# Keeyask Generation Project

**Environmental Impact Statement**

Supporting Volume

## Aquatic Environment

KEEYASK

Hydropower Limited Partnership

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# SECTION 7
# FISH QUALITY

Hydropower Limited Partnership

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# 7.0 FISH QUALITY

# 7.1 GENERAL INTRODUCTION

Fish are important indicators of ecosystem health, are valuable as a domestic and commercial resource, and are used in regional recreation. Changes in fish quality (either directly or indirectly via the food chain) can affect domestic and recreational fish consumption, may affect the livelihood of commercial fishers, and the health of any consumer. In this section, fish quality is viewed solely in terms of human consumption. The implications of potential effects of the Keeyask Project on fish quality components such as mercury concentration for wildlife consumers of fish are considered in the sections on mammals and birds in the Terrestrial Environment Volume.

Indicators of fish quality for human consumption that are considered in the assessment include:

- Mercury concentrations in the **epaxial** (*i.e.*, dorsal trunk) musculature (Section 7.2);
- Muscle concentrations of other trace metals (Section 7.3);
- Infection rates of lake whitefish (*Coregonus clupeaformis*) with the internal **cestode** parasite *Triaenophorus crassus* (Section 7.4); and
- Fish palatability (Section 7.5).

Three important domestic and commercial fish species, walleye (*Sander vitreus*), northern pike (*Esox lucius*), and lake whitefish were the main focus of the assessment. Some data on fish mercury concentrations were collected for several other species during the Keeyask environmental studies to address specific questions or because of special request by the KCNs. Except for forage fish, the results for these other species are presented in Appendix 7A.

For each indicator, the approach to the assessment (study area, sources of information, and assessment approach), information pertaining to the environmental setting (historic conditions, current conditions, current trends), and assessment of the Project effects (during construction and operating periods, mitigation, residual effects, cumulative effects, environmental monitoring and follow-up) are presented.

# 7.2 MERCURY

## 7.2.1 Introduction

Fish is the dominant source of dietary mercury exposure to top-level predators, including humans (Mergler *et al.* 2007). Mercury (also referred to as its chemical symbol, Hg) primarily exists as organic methylmercury in the **axial** (mainly trunk and tail) musculature of most freshwater fish (Lockhart *et al.*1972; Grieb *et al.* 1990; Johnston *et al.* 2001; Van Walleghem *et al.* 2007) and higher vertebrates (Wiener and Spry 1996). Methylmercury is the form of mercury that biomagnifies (Watras *et al.* 1998). Due to the detrimental neurotoxicological effects of relatively small amounts of mercury, the frequent consumption of fish with moderate to high mercury concentrations may pose a risk to human health (Clarkson 2002; Mergler *et al.* 2007). A human Health Risk Assessment for mercury exposure in the Keeyask Project area is part of the Socio-economic, Resource Use and Heritage Resources Supporting Volume (SE SV).

In addition to the health risk to humans, elevated levels of mercury are known to have deleterious effects on wildlife, particularly species that consume fish as part of their diet (Scheuhammer *et al.* 2007; for details see TE SV), and on the fish themselves. Sublethal and reproductive effects of mercury exposure have been documented in both laboratory and field studies and for multiple species of freshwater fish. Effects on biochemical processes, damage to cells and tissues, and reduced reproduction have been observed at methylmercury concentrations of approximately 0.3–0.7 parts per million (ppm) mercury wet weight in the whole body and about 0.5–1.2 ppm mercury wet weight in axial muscle (Sandheinrich and Wiener 2011). Such concentrations are common in adult piscivorous fish from natural freshwaters throughout North America (Kamman *et al.* 2005; Schetagne and Verdon 1999a) and even higher concentrations have been reported in northern pike, walleye, lake trout (*Salvelinus namaycush*) and burbot (*Lota lota*) from newly created reservoirs in Québec and Manitoba (Therrien & Schetagne 2008; Bodaly *et al.* 2007; Schetagne and Verdon 1999b). Despite the obvious potential for compromised fish health and reproductive impairment due to elevated body mercury concentrations, there is no clear evidence for associated population level effects in wild fish.

Mercury concentrations in aquatic systems and its bioamagnification up the food chain are the result of numerous processes. Mercury can be released directly into aquatic systems through natural weathering of rocks (Derksen 1978) or from deposition of emissions and effluents from natural and anthropogenic sources such as volcanoes, chlor-alkali plants, fossil fuel combustion, smelters, incinerators, gold mining, waste disposal, and others (Brouzes *et al.* 1977; Swain *et al.* 2007). Long range atmospheric transport has increased mercury deposition since the Industrial Revolution, even at the most remote locations, by a factor of approximately three (Lindberg *et al.* 2007). Human land-use changes can also make mercury more available to biota. For example, inundation of soils and vegetation as a result of reservoir creation introduces inorganic mercury and organic nutrients to the water, which in turn, increases microbial production of methylmercury (Ramlal *et al.* 1987; Kelly *et al.* 1997). Thus, the increase of methylmercury in new reservoirs is not due to increased atmospheric loading of inorganic mercury, but is primarily associated with increased activity of methylating microbes and an overall higher ecosystem efficiency in the conversion of inorganic mercury into methylmercury (Munthe *et al.* 2007).

Methylmercury released into the water column or sediment after bacterial decomposition is available for uptake by phytoplankton or detritivores, which in turn are consumed by zooplankton, other, larger invertebrates and fish. An alternative pathway of methylmercury entry into the food chain is the ingestion of live bacteria by protozoa or rotifers. Small amounts of methylmercury can also enter fish directly from the water via the gills (Hall *et al* 1997) or may be assimilated after bacterial methylation within a fish's intestinal tract (Rudd *et al* 1980). However, the primary pathway of methylmercury uptake in fish is via the diet (Harris and Snodgrass 1993; Hall *et al* 1997). Skeletal muscle is the principle storage tissue for mercury in fish and concentrations generally increase with increasing body size and age (Scott and Armstrong 1972; Green 1986; Peterson *et al* 2007). Methylmercury is biomagnified as its concentration increases with each higher level in the food chain (Jernelöv and Lann 1971; Cox *et al* 1979; Hall *et al* 1997) and the magnitude of bioaccumulation is positively related to food chain length (Harris and Snodgrass

1993; Cabana *et al*. 1994; Kidd *et al*. 1995). Other factors such as habitat preferences, metabolic rate, age, and size also potentially affect the bioaccumulation of mercury in fish (Jackson 1991).

The numerous reports of abnormally high mercury concentrations in fish soon after the impoundment of formerly riverine or lacustrine habitats from geographically distinct and environmentally diverse regions of the world (*e.g*., Cox *et al*. 1979; Bodaly *et al*. 1984a; Surma-Aho *et al*. 1986; Yingcharoen and Bodaly 1993) suggest that the above processes and patterns of mercury accumulation are a common consequence of reservoir creation. In northern Manitoba reservoirs, mercury concentrations generally show a characteristic pattern of increase and decline over time, with maximum values in northern pike and walleye typically occurring 3–8 years after flooding (Bodaly *et al*. 2007; Jansen and Strange 2007). Mercury concentrations in these two piscivores remain elevated over pre-impoundment concentrations for approximately 20–30 years (Bodaly *et al*. 2007; Jansen and Strange 2007).

Maximum mercury concentrations in piscivorous fish from boreal reservoirs often substantially exceed the recommended concentration of 0.2 ppm in fish muscle to be consumed by persons eating large quantities of fish (Wheatley 1979), as well as the standard of 0.5 ppm for fish to be commercially marketed in Canada (Health Canada 2007). Maximum mean mercury concentrations of 0.75–2.59 ppm for northern pike and 0.73–2.38 ppm for walleye have been reported for northern Manitoba lakes, with a significant proportion of the between-lake variability being explained by the percentage of flooded area (Bodaly *et al*. 2007). However, elevated fish mercury concentrations are not unique to reservoirs, and concentrations in piscivores from undisturbed, remote northern lakes can also surpass recommended consumption and marketing thresholds. For example, northern pike of 700 millimetres (mm) standard length from 59 non-impounded lakes in northern Québec had mean mercury concentrations ranging from 0.30 to 1.81 ppm (Schetagne and Verdon 1999a), and Bodaly *et al*. (1993) found that mean mercury concentrations of northern pike of 600 mm standard length approached, or slightly exceeded, 1 ppm in two out of six study lakes in remote areas of northwestern Ontario. Several chemical variables, such as water acidity (Wiener *et al*. 1990) or dissolved organic carbon (Watras *et al*. 1998), or the amount of atmospheric mercury deposition (Wiener *et al*. 2006) are known to affect fish mercury concentrations in lakes. However, it can be assumed that these factors are similar over a relatively small and **physiographically** homogeneous region such as the Keeyask Project area. The relative size of the watershed area (McMurtry *et al*. 1989) and lake size (Bodaly *et al*. 1993) have also been shown to be proximate abiotic factors resulting in differences in fish mercury concentrations. Smaller lakes often have substantially higher epilimnetic water temperatures and a relative larger littoral area compared to bigger lakes, resulting in both higher rates of bacterial mercury methylation and methylmercury uptake in fish via elevated metabolic rates (Bodaly *et al*. 1993). In addition to the factors discussed above, differences in fish mercury content in waterbodies may also arise from basin geology. Derksen (1978) noted the close agreement between major geological fault zones within the Precambrian shield area of Manitoba and areas with relatively large numbers of "mercury contaminated" lakes. The author speculated that fish mercury concentrations "are likely to be higher in areas of base metals deposits".

The approach and methods used to describe the existing environment and to predict Project effects on fish mercury concentrations are outlined in Section 7.2.2. A description of the environmental setting, including an overview of historical fish mercury concentrations with an assessment of temporal trends, and a comparison of current conditions in waterbodies from the Split Lake, Keeyask, and Stephens Lake areas with off-system waterbodies is provided in Section 7.2.3. Construction and operation effects, potential mitigation measures, residual impacts, and monitoring and follow-up are provided in Section 7.2.4.

## 7.2.2 Approach and Methods

### 7.2.2.1 Overview to Approach

Generally, the approach taken for the impact assessment of fish mercury concentrations was similar to the approach applied for other aquatic components. The assessment comprised two major segments:

- A description of the existing conditions in the study area to provide the foundation for assessing the potential effects of the Project on fish mercury concentrations; and
- An impact assessment in which potential effects of the Project on fish mercury concentrations were described.

The assessment focused on fish of domestic and commercial importance for resource users (*i.e.*, walleye, northern pike, and lake whitefish). Walleye and northern pike are at the top of the aquatic food web and represent the worst case scenario in terms of fish mercury concentrations and for the transfer and further bioaccumulation of mercury in wildlife and humans. In addition, smaller-bodied species (*i.e.*, forage fish) including spottail shiner (*Notropis hudsonius*), emerald shiner (*N. atherinoides*), trout-perch (*Percopsis omiscomaycus*), rainbow smelt (*Osmerus mordax*), and juveniles of yellow perch (*Perca flavescens*) and white sucker (*Catostomus commersonii*) were analyzed for mercury to gain a better understanding of the main sources of mercury to fish at higher trophic levels. The analysis of mercury in rainbow smelt also addressed emerging questions regarding the effects of invasive species, specifically, the potential role of food web alterations related to the introduction of rainbow smelt on mercury concentrations in predatory fish within the Project area.

The current conditions in the "environmental setting" were defined for a period of 10 years (1997–2006), although most of the available data on fish mercury concentrations were collected from 2001–2006. The 10-year period was identified to capture recent conditions in the study area with sufficient duration to encapsulate inter-annual variability. Information used for this characterization included data gathered from sampling programs conducted over a number of years under the Keeyask environmental studies, as well as data collected by Fisheries and Oceans Canada (DFO; formerly known as the Department of Fisheries and Oceans). The 2006 data represented the most recent information at the time the EIS was written. Additional sampling of fish for mercury analysis from Split and Stephens lakes was conducted in 2007 under the program "Monitoring of mercury concentrations in fish in northern Manitoba reservoirs" (MMMR; Jansen 2010a), from Gull Lake and the Aiken River in 2009 as part of Keeyask pre-construction monitoring (Jansen 2010b), and from Stephens Lake in 2009 and Split and Assean lakes in 2010 under another monitoring program partially funded by Manitoba Hydro. These data have not been incorporated into the EIS.

Existing mercury concentrations were determined for fish from several off-system lakes, known as the AEA offsetting lakes that will serve as regional reference systems to monitor natural (*i.e.*, not Project-related) fluctuations or trends in fish mercury concentrations against which corresponding changes in study area lakes can be compared. Most AEA offsetting lakes were selected for study at the request of CNP as they may be fished in the future as an alternative to traditional waterbodies that are affected by the Project.

The environmental setting also included a description of historical information (*i.e.*, including data collected prior to1997) to provide an overview of how fish mercury concentrations have changed over time and background data against which future changes can be evaluated. Historic trends in fish mercury concentrations were described for some of the study area lakes using available long-term databases. The historic data were also used to determine if the current environment is relatively stable or still undergoing substantive changes over time.

Potential impacts of the Keeyask GS on mercury concentrations in lake whitefish, northern pike, and walleye were assessed by modelling expected maximum concentrations by:

- Using a predictive model from the scientific literature; and
- Using historic and recent data from a nearby reservoir (*i.e.*, Stephens Lake) as a proxy for the future Keeyask reservoir.

The possible duration of elevated fish mercury concentrations was estimated based on empirical information for existing reservoirs, particularly in Manitoba. Information sources used for the assessment included information obtained from the Keeyask environmental studies, predictions generated for Section 2 (Water and Sediment Quality) and the Physical Environment Supporting Volume [PE SV], and scientific literature pertaining to hydroelectric development in Manitoba and elsewhere.

To assist in characterizing the potential effects of the Project on human health, fish mercury concentrations for the existing environment, as well as for predicted post-Project environmental conditions, were compared to the Health Canada standard of 0.5 ppm maximum acceptable concentration of total mercury in commercially-sold fish (Health Canada 2007) and to the recommended maximum acceptable concentration of 0.2 ppm mercury for persons consuming large quantities of fish (Wheatley 1979). The Health Canada standard of 0.5 ppm is identical to the Manitoba Water Quality Standards, Objectives, and Guidelines (MWQSOGs; Williamson 2002) value for aquatic life tissue residue for the protection of human consumers.

#### 7.2.2.2 Study Area

The study area for fish mercury investigations extends from Split Lake along the Nelson River downstream to and including Stephens Lake in the east. Waterbodies sampled for fish mercury concentrations were primarily riverine lakes and mainstem sections of the Nelson River, but also included two tributary waterbodies, the Aiken River south of Split Lake and Assean Lake north of Split Lake (Map 7-1). The expected magnitude of physical change (*e.g.*, changes in water levels and flows) due to the Project is expected to differ substantially among areas (Project Description Supporting Volume [PD SV] and PE SV) and, consequently, the study area was divided into several areas (Section 1). Three of these areas are relevant for the fish mercury component:

- Split Lake area, including Split Lake and adjoining waterbodies such as Assean Lake and Clark Lake, (*i.e.*, the area upstream of any direct Project influence); this area was included in the study because of the potential for fish that accumulated mercury in the Keeyask area to move upstream.
- Keeyask area, including the Nelson River from the outlet of Clark Lake to approximately 4 kilometre (km) downstream of Gull Rapids (*i.e.*, 'hydraulic zone of influence'); this area was included in the study because the effects on fish mercury concentrations are expected to be strongest; and
- Stephens Lake; the lake was included in the study because it serves as a proxy for modelling future fish mercury concentrations in the Keeyask area and because of potential downstream effects on fish mercury levels.

Waterbodies from the above three areas will be jointly referred to as study area lakes in the following discussion. Fish for mercury analysis were also collected from eight lakes north of the Nelson River (Caldwell, Christie, Kiask, Limestone, Pelletier, Recluse, Thomas, Waskaiowaka) and four lakes located south of the Nelson River (Atkinson, Cyril, Moose Nose, War). These 12 lakes will be jointly referred to as AEA offsetting lakes in the following discussion.

#### 7.2.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for mercury are detailed in this section.

##### 7.2.2.3.1 Existing Published Information

Mercury concentrations in fish from Manitoba waterbodies including Split Lake were first analyzed in 1969/1970 (Bligh 1971) and data extending to December 1972 from lakes and rivers that include waterbodies in the study area were summarized in Derksen (1978). From 1983 to 1989 and 1992 to 2005 the federal government (mainly DFO), the Province of Manitoba, and Manitoba Hydro studied mercury concentrations in fish as part of the "Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion" and its successor programs, collectively referred to as MMMR (Jansen and Strange 2007). Most mercury data for northern Manitoba lakes have been compiled by Derksen (1978), Green (1986, 1990), Bodaly *et al.* (2007), and Jansen and Strange (2007). Long-term records of primarily length standardized mercury concentrations of northern pike, walleye and, to a lesser extent, lake whitefish that pre-date the Keeyask environmental studies exist for Split (1970–1996) and Stephens (1981/1983–1996) lakes. More limited historic data on fish mercury concentrations exist for other waterbodies in the study area, such as Gull Lake (1982), Aiken River (1982), Assean Lake (1981, 1982, 1985, and 1996), and a few of the AEA offsetting lakes, such as Recluse (1982), Kiask (1992), Limestone (seven years from 1978–1996), and Waskaiowaka (1978, 1982) lakes. Most of the above data were stored in the "Inspections" database of Fisheries and Oceans Canada and (since 1997) the "National Contaminants Information System" database of the Canadian Food Inspection Agency (formerly Inspection Branch, DFO), and could be accessed at the Freshwater Institute in Winnipeg. In 2006, the Canadian Food Inspection Agency closed their laboratory and offices at the Freshwater Institute. Since then, the database has been relocated to Ottawa (Hoeve *pers. comm.* 2009). All historic data on mercury concentrations for species collected from study area lakes were considered for the assessment; however, unpublished historic data from any of the AEA offsetting lakes were not included as they were not available.

The effects of previous hydroelectric development in northern Manitoba, including the effects on fish mercury concentrations, were assessed on the Split Lake Resource Management Area as part of the Split Lake Cree Post Project Environmental Review (PPER; Split Lake Cree - Manitoba Hydro Joint Study Group 1996a, b, c).

#### 7.2.2.3.2 Keeyask Environmental Studies

Fish were captured for mercury analysis as part of the Keeyask environmental studies during gillnetting programs in Split Lake (2001–2002, 2005), Clark Lake (2004, 2006), Gull Lake (1999–2002, 2006), Stephens Lake (1999, 2001–2003, 2005), Aiken River (2002–2003), Assean Lake (2001–2002), the Nelson River mainstem between Clark and Gull lakes (2006), and 12 of the AEA offsetting lakes (2004–2006). In addition, data collected by DFO at Split Lake in 1998 (Strange and Bodaly 1999) were used to describe current conditions in the study area. Most fish sampling was conducted between August and September with some samples being collected from selected waterbodies in May and June. A detailed account of fish sampling methods are provided in Appendix 7B.

While not available for inclusion in the EIS, additional sampling of rainbow smelt from Gull Lake and northern pike and walleye from the Aiken River was conducted in 2009 as part of the Keeyask environmental studies to provide more current information on mercury concentrations. Furthermore, mercury data for lake whitefish, northern pike, and walleye from Split and Stephens lakes were collected in 2007 under the MMMR program, and similar information is available from Stephens Lake in 2009, and Split and Assean lakes in 2010.

Muscle samples for mercury analysis were collected from a broad size range of fish within each species, applying guidelines established under MMMR (see Strange and Bodaly 1999) and earlier Manitoba fish mercury programs. All mercury analyses for samples from 1999 onwards were performed by the same analysts at the "trace metals" laboratory of the DFO Freshwater Institute in Winnipeg, using a modified hot block method described by Hendzel and Jamieson (1976) followed by cold vapour atomic absorption spectroscopy. Quality Assurance/Quality Control (QA/QC) measures for sample storage, processing, and data evaluation were similar to those established under MMMR. QA/QC samples included National Research Council of Canada reference materials, laboratory control samples, and triplicate analyses (Appendix 7B).

Fish accumulate mercury over their lifetime, such that older, larger individuals typically have higher concentrations than younger, smaller fish (Scott and Armstrong 1972; Green 1986; Peterson *et al.* 2007; see Appendix 7C for an example). Therefore, mean arithmetic mercury concentrations (micrograms (μg) Hg per gram (g) of tissue wet weight, or ppm) were standardized by fork length to facilitate comparisons between samples of fish from the same waterbodies or between samples of fish from different waterbodies over time (Appendix 7B). Only commercial samples taken by inspectors of the Canadian Food Inspection Agency were available for some of the earliest data years for Split and Stephens lakes. For these, a triplicate sample for Hg analysis was taken from the homogenized filets of five individual fish of each species (DFO 1987). Consequently, mercury concentrations could not be adjusted to fish size for commercial data. For forage fish, arithmetic means were used for comparisons because the regression of mercury concentration and fish length was generally not significant (see Appendix 7D for an example). A detailed description of the analytical methods and the data analysis is provided in Appendix 7B.

Mercury concentrations are mainly presented as mean, length standardized concentrations of total mercury with their 95% confidence limits (CL). Arithmetic means with standard error (SE) are also included in the tables and have been used in the assessment if the relationship between mercury concentrations and fish length was not significant. Unless otherwise indicated, mercury is used synonymously with methylmercury when referring to fish concentrations in this document.

#### 7.2.2.4 Assessment Approach

The general approach used to characterize the existing conditions for fish mercury concentrations in the study area involved compilation of existing data and information for the area and the conduct of baseline field studies to generate the information needed to support the impact assessment. Additionally, mercury concentrations in study area lakes were compared to those from the AEA offsetting lakes to evaluate if and how concentrations differ between on-system and off-system waterbodies. For a broader regional perspective of fish mercury concentrations, study area lakes were compared to waterbodies on the Churchill River Diversion (CRD). Furthermore, an evaluation of trends in fish mercury concentrations for the study area was undertaken to ascertain if conditions are notably changing or are relatively stable. Lastly, fish mercury concentrations were compared to established standards and guidelines.

Several approaches/information sources were used to describe anticipated effects of the Project on mercury concentrations in fish and their human health-related effects, including:

- Local knowledge;
- Results from other components of the Keeyask environmental studies, such as water and sediment quality;
- Modelling exercises aimed at quantifying potential effects of the Project on fish mercury concentrations;
- Use of empirical information for existing reservoirs in Manitoba, in particular the Stephens Lake reservoir;
- Estimates of post-Project fish mercury concentrations were compared to established standards and guidelines;
- Information gained from other existing hydroelectric reservoirs, such as reservoirs in Québec and Finland;
- Information gained from experiments with hydroelectric reservoirs such as the Experimental Lakes Area Reservoir Project (ELARP) and the Flooded Uplands Dynamics Experiment (FLUDEX) in

which small boreal reservoirs were created by flooding surrounding wetlands and uplands, respectively (*e.g.*, Kelly *et al.* 1997, Bodaly *et al.* 2004); and

- Other scientific literature pertaining to Project linkage pathways.

#### 7.2.2.4.1 Federal and Provincial Objectives and Guidelines

The Bureau of Chemical Safety within Health Canada applies a standard (*i.e.*, a maximum level that does not appear as a unique regulation in the Food and Drug Regulations) of 0.5 ppm total mercury to all commercially-sold freshwater fish (Health Canada 2007). At this and higher concentrations, retail fish are deemed unfit for human consumption. In addition to the 0.5 ppm standard, the Medical Service Branch of Health Canada recommended in 1976 that the maximum acceptable concentration of mercury in fish should be 0.2 ppm for those persons that eat large quantities of fish (Wheatley 1979). Although the latter guideline value no longer has official status, it is still unofficially considered by Health Canada and is frequently applied in the risk assessment of fish consumption by health professionals (Wilson *pers. comm.* 2012). Therefore, the 0.2 ppm guideline was also used in the current assessment of fish mercury concentrations.

Proposed Manitoba Quality Objectives and Guidelines for mercury tissue concentrations of aquatic biota for human consumption were directly adopted from the federal Health Canada guidelines for residues in fish tissue (Williamson 2002) and are thus identical to the 0.5 ppm standard.

#### 7.2.2.4.2 Modelling Approaches

Modelling was used to estimate maximum fish mercury concentrations in lake whitefish, northern pike, and walleye from the Keeyask reservoir and Stephens Lake that might be expected as a result of the Project. The first approach involved the evaluation of the applicability of three existing models at predicting fish mercury concentrations in the study area. The applicable models were then used to calculate fish mercury concentrations in the Keeyask reservoir and Stephens Lake. The second approach used current conditions in the nearby Stephens Lake as a proxy to approximate future conditions in the reservoir. A detailed description of the modelling approaches and methods is provided in Appendix 7E.

## 7.2.3 Environmental Setting

### 7.2.3.1 Pre-1997 Conditions

TCN has formally expressed concern over high concentrations of mercury in fish from Split and Clark lakes as a result of CRD and Lake Winnipeg Regulation (LWR; Split Lake Cree - Manitoba Hydro Joint Study Group1996c). As a consequence, there has been a reduction in domestic fishing and the consumption of country food (including fish) by the community as people are afraid to eat fish and there has been an increase consumption of store bought food (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). TCN Members stated that the reduced quality of valuable fish has also reduced the income of commercial fishers (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). The concerns over high environmental mercury concentrations and its health and socio-economic consequences have been voiced by all the KCN communities (CNP Keeyask Environmental Evaluation Report; YFFN Evaluation Report [*Kipekiskwaywinan*]; FLCN Environment Evaluation Report [Draft]).

Long-term records exist for mercury concentrations in northern pike, walleye, and, to a lesser extent, lake whitefish from Split and Stephens lakes. These lakes were monitored as part of the MMMR and its successor programs (Jansen and Strange 2007). However, data that pre-date the completion of CRD/LWR exist only for northern pike and walleye from Split Lake.

Mean mercury concentrations in northern pike and walleye from Split Lake have fluctuated greatly over the 20-year period from 1970–1990 (Figure 7-1) without showing any trends that could be attributed to the operation of either the LWR which was completed in 1976 or the CRD which went into full operation in 1977 (also see Bodaly *et al.* 2007). Maximum mean concentrations for non-commercial samples (*i.e.*, excluding 1970) were observed in 1982 for both northern pike (0.52 ppm) and walleye (0.75 ppm). These maxima were not significantly different from means recorded in many sampling years between 1973 and 1990. Starting in 1990, mercury concentrations began to decrease almost linearly and, by 1996, concentrations of 0.21 ppm in northern pike and 0.29 ppm in walleye were the lowest (northern pike) or second lowest (walleye) recorded throughout the historic record until that year. Lake whitefish, which feed mainly on invertebrates and for which data are only available since 1983, showed a much smaller range in mercury concentrations than the two piscivorous species, but also decreased significantly from maximum concentrations of 0.10 ppm in 1986 to 0.06 ppm in 1996 (Figure 7-1). Mercury concentrations measured in 1982 from northern pike and walleye from the Aiken River, a tributary of Split Lake, were similar to values recorded in fish from Split Lake, with mean concentrations of 0.49 ppm in northern pike and 0.67 ppm in walleye.

In Stephens Lake, mercury concentrations in lake whitefish (0.19 ppm), northern pike (1.05 ppm), and walleye (1.76 ppm) were highest when first sampled (second sample for lake whitefish) more than 10 years after impoundment in 1970 (Figure 7-2). Since mercury concentrations typically peak within three to eight years of flooding (Bodaly *et al.* 2007), maximum mercury concentrations in fish from Stephens Lake were likely somewhat higher than those first recorded in the early 1980s. As observed in Split Lake, mercury concentrations in northern pike and walleye from Stephens Lake steadily decreased over the historic record, reaching approximately 0.35 ppm in both species by 1996. Lake whitefish showed a similar, though more moderate, declining trend; by 1996 mercury concentrations (0.10 ppm) were about half of those initially recorded.

Few historic data exist on fish mercury concentrations at Gull Lake (1982) and Assean Lake (1981, 1982, 1985, and 1996). Mean mercury concentrations in northern pike (0.51 ppm) and walleye (0.78 ppm) from Gull Lake were higher (mostly significantly) than those measured at Assean Lake for these two species (0.18–0.27 and 0.24–0.30 ppm, respectively). As observed at Split and Stephens lakes, mean mercury concentrations in lake whitefish from Assean Lake in 1981 (0.045 ppm) and 1985 (0.027 ppm) were significantly lower than in both piscivorous species.

