.

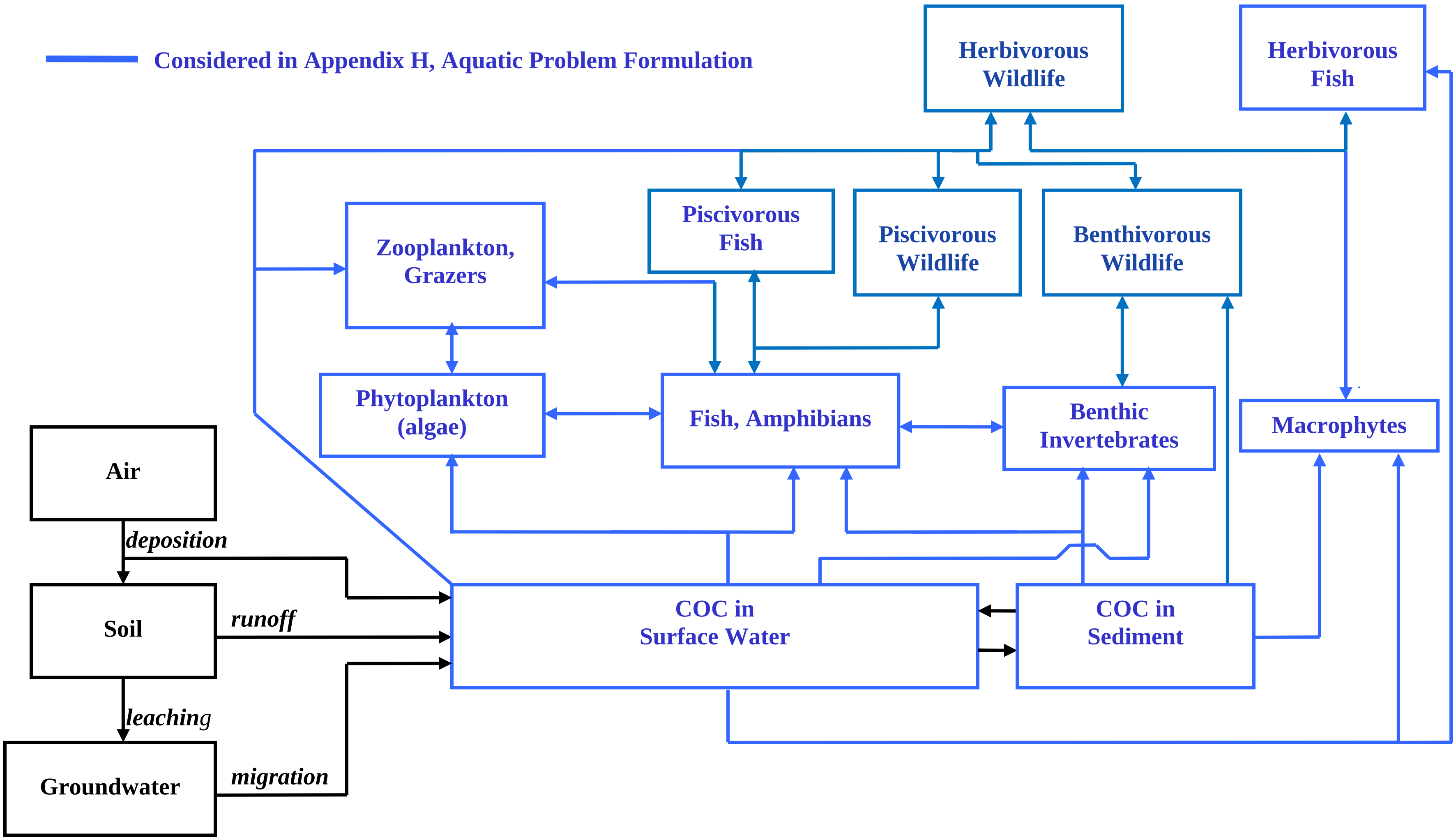



**Figure 5-5** **Preliminary Conceptual Model Illustrating Linkages between COC and Aquatic Organisms and the Terrestrial ERA (Chapter 4). Possible Indirect Effects are identified as Double-pointed Arrows**

The significance of indirect effects of metals on aquatic ecosystems has been recognized only recently (*e.g*., Preston, 2002; Fleeger *et al*., 2003; Sherwood *et al*., 2000; 2002; Campbell *et al*., 2003). However, failure to incorporate consideration of indirect effects into risk assessments has been criticized because it may be a significant source of uncertainty (Preston, 2002). Therefore, the following sections briefly describe what is known regarding some indirect effects on aquatic ecosystems, related to metals.

Risk assessments generally rely on data produced from single-species ecotoxicity tests. However, these tests cannot predict indirect effects on populations, communities or ecosystems (Fleeger *et al*., 2003). Therefore, many studies have utilized laboratory microcosms or mesocosms, or field surveys, because it is recognized that ecosystem studies are most relevant for assessing indirect effects (Preston, 2002). The following brief review focuses on these ecosystem studies, including several field studies which have been conducted around smelter sites, including the Horne copper smelter in Rouyn-Noranda, Quebec and the Sudbury smelters.

Indirect effects have been studied most intensively for pesticides (Fleeger *et al*., 2003), although some studies have been conducted on metals, and in particular copper. For example, caddisflies (*Hydropsyche morose*) were reported to be resistant to copper toxicity based on laboratory studies of direct toxicity (Clements *et al*., 1988). However, a microcosm experiment was subsequently conducted where it was observed that caddisfly predation by stoneflies (*Paragnetina media*) increased significantly after sublethal exposure to copper, presumably due to changes in predator avoidance behaviour of the caddisflies (Clements *et al*., 1989). Clements *et al*. (1988) also observed that copper reduced the size and diversity of the insect community in a laboratory stream, and that species dominance varied among copper treatments. Therefore, direct toxicity of copper to caddisflies was relatively unimportant, when compared to the indirect effects related to predator/prey interactions.

Indirect effects were suggested as the cause of effects observed on algae when they were exposed to elevated nutrient levels along with a mixture of metals. The algae were not affected by metals alone, possibly due to changes in the grazer community which feeds on algae (Breitburg *et al*., 1999). Keller *et al*. (2004) summarizes data from Sudbury lakes which contain unusual extensive growths of benthic filamentous algae, possibly as a result of an absence of grazers.

Several papers have been published that address indirect effects in lakes surrounding the Horne smelter in Rouyn-Noranda, Quebec. Yellow perch (*Perca flavescens*) were studied in four lakes along a gradient of metal contamination (Sherwood *et al*., 2000; 2002). The food web for the perch was very simple in metal-contaminated lakes compared to reference lakes. Perch normally start as zooplanktivores, then progress to benthivory and finally piscivory. In metal-contaminated lakes, perch had limited prey choices, and never reached piscivory. This was not necessarily due to the fact that prey fish were unavailable, but rather the perch were unable to reach sizes large enough to become piscivorous (Sherwood *et al*., 2002). In addition, the perch diet richness was low (as measured by stomach contents) in the most metal-polluted lakes, with zooplankton and dipterans (chironomids) dominating the diet, and other important prey groups such as ephemeropterans, odonates and fish being absent from the diet (Sherwood *et al*., 2000). Consumption rates were not different for fish in metal-contaminated lakes, and therefore the observed decrease in growth rates was considered to represent a significant increase in total energetic costs expended to meet dietary requirements (Sherwood *et al*., 2000).

Similar effects have been observed in lakes that have become mildly acidified (pH 5 to 6) (see discussion in Campbell *et al*., 2003). Yellow perch did not grow as large, and there were impacts on amphipods and burrowing mayflies that are important dietary components for perch. Other indirect effects, beyond those described above, are possible, but not well studied. For example, reduced survival of young perch could lead to increases in growth of those young fish that do survive. This was reported in yellow perch for moderately acidified lakes (Ryan and Harvey, 1980). Therefore, indirect effects caused by metals on yellow perch will be difficult or impossible to separate from similar indirect effects caused by acidification or metals combined with nutrients or other stressors.

The influence of a change in pH was studied at Little Rock Lake in Wisconsin, where the pH of one of two lake basins was reduced to between 4.7 and 5.6 over five years (see summary in Preston, 2002). Acidification caused a decline in populations of the cladoceran *Daphnia dubia* and the mayfly *Leptophlebia sp*. along with several species of invertebrate predators. Acidification also led to increases in benthic filamentous algae which led to indirect increases in benthic cladocerans (*Chydorus*) and caddisflies (*Oxyethira*) and provided increased refuge for juvenile yellow perch. Acidification also decreased the food availability for two species of rotifers (*Keratella cochlearis* and *K. taurocephala*). However, *K. taurocephala* increased in abundance, due to a reduction in invertebrate predators, whereas *K. cochelaris* decreased in abundance. Acidification also increased ultraviolet radiation (UVR) penetration into surface waters, which caused a stress on zooplankton community structure by reducing dissolved organic carbon. Other work conducted at Little Rock Lake (summarized in Preston, 2002) suggests that full recovery of the biota in the lake may occur once the pH stress is removed. The work conducted in Killarney Park, Ontario supports this (see summary in Campbell *et al*., 2003).

Metal-related indirect effects are difficult to distinguish from direct toxic effects (Fleeger *et al*., 2003). Campbell *et al*. (2003) attempted to determine the relative importance of indirect *vs.* direct effects of metals on yellow perch. Observed impacts on young perch were considered likely to be a result of direct toxicity. Young yellow perch (<2 years) feed on zooplankton, and the literature suggests the abundance of zooplankton in metal-contaminated lakes is adequate to meet the dietary needs of young perch. Perch change their diet as they grow, from zooplankton to benthos and finally fish. Reduced availability of larger prey (benthos) in metal-contaminated lakes resulted in increased bioenergetic costs and stunted growth of perch. This stunting of growth was not observed in younger fish. Therefore, the metals are likely having a significant indirect effect on the perch due to toxicity to (and therefore decreased availability of) larger benthic organisms on which they feed before becoming piscivorous (Campbell *et al*., 2003).

In summary, indirect effects of metals may have a greater influence than direct toxicity for species at a number of trophic levels and for specific life-stages. For perch, the indirect effects at the benthivorous stage may be significant, whereas direct metal toxicity may be more important to the young fish that feed on zooplankton (Campbell *et al*., 2003). Indirect effects of copper on predator avoidance behaviour in caddisflies were observed to be more important than direct toxicity (Clements *et al*., 1989). Algae were affected by changes in the grazer community (Breitburg *et al*., 1999; Keller *et al*., 2004). In addition, it may be difficult to distinguish between the direct and indirect effects of acidification, and those of metals (see summaries in Preston, 2002 and Campbell *et al*., 2003).

## 5.11 Selection of Lakes For Potential Further Study

The information presented in this section is intended to assist in the selection of lakes for further study in the Sudbury area. In selecting appropriate lakes to be considered in an aquatic ERA, several factors should be evaluated. Since the study is primarily interested in the effects of metal airborne emissions on aquatic ecosystems, a large number of lakes that receive other direct inputs should be removed from the initial list of potential selections. For example, those lakes that receive industrial or municipal effluents, including sewage and direct releases of mine tailings, should be eliminated. Also, the effects of metals on aquatic organisms may be influenced by the effects of acidification, and therefore the lake pH is important to consider (*e.g.*, lakes which have a pH below 6.0, as well as those lakes that have been artificially neutralized through the addition of $CaCO_3$ or $Ca(OH)_2$). Distance from the smelters may also be considered to provide a range of metals influences for study. It also should be noted whether or not the lakes are part of any monitoring programs (water and sediment chemistry) as well as biological or toxicity studies. It will be important to properly scope any future aquatic studies, including the identification of which lakes to study, to answer the question(s) being asked.

Section 5.11.1 summarizes the information available for many of the lakes in the Sudbury area. Section 5.11.2 identifies lakes that are not recommended for inclusion in a future ERA of smelter airborne emissions, due to the influence of other sources. Section 5.11.3 identifies the lakes and watersheds that have been neutralized, and Section 5.12 describes the marshes and wetlands of Sudbury.

### 5.11.1 Summary of Information Available for Lakes in the Sudbury Area

A review of the literature for studies conducted on lakes in the Sudbury area related to impacts from metals revealed that there are water chemistry data for over 75 lakes and sediment chemistry data for over 15 lakes (see Sections 5.5 and 5.6). However, biological data are available for only approximately 30 lakes, when considering only those publications which specifically name the lake(s) being studied, and which provide data for specific lakes, as opposed to describing trends for many lakes together. In fact, few lakes have been identified which included studies which considered more than one group of organisms (*e.g*., fish, algae, zooplankton, benthos), and of these, Kelly Lake, Long Lake and Whitson Lake have been impacted directly by mine effluent and therefore are not recommended for further consideration in this ERA. In addition, lakes north and east of Lake Wanapitei (greater than ~ 50 km northeast of Sudbury) and in Killarney Provincial Park (greater than 50 km southwest of Sudbury) were excluded because these lakes were primarily impacted by acid rain ($SO_2$ effects) and not metals from the smelters. This leaves several lakes including Clearwater, Hannah, Lohi, McFarlane, Middle, Nelson, Ramsey, Silver and Swan Lakes (Table 5.14) as lakes for which there are multiple studies of the biological communities. The studies conducted on these seven lakes, as well as information for other appropriate lakes in the Sudbury area, are summarized in section 5.14. Keller *et al*. (2004) summarized data related to recovery from metal and acid effects for crustacean zooplankton for 32 lakes (with pH $\geq$ 5.78), for walleye in 8 lakes, for fish in 17 lakes, and for periphyton in two acid, two limed and two reference lakes.

**Table 5.14 Location of Lakes where Multiple Biological Studies have been Conducted**

| Lake | Watershed | Distance from Copper Cliff Smelter |
|---|---|---|
| Clearwater Lake | Panache watershed | 12 km S |
| Hannah and Middle Lakes | Ramsey Lake watershed | 4 km S |
| Lohi Lake | Panache watershed | 11 km S |
| McFarlane Lake | Panache watershed | 10 km S |
| Nelson Lake | Nelson River watershed | 30 km N |
| Ramsey Lake | Ramsey watershed | Between Copper Cliff and Coniston |
| Silver Lake | Panache watershed | 6 km S |
| Swan Lake | Panache watershed | 12 km S |

Clearwater Lake is an example of a lake that was highly acidic in the 1970s (pH~4) but has now reached a pH greater than six without active neutralization (Keller *et al*., 2004). Patterns of copper and nickel concentrations in water analyzed from 1981 to 2003 also show a trend toward decreasing concentrations in this lake, although fluctuations may result from natural (*e.g*., drought) and anthropogenic (*e.g*., smelter shut-down) causes (Keller *et al*., 2004). Clearwater Lake was studied as part of the Sudbury Urban Lakes Study (1990; 2003) and continues to be studied as part of the SES Intensive and Extensive Monitoring Programs (Co-op, unpublished b; Keller *et al*., 2004).

Hannah Lake drains into Middle Lake, and is located approximately 4 to 5 km south of the smelters in Sudbury. Studies have been conducted on fish (Uutala and Smol, 1996; Eastwood and Couture, 2002; Couture and Kumar, 2003; Couture and Rajotte, 2003; Keller *et al*., 2004), primarily in Hannah Lake, and zooplankton (Yan *et al*., 1996a, b; Keller *et al*., 2004) in both lakes.

Lohi Lake is located approximately 11 km south of Copper Cliff in the Panache watershed. Lohi Lake was neutralized in 1973 and 1975. Macrophytes were surveyed in 1977 and 1978 in Lohi Lake (Nriagu, 1984). Ash free dry mass of periphyton collected from rocks was studied in 1996 (Keller *et al*., 2004).

Farlane Lake is situated approximately 10 km south of the Copper Cliff smelter, is upstream of Long Lake, and is part of the Panache watershed. The pH of McFarlane Lake is near neutral (Keller *et al*., 2004). McFarlane Lake is a popular fishing lake that has been stocked by community fisheries involvement program (CFIP) groups (Keller *et al*., 2004).

Nelson Lake is located approximately 30 km north of the Copper Cliff smelter. Nelson Lake was neutralized in 1975/76 and went from a pH of 5.8 (1973) to a pH of 6.4 (1979) (Keller *et al*., 1992b). Studies have been conducted on fish and zooplankton in Nelson Lake.

Ramsey Lake is located in the heart of Sudbury, and has a pH of approximately 7.4 (Keller *et al.*, 2004). Data are available for fish populations, zooplankton and crayfish, and this lake was part of the 1990 and 2003 Sudbury Urban Lakes Study. Ramsey Lake receives effluent from septic systems and urban road runoff, which could be reason for excluding it from further study. However, it is included for consideration in an ERA for three main reasons:

- It has been included in numerous studies, therefore there are significant data available regarding its physical and chemical characteristics;
- It is home to numerous species of fish, including game fish; and,
- It is a significant resource and focal point to Sudburians, and is therefore considered to be socially significant.

Silver Lake is located approximately 6 km south of the Copper Cliff smelter, and is part of the Panache watershed. It is considered to be the most heavily contaminated lake when considering those lakes impacted by atmospheric deposition only (Gunn and Keller, 1995). The pH of Silver Lake is approximately 6.0 (Keller *et al.*, 2004). Silver Lake was among the most heavily affected lakes, with a pH of 4.1 and copper and nickel concentrations of 430 and 880 µg/L, respectively in 1981 (Keller *et al.*, 1998). Biological improvements have been noted since reductions in emissions took place, beginning in the 1970s (Keller *et al.*, 1999).

Swan Lake is located approximately 12 km south of the Copper Cliff smelter. It was studied as a natural recovery lake (Keller *et al.*, 1992a,b). Recovery of crustacean zooplankton (for the period 1973 to 1986) was determined by comparing species richness in Swan and six other lakes to that found in one consistently near-neutral (pH 6.9) Sudbury lake (Welcome) and 24 near-neutral (pH> 6.0) lakes in Dorset Ontario (Keller and Yan, 1991).

The Ontario Ministry of the Environment (MOE, 1992) studied whether there was a relationship between phytoplankton community structure and acidification in a large 111 lake set, as well as in a smaller group of seven lakes (including Swan Lake), within a 260 km radius of Sudbury.

Table 5.15 provides a summary of information that may be used in the process of selecting lakes to be included in an ERA. Table 5.16 and Figure 5-6 show the locations of watersheds in the Sudbury area, and which lakes are found within each watershed. The lack of published biological data should not prevent a lake from being considered part of some future study on aquatic ecological effects in the Sudbury area. These historical data simply allow an evaluation of changes over time.

**Table 5.15 Summary of Information for Sudbury Area Lakes**

| Lake | pH [a] | Watershed [b] | Water Chemistry | Sediment Chemistry | Biota Studies [c] | Comments |
|---|---|---|---|---|---|---|
| **SES Extensive Lakes [d]** | | | | | | |
| Annie | 6.29 | | Yes | | | |
| Bell | 6.25 | Lower Vermilion | Yes | | | |
| Blue Sucker | 5.46 | | Yes | | | |
| Bob | 5.25 | | Yes | | | |
| Chiniguchi | 5.48 | Kukagami | Yes | | | |
| David | 5.11 | | Yes | | | |
| Donald | 5.28 | | Yes | | | |
| Dougherty | 4.87 | | Yes | | | |
| Florence | 5.23 | | Yes | | | |
| Fraleck | 6.07 | Wanapitei | Yes | | | |
| Frederick | 5.09 | | Yes | | | |
| Fullrock | 6.43 | | Yes | | | |
| George | 6.11 | | Yes | | | |
| Jim Edwards | 5.15 | | Yes | | | |
| Johnnie | 5.89 | | Yes | Yes | | |
| Killarney | 5.12 | | Yes | | | |
| Klock | 5.73 | | Yes | | | |
| Landers | 4.99 | | Yes | | | |
| Laundrie | 5.46 | Kukagami | Yes | | Yes | Studied as "natural recovery lake" by Keller *et al*., 1992a,b although influenced by neutralization of Bowland Lake |
| Mahzenazing | 6.13 | | Yes | | | |
| Marjorie | 4.61 | Kukagami | Yes | | | |
| Matagamasi | 5.83 | Kukagami | Yes | | | |
| Nellie | 4.66 | | Yes | | | |
| O.S.A | 4.91 | | Yes | | | |
| Pilgrim | 5.6 | | Yes | | | |
| Reef | 6.14 | | Yes | | | |
| Ruth Roy | 4.83 | | Yes | | | |
| Sans Chambre | 6.51 | | Yes | | Yes | Studied as "natural recovery lake" by Keller *et al*., 1992a,b |
| Seagram | 5.48 | | Yes | | | |
| Silvester | 4.91 | Kukagami | Yes | | | |
| Sunny Water | 4.77 | | Yes | | | |
| Telfer | 5.15 | | Yes | | | |
| Tillie | 5.09 | | Yes | | | |
| Tyson | 6.15 | | Yes | Yes | | |
| Wabun | 5.1 | | Yes | | | |
| Wavy | 4.99 | | Yes | | Yes | Studied as "natural recovery lake" by Keller *et al*., 1992a,b |
| White Oak | 5.84 | | Yes | | | |
| White Pine | 5.18 | | Yes | | Yes | Studied as "natural recovery lake" by Keller *et al*., 1992a,b |
| Whitson | 6.77 | | Yes | Yes | Yes | Receives mine wastes |
| Wolf | 4.98 | Kukagami | Yes | | | |

**Table 5.15 Summary of Information for Sudbury Area Lakes**

| Lake | pH [a] | Watershed [b] | Water Chemistry | Sediment Chemistry | Biota Studies [c] | Comments |
|---|---|---|---|---|---|---|
| **Urban Lakes** | | | | | | |
| Bennet | 6.7 | Ramsey | Yes | | Yes | |
| Bethel | 9 | Ramsey | Yes | | Yes | |
| Broder #23 | 6.38 | East Wanapitei | Yes | | | |
| Brodill | 6.0 | East Wanapitei | Yes | | | |
| Clearwater [e] | 6.3 | Panache | Yes | Yes | Yes | Studied as "natural recovery lake" by Keller *et al.*, 1992a,b |
| Crooked [e] | 5.78 | Upper Junction Creek | Yes | | Yes | |
| Crowley | 6.3 | Panache | Yes | | Yes | |
| Daisy | 6.2 | Panache | Yes | Yes | Yes | Small catchment neutralized in 1995 |
| Dill (T) | 6.61 | East Wanapitei | Yes | | Yes | |
| Forest | 6.18 | Panache | Yes | | Yes | |
| Grant | 7.2 | | Yes | | Yes | |
| Hannah | 7.25 | Ramsey | Yes | Yes (limited) | Yes | Lake (1973) and catchment (1983/4) neutralized |
| Johnny | 6.6 | | Yes | | Yes | |
| Kelly (Kelley) | 6.95 | Upper Junction Creek | Yes | Yes | Yes | Receives city sewage, mining effluent |
| Laurentian | 6.3 to 6.5 | Ramsey | Yes | | Yes | |
| Linton | 6.2 | Panache | Yes | | Yes | |
| Little Raft | 7 | East Wanapitei | Yes | | Yes | |
| Lohi | 6.3 | Panache | Yes | | Yes | Neutralized in 1973 and 1975 |
| Long | 7.1 | Panache | Yes | Yes | Yes | 1909 to 1916 - Long Lake Gold Mine |
| McFarlane | 7.5 | Panache | Yes | Yes | Yes | |
| Middle | 6.9 | Ramsey | Yes | | Yes | Lake (1973) and catchment (1983/4) neutralized |
| Minnow | 8.8 | Ramsey | Yes | Yes (limited) | | Historically impacted by lumber mill effluent and oil spills |
| Nepahwin | 7.5 | Ramsey | Yes | Yes | | |
| Raft | 6.8 | East Wanapitei | Yes | | Yes | |
| Ramsey | 7.67 | Ramsey | Yes | Yes | Yes | City drinking water source, receives effluent from septic and saw mill |
| Richard | 7.25 | Panache | Yes | | | |
| Robinson | 7.7 | Ramsey | Yes | | | |
| Silver[e] | 6.0 | Panache | Yes | Yes | Yes | Catchment neutralized |
| St. Charles | 7.0 | Ramsey | Yes | | Yes | |
| Still | 7.1 | | Yes | | Yes | |
| Tilton[e] | 6.3 | Panache | Yes | | Yes | |
| **Other Lakes** | | | | | | |

**Table 5.15 Summary of Information for Sudbury Area Lakes**

| Lake | pH [a] | Watershed [b] | Water Chemistry | Sediment Chemistry | Biota Studies [c] | Comments |
|---|---|---|---|---|---|---|
| Baby | 4.05 | East Wanapitei | | | Yes | Severely affected by acid; studied as "natural recovery lake" by Keller *et al*., 1992a,b |
| Bibby | 6.1 | East Wanapitei | | | | |
| Boucher | 6.7 | | | | Yes | |
| Camp | 6.4 | Panache | | | | |
| Chief | 4.8 | East Wanapitei | | | | |
| Fairbank | | Fairbank | | Yes | | |
| Geneva | 6.7 | Onaping | | Yes | | |
| Halfway | 6.7 to 6.8 | | Yes | Yes (limited) | Yes | Not influenced by smelters; used as "reference " lake |
| Larder | 8.0 to 8.2 | | Yes | | Yes | Received waste from gold mine, tailings |
| Michiwakenda | 7.2 to 7.9 | | | | Yes | Not influenced by smelters; used as "reference " lake |
| Nelson | 6.25 | Nelson River | Yes | Yes (limited) | Yes | Lake neutralized in 1975 to 1976 |
| Perch | 6.5 | Ramsey | | | | |
| Pine | 4.6 | Wanapitei | | | | |
| Round | 8.4 | | | | Yes | No direct sources of metals to this lake |
| Swan | 4.8 | Panache | Yes | | Yes | Studied as "natural recovery lake" by Keller *et al*., 1992a,b |
| Vermillion | 7.5 to 7.6 | Mid Vermilion | | | Yes | Receives mining effluents |
| Whitewater | | Whitewater | | Yes (limited) | Yes | Receives effluent from tailings dams |

[a] pH data from multiple sources (Keller *et al*., 2004; Bradley and Morris, 1986; Co-Op, unpublished, and the City of Greater Sudbury website) and therefore the pH of the lakes in 2005 may be different than those reported, depending on the age of the data.