#### 7.2.3.2 Current Conditions (Post-1996)

##### 7.2.3.2.1 Overview and Regional Context

More than 3,000 fish from the study area were analyzed for mercury. Biological data for these fish are summarized in Appendix 7F and Appendix 7G. For all species, standardized mean mercury concentrations were usually lower than the arithmetic values, reflecting the fact that, in most cases, the mean length of the fish analyzed for mercury was higher than the standard length for each species (Appendix 7F to Appendix 7I).

In all years and waterbodies sampled, mean mercury concentrations were substantially higher in the piscivorous walleye (0.12–0.43 ppm) and northern pike (0.16–0.43 ppm), than in the benthivorous lake whitefish (0.03–0.10 ppm; Appendix 7H). For these three species, standardized mean mercury concentrations were relatively similar among the waterbodies sampled, although fish from Stephens Lake tended to have the highest concentrations (Figure 7-3 to Figure 7-5).

Mercury concentrations in lake whitefish, northern pike, and walleye from Split and Stephens lakes have continued the pattern of decline observed historically (as described in Section 7.2.3.1), such that the most recent (2005) concentrations of 0.18 ppm in northern pike, 0.12–0.20 ppm walleye, and 0.03 ppm in lake whitefish are the lowest of the entire record (Figure 7-1 and Figure 7-2). For both lakes, the current (minimum) concentrations are lower (substantially lower in northern pike and walleye) than the lower 95% confidence limit of the mean of unregulated lakes in northern Manitoba used to calculate background concentrations in Bodaly *et al.* (2007). However, the data for the lakes used in Bodaly *et al.* (2007) to assess background conditions were mainly available only as small samples of large fish for years prior to 1985, conditions that tend to result in elevated concentrations compared to means from more recent collections (Jansen *unpubl. data* 2006). Therefore, the current mercury concentrations in lake whitefish, northern pike, and walleye should not necessarily be interpreted to fall below natural background concentrations.

In all years, the mean mercury concentrations in fish species sampled from study area lakes were below, in most cases substantially below, the Health Canada (2007) 0.5 ppm standard for commercial marketing (Figure 7-3 to Figure 7-5; Appendix 7H and Appendix 7I). While the mean mercury concentrations in lake whitefish never exceeded the 0.2 ppm threshold for safe consumption for persons eating large quantities of fish (Wheatley 1979), average concentrations in northern pike and walleye exceeded this threshold in some years for some waterbodies (Appendix 7H).

When considering mercury concentrations of individual fish rather than mean values, from 0–27% of the lake whitefish, 36–90% of the northern pike, and 4–100% of the walleye exceeded the 0.2 ppm threshold, and from 0–55% of the northern pike and 0–44% of the walleye had mercury concentrations above the 0.5 ppm standard (Appendix 7J). In more recent years (2004–2006), the percentage of individuals in excess of these thresholds has generally decreased. This pattern was not clearly associated with concomitant changes in fish size or age distribution.

Generally, forage fish (mainly less than 120 mm fork length) had substantially lower mercury concentrations (0.02–0.15 ppm, arithmetic mean) than most of the large-bodied fish in all years and waterbodies (Figure 7-6; Appendix 7I). Mercury concentrations in the two shiner species were mostly significantly higher than those in rainbow smelt, trout-perch, and juvenile yellow perch and white sucker, reaching mean arithmetic concentrations of approximately 0.15 ppm in Stephens Lake. Mean arithmetic concentrations of all forage fish species from Stephens Lake in 2003 were, for the most part, significantly higher compared to their conspecifics from Gull Lake.

In order to provide regional context to the mercury concentrations measured in study area lakes, lake whitefish, northern pike, and walleye were sampled for mercury from the AEA offsetting lakes in 2004–2006. Biological data recorded from these fish are summarized in Appendix 7K. Mean mercury concentrations in lake whitefish from seven of the AEA offsetting lakes ranged from 0.04 to 0.06 ppm and did not significantly differ among waterbodies except for Kiask Lake, which had higher concentrations than lake whitefish in Christie and Waskaiowaka lakes (Table 7-1). Mercury concentrations in the two piscivorous species from eight of the AEA offsetting lakes were generally higher and much more variable between lakes than observed for lake whitefish. Mean concentrations ranged from 0.12–0.22 ppm in northern pike and from 0.11–0.38 ppm in walleye (Table 7-1). Concentrations regularly differed significantly between lakes, and, particularly in walleye, were often higher in the lakes located to the north of the Nelson River compared to the south (Figure 7-7).

As observed in fish from the study area lakes, mean mercury concentrations in all species captured from the AEA offsetting lakes were always below the 0.05 ppm standard for commercial marketability (Figure 7-7). Furthermore, mean concentrations in lake whitefish, northern pike, and walleye were always, mostly, and sometimes, respectively, below the 0.02 ppm threshold for safe consumption of large amounts of fish. The proportion of individual fish exceeding the 0.2 ppm threshold was also similar, if not slightly lower, to study area lakes, ranging from 0–16% of lake whitefish, 19–78% of northern pike, and 5–91% of walleye. Up to 16% and 46% of the individual northern pike and walleye, respectively, exceeded a concentration of 0.5 ppm (Appendix 7L).

Lake whitefish from study area lakes in 2005 generally had similar mercury concentrations to their conspecifics from the AEA offsetting lakes, with the exception of Kiask Lake where concentrations were significantly higher than at either Split or Stephens lakes (Figure 7-7). For northern pike, mercury concentrations in study area lakes were also generally similar to fish from the AEA offsetting lakes, except for Thomas, Cyril, and Atkinson lakes, where fish had significantly lower mercury concentrations (Figure 7-7). Differences in mercury concentrations between study area lakes and the AEA offsetting lakes were most pronounced for walleye. Fish from the northern AEA offsetting lakes generally had higher mercury concentrations than their conspecifics from study area lakes, and, for Pelletier and Christie lakes, this difference was significant (Figure 7-7). Excluding the very low value for Split Lake in 2005, concentrations in walleye from the two southern AEA offsetting lakes were lower compared to the rest of the study area lakes, and, in War Lake, were significantly so.

For a broader regional context, fish mercury concentrations from Keeyask Study Area lakes were compared to those from lakes along the CRD route that were sampled as part of the Wuskwatim Project environmental studies. Concentrations, particularly in walleye and northern pike from study area lakes in 2002–2006, were often considerably and significantly lower than those from lakes along the CRD route in 2000–2007 (Jansen and Barth 2003; Jansen and Strange 2009; Jansen 2009). Furthermore, the declining trend in mercury concentrations observed for study area lakes from 1999–2006 was not evident in Wuskwatim Project lakes between 2001 and 2007 (Jansen 2009).

The following sections provide detailed results for each of the areas from which fish mercury data were collected.

#### 7.2.3.2.2 Split Lake Area

Recent (1998–2006) fish mercury data are available from four waterbodies in the Split Lake area: Split Lake, Clark Lake, Assean Lake, and the Aiken River (Map 7-1). A total of 409 northern pike, 439 walleye, 179 lake whitefish, and 122 rainbow smelt were sampled from these waterbodies (Appendix 7H and Appendix 7I). Of these fish, 111 lake whitefish, northern pike, and walleye were collected by DFO in 1998 (*i.e.*, not part of the Keeyask environmental studies).

Mean mercury concentrations in lake whitefish, northern pike, and walleye from Split Lake have continued to decline from historic peak values (see Section 7.2.3.1). Lake whitefish and walleye had significantly lower mercury concentrations in 2005 than in earlier sampling years, and, while not significant, a similar pattern was observed for northern pike. With 0.03 ppm for lake whitefish, 0.18 ppm for northern pike, and 0.12 ppm for walleye, mercury concentrations in 2005 were the lowest measured during all study years (Figure 7-1).

Apart from the long-term declining trend in Split Lake, mercury concentrations in the large-bodied fish have generally been relatively stable and low in area waterbodies since 2001 (Figure 7-3 to Figure 7-5; Appendix 7H). Concentrations in lake whitefish were higher (but not significantly so) in Split Lake, which is located on the CRD route, than in Assean Lake, which is connected to, but located off of the CRD route, in common data years. However, mercury concentrations in lake whitefish from both these lakes remained well below 0.1 ppm (Figure 7-3; Appendix 7H). Mean mercury concentrations for the two large-bodied piscivores, northern pike and walleye, were consistently below 0.35 ppm, and only a few significant differences existed between sampling years and waterbodies (Figure 7-4 and Figure 7-5). Similar to the results for lake whitefish, northern pike and walleye from Assean Lake consistently (although not significantly) had the lowest mercury concentrations in the Split Lake area in the two years data were collected (2001 and 2002).

Mercury concentrations in walleye and northern pike from Split Lake, Clark Lake, Assean Lake, and the Aiken River were generally similar between sampling years. The only significant difference in concentrations observed for both species was for Clark Lake in 2004 and 2006, with the 2006 concentration being higher (Figure 7-4 and Figure 7-5). This difference might be at least partially due to the fact that the walleye and northern pike analyzed for mercury from Clark Lake in 2006 were on average at least one year older than their conspecifics sampled in 2004 (Appendix 7F). Mercury concentrations have been shown to be more strongly associated with fish age than with fish length, particularly in walleye (Green 1986; Jansen and Barth 2003), and length standardization may not adequately standardize mercury concentrations if the age at a certain length (*i.e.*, growth rate) differs between fish samples.

The mean mercury concentration of walleye (0.12 ppm) from Split Lake in 2005 was not significantly different from the length standardized mean of 0.15 ppm obtained for 15 walleye analyzed for trace metals in 2004 using a different analytical laboratory than the fish mercury component (see Section 7.3.3.2.1).

Northern pike and walleye from the Aiken River were sampled from two locations in 2006, near York Landing and near Ilford (Map 7-1). Mercury concentrations in northern pike from York Landing

(0.26 ppm) and Ilford (0.25 ppm) were almost identical, whereas walleye from York Landing (0.19 ppm) had significantly lower mercury concentrations than their conspecifics caught near Ilford (0.24 ppm; Appendix 7H). The latter fish were on average more than a year older than those sampled at York Landing (Appendix 7F), which could have contributed to their higher mercury concentrations.

Except for fish from Split Lake in 2002, rainbow smelt mercury concentrations did not significantly correlate with fork length in the Split Lake area (Appendix 7D), and therefore arithmetic mean concentrations were used for comparisons. Mercury concentrations were generally very low in the area, ranging from 0.017 to 0.039 ppm (excluding the mean of 0.068 ppm for a sample of only six fish for Split Lake in 2005; Figure 7-6). Concentrations for the three years from Split Lake were all significantly higher than for the one sample from Clark Lake in 2004, a year for which no data existed for Split Lake.

#### 7.2.3.2.3 Keeyask Area

Recent (1999–2006) fish mercury data are available in the Keeyask area for Gull Lake and the section of the Nelson River west of Gull Lake extending to Birthday Rapids (Map 7-1) from a total of 183 northern pike, 124 walleye, 69 lake whitefish, 218 rainbow smelt, 51 emerald shiner, 94 spottail shiner, 77 trout-perch, 95 juvenile yellow perch, and 38 juvenile white sucker.

Mean mercury concentrations in lake whitefish from Gull Lake were consistently less than 0.10 ppm (Figure 7-3), and concentrations in northern pike and walleye were consistently less than 0.32 ppm (Figure 7-4 and Figure 7-5). Concentrations in the two piscivores were higher (significantly for northern pike) in 1999 than in subsequent years. Generally, mercury concentrations in lake whitefish, northern pike and walleye from Gull Lake have been relatively stable since 1999, with the exception of the high value for northern pike in 1999 and considering that data on lake whitefish are only available for the three years from 1999–2001 (Figure 7-4).

Mean mercury concentrations of northern pike and walleye from the Nelson River between Clark and Gull lakes in 2006 were lower, but not significantly so, than of their conspecifics from Gull Lake in the same year (Appendix 7H). The mean concentration in northern pike was identical to the length standardized mean of 0.16 ppm obtained for northern pike from the Nelson River that were analyzed for trace metals in 2004 using a different analytical laboratory than for the fish mercury component (see Section 7.3.3.2.1).

Mean arithmetic mercury concentrations in rainbow smelt were generally very low in the Keeyask area, ranging from 0.02–0.07 ppm (Figure 7-6). Concentrations in rainbow smelt from Gull Lake decreased significantly from 0.052 to 0.016 ppm between 2001 and 2004, but returned to 0.052 ppm in 2006 (Figure 7-6). With arithmetic concentrations ranging from 0.05–0.10 ppm in 2003 and 2004, spottail shiner and emerald shiner had mercury concentrations that were 3–6 times higher than those of rainbow smelt in the same years (Appendix 7I). Trout-perch and juvenile yellow perch and white sucker from Gull Lake in 2004 had mercury concentrations that were only slightly elevated compared to rainbow smelt. Intraspecific (*i.e.*, individuals from the same species) comparisons of forage fish from backwater and mainstem habitat in the Nelson River within Gull Lake showed that mercury concentrations were similar in 2004 (Appendix 7I). These results indicated that pathways and rates of mercury bioaccumulation were not substantially different in these two habitat types and/or that fish habitat fidelity, and thus opportunities for differences in mercury bioaccumulation, was not sufficient to result in divergent mercury concentrations in fish from backwaters and mainstream locations.

##### 7.2.3.2.4 Stephens Lake Area

Recent (1999–2005) fish mercury data for Stephens Lake are available from a total of 149 lake whitefish, 204 northern pike, 226 walleye, 136 rainbow smelt, 53 emerald shiner, 40 spottail shiner, 40 trout-perch, and 44 juvenile yellow perch.

Mercury concentrations in lake whitefish, northern pike, and walleye from Stephens Lake have continued to decline from historic peak values, except for a transient and relatively small increase in 2002 (Figure 7-2). Concentrations of 0.03, 0.18, and 0.20 ppm in 2005 for lake whitefish, northern pike, and walleye, respectively, were significantly lower than in previous years and represent the lowest mercury concentrations on record for all three species from this waterbody (Figure 7-2).

The mean mercury concentrations of lake whitefish from Stephens Lake in 2003 and 2005 (Appendix 7H) were not significantly different from the length standardized mean of 0.05 ppm obtained for lake whitefish analyzed for trace metals in 2004 using a different analytical laboratory than for the fish mercury component (see Section 7.3.3.2.1).

Mean arithmetic mercury concentrations in rainbow smelt were low and similar among the four sampling years, ranging from 0.04–0.06 ppm (Figure 7-6). Mean arithmetic mercury concentrations in trout-perch and juvenile yellow perch were similarly low at just over 0.05 ppm in the one year (2003) these species were sampled (Appendix 7I). In contrast, mean arithmetic concentrations in spottail shiner (0.16 ppm) and emerald shiner (0.15 ppm) were more than three times higher in 2003 than in rainbow smelt for the same year.

#### 7.2.3.3 Current Trends/Future Conditions

Current trends in mercury concentrations in fish from Keeyask Study Area lakes are generally well understood. Mercury concentrations have been measured in several fish species in this area since 1999 as part of the Keeyask environmental studies and in Split and Stephens lakes for several decades (up to 35 years) as part of long-term monitoring programs related to previous hydroelectric developments. The long-term pattern of mercury concentrations in Stephens Lake, which was created as a result of impoundment by the Kettle GS, is similar to that observed in several lakes that were flooded as part of CRD (Bodaly *et al.* 2007; Jansen and Strange 2007).

Based on the long-term datasets for Split and Stephens lakes and the several shorter-term (5–6 years) sets for other waterbodies in the Study Area lakes, current mercury concentrations in lake whitefish, northern pike, and walleye are either still declining or stabilizing at low concentrations. It is expected that without any further hydroelectric development mercury concentrations will continue to fluctuate slightly around current (1999-2005) mean values because concentrations are presently similar to those observed in natural reference lakes in the general area. The potential effects of climate change on long-term fish mercury concentrations in the Keeyask Project area described in Section 8.6.

Because of the almost complete lack of historic data on mercury concentrations in forage fish and the limited collection of such information for species other than rainbow smelt during the Keeyask environmental studies, it is difficult to evaluate temporal trends in the concentrations of these small-bodied species that serve as important prey items for large-bodied fish and other vertebrates. The arithmetic concentrations of 0.05, 0.15, and 0.16 ppm recorded in juvenile yellow perch, emerald shiner, and spottail shiner, respectively, from Stephens Lake in 2003, were substantially lower than those measured in 1989 by Ramsey (1990) for a small sample (n = 6) of their conspecifics (0.16, 0.29, and 0.27 ppm, respectively). While these two datasets do not constitute a trend, the results for the three forage fish species are consistent with the decline in mercury concentrations observed in lake whitefish, northern pike, and walleye over the same time period.

The decline in rainbow smelt mercury concentrations observed in Gull Lake between 2001 and 2004 seems to have been transient as it was completely reversed in 2006. Mercury concentrations of this invasive species from all locations are perhaps best characterized as variable at a low level. While the introduction of rainbow smelt has been associated with an increase in mercury concentrations in piscivorous fish in other waterbodies (Franzin *et al*. 1994), the appearance of rainbow smelt in many of the study area lakes does not seem to have resulted in such an increase to date. To the contrary, mercury concentrations in rainbow smelt in these waterbodies are currently below concentrations in other forage species frequently consumed by large-bodied fish, such as shiners (*Notropis* spp.), and their introduction may actually be contributing to the recent declines in mercury concentrations in the piscivore populations (see Section 7.2.3.2.4).

Current (2001–2006) mercury concentrations in all large-bodied fish populations examined in the study area fall below, and mostly substantially below, the current standard of 0.5 ppm for retail fish in Canada (Health Canada 2007). Furthermore, the mean concentration of all lake whitefish and cisco (see Appendix 7A) populations, and a few of the northern pike and walleye populations in the most recent years were below the 0.2 ppm consumption threshold for persons eating large quantities of fish (Wheatley 1979). In addition, the percentage of individual fish with mercury concentrations in excess of 0.2 ppm has decreased for the most recent (2004–2006) data years. However, many of the larger individuals of the large-bodied species considered in this study still exceed one or both of the above threshold or standard values relating to human fish consumption.

## 7.2.4 Projects Effects, Mitigation and Monitoring

### 7.2.4.1 Construction Period

Effects on fish mercury concentrations will be mainly related to the release of methylmercury and organic carbon from disturbed soils and vegetation during construction activities and from areas flooded during the construction period (see below), and to conditions favourable to increased net methylation rates in flooded areas.

The site preparation at the construction camp and work areas north and east of PR 280, the construction of the north and south access roads and the generating station (PE SV, Section 4.4.1; PD SV) and the removal of standing trees from the future reservoir area (PD SV,) will disturb vegetation and soils and likely create additional runoff into the Nelson River. The disturbance of the soil litter/organic layer and the removal of terrestrial vegetation can dramatically increase methylmercury concentrations in the runoff, and has been identified as a major mercury source to aquatic ecosystems (Munthe and Hultberg 2004) and linked to elevated mercury concentrations in fish (Bishop *et al.* 2009; Porvari *et al.* 2009). Although several mitigation measures will be in place to minimize the effects of construction activities on water quality, none of these directly address the runoff or treatment of water with increased mercury/methylmercury concentrations. The clearing of trees in the reservoir area and many other construction activities will be mainly carried out during the winter months, thus minimizing the disturbance of vegetation and soils and avoiding runoff at that time. Bishop *et al.* (2009) have estimated that 15–20% of the mercury in fish from Finnish forest lakes is attributable to forest disturbance during harvest. Most of these lakes are small (from less than one to a few square kilometres) but receive runoff from relatively large catchment areas compared to the conditions in the Keeyask Project area. Given the dilution capacity of the Nelson River as the receiving waterbody and the existing relatively large mercury methylation potential in the environment of the Project area, it is unlikely that the amounts of mercury/methylmercury entering on-system locations via runoff from construction sites will measurably affect the rates of mercury bioaccumulation into fishes compared to the effects of flooding.

To verify this hypothesis, monitoring of fish mercury concentrations will continue during the Project construction phase. Details of fish mercury monitoring planned for the Project are outlined in Section 7.2.4.4.

The flooding of terrestrial vegetation and soils in the future Keeyask reservoir will begin in the later stages of the construction period. During the majority of the Stage I and Stage II diversion periods, water levels in Gull Lake and upstream reaches of the Nelson River are predicted to be only marginally affected and will remain within the limits of existing maximum water levels. Therefore, effects would be expected to be similar to those that would occur under naturally high water levels in the area. During the latter stages of the Stage II diversion, when water levels are increased to near full supply level (PE SV, Section 4.4.1), flooding of soil and vegetation are expected to lead to a mobilization of mercury and microbial decomposition of (formerly) terrestrial organic matter. These effects of reservoir impoundment resulting in accelerated production and bioaccumulation of methylmercury will continue into the operation phase and are discussed in detail in the assessment of operation-related effects in the following section.

### 7.2.4.2 Operation Period

#### 7.2.4.2.1 Time to Reach Maximum Mercury Concentrations in Fish

The internal production and cycling of methylmercury in sediments and water, and the rates of bioaccumulation in biota are expected to increase rapidly in the Keeyask reservoir in response to flooding of wetlands and terrestrial soils that will be associated with reservoir creation. For example, flooded wetlands/peatlands produce methylmercury both at a higher rate (approximately 2 times) and for a more sustained time period than uplands (St. Louis *et al.* 2004; Bodaly *et al.* 2004; Hall *et al.* 2005). In addition to the type of flooded terrain, the response of environmental mercury concentrations in new reservoirs depends on the speed of bank erosion, water level fluctuations, hydraulic residence time, and general ecosystem efficiency in mercury cycling (Harris and Hutchinson 2009). Typically, a time lag exists between the occurrence of maximum mercury concentrations in the water column, lower trophic level organisms, forage fish, and predatory fish, such that older fish at the top of the food chain will be the last to reach peak concentrations. Also, within species, mercury concentrations of younger individuals tend to increase faster than those of older fish (Schetagne and Verdon 1999b; Harris and Hutchinson 2009). The time required to attain maximum concentration in adult (*i.e.*, standard sized) northern pike and walleye varies among boreal reservoirs, and has been reported to occur in 2–13 (mostly 4–8) years for several northern Manitoba reservoirs (Bodaly *et al.* 2007), in 10–13 years for the La Grande complex in Québec (Schetagne *et al.* 2003; Schetagne and Verdon 1999b), and 3–9 years in Finland (Verta and Porvari 1995, cited in Bodaly *et al.* 2007). A fast (less than 1 year) reservoir filling time has been associated with a much earlier (2–3 years post-fill, no species given) occurrence of maximum fish mercury concentrations compared to 6–10 years for a reservoir that took 35 months to fill (Schetagne *et al.* 2003). Based on these studies, and considering that the Keeyask reservoir will reach the initial fill level within a few months of dam construction (PE SV, Section 4.4.1), it can be expected that maximum mercury concentrations will be reached in the reservoir within 3–5 years in lake whitefish and 4–7 years in northern pike and walleye of standard size. These time lines are expected to be similar for fish in Stephens Lake

When referring to predicted mercury concentrations in fish from Stephens Lake, there is potential for concentrations to differ between different parts of the lake. This possibility exists because Stephens Lake will only receive methylmercury from water and biota exported from the Keeyask reservoir, the riverine portion of the Lake is relatively small compared to its total area, and most of the resident lake whitefish, northern pike, and walleye (and their fish prey) will feed primarily in habitats distant from the mainstem. Differences in mercury concentrations in fish caught from different lake areas are known from Southern Indian Lake (SIL; Bodaly *et al.* 1984a) and Cross Lake in northern Manitoba, and in other areas of North America have been associated with habitat-specific differences in mercury supply at the base of the food web (Choy *et al.* 2008) or from arthropod prey (Chumchal *et al.* 2008).

#### 7.2.4.2.2 Predictions of Maximum Mercury Concentrations in Fish

As outlined in Section 7.2.2.4, several modelling approaches were applied to estimate mean maximum mercury concentrations in lake whitefish, northern pike, and walleye in response to Keeyask-related flooding. All models predicted increases in concentrations above current concentrations for both the Keeyask reservoir and Stephens Lake (Table 7-2).

##### *Keeyask Reservoir*

Concentrations in lake whitefish are expected to more than double in the Keeyask reservoir, but will remain below 0.2 ppm. Larger increases are predicted for piscivorous species, with mercury concentrations increasing by 0.6–1.2 ppm over current concentrations in Gull Lake (Table 7-2). The values calculated using the Year 5 percentage flooding estimates were similar for lake whitefish and only marginally higher for northern pike and walleye than those obtained using the initial fill level of the reservoir (Table 7-2). The values derived from the Stephens Lake proxy model were much higher (1.3 ppm for northern pike, 1.4 ppm for walleye) than those estimated using the Johnston *et al.* (1991) Percentage Flooding model (0.8 ppm for both species; see Appendix 7E).

There are several limitations to the Johnston *et al.* (1991) model(s) that must be considered when interpreting its predictions for fish mercury levels in the Keeyask reservoir and Stephens Lake:

- Few of the reservoirs used to build the model(s) had extensive in-lake flooding with no upstream effects, as is predicted to occur in the Keeyask reservoir;
- The Percentage Flooding model explained between 38% (for northern pike) and 57% (for walleye) of the variation in fish mercury burden (Johnston *et al.* 2001), resulting in considerable uncertainties when the model is applied to predict mercury concentrations;
- The measurement of fish mercury concentrations used in the Johnston *et al.* (1991) model(s) generally began after peak concentrations occurred, such that maximum mercury burdens used for modelling were likely lower than actual burdens. This may have resulted in an underestimation of predicted concentrations in the Keeyask reservoir; and
- The model(s) does not include the effect of flow rate

The last issue may be of particular relevance for the Keeyask reservoir, which is expected to have a relatively short hydraulic residence time of up to 30 hours within the mainstem, approximately 30 days within the newly formed back-bay, and only longer in more sheltered, shallower areas farthest from the river mainstem (PE SV, Section 4.4.2.2). Fast flows and a short reservoir residence time have the potential to dilute and/or remove newly generated methylmercury in the water column before it enters the food web and is biomagnified in consumers at higher trophic levels. For a given amount of flooding, fish mercury concentrations will be lower where flow through the reservoir is high. Although most reservoirs used to build the Johnston *et al.* (2001) models were riverine in nature, the hydraulic residence times and the ratios of lacustrine to riverine areas were likely larger than is expected for the Keeyask reservoir. Such differences in hydrology also apply to the Stephens Lake proxy model, and suggest that based on flow rates alone, the predicted fish mercury concentrations for the Keeyask reservoir tend to be an overestimate.

When considering all of the above factors that could not be (fully) accounted for in the models used to make quantitative predictions of mercury concentrations in Keeyask reservoir fish, maximum concentrations in northern pike and walleye can be expected to reach or slightly exceed 1.0 ppm.

### *Stephens Lake*

Fish mercury concentrations in Stephens Lake are expected to increase much less than in the Keeyask reservoir. The model predicted maximum values to increase by approximately 30% in lake whitefish to 0.12 ppm and approximately 50% in northern pike and walleye, such that concentrations will peak just above 0.4 ppm for the two predatory species (Table 7-2). These estimates could be considered high because Stephens Lake and the Keeyask reservoir were treated as one waterbody for modelling purposes, although no actual flooding will occur within Stephens Lake. Conversely, the expected increase in methylmercury concentrations in the Keeyask reservoir (see above) combined with the predicted short hydraulic residence time will potentially result in substantial downstream transport of methylmercury in the water and, perhaps, lower trophic levels plants and animals that may more than compensate for the lack of increased *in situ* methylation in Stephens Lake. Increases in fish mercury concentrations in downstream environments have been attributed to the export of methylmercury from reservoirs in Manitoba (Bodaly *et al.* 2007), Québec (Schetagne *et al.* 2003) and Newfoundland and Labrador (Harris and Hutchinson 2007). It is difficult to quantify the effect of downstream transport on fish mercury concentrations because of the paucity of similar assessments in the literature and the site and species specificity of the downstream effect (Bodaly *et al.* 2007; Harris and Hutchinson 2007). Considering the multiple sources of uncertainty associated with predicting mercury concentrations in fish from Stephens Lake and using a conservative approach, it is expected that concentrations may reach 0.5 ppm in northern pike and walleye and may also be higher (*i.e.*, 0.15 ppm) in lake whitefish, at least in those areas of Stephens Lake close to the riverine corridor. Mercury concentrations in lake whitefish have the potential to be higher than the model predictions for a reason that does not apply to the two piscivorous species. Lake whitefish has been shown to switch from a diet of mainly invertebrates to one almost entirely consisting of fish downstream of Québec generation stations (Schetagne and Verdon 1999b; Schetagne *et al.* 2003). This behaviour seems to be particular to large fish (greater than 450 mm) that feed on stunned and dead forage fish in the tail waters of turbine outlets and spillways. Mean mercury concentrations in this segment of lake whitefish populations have been reported to be as high as 3.0 ppm, whereas, their similar-sized conspecifics in the reservoir had concentrations of approximately 0.5 ppm (Schetagne and Verdon 1999b; Schetagne *et al.* 2003). Such a large (or any) difference in lake whitefish mercury concentrations upstream and downstream of hydroelectric dams is unknown from Manitoba GSs. Thus, maximum mean mercury concentrations in lake whitefish are not expected to exceed 0.15 ppm in Stephens Lake

It must be emphasized that although an attempt was made to provide quantitative estimates of future mercury concentrations in the Keeyask reservoir and downstream areas, all predicted values should be treated more as indicators and not as precise quantitative predictions.