[b] Watershed information from City of Greater Sudbury web site at http://www.city.greatersudbury.on.ca/lakewaterquality/maps/lakeindex.cfm

[c] Biota studies may include any or multiple studies on fish, invertebrate or algal populations, however, studies only of tissue metal concentrations were excluded

[d] These SES Extensive lakes are not recommended for inclusion in the aquatic ERA due to their location outside the study area

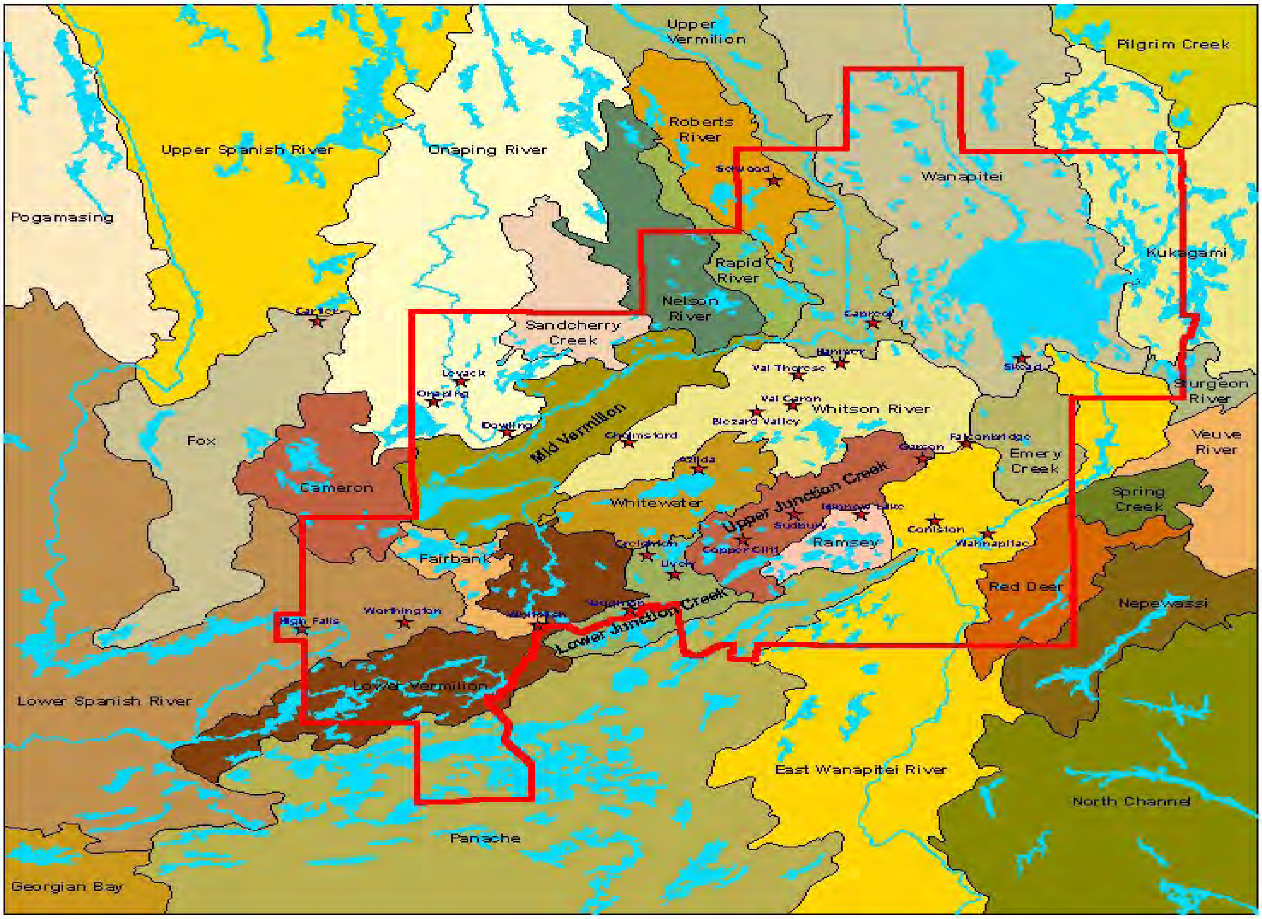
[e] Lake is part of both Urban Lake program and SES Extensive Monitoring program

**Table 5.16 Lakes within Sudbury Area Watersheds**

| Nelson River | Rapid River | Roberts River | Upper Vermilion |
|---|---|---|---|
| Foster | Joe | Ironside | Bass |
| Nelson | Pigeon | Kumska | Blueberry |
| Noland | Tank | | Cache |
| Rand | | **Onaping River** | Farm |
| Towermans | **Kukagami** | Clear | Fraser |
| | Bad | Diabase | Frenchman |
| **Wanapitei** | Bassfin | High Cliff | Gibson |
| Amy | Big Valley | Moose | Greens |
| Bannagan | Bonesteel | Pike | Hanmer |
| Barnett | Boot | Seal | Hutton |
| Bass | Cathro | Webfoot | Long |
| Beaver | Dewdney | Windy | Marshy |
| Bernard | Doon | | Onwatin |
| Blackthorn | Franks | **Sandcherry Creek** | Rockcut |
| Blue | Houston | Island | Ross |
| Bolands | Irish | Longvack | Wallin |
| Bonhomme | Jones | Morgan | Wisner |
| Bottom | Kukagami | West Morgan | |
| Bugg | Laundry | | **Ramsey** |
| Bushy | Matagamasi | **Mid Vermilion** | Bennet |
| Capre | McLaren | Gordon | Bethel |
| Caswel | Pelo | Simmons | Hannah |
| Connelly | Portage | Snider | Laurentian |
| Dean | Rat | Sweezey | Middle |
| Drill | Rathwell | Upper Gordon | Minnow |
| East Bass | Shed | Vermilion | Nepahwin |
| Ella | Silvester | | Perch |
| Fire | Thomas | **Whitson River** | Ramsey |
| Fraleck | Upper Thomas | Garson | Robinson |
| Framan | Wessel | McCrea | St. Charles |
| Goat | Wolf | Mosse | |
| Hagarty | | Whitson | **East Wanapitei** |
| Irving | **Emery Creek** | | Alice |
| Kosmerly | Falcon Gold | **Fairbank** | Baby |
| Lac St. Jean | | Bass | Bonanza |
| Lawlor | **Cameron** | Ethal | Brodill |
| Little Amy | Cameron | Fairbank | Chief |
| Little Italy | Ross | Little Fairbank | Little Raft |
| Little Otter | West Cameron | Mond | Raft |
| Lynn | | Skill | T (Dill) |
| Malbeuf | **Panache** | | |
| McFie | Bassoon | **Upper Junction Creek** | **Lower Spanish River** |
| Minnow | Bear | Clara Belle | Agnew |
| Moose | Brady | Crooked | Perch |
| Mowat | Camp | Kelly | St. Pothier |
| Otter | Clearwater | Lady MacDonald | |
| Overhead | Crowley | | **Lower Vermilion** |
| Parkin | Daisy | **Lower Junction Creek** | Ann |
| Peterson | Forest | Echo | Beavers |
| Pike | High | McCharles | Bell |
| Pine | Linton | Meatbird | Ella |

**Table 5.16 Lakes within Sudbury Area Watersheds**

| Nelson River | Rapid River | Roberts River | Upper Vermilion |
|---|---|---|---|
| Rathburn | Little Panache | Mud | Grassy |
| Sam Martin | Little Round | Simon | Happy's |
| Selwyn | Lohi | | Karstula |
| Skynner | Long | **Red Deer** | Little Ella |
| Spar | Makada | Jumbo | Little Rat |
| Stake | McFarlane | Red Deer | Louie |
| Tower | Norwest | Southeast Baby | Margaret |
| Upper Gipsy | Page | | Monk |
| Upper Mowat | Panache | **Sturgeon River** | Northweat |
| Waddell | Pig | Ashigami | Number Ten |
| Wanapitei | Pine | | Rat |
| Windy | Richard | **Whitewater** | Threecorner |
| | Silver | Emma | Wabagishik |
| | Tilton | Moore | West |
| | | Pump | |
| | | Turner | |
| | | Whitewater | |




Source: See interactive watershed map on the City of Greater Sudbury website, at
http://www.city.greatersudbury.on.ca/lakewaterquality/maps/lakeindex.cfm

**Figure 5-6 Locations of Watersheds in the City of Greater Sudbury and Surrounding Area**

## 5.11.2 Waterbodies Recommended for Exclusion from Further Consideration

There are several lakes and rivers that have been impacted by sources other than particulate emissions from the Sudbury smelters (Table 5.17). Seven of these (Junction Creek, Kelly Lake, Larder Lake, Long Lake, Vermilion Lake, Whitewater Lake and Whitson Lake) are described below, including the rationale for excluding them from further consideration in an ERA. In addition, lakes north and east of Lake Wanapitei (greater than ~ 50 km northeast of Sudbury) and in Killarney Provincial Park (~ 50 km southwest of Sudbury) were excluded because these lakes were primarily impacted by acid rain ($SO_2$ effects) and not metals from the smelters. These include many of the lakes from the SES Extensive Lakes survey (see Table 5.15 and COC screening data in Section 5.5.3). Despite the fact that these waterbodies may have been impacted by other sources or predominantly from acid rain and not metals, and are, therefore, not recommended for further assessment in this ERA, many studies have been conducted on these lakes and rivers, and therefore, some relevant information is provided below.

**Junction Creek**

Junction Creek flows southwest through the City of Greater Sudbury, and through a series of small lakes, beginning with Kelly Lake and ending at McCharles Lake, where it joins the Vermilion River system. Much of the Junction Creek watershed is within the zone of atmospheric deposition from the smelters. However, there are many other sources of contaminants along the creek. An active source of metals is the Frood Mine area and the Copper Cliff east drainage to Nolins Creek. In fact, Nolins Creek, Garson Creek and Copper Cliff Creek all have headwaters at mine sites and are tributaries of Junction Creek. Storm sewers also may contribute significant levels of metals to the system. Elevated PCBs were observed at one location in the creek, and possible sources were identified as the Creighton mine site, Vale Inco's tailings, and the Lively Golf Club. PAHs were observed at high levels in the creek downstream of the former Canada Creosote site; a layer of contaminated soils adjacent to the creek was observed in this area (Jaagumagi and Bedard, 2002).

An extensive study was conducted by the Sudbury District office of the MOE which included water and sediment quality analysis, benthic community composition, sediment toxicity bioassays and young-of-the-year fish tissue analysis (Jaagumagi and Bedard, 2002). In addition, since 1999, the Junction Creek Steward Committee has been actively working towards environmental restoration of this creek (Environment Canada, 2004). The committee is an independent body, comprised of individuals from local, regional and provincial government agencies such as the Ontario Ministry of Natural Resources and the Ontario Ministry of the Environment; universities and colleges; businesses; industries; community groups; the Vegetation Enhancement Technical Advisory Committee (VETAC); and other individuals. Vale Inco is a primary sponsor of the Junction Creek Stewardship Committee (Environment Canada, 2004). The committee activities include efforts to re-green adjacent lands, reduce soil erosion, re-engineer the creek bed and banks, improve water quality, and keep garbage out of the creek. In 2000 and 2001, 6,000 brook trout were reintroduced into the upper reaches of the creek. Research and monitoring will evaluate the success of this release (Environment Canada, 2004).

**Kelly (Kelley) Lake**

Kelly Lake receives treated effluent from Copper Cliff, as well as from the municipal sewage plant (Gunn and Keller, 1995). In fact, Kelly Lake has received effluent from the mining and smelting operations at

Inco Limited as well as residential effluent since the 1880s (Pearson *et al*., 2002b). There are several mine sites, waste rock dumps and wastewater ponds within its watershed. Air pollution has caused extensive devegetation in the area surrounding Kelly Lake, leading to severe erosion of the watershed and a rapid growth of the delta where Junction Creek meets Kelly Lake. The exposed area of the delta was estimated to grow at an average rate of 2,250 $m^2$ per year from 1875 to 1928. The natural growth of ground cover as a result of declines in smelter pollution since the 1970s and the revegetation program have helped to greatly reduce erosion and sedimentation (Pearson *et al*., 2002b).

Sediment cores from Kelly Lake indicate that a pine-dominated landscape was succeeded by one that was dominated by weeds. Copper and nickel concentrations in the lake dropped between 1968 and 1998 as a result of new vegetation that provided organic matter for metal ions to absorb to, as well as improved technologies for the treatment of wastewater and smelter emissions. The pH of the lake stabilized within a range of 6.5 to 8.0 in the mid 1980s. Previously, it was known to change from 4.5 to 10.0 within a period of a few weeks (Pearson *et al*., 2002b).

Kelly Lake has been the subject of numerous investigations for several years. For example, Hutchinson and Stokes (1973) studied the availability and toxicity of metals in Kelly Lake water through the use of bioassays with four different species of unicellular green algae. Species richness of zooplankton in the lake (in 1990 compared to 2003) is reported in Keller *et al*. (2004). Couture and Rajotte (2003) included Kelly Lake in their study of metabolic indicators of metal stress in yellow perch. Kelly Lake is the subject of ongoing study by various educational institutions such as Laurentian University (*e.g*., Dr. Graeme Spiers) and the University of Toronto (*e.g*., Dr. Miriam Diamond).

**Larder Lake and Whitson Lake**

Larder Lake and Whitson Lake are affected by direct input of liquid effluent from mine tailings and other mine related wastes (Chen *et al*., 2001; Eastwood and Couture, 2002), which precludes them from being considered further in an ERA. Larder Lake is located between 60 and 95 km from Sudbury (Eastwood and Couture, 2002). Whitson Lake is located near Val Caron (northeast of the Copper Cliff smelter, west of Falconbridge), approximately 10 km from the smelters (Keller *et al*., 2004). In a 2003 survey, six fish species were found in Whitson Lake (Keller *et al*., 2004) although the City of Greater Sudbury web site now lists seven species (Sudbury, 2004) based on a 2004 survey. Both lakes were included in studies by Chen *et al*. (2001) who measured concentrations of Se and Hg in the muscle tissues of perch (*Perca flavescens*) and walleye (*Stizosedion vitreum*) in lakes located from 4 to 204 km of the Sudbury smelters. Concentrations of mercury in the tissues of Sudbury fish were generally quite low, and this has been suggested to be the result of an antagonistic effect of environmental selenium on the assimilation of mercury (Chen *et al*., 2001). Both lakes also were studied by Eastwood and Couture (2002) who evaluated fish condition and liver concentrations of Cu, Zn and Ni in yellow perch from several lakes in the spring and fall of 1997. Additional studies are available on perch metal levels and condition for Whitson Lake (Rajotte and Couture, 2002; Couture and Kumar, 2003; Audet and Couture, 2003). Whitson Lake also is studied as part of the SES Extensive Monitoring Program.

**Long Lake**

Long Lake is situated approximately 10 km south of the Copper Cliff smelter, is joined to McFarlane Lake to the northeast, and is part of the Panache watershed. The pH of Long Lake is near neutral (Keller *et al*., 2004). Long Lake is a popular fishing lake that has been stocked by community fisheries involvement program (CFIP) groups (Keller *et al*., 2004). From 1909 until 1916, the Long Lake Gold Mine operated on the west side of Long Lake, adjacent to the Whitefish Lake First Nation. In 1912, this was the largest gold producing mine in Ontario (Sudbury, 2005). The historical presence of this gold mine is the reason Long Lake may be considered for exclusion from the aquatic ERA.

Long Lake is one of the lakes monitored in the Urban Lake Recovery Program. Twelve species of fish were found in McFarlane Lake during a 2004 survey (Keller *et al*., 2004). Studies are ongoing to assess the effects of altered fish prey communities on walleye growth in eight lakes including Long Lake (Keller *et al*., 2004). Studies will also include the effects of the loss of benthic invertebrates on food web functions (Keller *et al*., 2004). Species richness of zooplankton in Long Lake in both 1990 and 2003 was comparable to that in Dorset non-acidic reference lakes (Keller *et al*., 2004). The species assemblage, however, would have to be evaluated more closely to determine whether or not it is more typical of a recovering lake than a “natural” lake (Keller *et al*., 2004). Previously, phytoplankton growth rate was evaluated when exposed to water from five lakes (Alice, Baby, Boucher, Long and Kelly) collected in 1970 (Hutchinson and Stokes, 1973).

**Vermilion Lake**

Vermilion Lake is located northwest of downtown Sudbury, within 20 km of the smelters (approximately 30 km from Copper Cliff). Vermilion Lake is not recommended for further study because a tributary of the Vermilion River receives mining effluents from an ore-processing area, which has led to elevated levels of nickel (Bolger, 1980). A survey in 2003 found 13 species of fish in Vermilion Lake (Keller *et al*., 2004) although the City of Greater Sudbury web site now lists 14 species (Sudbury, 2004) based on a 2004 survey. Three studies have evaluated fish tissue metal levels and fish condition. Eastwood and

Couture (2002) studied fish condition and liver concentrations of Cu, Zn and Ni in yellow perch from Vermilion Lakes in the spring and fall of 1997. Rajotte and Couture (2002) studied Al, Cd, Cu, Ni, and Zn concentrations in tissues of yellow perch, as well as fish condition, swimming capacities, and tissue metabolic capacities. Bradley and Morris (1986) examined metal stress in fish from 10 Sudbury lakes through concentrations of metals (Ni, Cu and Zn) in muscle, liver and kidney tissues, and examined the relationship between concentrations in fish and concentrations in lake sediments. Three lakes were located within Sudbury, approximately 10 km from the smelters (Nepahwin, Minnow and Whitewater Lakes), six were within 30 to 50 km of Sudbury (Nelson, Ashigami, Vermilion Fairbank, Kukagami, Tyson), and one was located 180 km from Sudbury (Skeleton).

**Whitewater Lake**

Whitewater Lake is located approximately 10 km north of the Copper Cliff smelter. Tailing dams are situated in the Whitewater watershed. Some runoff flows into Pump and Clara Belle Lakes, which drain into Whitewater Lake. This effluent is high in copper, nickel, and iron, and has a pH of 7.8 (Sudbury, 2005).

Changes in aquatic macrophyte flora from 1947 to 1977 were described by Dale and Miller (1978). Bradley and Morris (1986) examined metal stress in fish from 10 Sudbury lakes through concentrations of metals (Ni, Cu and Zn) in muscle, liver and kidney tissues, and examined the relationship between concentrations in fish and concentrations in lake sediments. Three lakes were located within Sudbury, approximately 10 km from the smelters (Nepahwin, Minnow and Whitewater Lakes), six were within 30 to 50 km of Sudbury (Nelson, Ashigami, Vermilion Fairbank, Kukagami, Tyson), and one was located 180 km from Sudbury (Skeleton).

**Table 5.17 Rationale for Excluding Lakes and Rivers from an ERA**

| Waterbody | Reason for Exclusion from ERA | Example Relevant Studies Conducted on Waterbody |
|---|---|---|
| Junction Creek | Numerous active and historical sources including mines, tailings, a creosote site and golf course, and flows through Kelly Lake | Jaagumagi and Bedard (2002) |
| Kelly Lake | Receives mining effluent, sewage discharge, and is connected to Junction Creek | Hutchinson and Stokes (1973); Couture and Rajotte (2003); Pearson *et al*. (2002a); Keller *et al*. (2004) |
| Killarney Provincial Park Lakes | Primarily affected by acid rain and not metals from smelter air emissions | |
| Larder Lake | Direct input of liquid effluent from mine tailings and other mine-related wastes | Chen *et al*. (2001)<br>Eastwood and Couture (2002) |
| Long Lake | From 1909 to 1916, the Long Lake Gold Mine operated on the lake. | Hutchinson and Stokes (1973)<br>Keller *et al*. (2004) |

**Table 5.17 Rationale for Excluding Lakes and Rivers from an ERA**

| Waterbody | Reason for Exclusion from ERA | Example Relevant Studies Conducted on Waterbody |
|---|---|---|
| SES Extensive Monitoring Lakes | Primarily affected by acid rain and not metals from smelter air emissions; lower metals levels than Urban Lakes | |
| Vermilion Lake | A tributary of the Vermilion River receives mining effluents from an ore-processing area | Bradley and Morris (1986); Eastwood and Couture (2002); Rajotte and Couture (2002); Keller *et al*. (2004) |
| Lake Wanapitei | Primarily affected by acid rain and not metals from smelter air emissions | |
| Whitewater Lake | Effluent from tailings dams | Bradley and Morris (1986); Dale and Miller (1978) |
| Whitson Lake | Direct input of liquid effluent from mine tailings and other mine-related wastes | Chen *et al*. (2001); Eastwood and Couture (2002); Rajotte and Couture (2002); Couture and Kumar (2003); Audet and Couture (2003); Keller *et al*. (2004) |

### 5.11.3 Lakes and Watersheds that have been Neutralized

Several lakes and/or their watersheds in the Sudbury area have been neutralized to offset the effects of acid precipitation. Bowland Lake was neutralized in 1983 (Keller *et al.*, 1992a,b). Laundrie Lake is downstream of Bowland Lake, and therefore may have been affected by the neutralization of Bowland Lake, although Laundrie Lake itself has not been manipulated (Keller and Yan, 1991). Lohi Lake (downstream of Clearwater) was limed in 1973 and 1975 (Keller, 2005). Two other lakes, located approximatley 90 km north of Sudbury, were limed: Little Whitepine in 1989; and, Whirligig in 1989, 1993 and 1995. These two lakes were part of the Aurora Trout restoration project (Keller, 2005).

Hannah, Middle and Nelson Lakes were neutralized in the mid-1970s (Yan *et al.*, 1996b). $CaCO_3$ and $Ca(OH)_2$ were added to Middle Lake in the fall of 1973, to Hannah Lake in the spring of 1975 and to Nelson Lake in fall of 1975 and spring of 1976 (Yan and Dillon, 1984). Small amounts of phosphorus also were added to Middle Lake in 1975 to 1978 and to Hannah Lake from 1976 to 1978 (Yan and LaFrance, 1984). Granular limestone and fertilizer were added to the catchments of Middle and Hannah Lakes between 1983 and 1984 as part of Sudbury's reclamation program (Lautenbach, 1987). The catchment of Nelson Lake was not treated (Yan *et al*., 1996b).

A small catchment of Daisy Lake (near Coniston) was limed in 1995 (and perhaps at a later date, also) to examine effects on stream quality (Keller, 2005). Other lakes in Sudbury (*e.g.*, Silver) have been affected by terrestrial liming associated with the regional land reclamation program, although this is not well documented (Keller, 2005).

In summary, the following lakes have been neutralized:

- Bowland
- Laundrie (influenced by neutralization of Bowland)
- Lohi
- Hannah
- Little Whitepine
- Middle
- Nelson
- Whirligig

Granular limestone has been applied to the following catchments:

- Daisy
- Hannah
- Middle
- Silver

None of the following lakes (distance in km from Sudbury in brackets) have been directly subjected to water chemistry manipulations (Keller and Yan, 1991; Keller *et al*., 1992a,b). In fact, these lakes are considered "natural recovery lakes" (Keller *et al*., 1992a,b):

- Baby (1 km from Coniston)
- Clearwater (13)
- Swan (15)
- Wavy (21)
- Joe (28)
- Sans Chambre (29)
- Whitepine (89)

## 5.12 Marshes and Wetlands of Sudbury

The wetlands of the Sudbury region serve an important role as both a unique habitat to numerous species of wildlife and as a source of purification, removing metals and high levels of nutrients from water. One very important wetland within Sudbury is the Lily Creek Marsh. It is a 44.5 ha marsh that extends from

Ramsey Lake to Robinson Lake. Included within this overall area is the Robinson Marsh (25.2 hectares) (L'Institut Canadien de gestion des richesses naturelles-Cambrian College, 1994). The Science North Boardwalk extends through the Lily Creek Marsh, adding to the educational and recreational value of the area. The biodiversity of this area has been extensively documented, including numerous species of birds, vegetation, mammals, reptiles, amphibians, and fish (L'Institut Canadien de gestion des richesses naturelles-Cambrian College, 1994).