#### 7.2.4.2.3 Duration of Maximum Mercury Concentrations in Fish

The duration of peak mercury concentrations is not well known for most northern Manitoba reservoirs because of year to year variability in concentrations and insufficient temporal resolution of the data. However, the temporal pattern in fish mercury concentrations for the few northern Manitoba reservoirs that were frequently sampled during the time when maximum concentrations likely occurred (*e.g.*, South Bay of SIL, Threepoint and Wuskwatim lakes, Limestone reservoir) indicates that peak concentrations do not last for more than a few years (Jansen and Strange 2007). This conclusion is further supported by the evolution of fish mercury concentrations from the reservoirs of the La Grande complex in Québec (Schetagne *et al.* 2003). Most of these boreal reservoirs likely experienced mineral shoreline erosion and disintegration of fringing peatlands for several decades after the initial fill level was reached (for SIL see Newbury and McCulloch 1984), potentially stimulating mercury methylation beyond the initial pulse soon after reservoir flooding. For the Keeyask reservoir, the shoreline erosion is predicted to be mainly due to peat disintegration (a good source of inorganic mercury and carbon energy for methylating bacteria; see Section 7.2.1) which will increase the reservoir area by approximately 7.5 square kilometres ($km^2$) (*i.e.*, 8% of initial reservoir area) 30 years after impoundment (PE SV, Section 6.4.2.1). Most of the peatland disintegration is expected to occur in the first 10–15 years after reservoir creation, *i.e.*, during the time when maximum fish mercury concentrations will be established. Thus, maximum mean concentrations in fish from the Keeyask reservoir may persist slightly longer than in some of the CRD reservoirs.

#### 7.2.4.3 Residual Effects

Several studies from boreal reservoirs have shown that fish mercury concentrations will eventually return to pre-flooding or background (*i.e.*, local reference lakes) concentrations; however, the time course in which this occurs can be in the order of 20–30 years for piscivorous species (Schetagne and Verdon 1999; Schetagne *et al*, 2003; Bodaly *et al.* 2007; Jansen and Strange 2007). The precise mechanisms that determine the return time (*i.e.*, residual effect) are not well understood, but likely include several reservoir-specific factors, such as the trophic structure of the resident fish community, maximum fish mercury concentrations attained, reservoir morphometry, and the magnitude and duration of peatland disintegration in the reservoir. It is expected that the return time for the Keeyask reservoir, which will experience continuing, although decelerating peatland disintegration for several decades post-impoundment (PE SV, Section 6.4.2.1), will be similar, though slightly shorter compared to that observed at Stephens Lake after the construction of the Kettle GS. It took between 20 and more than 30 years for peak mercury concentrations in lake whitefish, northern pike, and walleye to return to background concentrations after the lake experienced approximately 70% flooding in 1970 (Derksen and Green 1987; CMAMM 1987)

##### 7.2.4.3.1 Construction Period

No residual effects of the construction period on fish mercury concentrations are expected.

##### 7.2.4.3.2 Operation Period

The residual effects on fish mercury concentrations will vary between the three areas considered:

- No or minimal effect are expected for Split/Clark Lake because the few fish that might migrate into the area from the Keeyask reservoir (and thus may have elevated mercury concentrations) will not measurably affect the average mercury level in area fish.
- Flooding of terrestrial areas, eroding shorelines and weekly water level fluctuation in the reservoir (*i.e.*, extending to the beginning of Clark Lake) will result in increased fish mercury concentrations. Predicted maximum mean mercury concentrations are expected to reach just above 1.0 ppm in northern pike and walleye within three to eight years after reservoir creation, and concentration higher than background may persist in these two species for up to 30 years.
- Export of water and biota with elevated mercury concentrations from the reservoir into Stephens Lake is expected to result in mean maximum mercury concentrations of approximately 0.5 ppm for northern pike and walleye with concentrations differing depending on where fish spend most of their lives in the lake. Pre-Project or background concentrations may not be reached for up to 25 years in fish from some areas of Stephens Lake.

##### 7.2.4.3.3 Summary of Residual Effects

The Project effects on fish mercury concentrations in the Keeyask reservoir are considered to be large, of medium geographic extent, and long-term (up to approximately 30 years) (Table 7-3). For Stephens Lake, the effects are expected to be moderate, of medium geographic extent, and of medium- to long-term (less than 25 years) duration.

The technical assessment of fish mercury concentrations is based on models, scientific literature, and information collected from a proxy reservoir (*i.e.*, Stephens Lake) and the overall certainty associated with the predictions is moderate to high. There is high certainty regarding the nature and direction of effects and moderate to high certainty with regard to the magnitude and the duration of effects predicted for the Keeyask reservoir and areas of Stephens Lake near the river mainstem. There is low certainty as to how far effects will extend into the northern parts of Stephens Lake; therefore the assessment was based on the worst case of effects to all fish in Stephens Lake.

#### 7.2.4.4 Environmental Monitoring and Follow-up

As described in Chapter 8 of the Keeyask Generation Project: Response to EIS Guidelines, Environmental Monitoring Plans are being developed as part of the Environmental Protection Program for the Project. The intent of the monitoring plans is to determine whether effects of the Project are as predicted and mitigation measures are functioning as intended. The monitoring plans will also provide for follow-up actions if effects are greater than predicted: the actions that would be taken depend on the nature and magnitude of the effect. The design of the monitoring plans will also consider uncertainties identified during the analysis and/or raised by the KCNs or during the regulatory review process. For example, the technical analysis predicts that effects to water quality will occur within the reservoir and downstream but that no effects will occur upstream in Split Lake; based on local knowledge, the KCNs have identified effects to Split Lake and therefore, Split Lake is being included in the monitoring program.

An outline of monitoring planned for the mercury in fish tissue component of the aquatic environment is provided below. A detailed monitoring plan will be provided in the Aquatic Effects Monitoring Plan (AEMP). This document will provide a detailed description of the rationale, schedule, sampling locations and sampling methods for the technical monitoring that is proposed for the Project. This plan will be implemented in consultation with regulators, in particular DFO and Manitoba Conservation and Water Stewardship, and it is expected that it will change based on regulatory review and on-going review of monitoring results. This monitoring plan will be implemented during the construction phase of the Project and will continue into the operations phase. Reports detailing the outcomes of monitoring programs will be prepared and submitted to regulators, to meet conditions of the Environment Act licence and other authorizations for the Project.

Mercury levels in the tissues of selected fish species (walleye, northern pike, lake whitefish, and one-year old yellow perch) will be measured to continue with pre-Project baseline monitoring and to verify post-Project predicted increases for fish from the Keeyask reservoir and Stephens Lake including uncertainties regarding the direction and magnitude of increases. In addition to the Keeyask reservoir and Stephens Lake, a number of waterbodies not predicted to experience increased fish mercury levels will be monitored. These include Split Lake, Assean Lake, and the Aiken River (also to address the concerns of the KCNs) and one off-system (reference) lake still to be determined. Results on mercury levels in fish tissue will be used to inform health communications undertaken as part of the mercury-in-foods programs outlined in the SE SV (Section 5.4.2.3). Fish mercury concentrations will be monitored regularly starting in 2013. Once the full supply level of the reservoir is reached, concentrations will be measured annually until maximum concentrations are reached, and every three years thereafter until concentrations are stable. For a more detailed description of the monitoring of fish mercury concentrations see the AEMP.

# 7.3 TRACE ELEMENTS

## 7.3.1 Introduction

With the exception of mercury, hydroelectric development generally does not result in increased accumulation of trace elements in biota. Fish from the study area were analyzed for muscle concentrations of trace elements (particularly trace metals/metalloids) in order to establish a concentration baseline. Fish muscle was selected for the analysis because it is the main exposure route for humans. These analyses included mercury because it is part of the whole suite of elements that is routinely analysed by the laboratory when a "trace metal" analysis is requested. Concentrations of trace metals have been studied extensively in aquatic biota at metal-contaminated sites; these and other studies have shown that trace metals other than mercury do not biomagnify and that fish muscle tissue is not a good indicator of environmental exposure (*e.g.*, Miller *et al.* 1992; Pip and Stepaniuk 1997). Pip and Stepaniuk (1997) found an inverse relationship between fish size or age and muscle concentrations of lead, copper, and cadmium in lake whitefish and northern pike from the Nelson River, indicating that as these fish grow and age at least some elements are removed faster from muscle tissue than they are taken up. Under normal conditions (*i.e.*, no local sources of contamination), trace element concentrations in fish muscle are much lower than existing tissue residue guidelines for the protection of human consumers of aquatic biota (*e.g.*, lead and arsenic; Williamson 2002).

## 7.3.2 Approach and Methods

### 7.3.2.1 Overview to Approach

The general approach taken for the impact assessment of trace metals in fish differed slightly than that for mercury (see Section 7.2.2.). The existing conditions in the study area were described but a quantitative assessment of the potential impacts was not conducted because trace element concentrations in fish (except for mercury) are not expected to be increased by the construction and operation of a hydroelectric GS. Instead, simple conceptual models based on the scientific literature and existing data from other EIS studies were used to produce qualitative predictions of future concentrations of trace elements in fish.

A description of the existing conditions focused on fish of domestic and commercial importance for resource users (*i.e.*, walleye, northern pike, and lake whitefish). Current trace element concentrations were compared to concentrations from other waterbodies in the larger region of the study area and to existing Manitoba tissue residue guidelines for the protection of human consumers that exist for lead and arsenic (Williamson 2002).

The current conditions in the "environmental setting" were defined for a period of 10 years (1997–2006), although the only available data on fish trace metal concentrations were collected in 2004 as part of the

Keeyask environmental studies. Historical information (*i.e.*, data collected prior to1997) on fish trace element concentrations was not available from the study area.

#### 7.3.2.2 Study Area

The study area for fish trace elements investigations extends along the Nelson River from Split Lake in the west downstream to and including Stephens Lake in the east. The expected magnitude of physical change (*e.g.*, changes in water levels and flows) due to the Project is expected to differ substantially among areas (as described in the PD SV and PE SV) and, consequently, the study area was divided into several areas (Section 1). Three of these areas are relevant for the fish trace metal component:

- Split Lake area, including Split Lake and adjoining waterbodies such as Assean Lake and Clark Lake, (*i.e.*, the area upstream of any direct Project influence);
- Keeyask area, including the Nelson River from the outlet of Clark Lake to approximately 4 km downstream of Gull Rapids (*i.e.*, "hydraulic zone of influence"), and tributary streams; and
- Stephens Lake area, including Stephens Lake and adjoining waterbodies.

Actual waterbodies sampled for trace metal concentrations within the study area included Split Lake just north of the community of Split Lake, the Nelson River between Clark and Gull lakes, and Stephens Lake with one sampling site downstream of Gull Rapids (see Section 7.2.2.2 and Map 7M-1). The three waterbodies will be jointly referred to as study area lakes in the following discussion.

#### 7.3.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for trace elements are detailed below.

##### 7.3.2.3.1 Existing Published Information

There is limited scientific information on trace element concentrations (other than mercury) in fish along the Rat/Burntwood/Nelson River corridor. The only other known studies measured trace metals in fish captured in the Nelson River above and below the Limestone GS in 1988 (Pip and Stepaniuk 1997), and in fish captured along the Rat/Burntwood River system in 2001 and 2002 as part of Wuskwatim GS environmental studies (Jansen 2005). No historic information on trace element concentrations in fish from the study area is available.

##### 7.3.2.3.2 Keeyask Environmental Studies

To describe existing conditions in the study area, fish were captured from study area lakes for trace element analysis between 05 and 12 October 2004. Sampling included three lake whitefish and 15 walleye from Split Lake, 20 northern pike from Gull Lake, and 10 lake whitefish and one walleye from Stephens Lake. A detailed description of the fish capture and analytical methods and the data analysis is provided in Appendix 7M. Calcium, magnesium, potassium, and sodium were part of the suite of elements that were analyzed. These metals usually occur in vertebrate tissues in concentrations of greater than 100 µg/g (ppm) and are not considered trace elements. Therefore, they were not included in the present assessment.

#### 7.3.2.4 Assessment Approach

The general approach applied for characterizing the existing conditions for fish trace metal concentrations in the study area involved the compilation of existing data and information for the area, and the conduct of baseline field studies to generate information needed to support the assessment. Additionally, fish trace element concentrations from study area lakes were compared to those from waterbodies in the same geographical region to evaluate if and how concentrations differ regionally. Lastly, trace element concentrations in fish were compared to established standards and guidelines.

The information sources used to describe anticipated effects of the Project on trace element concentrations in fish and their human health-related effects included:

- Local knowledge;
- Results from other components of the Keeyask environmental studies, such as water and sediment quality;
- Use of empirical information for existing reservoirs in Manitoba; and
- Other information in the published literature.

##### 7.3.2.4.1 Federal and Provincial Objectives and Guidelines

Proposed Manitoba quality objectives and guidelines for tissue concentrations of aquatic biota for human consumption exist for only a few of the trace elements considered in this report, namely mercury (discussed in Section 7.2), arsenic, and lead (Williamson 2002). The respective guideline values are included in Table 7-4.

## 7.3.3 Environmental Setting

#### 7.3.3.1 Pre-1997 Conditions

No historic scientific data exist for trace metals other than mercury in study area lakes. However, the Split Lake Cree have expressed concern for the reduced quality of fish from Split and Clark lakes, particularly the high concentrations of mercury (see Section 7.2), as a result of CRD and LWR (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). As a consequence, fishing has been reduced and the consumption of country food (including fish) has gone down because people are afraid to eat fish; people now consume more store bought food. The reduced quality of valuable fish had also reduced the income of commercial fishers (Split Lake Cree - Manitoba Hydro Joint Study Group1996c).

#### 7.3.3.2 Current Conditions (Post-1996)

##### 7.3.3.2.1 Overview and Regional Context

Lake whitefish analyzed for trace elements had a fork length of 410–536 mm and were 7–20 years old; northern pike ranged in length between 552 and 705 mm and in age from 4–10 years, and walleye measured 382–485 mm at ages of 6–11 years. The mean lengths, weights, and ages of the three species are presented by waterbody in Table 7-5. The concentrations of most trace elements were at or below the detection limits of the analytical method used (Table 7-4). All mean and individual fish concentrations of arsenic, lead, and mercury, the elements with existing Manitoba guidelines for the protection of human health (Williamson 2002), were below (mercury) or well below (arsenic and lead) the guideline limits. Both arithmetic and length standardized mean mercury concentrations were not significantly different and were typically very similar to those concentrations independently obtained for the same species and waterbodies in 2002–2006 as part of the fish mercury component of the Keeyask environmental studies (see Section 7.2.3.2; Appendix 7H). This congruence with data from a much larger data set also provides some measure of confidence in the results from the trace metal analyses in general, that for most lakes were based on relatively small sample sizes.

A few differences existed in element concentrations among fish species. For example, lake whitefish had higher concentrations of cobalt and strontium than both northern pike and walleye (which were below detection limit; Table 7-4). Furthermore, lake whitefish and northern pike had similar and significantly higher concentrations of manganese and zinc than walleye. However, it must be cautioned that some of these differences could be due to a location effect, because the interspecific comparisons involved mostly fish from different waterbodies. Pip and Stepaniuk (1997) found no significant differences in the muscle concentrations of lead, copper, and cadmium between lake whitefish and northern pike from the Nelson River downstream of the study area (see below).

Generally, trace element concentrations in the fish collected as part of the Keeyask environmental studies were similar to concentrations seen in four waterbodies on the Burntwood River section of the CRD route (Jansen 2005). A notable difference existed in that all three species from Keeyask Study Area lakes (sample sizes of 1–3 fish were excluded) had 2–3 times higher concentrations of copper than their conspecifics from lakes and rivers on the diversion route, while mean iron concentrations were (mostly) 2–3 times higher in fish from the CRD waterbodies than those from study area lakes. These differences were significant. Copper concentrations in lake whitefish and northern pike from study area lakes were at least twice the concentrations observed in these species obtained near the Limestone GS in 1988, whereas concentrations in lead and cadmium were at least an order of magnitude lower (Pip and Stepaniuk 1997). These authors further reported that copper was higher in the skeletal muscle of northern pike captured downstream of the Limestone GS relative to upstream, which was thought to reflect a difference in copper concentrations in the environment (*e.g.*, water and sediment). No such spatial difference was noted for the other elements (*i.e.*, cadmium, lead) in northern pike or in any of the three trace metals that were analyzed from lake whitefish (Pip and Stepaniuk 1997).

### 7.3.3.3 Current Trends

No trends can be observed as there is no data for the study area with which to compare the 2004 results.

## 7.3.4 Project Effects, Mitigation and Monitoring

### 7.3.4.1 Construction Period

No marked increases of trace elements are expected in the aquatic environment due to construction activities (for mercury see Section 7.2) and, as a result, no measurable changes in fish trace metal concentrations are anticipated.

### 7.3.4.2 Operation Period

Most trace element concentrations (*i.e.*, total metals) are not expected to increase measurably in the Keeyask reservoir water column during Project operation. Furthermore, the bioavailability of dissolved metals is predicted to be reduced because of complex formation with organic acids. The few elements that are expected to increase in water concentrations are currently well above (iron and aluminum) or occasionally exceed (silver and selenium) Manitoba Water Quality Standards, Objectives, and Guidelines (MWQSOGs; Williamson 2002). No tissue residue guidelines for the protection of human consumers exist for these four elements. Despite their regular exceedance of guideline values for water, concentrations of iron, aluminum, silver, and selenium in fish muscle from all study area lakes sampled in 2004 were quite low and, with the exception of aluminum, were at or slightly above the detection limit of the analytical method used. Except for mercury, no substantial changes in total trace element concentrations in fish are anticipated as a result of the operation of the Keeyask GS. The predicted increases in methylmercury concentrations are considered in detail in Section 7.2.4.

### 7.3.4.3 Residual Effects

No residual effects of the construction and operation of the Keeyask GS are expected to measurably affect fish trace metal concentrations (Table 7-6). The technical assessment of trace elements in fish tissue is based on scientific literature. The overall certainty associated with the predictions is high.

### 7.3.4.4 Environmental Monitoring and Follow-up

No monitoring is required

# 7.4 *T. CRASSUS* INFECTION

## 7.4.1 Introduction

Several coregonine species, including lake whitefish and cisco, are second intermediate hosts of the cestode *Triaenophorus crassus* (Miller 1952; Lawler and Scott 1954; Watson and Dick 1979). The **plerocercoid** (immature worm) of this parasite forms visible yellowish cysts in the musculature of the host fish. The larvae will not further develop in humans (Miller 1952) and, thus, do not present a direct health concern. However, the cysts are objectionable in appearance and, depending on the rate of infection, limit the commercial marketability of lake whitefish. Therefore, the monitoring of *T. crassus* infection rates in lake whitefish from Manitoba lakes is directly relevant to concerns regarding the economic viability of many fisheries. For example, increased cysts counts and downgraded quality classification have been associated with the collapse of the largest commercial lake whitefish fishery in northern Manitoba at SIL in the early 1980s (Bodaly *et al.* 1984b). The Freshwater Fish Marketing Corporation (FFMC) routinely samples lake whitefish from Manitoba lakes with commercial fisheries and counts cysts from dressed fish in their laboratory in Winnipeg. Fish are then graded based on their "rate of infestation" (RI). The presence of internal parasites is commonly referred to as an (parasitic) infection (Hoffman 1999) and this term is used throughout this document, except when directly referring to RI values as calculated by the FFMC. Historically cisco has occasionally been tested for *T. crassus* infections in northern Manitoba lakes (*e.g.*, Sunde 1964a). However, only lake whitefish were used to describe the current environment and assess Project impacts because commercial fisheries in the study area exist only for this species.

## 7.4.2 Approach and Methods

### 7.4.2.1 Overview to Approach

The general approach taken for the impact assessment of *T. crassus* infections in lake whitefish slightly differed from that for fish mercury concentrations and some of the other aquatic components. The existing conditions in the study area were described but quantitative predictions of Project effects on *T. crassus* infection rates were not attempted because of a generally poor understanding of the effects of reservoir flooding on lake whitefish infection rates by *T. crassus* and a lack of empirical data that would allow estimates of infection rates for the Keeyask reservoir. Instead, the potential for changes in post-Project infection rates was assessed qualitatively based on the available empirical data from the study area and simple conceptual models based on the scientific literature.

To be consistent with other aquatic components, the current conditions in the "environmental setting" were defined for a period of 10 years (1997–2006). However, almost all available data on lake whitefish infection rates were collected in 2003 or later.

Current *T. crassus* infection rates were graded using FFMC protocol (Appendix 7N) to determine two levels of marketability: "export" and "other". Infection rates and grades of lake whitefish were compared to those from some AEA offsetting lakes (see below) and other waterbodies in the larger region of the study area.

### 7.4.2.2 Study Area

The study area for the investigations of lake whitefish infection rates with *T. crassus* extended along the Nelson River from Split Lake in the west downstream to and including Stephens Lake in the east. The expected magnitude of physical change (*e.g.*, changes in water levels and flows) due to the Project is expected to differ substantially among areas (as described in the PD SV and PE SV) and, consequently, the study area was divided into several areas (Section 1). Three of these areas are relevant for the *T. crassus* infection component:

- Split Lake area, including Split Lake and adjoining waterbodies such as Assean Lake and Clark Lake, (*i.e.*, the area upstream of any direct Project influence);

- Keeyask area, including the Nelson River from the outlet of Clark Lake to approximately 4 km downstream of Gull Rapids (*i.e.*, 'hydraulic zone of influence'), and tributary streams; and
- Stephens Lake area, including Stephens Lake and adjoining waterbodies.

Actual waterbodies sampled for *T. crassus* infections within the study area included Split Lake, Gull Lake, and Stephens Lake (Map 7-1). The three waterbodies are jointly referred to as study area lakes in the following discussion. Lake whitefish were also collected for the assessment of *T. crassus* infection from Waskaiowaka Lake, one of the AEA offsetting lakes north of the Nelson River (see Section 7.2.2.2 and Map 7-1). The AEA offsetting lakes were used as regional reference lakes that will not experience direct physical effects of the Project and because of specific request by the KCNs.

### 7.4.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for *T. crassus* infection are detailed below.

#### 7.4.2.3.1 Existing Published Information

Lake whitefish infection rates with *T. crassus* cysts have been recorded in Manitoba since the 1950s (*e.g.*, Schlick 1967a) and the extensive records of the early 1960s have been summarized by Sunde (1965a). These "*T. crassus* surveys" included study area lakes and AEA offsetting lakes. Lake whitefish infection tests have been conducted at Split Lake in 1959 and from 1962–1966, and at the four AEA offsetting lakes Atkinson Lake (1963/64), War Lake (1962/63), Moose Nose Lake (1968), and Waskaiowaka Lake (1972) by the Federal Department of Fisheries (DFO) and/or by the Fisheries Branch of the Manitoba Department of Mines and Natural Resources (MDMNR; Sunde 1965a; Schlick 1967a, 1979; Moshenko 1968). Information on *T. crassus* infections are also available for lake whitefish from Split Lake in 1985, 1996, 1998, and 2004 from commercial monitoring by the FFMC

The effects of previous hydroelectric development in northern Manitoba, including the effects on whitefish infection rates with *T. crassus*, were assessed on the Split Lake Resource Management Area as part of the Split Lake Cree Post Project Environmental Review (PPER; Split Lake Cree - Manitoba Hydro Joint Study Group 1996a, b, c).

#### 7.4.2.3.2 Keeyask Environmental Studies

Eighty-seven lake whitefish were captured for *T. crassus* analysis as part of the Keeyask environmental studies from study area lakes and one Offset Lake (Waskaiowaka Lake) during summer/fall of 2003–2006. Biological information for these lake whitefish is presented in Table 7-7. Cyst counts of *T. crassus* in lake whitefish from Split Lake were also obtained from FFMC commercial monitoring between 1985 and 2004 (Kjarsgaard *pers. comm*. 2007)).

Sampled fish were gutted and stored at -18°C in freezers for subsequent delivery to the FFMC (Winnipeg, MB). At the FFMC, fish were thawed, filleted, and inspected for the presence of *T. crassus* cysts in body musculature according to the "Whitefish Inspection Protocol" (FFMC, a copy is provided in Appendix 7N). The rate of infestation (RI) was calculated as:

RI = number of cysts ÷ weight of fish (lb) ×100

Until 1999, RI values of less than 50 were graded as "export", values of 50–80 were graded as "continental", and values greater than 80 were graded as "cutter" by the FFMC (Kjarsgaard *pers. comm.* 2009). Starting in 2000, all RI values of 50 and higher were graded as "other" (Kjarsgaard *pers. comm.* 2009), and this grading system was also applied to earlier data in this document. Results for 2003–2005 were provided by the FFMC as a single RI value for the entire sample from each lake and as the total dressed weight of all inspected fish, whereas in 2006 cyst counts and dressed weights were available for individual fish.

#### 7.4.2.4 Assessment Approach

The general approach applied for characterizing the existing conditions for *T. crassus* infections in the study area involved compilation of available data and information for the Area, and the conduct of baseline field studies to generate information needed to support the assessment. Additionally, current and historic rates of *T. crassus* infections from study area lakes were compared to those from AEA offsetting lakes and other waterbodies in the general geographical area to evaluate if and how infection rates differ regionally and if effects of hydroelectric development on infection rates can be discerned. The current grading system for *T. crassus* infection rates applied by the FFMC was used to assess potential Project effects on lake whitefish commercial marketability.

The information sources used to describe anticipated effects of the Project on *T. crassus* infection rates of lake whitefish and their human health related effects included:

- Local knowledge;
- Results from other components of the Keeyask environmental studies, such as lower trophic levels;
- Simple conceptual models based on the scientific literature;
- Use of empirical information for existing reservoirs in Manitoba; and
- Other information in the published literature.

### 7.4.3 Environmental Setting

#### 7.4.3.1 Pre-1997 Conditions

Historically, RI values greater than 50 have been regularly observed in northern Manitoba lakes that are considered both impacted or unaffected by anthropogenic activities, and that are fished either recreationally or commercially (Sunde 1964a, 1965a, b). Many of the waterbodies surveyed in the 1960s exceeded the RI threshold of 50 for unrestricted marketability in at least one sampling period (Sunde 1964a, 1965a, b), reaching values of up to 1,062 cysts per 100 lbs of lake whitefish (Sunde 1965a). However, these high RI values were often associated with small sample sizes (*i.e.*, lower total weights) of fish tested, and almost all of the lake whitefish from lakes affected by CRD and LWR that were tested for *T. crassus* infection in multiple years and with larger fish samples had RI values below 50 (Sunde 1964a, 1965a, b; Schlick 1966, 1967a, b).

Lake whitefish from Split Lake have had low RI values in every year they were inspected. In 1959 and the 1960s, RI values ranged from 6 to 18 (Sunde 1965b; Schlick 1967a; one RI value of zero from a sample of 4 lbs was excluded), and in 1985 and 1996 were zero when sampled as part of the FFMC monitoring (Table 7-7). The historical infection rates of lake whitefish from Waskaiowaka and Moose Nose lakes were much higher than observed in Split Lake, with RI values of 108 in 1972 (Schlick 1979) and 198 in 1968 (Moshenko 1968), respectively. Lake whitefish from Atkinson Lake in 1963/64 had RI values of 24 and 106, whereas the infection rate of lake whitefish from War Lake in 1962/63 was only 4 cysts per 100 lbs (Sunde 1965a).

TCN has expressed concern for the reduced quality of fish from Split and Clark lakes, including a perceived increase in lake whitefish parasites, as a result of the CRD and LWR (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). The reduced quality of valuable fish has also been associated with a reduction in the income of commercial fishers. The concerns over high fish infection rates with *T.crassus* and its aesthetic and socio-economic consequences have been expressed by all the KCN communities.

### 7.4.3.2 Current Conditions (Post-1996)

#### 7.4.3.2.1 Overview and Regional Context

All of the lake whitefish sampled from study area lakes as part of the Keeyask environmental studies and FFMC commercial catch monitoring were graded as "export" (Table 7-8). In all years, lake whitefish from Stephens Lake had higher RI values than lake whitefish from either Gull Lake or Split Lake. Lake whitefish sampled from off-system Waskaiowaka Lake in 2003 had a substantially higher RI value of 85 compared to fish sampled from on-system lakes. Lake whitefish from Waskaiowaka Lake were graded as "other" and would have been rejected for export by the FFMC. While it is possible that the current high RI score in Waskaiowaka Lake may have been a result of the small number of fish sampled from this lake, the only previous sample of lake whitefish from this lake in 1972 showed an even higher RI value (Schlick 1979; see Section 7.4.3.1). Lake whitefish from the other three AEA offsetting lakes with existing (historical) data on *T. crassus* infection differed in their RI values and grades. The single samples for War and Moose Nose lakes resulted in a very low and a very high RI score, indicating "Export" and "Other" grade, respectively (Table 7-8). Samples from Atkinson Lake were graded as both "export" and "other" in the same sampling year (Table 7-8).

Lake whitefish collected further upstream along the CRD route from Wuskwatim Lake between 1998 and 2002, were, with the exception of one sample in 2001, all graded as "export", having RI values ranging from 0 to 46 (Jansen 2005). The author partially attributed the RI value of 62 observed in 2001 to the small number of fish sampled. Five of the six lake whitefish samples collected from Wuskwatim Lake between 1961 and 1965 resulted in RI scores of 0–25 with a single outlier value of 64 (Schlick 1966). These results show that current and historic infection rates of lake whitefish from study area lakes are similar to those from other lakes in the region.

#### 7.4.3.2.2 Split Lake Area

Few cysts were observed in lake whitefish taken as part of the Keeyask environmental studies from Split Lake in 2003–2006. All but one sample had a RI value of 0 and the maximum value was 5 (Table 7-8). The seven samples taken by the FFMC in 1998 consistently had higher RI values, ranging from 8–31 with a mean of 16.7 (15 cysts in 90 lbs). In all cases, lake whitefish from Split Lake have been graded as "export", a classification that is consistent with the historic data for this lake (Section 7.4.3.1).

#### 7.4.3.2.3 Keeyask Area

None of the 22 lake whitefish sampled from Gull Lake in 2004 and 2006 had *T. crassus* cysts, resulting in a RI value of 0 and a grade of "export" (Table 7-8).

#### 7.4.3.2.4 Stephens Lake Area

Lake whitefish from Stephens Lake analyzed for infections with *T. crassus* in 2003, 2005, and 2006 had RI scores of 21–37 and fish were graded "export" in all three years (Table 7-8). In 2006, when inspection records were available for individual fish, 23 of the 26 cysts counted were observed in only three of the fish sampled. A high concentration of cysts in a few individuals is not unusual for lake whitefish from northern Manitoba (*e.g.*, Cousins Lake; Schlick 1963) and lake whitefish in general (Pulkinen and Valtonen 1999). Such a pattern has been attributed to a positive relationship between cyst counts and host age in European whitefish (*Coregonus lavaretus*; Pulkinen and Valtonen 1999), although Watson and Dick (1979) were unable to find a dependence of *T. crassus* cyst abundance and fish age for lake whitefish from SIL. Age alone can also not explain the infection pattern observed in lake whitefish from Stephens Lake in 2006, as the age of the three most heavily infected fish (11, 14, 16 years) was similar to the mean age (12.4 years) of all 26 fish. The transmission of *T. crassus* from the first intermediate host (*i.e.*, planktonic copepods) to the second intermediate host (*e.g.*, lake whitefish) occurs during only a short time in spring in the littoral zone (Miller 1952). The resulting yearly parasite accumulation rate may be steady for several years, but can increase substantially once a threshold intensity is reached above which the probability of acquiring further parasites increases and individuals become heavily infected (Pulkkinen and Valtonen 1999). Such a parasite-host dynamic could have been responsible for the relative high cyst counts in the three fish from Stephens Lake in 2006.

### 7.4.3.3 Current Trends

There does not appear to be a distinct temporal trend in the rates of *T. crassus* infection for study area lakes. Lake whitefish from Split Lake, for which the longest record exists, have consistently had RI values below the threshold for "export" grade in all years sampled, with RI values ranging from 0 to 17 between 1959 and 2006 (for 1998 the mean RI for the seven samples was used). With the exception of 1998, the level of lake whitefish infection before and after the implementation of CRD/LWR (approximately 1976) was consistently close to zero, with a maximum RI value of 5. Without additional information, such as the age structure of the lake whitefish that were sampled for *T. crassus* cysts in 1998 or individual cyst counts, there are no obvious reasons for the higher, but still relatively low mean RI of 17 in 1998.

The consistently low infection rates in lake whitefish from Split and Gull lakes are noteworthy considering that the percentage of lake whitefish in a population infected with *T. crassus* can vary substantially between years and has been observed to reach 100% in European whitefish (*C. lavaretus*; Pulkkinen *et al.* 1999). The intensity of *T. crassus* infections is connected to the degree of plankton feeding in coregonines (Miller 1952, Watson and Dick 1979; Pulkkinen *et al.* 1999). Consistent with the low *T. crassus* infections rates, dietary data obtained as part of the Keeyask environmental studies do not indicate extensive zooplanktivory of adult lake whitefish in any study area lakes (see Section 5).

## 7.4.4 Project Effects, Mitigation and Monitoring

### 7.4.4.1 Construction Period

Effects on *T. crassus* infection rates in lake whitefish are mainly related to the relative abundance of its intermediate and final hosts. Such a restructuring of the zooplankton and, particularly fish communities will occur, if at all, only over a longer time period. Although the habitat changes that potentially cause changes in zooplankton and fish community structure may begin during the construction period, their biological consequences will likely not be measurable until sometime during the operation period. Therefore they will be covered in the assessment of operation-related effects in the following section.

### 7.4.4.2 Operation Period

Many ecological factors affect the manifestation of *T. crassus* cyst in the secondary host lake whitefish, such as fish age, infection history, the abundance of primary host copepods, alternative invertebrate prey, other secondary hosts such as cisco, and northern pike as the final host (Miller 1952; Dymond 1965). More recently Pulkkinen *et al.* (1999) has shown that shifts in the abundance and intensity of food competition of coregonine species can affect the transmission rate of *T. crassus*. Some of these factors may be differentially affected by reservoir creation and the direction of an effect may change over time post-Project (*e.g.*, zooplankton abundance; Section 4.4). Therefore, predictions of the rate of *T. crassus* infection in response to the Project based on conceptual models and linkages to other EIS study components will incorporate uncertainties associated with the predictions (if existing) for each individual component. Considering these uncertainties, it can be expected that, *T. crassus* infection rates of lake whitefish in the Keeyask reservoir and Stephens Lake will not be strongly affected by the Project. The predicted reduced abundance (*i.e.*, CPUE) of lake whitefish (and other fish hosts of *T. crassus*) during the early (1–5 years) stages after the creation of the Keeyask reservoir (see Section 5) and the shift in the inshore zooplankton community composition from cyclopoid copepods (*i.e.*, *T. crassus* vectors) to larger daphnids (non-vectors; see Section 4.4) will potentially reduce opportunities for parasite transmission and thus, result in lower cyst counts of *T. crassus* in lake whitefish. This tendency may be reversed in later years, favoured by an expected 30–40% increase in lake whitefish abundance over current levels, the subsiding of conditions that favour daphnids and littoral cladocera, and a moderate decrease in drifting invertebrates, and attenuated by a large increase in benthic invertebrates (Section 4.4 and Section 4.5). *T. crassus* infection rates of lake whitefish from Stephens Lake are not expected to change measurably based on the few and relatively small Project-related changes in zooplankton and benthic invertebrate abundance and community composition (Section 4) and effects to lake whitefish abundance are predicted to be mitigable (Section 5).

As an alternative (to conceptual models) approach to predict post-Project infection rates of lake whitefish by *T. crassus* in the study area, other northern Manitoba reservoirs for which pre- and post-impoundment *T. crassus* cyst counts are available were used as a proxy. Lake whitefish from SIL (upstream on the CRD route and flooded in 1976) were examined for *T. crassus* cyst counts between 1952 and 1982 by DFO (Sunde 1964b, 1965a; Bodaly *et al.* 1983, 1984b). These data indicate that the percentage of lake whitefish classified as export deceased dramatically starting in 1978, such that by 1982 100% of the summer fishery was classified as "other" (Bodaly *et al.* 1984b). However, these results were not confirmed by experimental catches of lake whitefish conducted by DFO in 1978–1982 (Bodaly *et al.* 1983, 1984b), and it is likely that the increased *T. crassus* infestation rates in the commercial catch were mainly due to a switch in lake whitefish exploitation to more infected stocks within SIL (Bodaly *et al.* 1984b). Thus, it is doubtful that cysts levels generally increased in SIL lake whitefish post-impoundment. Such an increase had been predicted by Watson and Dick (1979) based on the assumption that lake whitefish in SIL will switch their diet more to cestode-vectoring copepods because of an expected post-impoundment decline in amphipod abundance.

Although not as extensive as for SIL, pre- and post-impoundment data on *T. crassus* infection rates also exist for Wuskwatim Lake, which is located on the CRD route approximately 200 km closer to the study area. Five of the six lake whitefish samples collected from Wuskwatim Lake between 1961 and 1965 resulted in RI scores of 0–25 with a single outlier value of 64 from a small sample (Schlick 1966). RI values for lake whitefish collected between 1998 and 2002 were similarly low (0–46), with the exception of one small sample in 2001 that scored a RI of 62 (Jansen 2005). Thus, *T. crassus* infection rates in lake whitefish from Wuskwatim Lake did not substantially change after CRD.

Based on both the empirical data from SIL and Wuskwatim Lake, lakes located in the general region of the study area, and on conceptual models it can be expected that cyst counts of *T. crassus* in lake whitefish from the Keeyask reservoir and Stephens Lake will not be measurably affected by the Project.

#### 7.4.4.3 Residual Effects

No operational effects on *T. crassus* infection rates of lake whitefish in the Keeyask reservoir are expected and, thus, no residual effects are anticipated (Table 7-9). The technical assessment of the prevalence of *T. crassus* cysts in lake whitefish is based on scientific literature and information from proxy reservoirs (*e.g.* Southern Indian Lake). The overall certainty associated with the predictions is moderate to high.

#### 7.4.4.4 Environmental Monitoring and Follow-up

No monitoring is required in addition to the monitoring done by the Freshwater Fish Marketing Corporation.

## 7.5 FISH PALATABILITY

### 7.5.1 Introduction

Palatability is an important component of fish quality since taste directly affects the acceptability of fish as a food source. Fish taste is also a very subjective quality as it is influenced by a person's age, food habits, cooking style, perceptions about the food source, and other factors resulting in individual preferences that make generalizations about the acceptability of fish as a food source difficult.

During the planning phase of the Keeyask Project, KCNs members expressed ongoing concerns about the poor flavour and mushy texture of fish they catch from their resource area waterbodies. Several communities requested studies to evaluate the quality of fish for human consumption when prepared using traditional methods.

## 7.5.2 Approach and Methods

### 7.5.2.1 Overview to Approach

The general approach taken for the assessment of fish palatability differed from that for fish mercury concentrations and most other aquatic components. Because of the complexities of what constitutes fish "taste" and the lack of and/or prohibitive costs associated with standard analytical methods for chemical components associated with odour and flavour, fish taste tests were conduct with members of study area communities to describe the existing conditions in study area lakes. Because of a generally poor understanding of the effects of reservoir flooding on fish taste and a lack of taste test studies designed to determine potential differences in palatability before and after reservoir creation, detailed predictions of Project effects on fish taste were not attempted for the Keeyask reservoir. Instead, some scientific results on changes in fish flavour associated with reservoir creation were presented (including linkages to other EIS study components), including a discussion on the potential for such a response in study area lakes.

Members of TCN, at Split Lake, YFFN at York Landing, and FLCN at Bird were asked to judge the palatability of lake whitefish, northern pike, and walleye from waterbodies of their choice in the vicinity of each community. The acceptability of the three species from lakes that had previously been affected by hydroelectric developments was compared by panellists from each community to fish from at least one nearby lake that had not been affected by hydroelectric development.

### 7.5.2.2 Study Area

The study area for the investigations of fish palatability extended along the Nelson River from Split Lake in the west downstream to and including Stephens Lake in the east. The expected magnitude of physical change (*e.g.*, changes in water levels and flows) due to the Project is expected to differ substantially among areas (as described in the Project Description and Physical Environment Technical Assessment Report) and, consequently, the study area was divided into several areas. Three of these areas are relevant for the fish palatability component:

- Split Lake area, including Split Lake and adjoining waterbodies such as Assean Lake and Clark Lake, (*i.e.*, the area upstream of any direct Project influence);
- Keeyask area, including the Nelson River from the outlet of Clark Lake to approximately 4 km downstream of Gull Rapids (*i.e.*, 'hydraulic zone of influence'), and tributary streams; and
- Stephens Lake area, including Stephens Lake and adjoining waterbodies.

Actual waterbodies sampled for fish used in the taste tests within the study area included the Aiken River, Split Lake, Assean Lake, Gull Lake, and Stephens Lake (Map 7-1). These five waterbodies will be referred to jointly as study area lakes in the following discussion. Fish for the palatability studies were also collected from three of the AEA offsetting lakes: Atkinson Lake, War Lake, and Waskaiowaka Lake (Map 7-1) and the Limestone reservoir downstream of the study area on the Nelson River. The AEA offsetting lakes were used as regional reference lakes that will not experience direct physical effects of the Project and because of specific request by the KCNs.

#### 7.5.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for fish palatability are detailed below.

##### 7.5.2.3.1 Existing Published Information

There is limited scientific information on the palatability of fish along the Rat/Burntwood/Nelson River corridor. The only other known study conducted taste tests on lake whitefish, northern pike, and walleye from Baldock, Footprint, Leftrook, and Wuskwatim lakes in the community of Nelson House as part of the Wuskwatim Project EIS in 2002 (Manitoba Hydro and Nisichawayasihk Cree Nation 2003; Ryland and Watts 2002a).

The effects of previous hydroelectric development in northern Manitoba, including the effects on fish palatability, were assessed on the Split Lake Resource Management Area as part of the Split Lake Cree Post Project Environmental Review (PPER; Split Lake Cree - Manitoba Hydro Joint Study Group 1996a, b, c).

##### 7.5.2.3.2 Keeyask Environmental Studies

To describe existing conditions in the study area, fish palatability studies were conducted in the communities of Split Lake (TCN) on fish from Split, Gull, Assean, and Waskaiowaka lakes from 20–22 October, 2003 (Ryland and Watts 2004b), York Landing (YFFN) on fish from Split Lake, War Lake[1], and the Aiken River from 29 April to 6 May, 2002 (Ryland and Watts 2002b), Bird (FLCN) on fish from Gull, Stephens, and Atkinson Lakes, and the Limestone reservoir from 28–30 October, 2003 (Ryland and Watts 2004a). A detailed description of the fish taste procedure and the data analysis is provided in Appendix 7O.

#### 7.5.2.4 Assessment Approach

The general approach applied for characterizing the existing conditions for fish palatability in the study area involved compilation of available data and information for the study area and the general region, including baseline fish taste studies to generate information needed to support the assessment. Results from these taste tests for study area lakes were compared to those from AEA offsetting lakes and other

[1] not Moose  Nose Lake as stated in Ryland and Watts (2002b)

waterbodies in the general geographical area to evaluate if and how fish palatability differs between on-system and off-system waterbodies and regionally.

The information sources used to discuss potential effects of the Project on fish palatability included:

- Local knowledge;
- Results from other components of the Keeyask environmental studies, such as the physical environment and phytoplankton;
- Use of empirical information for existing reservoirs in Manitoba; and
- Other information in the published literature.

## 7.5.3 Environmental Setting

### 7.5.3.1 Pre-1997 Conditions

There are no published data on fish palatability for the Keeyask study area prior to 1997. However, TCN has expressed concern for the reduced quality of fish from Split and Clark lakes and the upper Nelson River as a result of CRD and LWR (Split Lake Cree - Manitoba Hydro Joint Study Group1996c).

### 7.5.3.2 Current Conditions (Post-1996)

#### 7.5.3.2.1 Overview and Regional Context

The mean acceptability scores of walleye, northern pike, and lake whitefish sampled from all locations by panellists at all three communities ranged between 4.8 and 6.4 on the 7 point scale (Figure 7-8), which corresponds to ratings of “like slightly” (5.0) to “like moderately” (6.0).

TCN panellists rated walleye and northern pike from Gull Lake significantly less acceptable than fish from Split Lake, Assean Lake, and Waskaiowaka Lake (Figure 7-8). There was no such difference in the acceptability of lake whitefish from any of the lakes. Fox Lake Cree Nation panellists rated walleye from Gull Lake significantly less acceptable than fish from Atkinson Lake, but not from the Limestone reservoir or Stephens Lake (Figure 7-8). There was no such difference in the acceptability of northern pike or lake whitefish from any of the lakes. At both communities, lake whitefish, northern pike, and walleye from Gull Lake with significantly lower acceptability values corresponded to “like slightly” compared to “like moderately” for the corresponding species from other lakes. There was no significant difference in the mean acceptability of fish species sampled from different waterbodies by YFFN panellists. It should be noted that the results of the palatability study at York Landing do not reflect the views of most community members on the taste of fish species used in the study. In particular, there is concern that no fish were tested that were captured when water temperatures were warmer and taste differences would be more noticeable.

Panellists from all three communities showed some preference for fish from lakes that have not been previously affected by hydroelectric development. While not statistically significant, fish from Waskaiowaka Lake, War Lake, Atkinson Lake, and Assean Lake generally had the highest mean palatability scores. Conversely, fish from Gull Lake, with the exception of lake whitefish at the community of Split Lake, always received the lowest acceptability scores of the four test lakes by members of TCN and FLCN.

Fish from Gull Lake were tested by panellists from both TCN and FLCN, while fish from Split Lake were tested by panellists from both TCN and YFFN. Although the palatability tests were not set-up to compare acceptability scores between communities and test were not done on fish caught at exactly the same time (as was the case for within community tests), it is interesting to note that ratings for fish of the same species sometimes varied more between communities than between fish from the different lakes tested within each community. For example, northern pike from Gull Lake were scored 6.1 ("like moderately") by panellists from FLCN, whereas panellists from TCN scored the same species from Gull Lake with 4.8 ("like slightly"; Figure 7-8).

Similar taste tests with the same three fish species as in the three KCN communities were conducted in 2002 as part of the Wuskwatim Project EIS. It was found that fish sampled from lakes previously affected by hydroelectric development (Footprint and Wuskwatim) were of similar acceptability to panellists from Nelson House as those sampled from lakes that had not been affected by hydroelectric development (Baldock and Leftrook; Manitoba Hydro and Nisichawayasihk Cree Nation 2003; Ryland and Watts 2002a). The acceptability scores of fish from these lakes were similar to those from study area waterbodies with average scores ranging between 5.3 and 6.3 on the 7-point scale.

#### 7.5.3.3 Current Trends

There can be no discussion of trends as there is no historical data with which to compare.

### 7.5.4 Project Effects, Mitigation and Monitoring

#### 7.5.4.1 Construction Period

Hydrocarbons or other olfactory or **gustatory** competent substances that are readily incorporated into fish musculature may be introduced into the aquatic environment during the construction period from site runoff/drainage, cofferdam seepage or accidental spillages. However, the release of significant amounts of such substances is unlikely because of the development and implementation of good management practices and containment of areas sensitive to spillage.

#### 7.5.4.2 Operation Period

The presence of a mouldy or earthy odour and flavour in walleye, northern pike, lake whitefish, and cisco from Cedar Lake in northern Manitoba in the mid-1970s was linked to reservoir flooding by Yurkowski and Tabachek (1980). These authors identified the chemical compounds responsible for the unacceptable odour and taste that had led to a temporary closure of the fishery. It was argued that the input of additional organic matter into the relatively shallow Cedar Lake could have enhanced the growth of blue-green algae, one group of organisms known to produce the odour causing chemicals (Yurkowski and Tabachek 1980). However, the source of the muddy flavour was never determined at the time it occurred and the problem has not been reported in later years. Although local knowledge indicates poor fish quality in study area lakes as a result of CRD and LWR, mouldy or muddy odour and flavour were not specifically mentioned (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). Furthermore, blue-green algae blooms are not known from study area lakes (Section 4.2), which are mostly relatively deep, cold, and well mixed. Because the areas of the Keeyask reservoir that provide suitable habitat for fish of domestic and commercial importance are predicted to remain deep, cold, and well mixed, flavour problems such as reported for Cedar Lake are not expected to occur in the reservoir or other study area lakes. Generally, there are no known pathways that connect normal operation related activities to fish palatability. As outlined in Section 7.5.4.1, accidental spillage of substances that have the potential to affect fish smell and flavour may also happen during GS operations. However, the release of significant amounts of such substances into the aquatic environment is unlikely because of the development and implementation of good management practices and containment of areas sensitive to spillage.

Based on the palatability studies conducted in 2002–2004, fish from Gull Lake currently receive the lowest acceptability scores by members of TCN and FLCN compared to other study area lakes and AEA offsetting lakes. The palatability of fish from Gull Lake is not expected to change after the creation of the Keeyask reservoir.

#### 7.5.4.3 Residual Effects

The preferences for certain fish species and for fish from particular lakes that have been expressed by members of TCN and FLCN in the past (Section 7.5.3.1) will likely not change throughout the operating period of the GS. Given the subjectivity of taste perception and the complexity of factors contributing to an enjoyable culinary experience make it difficult to predict effects to palatability based on examination of changes to fish diet, *etc.*, alone. No long-term residual effects of the Project on fish palatability are anticipated, and because of the subjectivity of the assessment procedure, the certainty of these predictions is high (Table 7-10).

#### 7.5.4.4 Environmental Monitoring and Follow-up

No monitoring is required.

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# TABLES, FIGURES, AND MAPS

**Table 7-1: Mean arithmetic (± standard error [SE]) and standardized (± 95% confidence level [CL]) mercury concentration of lake whitefish, northern pike, and walleye from the AEA offsetting lakes in 2004–2006**

| Species | Lake | Year | n[1] | Arithmetic | SE | Standard | 95% CL |
|---|---|---|---|---|---|---|---|
| Lake whitefish | Caldwell | 2005 | 25 | 0.049 | 0.007 | 0.039 | 0.033–0.047 |
| | Christie | 2005 | 23 | 0.041 | 0.005 | 0.036 | 0.031–0.043 |
| | Kiask | 2005 | 32 | 0.062 | 0.009 | 0.056 | 0.045–0.069 |
| | Limestone | 2005 | 25 | 0.103 | 0.021 | 0.049 | 0.035–0.070 |
| | Thomas | 2005 | 23 | 0.041 | 0.005 | 0.042 | 0.036–0.048 |
| | Waskaiowaka | 2005 | 23 | 0.045 | 0.004 | 0.036 | 0.031–0.042 |
| | Cyril | 2006 | 51 | 0.057 | 0.005 | 0.042 | 0.038–0.046 |
| Northern pike | Christie | 2005 | 31 | 0.313 | 0.040 | 0.218 | 0.194–0.244 |
| | Kiask | 2005 | 19 | 0.335 | 0.044 | 0.205 | 0.172–0.245 |
| | Thomas | 2005 | 27 | 0.223 | 0.038 | 0.134 | 0.111–0.162 |
| | Waskaiowaka | 2005 | 14 | 0.233 | 0.021 | 0.190[2] | 0.143–0.253 |
| | Atkinson | 2006 | 61 | 0.148 | 0.014 | 0.137 | 0.125–0.150 |
| | Cyril | 2006 | 53 | 0.122 | 0.017 | 0.111 | 0.102–0.121 |
| | Moose Nose | 2004 | 39 | 0.174 | 0.009 | 0.153 | 0.136–0.171 |
| | War | 2006 | 50 | 0.159 | 0.017 | 0.152 | 0.137–0.168 |
| Walleye | Caldwell | 2005 | 25 | 0.305 | 0.042 | 0.247 | 0.218–0.280 |
| | Christie | 2005 | 25 | 0.340 | 0.049 | 0.290 | 0.258–0.327 |
| | Pelletier | 2005 | 35 | 0.464 | 0.037 | 0.381 | 0.348–0.417 |
| | Recluse | 2005 | 26 | 0.382 | 0.034 | 0.250 | 0.206–0.304 |
| | Thomas | 2005 | 24 | 0.186 | 0.029 | 0.140 | 0.121–0.162 |
| | Waskaiowaka | 2005 | 28 | 0.364 | 0.026 | 0.266 | 0.227–0.311 |
| | Atkinson | 2006 | 38 | 0.146 | 0.013 | 0.132 | 0.116–0.150 |
| | War | 2006 | 45 | 0.129 | 0.006 | 0.107 | 0.099–0.116 |

1. n = the number of fish sampled
2. The relationship between mercury concentration and fish length was not significant.

**Table 7-2: Model-derived estimates of mean maximum mercury concentrations (ppm) in lake whitefish, northern pike, and walleye for the Keeyask reservoir and Stephens Lake after the construction of the Keeyask Generating Station compared to current mercury concentrations in Gull Lake (2002–2006) and Stephens Lake (2001–2005)**

| Model | Species | | |
|---|---|---|---|
| | Lake whitefish | Northern pike | Walleye |
| **Keeyask Reservoir** | | | |
| Current (Gull Lake) | 0.07 | 0.22 | 0.23 |
| **PF model [1]** | | | |
| Day 1 [2] | 0.18 | 0.81 | 0.83 |
| Year 5 [3] | 0.18 | 0.83 | 0.85 |
| **Proxy model [4]** | | | |
| Day 1 | 0.19 | 1.30 | 1.42 |
| Year 5 | 0.19 | 1.33 | 1.46 |
| **Stephens Lake** | | | |
| Current | 0.09 | 0.26 | 0.29 |
| **PF model [5]** | | | |
| Day 1 | 0.12 | 0.40 | 0.43 |
| Year 5 | 0.12 | 0.41 | 0.43 |

1. Percent flooded (PF) regression model modified after Johnston *et al.* (1991).
2. Day 1 uses the reservoir area at the first time the initial fill level is in effect.
3. Year 5 uses the estimated reservoir area at Year 5 post-flooding (PE SV, Section 6, Shoreline Erosion Processes; also see Appendix 7E, Model selection).
4. Stephens Lake is used as a proxy for future conditions in the Keeyask reservoir.
5. Applies the proportion of flooded area to the combined area of Stephens Lake and the Keeyask reservoir.