The pH of the Lily Creek Marsh has been found to be high, with an average pH of 7.4 in the main channel and levels as high as 10.2 in the west tributary. As a result of these elevated pH levels, the uptake of metals by organisms in the marsh is restricted. Researchers have found that the Lily Creek Marsh is quite healthy overall, despite the fact that it may be receiving significant inputs of metals (Kujanpaa *et al*., 1984).

The concentrations and bioavailability of metals in wetland soils of Sudbury have been studied, although not recently. Twenty-five wetlands located from 1.0 to 75.6 km from the Copper Cliff smelter, as well as several near the Falconbridge and Coniston Smelters, were sampled in July 1980 (Taylor and Crowder, 1983). The substrate sampled ranged from possessing characteristics of soils to true sediments. Concentrations of metals were measured, as well as pH, Eh, organic carbon content and the presence of other ions. The levels of metals found in wetland substrate were similar in magnitude and distribution as found for soils and sediments. There were two highly-contaminated sites located 2.0 and 3.1 km from the Copper Cliff smelter which contained concentrations of copper of 3,738 and 6,912 µg/g, and 9,372 and 5,518 µg/g of nickel, respectively. Overall, the levels of metals decreased significantly with distance from the smelter (Taylor and Crowder, 1983). The organic carbon content of the substrate increased with distance from the smelters, possibly as a result of increased microbial activity leading to a higher incorporation of organic compounds from decomposition into the substrate (Freedman and Hutchinson, 1980).

Overall, concentrations of Cu, Ni, Fe, and Mg in wetland substrates decreased with increasing distance from the smelters, and concentrations of Zn, Mn and Ca did not show a significant change with distance (Taylor and Crowder, 1983). The decreased level of organic carbon in the wetlands located closer to the smelters could result in increased bioavailability of metals and greater toxicity to organisms within these habitats.

## 5.13 Summary and Recommendations of Lakes for Further Study

A review of the available literature describing the physical, chemical, and biological characteristics of Sudbury lakes, as well as the potential sources of metal and biological contaminants, has provided insight for the selection and exclusion of lakes for further consideration in an aquatic ERA. Several lakes have been the focus of biological studies and surveys creating a basis from which to expand further characterizations. However, lakes such as Kelly, Long and Whitson have been directly impacted by mine effluent and should, therefore, not be included in an assessment focused on the risks associated with atmospheric smelter emissions. Other lakes, such as those found within Killarney Provincial Park and those northeast of Lake Wanapitei, have been well studied but are located at distances from the smelters great enough that they have not been significantly impacted by emissions of metals. Several other lakes have been included in, or indirectly affected by, neutralization projects designed to mitigate the impact of acid precipitation. While it is not recommended that these lakes be excluded from further consideration, it is recommended that this influence be recognized should these lakes be selected.

Overall, considering the availability of lake characterization data, distance from the smelters, and potential sources of metals, seven lakes may be recommended for potential inclusion in an aquatic ERA focusing on the impacts of metals from smelter emissions:

- Clearwater
- Hannah
- Middle
- McFarlane
- Nelson
- Ramsey
- Silver

It is expected that this list of lakes may be modified or expanded, particularly since the above-mentioned lakes are located only in the Ramsey and Panache watersheds. It is recommended that the final selection of lakes for further study be completed in consultation with stakeholders (including researchers at Laurentian University), and that a detailed description of the scope for the study be used to help guide this process.

It is also recommended that the marshes and wetlands of the Sudbury area be represented in a risk assessment, as they serve an important role in the purification of sediments and surface waters, as well as providing unique habitat to numerous species of plants and wildlife.

## 5.14 Review of Sudbury-Specific Aquatic Effects Data

The available Sudbury-specific information on effects for each VEC group is summarized in this section. Effects on fish, zooplankton, benthic invertebrates, algae and macrophyes, and amphibians are presented in Sections 5.14.1 through 5.14.5, respectively.

### 5.14.1 Fish

Of the five species of fish selected as VECs (walleye, yellow perch, lake trout, white sucker, and common shiner), Sudbury-specific studies that could be used to assess potential risks were only available for yellow perch and lake trout. These studies, along with a presentation of muscle tissue concentrations measured in perch, walleye and lake herring, are described below.

Many studies have attempted to characterize the impacts that smelter emissions have had on Sudbury fish populations. This has included fish sampling and analysis of historical records, comparisons of concentrations within various tissues to concentrations in the surrounding media, and relating environmental concentrations to physical characteristics and metabolic performance. Some species have been affected to a greater extent than others as a result of sensitivity to acidic conditions or elevated levels of certain metals. Lake trout (*Salvelinus namaycush*) have been most seriously affected by acidification. Results indicated that approximately 94 lakes in the Sudbury region that at one time contained populations of lake trout were no longer able to support these populations (Matuszek *et al*., 1990). Loss of sensitive piscivorous species such as lake trout allowed the more tolerant yellow perch to flourish under conditions with limited predation or competition. However, there is evidence to indicate that, although populations of perch are found in the majority of Sudbury area lakes, environmental conditions may be limiting their growth and success.

Diminished water quality among Sudbury lakes resulted in the loss of lake trout from approximately 60 lakes from 1950 to 1980 (Gunn, 1982; Beggs *et al*., 1985). This occurred in many lakes within 100 km of the Sudbury smelters that were acidified and had elevated concentrations of smelter associated metals, such as Cu and Ni, as well as elevated concentrations of naturally occurring metals such as Al, Mn, and Zn that are released from soils and sediments as a result of acidic conditions. In an effort to restore the local recreational lake trout fishery, hatchery raised fish have been introduced to several Sudbury lakes. Lakes with pH greater than 5.1 have been shown to support survival and growth of stocked lake trout (Bowlby *et al.*, 1988).

A number of fish sampled from Sudbury lakes have been found to contain levels of Hg that surpassed the human consumption guideline of 0.5 ppm, but overall, Sudbury trout have not been shown to have Hg concentrations that would be considered elevated relative to levels in fish from other parts of Ontario. Generally, fish, invertebrates and aquatic mammals in the Sudbury area were found to have relatively low levels of Hg (Wren and Stokes, 1988). This observation was suggested to be the result of high levels of Se in Sudbury lakes resulting from the smelters, acting to retard the rate of Hg uptake from food and water (Rudd *et al*, 1980; Turner and Swick, 1983; Nriagu and Wong, 1983).

Overall, it was demonstrated that metals found within the muscle tissue of fish taken from lakes that are not impacted by the Sudbury smelters were similar to those concentrations found in fish taken from lakes within the Sudbury area that experience higher levels of deposition from the smelting operations. This finding is supported by several other studies that have found that metals, such as lead and cadmium, show low rates of bioaccumulation within fish muscle tissue (Uthe and Bligh, 1971; Falk *et al*., 1973; Benoit *et al*., 1976; Phillips and Russo, 1978; Holcombe *et al*., 1979; Abo-Rady, 1979; Wilson *et al*., 1981). In addition, concentrations of copper, nickel and zinc in fish tissues generally do not vary by fish species or between lakes from 10 to 180 km from the Sudbury smelters (Bradley and Morris, 1986). Significant differences were only found for liver Cu levels among fish species and among lakes (Bradley and Morris, 1986).

Bradley and Morris (1986) note that using sediment levels of Cu alone is not a good method for predicting toxicity to fish. Instead, other water quality parameters such as alkalinity and DOC must also be considered to determine Cu bioavailability. Some studies have gone beyond simply relating surface water and sediment concentrations to tissue concentrations and have attempted to relate uptake to fish condition. To examine the relationships between liver metal (Cu, Zn, and Ni) concentrations and physical condition of yellow perch, northeastern lakes were sampled in the spring and fall of 1997 (Eastwood and Couture, 2002). Of the seven lakes sampled, four (Hannah, Ramsey, Whitson, and Vermillion) were located in Sudbury within 20 km of the smelters, and the remaining three (Michiwakenda, Larder, and Round) were located 60 to 95 km from the Sudbury area. Overall, the Sudbury area lakes had higher aqueous metal concentrations (Hannah and Whitson lakes in particular) than those outside the Sudbury area. All of the lakes in the study had circumneutral pH values (7.05 to 8.41), with the lower values found in Sudbury lakes (Hannah and Whitson primarily). There was a general trend of decrease in pH, alkalinity and DOC and an increase of Cu concentrations closer to the emission sources.

In the spring, concentrations of Cu in the livers of perch from highly contaminated lakes (Hannah and Whitson) were much higher than those from cleaner lakes (Eastwood and Couture, 2002). These differences had decreased or been eliminated by fall. It was suggested that these differences may be the result of elevated metal inputs in spring from snow melt or lake turnover. These results suggest that in all metal-contaminated lakes, there is a point within the year that the capacity to homeostaticaly control Cu is overwhelmed. This is in agreement with laboratory studies and literature reviews by Taylor *et al*. (2000) who found that tissue Cu concentrations only increase when fish are exposed to high concentrations of Cu. Concentrations of Zn within the liver were also influenced by season and concentrations within the lakes, but not to the extent of Cu, although the aqueous concentrations of Zn did not exceed the PWQO, whereas Cu did. This and other studies (*e.g.*, Lucas *et al*., 1970; Falk *et al*., 1973) indicate that Zn concentrations are more effectively regulated by fish.

Metal bioaccumulation in yellow perch collected from eight lakes along a metal contamination gradient was examined by Giguere *et al.* (2005). Four of the lakes were located in the vicinity of Rouyn-Noranda, with the other four in the Sudbury area (Wavy, Hannah, Raft and Laurentian). Fish exposure to Cd, Cu, Ni and Zn was estimated on the basis of calculated free metal ion concentrations in the epilimnion of the sampled lakes. Hepatic Ni concentrations were significantly related to free $Ni^{2+}$ concentrations in the eight lakes examined ($p=0.04$). However, total Cd, Cu and Zn accumulation in the liver was not significantly related to the corresponding ambient free metal ion concentrations or total dissolved metal concentrations ($p>0.05$). In addition, considering potential ionic competition between $M^{2+}$ and $Ca^{2+}$, $H^+$ or other metals, did not yield a significant relationship. During an initial study of yellow perch from eight lakes in the Rouyn-Noranda area, interlake variability in metal bioaccumulation (Cd, Cu and Zn) could be explained in terms of changes in ambient free metal ion and Ca concentrations. However, with the addition of the four Sudbury lakes, aqueous free metal ion concentrations did not explain hepatic metal bioaccumulation in perch. The relationship between aqueous metal concentrations and bioaccumulation in the liver may be hidden by the influence of a food vector or varying aqueous metal concentrations. Varying aqueous concentrations are particularly important for the liver, which integrates exposure over a long period of time. Attempting to predict trace element concentrations in animal organs based on aqueous free metal ion concentrations may be too simplistic, as this approach does not consider changes in metal accumulation pathways, internal redistribution of metals or the physiological condition of the animal (Giguere *et al*., 2005).

Based on comparisons of weight, length, and age of samples, fish from Whitson Lake (and Hannah to a smaller extent) showed slower growth than fish from other lakes. Other indicators such as the relative condition factor (Kn) and scaling coefficients also indicate a lower condition of fish from these lakes ($Kn = W/ L^b$, where W is weight in g, L is fish fork length in cm, and b is the scaling coefficient). The Kn value is a measure of the girth of a fish, or a measure of the storage of energy (protein, lipids, glycogen). This value will increase as a result of a greater than average weight for a particular weight. Fish sampled from Whitson and Hannah lakes had lower Kn values than those from less contaminated lakes. There were consistent seasonal variations in the sampled populations, supporting the use of Kn as a bioindicator of effects resulting from metal contamination (Eastwood and Couture, 2002).

The concept of homeostatic control of metals was further explored by Rajotte and Couture (2002). To examine the effects of Al, Cd, Cu, Ni, and Zn tissue concentrations on the condition, swimming capacities, and tissue metabolic capacities of wild yellow perch, perch from Lakes Nelson, Ramsey, Vermilion, and Whitson were collected in fall 1998 (Rajotte and Couture, 2002). Cadmium and Al are considered to be toxic metals, neither of which are believed to be under homeostatic control. Copper and Zn are considered to be essential metals and are believed to be under homeostatic control. Based on the results of the Eastwood and Couture (2002) study, it was suggested that when there is an overload of homeostatic control, direct toxic effects of metals are more likely to occur. When these mechanisms are not overloaded, the effect of metals may simply be the costs associated with regulating these metals. Nickel is considered to be an essential metal, but regulatory mechanisms are not known for fish (Rajotte and Couture, 2002).

Overall, fish with higher tissue metal concentrations showed reduced conditions relative to fish with lower tissue concentrations. Fish sampled from Whitson Lake (a lake not recommended for further assessment; see Section 5.11.2) were found to have higher liver concentrations, lower growth rates and lower condition than fish from less contaminated lakes studied. Fish from the less contaminated Nelson Lake had lower tissue concentrations and were in better condition. These results are consistent with those of the Eastwood and Couture (2002) study in which the authors found elevated hepatic concentrations of Cu in perch from Whitson Lake and reduced condition and growth in the spring and fall of 1997. Reduced growth rates are suggested to be the result of increased protein anabolism and catabolism, likely involved in repair mechanisms or the regulation of metals. Sherwood *et al*. (2000) found that yellow perch found in lakes contaminated with high metal concentrations required higher energetic costs.

Fish were collected for the purpose of tissue analysis from eight lakes in the Sudbury area as part of the Sudbury Soils Study (SARA, 2006). Six of the lakes sampled (Ashigami, Massey, Long, Crooked, McFarlane and Ramsey) are of particular interest for the current ERA. Ashigami and Massey are located near the northeastern boundary of the recommended study area, and the other four lakes are part of the Urban Lakes monitoring program. Fish also were collected from Vermillion and Whitson Lakes. These data are not discussed further because the lakes have been impacted by direct mining effluents (Section 5.11.2). Long Lake also may have had historical input of metals due to the presence of a gold mine on the lake, but it is included here for consideration. Yellow perch (separated into two size classes) and walleye were caught from each lake, and lake herring from Crooked and Long Lakes.

Metal concentrations (As, Co, Cu, Pb, Ni, and Se) were measured in muscle tissue of the large perch and walleye, and in whole small perch and herring. These concentrations were compared to the lowest relevant effect levels from Jarvinen and Ankley (1999) (Tables 5.18 and 5.19). The database of tissue residue effect levels by Jarvinen and Ankley, both of U.S. EPA, is the most comprehensive review and compilation of data that relate chemical concentrations in tissues of aquatic organisms with measured biological effects. The lowest relevant effect levels represented reduced survival or growth. For example, only data from freshwater fish species, and for juvenile or adult fish (not eggs or embryos), were used. Also, muscle effect levels were used in the comparison with Sudbury fish muscle data, and whole body effect levels were used in the comparison with Sudbury whole body fish metal levels. No tissue residue effect data were available for cobalt in either muscle or whole fish, for lead in muscle, or for nickel in whole fish (Jarvinen and Ankley, 1999). A detailed aquatic ERA could review the available tissue residue data, from Jarvinen and Ankley (1999) and other sources, and develop a less conservative set of screening values for the Sudbury fish tissue data. However, Tables 5.18 and 5.19 provide a preliminary indication of the potential for adverse effects due to metal levels in fish. In addition, it is noted that no fish tissue data are available for other fish species of potential interest (*e.g.,* only one composite sample of golden shiner was analyzed from one lake, therefore, this data point was not included in the analysis); therefore, it is unknown how fissue tissue levels would compare between these and other species.

**Table 5.18 Comparison of Maximum Metal Concentrations Measured in Sudbury Fish Muscle Tissue to Tissue Residue Effect Levels (µg/g wet weight)**

| | | Arsenic | Cobalt | Copper | Lead | Nickel | Selenium |
|---|---|---|---|---|---|---|---|
| | **Effect Level[a]:** | 6 | NA | 0.5 | NA | 58 | 3.1 |
| **Lake** | **Species** | | | | | | |
| Ashigami | Perch (>15cm) | 0.12 | 0.036 | **4.98** | 0.088 | 0.42 | **3.93** |
| | Walleye | 0.23 | 0.0075 | 0.34 | 0.039 | 0.11 | 2.2 |
| Massey | Perch (>15cm) | 0.056 | 0.015 | 0.355 | 0.065 | 0.395 | **3.21** |
| | Walleye | 0.411 | 0.026 | **0.53** | 0.02 | 0.076 | 2.2 |
| Long | Walleye | 2.6 | 0.043 | **1.1** | 0.055 | 2.6 | 1.9 |
| Crooked | Perch (>15cm) | 0.066 | 0.005 | 0.14 | 0.004 | 0.16 | 0.44 |
| | Walleye | 0.12 | 0.026 | 0.46 | 0.044 | 0.51 | 0.74 |
| McFarlane | Perch(>15cm) | 0.03 | 0.021 | 0.39 | 0.04 | 0.21 | 1.1 |
| | Walleye | 0.071 | 0.005 | 0.23 | 0.052 | 0.062 | 0.85 |
| Ramsey | Perch(>15cm) | 0.657 | 0.048 | 0.415 | 0.025 | 0.30 | **3.54** |
| | Walleye | 0.1 | 0.013 | **0.73** | 0.02 | 0.17 | 2.68 |

NA: Not Available
Values **bolded** in grey shading exceeded the corresponding effect level.
[a] Tissue residue effect levels taken from Jarvinen and Ankley (1999).

Concentrations of copper in fish muscle tissue exceeded the tissue residue effect level in one fish from both Ashigami and Massey Lakes (Table 5.18). There were several exceedances in walleye from Long and Ramsey Lakes, although the magnitude of the exceedance was small (generally less than a factor of two). Concentrations of selenium in fish muscle tissue exceeded the effect level in several fish from Ashigami Lake, one fish from Massey Lake, and two fish from Ramsey Lake. Muscle tissue concentrations of all other metals were below their corresponding effect levels (Table 5.18). There was no consistency in the exceedances, and the lakes furthest from the smelters (Ashigami and Massey) had several exceedances. This may be a function of the conservative nature of this evaluation, as the maximum measured tissue concentration was compared to the lowest relevant tissue residue effect level.

Tissue residue effect levels are available for whole fish concentrations (Table 5.19). Only selenium concentrations in fish from Sudbury exceeded the tissue residue effect level. This was observed in several lakes, including the lakes furthest from the smelters (Ashigami and Massey Lakes).

**Table 5.19 Comparison of Maximum Metal Concentrations Measured in Sudbury Whole Fish to Tissue Residue Effect Levels (µg/g wet weight)**

| | | Arsenic | Cobalt | Copper | Lead | Nickel | Selenium |
|---|---|---|---|---|---|---|---|
| | **Effect Level[a]:** | 1 | NA | 7.4 | 4 | NA | 1 |
| **Lake** | **Species** | | | | | | |
| Ashigami | Perch (>15cm) | 0.206 | 0.126 | 1.5 | 0.6 | 0.59 | **2.83** |
| Massey | Perch (>15cm) | 0.835 | 0.073 | 0.65 | 0.052 | 0.54 | **2.3** |
| Long | Perch (>15cm) | 0.483 | 0.295 | 7 | 0.28 | 7.2 | **2.78** |
| | Herring | 0.655 | 0.063 | 0.78 | 0.2 | 0.878 | **2.2** |
| Crooked | Perch(>15cm) | 0.077 | 0.07 | 1.0 | 0.17 | 0.778 | 0.685 |
| | Herring | 0.071 | 0.041 | 0.6 | 0.011 | 0.21 | 0.56 |
| McFarlane | Perch (>15cm) | 0.445 | 0.049 | 0.76 | 0.063 | 0.84 | 1.00 |
| Ramsey | Perch (>15cm) | 0.12 | 0.046 | 0.78 | 0.13 | 0.65 | **2.70** |

NA: Not Available
Values **bolded** in grey shading exceeded the corresponding effect level.
[a] Tissue residue effect levels taken from Jarvinen and Ankley (1999).

Another method used to assess the potential effects of metals on Sudbury fish populations was to determine the resting and active metabolic rates in wild yellow perch from lakes contaminated with copper and cadmium and compare them to fish from unaffected lakes (Couture and Kumar, 2003). This study is a follow up to the Eastwood and Couture (2002) study in which it was demonstrated that there are condition indicators that decrease with increasing metal concentrations. Four lakes from the Sudbury area were chosen to provide a range of values for Cu and Cd contamination (both aqueous and sediment). Halfway Lake is located approximately 90 km north of the Greater City of Sudbury and is not considered to be influenced by mining and smelting activities (representative of background conditions). Hannah, Ramsey and Whitson Lakes are located within the Sudbury basin and have varying concentrations of Cu and Cd. Perch were collected from each of these lakes in June and July of 2000. Liver Cu concentrations reflected the metal concentrations measured in the lake sediments (Couture and Kumar, 2003).

A correlation between liver Cu and Cd concentrations with sediment concentrations supported the suggestion that uptake from the diet is a major route of metal accumulation (Couture and Kumar, 2003). This is also supported by the findings of a Rouyn-Noranda study which showed that yellow perch had a diet primarily composed of chironomids, a benthic invertebrate that has been shown to accumulate metals (Warren *et al*., 1998). Audet and Couture (2003) showed that liver Cu and Cd concentrations were highly influenced by dietary uptake which was heavily influenced by sediment concentrations. When perch shift from a diet of predominantly zooplankton to benthic invertebrates, dietary metal bioaccumulation of metals is significantly increased (Kovecses *et al*., 2002).

Couture and Kumar (2003) showed that liver Cu concentrations in wild yellow perch had a strong relationship with decreases in aerobic capacities of whole fish and tissues. Impaired aerobic capacities in wild fish would reduce the availability of energy for aerobic processes such as locomotion, digestion and growth. This is believed to occur as a result of Cu targeting and inhibiting mitochondrial enzymes. A decrease in aerobic capacity as a result of metal exposure appeared to be compensated for by an increase in anaerobic capacities. This may be a disadvantage because vital processes such as growth, digestion, and sustained swimming are primarily aerobic activities. Any anaerobic activities would ultimately have to be recharged aerobically. This study did not show any correlation between liver Cd and aerobic capacities (Couture and Kumar, 2003). The results of this study show a strong relationship between Cu exposure and the overall health of wild yellow perch (Couture and Kumar, 2003). The four lakes used in the study have near neutral pH, thus eliminating the influence of acidification.

The results of these and other studies have provided evidence that fish within Sudbury area lakes have been, and likely continue to be, adversely affected by the influence of smelter emissions on water and sediment quality. The extent of this impact will vary from lake to lake, and species to species, and may become less significant as the full impact of reductions in smelter emissions are more completely represented in sediment and water quality. However, further recovery of fish populations in lakes that may now offer a hospitable environment for growth and reproduction may be limited by factors such as geographical separation and poor dispersal mechanisms.

### 5.14.2 Zooplankton Communities

In the Sudbury area, lake zooplankton populations and communities began to be studied more than 20 years ago, when acid rain, $SO_2$ emissions and lake pH were of greater concern than today. Planktonic rotifers and crustacean zooplankton, such as daphnids, have been studied relative to their sensitivity to acid pH, as well as to monitor their recovery with changes in pH and other factors (*e.g.*, improved water quality). Section 5.14.2.1 summarizes the effects and “recovery” of plankton communities, for Sudbury in general. Section 5.14.2.2 summarizes the information available on crustacean zooplankton in Middle, Hannah and Nelson Lakes, three of the more intensively studied lakes.

#### *5.14.2.1 Effects and Recovery of Plankton Communities in Sudbury*

MacIsaac *et al*. (1987) examined the planktonic rotifer communities from 47 lakes within 175 km of Sudbury to determine the effects of acid and metals on species distributions. Rotifers were studied because their diversity, abundance and fertility often exceed those of crustacean zooplankton. Water samples and samples of zooplankton were taken in 1984 from the 47 lakes primarily located to the northeast and southwest of Sudbury along prevailing wind directions (MacIsaac *et al*., 1987). Data for individual lakes are not provided; in fact, the names of the lakes studied are not listed within this publication.

The median number of species was significantly higher in non-acid than in acid lakes, but overall, the strongest correlation (in this case, positive) was found between total phosphorous levels and rotifer density (MacIsaac *et al*., 1987). The effect of pH was considered to be indirect. *Keratella taurocephala* and *Gastropus* together accounted for greater than 70% of the total rotifer assemblages in acid lakes, and less than 5% in non-acid lakes. Strong declines in *K. taurocephala* occurred at a pH of 4.8, and this species was almost completely absent at pH greater than 6.0. *Keratella cochlearis* was rarely found in lakes with a pH below 5.3, but was dominant in 88% of lakes with a pH greater than 5.5. Neither *Keratella cochlearis* nor *Polyarthra vulgaris* were found in lakes with a pH below 4.5 (MacIsaac *et al*., 1987). In summary, based on the data from 1984, it appears as though the total phosphorus of a lake (which can influence productivity) may have been more of a factor on rotifer assemblages than pH or metal concentrations. *Keratella cochlearis* was almost exclusively found in lakes that have low levels of metals and pH greater than 5.5. Community composition was also related to alkalinity, Ca, Mg, and Al. No relationships were found with Ni or Cu concentrations (MacIsaac *et al*., 1987).

Recovery of crustacean zooplankton (for the period 1973 to 1986) was determined by comparing species richness in seven acid-impacted Sudbury lakes (Clearwater, Swan, Wavy, Joe, Sans Chambre, Laundrie, Whitepine) to that found in one consistently near-neutral (pH 6.9) Sudbury lake (Welcome) and 24 near-neutral (pH> 6.0) lakes in Dorset Ontario (Keller and Yan, 1991). Only Clearwater and Swan Lakes are within 20 km of Sudbury; Wavy, Joe and Sans Chambre Lakes are 21 to 30 km from Sudbury and the remainder are more than 80 km from Sudbury (Keller and Yan, 1991). The lake water pH, at the time of this study, was between four and five for Clearwater, Swan and Wavy Lakes, and between five and six for the other four acid-impacted Sudbury lakes (Keller and Yan, 1991).

Species richness in the Sudbury lakes was positively correlated with pH (r= 0.84) and negatively correlated with concentrations of Al, Cu and Ni (r= -0.78 to -0.82) (Keller and Yan, 1991). Due to the strong correlation between acidity and trace metal concentrations within Sudbury lakes, it was not possible to separate the effects of metals and the effects of pH on species richness (Keller and Yan, 1991). The absence of *Holopedium gibberum*, *Diaptomus minutus*, and *Mesocyclops edax* from Sudbury lakes may be an indication of metal toxicity since these species are known to not be acid sensitive (Keller and

Pitblado, 1984). Changes in the total number of crustacean zooplankton species and the occurrence of new species are documented over the time period from 1973 to 1986 (Keller and Yan, 1991). The recovery of the *Holopedium gibberum* population in Wavy Lake may be a good indicator that concentrations of trace metals have decreased considering this species is not acid sensitive but is sensitive to metals (Lawrence and Holoka, 1987).

Keller *et al*. (2004) provides a recent review of recovery of Sudbury area lakes (Appendix H1). Populations of a number of acid and/or metal sensitive invertebrate species (*e.g.*, *Holopedium* and *Daphnia mendotae*) have recolonized some lakes (Keller and Yan, 1991; Yan *et al*., 1996b). In 1999, littoral zone mayflies (*Stenonema*) were found in Ramsey Lake; *Stenonema* were not found in surveys of Ramsey Lake in 1995 and 1996 (W. Keller, unpublished data). These changes are considered to be largely a response to reduced metal concentrations (Keller *et al*., 2004). Many lakes close to the smelters still have crustacean zooplankton communities that have fewer species than expected in more pristine, near-neutral lakes. Copepod assemblages in Sudbury lakes appear to be somewhat more typical and have shown greater recovery than cladoceran assemblages, possibly due to the greater sensitivity of cladocerans to metals (Yan *et al.,* 2004). The crustacean zooplankton community of Clearwater Lake has improved, and Keller *et al*. (2004) consider the substantial biological recovery observed in Clearwater Lake to be promising, because this lake was extremely damaged. The rate and extent of biological recovery appear to be related to the initial severity of damage, habitat limitations, and failure to reach the lakes to permit colonization (Yan *et al*., 1996b; Gunn and Keller, 1995).

Metal concentrations in lake water and sediments may still affect aquatic communities in some lakes close to the Sudbury smelters (Keller *et al*., 2004). However, several studies suggest that habitat limitations and colonization issues also are important. Dispersal and colonization may simply be a matter of time, although not all attempts at reintroductions have been successful, suggesting that habitat suitability may not be adequate. There could also be other natural and anthropogenic stressors affecting the aquatic communities (Keller *et al*., 2004).

#### *5.14.2.2 Zooplankton Communities in Hannah, Middle and Nelson Lakes*

Concentrations of several metals (*e.g*., copper, nickel) still exceed water quality guidelines in Hannah and Middle Lakes; few data are available for Nelson Lake (Table 5.4). An exceedance of a guideline does not imply adverse effects on zooplankton communities, but rather identifies the need for further consideration of potential adverse effects. Guidelines rarely account for toxicity modifying factors and other considerations that impact on potential toxicity.

Yan *et al*. (1996a,b) have studied *Daphnia galeata mendotae* in Hannah, Middle and other Lakes. *D. g. mendotae* is a common invertebrate found in North America, and due to its abundance and large size, often makes up a major portion of the average biomass of crustacean zooplankton in the lakes of the southern Canadian Shield (Yan *et al*., 1988). Populations of this species are often impacted by acidification as a result of the occurrence of ionregulatory failure when lake water reaches a pH below 6.0 (Havens, 1992). This species also has demonstrated a strong sensitivity to metals, mainly copper and nickel.

Paleolimnological evidence revealed that Hannah Lake supported a large population of *Daphnia* from 1870 to 1960. Since Middle Lake is located directly downstream of Hannah Lake, it was assumed that Middle Lake also supported Daphnid populations. The loss of populations of *D. g. mendotae* in Middle and Hannah Lakes in the early 1960s originally appeared to be the result of acidification, with pH levels falling below 4.5 (Yan *et al*., 1996b). Liming efforts in the mid-1970s, along with emission reductions during the 1970s and 1980s, allowed the lakes to return to a more neutral pH above 6.0. In fact, Hannah and Middle Lakes were manipulated in several ways: $CaCO_3$ and $Ca(OH)_2$ were added to Middle Lake in the fall of 1973 and to Hannah Lake in the spring of 1975; small amounts of P were added to the lakes in the mid-1970s; Middle Lake was stocked with smallmouth bass, Iowa darter and brook stickleback in 1976 (Yan *et al*., 1996a); a small portion (<10%) of the Hannah Lake watershed was limed, fertilized and seeded with a legume/grass mixture in the early 1980s; and, most of the watersheds of both lakes were planted with conifer seedlings since 1980 (City of Greater Sudbury, 2006). However, the additions of base to the lakes and their catchments were considered the only activities having long-term effects (*e.g*., the stocked fish died, likely due to metal toxicity) (Yan *et al*., 1996a).

The pH of Middle and Hannah Lakes has remained above 6.0 since the mid-1970s. *D. g. mendotae* reappeared in Hannah Lake in 1979, 4 years following liming, and in Middle Lake in 1986, 13 years following liming. In comparison to reference lakes, the populations of *D. g. mendotae* had recovered by 1985 in Hannah Lake and by 1988 in Middle Lake. The timing and pace of recovery of the populations of *D. g. mendotae* in the two lakes was different. Since the effects of acidity were eliminated, the pace of recovery was attributed to elevated concentrations of metals (Yan *et al*., 1996a).

To test the hypothesis that metal concentrations were regulating the pace and extent of recovery, 21-day bioassays were performed in which concentrations of metals were added to lake water to mimic the changing levels during the 1970s and 1980s (specifically, 1976, 1981, 1985, 1989, and 1993) (Yan *et al*., 1996a). Overall, the responses of *D. g. mendotae* to the experimental bioassays were consistent with the historical changes in the abundance of the populations in Middle and Hannah Lakes. The sensitivity of Daphnids to Cu has been well documented. Reductions in longevity have been observed at concentrations of 5 to 10 µg Cu/L (Ingersoll and Winner, 1982; Koivisto *et al*., 1992), a 16% reduction in reproduction at 22 µg/L (Biesinger and Christensen, 1972), and 72-hour LC50s for four species of *Daphnia* (*D. magna, D. pulex, D. parvula*, and *D. ambigua*) ranged from 54 to 87 µg/L (Winner and Farrell, 1976), indicating that populations would be able to survive at concentrations of about 27 to 44 µg/L. Concentrations of Cu averaged higher than 50 µg/L in 1976 in Middle and Hannah Lakes, indicating that lake water was potentially lethal to Daphnids at this time. Concentrations of Cu had dropped to 21 and 28 µg/L in Hannah and Middle Lakes, respectively, by the late 1980s. According to toxicity levels reported by Winner and Farrell (1976), Daphnids would be able to survive in these lakes at this time. This is in agreement with the historical trends in *D. g. mendotae* recovery, indicating that copper alone may have dictated the recovery process (Yan *et al*., 1996a).

Nickel may have also influenced the occurrence of *D. g. mendotae* in Middle and Hannah Lakes (Yan *et al*., 1996a). Munzinger (1990; 1994) reported that nickel at levels greater than 40 µg/L can cause reduced longevity in hard waters, and levels greater than 80 µg/L can reduce clutch size. Levels of nickel in Middle and Hannah Lakes remained greater than 100 µg/L in 1993 (Yan *et al*., 1996a) and still in 2003 (see Table H.4). Results for cadmium are questionable, although it appears cadmium levels would not have interfered with the recovery process (Yan *et al*., 1996a).

Yan *et al*. (1996b) examined the recovery of crustacean zooplankton communities in three Sudbury lakes (Hannah, Middle, and Nelson) 14 to 16 years following the addition of $CaCO_3$ and $Ca(OH)_2$ through comparisons with zooplankton communities in reference lakes. There were 47 spatial reference lakes chosen from Haliburton, Muskoka, Parry Sound, and Nipissing, which were selected because they represented an appropriate range in acidity, nutrients, and morphometry of Canadian Shield Lakes.

A multivariate analysis showed that the zooplankton community of Nelson Lake improved from one characteristic of a shallow, acidic lake during the 1970s to one characteristic of a large, non-acidic, deep, nutrient-poor lake by the mid-1980s. This was considered to be strong evidence of recovery from acidification. Conversely, the zooplankton communities of Middle and Hannah Lake did not recover during the 1980s despite the circumneutral pH. Annual changes in relative abundance of various zooplankton species are shown for the years 1973 through 1989 in Yan *et al*. (1996b). The taxa of zooplankton in these lakes reflected those that would be found in an acidic lake, although Hannah was showing more signs of recovery than Middle (Yan *et al*., 1996b).

Keller *et al*. (1999) evaluated species richness of Sudbury area lakes, and found that even in those lakes in Sudbury that had near-neutral pH, there was often an absence or extreme scarcity of mollusks, amphipods, mayflies and crayfish, all organisms that would be expected to occur in lakes of this nature (Gunn and Keller, 1995; Heneberry, 1997; Reasbeck, 1997; Borgmann *et al*., 1998). The absence of grazers may limit nutrient cycling in many Sudbury lakes. This was observed in Middle and Hannah Lakes, which were artificially neutralized in the 1970s but experience extensive growths of periphyton (Heneberry, 1997; Keller *et al*., 1999). This reduction in energy transfer may be the cause of low numbers of fish within Sudbury lakes (Wright, 1995; Keller *et al*., 1999). The overall absence of key aquatic organisms has been suggested to be the result of either a failure of the ability to reach these lakes to allow for colonization, or an inability to colonize as a result of poor habitat conditions (Gunn and Keller, 1995). In fact, it has been suggested that zooplankton communities may not fully recover until a normal predator community is established, and clearly this is not the case in Hannah and Middle Lakes, where there are few fish species (Yan *et al*., 1996b).

Overall, indicators showed that the zooplankton communities fully recovered in Nelson Lake, partially recovered in Hannah Lake, and did not recover in Middle Lake. This may have been a result of the fact that Middle and Hannah Lake have been acidified for over 60 years, while Nelson had only recently become acidified, as indicated by the presence of populations of lake trout. It may also be a result of elevated concentrations of copper in Middle and Hannah Lakes but not in Nelson Lake. From 1985 to 1989, concentrations of copper in Middle Lake declined from 37 to 28 µg/L, and from 31 to 22 µg/L in Hannah Lake (Yan *et al*., 1996b). Concentrations of copper measured in 2003 (see Table H.4) were 24 µg/L in Middle Lake and 22 µg/L in Hannah Lake. Copper is considered to be toxic to zooplankton in the range of 20 to 50 µg/L (Winner, 1985). There is also a greater number of sources of colonists for Nelson Lake (Yan *et al*. 1996b).

**Summary for Zooplankton Communities in Hannah, Middle and Nelson Lakes**

Zooplankton communities are considered fully recovered in Nelson Lake (Yan *et al*., 1996b). There is strong evidence that populations of the acid-sensitive crustacean zooplankton *D. g. mendotae* in Hannah and Middle Lakes have recovered since the pH of the lake has risen to around 7.0. However, the unusually large populations may be a function of the lack of competitors for the large grazer niche. Current metal levels in Hannah and Middle Lakes generally are not expected to adversely impact this species (Yan *et al*., 1996a; Table 5.4). However, the zooplankton communities of Hannah and Middle Lakes are not considered to have recovered fully (Yan *et al*., 1996b). This may be due to Cu levels in water, which are at the lower end of the range considered to be toxic, or due to a lack of colonizers, and the absence of normal trophic interactions (Yan *et al*., 1996b).

### 5.14.3 Benthic Invertebrates

There are few recent sediment quality data available for Sudbury area lakes (Table 5.8), and for many lakes, only a few metals were measured. There also are relatively few data on the benthic macroinvertebrate communities of Sudbury area lakes. Changes in the benthic macroinvertebrate communities relative to changes in pH were studied (*e.g*., Keller *et al*., 1990; Keller *et al*., 1992b; Keller and Gunn, 1994) but these studies focussed on lakes fairly remote from Sudbury (*e.g*., in Killarney Park or >80 km north of Sudbury) and did not attempt to relate changes in the community with metals levels. However, the biological monitoring studies conducted by researchers in the Sudbury area provide valuable information that can be integrated with metal exposure data in a detailed risk assessment.

For example, Keller *et al*. (1990) evaluated the changes in the community of benthic invertebrates and the resulting effects on the fish community that occurred following the neutralization of Bowland Lake from a pH of 4.9 to greater than 6.0 and the reintroduction of lake trout. The recovery of acid sensitive species appeared to be a slow process following neutralization. In Bowland Lake, there was an observed increase in the number of taxa that are typically found in non-acidic conditions, such as oligochaetes, but even two years after neutralization, recovery was slow. Predation of zoobenthos by fish was determined to be an important factor in establishing the abundance, biomass, and size structure of the zoobenthos community (Keller *et al*., 1990). This study did not measure levels of metals in Bowland Lake and did not relate the structure of the zoobenthos community to metal levels.

A few studies have focussed on accumulation and effects of selected metals on crayfish. However, these studies are not particularly useful for evaluating the risks to these organisms or the benthic macroinvertebrate community in general. For example, two studies have been conducted to compare concentrations of copper, cadmium and nickel (Bagatto and Alikhan, 1987a) and zinc, iron and manganese (Bagatto and Alikhan, 1987b) in freshwater crayfish populations (*Orconectes virilis* and *Cambarus bartoni*) at selected distances from the smelter emission source. Intermoult adults were collected from contaminated lakes (Ramsey and Joe Lake) near the emission source, and from an uncontaminated lake (Wizard Lake) 150 km from the smelter. The general observed relationship for crayfish tissue concentrations was Cd <Ni <Cu (Bagatto and Alikhan, 1987a) and Mn < Zn < Fe < Mg (Bagatto and Alikhan, 1987b). Tissue concentrations for Cd, Ni, Cu, Zn and Fe were higher at the near sites (*i.e*. Lake Ramsey and Lake Joe) in comparison to the reference site (*i.e*. Lake Wizard) (Bagatto and Alikhan, 1987a, b). These studies support research that heavy metals tend to accumulate in crustaceans in the form of granules within the hepatopancreas (Loizzi and Peterson, 1971). As is seen in fish studies, it appears that crustaceans have a threshold above which they are not able to effectively regulate internal Cu concentrations (Bagatto and Alikhan, 1987a).

Taylor *et al*. (1995) compared the sensitivity of the freshwater crayfish, *Cambarus robustus*, to various copper concentrations from an acidified metal-contaminated reservoir (Wavy Lake, 26 km downwind of Copper Cliff) in Sudbury and a fast-flowing, circumneutral, uncontaminated stream (Pike Creek, 150 km northeast of Copper Cliff). The 24-hour LC50s were 4.07 mg/L for Wavy Lake crayfish and 3.48 mg/L for Pike Creek crayfish (Taylor *et al*., 1995). The EC50s from this study indicate that the crayfish from Wavy Lake have an increased tolerance for Cu compared to Pike Creek crayfish (Taylor *et al*., 1995).

The impact of atmospherically deposited metals on aquatic ecosystems in Sudbury was evaluated by Borgmann *et al*. (1998, 2001a, 2001b) in 1996 and 1998 through an examination of the benthic invertebrate communities. Four potentially impacted lakes in the immediate Sudbury area (<13 km from Copper Cliff) were compared against eight lakes located at intermediate (35 to 52 km) and considerable (94 to 154 km) distances from Copper Cliff (Table 5.20). All of the lakes were circum neutral, with pH values of 6.6 to 8.3 near the surface.

**Table 5.20 Test and Reference Sites at Which *in-situ* Benthic Invertebrate Abundances and Sediment Toxicity Were Evaluated[a]**

| Sudbury Lakes (<13km) | Intermediate Lakes (35-52 km) | Reference Lakes (94-154 km) |
|---|---|---|
| Ramsey | Nepewassi | Tomiko |
| McFarlane | Kakaswaganda | Restoule |
| Raft | Trout | Nosbonsing |
| Richard | Lower Sturgeon | Talon |

[a] Distances are described as proximity to Copper Cliff (Borgmann *et al*., 1998).

Concentrations of metals were found to be higher within the sediments of the Sudbury lakes relative to those of the intermediate or reference lakes. Increased bioaccumulation of Cd, Co and Ni in laboratory exposed *Hyalella azteca* was observed, with body concentrations that were respectively, 17, 4.5 and 3.7 fold greater than amphipods exposed to sediments from the reference lakes (Borgmann *et al*., 2001b). Conversely, As, Cr, Mn, Pb, Se, and Tl concentrations observed in *H. azteca* did not differ significantly among amphipods exposed to sediments from the Sudbury, reference, and intermediate lakes (Borgmann *et al*., 2001b).

Metal concentrations in test organisms were found to be the most direct indicator of metal bioavailability compared to concentrations in overlying water. However, concentrations in overlying water can provide supplemental information and are useful, for example, when investigating toxic effects of Cu as it is difficult to determine concentrations of Cu in test organisms. Measured concentrations of metals in overlying water are only useful if the observed toxicity can be attributed to the dissolved phase rather than the portion found within the sediment. To determine the cause of observed toxicity during chronic exposure experiments, Borgmann *et al.* (1998, 2001a, 2001b) exposed two groups of test organisms within the same container to different treatments. One group was exposed to the solid phase sediment while the second group was placed in cages above the sediment. Survival in the cages correlated well with survival of amphipods exposed directly to the sediment, and there were no cases in which high mortality in the sediment was not also associated with high mortality in the cage. It was therefore proposed that in the Sudbury lakes, toxicity was due to the dissolved phase of a substance, rather than that found within the bed sediments.

*In situ* benthic invertebrate communities were sampled at each of the sample lakes. Midges belonging to *Chaoboridae*, *Chironomini* and *Tanypodinae*, and *Oligochaetes* were the most frequently observed organisms in all of the lakes. Their abundance did not correlate with the distance from Sudbury (Copper Cliff) (Borgmann *et al*., 2001a, 2001b). A significant difference in clam abundance between the Sudbury and reference lakes was observed in both the 1996 and 1998 data sets, as fingernail clams (*Pisidiidae*) were completely absent from the Sudbury lakes (Borgmann *et al*., 1998, 2001b). Based on the pooled data from both sample years, there was a significant absence of midges (*Tanytarsini*) from all but one of the Sudbury lakes relative to the reference sites ($p<0.05$, Mann-Whitney U test). In addition, amphipods were mostly absent from the Sudbury lakes, however, amphipods were also absent from six of the reference and intermediate lakes. Therefore, no significant difference was determined (Borgmann *et al*., 2001a, 2001b).

To discriminate response patterns at the community level between lakes, total abundance, number of taxa (species) and ordination summaries at the lowest taxonomic level (species) were evaluated. No relationship was found between abundance, number of taxa, or the ordination axes with increasing distance from Sudbury, based on correlation analysis or ANOVA (Borgmann *et al*., 1998).

Chronic sediment toxicity tests were conducted on four benthic invertebrate species in 1996. Tests on *Chironomus* and *Hyalella azteca* were repeated in 1998 (Borgmann *et al*., 2001a, 2001b). Test organisms were exposed to various treatments including sediment obtained from the Sudbury, intermediate and reference lakes. No relationship could be detected between survival or growth of *Chironomus riparius* and distance from Copper Cliff (Borgmann *et al*., 2001a). Significantly lower growth of *Hexagenia* was observed in the Sudbury area sediments than in all others, and there were no significant differences between the other groups (ANOVA with Tukey test, $p<0.001$) (Borgmann *et al*., 2001a). The mean survival and growth of *Hyalella azteca* were both significantly lower in the Sudbury lakes than in either the intermediate or reference lakes (ANOVA with Tukey test, $p<0.01$). Sediment toxicity to *Hexagenia* and *Hyalella* correlated closely with the measured Ni concentration in the overlying water, with the exception of *Hexagenia* growth. Most of the *Tubifex tubifex* survived the toxicity tests. However, the total number of live young produced was lower in the Sudbury area sediments than in intermediate or reference lake sediments (ANOVA with Tukey test, $p<0.01$) (Borgmann *et al*., 2001a). In addition, there was no significant difference in the presence of empty or full cocoons among the sediment groups.

Benthic community data and chronic toxicity data collected in 1998 supported those findings from the previous study in 1996 (Borgmann *et al.*, 2001a, 2001b). In addition, a strong correlation was observed between the benthic survey and sediment toxicity test results. Taken together, *in-situ* invertebrate abundance and toxicity tests indicated that there are biological impacts associated with the sediments at water depths of $\geq$ 10 m. Metal bioaccumulation and comparison of metals in overlying water demonstrated that Ni was likely the primary cause of sediment toxicity to benthic invertebrates (Borgmann *et al*., 1998, 2001a, 2001b).

In order to obtain a better understanding of the biological significance of metal profiles within sediment cores from the Sudbury area, a follow-up study conducted by Borgmann and Norwood (2002) examined the impact of sediment samples on benthic invertebrates. Several sediment cores were collected from Richard Lake (<13 km from Copper Cliff) and total metal concentrations, toxicity to *Hyalella*, and metal bioaccumulation by *Hyalella* were measured. Metal concentrations were lowest in the deepest part of the core, but increased rapidly until a depth of approximately 5 cm. Chronic toxicity (4 weeks) of the sediment to *Hyalella* produced a profile that matched metal concentrations found in the sediment. Mortality was almost complete for amphipods exposed to sediments from 3 to 14 cm. However, survival was high ($\geq$80%) in the deepest sediment. Chronic mortality of the young amphipods exposed to sediment correlated with Ni bioaccumulation in the caged adult *Hyalella* exposed to water overlying the sediment for one week. Nickel was the only metal that accumulated to levels greater than the chronic lethal body concentration. Borgmann and Norwood (2002) concluded that sediment cores obtained from the Sudbury area are toxic to amphipods, that Ni appears to be the primary cause of toxicity, and that toxicity is due to a dissolved substance. In addition, based on Pb-210 dating and Ni trends in the sediment core samples, Borgmann and Norwood (2002) suggest that the chronic toxicity of the surface sediments of Richard lake may exist for approximately 15 years.

### 5.14.4 Algae and Macrophyte Communities

No recent studies of algal communities in Sudbury area lakes were found. The only recent information on phytoplankton communities comes from a brief submitted for inclusion in the 2004 National Acid Rain Assessment Report (Winter *et al*., 2004) which states that the phytoplankton community of Clearwater Lake is similar to communities of near-neutral, more pristine lakes (Winter *et al*., 2004). In addition, there have been complaints of large algal blooms in three residential lakes, 5 km downstream of Kelly Lake (Wainio *et al*., 2003).

Older studies focussed on the influence of water pH (acidity) and aluminium on algae and diatoms, but in acid lakes outside of the ERA Study Area (*e.g*., in Killarney Park or north of Sudbury) (*e.g*., Dixit *et al*., 1992; MOE, 1992).

The Ontario Ministry of the Environment (MOE, 1992) study was conducted to determine if there was a relationship between phytoplankton community structure and acidification in a large 111 lake set, as well as in a smaller group of seven lakes, within a 260 km radius of Sudbury. Data were collected between 1974 and 1986. The group of seven lakes included Swan, Wavy, Joe, Sans Chambre, Laundrie, Welcome and Whitepine Lakes. Swan Lake is the closest to Sudbury (15 km from smelters), with others being a significant distance from the smelters (e.g. Whitepine at 90 km from smelters). A detailed analysis of taxonomic changes was undertaken for Joe, Laundrie and Welcome Lakes. This study focussed on the relationship between phytoplankton communities and environmental factors such as alkalinity and pH. Alkalinity, total phosphorus, and aluminium were found to be the most important factors related to the number of phytoplankton taxa (MOE, 1992). All of the bloom-forming blue-green algae (*Bicosoeca*, *Trachelomonas*, *Planctonema*, *Gomphonema*, *Stauroneis*, *Coelastrum*, *Tetrastrum*, *Eudorina* and *Dictyosphaerium*) appeared to have low tolerance for metals. There was a highly significant linear correlation between lake pH and the number of taxonomic groups of phytoplankton in the seven-lake set, and between alkalinity and the number of taxonomic groups in the larger 111 lake set (MOE, 1992).