**Table 7-3: Residual effects on fish quality during operation — mercury**

| Environmental Effect | Mitigation/Enhancement | Residual Effect |
|---|---|---|
| **Split/Clark Lake**<br>Potential effect due to the harvest and consumption of fish with high mercury levels that moved from the Keeyask reservoir into Split/Clark Lake but probability of catching such a fish is low. Tagging data to date suggest that only a small proportion of fish in upstream waterbodies originated from the Keeyask reach. | Information and awareness programs for local resource users on how to minimize mercury uptake under current fish harvest practices.<br>Continuous monitoring of fish mercury concentrations and incorporation of current results into updated local consumption guidelines | Negligible |
| **Below Clark Lake to GS**<br>Flooding of terrestrial areas, eroding shorelines and weekly water level fluctuation in the reservoir will result in increases in fish mercury concentrations. Predicted maximum mean mercury concentrations are in the range of 0.81-1.33 ppm for northern pike and 0.83-1.46 ppm for walleye and likely will just exceed 1.0 ppm. | Reduction in potential flooded area through selection of 159 m forebay elevation.<br>Information and awareness programs for local resource users on how to minimize mercury uptake without abandoning fish harvest;<br>Use of AEA offsetting lakes to provide fish for local consumption;<br>Continuous monitoring of fish mercury concentrations and incorporation of current results into updated local consumption guidelines. | Large, of medium geographic extent, long-term (up to ~30 years). |

**Table 7-3: Residual effects on fish quality during operation — mercury**

| Environmental Effect | Mitigation/Enhancement | Residual Effect |
|---|---|---|
| **Downstream of GS/ Stephens Lake** | | |
| Export of water and biota with elevated mercury concentrations from the upstream reservoir into Stephens Lake would likely result in moderate increases in fish mercury concentrations; estimated mean maximum mercury concentrations are 0.5 ppm for northern pike and walleye. Concentrations may differ depending on where fish spend most of their lives in the lake. | Information and awareness programs for local resource users on how to minimize mercury uptake under current fish harvest practices;<br>Suggesting alternative waterbodies for fish harvest based on measurements of mercury concentrations;<br>Continuous monitoring of fish mercury concentrations and incorporation of current results into updated local consumption guidelines. | Moderate, of medium geographic extent, medium- to long-term. |

**Table 7-4: Mean (±standard error) trace element concentrations in muscle of walleye, lake whitefish, and northern pike captured in Split and Stephens lakes and the Nelson River in 2004**

| Element | DL[1] | GV[2] | Walleye[3] | | Lake whitefish | | Northern pike |
|---|---|---|---|---|---|---|---|
| | | | Split L (n = 15) | Stephens L (n = 1) | Split L (n = 3) | Stephens L (n = 10) | Nelson R (n = 20) |
| Aluminum (Al) | 0.3 | - | 2.2 ± 0.1 | 0.9 | 1.6 ± 0.2 | 1.3 ± 0.2 | 1.7 ± 0.3 |
| Antimony (Sb) | 0.03 | - | 0.04 ± 0.02 | 0.22 | <0.03 ± 0.00 | <0.03 ± 0.00 | <0.03 ± 0.00 |
| Arsenic (As) | 0.1 | 3.5 | <0.10 ± 0.00 | <0.10 | <0.10 ± 0.00 | <0.10 ± 0.02 | <0.10 ± 0.00 |
| Barium (Ba) | 0.3 | - | 0.63 ± 0.06 | 0.40 | 0.77 ± 0.03 | 0.38 ± 0.09 | 0.60 ± 0.06 |
| Beryllium (Be) | 0.03 | - | <0.03 ± 0.00 | <0.03 | <0.03 ± 0.00 | <0.03 ± 0.00 | <0.03 ± 0.00 |
| Bismuth (Bi) | 0.05 | - | <0.05 ± 0.00 | <0.05 | <0.05 ± 0.00 | <0.05 ± 0.00 | <0.05 ± 0.00 |
| Boron (B) | 0.3 | - | 1.72 ± 0.29 | 1.10 | 2.17 ± 0.53 | 1.30 ± 0.51 | 2.47 ± 0.43 |
| Cadmium (Cd) | 0.005 | - | <0.005 ± 0.000 | 0.008 | <0.005 ± 0.000 | <0.005 ± 0.000 | <0.005 ± 0.001 |
| Chromium (Cr) | 0.3 | - | 0.31 ± 0.04 | <0.30 | <0.30 ± 0.00 | <0.30 ± 0.08 | <0.30 ± 0.00 |
| Cobalt (Co) | 0.005 | - | <0.005 ± 0.000 | <0.005 | 0.0123 ± 0.004 | 0.0109 ± 0.001 | <0.005 ± 0.000 |
| Copper (Cu) | 0.03 | - | 0.90 ± 0.08 | 3.37 | 0.64 ± 0.08 | 1.78 ± 0.40 | 0.87 ± 0.06 |
| Iron (Fe) | 2 | - | <2.0 ± 0.4 | <2.0 | <2.0 ± 0.7 | 5.4 ± 1.7 | 3.0 ± 0.5 |
| Lead (Pb) | 0.03 | 0.5 | <0.03 ± 0.00 | 0.10 | <0.03 ± 0.00 | <0.03 ± 0.00 | <0.03 ± 0.00 |
| Manganese (Mn) | 0.3 | - | 0.7 ± 0.1 | 0.4 | 1.5 ± 0.3 | 1.1 ± 0.2 | 1.7 ± 0.1 |
| Mercury (Hg) | 0.01 | 0.5 | 0.15 ± 0.01 | 0.15 | 0.06 ± 0.01 | 0.09 ± 0.01 | 0.21 ± 0.01 |
| Molybdenum | 0.05 | - | <0.05 ± 0.00 | <0.05 | <0.05 ± 0.00 | <0.05 ± 0.00 | <0.05 ± 0.00 |
| Nickel (Ni) | 0.05 | - | <0.05 ± 0.00 | <0.05 | <0.05 ± 0.00 | <0.05 ± 0.00 | <0.05 ± 0.00 |
| Selenium (Se) | 0.1 | - | 0.3 ± 0.0 | 0.2 | 0.2 ± 0.0 | 0.2 ± 0.0 | 0.2 ± 0.0 |
| Silver (Ag) | 0.005 | - | <0.005 ± 0.000 | <0.005 | <0.005 ± 0.000 | <0.005 ± 0.000 | <0.005 ± 0.000 |
| Strontium (Sr) | 0.05 | - | 5.5 ± 0.4 | 4.8 | 14.2 ± 0.5 | 11.6 ± 2.6 | 4.3 ± 0.4 |
| Thallium (Tl) | 0.003 | - | 0.005 ± 0.000 | 0.004 | 0.003 ± 0.001 | <0.003 ± 0.000 | <0.003 ± 0.000 |
| Tin (Sn) | 0.05 | - | 0.16 ± 0.02 | 0.24 | 0.11 ± 0.06 | 0.16 ± 0.04 | 0.11 ± 0.02 |

**Table 7-4: Mean (±standard error) trace element concentrations in muscle of walleye, lake whitefish, and northern pike captured in Split and Stephens lakes and the Nelson River in 2004**

| Element | DL[1] | GV[2] | Walleye[3] | | Lake whitefish | | Northern pike |
|---|---|---|---|---|---|---|---|
| | | | Split L (n = 15) | Stephens L (n = 1) | Split L (n = 3) | Stephens L (n = 10) | Nelson R (n = 20) |
| Titanium (Ti) | 0.3 | - | <0.3 ± 0.0 | <0.3 | <0.3 ± 0.0 | <0.3 ± 0.0 | <0.3 ± 0.0 |
| Uranium (U) | 0.005 | - | <0.005 ± 0.000 | <0.005 | <0.005 ± 0.000 | <0.005 ± 0.000 | <0.005 ± 0.000 |
| Vanadium (V) | 0.03 | - | <0.03 ± 0.00 | <0.03 | 0.03 ± 0.01 | <0.03 ± 0.01 | <0.03 ± 0.00 |
| Zinc (Zn) | 0.3 | - | 8.2 ± 0.3 | 8.0 | 11.6 ± 2.5 | 12.2 ± 1.4 | 13.7 ± 0.6 |

1. Analytical detection limit (DL).
2. Provincial (Williamson 2002) guideline value (GV) for aquatic life tissue residues for the protection of human consumers.
3. All values are expressed as μg/g (wet weight); the number of fish analyzed is indicated in brackets.

**Table 7-5: Mean (± standard error) fork length, total weight, and age of walleye, lake whitefish, and northern pike captured for trace element analysis in Split Lake, the Nelson River between Birthday and Gull rapids, and Stephens Lake below Gull Rapids, fall 2004**

| Waterbody | Species | n | Fork Length (mm) | Weight (g) | Age (y) |
|---|---|---|---|---|---|
| Split Lake | Lake whitefish | 3 | 449 ± 25 | 1833 ± 504 | 9.0 ± 1.2 |
| | Walleye | 15 | 427 ± 10 | 920 ± 86 | 7.7 ± 0.5 |
| Nelson River | Northern pike | 20 | 637 ± 11 | 1821 ± 119 | 6.7 ± 0.4 |
| Stephens Lake | Lake whitefish | 10 | 478 ± 32 | 1915 ± 136 | 10.6 ± 1.1 |
| | Walleye | 1 | 421 | 900 | 7.0 |

**Table 7-6: Residual effects on fish quality — trace elements**

| Environmental Effect | Mitigation/Enhancement | Residual Effect |
|---|---|---|
| **Split/Clark Lake**<br>There will be no measurable effect on fish tissue concentrations of trace elements other than mercury. | N/A | None |
| **Below Clark Lake to GS**<br>No measurable increases in fish tissue concentrations of trace elements other than mercury are expected due to the construction and operation of the Keeyask GS | N/A | None |
| **Downstream of GS/ Stephens Lake**<br>There will be no measurable effect on fish tissue concentrations of trace elements other than mercury. | N/A | None |

**Table 7-7: Mean (± standard error) fork length (mm), round weight (g), condition factor (K), and age (years) of lake whitefish inspected for cysts of *Triaenophorus crassus* as part of Keeyask environmental studies from 2003–2006**

| Lake | Year | n | Length | Weight | K | Age |
|---|---|---|---|---|---|---|
| Split | 2006 | 6[a] | 510.3 ± 8.0 | 2241.7 ± 207.3 | 1.68 ± 0.08 | 10.3 ± 1.2 |
| | 2005 | 12 | 466.3 ± 9.4 | 1950.0 ± 129.7 | 1.90 ± 0.04 | 11.7 ± 0.7 |
| | 2003 | 6 | - | - | - | - |
| Gull | 2006 | 10[b] | 496.3 ± 26.5 | 2396.0 ± 549.8 | 1.89 ± 0.16 | 11.2 ± 1.1 |
| | 2004 | 12 | - | - | - | - |
| Stephens | 2006 | 17[c] | 496.7 ± 13.2 | - | - | 12.4 ± 1.2 |
| | 2005 | 12 | 498.9 ± 9.5 | 2366.7 ± 161.5 | 1.87 ± 0.05 | 11.8 ± 1.0 |
| | 2003 | 6 | - | - | - | - |
| Waskaiowaka | 2003 | 6 | - | - | - | - |

a. n=3 for length, weight, and K
b. n=4 for length, weight, and K
c. n=16 for length

**Table 7-8: Summary of information on rate of *Triaenophorus crassus* infestation (RI) and assigned grade of lake whitefish for three study area lakes and four AEA offsetting lakes**

| Lake | Year | Date[1] | Collector[2] | Fish (n) | Dressed weight (lbs) | Cysts (n) | RI | Grade[3] |
|---|---|---|---|---|---|---|---|---|
| Split | 2006 | 21 Jun–11 Oct | NSC | 6 | 211 | 0 | 0 | Export |
| | 2005 | 6–8 Oct | NSC | 12 | 39.6 | 2 | 5.1 | Export |
| | 2004 | 15 Sep | FFMC | - | 9 | 0 | 0 | Export |
| | | 18 Jun | FFMC | - | 15 | 0 | 0 | Export |
| | 2003 | 10–18 Oct | NSC | 6 | 20.3 | 0 | 0 | Export |
| | 1998 | 20 Sep | FFMC | - | 13 | 1 | 8 | Export |
| | | 18 Sep | FFMC | - | 15 | 3 | 20 | Export |
| | | 17 Sep | FFMC | - | 13 | 1 | 8 | Export |
| | | 11 Sep | FFMC | - | 13 | 4 | 31 | Export |
| | | 8 Sep | FFMC | - | 12 | 1 | 9 | Export |
| | | 3 Sep | FFMC | - | 13 | 3 | 23 | Export |
| | | 19 Jun | FFMC | - | 11 | 2 | 18 | Export |
| | 1996 | 9 Jun | FFMC | - | 21 | 0 | 0 | Export |
| | 1985 | 2 Sep | FFMC | - | 13 | 0 | 0 | Export |
| | 1966 | 1-9 Jul | MDMNR | 53 | 96.9 | 14 | 14.4 | Export |
| | 1965/66 | - | DFO | - | 234 | 29 | 12 | Export |
| | 1964/65 | - | DFO | - | 259 | 16 | 6 | Export |
| | 1963/64 | - | DFO | - | 217 | 16 | 7 | Export |
| | 1962/63 | - | DFO | - | 219 | 40 | 18 | Export |
| | 1959 | - | MDMNR | 133 | 290 | 28 | 9.6 | Export |
| Gull | 2006 | 23 Aug–3 Oct | NSC | 10 | 40.8 | 0 | 0 | Export |
| | 2004 | 18–22 Oct | NSC | 12 | 39.3 | 0 | 0 | Export |
| Stephens | 2006 | 20 Aug–26 Sep | NSC | 17 | 69.8 | 26 | 37.3 | Export |
| | 2005 | 10–11 Oct | NSC | 12 | 56.5 | 12 | 21.2 | Export |
| | 2003 | 26–30 Sep | NSC | 6 | 23.9 | 5 | 20.9 | Export |
| Atkinson | 1963/64 | - | DFO[4] | - | 169 | - | 24 | Export |
| | 1963/64 | - | DFO[5] | - | 17 | - | 106 | Other |
| Moose Nose | 1968 | 1 Jan | MDMNR | 94 | 152.7 | 302 | 197.8 | Other |
| War | 1962/63 | - | DFO[5] | - | 57 | - | 4 | Export |

**Table 7-8: Summary of information on rate of *Triaenophorus crassus* infestation (RI) and assigned grade of lake whitefish for three study area lakes and four AEA offsetting lakes**

| Lake | Year | Date[1] | Collector[2] | Fish (n) | Dressed weight (lbs) | Cysts (n) | RI | Grade[3] |
|---|---|---|---|---|---|---|---|---|
| Waskaiowaka | 2003 | 3–5 Oct | NSC | 6 | 12.9 | 11 | 85.3 | Other |
| | 1972 | June | MDMNR | 64 | 105 | 113 | 107.6 | Other |

1. Inspection date for FFMC collection, sampling date(s) for all other collections.
2. NSC = North/South Consultants Inc.; FFMC = Freshwater Fish Marketing Corporation; MDMNR= Manitoba Department of Mines and Natural Resources; DFO= Federal Department of Fisheries ([4]= sample analysed in eastern Canada; [5]= sample analysed in Manitoba).
3. Export= <50 cysts per 100 lbs of fish; Other=≥50 cysts per 100 lbs of fish

**Table 7-9: Residual effects on fish quality — *Triaenophorus crassus* infection**

| Environmental Effect | Mitigation/Enhancement | Residual Effect |
|---|---|---|
| **Split/Clark Lake**<br>There will be no measurable effect on the infection rate of lake whitefish by the cestode *T. crassus*. | N/A | None |
| **Below Clark Lake to GS**<br>Changes in habitat, zooplankton community composition, and the fish community may affect the infection rate of *T. crassus* in lake whitefish. These effects likely will change over time and will be partially compensatory. No measurable effects on *T. crassus* infection rates of lake whitefish are expected. | Depending on the perceived change in the esthetic quality of lake whitefish caused by *T. crassus* cysts by resource users, other waterbodies (that have low *T. crassus* infection rates of whitefish) could be harvested for this fish species. | None |
| **Downstream of GS/ Stephens Lake**<br>There will be no or minimal effects to the infection rate of *T. crassus* in lake whitefish in Stephens Lake. | Depending on the perceived change in the esthetic quality of lake whitefish caused by *T. crassus* cysts by resource users, other waterbodies (that have low *T. crassus* infection rates of whitefish) could be harvested for this fish species. | None |

**Table 7-10: Residual effects on fish quality — palatability**

| Environmental Effect | Mitigation/Enhancement | Residual Effect |
|---|---|---|
| **Split/Clark Lake**<br>Average palatability could be affected if sufficient fish moved from the Keeyask reservoir to comprise a significant portion of the catch. Tagging data to date suggest that only a small proportion of fish in upstream waterbodies originated from the Keeyask reach. | None | The preferences for certain fish species and for fish from particular lakes that have been expressed by members of KCN communities in the past will likely persist after the construction of the GS. |

**Table 7-10: Residual effects on fish quality — palatability**

| Environmental Effect | Mitigation/Enhancement | Residual Effect |
|---|---|---|
| **Below Clark Lake to GS** | | |
| The changes from a lotic to a mainly lentic environment and the associated shifts in the community composition and relative abundances of invertebrates and fishes have the potential to influence fish palatability via changes in diet, *etc.* “Palatability” is affected by a suite of factors. The limited available data based on “taste tests” using lake whitefish, northern pike (jackfish), and walleye (pickerel) in KCN communities indicate that statistically significant taste differences between fish from different lakes on and off system cannot be detected. However, factors other than taste from a prepared meal affect palatability. Given the subjectivity of taste perception and the complexity of factors contributing to an enjoyable culinary experience make it difficult to predict effects to palatability based on examination of changes to fish diet, *etc.* alone. | Offset fishing programs | The preferences for certain fish species and for fish from particular lakes that have been expressed by members of KCN communities in the past will likely persist after the construction of the GS. |
| Hydrocarbons or other olfactory or gustatory competent substances that are readily incorporated into fish musculature may be introduced into the aquatic environment during the construction period from site runoff/drainage, cofferdam seepage or accidental spillages. | Development and implementation of good management practices and containment of areas sensitive to spillage. | Potential for medium geographic extent, moderate, short-term effects. |

**Table 7-10: Residual effects on fish quality — palatability**

| Environmental Effect | Mitigation/Enhancement | Residual Effect |
|---|---|---|
| **Downstream of GS/ Stephens Lake**<br>Palatability could be affected by factors such as accidental hydrocarbon spills from the GS as well as other factors discussed under other areas. Potential effects will likely differ in strength between areas of Stephens Lake, diminishing with distance from the river mainstem. | Mitigation measures will be in place to reduce and localize the effects of accidental spills.<br><br>Offset fishing programs | The preferences for certain fish species and for fish from particular lakes that have been expressed by members of KCN communities in the past will likely persist after the construction of the GS. |

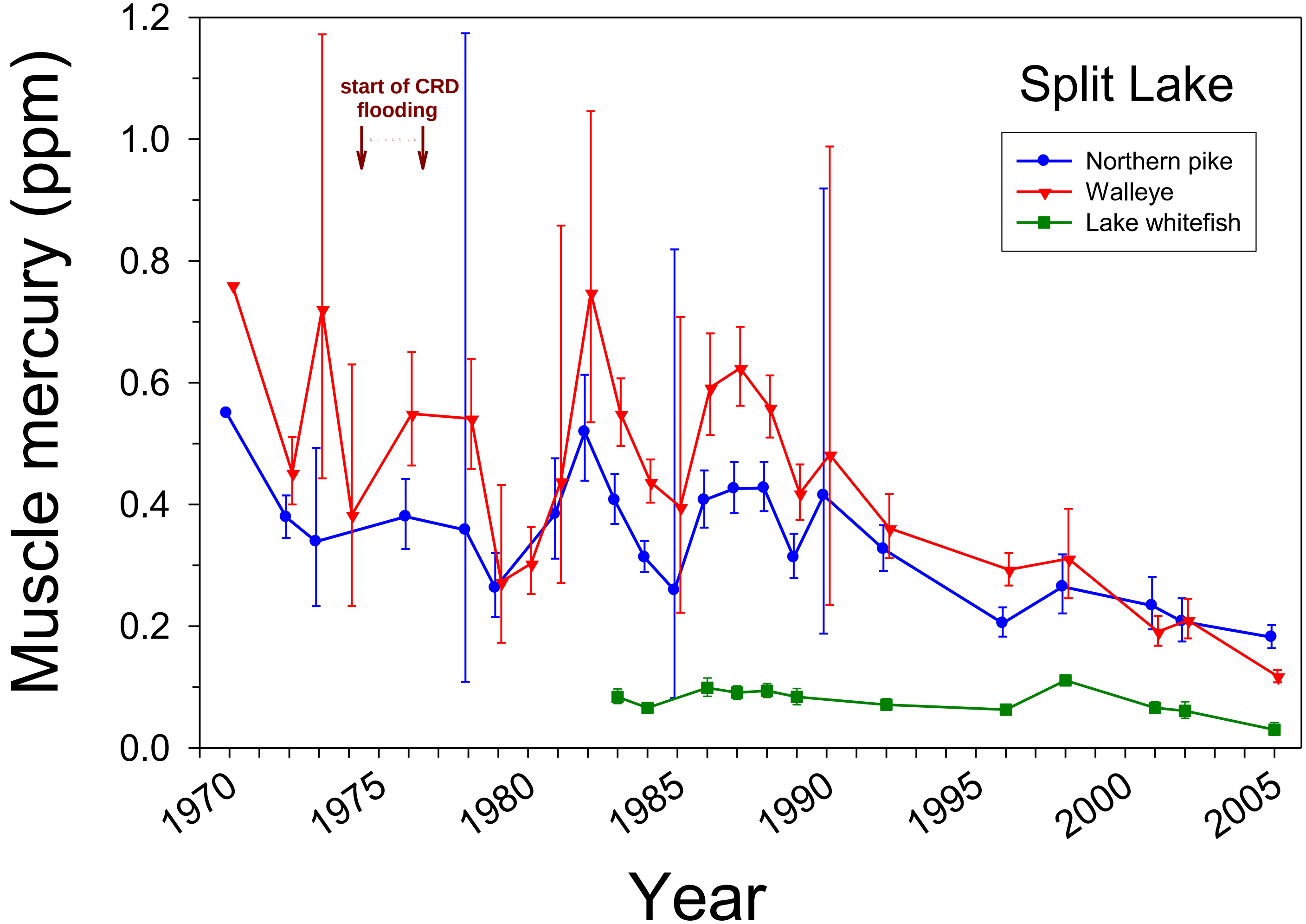



**Figure 7-1: Mean (± 95% confidence limit) standardized mercury concentrations of northern pike, walleye, and lake whitefish from Split Lake from 1970 to 2005**

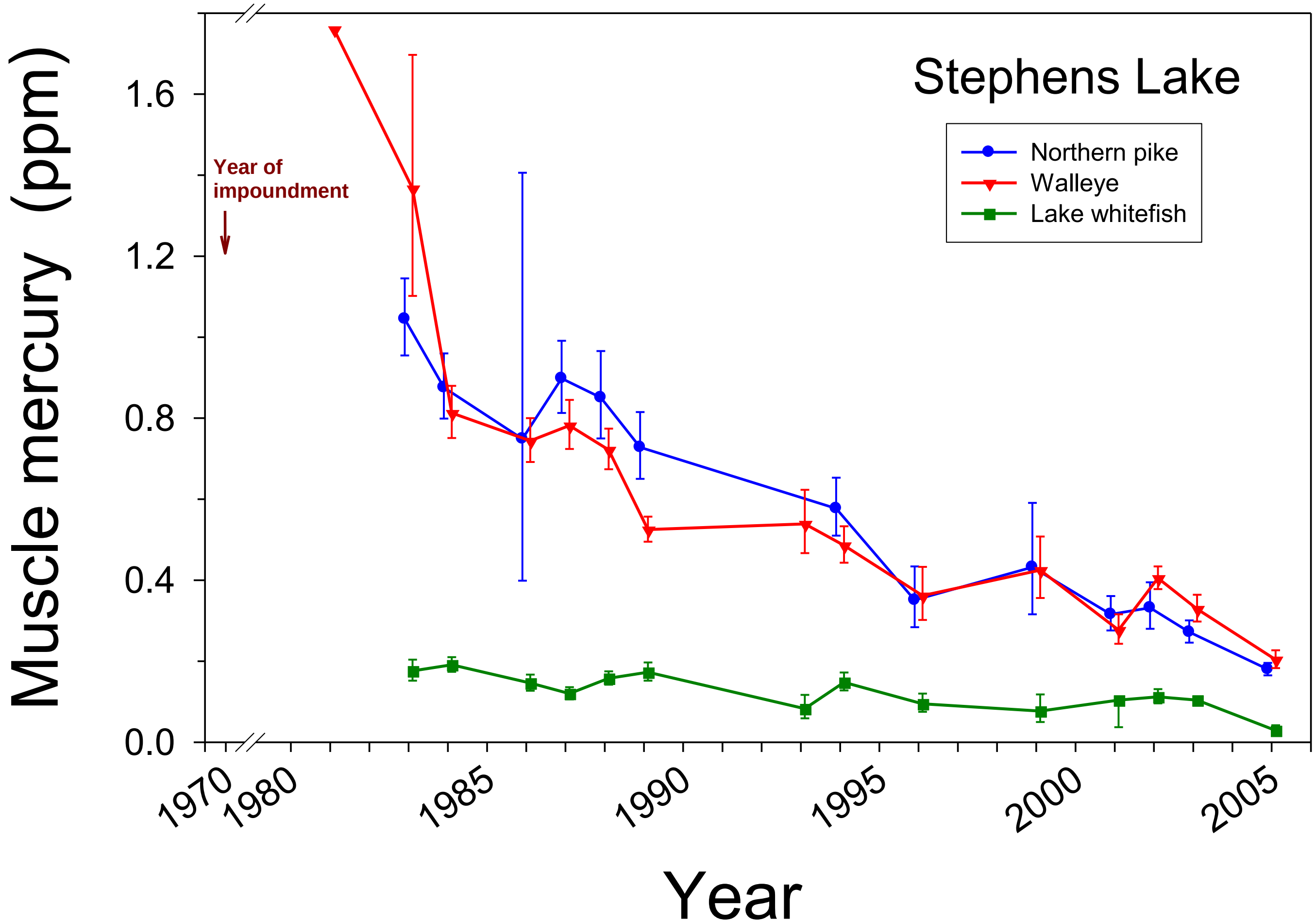
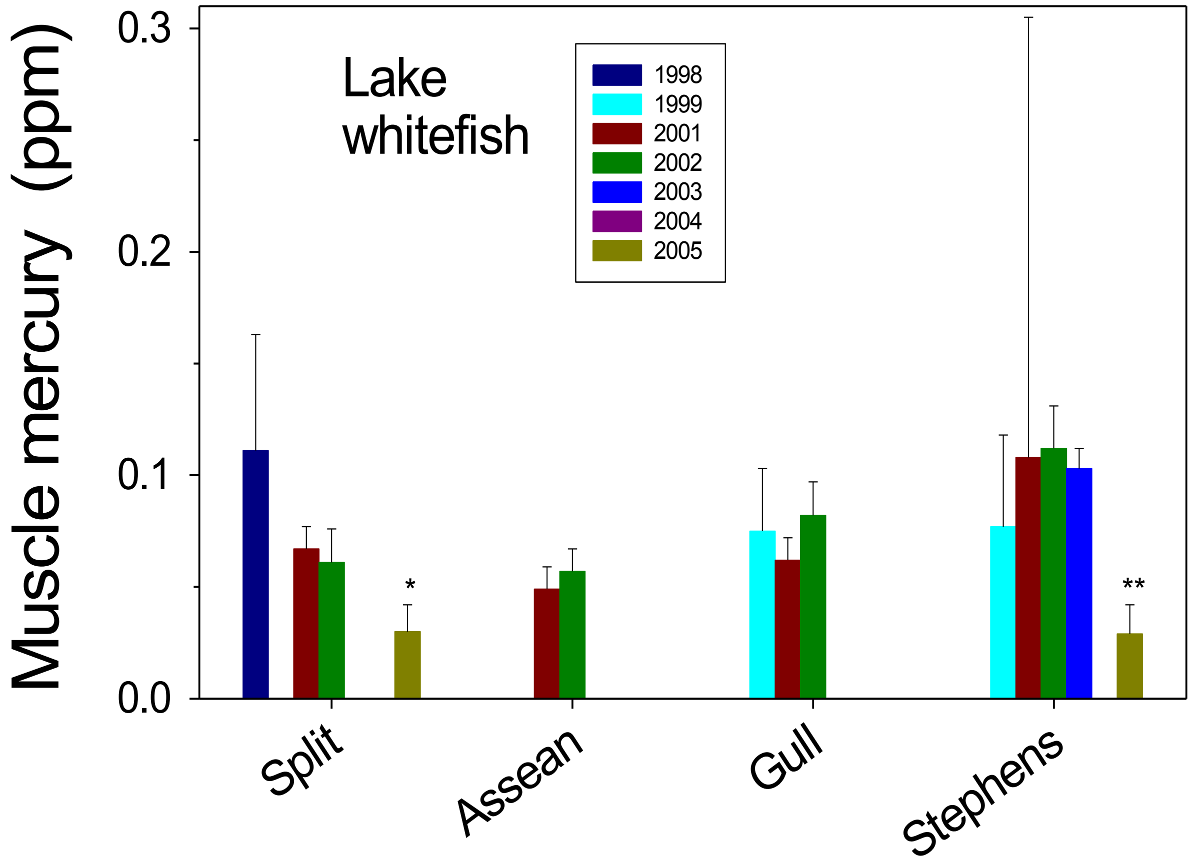
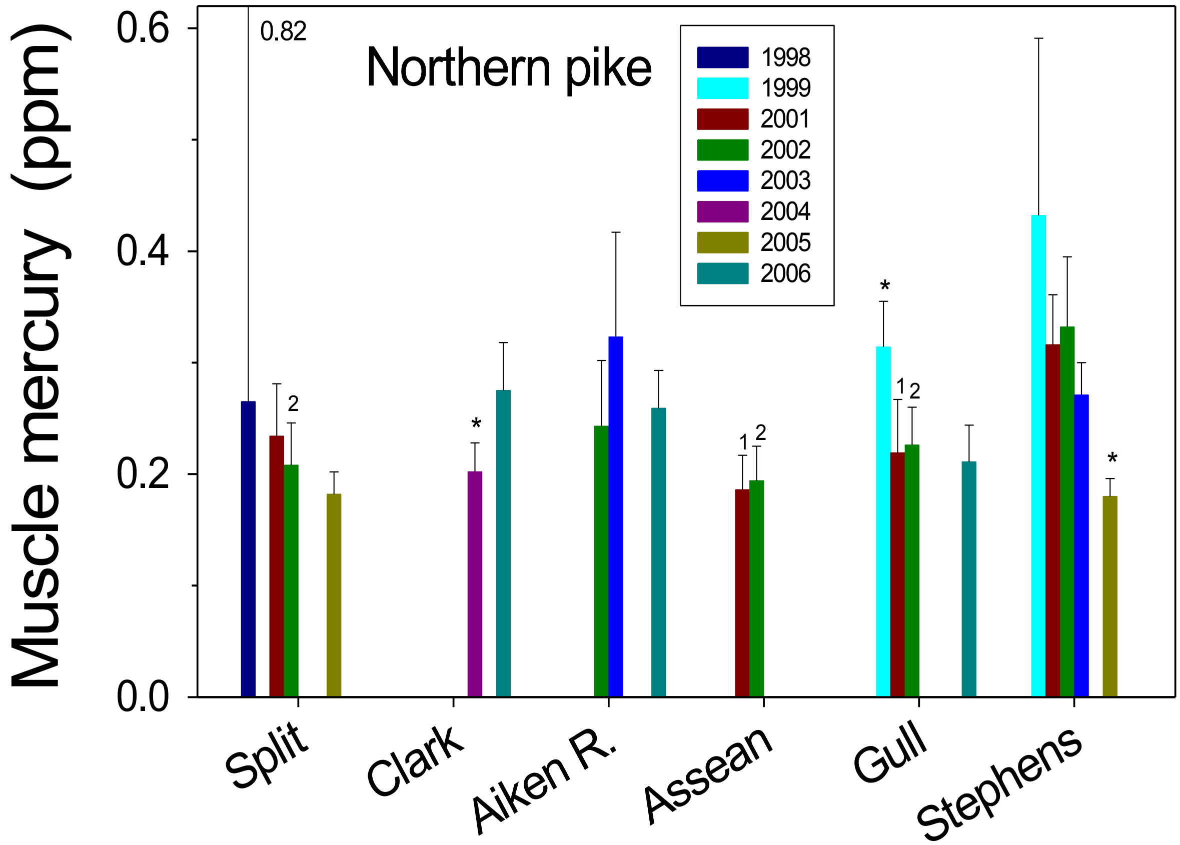



**Figure 7-2: Mean (± 95% confidence limit) standardized mercury concentrations of northern pike, walleye, and lake whitefish from Stephens Lake from 1981 to 2005. The mean without a confidence limit for walleye in 1981 is from a commercial sample**




* Significantly different from other yearly means for that waterbody.
** Significantly different from other yearly means for that waterbody except for 2001.