Several studies were conducted in the early 1970s, when lake pH was lower and $SO_2$ and metal emissions from the smelter were higher than they are today (*e.g*., Hutchinson, 1973; Hutchinson and Stokes, 1973; Stokes *et al*., 1973). In the Stokes and Hutchinson series of studies, it was found that the pattern of responses to metals differed between species harvested from Sudbury Lakes (*Chlorella fusca* from Baby

Lake and *Scenedesmus acutiformis* from Boucher Lake) and those typically tested in laboratory bioassays (*Chlorella vulgaris* and *Scenedesmus acuminatus*). In laboratory species, a threshold effect was observed, in which inhibition and death occurred above a critical level, with little inhibition below this concentration. Algae taken from the polluted Sudbury lakes did not show a threshold effect, and instead had a gradual reduction in growth with increasing concentrations of metals (Stokes *et al*., 1973). Bioassays also were conducted on four laboratory species of algae (*Chlorella vulgaris, Scenedesmus acuminate, Haematococcus capensis* and *Chlamydomonas eugametos*) exposed to water from five lakes (Boucher, Alice, Baby, Kelley, and Long), again from 1970. Algal growth was adversely affected in all lake waters, and the algae also accumulated significant quantities of metals (Hutchinson and Stokes, 1973).

No recent data are available on macrophyte communities of Sudbury lakes of interest for this ERA. Macrophytes are being considered as part of a remediation strategy in Sudbury, for example, in Kelley Lake (Wainio *et al*., 2003). Kelley Lake has numerous other sources of contamination, and therefore is not recommended for study within this ERA (see Section H-6.2). However, a vegetation survey was conducted in 2000 for this lake, and eight species of submergent plants and 20 species of marsh plants were found (Wainio *et al*., 2003). Macrophytes were surveyed in six lakes during 1977 and 1978: Clearwater, Lohi, Hannah, Middle, Nelson and Labelle Lakes (Nriagu, 1984). The vascular macrophyte species present in Clearwater Lake were *Eriocaulon septangulare*. *Eleocharis acicularis*, *Juncus pelocarpus*, *Myriophyllum tenellum* and *Lycopus spp*. Concentrations of Ni, Cu, Zn, Cd, and Mn in the tissues of pipewort (*Eriocaulon septangulare)* taken from sites in Clearwater Lake were significantly higher than those in the adjacent littoral sediments (Nriagu, 1984). However, these data are too old to be of use in the ERA.

### 5.14.5 Amphibians

There are a limited number of studies available that have described the distribution and status of amphibian populations in the Sudbury area. A current list of species present in the Sudbury region is presented in Section 5.9.5. The most extensive survey reviewed was conducted in 1982 and 1983 in which 118 potential amphibian breeding sites, between nine and 66 km northeast or southwest of Sudbury, were studied (Glooschenko *et al*., 1992). Of the 13 species of amphibians that were anticipated to occur in this area (Cook, 1984), 12 were identified within the survey. The two most common species were *Hyla crucifer* and *Rana sylvatica*, which were each found in 32 locations. *Rana pipiens* was found in 14 locations, *Bufo americanus* in 13, *Rana clamitans* in 10, and *Hyla versicolor* in nine. The remaining six species identified in the Sudbury area (*Plethodon cinereus*, *Rana septentrionalis*, *Rana catesbeiana*, *Ambystoma maculatum*, *Ambystoma jeffersonianum* complex, and *Ambystoma laterale*) were found in six or fewer locations.

Results of this study show a significant relationship between the buffering status of surface water (which decreases at lower pH) and the occurrence of *H. crucifer, R. pipiens* and *R. clamitans,* indicating that the acidification of breeding pools may have resulted in the loss of some species in Sudbury locations (Glooschenko *et al*., 1992). Although the Sudbury area is considered to be within the range of *A. maculatum*, this species was only identified at two of the 118 locations surveyed. This may be the result of the sensitivity of *A. maculatum* to acidification, with pH in the range of 4.0 to 4.5 resulting in mortality in 85% or more of embryos within a few hours, and pH of 4.5 to 5.0 resulting in high embryonic mortality and other adverse effects (Clark and LaZerte, 1987; Dale *et al*., 1985; Freda and Dunson, 1985; Pough and Wilson, 1977). Low pH may also be responsible for the reduced number of *R. sylvatica* egg masses observed in Sudbury ponds.

Numerous studies have demonstrated the sensitivity of amphibian reproductive success to low pH in breeding water (Karns, 1983; Freda *et al*., 1990). The number of *Rana sylvatica* egg masses per pond and per communal aggregate was found to be lower in the Sudbury survey than in several other studies. This observation was suggested to be the result of the inhibitory effects of low pH (Glooschenko *et al*., 1992).

Levels of copper, zinc, iron, and aluminium in at least some of the Sudbury ponds were above the levels associated with adverse effects in amphibians. A negative relationship was found for aluminium and *R. clamitans*, and the absence of *B. americanus* and *R. pipiens* in some locations was associated with elevated levels of nickel and zinc, respectively (Glooschenko *et al*., 1992). Overall, high concentrations of metals that have been associated with smelter emissions may have had adverse effects on amphibian populations in ponds in the Sudbury area, at least in the past (this survey was conducted in 1982/3).

No recent studies were found that evaluated amphibians in the Sudbury area, and therefore current levels of impact to amphibian populations are unknown. In addition, water quality data are not available for Sudbury area wetlands or woodland ponds in which many amphibians breed. This represents a significant data gap for an aquatic ERA. Therefore, to provide a preliminary indication of potential impacts to amphibians at the present time, water quality data from Table 5.4 were compared to available toxicity benchmarks for amphibians.

The Lily Creek Marsh is a well-known wetland area within the City of Greater Sudbury, which extends from Ramsey Lake to Robinson Lake (Section 5.12). Concentrations of COC in water from Ramsey and Robinson Lakes can be compared to amphibian toxicity benchmarks as a preliminary evaluation of potential risks to amphibians in this wetland (Table 5.21). As a point of comparison, the maximum measured concentration of metals from any lake (Table 5.4), with the exception of Whitson and Kelly Lakes (two lakes known to receive mine effluent), is also presented.

Limited metal toxicity data exist for amphibians, with the exception of LC50 data (the dose that results in 50% mortality in a toxicity test). A recent summary of metal toxicity data for amphibians is provided in Linder and Grillitsch (2000). Concentrations of cadmium, cobalt, lead and zinc are below the lowest amphibian toxicity values (Table 5.21). The concentration of copper in only one lake (Crooked) marginally exceeds the amphibian toxicity value. Concentrations of nickel in Ramsey Lake exceed the lowest toxicity value. However, few toxicity data are available for nickel, and the lowest toxicity values are for species not native to the Sudbury area. Only a few Sudbury-area lakes contain nickel at concentrations that exceed 150 µg/L (Table 5.4).

**Table 5.21 Comparison of Metal Concentrations in Sudbury Lakes to Amphibian Toxicity Benchmarks**

| | | **Concentration in Lake Surface Water[c]** | | | |
|---|---|---|---|---|---|
| **COC[a]** | **Toxicity Value[b]** | **Maximum[d]** | **Ramsey** | **Robinson** | **Comment** |
| Cadmium | 9 | 0.99 | 0.1 | <1 (nd) | Lowest toxicity data is a NOAEL for growth. Lowest LC50 = 40 µg/L |
| Cobalt | 50 | 9.2 | <1.5 (nd) | <1.5 (nd) | Lowest toxicity data is an LC50 for a toad native to the southeastern US. |
| Copper | 40 | **48** | 14.4 | 10 | Lowest toxicity data is an LC50 for three species. |
| Lead | 40 | 22 | <11 (nd) | <11 (nd) | Lowest toxicity data is an LC50 for a toad native to the southeastern US. Next lowest toxicity data is a NOEC for avoidance in the American Toad of 500 µg/L. |
| Nickel | 50 | **319** | **74** | 36 | Lowest toxicity data is an LC50 for a toad native to the southeastern US. Next lowest toxicity data is an EC50 of 150 µg/L for the African clawed frog. |
| Zinc | 800 | 36 | 2 | 1 | Lowest toxicity data (10 µg/L) is an LC50 for a toad native to the southeastern US, and this value is |

**Table 5.21 Comparison of Metal Concentrations in Sudbury Lakes to Amphibian Toxicity Benchmarks**

| COC[a] | Toxicity Value[b] | Concentration in Lake Surface Water[c] Maximum[d] | Ramsey | Robinson | Comment |
|---|---|---|---|---|---|
| | | | | | lower than the PWQO of 20 µg/L, thus the next lowest toxicity data was selected for screening. |

[a] COC for water from Section 5.5
[b] µg/L; taken from Linder and Grillitsch (2000) except for cobalt data that were obtained from Birge *et al*., 1979.
[c] µg/L; taken from Table 5.4; nd indicates non-detect concentration with detection limit given
[d] Maximum taken from Table 5.4, with exception of Whitson and Kelly Lakes
Shaded values in **bold** text exceed the corresponding toxicity value.

The comparison in Table 5.21 provides only a preliminary indication of potential risks to amphibians, from exposure to metals in water during the tadpole stage. A detailed aquatic ERA may require additional sampling of water and possibly amphibians from wetlands and ponds of the Sudbury area to more fully understand the potential impacts of metals on amphibians.

### 5.14.6 Summary of Effects Data

The availability of Sudbury-specific data to describe the potential effects of elevated metal levels in surface water and sediment on VEC groups recommended for the aquatic assessment is generally limited. General correlations between population characteristics and environmental concentrations have been made for several groups, but the nature of these relationships has not been fully explored. For example, the number of species in the planktonic rotifer community has been observed to be lower in lakes suffering from acidification, but this has been shown to be more heavily influenced by lake productivity (*i.e.*, phosphorous) than pH or levels of metals. Crustacean zooplankton species richness was positively correlated with pH and negatively correlated with concentrations of aluminium, copper and nickel, although limitations on organism dispersal and colonization also may be significant factors influencing the populations. Few Sudbury-based studies have focused on the effects of metals on benthic invertebrates. This is a significant data gap that will need to be addressed in further studies. There are limited recent studies on algae and macrophyte communities in Sudbury lakes. Alkalinity, total phosphorous and aluminium were found to be the most influential factors related to the number of phytoplankton taxa in Sudbury lakes. There are also a limited number of studies describing the effects of metals on the amphibian populations in Sudbury. A significant relationship was observed between the buffering status of surface water and the occurrence of certain species, and it has been suggested that levels of metals such as nickel may be limiting the distribution of amphibians in the area.

The majority of relevant effects data for Sudbury lakes is available for fish, and in particular, yellow perch. The Sudbury-specific data for yellow perch include physical characteristics and metabolic performance, as well as comparisons between environmental concentrations and levels within various tissues. Fish muscle tissue typically shows a low rate of bioaccumulation, whereas the liver has been shown to significantly accumulate metals such as copper, particularly during certain periods of the year. Accumulation generally occurs when surface water and sediment concentrations reach levels that overwhelm the capacity of the fish for homeostatic control (*e.g.,* copper and zinc). Slower growth and reduced fish condition have been observed in fish taken from heavily-impacted lakes relative to those found in more remote areas. Fish tissue analyses conducted as part of the Sudbury Soils Study generally found that levels of metals in sampled fish were below levels associated with potentially toxic effects, with the exception of copper and selenium, which had slight exceedences.

## 5.15 Evaluation of Risks to Wildlife with an Aquatic-based Diet

Three VECs were selected to represent wildlife with a significant proportion of their diet arising from the aquatic environment: common loon, mallard and mink. The process used to select these VECs is described in Chapter 2 of this report. Methods used to estimate risks are detailed in Chapter 4 and associated appendices. The following subsections provide additional detail for loon, mallard and mink with regards to the exposure assessment (Section 5.15.1), the effects assessment (Section 5.15.2) and the risk characterization (Section 5.16).

### 5.15.1 Exposure Assessment

The methodology used to model exposures to loon, mallard and mink is described in detail in Section 4.1.1 of Chapter 4 this report. Probabilistic methods were used to estimate exposures, using the full distribution of COC concentrations in various media. General descriptions of the various parameters are provided in Section 4.1.1, and life history data used in the model are provided in Appendix D4. Surface water, sediment, fish and soil data used to estimate exposures are described in Chapter 4. Histograms illustrating the surface water, sediment and fish data are provided in Appendix D1. Biological details (species, fork length, total length, weight) for the forage fish collected from Sudbury-area lakes are provided in Appendix H2.

Exposures were estimated only for those COC that were identified in Chapter 2: arsenic, cadmium, cobalt, copper, lead, nickel and selenium. Additional COC (*i.e.*, aluminium, iron, chromium, manganese, mercury, vanadium and zinc) that were identified in Section 5.4 were not assessed because they were selected based on toxicity to aquatic life (*i.e.*, aquatic plants, invertebrates and fish).

### 5.15.2 Effects Assessment

The effects assessment is divided into a discussion of Toxicity Reference Values (TRVs) in Section 5.15.3 and field information on reproductive success and population trends in Section 5.15.4

### 5.15.3 Toxicity Reference Values

Table 5.22 presents the TRVs for each VEC and COC combination. Detailed discussions of the TRV derivation procedures for each COC are provided in Chapter 4 and discussed briefly below. Toxicological profiles for the COC are provided in Appendix F.

**Table 5.22 Toxicity Reference Values (TRVs) for Valued Ecosystem Components (VECs) Exposed to Chemicals of Concern (COC)**

| COC | VEC | Test species | Test Species Effect Level (mg/kg/d) | IC20, LOAEL or NOAEL | TRV (mg/kg/d) | Reference |
|---|---|---|---|---|---|---|
| Arsenic | Common Loon | Mallard | 14 | IC20 | 14 | U.S.EPA, 2001 derived from Stanley *et al.*, 1994 |
| | Mallard | Mallard | 14 | IC20 | 14 | |
| | Mink | Mouse | 10 | IC20 | 3 | Byron *et al.*, 1967 |
| Cadmium | Common Loon | Chicken | 3.1 | LOAEL | 3.1 | U.S. EPA, 2005 geomean of 35 studies |
| | Mallard | Chicken | 3.1 | LOAEL | 3.1 | |
| | Mink | Rodents, sheep, pig | 5.4 | LOAEL | 5.4 | U.S. EPA, 2005 geomean of 28 studies |
| Cobalt | Common Loon | Chicken | 16 | LOAEL | 16 | U.S. EPA, 2005 geomean of 11 studies |
| | Mallard | Chicken | 16 | LOAEL | 16 | |
| | Mink | Rodents, Pig, Cow | 19 | LOAEL | 19 | U.S. EPA, 2005 geomean of 14 studies |
| Copper | Common Loon | Chicken | 60 | IC20 | 60 | U.S. EPA, 2001 derived from Mehring *et al.*, 1960 |
| | Mallard | Chicken | 60 | IC20 | 60 | |
| | Mink | Mink | 19 | IC20 | 19 | U.S. EPA, 2001 derived from Aulerich *et al.*,1982 |
| Lead | Common Loon | Chicken | 10 | IC20 | 10 | U.S. EPA, 2001 derived from Edens and Garlich, 1983 |
| | Mallard | Chicken | 10 | IC20 | 10 | |
| | Mink | Rat | 80 | LOAEL | 27 | Azar *et al.*, 1973 |
| Nickel | Common Loon | Mallard | 77 | LOAEL | 77 | Cain and Pafford, 1981 |
| | Mallard | Mallard | 77 | LOAEL | 77 | Sample *et al.*, 1996 |

**Table 5.22 Toxicity Reference Values (TRVs) for Valued Ecosystem Components (VECs) Exposed to Chemicals of Concern (COC)**

| COC | VEC | Test species | Test Species Effect Level (mg/kg/d) | IC20, LOAEL or NOAEL | TRV (mg/kg/d) | Reference |
|---|---|---|---|---|---|---|
| | Mink | Rat | 50 | LOAEL | 17 | based on Ambrose *et al.*, 1976 |
| Selenium | Common Loon | Mallard, Chicken, Kestrel | 0.4 | LOAEL | 0.4 | Heinz *et al.*, 1989; Ort and Latshaw, 1978; Santolo *et al.*, 1999 |
| | Mallard | Mallard | 0.4 | LOAEL | 0.4 | Heinz *et al.*, 1989 |
| | Mink | Rats | 0.33 | LOAEL | 0.1 | Sample *et al.*, 1996 based on Rosenfeld and Beath, 1954 |

Effect level for test species is either a LOAEL = Lowest-observed-adverse-effect level) or an IC20 = concentration resulting in 20% level of effect on test organism.

**Arsenic**

*Birds*

TRVs were developed from a study of 99 breeding pairs of mallards (Stanley *et al.*, 1994) administered sodium arsenate in their diet during their reproductive cycle. U.S. EPA (2001) estimated a NOAEL of 9 mg/kg/d, a LOAEL of 40 mg/kg/d, and derived an IC20 of 14 mg/kg/d from this study. The IC20 of 14 mg/kg/d was the TRV for the loon and mallard.

*Mammals*

Mink are not closely related to any of the test species for which toxicity data are available (Appendix F). Therefore, an uncertainty factor of three was applied to the small mammal IC20 of 10 mg/kg/d (Byron *et al.*, 1967) to derive the TRV for mink of 3 mg/kg/d.

**Cadmium**

*Birds*

The U.S. EPA (2005) conducted a detailed review of published studies on the toxicity of cadmium to avian species, 35 of which met the quality criteria for use in developing an Eco-SSL. Endpoints evaluated included reproduction, growth, and survival. A total of 11 bounded LOAELs (five with reproductive endpoints, six with growth endpoints) were considered in the calculation of a TRV for loons. Three of the reproduction LOAELs and one of the growth LOAELs were significantly greater (7.1 to 37.6

mg Cd/kg/day) than the other data points, and therefore, were omitted from the analysis in the interest of conservatism. The geometric mean of the remaining seven bounded LOAELs was 3.1 mg Cd/kg/day and was used as the TRV for loon and mallard.

*Mammals*

The U.S. EPA (2005) conducted a detailed review of published studies on the toxicity of cadmium to mammalian species, 145 of which met the quality criteria for use in developing an Eco-SSL. Endpoints evaluated included reproduction, growth, and survival. A total of 35 bounded LOAELs (11 with reproductive endpoints, 24 with growth endpoints) were considered in the calculation of a mammalian TRV. Three of the reproduction LOAELs and four of the growth LOAELs were significantly greater than the other data points (doses greater than 40 mg Cd/kg/d), and were, therefore, omitted from the analysis in the interest of conservatism. The geometric mean of the remaining 28 bounded LOAELs was 5.4 mg Cd/kg/day and was used as the TRV for mink.

**Cobalt**

*Birds*

There are limited studies on the toxicity of cobalt to birds. The U.S. EPA (2005) conducted a detailed review of published studies on avian toxicity and determined that studies from 11 papers were of sufficient quality for use in developing an Eco-SSL. Endpoints included growth and mortality (no studies reported reproductive effects). The geometric mean of 11 of the 12 LOAELs (one was much higher than the others [148 mg/kg/d] and was omitted from the calculation) was 16.1 mg/kg/d (range of 7.8 to 38 mg/kg/d). The TRV for loon and mallard is 16 mg/kg/d (LOAEL).

*Mammals*

There are limited studies on the toxicity of cobalt to mammals, and none were chronic in duration. The U.S. EPA (2005) conducted a detailed review of published studies on mammalian toxicity and determined that studies from 18 papers were of sufficient quality for use in developing an Eco-SSL. Species included mice, rats, cows, pigs and guinea pigs. Endpoints included growth and reproduction. The geometric mean of the 14 LOAELs was 19 mg/kg/d (range of 1 to 122 mg/kg/d), which was used as the TRV for mink.

**Copper**

*Birds*

The dietary requirement of poultry has been estimated at between four and five mg Cu/kg diet (NRC, 1980). U.S. EPA (2001) selected Mehring *et al.* (1960) as the definitive study from which to estimate effect concentrations. The IC20 (copper as copper oxide) for chickens was 72 mg/kg/d for survival and 60 mg/kg/d for growth. The IC50 for growth of chickens was 94 mg/kg/d. The TRV for loon and mallard was selected as 60 mg/kg/d.

*Mammals*

An IC20 of 19 mg/kg/d was derived for mink exposed to copper sulphate (U.S. EPA, 2001). This was based on a study by Aulerich et al. (1982). The TRV for mink was selected as 19 mg/kg/d.

**Lead**

*Birds*

The data from one study on quail (Edens and Garlich, 1983), which reported the lowest LOAEL (1.1 mg/kg/d), could not be fit to a dose-response model (U.S. EPA, 2001). However, the data from the next lowest reported LOAEL could (also Edens and Garlich, 1983). The LOAEL from this study on chickens was 3.5 mg/kg/d (with lead as lead acetate), and the data were fit to a dose-response model to develop an IC20 of 9.9 mg/kg/d (U.S. EPA, 2001). Other reported reproductive LOAELs were much higher, including one of 70 mg/kg/d for chickens exposed to lead oxide (Hermayer *et al*., 1977), which may in fact be more relevant to the situation at Sudbury. However, the IC20, rounded to 10 mg/kg/d, was used as the TRV for loon and mallard.

*Mammals*

Because the bioavailability of lead in drinking fluid is significantly greater than that from diet or soil, the TRV was based on a study that exposed rats to lead acetate in diet over three generations (Azar *et al*., 1973). None of the exposure levels affected the number of pregnancies, the number of live births or other reproductive indices. However, a concentration of 1,000 mg/kg diet (converted to 80 mg/kg/d in U.S. EPA 2001) resulted in reduced offspring weights and kidney damage in young. The dose of 80 mg/kg/d was converted to the TRV for min of 27 mg/kg/d by applying an uncertainty factor of three.

**Nickel**

*Birds*

Cain and Pafford (1981) fed nickel sulphate to newly hatched mallard ducklings in their diet at three dose levels (176, 774 and 1,069 mg/kg) from one to 90 days of age. Ducklings exposed to 1,069 mg/kg diet displayed tremors and showed signs of paresis (impaired mental function) after 14 days, and at 28 days, weighed significantly less than birds fed the other diets. Seventy-one percent of this group died within 60 days of age. In the 774 mg/kg group, ducklings began to tremor at four weeks of age and two of the ducklings died (8%) before the scheduled necropsy at 60 days of age. By 90 days, an additional two ducklings had died among those remaining (17%). All surviving ducklings continued to display tremors throughout the 90-day experimental period. No adverse effects were reported in the 176 mg/kg dose group. Sample *et al*. (1996) used this study to derive TRVs. However, Sample *et al*. (1996) considered 1,069 mg/kg as the LOAEL. Due to the significant mortality, the SARA Group chose to use the dietary concentration of 774 mg/kg as the LOAEL and converted it to a dose of 77 mg/kg/d for use as the TRV for loon and mallard.

*Mammals*

Reduced offspring body weights were reported in rats fed 1,000 mg Ni/kg over two generations (Ambrose *et al.*, 1976). No adverse effects were reported at dietary concentrations of 250 or 500 mg Ni/kg. The dietary concentration of 1,000 mg Ni/kg was selected as a chronic LOAEL. Sample *et al.* (1996) converted this value to a dose of 50 mg/kg bw/d, assuming a body weight of 0.35 kg and a food ingestion rate of 0.0175 kg/d. This dose was divided by an uncertainty factor of three to derive the TRV for mink.

**Selenium**

*Birds*

For those bird species that have been the focus of selenium toxicity studies, reproduction appears to be the most sensitive endpoint, followed by growth, and then mortality. Of the six species for which toxicity data were available, it was possible to obtain LOAEL doses for reproductive effects for four: American kestrel (*Falco sparverius*), chicken (*Gallus gallus*), mallard (*Anas platyrhynchos*) and screech owl (*Otus asio*). Results were relatively consistent across species, with a single, most sensitive, chronic LOAEL for reproduction of 0.4 mg/kg bw/d applying to American kestrel, chicken and mallard. Thus the the chronic LOAEL of 0.4 mg Se/kg bw/d for hatch of fertile eggs (Heinz *et al.*, 1989) and hatching success (Stanley *et al*., 1996) was selected as the TRV for loon and mallard.

*Mammals*

The most sensitive study describing the toxic effects of selenium to small mammals was a multiple generation study involving female Wistar rats exposed to potassium selenate in drinking water for one year, encompassing two generations (Rosenfeld and Beath, 1954). Rosenfeld and Beath (1954) reported that $2^{nd}$ generation progeny were reduced by 50% relative to controls in the 0.33 mg/kg bw/d treatment while no effect on mortality was observed in the 0.2 mg/kg bw/d treatment. The TRV for mink of 0.1 mg/kg/d is based upon the LOAEL of 0.33 mg/kg/d divided by an uncertainty factor of three.

### 5.15.4 Field Information on Reproductive Success and Population Trends

**Mink**

There is little information available for mink in the Sudbury area. Mink are not managed by the OMNR and there are no hunting quotas placed on them; however, trappers have consistently reported good harvests throughout Sudbury (Biscaia, 2005 pers. comm.). Dobbyn *et al*. (1994) report that mink are common in the Sudbury area.

The ability of a habitat to support a mink population is limited by den availability (Allen, 1984). Several dens are used within a mink's home range for concealment, shelter, and litter rearing (Allen, 1984). The most commonly used dens are located in cavities beneath tree roots at the water's edge, with those located above the water line used preferentially (Allen, 1984).

**Waterfowl (Common Loons and Mallards)**

There are approximately 300 species of birds known to occur in Sudbury (Whitelaw, 1989) of which 183 breed in the City of Greater Sudbury (Monet and Boucher, 2005). Common waterfowl that breed in the Sudbury area are the common loon, common merganser (*Mergus merganser*), common goldeneye (*Bucephala clangula*), hooded merganser (*Lophodytes cucullatus*), American black duck (*Anas rubripes*), ring-necked duck (*Aythya collaris*), mallard (*Anas platyrhynchos*), and the wood duck (*Aix sponsa*) (McNicol *et al*., 1995). McNicol *et al*. (1995) stated that the breeding success of water birds in the Sudbury area appears to be recovering.

Weeber *et al.* (2004) reported data from extensive surveys of waterbird pairs in the Muskoka (1988 to 2002), Sudbury (1993 to 2002) and Algoma (1989 to 2001) regions of Ontario. The authors demonstrated a statistically significant increase (~9%) in pair counts of dabbling ducks on highly acidified (pH<5.3) lakes in the Sudbury area. A non-statistically significant increase (~7%) was also observed in lakes of pH

5.3 to 6.0, while a non-significant decrease (~1%) was observed in lakes of pH greater than 6.0. An increase in pair counts of diving water birds was observed in Sudbury lakes of all pH classes (~4%) (statistically significant for pH < 5.3 and pH 5.3 to 6.0). A statistically-significant increase in pair counts of piscivorous birds was observed in Sudbury lakes of all pH classes (pH <5.3: ~12%; pH 5.3 to 6.0: ~3%; pH >6.0 ~5%) (Weeber *et al*., 2004). Increases in pair counts of piscivorous birds in the Sudbury area are similar to, or in excess of in the case of highly acidified lakes, the average Ontario annual increase in common loon counts of 4.7% reported by Burgess and Collins (2004 per. comm.).

**Common Loon**

Smaller lakes, between five and 50 ha, can accommodate only a single pair of loons. Larger lakes may accommodate several pairs of loons, with each pair occupying a bay or section of the lake (Ashenden, 1994). Loons generally build their nests on the ground close to lakes, ponds or rivers (CWS, 2002), and, if possible, will build nests so that they are completely surrounded by water, such as on an island, muskrat house, half-submerged log, or on a clump of grass-like water plants (Ashenden, 1994). Loons will use whatever material is available to build their nests, sometimes even using clumps of mud and vegetation from the lake bottom (Ashenden, 1994). If no plant materials are available, loons will lay their eggs directly on mud or rock (Ashenden, 1994).

A pair of loons usually only produces one brood per season, with a maximum of two eggs in the clutch (CWS, 2002). The eggs are usually laid in June with the chicks appearing later in the month (Ashenden, 1994). The most dangerous predators to loon chicks are large carnivorous fish, snapping turtles, gulls, eagles, and crows (Ashenden, 1994). The life expectancy of the loon is 15 to 30 years (Ashenden, 1994).

Loons abandon nesting areas due to several factors that destroy breeding habitat (Ashenden, 1994), such as:

- Lakeshore development;
- Spills of oil and other pollutants;
- Physical interferences with the nests or young; and
- Increased boat wakes on lakes.

There is evidence of breeding loons in the Sudbury area for the period of 1980 to 1985 (Cadman *et al*., 1987). The breeding density (measured by pairs per 100 $km^2$) of loons in the Sudbury area increased between 1985 and 1989 (McNicol *et al*., 1995). Loon breeding density in Sudbury correlated strongly with open water area and pH. Where pH was greater than 5.5, loons were abundant (Lake Onaping area, Lady Evelyn area and Lake Panache area) (McNicol *et al.*, 1995). Persistently low pH and unsuitable nesting habitat in close proximity to Sudbury restricted the recruitment of loons to these areas. However, an increase in recruitment close to Sudbury was observed in the 1990s (McNicol *et al.*, 1995). Between 1993 and 2002, an 11.7% annual increase in pair counts of pisciverous birds (Common Loon and Common Merganser) was observed in Sudbury for lakes of pH <5.3 (Weeber *et al.*, 2004) (Figure 5-7). Lakes of pH 5.3 to 6.0 and > pH 6.0 also experienced increases in piscivorous birds (Figure 5-7) (Weeber *et al.*, 2004). The data may suggest that increased numbers of pairs are using lake habitats that were previously unoccupied (Weeber *et al.*, 2004). In the case of the Sudbury area, the increase in piscivorous pairs on highly acidified lakes may indicate a response to habitat quality improvements (Weeber *et al.*, 2004). Increases in pair counts in the Sudbury area are similar to, or in excess of the case of highly acidified lakes, the average Ontario annual increase of 4.7% reported by Burgess and Collins, 2004 pers. comm.

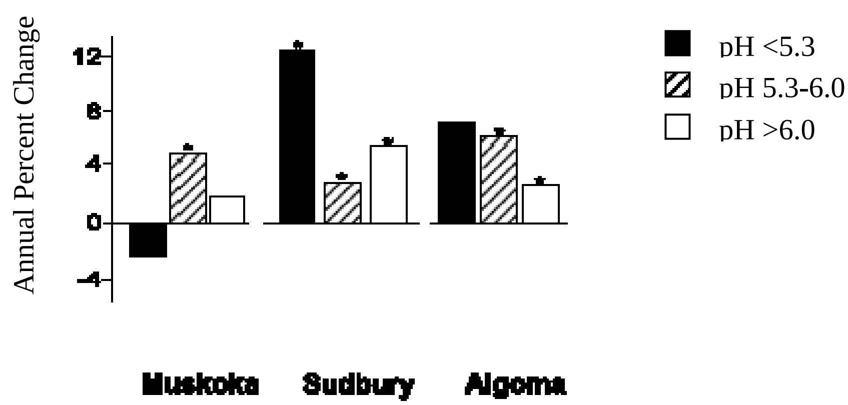



**Figure 5-7 Average Annual Percent Change in the Number of Breeding Pairs of Piscivores (Common Merganser and Common Loon) Observed on Lakes in the Muskoka (1988-2002), Sudbury (1993-2002) and Algoma (1989-2001) Regions of ON (Weeber *et al.*, 2004). Trends are Shown by Region and are Classified According to Three Lake pH Classes (pH < 5.3 (solid), pH 5.3 – 6.0 (hatched), pH > 6.0 (open)). Asterisks Indicate Statistically Significant Trends (Weeber *et al.*, 2004)**

Over the past 20 years (1985 to 2005), 59 lakes in Greater Sudbury have been monitored off and on by lake residents and community volunteers to track the breeding success and chick survival of loons (Schoenefeld, 2005 pers.comm.). Preliminary data suggest that the common loon was breeding in 37 areas (10 by 10 km areas) in the City of Greater Sudbury in 1980 to 1985 and in 35 areas in 2000 to 2005 (Monet and Boucher, 2005). The significance of this change is unknown. In 2003, 13 lakes were monitored to track the arrival of adult loon pairs and survival of loon chicks. In 2003, loon chicks were observed on Daisy Lake in Sudbury for the first time in 20 years; these chicks survived to migration (Schoenefeld, 2005 pers.comm.).

Environment Canada (EC) has been assessing the impact of acid rain for the last 25 years. A general increase in the number of breeding common loons was observed in much of southeastern Canada (*i.e.* Ontario, Quebec and Newfoundland) (Environment Canada, 2004). Observations of breeding common loons in the Sudbury region suggest that these increases may be related to improved habitat conditions in previously damaged lakes (Environment Canada, 2004).

**Mallard**

Mallard population density is positively correlated with availability of terrestrial cover for nesting and the availability of wetlands and ponds that provide the mallard's aquatic diet (U.S. EPA, 1993). Mallard nests are usually found within half a kilometre of water, however this is not a requirement and nests are occasionally found at some distance from a water body (U.S. EPA, 1993; Lister, 1996). Nests are built on the ground and are little more than a depression lined with bits of rushes, grass, or weeds (Lister, 1996).

The preferred areas for nesting contain dense grassy vegetation that is at least one metre high because it serves as good cover (U.S. EPA, 1993).

Nest success is an important factor in determining mallard populations (U.S. EPA, 1993). Mammalian predation, followed by human disturbance are the primary causes of nest failure (U.S. EPA, 1993). If their nests are destroyed, mallards may re-nest up to three or four times with each successive nest having fewer eggs. Mallards do not raise more than a single brood of ducklings each year (Lister, 1996). Juvenile survival strongly depends on food and habitat availability (U.S. EPA, 1993).

The mallard is a common breeding bird and the most abundant duck species in the Sudbury area (McNicol *et al*., 1995). Sudbury populations of the mallard are stable (McNicol *et al*., 1995). Weeber *et al.* (2004) reported data from extensive surveys of waterbird pairs in the Muskoka, Sudbury and Algoma regions of Ontario showing a statistically significant increase (~9%) from 1993 to 2002 in pair counts of dabbling ducks on highly acidified (pH<5.3) lakes in the Sudbury area (Figure 5-8). A non-statistically significant increase (~7%) was also observed in lakes of pH 5.3 to 6.0, while a non-significant decrease (~1%) was observed in lakes of pH greater than 6.0. Although the authors of the study do not explain the slight decrease in dabbling ducks for lakes of pH > 6.0, they do comment that the Sudbury data are not consistent with the data (1985 to 1989) presented in McNicol *et al*. (1995) for slightly- or non-acidic lakes (Weeber *et al.*, 2004).

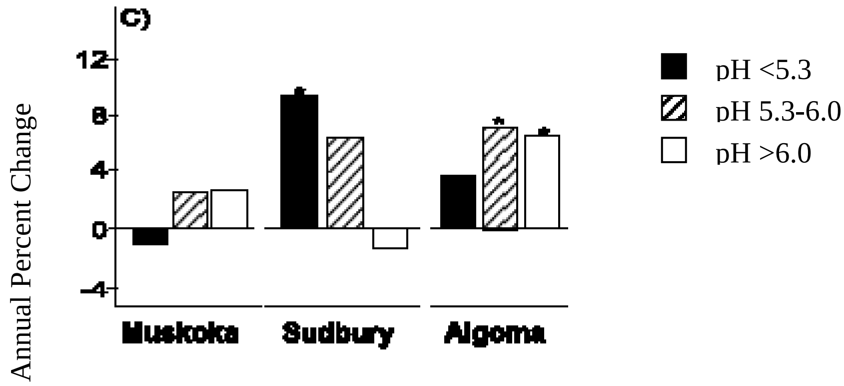



**Figure 5-8 Average Annual Percent Change in the Number of Breeding Pairs of Dabblers (Mallard, American Black Duck and Wood Duck) Observed on Lakes in the Muskoka (1988-2002), Sudbury (1993-2002) and Algoma (1989-2001) Regions of ON. Trends are Shown by Region and are Classified According to Three Lake pH Classes (pH < 5.3 (solid), pH 5.3 – 6.0 (hatched), pH > 6.0 (open)). Asterisks Indicate Statistically Significant Trends (Weeber *et al*., 2004)**

Weeber *et al.* (2004) note that it is currently unclear whether food resources and other habitat qualities have improved such that more breeding pairs of water birds are supported now relative to 1985-1989. They also note that the data suggest that the birds are using lake habitats that were previously unoccupied (Weeber *et al.*, 2004). Preliminary data suggest that mallards were breeding in 39 areas (10 by 10 kilometre areas) in the City of Greater Sudbury in 1980 to 1985 and in 35 areas in 2000 to 2005 (Monet and Boucher, 2005). The significance of this change is unknown.

## 5.16 Risk Characterization

Risks were characterized by estimating the probability that wildlife exposures exceeded the TRV. These modelling results were then put into context based on what is known about populations in the Sudbury area (Section 5.15).

Risks were estimated for four separate areas within the Study Area (Tables 5.23 through 5.26) that contain significant aquatic habitat: Zone 1, Zone 2, Zone 3 (as identified in Chapter 2) and Sudbury-centre. In all cases, unacceptable risks are not predicted for mink, loon or mallard exposed to arsenic, cadmium, cobalt, copper, lead or nickel in any portion of the study area. However, there was a greater than 90% probability that selenium exposures to mink, loon and mallard exceeded TRVs in all portions of the study area. The 90$^{th}$ percentile ERs were 110, 6.7 and 23 for mink, loon and mallard, respectively, in all areas.

**Table 5.23 Probability of ER > 1.0 for Zone 1**

| VEC | Arsenic | Cadmium | Cobalt | Copper | Lead | Nickel | Selenium |
|---|---|---|---|---|---|---|---|
| Mink | 0 | 0 | 0 | 0 | 0 | 1 | **100** |
| Loon | 0 | 0 | 0 | 0 | 0 | 0 | **88** |
| Mallard | 0 | 0 | 0 | 0 | 0 | 0 | **96** |

Values in bold and highlighted indicate probability is > 10% that ER>1.0

**Table 5.24 Probability of ER > 1.0 for Zone 2**

| VEC | Arsenic | Cadmium | Cobalt | Copper | Lead | Nickel | Selenium |
|---|---|---|---|---|---|---|---|
| Mink | 0 | 0 | 0 | 0 | 0 | 1 | **100** |
| Loon | 0 | 0 | 0 | 0 | 0 | 0 | **91** |
| Mallard | 0 | 0 | 0 | 0 | 0 | 0 | **99** |

Values in bold and highlighted indicate probability is > 10% that ER>1.0

**Table 5.25 Probability of ER > 1.0 for Zone 3**

| VEC | Arsenic | Cadmium | Cobalt | Copper | Lead | Nickel | Selenium |
|---|---|---|---|---|---|---|---|
| Mink | 0 | 0 | 0 | 0 | 0 | 1 | **100** |
| Loon | 0 | 0 | 0 | 0 | 0 | 0 | **91** |
| Mallard | 0 | 0 | 0 | 0 | 0 | 0 | **99** |

Values in bold and highlighted indicate probability is > 10% that ER>1.0

**Table 5.26 Probability of ER > 1.0 for Sudbury-central**

| VEC | Arsenic | Cadmium | Cobalt | Copper | Lead | Nickel | Selenium |
|---|---|---|---|---|---|---|---|
| Mink | 0 | 0 | 0 | 0 | 0 | 1 | **100** |
| Loon | 0 | 0 | 0 | 0 | 0 | 0 | **91** |
| Mallard | 0 | 0 | 0 | 0 | 0 | 0 | **99** |

Values in bold and highlighted indicate probability is > 10% that ER>1.0

Risks to mink, loon and mallard result predominantly from exposure to benthic invertebrates. There is considerable uncertainty surrounding these risk estimates. Selenium was not measured in benthic invertebrates, but was modelled based on selenium concentrations in sediment. Selenium was measured in single samples taken from only eight lakes in the study area. In addition, little information is available for uptake factors for selenium from sediment to benthic invertebrates. Therefore, there is low confidence in the results of the risk modelling for mink, loon and mallard from exposure to selenium.

Mink are not managed by the Ontario Ministry of Natural Resources and there are no hunting quotas placed on them; however, trappers have consistently reported good harvests throughout Sudbury (Biscaia, 2005 pers. comm.). Mink, marten and fisher each comprised 4 to 8 % of the fur harvest in 1997-98 and 1998-99 (Robitaille, 2005 pers. comm.). Dobbyn *et al*. (1994) report that mink are common in the Sudbury area.

Between 1993 and 2002, an 11.7% annual increase in pair counts of piscivorous birds (Common Loon and Common Merganser) was observed in Sudbury for lakes of pH <5.3 (Weeber *et al*., 2004). Lakes of pH 5.3 to 6.0 and > pH 6.0 also experienced increases in piscivorous birds (Weeber *et al*., 2004). The data may suggest that increased numbers of pairs are using lake habitats that were previously unoccupied (Weeber *et al*., 2004). In the case of the Sudbury area, the increase in piscivorous pairs on highly acidified lakes may indicate a response to habitat quality improvements (Environment Canada, 2004; Weeber *et al.*, 2004). In 2003, 13 lakes were monitored to track the arrival of adult loon pairs and survival of loon chicks. Loon chicks were observed on Daisy Lake in Sudbury for the first time in 20 years; these chicks survived to migration (Schoenefeld, 2005 pers.comm.).

The mallard is a common breeding bird and the most abundant duck in the Sudbury area (McNicol *et al*., 1995). Weeber *et al.* (2004) reported data from extensive surveys of waterbird pairs showing a statistically significant increase (~9%) from 1993 to 2002 in pair counts of dabbling ducks on highly acidified (pH<5.3) lakes in the Sudbury area. There were no statistically significant changes for lakes of pH 5.3 to 6.0 or pH > 6.0. Weeber *et al.* (2004) also note that the data suggest that the birds are using lake habitats that were previously unoccupied.

These observations for mink, loon and mallards suggest that these animals are common in the Sudbury area. In fact, populations of piscivorous and benthivorous birds and mammals may be increasing in size, possibly due to habitat improvements. Breeding success and abundance data have been collected for birds for the period from 2001 to 2005; the results will not be published until 2007. It is recommended that those results be reviewed, when available, relative to the information available within this ERA.

## 5.17 Uncertainties and Data Gaps

The goals of any future ERA for aquatic life should determine the scope of the assessment. These will also assist with the delineation of the study area for an aquatic ERA. The numerous studies and long-term monitoring programs conducted by researchers in the Sudbury area will provide important ecological data that may be integrated into the detailed aquatic ERA. These studies have linked lake water pH to species abundance and community composition, dealing with fish, invertebrates, plants, and algae. The results from these programs can be used to help focus research efforts by illustrating the long-term trends in monitoring data, identifying which lakes have been significantly affected by acidification and/or metals, and which lakes may be disregarded from further consideration.

Data gaps for many COC and VECs in Sudbury lakes have been identified in this aquatic problem formulation. Although concentrations of metals in water of Sudbury lakes are well documented in the scientific literature, few recent data for sediment or biota are available and further efforts must be made to find a direct association between these concentrations and population effects for many groups of organisms. Those studies that have related metal contamination to populations have dealt primarily with yellow perch. There is a definite lack of information on Sudbury-specific metal impacts on other fish species, invertebrates, amphibians, macrophytes, and algae. Marshes and wetlands are not well studied, and existing data are more than 20 years old. From the information available, it is difficult to determine if metals are having a significant deleterious effect on these populations directly or through reductions in food or habitat quality.

Risk modelling predicted unacceptable risks from selenium exposure for mink, loon and mallards in all areas in which they were assessed. However, the same water and sediment dataset was used for all zones/areas, and included only a single selenium analysis for sediment from each of eight lakes. Information from local naturalists suggests that these animals are common in the Sudbury area. In fact, populations of piscivorous and benthivorous birds and mammals may be increasing in size, possibly due to habitat improvements. Breeding success and abundance data have been collected for birds for the period from 2001 to 2005; the results will not be published until 2007. It is recommended that those results be reviewed, when available, relative to the information within this ERA. Potential risks to these species are related to sediment concentrations of selenium, and uptake of selenium into benthic invertebrates. Uncertainties related to risk modelling for wildlife are detailed in Chapter 4 of this report.

There are numerous uncertainties and data gaps that should to be filled before a comprehensive aquatic ERA could be completed. It is recommended that any future aquatic ERA consider the following data gaps and methods to fill them. However, it is anticipated that the list of studies required to complete the aquatic ERA may differ from this suggested list, based on future discussions between stakeholders. In addition, it is recommended that MOE and other stakeholders be contacted regarding data and information they may have which is unpublished but which may be useful to future aquatic studies. Potential data needs could include:

- Comprehensive water chemistry, including but not limited to parameters such as pH and hardness, metal concentrations (for all COC and for metals not retained as COC, but for which data are limited [*e.g*., antimony, selenium]), for each lake selected for in-depth study;

- Comprehensive sediment chemistry, including but not limited to parameters such as organic carbon content, acid volatile sulfide, and sediment texture, metal concentrations (for all COC, and for metals not retained as COC, but for which data are limited [*e.g.*, antimony, magnesium, mercury]) for each lake selected for in-depth study;
- Background sediment chemistry data;
- Sequential extraction analysis of sediments to evaluate bioavailability;
- Chemical and biological data for marshes and wetlands in the Sudbury area, since data from the 1980s suggest potential impacts on wetlands near the smelters which should be investigated;
- Biological or ecological data for the fish species identified as VECs (particularly common shiner, white sucker and walleye) in the lakes of interest, since most data are for perch in Hannah and Ramsey Lakes;
- Metal accumulation data for additional whole fish species of potential interest;
- Effects of the mixture of metals (and other influences, such as pH) on various aquatic VECs could be assessed by using laboratory- or field-based toxicity tests or field population or community survey techniques;
- Benthic invertebrate community data for all lakes which may be part of an aquatic ERA; laboratory bioassays and measurements of uptake from sediment could be considered, using sediment from lakes of interest;
- Data are available for zooplankton communities in many lakes, particularly Middle and Hannah Lakes. Community metrics and laboratory bioassays should be considered for inclusion in the aquatic ERA for other lakes of interest;
- Little recent data are available for algal or macrophyte communities in the Sudbury area; surveys can be done for lakes of interest, and laboratory algal bioassays could be considered to evaluate water quality on particular algal species; and
- Few data are available for amphibians; consideration may be given to conducting amphibian call surveys to evaluate populations in the Sudbury area.

Lakes which have been studied in the past, and which may be considered for inclusion in a future aquatic ERA, include:

- Clearwater Lake;
- Hannah and Middle Lakes;
- McFarlane Lake;
- Ramsey Lake; and
- Silver Lake.

However, depending on the goals of the aquatic ERA, several other lakes could be considered for inclusion in the ERA, particularly since the above-mentioned lakes are located only in the Ramsey and Panache watersheds.

## 5.18 References

Abo-Rady, M. 1979. Lebensm. Unters. Forsch. 168: 259. Cited In: Bradley and Morris, 1986.

Adamus, P.R., and Brandt, K. 1990. Impacts on quality of inland wetlands of the United States: A survey of indicators, techniques, and applications of community level biomonitoring data. Corvallis OR: U.S. Environmental Protection Agency, Environmental Research Laboratory. EPA/600/3-90/073.

Allen, A.W. 1984. Habitat suitability index models: Mink. U.S. Fish Wildl. Serv. FWS/OBS-82/10.61 Revised. 19pp.

Ambrose, A.M., Larson, P.S., Borzelleca, J.F., and Hennigar, G.R. 1976. Long-term Toxicologic Assessment of Nickel in Rats and Dogs. J Food Sci Technol 13:181-187.

Arafat, N., and Nriagu, J.O. 1986. pH induced release of metals from polluted lake sediments. Water, Air, Soil Pollut. 31: 991- 998. Cited In: Nriagu and Rao, 1987.

Ashenden, J. 1994. Bird Fact Sheet – Loon. Hinterland's Who's Who. Environment Canada. CW69-4/4-2003E-HTML. http://www.hww.ca/hww2p.asp?id=53&cid=0.

ATSDR. 1999. Toxicological profile for lead. Agency for Toxic Substances and Disease Registry (ATSDR), Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service.

Audet, D., and P. Couture. 2003. Seasonal variations in tissue metabolic capacities of yellow perch (perca flavescens) from clean and metal-contaminated environments. Canadian Journal of Fisheries and Aquatic Sciences. 60:269-278.

Aulerich, R.J., Ringer, R.K., Bleavins, M.R., and Napolitano, A. 1982. Effects of supplemental dietary copper on growth, reproductive performance and kit survival of standard mink and the acute toxicity of copper to mink. J Animal Science 55(2):337-343.

Azar, A., Trochimowicz, H.J., Maxfield, M.E. 1973. Review of lead studies in animals carried out at Haskell Laboratory: Two year feeding study and response to hemorrhage study. In: Barth D, Berlin A, Engel R, *et al*., eds. Environmental health aspects of lead: Proceedings, International Symposium, October 1972, Amsterdam, The Netherlands. Luxembourg: Commission of the European Communities, 199-210. Cited In: ATSDR, 1999.