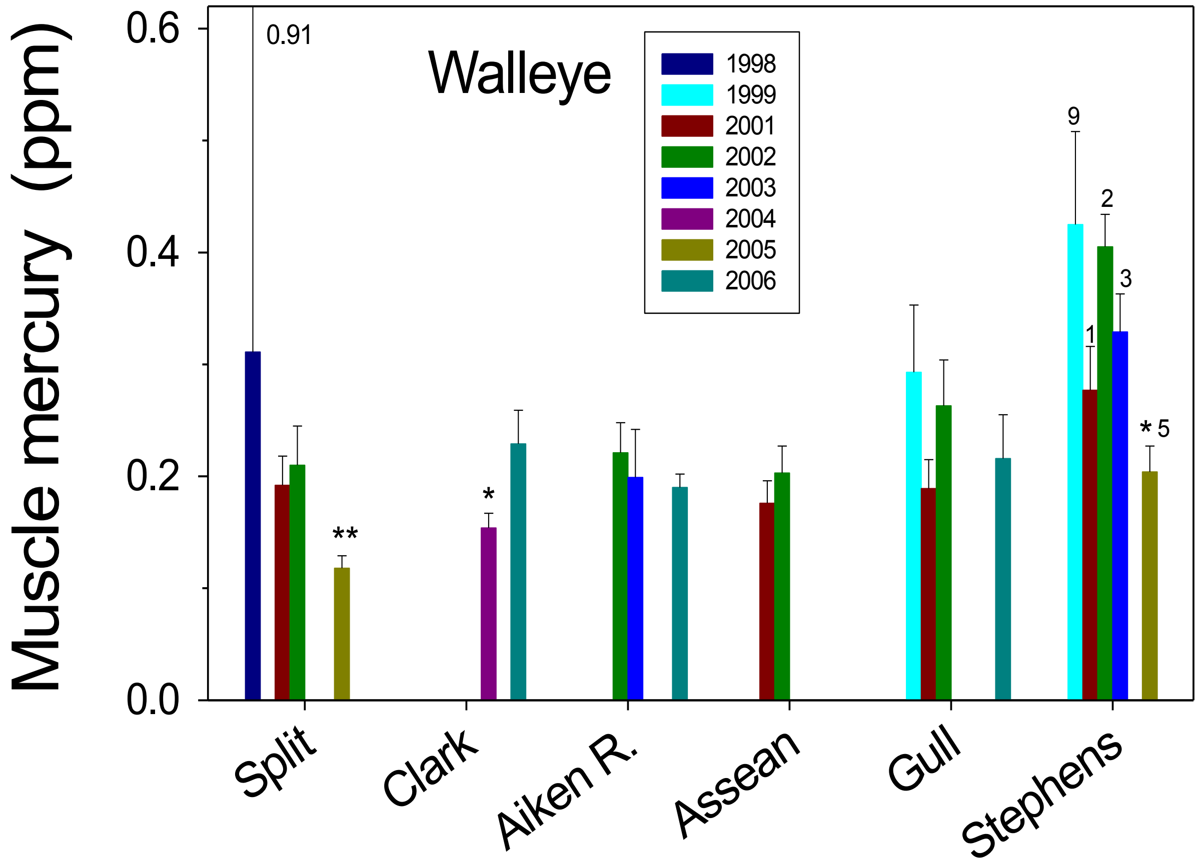
**Figure 7-3: Mean (+ upper 95% confidence limit) standardized mercury concentrations of lake whitefish from Split, Assean, Gull, and Stephens lakes in 1998–2005**




* Significantly different from other yearly means for that waterbody.
1. Significantly different from Stephens Lake in 2001.
2. Significantly different from Stephens Lake in 2002.

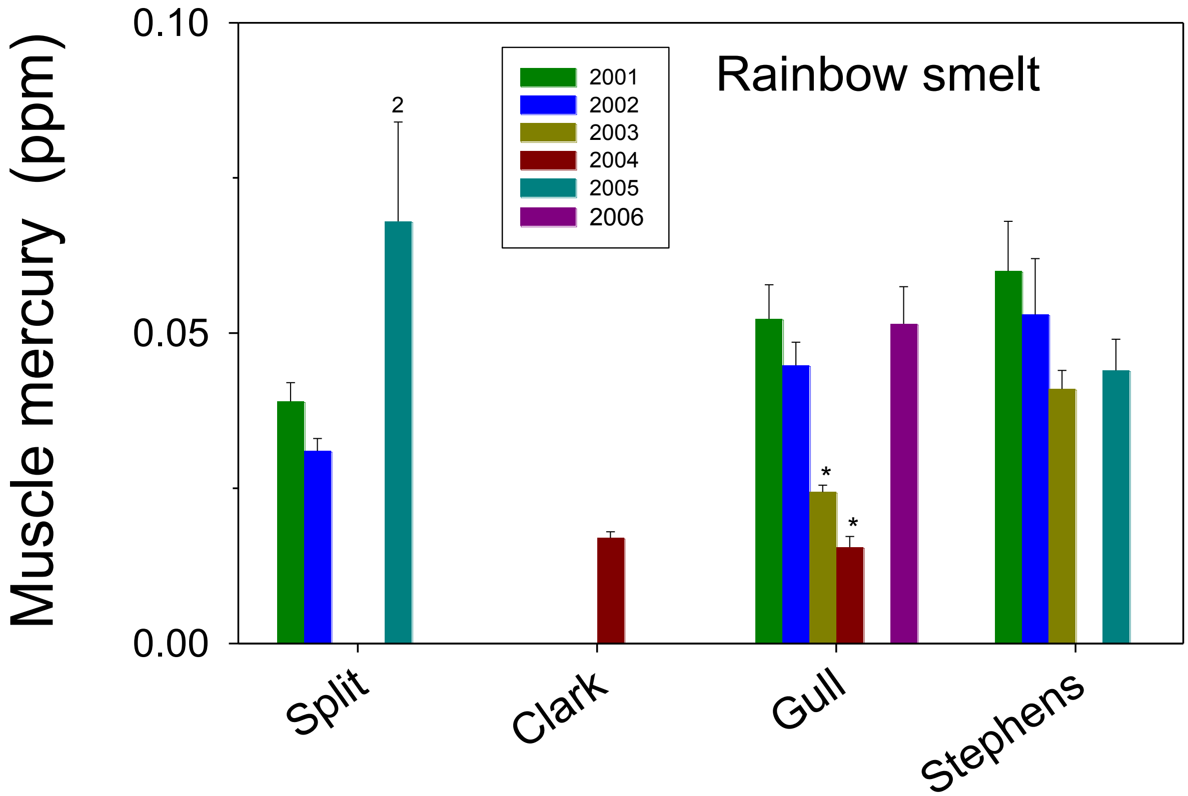
Note: The confidence limit for Split Lake in 1998 is shown numerically.

**Figure 7-4: Mean (+ upper 95% confidence limit) standardized mercury concentrations of northern pike from four waterbodies in the Split Lake area, and Gull and Stephens lakes in 1998–2006**



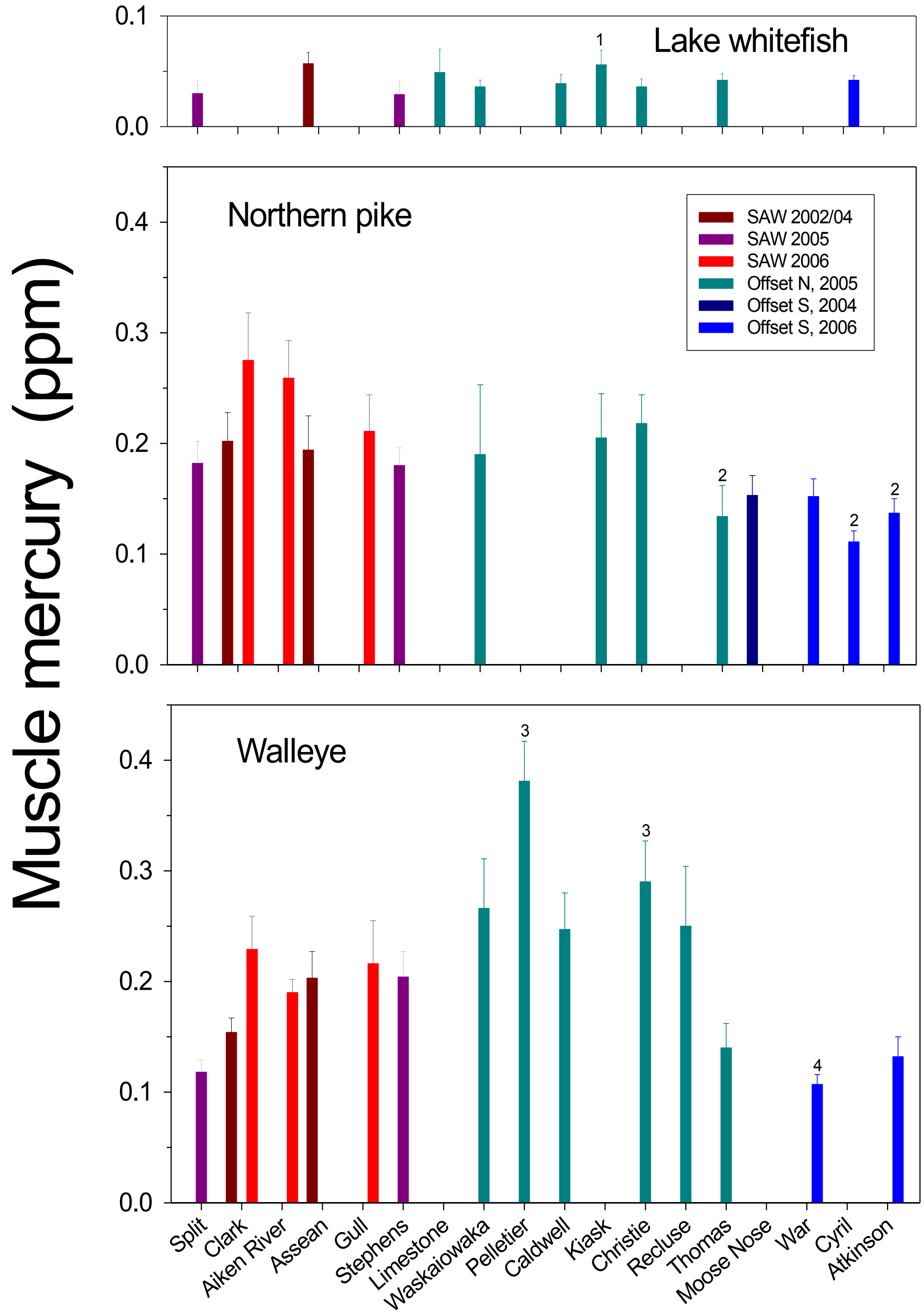

* Significantly different from other yearly means for that waterbody.
** Significantly different from other yearly means for that waterbody except for 1998.
Note: numbers above the confidence limit bar indicate that the mean is significantly different from all other waterbodies for that year. The confidence limit for Split Lake in 1998 is shown numerically.

**Figure 7-5: Mean (+ upper 95% confidence limit) standardized mercury concentrations of walleye from four waterbodies in the Split Lake area, and Gull and Stephens lakes in 1998–2006**



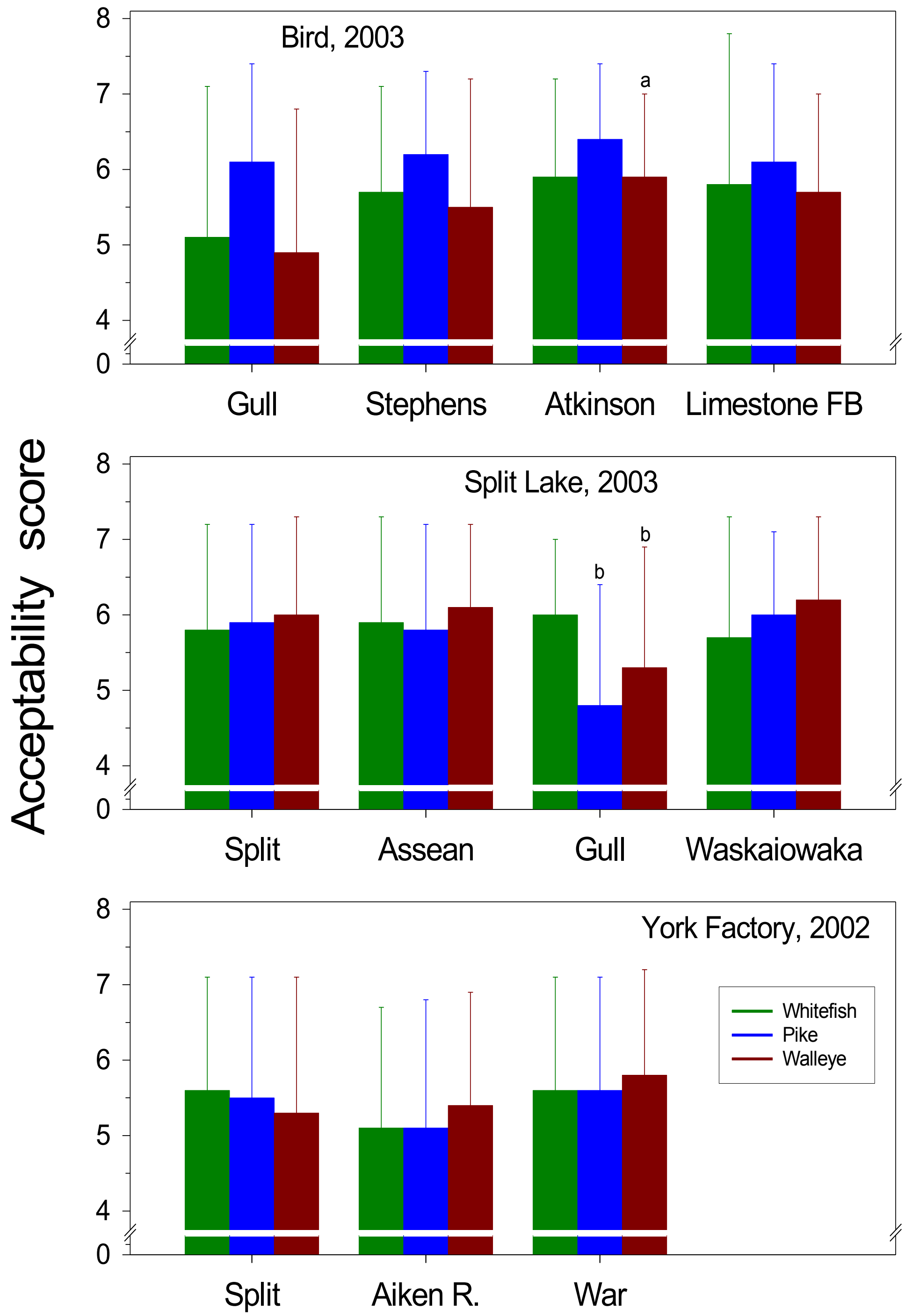

* Significantly different from other yearly means for that waterbody.
2. Significantly different from 2002

**Figure 7-6: Mean (+standard error) arithmetic mercury concentrations of rainbow smelt from two waterbodies in the Split Lake area, and Gull (mainstem locations) and Stephens lakes for 2001–2006**



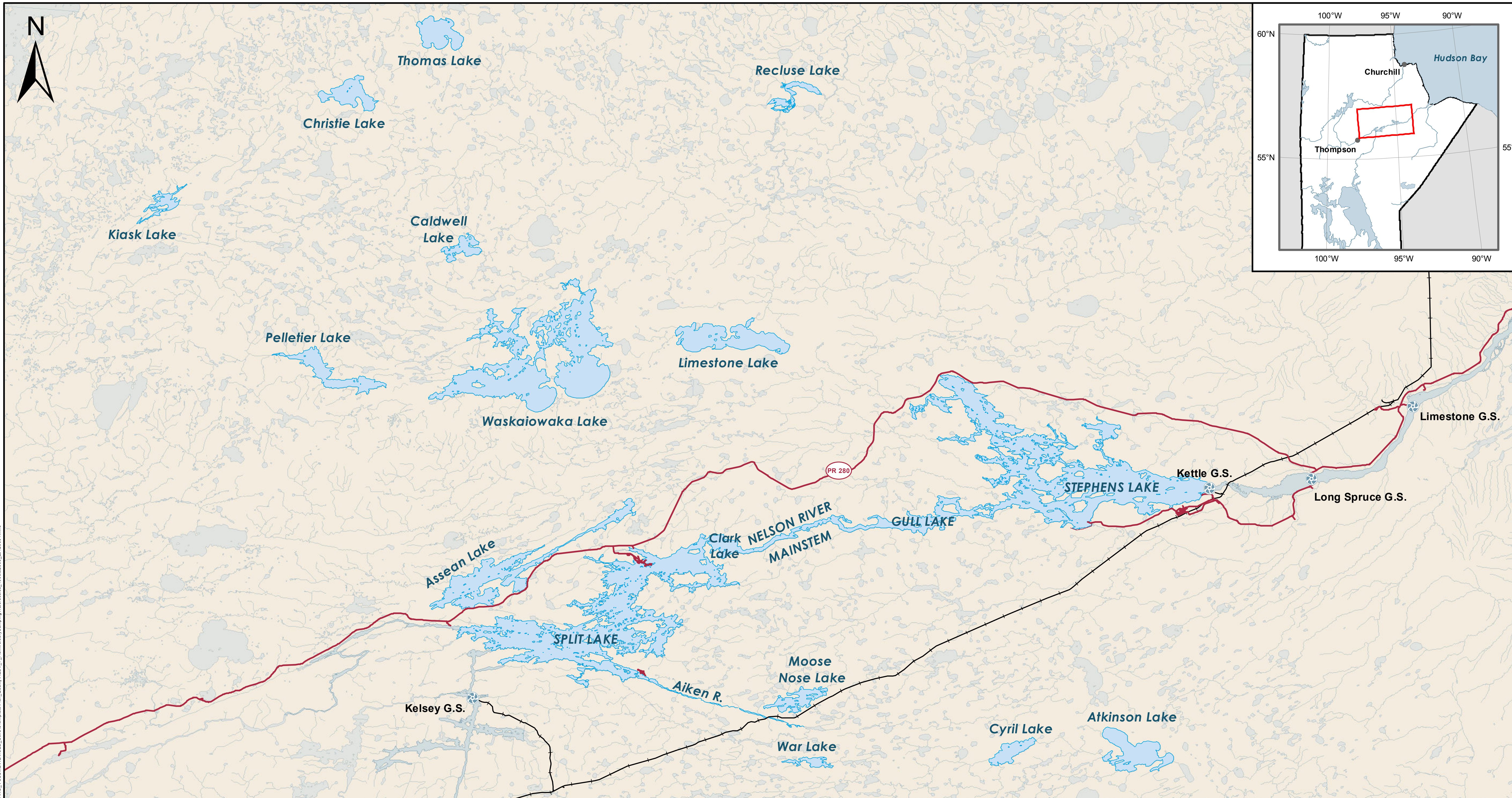

1. Significantly higher than Split and Stephens lakes.
2. Significantly lower than all study area waterbodies (SAW).
3. Significantly higher than all SAW.
4. Significantly lower than all SAW except for Split Lake.

**Figure 7-7: Mean (+ upper 95% confidence limit) standardized mercury concentrations of lake whitefish, northern pike, and walleye from study area waterbodies (SAW) and AEA offsetting lakes north (N) and south (S) of the Nelson River in 2002–2006**




a Significantly (p<0.10) higher than Gull Lake for that species.
b Significantly (p<0.05) lower than all other lakes for that species
FB= forebay (reservoir).

**Figure 7-8: Mean (+ standard deviation) acceptability scores of whitefish, pike, and walleye from Keeyask Project waterbodies tested for palatability at three First Nation communities in 2002 and 2003**




KEEYASK Hydropower Limited Partnership

0 25 50 Kilometres

0 10 20 Miles

Projection: UTM Zone 15, NAD 83
Data Source: NTS base 1:250 000

# Fish Mercury Sampling Waterbodies 1999-2006

Map 7-1

# APPENDICES

# APPENDIX 7A
# MERCURY CONCENTRATIONS IN CISCO, LARGE-BODIED WHITE SUCKER, AND LAKE STURGEON

In addition to lake whitefish, northern pike, walleye, and forage fish, which are the focus of the main body of the document, mercury data were also collected as part of the Keeyask environmental studies from cisco, white sucker, and lake sturgeon (Table 7A-1). These species were selected to obtain information on mercury concentrations in fish at lower trophic levels and/or in response to First Nation requests. Because limited data are available for these species, the results have been summarized as an appendix. Note that results for small-bodied white sucker are included in the sections on forage fish of the main document.

# 7A.1 CISCO

Mercury data for cisco were collected as part of the Keeyask environmental studies from Split, Stephens, and Assean lakes between 1998 and 2005. Current (1998–2005) arithmetic (0.03–0.10 ppm) and standardized (to a length of 300 mm; 0.04–0.11 ppm) concentrations of cisco from Split Lake are slightly more variable between years (Table 7A-2), but generally fall within a similar range as was observed historically. Historic means for four data years between 1983 and 1996 ranged from 0.06 to 0.09 ppm (unpublished analyses by North/South Consultants [NSC] based on data from the Canadian Food Inspection Agency [CFIA] database; also see Green 1990). However, all of the current means from Split Lake should be interpreted with caution because of the associated small sample sizes and the non-significant relationships between mercury concentration and fish length (Table 7A-2).

Current standardized mean concentrations in cisco from Stephens Lake (0.14 ppm) in years with adequate sample size (*i.e.*, 2002 and 2003; Table 7A-2) are higher (although mainly not significantly due to the small size of the Split Lake samples) than those of their conspecifics from Split and Assean lakes. The recent concentrations from Stephens Lake are also higher (significantly in 2002) than the value of 0.088 (95% CL = 0.073–0.107) obtained in 1993, the last data year prior to the Keeyask Environmental studies (CFIA database and unpublished analyses by NSC). The mean concentrations in 1993 was also significantly lower than the value of 0.298 ppm (0.245–0.362 ppm) measured in 1984, which represents the highest mean concentration measured in cisco from Stephens Lake, including additional samples taken in 1983 and 1986 (CFIA database and unpublished analyses by NSC; also see Green 1990).

Standardized (but not arithmetic) mean mercury concentration differed significantly between cisco from Assean Lake in 2001 (when the correlation between length and mercury concentration was not significant) and 2002 (Table 7A-2). The value for 2002 also was significantly higher than the mean of 0.064 ppm measured in 1981, the only other data year from Assean Lake (CFIA database and unpublished analyses by NSC). In all three years, mercury concentrations were similar to the range of means obtained from Split Lake in 1983–2005. Mercury concentrations in cisco from Gull Lake could not been meaningfully assessed because of small fish sample sizes (Table 7A-2) and historic data are unavailable.

When considering mercury concentrations of individual fish, 91% of all 163 cisco analyzed as part of the Keeyask environmental studies had mercury concentrations below 0.2 ppm (Table 7A-3), the threshold for safe consumption for persons eating large quantities of fish (Wheatley 1979).

# 7A.2 WHITE SUCKER

The results of mercury analyses on small-bodied white sucker (61–130 mm length) are included in the main document with forage fish. Large-bodied white sucker (Table 7A-1) were collected as part of the Keeyask environmental studies in two years from one waterbody. White sucker from the Aiken River had standardized (to a length of 400 mm) mean mercury concentrations of less than 0.1 ppm in 2002 and 2003 (Table 7A-2). These values are slightly lower than the standardized concentrations of 0.13–0.17 ppm recorded for white sucker (mean length of 351–401 mm) from Split Lake in 1983, 1984, and 1986, and the 0.12–0.20 ppm measured in conspecifics (mean length of 325–379 mm) from Stephens Lake for the same three years (unpublished analyses by NSC based on data from the CFIA database;; also see Green 1990). In 1993 (0.094 ppm) and 1996 (0.084 ppm) mercury concentrations of white sucker in Stephens Lake were significantly lower than in most previous years and were almost identical to the current levels measured in fish from the Aiken River.

Although the recent mean mercury concentrations of white sucker from the Aiken River are less than half the 0.2 ppm threshold for safe consumption for persons eating large quantities of fish (Wheatley 1979), 29% of the 51 individuals exceeded this limit, with maximum values reaching 0.36 ppm (Table 7A-3).

# 7A.3 LAKE STURGEON

Lake sturgeon were collected as part of the Keeyask environmental studies in two years from one waterbody. In 2002 and 2004, three and 10 lake sturgeon, respectively, captured by domestic harvesters in Gull Lake that measured between 1,050 and 1,439 mm total length were sampled for mercury. Arithmetic mean concentrations (the relationship between total length and mercury concentrations was not significant) were 0.17 ppm in 2002 and 0.21 ppm in 2004 (Table 7A-2). Of the 13 sturgeon, four (31%) had higher concentrations than the 0.2 ppm threshold for safe consumption for persons eating large quantities of fish (Wheatley 1979). A maximum concentration of 0.67 ppm was measured in the largest fish (Table 7A-3).

To our knowledge, no other measurements exist of mercury levels in lake sturgeon from waterbodies within the study area to provide a historical context. The only other data on sturgeon mercury concentrations from the lower Nelson River are from northeast of the Limestone GS. In 2003, seven lake sturgeon from two mainstem reaches of the river were analyzed for mercury. The fish measured between 725 and 1,200 mm (mean = 841 mm) and had an arithmetic mean concentration of 0.19 ppm (NSC *unpubl. data*).