Bagatto, G., and Alikhan, M.A. 1987a. Copper, cadmium and nickel accumulation in crayfish populations near copper-nickel smelters at Sudbury, Ontario, Canada. Bulletin of Environmental Contamination and Toxicology 38:540-545

Bagatto, G., and Alikhan, M.A. 1987b. Zinc, iron and manganese accumulation in crayfish populations near copper-nickel smelters at Sudbury, Ontario, Canada. Bulletin of Environmental Contamination and Toxicology 38:1076-1081

BC MELP. 1998a. B.C. Ministry of Environment, Lands and Parks. British Columbia Approved Water Quality Guidelines (Criteria). Water Management Branch, Environment and Resource Management Department, Ministry of Environment, Lands and Parks. Updated January 17, 2001.

BC MELP. 1998b. B.C. Ministry of Environment, Lands and Parks. A Compendium of Working Water Quality Guidelines for British Columbia. Water Management Branch, Environment and Resource Management Department, Ministry of Environment, Lands and Parks. Updated January 17, 2001.

Beamish, R.J., and H.H. Harvey. 1972. Acidification of the La Cloche Mountain lakes, Ontario, and resulting fish mortalities. J. Fish. Res. Bd. Canada 29:1131-1143.

Becker, J., Brandt, C., and Dauble, D. 1998. Species selection for an ecological risk assessment of the Columbia River at the Hanford site, Washington, USA. Environmental Toxicology and Chemistry 17(11): 2354-2357.

Beebee. T.J.C., and Griffin, J.R. 1977. A preliminary investigation into natterjack toad (*Bufo calamita*) breeding site characteristics in Great Britain. J. Zool. 181:341-350. Cited In: Glooschenko *et al.*, 1992.

Beggs, G.L., Gunn, J.M., and Olver, C.H. 1985. Ont. Fish. Res. Board Can. 34: 649. Cited In: Bowlby *et al.*, 1988.

Beiswenger, R.E. 1988. Integrating anuran amphibian species into environmental assessment programs. In: Szaro RC, Severson KE, Patton DR, technical coordinators. Management of amphibians, reptiles, and small mammals in North America. Fort Collins CO: USDA Forest Service. General Technical Report RM-166.

Benoit, D.A., Leonard, E.N., Christensen, G.M., and Fiandt, J.T. 1976. Trans. Am. Fish. Soc. 105: 550. Cited In: Bradley and Morris, 1986.

Best. 2001. A Risk Management Strategy for PCB –Contaminated Sediments. Committee on Remediation of PCB-Contaminated Sediments. Board on Environmental Studies and Toxicology, National Research Council.

Biesinger, K.E., Christensen, G.M. 1972. Effects of various metals on survival, growth, reproduction, and metabolism of *Daphnia magna*. J. Fish. Res. Board Can. 29: 1691- 1700.

Birge, W.J., J.A. Black, and A.G. Westerman. 1979. Evaluation of Aquatic Pollutants Using Fish and Amphibian Eggs as Bioassay Organisms. In: S.W. Nielsen, G. Migaki, and D.G. Scarpelli (Eds.), Symp. Animals Monitors Environ. Pollut., 1977, Storrs, CT 12:108-118. Cited In: U.S. EPA, 2005.

Biscaia, P. 2005. Personal Communication to Cantox Environmental. Acting/Area Biologist, Sudbury District, Ministry of Natural Resources.

Blomme, C.G. and Laws, K.M. 1995. Partnerships for wildlife restoration: Peregrine falcons. In: Gunn, J.M.(Ed.) Restoration and Recovery of an Industrial Region. Ontario Ministry of Natural Resources, Cooperative Freshwater Ecology Unit, Laurentian University, Sudbury, Ontario, Canada. Spring-Verlag, New York.p.155-168.

Bolger, P.M. 1980. Ecological effects of liquid mining effluents on the Onaping River System in Ontario. Thesis, Laurentian University, Sudbury, Ontario. Cited In: Bradley and Morris, 1986.

Bonin, J., M. Ouellet, J. Rodrigue and Jean-Luc DesGranges. 1997. Measuring the Health of Frogs in Agricultural Habitats Subjected to Pesticides. ). In Amphibians In Decline, Canadian Studies of a Global Problem, D.M. Green (eds.). pp 246-257.

Borgmann, U., and Munawar, M. 1989. A new standardized sediment bioassay protocol using the amphipod *Hyalella azteca* (Saussure). Hydrobiologia, 188/189: 425-431.

Borgmann, U., and Norwood, W.P. 1997. Toxicity and accumulation of zinc and copper in *Hyalella azteca* exposed to metal-spiked sediments. Can. J. Fish. Aquat. Sci. 54: 1046-1054.

Borgmann, U., and Norwood, W.P. 2002. Metal bioavailability and toxicity through a sediment core. Environ. Pollut. 116: 159-168.

Borgmann, U., Jackson, T.A., Reynoldson, T.B., and Rosa, F. 1998. Interim report on the effects of atmospheric deposition of metals from the Sudbury smelters on aquatic benthic ecosystems. National Water Research Institute Rep. No. 98- 230. Burlington, Ontario, Canada. 148 pp.

Borgmann, U., Norwood, W.P., Reynoldson, T.B., and Rosa, F. 2001a. Identifying cause in sediment assessments: bioavailability and the Sediment Quality Triad. Can. J. Fish. Aquat. Sci 58: 950-960.

Borgmann, U., Reynoldson, T.B., Rosa, F. and Norwood, W.P. 2001b. Final Report on the effects of atmospheric deposition of metals from the Sudbury smelters on aquatic ecosystems. Part I: Major ions, nutrients, total metals, benthos, and toxicity and bioavailability of metals to benthic invertebrates. National Water Research Institute, Environment Canada, Burlington, Ontario, Canada, NWRI Contribution No. 01-023.

Bousfield, E.L. 1958. Fresh-water amphipod crustaceans of glaciated North America. Canadian Field-Naturalist 72:55-113. Cited In: Borgmann and Norwood, 1997.

Bowlby, J.M., Gunn, J.M. and Liimatainen, V.A. 1988. Metals in stocked lake Trout (*Salvelinus namaycush*) in lakes near Sudbury, Canada. Water, Air and Soil Pollution 39:217-230.

Bradley, R.W., and Morris, J.R. 1986. Heavy Metals in Fish from a Series of Metal-Contaminated Lakes near Sudbury, Ontario. Water, Air, and Soil Pollution 27: 341- 354.

Breitburg, D.L., Sanders, J.G., Gilmour, C.C., Hatfield, C.A., Osman, R.W., Riedel, G.F., Seitzinger, S.B., and Sellner, K.G. 1999. Variability in responses to nutrients and trace elements, and transmission of stressor effects through an estuarine food web. Limnol. Oceanogr. 44:837-863. Cited In: Fleeger *et al*., 2003.

Brix, K.V., DeForest, D.K., Burger, M., and Adams, W.J. 2005. Assessing the relative sensitivity of aquatic organisms to divalent metals and their representation in toxicity datasets compared to natural communities. Hum. Ecol. Risk Assess. 11:1139-1156.

Brooks, J.L., 1957. The systematics of North American *Daphnia*. Mem. Conn. Acad. Arts Lett. 13: 1-180. Cited In: Yan *et al*., 1996c.

Burgess, N. and Collins, B.T. 2004. Personal communication from Burgess and Collins, Canadian Wildlife Service, Environment Canada. Cited In: Weeber *et al*., 2004.

Byron, W.R., Bierbower, G.W., Brouwer, J.B., and Hansen, W.H. 1967. Pathologic changes in rats and dogs from two-year feeding of sodium arsenite or sodium arsenate. Toxicol. Appl. Pharmacol. 10:132-147.

Cadman, M. D. Eagles, P. F. J. and Helleiner, F. M. 1987. Atlas of the Breeding Birds of Ontario. Federation of Ontario Naturalists and Long Point Bird Observatory.

Cain, B.W., and Pafford, E.A. 1981. Effects of dietary nickel on survival and growth of Mallard ducklings. Arch Environ Contam Toxicol 10:737-745.

Campbell, P.G.C., Hontela, A., Rasmussen, J.B. 2003. Differentiating between direct (physiological) and food-chain mediated (bioenergetic) effects on fish in metal-impacted lakes. Human and Ecological Risk Assessment 9(4):847-866.

Carignan, R., and Nriagu, J.O. 1985. Trace metal deposition and mobility in the sediments of two lakes near Sudbury, Ontario. Geochem. Cosmochim. Acta. 49: 1753- 1764. Cited In: Nriagu *et al*., 1998.

Carter, J.C.H., Dadswell, M.J., Roff, J.C., and Sprules, W.G. 1980. Distribution and zoogeography of planktonic crustaceans and dipterans in glaciated eastern North America. Can. J. Zool. 58: 1355-1387. Cited In: Yan *et al*., 1996a.

CCME. 1996. A Framework for Ecological Risk Assessment: General Guidance. The National Contaminated Sites Remediation Program. Winnipeg, MB.

CCME. 1997. A Framework for Ecological Risk Assessment: Technical Appendices. The National Contaminated Sites Remediation Program. Winnipeg, MB.

CCME. 2002. Canadian Environmental Quality Guidelines. Canadian Council of Ministers of the Environment. Winnipeg, MB.

Chan, W.H., and Lusis, M.A. 1985. Post-Superstack Sudbury Emissions and their Fate in the Atmosphere: An Overview of the Sudbury Environment Study Results. Water, Air, Soil Pollut. 26: 43-58. Cited In: Nriagu and Rao, 1987.

Chapman, P.M., Wang, F., Janssen, C.R., Goulet, R.R., and Kamunde, C.N. 2003. Conducting ecological risk assessments of inorganic metals and metalloids: Current status. Human and Ecological Risk Assessment 9(4): 641-697.

Chappell, W.R. 1975. Transport and Biological Effects of Molybdenum in the Environment. In: Heavy Metals in the Aquatic Environment. P.A. Krenkel, ed. Pergamon Press Ltd., Oxford. Cited In: CCME, 2002.

Chen, Y.W., N. Belzile, and J.M. Gunn. 2001. Antagonistic effect of selenium on mercury assimilation by fish populations near Sudbury metal smelters? Limnol. Oceanogr., 46(7):1814-1818.

City of Greater Sudbury. 2006. Personal communication (email) between Stephen Monet (City of Greater Sudbury) and the SARA Group, regarding Hannah and Middle Lake Watersheds. Email received June 21, 2006.

Clark, K.L., and LaZerte, B.D. 1987. I ntraspecific variation in hydrogen ion and aluminium toxicity in *Bufo americanus* and *Ambystoma maculatum*. Can. J. Fish. Aquat. Sci. 44: 1622-1628. Cited In: Glooschenko *et al*., 1992.

Clements, W.H., Cherry, D.S., and Cairns, Jr. J. 1988. The impact of heavy metals on aquatic insect communities in streams: a comparison of observations and experimental results. C. J. Fish. Aquat. Sci. 11: 2017-2025. Cited In: Preston, 2002.

Clements, W.H., Cherry, D.S., and Cairns, Jr. J. 1989. The influence of copper exposure on predator-prey interactions in aquatic insect communities. Freshwater Biology 21: 483-488. Cited In: Preston, 2002.

Conroy, N.I., Hawley, K., and Keller, W. 1978. Extensive monitoring of lakes in the greater Sudbury area. 1974- 1976. Tech. Rep. Ontario Ministry of Environment. Sudbury, Ontario. Canada. 40 pp plus appendices.

Cook, F.R. 1984. Introduction to Canadian Amphibians and Reptiles. National Museums of Canada. Ottawa, ON. 200 p. Cited In: Glooschenko *et al*., 1992.

Co-Op. 2004. Co-Operative Freshwater Ecology Unit – 2004 Program. Available online: http://www.city.greatersudbury.on.ca/content/div_lakewaterquality/documents/Co-op_Unit_2004Report_Website_Format.pdf

Co-Op. Unpublished a. Co-Operative Freshwater Ecology Unit. Unpublished data for SES Extensive Study Lakes (1981-2002). Excel spreadsheet "SES Lakes Data.xls". Provided by Bill Keller. Department of Biology, Laurentian University. Sudbury, Ontario, Canada.

Co-Op. Unpublished b. Co-Operative Freshwater Ecology Unit. Unpublished data for SES Intensive Study Lakes (1984-2002). Excel spreadsheet "Intensive Annual Averages.xls". Provided by Bill Keller. Department of Biology, Laurentian University. Sudbury, Ontario, Canada.

Couture, P., and P.R. Kumar. 2003. Impairment of metabolic capacities in copper and cadmium contaminated wild yellow perch (*perca flavescens*). Aquatic Toxicology. 64(1):107-120.

Couture, P., and J.W. Rajotte. 2003. Morphometric and metabolic indicators of metal stress in wild yellow perch (*Perca flavescens*) from Sudbury, Ontario: A review. J. Environ. Monit. 5:216-221.

CWS. 2002. Canadian Wildlife Service - Ontario Region.. http://www.on.ec.gc.ca/wildlife_e.html. Last updated 12/31/2002. Accessed January 2005.

Dale, H.M., and Miller, G.E. 1978. Changes in the aquatic macrophyte flora, Whitewater Lake near Sudbury, Ontario from 1947 to 1977. Can. Field-Nat. 92: 264-270. Cited In: SARA, 2005.

Dale, J.M., Freedman, B., and Kerekes, J. 1985. Acidity and associated water chemistry of habitats in Nova Scotia. Can. J. Zool. 63: 97-105. Cited In: Glooschenko *et al*., 1992.

Dixit, A.S., Dixit, S.S., and Smol, J.P. 1992. Long-term trends in lake water pH and metal concentrations inferred from diatoms and chrysophytes in three lakes near Sudbury, Ontario. Can. J. Fish. Aquat. Sci. Vol. 49(Suppl. 1).

Dobbyn, J., Eger and Wilson, N. 1994. Atlas of the Mammals of Ontario. Federation of Ontario Naturalists ISBN 1-896059-02-3.

Dudka, S., Ponce-Hernandez, R., and Hutchinson, T.C. 1995. Current level of total element concentrations in the surface layer of Sudbury's soil. Sci. Total Environ. 162: 161- 171. Cited In: Nriagu *et al*., 1998.

Dunson, W.A., and Connell, J. 1982. Specific inhibition of hatching in amphibian embryos by low pH. J. Herpitol. 16: 314-316. Cited In: Glooschenko *et al*., 1992.

Eastwood, S. and P. Couture. 2002. Seasonal Variations in Condition and Liver Metal Concentrations of Yellow Perch (Perca flavescens) from a Metal-Contaminated Environment. Aquatic Toxicology. 58:43-56.

Edens, F.W., and Garlich, J.D. 1983. Lead-induced egg production decrease in Leghorn and Japanese quail hens. Poult Sci 62:1757-1763.

Environment Canada. 1997. Fish Survey Expert Working Group Final Report: Recommendations from Cycle 1 Review. Prepared for Environmental Effects Monitoring Technical Management Committee for the Pulp and Paper Sector. Ottawa. Report EEM/1997/6.

Environment Canada. 2004. State of the Lakes Ecosystem Conference (SOLEC) 2004 Success Story: The Junction Creek Stewardship Committee, Sudbury, Ontario. http://www.on.ec.gc.ca/press/solec_2004_junction_creek_e.html

Falk, M.R., Miller, M.D., and Kostiuk, S.J.M. 1973. Biological effects of mining wastes in the Northwest Territories. Fish. and Mar. Serv. Tech. Rep. Ser. No. CEN/T-73-10, Environment Canada. Cited In: Bradley and Morris, 1986.

Fitchko, Y. 1978. The distribution, mobility and accumulation of nickel, copper and zinc in a river system draining the eastern part of the metal-polluted Sudbury smelting area; unpublished PhD thesis, University of Toronto, Toronto, 209p. Cited In: Pearson *et al*., 2002a.

Fleeger, J.W., Carman, K.R., and Nisbet, R.M. 2003. Indirect effects of contaminants in aquatic ecosystems. Review. The Science of the Total Environment 317:207-233.

Freda, J. 1991. The effects of aluminium and other metals on amphibians. Environ. Pollut. 71: 305- 328. Cited In: Freda and McDonald, 1993.

Freda, J., and Dunson, W.A. 1985. Field and laboratory studies of ion balance and growth rates of ranid tadpoles chronically exposed to low pH. Copeia 1985: 415-423.

Freda, J., and Dunson, W.A. 1986. Effects of low pH and other chemical variables on the local distribution of amphibians. Copeia. 1986: 454-466. Cited In: Glooschenko *et al*., 1992.

Freda, J., and McDonald, D. G. 1993. Toxicity of Amphibian Breeding Ponds in the Sudbury Region. Can. J. Fish. Aquat. Sci. 50: 1497-1503.

Freda, J., Cavdek, V., and McDonald, D.G. 1990. Role of organic complexation in the toxicity of aluminium to *Rana pipiens* embryos and *Bufo americanus* tadpoles. Can. J. Aquat. Sci. 47: 217-224. Cited In: Freda and McDonald, 1993.

Freda, J., Sadinski, W.J., and Dunson, W.A. 1991. L ong term monitoring of amphibian populations with respect to the effects of acidic deposition. Water Air Soil Pollut. 55: 445- 462. Cited In: Glooschenko *et al*., 1992.

Freedman, B., and Hutchinson, T.C. 1980. Can. J. Bot. 58: 1722. Cited In: Taylor and Crowder, 1983.

Gascon, C., and Planas, D. 1986. Spring pond water chemistry and the reproduction of the wood frog (*Rana sylvatica*). Can. J. Zool. 64: 543-550. Cited In: Glooschenko *et al*., 1992.

Giguere, A., Campbell, P.G.C., Hare, L., and Cossu-Leguille, C. 2005. Metal bioaccumulation and oxidative stress in yellow perch (*Perca flavescens)* collected from eight lakes along a metal contamination gradient (Cd, Cu, Zn, Ni). Can. J. Fish. Aquat. Sci. 62: 563-577.

Glooschenko, V., Weller, W.F., Smith, P.G.R., Alvo, R., and Archbold, J.H.G. 1992. Amphibian Distribution with Respect to Pond Water Chemistry near Sudbury, Ontario. Can. J. Fish. Aquat. Sci. vol. 49(Suppl. 1): 114-121.

Grzela, T.K., and Wright, P. 2003. Aquatic Organisms of the City of Greater Sudbury Area Lakes. Available online: http://www.city.greatersudbury.on.ca/cms/index.cfm?app=div_lakewaterquality&lang=en&currID=683&parID=0

Gunn, J.M. 1982. Acidification of lake trout (*Salvelinus namaycush*) lakes near Sudbury Ontario. In Acid Rain/ Fisheries, Proceedings of the International Symposium on Acidic Precipitation and Fisheries Impacts in Northeastern North America, Cornell University, Ithaca. 351 pp. Cited In: Bowlby *et al*., 1988.

Gunn, J.M., and Keller, W. 1995. Urban lakes: Integrators of environmental damage and recovery. In: Gunn, J.M.(Ed.) Restoration and Recovery of an Industrial Region. Ontario Ministry of Natural Resources, Cooperative Freshwater Ecology Unit, Laurentian University, Sudbury, Ontario, Canada. Spring-Verlag, New York. p.257-270.

Fisheries and Oceans Canada. 2006. Fish and Aquatic life, Underwater World, Lake trout. http://www.dfo-mpo.gc.ca/zone/underwater_sous-marin/touladi/laketrout-touladi_e.htm Accessed June 28, 2006

Harvey, H.H. 1975. Fish populations in a large group of acid-stressed lakes. Verh. Int. Ver. Limnol. 19:2406-2417. Cited In: Dixit *et al*., 1992.

Havens, K.E. 1992. Acid and aluminium effects on sodium homeostasis and survival of acid-sensitive and acid-tolerant Cladocera. Can. J. Fish. Aquat. Sci. 49: 2392- 2398. Cited In: Yan *et al*., 1996a.

Hebert, C.E., Bishop, C.A., Weseloh, D.V. 1993. Evaluation of wetland biomonitors for the Great Lakes: a review of contaminant levels and effects in five vertebrate classes. Ottawa, Canada: Canadian Wildlife Service, Ontario Region. Technical Report Ser 182.

Heinz, G.H., Hoffman, D.J., and Gold, L.G. 1989. Impaired reproduction of mallards fed an organic form of selenium. J Wildl Mgmt 53:418-428.

Heneberry, J.H. 1997. Absence of large grazers: an obstacle to recovery in acid and metal damaged lakes. M.Sc. Thesis, Laurentian University, Sudbury, Ontario, Canada. 108 pp. Cited In: Keller *et al*., 1999.

Hermayer, K.L., Stake, P.E., and Shippe, R.L. 1977. Evaluation of dietary zinc, cadmium, tin, lead, bismuth and arsenic toxicity in hens. Poult. Sci. 56:1721.

Holocombe, G.W., Benoit, D.A., and Leonard, E.N. 1979. Trans. Am. Fish. Soc. 108: 76. Cited In: Bradley and Morris, 1986.

Hutchinson, T.C. 1973. Comparative studies of the phytotoxicity of heavy metals to phytoplankton and their synergistic interactions. Water Pollution Res. Can., Proc. 8th Can. Symp. Cited In: Stokes *et al*., 1973.

Hutchinson, T.C., and P.M. Stokes. 1973. Heavy Metal Toxicity and Algal Bioassays. Paper given at Water Quality Parameters Conference, Chemical Institute of Canada, Canada Centre for Inland Waters. November 19-21, 1973.

Iles, A.C. 2003. Indirect effects of metal contamination on energetics of yellow perch (Perca flavescens)in Sudbury area lakes, resulting from food web simplification. M.Sc. Thesis, McGill University, Montreal, QC. Cited In: Keller *et al*., 2004.

Ingersoll, C.G., and Winner, R.W. 1982. Effect on *Daphnia pulex* (de Geer) of daily pulse exposures to copper or cadmium. Environ. Toxicol. Chem. 1: 321- 327. Cited In: Yan *et al*., 1996a.

Jaagumagi, R., and Bedard, D. 2002. Junction Creek System (Sudbury) Environmental Monitoring Study, September 1999. Ontario Ministry of the Environment, Northern Region, Sudbury District Office.

Jarvinen, A.W., and Ankley, G.T. 1999. Linkage of Effects to Tissue Residues: Development of a Comprehensive Database for Aquatic Organisms Exposed to Inorganic and Organic Chemicals. SETAC Technical Publications Series. Society of Environmental Toxicology and Chemistry.

Jones, D.S., G.W. Suter II, and R. N. Hull. 1997. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision. Oak Ridge National Laboratory. DE-AC05-84OR21400.

Karns, D.R. 1983. The Role of Low pH in the Toxicity of Bog Water to Amphibian Embryos. Bull. Ecol. Soc. Am. 64: 188. Cited In: Glooschenko *et al*., 1992.

Keller, W. 2005. Personal communication with Chris Wren of the SARA Group regarding neutralized lakes. March 24, 2005.

Keller, W. and J.R. Pitblado. 1984. Crustacean plankton in northeastern Ontario lakes subjected to acidic deposition. Water Air Soil Pollut. 23: 271-291. Cited In: Keller and Yan, 1991.

Keller, W., Molot, L.A., Griffiths, R.W., and Yan, N.D. 1990. Changes in the Zoobenthos Community of Acidified Bowland Lake after Whole-Lake Neutralization and Lake Trout (*Salvelinus namaycush*) Reintroduction. Can. J. Fish. Aquat. Sci. 47: 440- 445.

Keller, W., and Conlon, M. 1995. Crustacean zooplankton communities and lake morphometry in Precambrian Shield lakes. Can. J. Fish. Aquat. Sci. 51: 2424-2434. Yan *et al*., 1996a.

Keller, W., and Yan, N.D. 1991. Recovery of Crustacean Zooplankton Species Richness in Sudbury Area Lakes Following Water Quality Improvements. Can. J. Fish. Aquat. Sci. 48: 1635-1644.

Keller, W., J.R. Pitblado, and J. Carbone. 1992a. Chemical responses of acidic lakes in the Sudbury, Ontario, area to reduced smelter emissions, 1981-89. Canadian Journal of Fisheries and Aquatic Sciences. 49(suppl.1):25-32.

Keller, W., J.M. Gunn., and N.D., Yan. 1992b. Evidence of biological recovery in acid-stressed lakes near Sudbury, Canada. Environmental Pollution 78:79-85.

Keller, W. and Gunn, J.M. 1994. Effects of Sudbury smelter emissions on lakes in Ontario: a review and update. PIB# 3178. Ontario Ministry of Environment and Energy, Cooperative Freshwater Ecology Unit.

Keller, W., J. Carbone, P. Gale, J. Heneberry, M. Malette and J. Gunn. 1998. Data report: Extensive monitoring of acidified lakes in the Sudbury area. 1981 to 1997. Cooperative Freshwater Ecology Unit. Rep. Laurentian University, Sudbury Ontario, Canada. 153 pp.

Keller, W., J.H. Heneberry., and J.M. Gunn. 1999. Effects of emission reductions from the Sudbury smelters on the recovery of acid- and metal-damaged lakes. Journal of Aquatic Ecosystem Stress and Recovery 6: 189-198.