# 7A.4 REFERENCES

## 7A.4.1 LITERATURE CITED

Green, D.J. 1990. Updated summary of fish mercury data collected from six lakes on the Rat Burntwood and Nelson River systems, 1983 1989. MS Report No. 90-10, Manitoba Natural Resources, Fisheries Branch. 277 pp.

Wheatley, B. 1979. Methylmercury in Canada; exposure of Indian and Inuit residents to methylmercury in the Canadian environment. Health and Welfare Canada, Medical Service Branch, Ottawa, ON. 200 pp.

## 7A.4.2 PERSONAL COMMUNICATIONS

Canadian Food Inspection Agency (CFIA) 2009. Data base on contaminants in freshwater fish from Alberta, Saskatchewan, Manitoba, and Ontario; managed since 2006 by John Hoeve, Fish, Seafood and Production Division, Canadian Food Inspection Agency, Ottawa, ON. Email and telephone correspondence with Wolfgang Jansen, North/South Consultants Inc., Winnipeg, MB, August 26, 2009.

**Table 7A-1: Mean (± standard error [SE]) fork length, round weight, and age of cisco, white sucker, and lake sturgeon from study area waterbodies, 1998-2005**

| Species | Waterbody | Year | Length (mm) | n[1] | Weight (g) | n | Age (years) | n |
|---|---|---|---|---|---|---|---|---|
| Cisco | Split Lake | 1998 | 256.7 ± 8.2 | 24 | 267.9 ± 28.0 | 24 | 5.5 ± 0.3 | 19 |
| | | 2001 | 272.4 ± 41.7 | 7 | 463.4 ± 153.3 | 7 | 5.8 ± 1.6 | 6 |
| | | 2002 | 251.5 ± 24.6 | 11 | 320.6 ± 88.8 | 11 | 5.6 ± 0.9 | 10 |
| | | 2005 | 243.3 ± 45.7 | 4 | 358.3 ± 188.4 | 4 | 5.0 ± 0.7 | 4 |
| | Gull Lake | 1999 | 242.5 ± 27.5 | 2 | 187.5 ± 112.5 | 2 | 3.5 ± 0.5 | 2 |
| | | 2001 | 336.5 ± 38.5 | 2 | 662.5 ± 237.5 | 2 | 6.0 ± 1.0 | 2 |
| | | 2002 | 216.3 ± 50.6 | 4 | 239.3 ± 179.2 | 4 | - | 0 |
| | Stephens Lake | 2002 | 256.1 ± 11.4 | 30 | 325.9 ± 47.3 | 28 | 4.9 ± 0.4 | 29 |
| | | 2003 | 260.6 ± 15.1 | 24 | 486.1 ± 51.4 | 18 | 6.9 ± 0.7 | 24 |
| | | 2005 | 355 | 1 | 800 | 1 | 12 | 1 |
| | Assean Lake | 2001 | 245.7 ± 15.1 | 24 | 261.5 ± 49.2 | 24 | 3.7 ± 0.4 | 24 |
| | | 2002 | 208.4 ± 16.2 | 28 | 198.7 ± 45.7 | 26 | 4.4 ± 0.5 | 28 |
| White sucker | Aiken River | 2002 | 441.3 ± 9.6 | 36 | 1359.8 ± 116.8 | 28 | 10.0 ± 0.7 | 33 |
| | | 2003 | 402.4 ± 11.8 | 15 | 896.2 ± 74.1 | 13 | 7.9 ± 1.2 | 14 |
| Lake sturgeon | Gull Lake | 2002 | 1263.5* ± 139.0 | 2 | - | 0 | - | 0 |
| | | 2004 | 1290.5* ± 29.0 | 10 | 18 000.0 ± 2500 | 2 | 26.8 ± 2.0 | 10 |

* Total length.
1. Number of fish measured.

**Table 7A-2: Mean arithmetic (± standard error [SE]) and standardized (± 95% CL) mercury concentration (ppm) of cisco, large-bodied white sucker, and lake sturgeon from study area waterbodies, 1998–2005**

| Species | Waterbody | Year | n[1] | Arithmetic | SE | Standard | 95% CL |
|---|---|---|---|---|---|---|---|
| Cisco | Split Lake | 1998 | 24 | 0.096 | 0.008 | 0.105* | 0.046–0.240 |
| | | 2001 | 7 | 0.073 | 0.014 | 0.072* | 0.039–0.132 |
| | | 2002 | 11 | 0.078 | 0.015 | 0.080* | 0.037–0.174 |
| | | 2005 | 4 | 0.033 | 0.007 | 0.039* | 0.016–0.097 |
| | Assean Lake | 2001 | 24 | 0.067 | 0.004 | 0.069* | 0.058–0.083 |
| | | 2002 | 28 | 0.073 | 0.008 | 0.114 | 0.101–0.129 |
| | Gull Lake | 1999 | 2 | 0.123 | 0.029 | 0.190* | - |
| | | 2001 | 2 | 0.070 | 0.008 | 0.062* | - |
| | | 2002 | 4 | 0.118 | 0.054 | 0.175* | 0.023–1.330 |
| | Stephens Lake | 1999 | 2 | 0.057 | 0.002 | 0.066 | - |
| | | 2002 | 30 | 0.132 | 0.012 | 0.138 | 0.115–0.167 |
| | | 2003 | 24 | 0.139 | 0.016 | 0.139 | 0.105–0.184 |
| | | 2005 | 1 | 0.071 | - | - | - |
| White sucker | Aiken River[1] | 2002 | 36 | 0.161 | 0.014 | 0.094 | 0.081–0.110 |
| | | 2003 | 15 | 0.109 | 0.019 | 0.087 | 0.068–0.111 |
| Lake sturgeon | Gull Lake | 2002 | 3 | 0.166 | 0.033 | - | - |
| | | 2004 | 10 | 0.207 | 0.060 | 0.161* | 0.096–0.270 |

* The relationship between mercury concentration and fish length was not significant.
1. Number of fish measured.
2. At York Landing.

**Table 7A-3: Percentage of cisco, large-bodied white sucker, and lake sturgeon from study area waterbodies in 1998–2006 with mercury concentrations of ≥ 0.2 ppm or ≥ 0.5 ppm**

| Species | Waterbody | Year | n[1] | ≥ 0.2 ppm | % | ≥ 0.5 ppm | % | Max[2] (ppm) |
|---|---|---|---|---|---|---|---|---|
| Cisco | Split Lake | 1998 | 24 | 1 | 4.2 | 0 | 0.0 | 0.205 |
| | | 2001 | 7 | 0 | 0.0 | 0 | 0.0 | 0.130 |
| | | 2002 | 11 | 0 | 0.0 | 0 | 0.0 | 0.178 |
| | | 2005 | 4 | 0 | 0.0 | 0 | 0.0 | 0.044 |
| | Assean Lake | 2001 | 24 | 0 | 0.0 | 0 | 0.0 | 0.115 |
| | | 2002 | 28 | 1 | 3.6 | 0 | 0.0 | 0.213 |
| | Gull Lake | 1999 | 2 | 0 | 0.0 | 0 | 0.0 | 0.150 |
| | | 2001 | 2 | 0 | 0.0 | 0 | 0.0 | 0.078 |
| | | 2002 | 4 | 1 | 25.0 | 0 | 0.0 | 0.277 |
| | Stephens Lake | 1999 | 2 | 0 | 0.0 | 0 | 0.0 | 0.058 |
| | | 2002 | 30 | 3 | 10.0 | 0 | 0.0 | 0.367 |
| | | 2003 | 24 | 9 | 37.5 | 0 | 0.0 | 0.289 |
| | | 2005 | 1 | 0 | 0.0 | 0 | 0.0 | 0.071 |
| White sucker | Aiken River[3] | 2002 | 36 | 12 | 36.3 | 0 | 0.0 | 0.355 |
| | | 2003 | 15 | 3 | 20.0 | 0 | 0.0 | 0.227 |
| Lake sturgeon | Gull Lake | 2002 | 3 | 1 | 33.3 | 0 | 0.0 | 0.205 |
| | | 2004 | 10 | 3 | 30.0 | 1 | 10.0 | 0.667 |

1. Number of fish measured.
2. Maximum mercury concentration.
3. At York Landing.

# APPENDIX 7B
# METHODS FOR FISH SAMPLING AND AGEING, TISSUE MERCURY ANALYSIS, AND DATA TREATMENT

# 7B.1 FISH COLLECTION

Fish for mercury analysis were collected from several sites at each waterbody over 1-3 week periods usually between August and October of a sampling year. Table 7B-1 summarizes fish collection information for all study area lakes and the AEA offsetting lakes.

Fish were mostly sampled as part of an experimental, standard gang index gillnetting program (Table 7B-1). These programs used gillnet gangs of six 25 yard (22.9 m) long by 6 feet (1.8 m) deep panels 1.5, 2, 3, 3.75, 4.25, and 5 inch ("; 38, 51, 76, 95, 108, and 127 mm) stretched mesh made out of twisted nylon thread, which were set overnight for approximately 24 hours (h). Forage fish were captured mainly with small mesh index gillnets (*i.e.,* Swedish gill nets). These nets were primarily used to catch rainbow smelt in overnight sets and consisted of three 10 metres (m) long by 1.8 m deep twisted nylon panels of 16, 20, and 25 mm stretched mesh. Some of the forage fish at the backwater and mainstem sites on the Nelson River were captured using a 17 m long and 1.4 m deep seine net with 4 mm mesh that was set from shore.

Some fish for mercury analysis were obtained during spring and fall tagging programs that used gill nets of 3.75 (95 mm), 4.25 (108 mm), and 5 " (127 mm) twisted monofilament stretched mesh set for 1–4 h during the day. Using the same type of nets, fish were specifically collected for mercury analysis (fish mercury collection in Table 7B-1) at Moose Nose Lake in 2004 and the other AEA offsetting lakes in 2005 and 2006 during 3–8 h sets at two to five different locations within a lake.

To be consistent with the methods described by the MMMR (see Table 1 in Strange and Bodaly 1999 for an example of numbers of fish and stratification of length classes), a broad size range of fish was collected whenever possible. Upon capture, individuals from commercially important species were measured for fork length (± 1 mm) and total weight (± 25 g, pan balance for most fish greater than 200 g; ± 1 g, digital balance for most fish less than 200 g), examined internally to determine sex and maturity, and bony structures were removed for age analysis. Dorsal spines were taken from walleye, cleithra were collected from northern pike, and a portion of the pelvic fin was removed from cisco, lake whitefish, and white sucker. In addition to the pelvic fin rays, otoliths were removed and used for aging cisco and lake whitefish. A sample for mercury analysis of axial muscle (fillet) weighing approximately 10 g was removed from each fish just anterior to the caudal (tail) fin. The muscle with skin attached was placed in a mercury-free plastic bag and stored on ice until it could be frozen. Forage fish were not measured and dissected in the field but placed individually into labelled mercury-free plastic bags and stored on ice before freezing. All frozen tissue samples and whole fish were shipped to the DFO Freshwater Institute in Winnipeg for mercury analysis.

# 7B.2 LABORATORY DETERMINATIONS

A subsample of approximately 0.2 g was removed from the middle of each 10 g muscle sample after slicing away the skin on one side and a thin outside layer on the other side. This procedure ensured that the percentage of water in the subsample was representative of the original sample taken from the fish. A similar sized muscle sample free of skin and bones was obtained from each forage fish.

Mercury analysis was performed using a modified hot block method described by Hendzel and Jamieson (1976) followed by cold vapour atomic absorption spectroscopy. Two samples of five different National Research Council of Canada reference materials were typically analyzed with each sample run. With one exception (mussel in 2000), yearly mean mercury concentrations obtained from the mussel and lobster hepatopancreas reference materials were within 7% of the mean certified value (Table 7B-2). Mean concentrations from sample runs of reference materials with mercury levels exceeding 2 ppm (*i.e.*, DOLT and DORM) were 5-16% lower than the certified concentrations (Table 7B-2).

Fork length (±1 mm), weight (±0.1 g), and sex (if possible) of forage fish were obtained from thawed fish prior to mercury analysis. After removing the muscle sample, bony structures were removed for age analysis of all individuals, or from selected individuals from length stratified groups of fish. Part of the pectoral fin including the first three to four fin rays were taken from white sucker, otoliths were collected from emerald and spottail shiner, rainbow smelt, and trout-perch, and dorsal spines were used for yellow perch.

Dried ageing structures of all fish were prepared and analyzed using a variety of techniques (Mackay *et al.* 1990). Fin rays or spines were coated in epoxy and sectioned with a Struers microtome saw. Sections were then fixed on glass slides with Cytoseal 280 and fish ages were determined by examining the slides with a Wild M3 dissecting microscope. Otoliths were slightly polished on a whetstone, immersed in synthetic wintergreen oil, and viewed with a dissecting microscope. Cleithra were cleaned and examined under reflected light.

# 7B.3 DATA ANALYSIS

A condition factor (K) was calculated for each fish as:

$K= (W / (L/10)^3) \times 100)$

where: W = round weight (g); and

L = fork length (mm).

Fish obtained from yearly samples from a group of lakes will invariably differ in mean size between years and lakes. Because fish accumulate mercury over their life time, so that older and, normally, larger individuals have higher levels than younger, smaller fish, mean mercury concentrations have been standardized under the MMMR program to facilitate comparisons between samples of fish from the same location or between samples of fish from different waterbodies over time (Jansen and Strange 2007). The standard lengths for cisco, lake whitefish, northern pike, and walleye are presented in Table 7B-3 together with standard lengths for the other fish species analyzed for mercury during the Keeyask environmental studies. For species without established standard lengths, a length was chosen that approximated the mean size of individuals captured as part of the Keeyask environmental studies and from other northern Manitoba waterbodies between 1998 and 2002.

In addition to arithmetic means, standardized mean mercury concentrations were calculated from unique regression equations for each species and lake based on the analysis of logarithmic transformations of muscle mercury concentration and fork lengths using the following relationship:

$$\text{Log}_{10}[\text{Hg}] = a + b\,(\text{Log}_{10}L)$$

where [Hg] = muscle mercury concentration (µg/g),

L = fork length (mm),

a = Y-intercept (constant), and

b = slope of the regression line (coefficient)

To present data in more familiar units, all standardized means and their confidence limits were retransformed to arithmetic values.

Differences in mean fish length among lakes or between habitat types, and differences in arithmetic mean mercury concentrations were ascertained employing one-way and two-way analysis of variance (ANOVA). If F-values were significant, differences between individual means were confirmed by Holm-Sidak's pairwise multiple comparison tests. If normality of data distribution or equality of variances could not be achieved by logarithmic transformation of the data, Kruskal-Wallis one way ANOVA on ranks was performed. In all cases, significance was established at P greater than or equal to 0.05. Differences in standardized mean mercury concentrations between lakes or years were established if the 95% CL of two means did not overlap. Statistical analyses were run using Sigma Stat V. 3.01 (SPSS Inc. 2003) and SAS for Windows V. 8 (SAS 1999) software.

# 7B.4 REFERENCES

## 7.B.4.1 LITERATURE CITED

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**Table 7B-1: Waterbody with number of sites, sampling year, date and program, and fish species analyzed for mercury concentrations under the Keeyask Project**

| Waterbody | Year | Date | Sites (n[1]) | Program | Species[2] |
|---|---|---|---|---|---|
| Split Lake | 2001 | 15–26 Aug | 16 | Index gillnetting | Cisco, whitefish, pike, walleye, smelt |
| | 2002 | 13–24 Aug / 4–10 Oct | 18 | Index gillnetting, fish mercury collection | Cisco, whitefish, pike, walleye, smelt |
| | 2005 | 20–23 Aug / 6–9 Oct | 14 | Index gillnetting, fish mercury collection | Cisco, whitefish, pike, walleye, smelt |
| Clark Lake | 2004 | 17–21 Aug | 5 | Index gillnetting | Pike, walleye, smelt |
| | 2006 | 13–23 Jun | 7 | Fish mercury collection | Pike, walleye |
| Gull Lake | 1999 | 05–10 Oct | 11 | Index gillnetting | Cisco, whitefish, pike, walleye |
| | 2001 | 15–25 Aug | 17 | Index gillnetting | Cisco, whitefish, pike, walleye, smelt |
| | 2002 | 6–14 Aug | 17 | Index gillnetting; commercial fishing | Cisco, whitefish, pike, walleye, smelt; sturgeon |
| | 2003 | 25 July–22 Aug | 12 | Forage fish gillnetting | Smelt, Sp shiner, Em shiner, T perch, Y perch |
| | 2004 | 11 Aug – 05 Sep | 9 | Forage fish gillnetting [3]; commercial fishing | W sucker, smelt, Sp shiner, Em shiner, T perch, Y perch; sturgeon |
| | 2006 | 31 May–30 Jun / 18–27 Aug | 10<br>8 | Fish mercury collection | Pike, walleye, smelt |
| Stephens Lake | 2001 | 25–27 Aug / 29 Sep–11 Oct | 13 | Index gillnetting, fall tagging | Whitefish, pike, walleye, smelt |

**Table 7B-1: Waterbody with number of sites, sampling year, date and program, and fish species analyzed for mercury concentrations under the Keeyask Project**

| Waterbody | Year | Date | Sites (n[1]) | Program | Species[2] |
|---|---|---|---|---|---|
| Stephens Lake (Continued) | 2002 | 24 July–Aug 8 / 2–5 Oct | 30 | Standard gang & small mesh index gillnetting, fall tagging | Cisco, whitefish, pike, walleye, R smelt |
| | 2003 | 23 July–5 Aug | 42 | Index gillnetting, forage fish gillnetting | Cisco, whitefish, pike, walleye, smelt, Sp shiner, Em shiner, T perch, Y perch |
| | 2005 | 31 Aug–5 Sept / 26–29 Sept | 13 | Index gillnetting, fish mercury collection | Cisco, whitefish, pike, walleye, smelt |
| Aiken River | 2002 | 31 May–9 June | 3 | Spring tagging | Pike, walleye, W sucker |
| | 2003 | 16–20 May | 5 | Spring tagging | Pike, walleye, W sucker |
| Aiken River, York Landing | 2006 | 20–21 May | 5 | Fish mercury collection | Pike, walleye |
| Aiken River, Ilford | 2006 | 17–19 May | 9 | Fish mercury collection | Pike, walleye |
| Assean Lake | 2001 | 29 May–3 Sept | 10 | Index gillnetting | Cisco, whitefish, pike, walleye |
| | 2002 | 20–25 Aug | 11 | Index gillnetting | Cisco, whitefish, pike, walleye |
| NR mainstem | 2006 | 31 May–24 Jun | 8 | Index gillnetting, fish mercury collection | Pike, walleye |
| **AEA Offsetting Lakes** | | | | | |
| - Atkinson Lake | 2006 | 2–5 Aug | 5 | Fish mercury collection | Pike, walleye |
| - Caldwell Lake | 2005 | 8–9 Aug | 4 | Fish mercury collection | Whitefish, walleye |
| - Cyril Lake | 2006 | 5–7 Aug | 2 | Fish mercury collection | Whitefish, pike |

**Table 7B-1: Waterbody with number of sites, sampling year, date and program, and fish species analyzed for mercury concentrations under the Keeyask Project**

| Waterbody | Year | Date | Sites (n[1]) | Program | Species[2] |
|---|---|---|---|---|---|
| - Christie Lake | 2005 | 14–15 Aug | 4 | Fish mercury collection | Whitefish, pike |
| - Kiask Lake | 2005 | 11–12 Aug | 4 | Fish mercury collection | Whitefish, pike, lake trout |
| - Limestone Lake | 2005 | 8–9 Aug | 4 | Fish mercury collection | Whitefish |
| - Moose Nose Lake | 2004 | 09–11 Sep | 4 | Fish mercury collection | Pike |
| - Pelletier Lake | 2005 | 3–4 Aug | 2 | Fish mercury collection | Walleye |
| - Recluse Lake | 2005 | 10–12 Aug | 5 | Fish mercury collection | Walleye |
| - Thomas Lake | 2005 | 3–4 Aug | 3 | Fish mercury collection | Whitefish, pike |
| - War Lake | 2006 | 8–11 Aug | 3 | Fish mercury collection | Pike, walleye |
| - Waskaiowaka Lake | 2005 | 6–7 Aug | 5 | Fish mercury collection | Whitefish, pike, walleye |

1. n=sample size.
2. Whitefish= lake whitefish, pike= northern pike, smelt= rainbow smelt, sturgeon= lake sturgeon, Sp shiner= spottail shiner, Em shiner= emerald shiner, T perch= trout-perch, Y perch= yellow perch, W sucker= white sucker.
3. Sample sites for forage fish gillnetting (including one location just west of Gull Lake) are referred to in Section 7.2.3.2.3 as Nelson River mainstem and backwater sites.

**Table 7B-2: Comparison of certified total mercury concentrations (ppm; mean ± 95% C.L. or expanded uncertainty; see http://inms-ienm.nrc-cnrc.gc.ca/en/calserv/crm_e.php, last accessed 2 Jan 2007) of certified reference materials and laboratory control samples (muscle from four fish) from the Canadian Food Inspection Agency in 2005 with analyses done at the DFO Freshwater Institute "metals" laboratory from 1999-2006**

| Year | Statistic | Mussel[1] (0.0610 ± .0036) | Tort-2 (0.27 ± .06) | Dolt-1 (0.225 ± .037) | Dolt-2* (2.14 ± .28) | Dolt-3 (3.37 ± .14) | Dorm-1 (0.798 ± .074) | Dorm-2 (4.64 ± .26) | Fish-368 (0.782 ± .09) | Fish-369 (0.494 ± .06) | Fish-370 (0.280 ± .03) | Fish-371 (0.399 ± .05) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1999 | Mean | 0.0575 | 0.26 | - | 2.03 | - | - | 4.32 | - | - | - | - |
| | n[2] | 5 | 22 | | 27 | | | 12 | | | | |
| 2000 | Mean | 0.0427 | 0.26 | - | 1.85 | - | - | 3.92 | - | - | - | - |
| | n | 4 | 6 | | 6 | | | 2 | | | | |
| 2001 | Mean | 0.0622 | 0.28 | - | 1.93 | - | - | 3.89 | - | - | - | - |
| | n | 17 | 32 | | 25 | | | 13 | | | | |
| 2002 | Mean | 0.0633 | 0.28 | - | 1.93 | - | - | 4.12 | - | - | - | - |
| | n | 14 | 27 | | 29 | | | 11 | | | | |
| 2003 | Mean | 0.0625 | 0.29 | - | 1.90 | 3.20 | - | 4.02 | - | - | - | - |
| | n | 12 | 22 | | 4 | 8 | | 6 | | | | |
| 2004 | Mean | 0.0613 | 0.27 | - | - | 3.11 | - | 4.02 | - | - | - | - |
| | n | 13 | 26 | | | 11 | | 9 | | | | |
| 2005 | Mean | 0.0560 | 0.30 | - | - | 3.11 | - | 3.95 | 0.809 | 0.534 | 0.263 | 0.372 |
| | n | 17 | 45 | | | 21 | | 11 | 6 | 6 | 26 | 26 |
| 2006 | Mean | 0.0564 | 0.271 | 0.262 | - | 2.843 | 0.775 | | - | 0.520 | 0.264 | 0.369 |
| | n | 5 | 29 | 24 | | 25 | 6 | | - | 20 | 23 | 16 |

1. Certified reference materials: mussel tissue (CRM 2976), lobster hepatopancreas (Tort-2), dogfish liver (Dolt-1, Dolt-2, Dolt-3), and dogfish red muscle (Dorm-1, Dorm-2).
2. n represents the number of analyses in a year.
* Dolt-2 reference material was no longer available after 2003.

**Table 7B-3: List of fish species with their standard length (total length for lake sturgeon, fork length for all other species) used for the determination of mercury concentrations under the Keeyask environmental studies. Standard lengths of small individuals of a species analyzed as forage fish are given in brackets**

| Species | Scientific name | Standard length (mm) |
|---|---|---|
| Cisco | *Coregonus artedi* | 300 |
| Emerald shiner | *Notropis atherinoides* | 75 |
| Lake sturgeon | *Acipenser fulvescens* | 1300 |
| Lake trout | *Salvelinus namaycush* | 500 |
| Lake whitefish | *Coregonus clupeaformis* | 350 |
| Northern pike | *Esox lucius* | 550 |
| Rainbow smelt | *Osmerus mordax* | 100 |
| Spottail shiner | *Notropis hudsonius* | 75 |
| Trout-perch | *Percopsis omiscomaycus* | 75 |
| Walleye | *Sander vitreus* | 400 |
| White sucker | *Catostomus commersonii* | 400 (80) |
| Yellow perch | *Perca flavescens* | (75) |

# APPENDIX 7C
# EXAMPLE OF RELATIONSHIP BETWEEN MERCURY CONCENTRATION AND FISH LENGTH (LARGE-BODIED FISH)

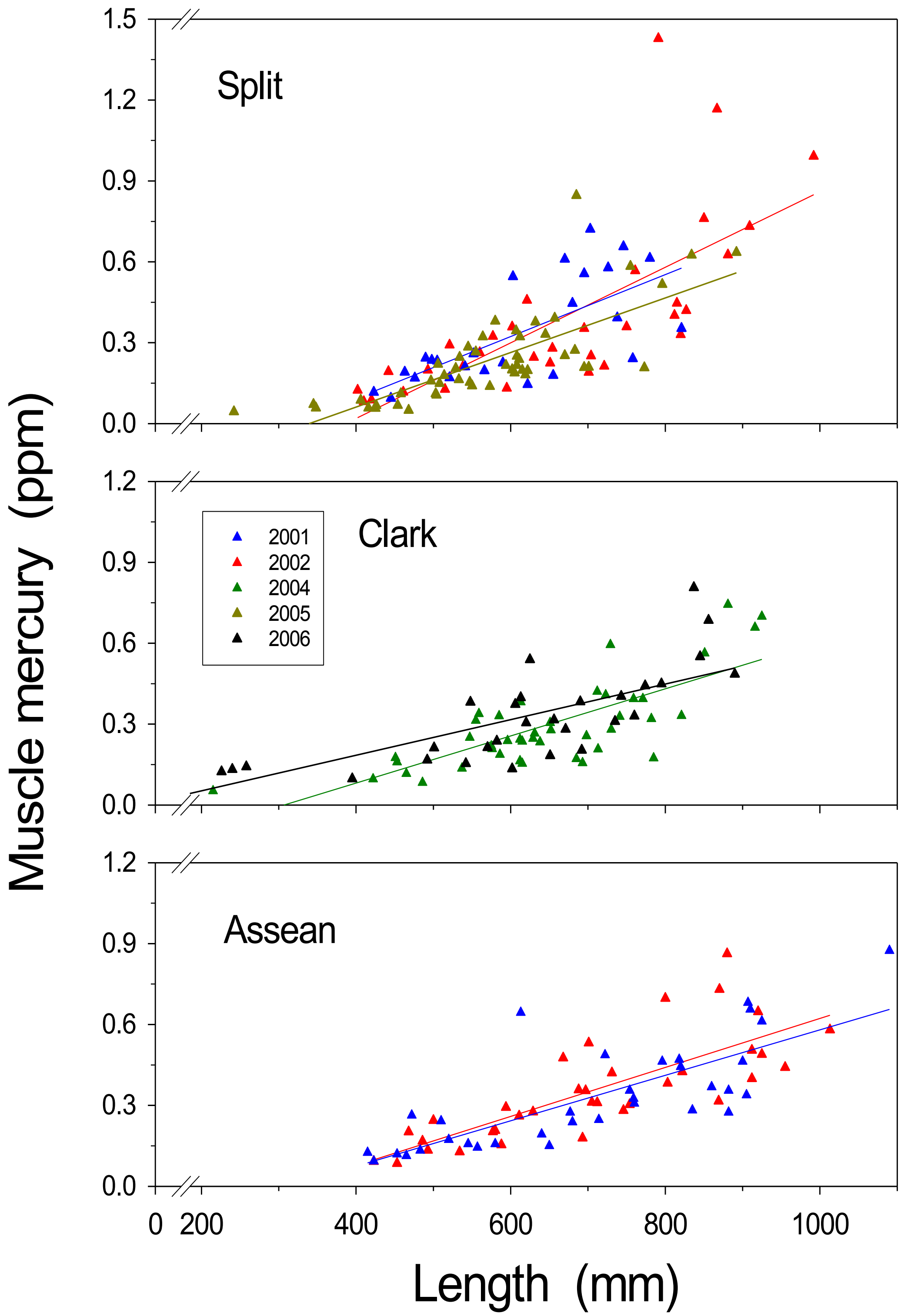


**Figure 7C-1: Relationship between mercury concentration and fish length for northern pike from Split, Clark, and Assean lakes for 2001–2006. Significant linear regression lines are shown**