Keller, W., Heneberry, J., Gunn, J.M., Snucins, E., Morgan, G., and Leduc., J. 2004. Recovery of Acid and Metal Damaged Lakes Near Sudbury Ontario: Trends and Status. Cooperative Freshwater Ecology Unit. Department of Biology, Laurentian University, Sudbury, Ontario.

Kelso, J., Love, R., Lipsit, J., and Dermott, R. 1982. Chemical and biological status of headwater lakes in the Sault St. Marie District, Ontario. 165-208. In F. D'Itri [ed.] Acid Precipitation: effects on ecological systems. Ann Arbor Science Publishers, Ann Arbor, MI. Cited In: Glooschenko *et al*., 1992.

Khanharot, B.S., Sehgal, A., and Bhasin, M.K. 1985. "Man and Biosphere" – Studies on the Sikkim Himalayas. Part 5: Acute Toxicity of Selected Heavy Metals on the Tadpoles of *Rana hexadactyla*. Acta Hydrochim. Et Hydrobiol. 18(2): 259-263.

Koivisto, S., Ketola, M., and Walls, M. 1992. Comparison of 5 cladoceran species in short-term and long-term copper exposure. Hydrobiologia 248: 125- 136. Cited In: Yan *et al*., 1996a.

Kovecses, J., Sherwood, G., and Rasmussen, J.B. 2002. The role of ontogenetic diet shifts in metal exposure to yellow perch (*Perca flavescens*). SETAC Globe 3(4): 42-44.

Kraft, C.E., D.M. Carlson, and S.C. Brown. 2003. *The On-line Fishes of New York State*, *Version 2.1*. Department of Natural Resources, Cornell University, Ithaca, NY. Accessed online: http://fish.dnr.cornell.edu/nyfish/fish.html

Kramer, J.R. 1976. Fate of Atmospheric Sulphur Dioxide and Related Substances as Indicated by Chemistry of Precipitation. Dept. of Geology Report. Hamilton, Ontario, McMaster University. Cited In: Nriagu and Rao, 1987.

Kujanpaa, T., C. Turcotte, L. Wakely and R. Manitouwabi. 1984. Biological Survey of Lily Creek, Sudbury, Ontario. Report prepared for Environment 2000.

Lande, S.P., and Guttman, S.I. 1973. The effects of copper sulphate on the growth and mortality of *Rana pipiens* tadpoles. Herpitologica. 29: 22-27. Cited In: Glooschenko *et al*., 1992.

Lauren, D.J., and MacDonald, D.G. 1986. Influence of water hardness, pH, and alkalinity on the mechanisms of copper toxicity in juvenile rainbow trout, *Salmo gairdneri*. Can. J. Fish. Aquat. Sci. 43: 1488- 1496. Cited In: Eastwood and Couture, 2002.

Lautenbach, W.E. 1987. The greening of Sudbury. J. Soil Water Conserv. 42:228-231. Cited In: Yan *et al*., 1996b.

Lawrence, S.G., and Holoka, M.H. 1987. Effects of low concentrations of cadmium on the crustacean zooplankton community of an artificially acidified lake. Can. J. Fish. Aquat. Sci. 44 (Suppl. 1): 163- 172. Cited In: Keller and Yan, 1991.

Lee, Y.H., and Stuebing, R.B. 1990. Heavy Metal Contamination in the River Toad, *Bufo juxtasper* (Inger), Near a Copper Mine in East Malaysia. Bull. Environ. Contam. Toxicol. 45: 272- 279.

LeGrande, G.R., and D.D. Runnells. 1975. Removal of Dissolved Molybdenum by Precipitation of Ferric Ion. Environ. Sci. Technol. 9: 744- 749. Cited In: CCME, 2002.

Lesbarreres, D. 2006. Personal Communication with Cantox Environmental March 30, 2006 re: references cited in your review of Sudbury ERA.

L'Institut canadien de gestion des richesses naturelles – Cambrian College. 1994. Wetland Evaluation – Robinson Marsh, and Lily Creek Marsh, Sudbury, Ontario. Prepared for the Ministry of Natural Resources.

Linder, G., and Grillitsch, B. 2000. Ecotoxicology of Metals. Chapter 7 In: Sparling DW, Linder G, Bishop CA, editors. Ecotoxicology of Amphibians and Reptiles. Society of Environmental Toxicology and Chemistry (SETAC) Technical Publication Series. Pensacola, FL.

Lister, R. 1996. Bird Fact Sheet – Mallard. Hinterland's Who's Who. Environment Canada. CW69-4/14-1996E. http://www.hww.ca/hww2p.asp?id=54&cid=0.

Loizzi, R.F., and D.R. Peterson. 1971. Lipolytic sites in crayfish hepatopancreas and correlation with fine structure. Comp. Biochem. Physiol. 39B: 227-236. Cited In: Bagatto and Alikhan, 1987b.

Lucas, H.F., Edington, D.N., and Colby, P.J. 1970. J. Fish. Res. Board Can. 27: 677. Cited In: Bradley and Morris, 1986.

MacLean, R.S., Borgmann, U., and Dixon, D.G. 1996 . Bioaccumulation kinetics and toxicity of lead in Hyalella azteca (Crustacea, Amphipoda). Can. J. Fish. Aquat. Sci. 53: 2212-2220.

Maclsaac, H.J., Hutchinson, T.C., and Keller, W. 1987. Analysis of Planktonic Rotifer Assemblages from Sudbury, Ontario, Area Lakes of Varying Chemical Composition. Can. J. Fish. Aquat. Sci. 44: 1692- 1701.

Matuszek, J.E., Goodier, J., and Wales, D.L. 1990. The occurrence of Cyprinidae and other small fish species in relation to pH in Ontario Lakes. Trans. Am. Fish. Soc. 119: 850- 861. Cited In: Matuszek and Wales, 1992.

Matuszek, J.E., Wales, D.L., and Gunn, J.M. 1992. Estimated impacts of $SO_2$ emissions from Sudbury smelters on Ontario's sportfish populations. Can. J. Fish. Aquat. Sci. 49(Suppl. 1):87-94.

McNicol, D.K., and Mallory, M.L. 1994. Trends in small lake water chemistry near Sudbury, Canada, 1983- 1991. Water, Air, Soil Pollut. 73: 105- 120. Cited In: Nriagu *et al*., 1998.

McNicol, D.K., Ross, R.K., Mallory, M.L. and Brisebois, L.A. 1995. Trends in waterfowl populations: Evidence of recovery from acidification. In: Gunn, J.M.(Ed.) Restoration and Recovery of an Industrial Region. Ontario Ministry of Natural Resources, Cooperative Freshwater Ecology Unit, Laurentian University, Sudbury, Ontario, Canada. Spring-Verlag, New York.p.155-168.

Mehring, A.L. Jr., Brumbaugh, J.H., Sutherland, A.J., and Titus, H.W. 1960. The tolerance of growing chickens for dietary copper. Poult Sci 39:713-719.

MNR. 2004. Species at Risk in Ontario List. Ontario Ministry of Natural Resources. April 26, 2004. http://www.ontarioparks.com/saro-list.pdf

MOE. 1992. The phytoplankton of Sudbury area lakes (Ontario) and the relationships with acidification status. Ontario Ministry of the Environment. March, 1992. PIBS 1898.

MOE. 1993. Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario. Ontario Ministry of the Environment. August, 1993.

MOE. 2005. Procedures for the Use of Risk Assessment under Part XV.1 of the Environmental Protection Act. Ontario Ministry of the Environment. October, 2005.

MOEE. 1994. Water Management. Policies, Guidelines, Provincial Water Quality Objectives of the Ministry of Environment and Energy. Ontario Ministry of Environment and Energy. July, 1994. PIBS 3303E.

Monet, S. and N. Boucher. 2005. The Natural Heritage Background Study. The Environmental Initiatives Group. The City of Greater Sudbury. Available from the City of Greater Sudbury website at: http://greatersudbury.ca/content/div_planning/documents/Natural_Heritage_Background_Study.pdf

Munzinger, A. 1990. Effects of nickel on *Daphnia magna* during chronic exposure and alterations in the toxicity to generations pre-exposed to nickel. Water Res. 24:845-852. Cited In: Yan *et al*., 1996a.

Munzinger, A. 1994. The influence of nickel on population dynamics and on some demographic parameters of *Daphnia magna*. Hydrobiologia 277:107-120. Cited In: Yan *et al*., 1996a.

Neary, B.P., Dillon, P.J., Munro, J.R., and B.J Clark. 1990. The acidification of Ontario lakes: an assessment of their sensitivity and current status with respect to biological damage; Technical Report, Ministry of the Environment, Dorset, Ontario, 171p. Cited In: Pearson *et al*., 2002a.

NRC. 1980. Mineral tolerance of domestic animals. Subcommittee on mineral toxicity in animals - Agriculture and renewable resources commission on natural resources. National Research Council. National Academy Press. Washington, D.C.

NRC. 2005. Mineral tolerance of animals 2nd revised edition. Committee on Minerals and Toxic Sunbstances in Diets of Water for Animals. Board on Agriculture and Natural Resources, Division on Earth and Life Studies. National Research Council of the National Academies. Washington, D.C.

Nriagu, J. O. 1984. (ed.), "Environmental Impacts of Smelters". Cited In: Sudbury, 2005a.

Nriagu, J.O., and Gaillard, J.F. 1984. The speciation of pollutant metals in lakes near the smelters at Sudbury, Ontario. Adv. Environ. Sci. Technol. 15: 349- 374. Cited In: Nriagu and Rao, 1987.

Nriagu, J.O., and S.S. Rao. 1987. Response of Lake Sediments to Changes in Trace Metal Emissions from the Smelters at Sudbury, Ontario. Environmental Pollution. 44: 211-218.

Nriagu, J.O. and H.K. Wong. 1983. Selenium pollution of lakes near the smelters at Sudbury, Ontario. Nature 301:55-57. Cited In: Bowlby *et al*., 1988.

Nriagu, J.O., H.K.T. Wong, G. Lawson, and P. Daniel. 1998. Saturation of ecosystems with toxic metals in Sudbury basin, Ontario, Canada. The Science of the Total Environment 223: 00-117.

Ort, J.F., and Latshaw, J.D. 1978. The toxic level of sodium selenite in the diet of laying chickens. J. Nutr. 1081114. Cited In : NRC 2005.

Pearson, D.A.B., Gunn, J.M., and W. Keller. 2002a. The Past, Present and Future of Sudbury's Lakes. In: Rousell D.H. and Janson, K.J. (Eds.) The Physical Environment of the City of Greater Sudbury. Ontario Geological Survey Special Volume 6. p. 195-216.

Pearson, D.A.B., Lock, A.S., Belzile, N., and R.J. Bowins. 2002b. Assessing Kelly Lake's History as a Natural Trap for Mining Industry and Municipal Effluent during the Growth of Sudbury. In: Rousell D.H. and Janson, K.J. (Eds.) The Physical Environment of the City of Greater Sudbury. Ontario Geological Survey Special Volume 6. p. 175-194.

Phillips, G.R., and Russo, R.C. 1978. Metal bioaccumulation in fishes and aquatic invertebrates: A literature review. EPA-600/3-78-103. U.S. Environmental Protection Agency. Cited In: Bradley and Morris, 1986.

Pierce, B.A., Hoskins, J.B., and Epstein, E. 1984. Acid tolerance in Connecticut wood frogs (Rana sylvatica). J. Heritol. 18: 159- 167. Cited In: Glooschenko *et al*., 1992.

Playle, R.C. 1998. Modelling metal interactions at fish gills. Sci. Total Eviron. 219: 147- 163. Cited In: Eastwood and Couture, 2002.

Playle, R.C., Dixon, D.G., and Burnison, K. 1993. Copper and cadmium binding to fish gills: estimates of metal-gill stability constants and modelling of metal accumulation. Can. J. Fish. Aquat. Sci. 50: 2678- 2687. Cited In: Eastwood and Couture, 2002.

Pough, F.H. 1976. Acid precipitation and embryonic mortality of spotted salamanders, Ambystoma maculatum. Science. 192: 68-70. Cited In: Glooschenko *et al*., 1992.

Pough, F.H., and Wilson, R.E. 1977. Acid precipitation and reproductive success of Ambystoma salamanders. Water Air Soil Pollut. 7: 307-316. Cited In: Glooschenko *et al*., 1992.

Porter, K.R., and Hakanson, D.E. 1976. Toxicity of mine drainage to embryonic and larval boreal toads. Copeia. 1976: 327-331. Cited In: Glooschenko *et al*., 1992.

Preston BL. 2002. Indirect effects in aquatic ecotoxicology: Implications for ecological risk assessment. Environmental Management 29(3):311-323.

Pyle, G.G., J.W. Rajotte, and P. Couture. 2005. Effects of Industrial Metals on Wild Fish Populations along a Metal Contamination Gradient. Ecotoxicology and Environmental Saftey 61(3):287-312.

Rajotte, J.W. and P. Couture. 2002. Effects of Environmental Metal Contamination on the Condition, Swimming Performance, and Tissue Metabolic Capacities of Wild Yellow Perch (*Perca flavescens*). Canadian Journal of Fisheries and Aquatic Sciences. 59: 1296-1304.

Reasbeck, J.A. 1997. Changes in benthic macroinvertebrate community structure in acid and metal contaminated lakes following water quality improvements. M.Sc. Thesis, Laurentian University, Sudbury, Ontario, Canada. 95 pp. Cited In: Keller *et al*., 1999.

Robitaille, J.F. 2005. Personal Communication to Cantox Environmental. Laurentian University, Department of Biology.

Rosenfeld, I., and Beath, O.A. 1954. Effect of Selenium on Reproduction in Rats. Proc Soc Exp Biol Med 87, 295-297.

Rudd, J.W., Turner, M.A., Townsend, B.E., Swick, A., and Furutani, A. 1980. Can. J. Fish. Aquat. Sci. 37: 848. Cited In: Bowlby *et al*., 1988.

Ryan, P.M., and Harvey, H.H. 1980. Growth responses of yellow perch (*Perca flavescens* Mitchill) to lake acidification in the La Cloche mountain lakes of Ontario. Env. Biol. Fishes 5:97-108.Cited In: Campbell *et al*., 2003.

Sample, B.E., Opresko, D.M., Suter II, G.W. 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Oak Ridge National Laboratory, Oak Ridge, Tennessee. ES/ER/TM-86/R3.

Sample, B.E., and Arenal, C.A. 1999. Allometric models for interspecies extrapolation of wildlife toxicity data. Bulletin of Environmental Contamination and Toxicology. 62: 653-663.

SARA. 2003. Have Your Say Workshop Summary Report. SARA Group. June 2003.

SARA. 2004. Summary Report. 2001 Sudbury Soils Data. SARA Group. July, 2004.

SARA. 2005. Cadmium as a Chemical of Concern. May 2005.

SARA. 2006. Metal Levels in Fish Tissues from Sudbury Lakes. Data Report. Volume II, Appendix G.

Semkin, R. and Kramer, J.R. 1976. Sediment geochemistry of Sudbury area lakes. Can. Mineralogist 14: 73- 90.

Schoenefeld, D. 2005. Personal Communication to Cantox Environmental Inc. February, 2005. Loon Survey for the Greater Sudbury Area.

Sherwood, G.D., Kovecses, J., Hontela, A., and Rasmussen, J.B. 2002. Simplified food webs lead to energetic bottlenecks in polluted lakes. Can. J. Fish. Aquat. Sci. 59:1-5.

Sherwood, G.D., Rasmussen, J.B., Rowan, D.J., Brodeur, J., and Hontela A. 2000. Bioenergetic costs of heavy metal exposure in yellow perch (*Perca flavescens*): in situ estimates with a radiotracer ($^{137}Cs$) technique. Can. J. Fish. Aquat. Sci. 57:441-450.

Smith, E. 2005. Personal Communication: Eric Smith, Ministry of the Environment. Email to Chris Wren of Chris Wren and Associates regarding groundwater impacts to surface water. April 18, 2005.

Stanley, T.R., Spann, J.W., Smith, G.J., and Rosscoe, R. 1994. Main and interactive effects of arsenic and selenium on mallard reproduction and duckling growth and survival. Arch. Environ. Contam. Toxicol. 26:444-451.

Stanley, T.R. Jr., Smith, G.J., Hoffman, D.J., Heinz, G.H. and Rosscoe, R. 1996. Effects of boron and selenium on mallard reproduction and ducking growth and survival. Environmental Toxicology and Chemistry 15(7):1124-1132.

Stokes, P.M., T.C. Hutchinson and K. Krauter. 1973. Heavy-metal tolerance in algae isolated from contaminated lakes near Sudbury, Ontario. Can. J. Bot. 51: 2155-2168.

Sudbury. 2004. Fish Species in Greater Sudbury Lakes. City of Greater Sudbury. http://www.city.greatersudbury.on.ca/cms/index.cfm?app=div_lakewaterquality&lang=en&currID=2271&parID=2177

Sudbury. 2005a. City of Greater Sudbury, Lake Water Quality.
http://www.city.greatersudbury.on.ca/cms/index.cfm?app=div_lakewaterquality&lang=en&currID=680&parID=619

Sudbury. 2005b. City of Greater Sudbury. Individual Lake Information.
http://www.city.greatersudbury.on.ca/cms/index.cfm?app=div_lakewaterquality&lang=en&currID=619&parID=0

Suter G.W. II. 1993. Ecological Risk Assessment. Lewis, Boca Raton, FL, USA.

Suter, G.W. II. 1999. Developing conceptual models for complex ecological risk assessments. Human and Ecological Risk Assessment 5(2):375-396.

Suter, G.W. II, and C.L. Tsao. 1996. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision. Oak Ridge National Laboratory, Oak Ridge, TN. 104 pp. ES/ER/TM-96/R2.

Taylor, G.J., and A.A. Crowder. 1983. Accumulation of atmospherically deposited metals in wetland soils of Sudbury, Ontario. Water, Air and Soil Pollution. 19: 29-42.

Taylor, R.T., G.D. Watson, and M.A. Alikahn. 1995. Comparative sub-lethal and lethal acute toxicity of copper to the freshwater crayfish, Cambarus robustus (Cambaridae, Decapoda, Crustacea) from an acidic metal-contaminated lake and a circumneutral uncontaminated stream. Wat. Res. 29(2):401-408.

Taylor, L.N., McGeer, J.C., Wood, C.M., and McDonald, D.G. 2000. Physiological effects of chronic copper exposure to rainbow trout (*Oncorhynchus mykiss*) in hard and soft water: evaluation of chronic indicators. Environ. Toxicol. Chem. 19(9): 2298- 2308.

Thompson, P.A., J. Kurias, and S. Mihok. 2005. Derivation and use of sediment quality guidelines for ecological risk assessment of metals and radionuclides released to the environment from uranium mining and milling activities in Canada. Environmental Monitoring and Assessment. 110: 71-85.

Turner, M.A., and Swick, A. L. 1983. Can. J. Fish. Aquat. Sci. 40: 2241. Cited In: Bowlby *et al*., 1988.

U.S. EPA. 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. Risk Assessment Forum, US Environmental Protection Agency, Washington, D.C.

U.S. EPA. 2001. Supplemental Guidance to RAGS: Region 4 Bulletins, Ecological Risk Assessment. Originally published November 1995. Website version last updated November 30, 2001: http://www.epa.gov/region4/waste/ots/ecolbul.htm

U.S. EPA. 2002. National Recommended Water Quality Criteria: 2002. U.S. Environmental Protection Agency. Office of Water.

U.S. EPA. 2005. Ecotox Database. http://www.epa.gov/cgi-bin/ecotox_quick_search. Last updated on Thursday, June 23rd, 2005 Accessed June 23, 2005.

Uthe, J.F. and Bligh, *E.G*. 1971. J. Fish. Res. Board Can. 28: 786. Cited In: Bradley and Morris, 1986.

Uutala, A.J. and Smol, J.P. 1996. Paleolimnological reconstructions of long-term changes in fisheries status in Sudbury area lakes. Can. J. Fish. Aquat. Sci. 53: 174- 180.

Wainio, S.E., Beckett, P.J., Pearson, D.A.B. and G.M. Courtin. 2003. A potential remediation strategy using wetland vegetation and aquatic macrophytes in Kelley Lake, Sudbury, Ontario. Proceedings of the Sudbury 2003 Mining and the Environment Conference, Sudbury, Ontario. http://www.ott.wrcc.osmre.gov/library/proceed/sudbury2003/sudbury03/162.pdf

Warren, L.A., Tessier, A., Hare, L. 1998. Modelling cadmium accumulation by benthic invertebrates in situ: the relative contributions of sediment and overlying water reservoirs to organism cadmium concentrations. Limnol. Oceanogr. 43(7): 1442- 1454. Cited In: Couture and Kumar, 2003.

Webb, J.S., I. Thornton, and K. Fletcher. 1968. Geochemical Reconnaisance and Hypocuprosis. Nature. 212: 1010. Cited In: CCME, 2002.

Weeber, R.C., Brook, R.W., Burgess, N., Champoux, L., Masse, D. and McNicol, D.K. 2004. Breeding waterbirds. In: EC/MSC (Environment Canada/Meteorological Service of Canada). 2004. 2004 Acid Deposition, Science Assessment and Summary of Key Results. PDF format.

Wilson, D., Finlayson, B., and Morgan, N. 1981. Calif. Fish and Game. 67: 176. Cited In: Bradley and Morris, 1986.

Winner, R.W., and Farrell, M.P. 1976. Acute and chronic toxicity of copper to four species of *Daphnia*. J. Fish. Res. Board Can. 33: 1685- 1691. Cited In: Yan *et al*., 1996a.

Winner, R.W. 1985. Bioaccumulation and toxicity of copper as affected by interactions between humic acid and water hardness. Water Res. 19:449-455. Cited In: Yan *et al*., 1996b

Winter, J.G., W. Keller, A.M. Paterson, K.M. Somers., and N.D. Yan. 2004. Monitoring biological recovery of Clearwater Lake from acid and metal contamination using phytoplankton community composition (Brief submitted for inclusion in the 2004 National Acid Rain Assessment Report). Cited In: Keller *et al*., 2004.

Whitelaw, C. 1989. Seasonal occurrence of birds in the Sudbury District. Sudbury. Unpublished report. Cited In: Blomme and Laws, 1995.

Wren, C.D., and Stokes, P.M. 1988. Ambio. 17: 28. Cited In: Bowlby *et al*., 1988.

Wren, C.D. 1996. Survey of sediment quality in lakes in the Timmins area. Unpublished data prepared for Royal Oak Mines Ltd.

Wright, E.M. 1995. Assessment of temporal changes in yellow perch (*Perca flavescens* (Mitchill)) biomass as evidence of recovery of acid and metal stressed lakes near Sudbury, Ontario. M.Sc. Thesis, Laurentian University, Sudbury, Ontario, Canada, 91 pp. Cited In: Keller *et al*., 1999.

Yan, N.D., and Dillon, P.J. 1984. Experimental neutralization of lakes near Sudbury, Ontario. In: Environmental impacts of smelters. Edited by J. Nriagu. J. Wiley & Sons, Inc., New York. pp. 417-456. Cited In: Yan *et al*., 1996b.

Yan, N.D., and LaFrance, C.J. 1984. Responses of acidic and neutralized lakes near Sudbury, Ontario, to nutrient enrichment. In: Environmental impacts of smelters. Edited by J. Nriagu. J. Wiley & Sons, Inc., New York. pp. 457-521. Cited In: Yan *et al*., 1996b.

Yan, N.D., Keller, W., Pitblado, J.R., and Mackie, G.L. 1988. *Daphnia-Holopedium* relationships in Canadian Shield lakes ranging in acidity. Verh. Int. Ver. Theor. Angew. Limnol. 23: 252- 257. Cited In: Yan *et al*., 1996a.

Yan, N.D., Welsh, P.G., Lin, H., Taylor, D.J., and Filion, J.M. 1996a. Demographic and Genetic Evidence of the Long-Term Recovery of *Daphnia galeata mendotae* (Crustacea: Daphniidae) in Sudbury Lakes Following Additions of Base: the Role of Metal Toxicity. Can. J. Fish. Aquat. Sci. 53: 1328- 1344.

Yan, N.D., Keller, W., Somers, K.M., Pawson, T.W., and Girard, R.E. 1996b. Recovery of crustacean zooplankton communities from acid and metal contamination: Comparing manipulated and reference lakes. Can. J. Fish. Aquat. Sci. 53: 1301- 1327.

Yan, N.D., Keller, W., Scully, N.M., Lean, D.R.S., and Dillon, P.J. 1996c. Increased UV-B penetration in a lake owing to drought-induced acidification. Nature 382: 141-143. Cited In: Keller *et al*., 2004.

Yan, N.D., Girard, R., Heneberry, J., Keller, W., Gunn, J., Dillion, P.J., and Somers, K.M. 2004. The recovery of copepod, but not cladoceran, zooplankton from severe and chronic effects of multiple stressors. Ecology Letters 7: 452-460.