# APPENDIX 7D
# EXAMPLE OF RELATIONSHIP BETWEEN MERCURY CONCENTRATION AND FISH LENGTH (SMALL-BODIED FISH)

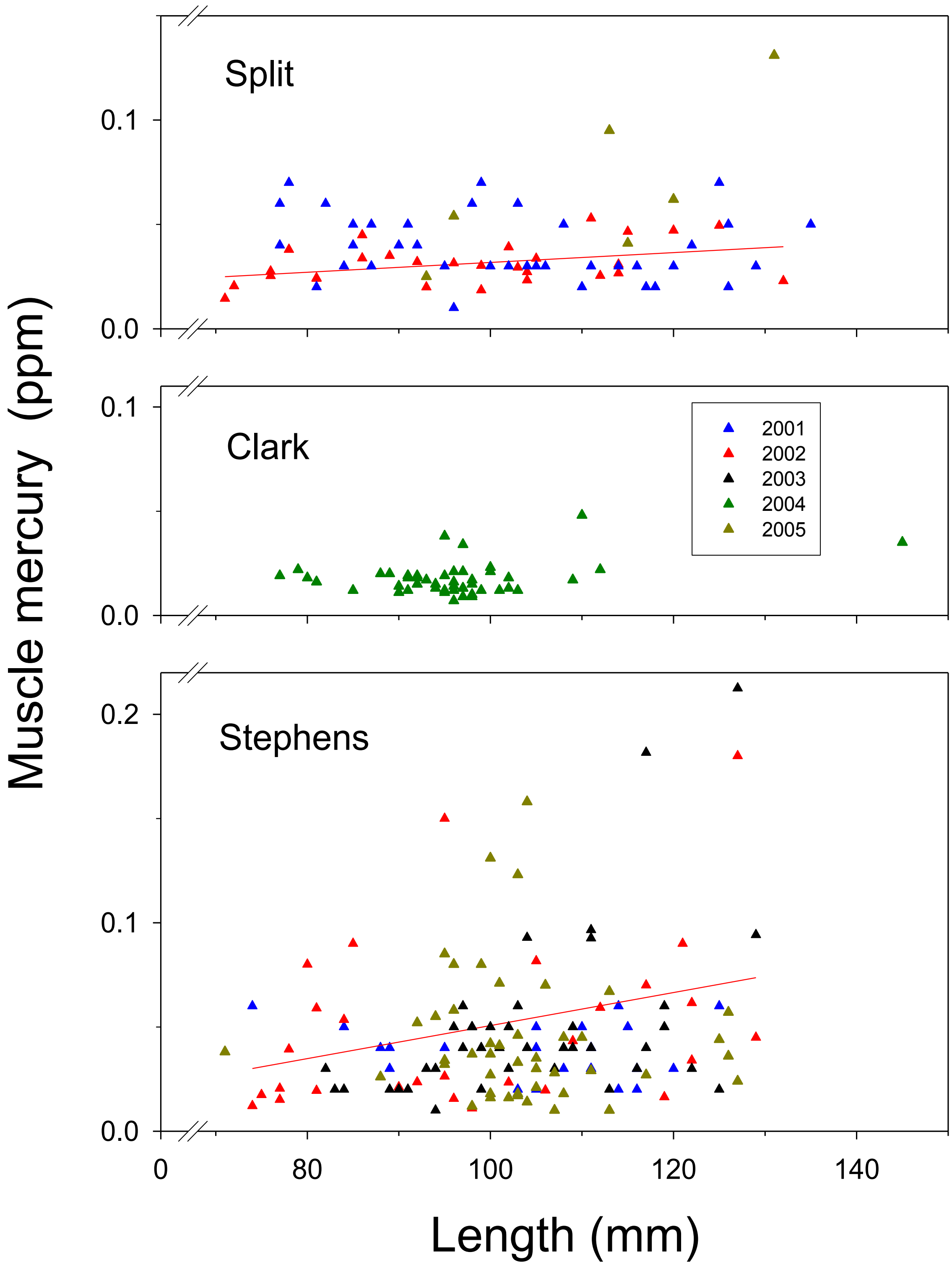


**Figure 7D-1: Relationship between mercury concentration and fish length for rainbow smelt from Split, Clark, and Stephens lakes for 2001–2005. Significant linear regression lines are shown**

# APPENDIX 7E
# MODELLING APPROACHES AND METHODS FOR THE PREDICTION OF POST-PROJECT MAXIMUM MERCURY CONCENTRATIONS IN FISH

# 7E.1 DESCRIPTIONS OF MODELS USED IN PROJECT ASSESSMENT

To estimate the impact of the Project on maximum mercury concentrations in lake whitefish, northern pike, and walleye from the Keeyask reservoir and Stephens Lake, three existing models for predicting mercury concentrations in fish were considered: a mechanistic; a semi empirical-semi mechanistic; and an empirical model. In the following sections, the applicability of each type of model in predicting mercury concentrations in the Keeyask reservoir is discussed, as well as the use of the selected predictive model. A second approach that uses a nearby reservoir (Stephens Lake) as a proxy for future conditions in the Keeyask reservoir is also described.

For modelling purposes, the weighted (by fish sample size) mean standard concentrations for the three last data years (2001, 2002, 2006) for Gull Lake and the four last data years (2001, 2002, 2003, 2005) for Stephens Lake were used to represent current fish mercury levels. There was no discernable temporal trend in the concentrations for these years, and all data were used to minimize chance events that may be associated with using only the data from the most recent year.

## 7E.1.1 MECHANISTIC MODEL

A mechanistic model (*i.e.*, based on mathematical equations that describe natural processes) was recently developed for predicting mercury cycling and bioaccumulation in shallow flood zones of reservoirs using data collected as part of the Flooded Uplands Dynamics Experiment (FLUDEX) and the Experimental Lakes Area Reservoir Project (ELARP) experiments at the Experimental Lakes Area (ELA) near Kenora, Ontario (Harris and Hutchinson 2009). This so-called RESMERC model used a mass balance approach to adequately reproduce the key trends in fish mercury concentrations observed in FLUDEX and ELARP reservoirs, and a natural lake at the ELA. However, the model was not very successful in predicting the mercury dynamics of a relatively large, morphometrically complex, old reservoir (*i.e.*, Notigi Lake in northern Manitoba). The authors concluded that additional model testing was necessary at other full-scale reservoirs to better calibrate and constrain the model and to account for potential effects of, for example, hypolimnetic mercury production, spatial heterogeneity in mercury concentrations, and differences in flow rate/hydraulic residence time (Harris and Hutchinson 2009). Furthermore, most of the model's input parameters are not known for Gull Lake, and would have to be substituted by generic values or, if available, data from other northern Manitoba reservoirs. For these reasons, the RESMERC model was not used to predict fish mercury concentrations for the Keeyask project.

## 7E.1.2 SEMI MECHANISTIC-SEMI EMPIRICAL MODEL

A so-called semi mechanistic-semi empirical model that incorporates some of the functional geochemical and biological relationships that govern mercury cycling and bioaccumulation in reservoirs is currently being developed for Hydro Québec at the University of Sherbrooke. This model mainly uses inputs obtained from the monitoring of the La Grande hydroelectric complex and is not ready/available for more general use (Schetagne *pers. comm.* 2009).

### 7E.1.3 EMPIRICAL MODEL

An empirical model (*i.e.*, based on observed data) was developed by Johnston *et al.* (1991) to predict maximum fish mercury body burdens following reservoir creation using regression equations developed from data collected from 21 northern Manitoba lakes and reservoirs on or near the route of the CRD. Body burden refers to the total amount of mercury in a fish (*i.e.*, the product of mercury concentration and fish weight). The physical variables used in the model equations to predict mercury burden include: change in surface water level; percent flooding (%F); upstream percent flooding (U%F); flooded area to volume ratio; and upstream flooded area to volume ratio. Statistical analysis was used to relate the observed mercury burden of lake whitefish, northern pike, and walleye in the different lakes to the five physical variables to show which of these variables are the best predictor(s) of peak mercury concentrations.

## 7E.2 MODEL SELECTION

Because both the RESMERC model and the Hydro Québec model still need to be improved or are not yet available, the empirical model(s) developed by Johnston *et al.* (1991) was selected to predict maximum mercury concentrations in the Keeyask reservoir. The model(s) is published in the peer-reviewed literature, is based on data for lakes from the same geographical area as the Keeyask Project, and all the necessary data are available from the Keeyask Project. It is important to recognize that the Johnston *et al.* (1991) model(s) can only be applied to estimate maximum mercury burdens/concentrations in lake whitefish, northern pike, and walleye, but does not predict the timing of maximum mercury concentrations and the duration of elevated concentrations in these species.

Because upstream flooding is not an issue for the Keeyask reservoir, the two Johnston *et al.* (1991) model equations initially considered in the impact analysis were:

(1) MERC = $b_0 + b_1$ %F; and

(2) MERC = $b_0 + b_1$ AVR

where MERC = mean peak mercury burden calculated as the product of fish wet weight and muscle mercury concentration;

%F = the percentage of reservoir flooding (*i.e.*, flooded area/total area);

AVR = the ratio of flooded area to reservoir volume;

$b_0$ = regression constant related to the baseline mercury burden without flooding; and

$b_1$ = regression constant related to the flooding contribution to the burden.

These models were developed for fish of a standard fork length of 550 mm for northern pike, 400 mm for walleye, and 350 mm for lake whitefish, resulting in the following regression equations for the %F models (coefficient of determination):

| | | |
|---|---|---|
| (1a) Northern pike: | 565.0 + 15.70 %F | ($r^2$=0.38) |
| (1b) Walleye: | 292.7 +10.18 %F | ($r^2$=0.57) |
| (1c) Lake whitefish: | 60.1 + 1.60 %F | ($r^2$=0.52) |

Model equations (1a-c) and (2a-c, not shown) were modified to better represent the specific conditions in the Keeyask reservoir and Stephens Lake. This was accomplished by replacing the "generic" intercept of the equations with a specific intercept that reflects existing fish mercury concentrations and body mass for each of the three species in Gull and Stephens lakes. The intercept was calculated by multiplying the current species-specific mean mercury concentration by the average weight of a fish of standard length obtained from the weight-length regression from a large sample (greater than 500) of fish. The adjusted regression equations for Gull Lake were:

| | |
|---|---|
| (1aa) Northern pike: | 285.2 + 15.70 %F |
| (1ba) Walleye: | 186.6 +10.18 %F |
| (1ca) Lake whitefish: | 54.4 + 1.60 %F |

Predicted mercury body burdens were recalculated as concentrations. The percentage flooding for the Keeyask reservoir at day 1 (*i.e.*, first day the initial fill level is in effect) was calculated according to Johnston *et al.* (1991) as 49.6%, applying a pre-flood area of 46.6 $km^2$ and a reservoir area of 92.5 $km^2$ (*i.e.*, 45.9 $km^2$ flooded area) (PE SV, Section 6). Over time, more peat is expected to disintegrate in the Keeyask reservoir, increasing the total flooded area in Year 5 post-flooding by 2.9 $km^2$ (PE SV, Section 6.) or 1.6%. The continuing peat disintegration could enhance methylmercury production in sediments (see Section 2.5) and, subsequently, the magnitude and the duration of elevated mercury concentrations in fish. To obtain a worst case quantitative estimate of the potential effect of peat disintegration on fish mercury concentrations, percentage flooding at Day 1 was also calculated by adding the entire 2.9 $km^2$ increase in flooded area predicted for Year 5 to the 45.9 $km^2$ initial flooded area, resulting in a percentage flooding of 51.2% (PE SV, Section 6).

The %F and the AVR models resulted in estimates of fish mercury concentrations that are within 2–5% of each other. Because reservoir volume is estimated with less certainty than percent flooded area, only the results for the %F models are presented in this document.

To test the Johnston *et al.* (1991) %F model (1) on a reservoir that was not part of the model building process, that is located within the study area, and that has a long record of fish mercury concentrations, the modified model (modifications as described above) was applied to Stephens Lake. For this hind cast prediction of maximum values, it was assumed that the mercury concentrations and length-weight relationships obtained for whitefish, pike, and walleye from Stephens Lake in 2003 are similar to those during pre-impoundment years (*i.e.*, prior to the construction of the Kettle GS). Although maximum fish mercury concentrations for Stephens Lake are not known because of the time lag between impoundment and first mercury measurements in fish (see below) and the first measured values had to be used for comparison, the results of this test indicated that the model severely underestimated maximum concentrations in walleye and likely underestimated those in pike (Table 7E-1).

## 7E.2.1 STEPHENS LAKE PROXY MODEL

Because of the disjunct between modeled and likely maximum mercury concentrations in at least two of the three fish species from Stephens Lake (Table 7E-1), a second empirical approach to estimate future fish mercury concentrations in the Keeyask reservoir was used. For this, Stephens Lake was used as a proxy for future mercury methylation and bioaccumulation conditions in the Keeyask reservoir. Stephens Lake was considered appropriate as it is adjacent to Gull Lake in a similar physiogeographic area, and flooding related to the construction of the Kettle GS affected areas with much peat and other wetlands, as will occur in the Keeyask reservoir (PE SV, Section 6.4.2.1). Natural wetlands (including peatlands) are a major source (Kelly *et al.* 1997) and play an important role in the transport (Driscoll *et al.* 1994) of methylmercury to lake ecosystems. Although trees were not cleared from the flooded zone of Stephens Lake prior to inundation, as is proposed for the Keeyask reservoir, this difference in site preparation is negligible for modeling purposes, because the elimination of standing trees alone removes only insignificant amounts of mobile carbon and mercury compared to that left in other vegetation and soil (Mailman *et al.* 2006).

As mentioned before, the exact maximum mercury concentrations in fish from Stephens Lake are unknown because of the long time lag between flooding (1970) and first mercury measurements (1981/1983). Concentrations in walleye, northern pike, and lake whitefish were 1.76, 1.05, and 0.19 ppm, respectively, when first (second for whitefish) measured 11–14 years after flooding (Table 7E-1). For more realistic estimates of maximum post-flooding concentrations, values of 2.0, 1.8, and 0.25 ppm were assumed for the three species, respectively. These estimates were obtained by applying the relationship between maximum fish mercury concentrations three to eight years after flooding and the concentrations four to eight additional years later from several northern Manitoba reservoirs to Stephens Lake. Expected maximum mercury concentrations of whitefish, pike, and walleye in the Keeyask reservoir were predicted by linear interpolation from the existing concentrations in Stephens Lake. For this, it was assumed that current (weighted mean for years 2001–2005) mercury concentrations of whitefish (0.09 ppm), pike (0.26 ppm), and walleye (0.29 ppm) in Stephens Lake represent natural, pre-impoundment conditions. Expected maximum concentration for whitefish, pike, and walleye in the Keeyask reservoir were then estimated based on the increases in concentrations for 70.3% flooding (Derksen and Green 1987; Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion [CMAMM] 1987) by interpolating for percentage flooding predicted for the Keeyask reservoir. The resultant concentrations were added to the current (weighted mean for years 2001–2006) concentrations for each species in Gull Lake. For example, mercury concentrations in walleye increased by 1.70 ppm (2.0–0.30 ppm) for 70.3% flooding in Stephens Lake, corresponding to 1.20 ppm and 1.23 ppm for the 49.6% (Day 1) and 51.2 % (Year 5) flooding of the Keeyask reservoir. With current Gull Lake concentrations of 0.23 ppm, the expected maximum concentration in walleye for the Keeyask reservoir is 1.42–1.46 ppm (differences due to rounding).

# 7E.2.2 MODEL TO PREDICT FISH MERCURY CONCENTRATIONS DOWNSTREAM OF THE KEEYASK RESERVOIR

The export of methylmercury in water and biota from flooded lakes resulting in elevated fish mercury concentrations downstream is known from Manitoba (Bodaly *et al.* 2007) and Québec (Schetagne and Verdon 1999b; Schetagne *et al.* 2000; Schetagne *et al.* 2003) hydroelectric reservoirs. Significant increases in downstream transport of methylmercury have also been observed during the experimental flooding studies of the ELA reservoir projects (Kelly *et al.* 1997; St. Louis *et al.* 2004; Hall *et al.* 2005). The geographical extent of downstream effects in large reservoirs is highly variable (Bodaly *et al.* 2007), but has been observed as far as 275 km on downstream river sections without large, deep bodies of water that promote biological uptake of mercury-rich particles originating from the reservoir (Schetagne and Verdon 1999b). In order to estimate downstream export of mercury from the Keeyask reservoir, maximum fish mercury concentrations in Stephens Lake were first estimated using the Johnston *et al.* (1991) two-variable model, which considers percent *in situ* flooding and upstream flooding:

(3) $\text{MERC} = b_0 + b_1\,\%F + b_2\,U\%F$

where: MERC = mean peak mercury burden calculated as the product of fish wet weight and muscle mercury concentration;

%F = the percentage of reservoir flooding (*i.e.*, flooded area/total area);

$b_0$ = regression constant related to the baseline mercury burden without flooding;

$b_1$ = regression constant related to the flooding contribution to the burden

U%F = percentage of upstream (*i.e.*, Keeyask reservoir) flooding (flooded area/total area); and

$b_2$ = regression constant related to the upstream contribution of fish mercury burden.

The resultant estimates of maximum mercury concentrations in Stephens Lake were unrealistically high (*e.g.*, 1.5 ppm for pike). A probable explanation for the inability of the %F/U%F model (3) to reasonably predict downstream export of mercury from the Keeyask reservoir into Stephens Lake is that the within-lake and upstream variables are given the same relative importance, such that upstream effects have a much larger effect on mercury burdens than within-reservoir effects (Johnston *et al.* 1991). Moreover, upstream flooding in Johnston *et al.* (1991) did not consider the distance between and the relative sizes of the flooded and downstream waterbody, and thus potential dilution effects. Finally, Stephens Lake (*i.e.*, the receiving waterbody) will experience no flooding due to the Project, a scenario that was not part of the Johnston *et al.* (1991) model building. Because of the apparent inadequacies of the 2-variable model (3) and the close proximity of the Keeyask reservoir and Stephens Lake (less than 4 km), the two waterbodies were treated as one and the %F model (1) was used to predict downstream mercury concentrations. Thus, the area flooded by the Keeyask GS (45.9 $km^2$ on Day 1, 48.8 $km^2$ in Year 5) (PE SV, Section 6 [Shoreline Erosion Processes]) was apportioned to the combined area of the Keeyask reservoir (92.5 $km^2$ on Day 1, 95.4 $km^2$ in Year 5) and the Stephens Lake water area of 332.02 $km^2$ (TE SV), resulting in a percentage flooding of 10.8% at day 1 and 11.4% at Year 5 post-impoundment. This approach likely represents a worst-case scenario as Stephens Lake may experience only downstream transport of water (see Section 2.5) and biota with elevated methylmercury concentrations rather than *in situ* increases in mercury methylation due to the Project.

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## 7E.3.2 PERSONAL COMMUNICATIONS

Schetagne, Roger. 2009. Mercury Program Manager, Hydro Québec. Email and telephone correspondence with Wolfgang Jansen, North/South Consultants Inc., Winnipeg, MB, November 21, 2009.

**Table 7E-1: Model-derived estimates of mean maximum mercury concentrations (ppm) in lake whitefish, northern pike, and walleye for Stephens Lake after the construction of the Kettle Generating Station in 1970 compared to the first (second for whitefish) measured mean (±95% CL) mercury concentrations in 1981/1984**

| Model | Species | | |
|---|---|---|---|
| | Lake Whitefish | Northern Pike | Walleye |
| %F model[1] | 0.26 | 1.16 | 1.20 |
| First measured | 0.19 (0.17–0.21) | 1.05 (0.96–1.15) | 1.76[2] |
| Year of measurement | 1984 | 1983 | 1981 |

1. Percent flooded (%F) regression model modified after Johnston *et al.* (1991).
2. Commercial sample; the first sample of individual fish in 1983 had a mean of 1.37 (± 1.10-1.70) ppm.

# APPENDIX 7F
# BIOLOGICAL INFORMATION FOR FISH SPECIES OF COMMERCIAL IMPORTANCE FROM STUDY AREA WATERBODIES, 1998–2006

**Table 7F-1: Mean (± standard error [SE]) fork length, round weight, and age of fish species of commercial importance from study area waterbodies, 1998–2006**

| Species | Waterbody | Year | Length (mm) | n[1] | Weight (g) | n | Age (years) | n |
|---|---|---|---|---|---|---|---|---|
| Lake whitefish | Split Lake | 1998 | 361.8 ± 15.6 | 24 | 852.7 ± 100.2 | 24 | 6.9 ± 1.0 | 24 |
| | | 2001 | 326.3 ± 23.1 | 28 | 803.7 ± 139.0 | 28 | 6.5 ± 1.0 | 28 |
| | | 2002 | 355.8 ± 24.6 | 30 | 1074.4 ± 171.8 | 30 | 7.0 ± 0.8 | 30 |
| | | 2005 | 464.6 ± 5.8 | 37 | 1929.7 ± 81.6 | 37 | 11.3 ± 0.4 | 37 |
| | Gull Lake | 1999 | 356.4 ± 22.8 | 22 | 1058.0 ± 153.8 | 22 | 5.8 ± 0.7 | 22 |
| | | 2001 | 415.4 ± 23.3 | 21 | 1584.5 ± 261.9 | 21 | 7.7 ± 1.1 | 21 |
| | | 2002 | 366.9 ± 30.1 | 26 | 1450.8 ± 239.5 | 25 | 7.8 ± 1.2 | 26 |
| | Stephens Lake | 2001 | 488.6 ± 9.0 | 15 | 2180.3 ± 123.2 | 15 | 13.2 ± 1.4 | 9 |
| | | 2002 | 402.7 ± 23.6 | 25 | 1363.6 ± 189.2 | 25 | 8.1 ± 0.9 | 25 |
| | | 2003 | 393.6 ± 14.9 | 78 | 1747.8± 134.6 | 69 | 9.5 ± 0.7 | 76 |
| | | 2005 | 488.3 ± 9.5 | 25 | 2234.0± 138.4 | 25 | 12.2 ± 0.8 | 25 |
| | Assean Lake | 2001 | 331.3 ± 21.5 | 32 | 783.5 ± 131.2 | 32 | 5.3 ± 0.7 | 32 |
| | | 2002 | 349.8 ± 22.9 | 28 | 822.2 ± 129.1 | 28 | 6.4 ± 0.9 | 27 |
| Northern pike | Split Lake | 1998 | 557.7 ± 23.5 | 38 | 1500.9 ± 186.4 | 38 | 4.9 ± 0.4 | 35 |
| | | 2001 | 610.7 ± 23.4 | 25 | 1906.0 ± 207.3 | 25 | 6.2 ± 0.3 | 25 |
| | | 2002 | 666.3 ± 28.0 | 33 | 2678.8 ± 331.5 | 33 | 7.8 ± 0.6 | 25 |
| | | 2005 | 573.6 ± 17.4 | 51 | 1571.6 ± 142.5 | 51 | 6.8 ± 0.4 | 51 |
| | Clark Lake | 2004 | 645.8 ± 20.9 | 44 | 2213.0 ± 219.9 | 23 | 6.8 ± 0.4 | 44 |
| | | 2006 | 613.9 ± 35.0 | 31 | 2245.0 ± 287.8 | 30 | 7.8 ± 0.6 | 31 |
| | NR Mainstem[2] | 2006 | 653.9 ± 17.6 | 22 | 2046.6 ± 167.7 | 22 | - | - |
| | Gull Lake | 1999 | 694.4 ± 27.8 | 40 | 3440.0 ± 412.6 | 40 | 8.0 ± 0.5 | 39 |
| | | 2001 | 687.8 ± 30.0 | 33 | 2966.7 ± 381.2 | 33 | 7.6 ± 0.5 | 31 |
| | | 2002 | 699.7 ± 29.9 | 35 | 3298.6 ± 411.5 | 35 | 9.2 ± 0.6 | 35 |
| | | 2006 | 501.3 ± 29.5 | 44 | 1361.0 ± 227.5 | 44 | 5.3 ± 0.5 | 44 |
| | Stephens Lake | 2001 | 641.1 ± 35.9 | 27 | 2376.9 ± 406.7 | 27 | 5.7 ± 0.5 | 265 |
| | | 2002 | 700.1 ± 29.9 | 35 | 2955.0 ± 357.1 | 35 | 9.3 ± 0.7 | 33 |
| | | 2003 | 631.5 ± 17.7 | 76 | 2276.8 ± 203.2 | 76 | 9.4 ± 0.5 | 73 |
| | | 2005 | 583.5 ± 20.0 | 52 | 1742.8 ± 206.9 | 52 | 6.7 ± 0.4 | 52 |
| | Aiken River | 2002 | 530.1 ± 34.3 | 17 | 1436.8 ± 399.4 | 17 | 5.1 ± 0.4 | 15 |
| | | 2003 | 492.5 ± 21.8 | 18 | 958.9 ± 104.8 | 14 | 6.2 ± 0.5 | 18 |
| | AR[3], York Landing | 2006 | 589.2 ± 12.2 | 33 | 1519.7 ± 101.5 | 33 | 7.2 ± 0.3 | 30 |
| | AR[3], Ilford | 2006 | 496.4 ± 8.1 | 50 | 949.0 ± 44.1 | 50 | 6.3 ± 0.2 | 50 |
| | Assean Lake | 2001 | 703.6 ± 30.5 | 34 | 2972.1 ± 385.4 | 34 | 7.4 ± 0.5 | 34 |
| | | 2002 | 706.1 ± 27.5 | 35 | 2998.6 ± 354.0 | 35 | 8.5 ± 0.5 | 35 |

**Table 7F-1: Mean (± standard error [SE]) fork length, round weight, and age of fish species of commercial importance from study area waterbodies, 1998–2006**

| Species | Waterbody | Year | Length (mm) | n[1] | Weight (g) | n | Age (years) | n |
|---|---|---|---|---|---|---|---|---|
| Walleye | Split Lake | 1998 | 362.0 ± 21.6 | 25 | 714.6 ± 138.3 | 25 | 7.6 ± 0.8 | 25 |
| | | 2001 | 398.1 ± 21.6 | 27 | 1033.1 ± 157.5 | 27 | 7.2 ± 0.7 | 26 |
| | | 2002 | 410.2 ± 21.7 | 30 | 1175.2 ± 169.6 | 29 | 7.6 ± 0.8 | 28 |
| | | 2005 | 330.2 ± 16.1 | 53 | 634.4 ± 83.7 | 53 | 6.1 ± 0.4 | 53 |
| | Clark Lake | 2004 | 399.7 ± 17.2 | 44 | 966.6 ± 103.8 | 41 | 6.7 ± 0.5 | 44 |
| | | 2006 | 338.0 ± 12.3 | 48 | 1127.1 ± 76.2 | 48 | 7.9 ± 0.4 | 47 |
| | NR mainstem[2] | 2006 | 525.9 ± 23.0 | 10 | 1880.0 ± 201.3 | 10 | - | - |
| | Gull Lake | 1999 | 444.6 ± 13.7 | 22 | 1350.0 ± 131.5 | 22 | 8.5 ± 0.8 | 22 |
| | | 2001 | 421.5 ± 20.7 | 26 | 1180.8 ± 165.0 | 26 | 7.0 ± 1.0 | 24 |
| | | 2002 | 423.3 ± 23.6 | 32 | 1339.6 ± 200.8 | 32 | 9.1 ± 1.1 | 32 |
| | | 2006 | 463.8 ± 19.5 | 34 | 1415.3 ± 149.6 | 34 | 9.9 ± 0.9 | 34 |
| | Stephens Lake | 2001 | 418.6 ± 20.5 | 29 | 1217.2 ± 174.1 | 29 | 7.9 ± 1.0 | 29 |
| | | 2002 | 437.9 ± 21.7 | 34 | 1320.6 ± 171.4 | 34 | 10.4 ± 0.9 | 33 |
| | | 2003 | 433.5 ± 11.8 | 70 | 1240.3 ± 94.6 | 69 | 10.2 ± 0.6 | 67 |
| | | 2005 | 400.8 ± 13.4 | 69 | 1141.3 ± 96.1 | 69 | 10.1 ± 0.7 | 69 |
| | Aiken River | 2002 | 387.2 ± 5.7 | 41 | 714.7 ± 39.2 | 39 | 6.7 ± 0.3 | 38 |
| | | 2003 | 396.7 ± 10.3 | 16 | 681.8 ± 71.2 | 11 | 8.1 ± 0.5 | 13 |
| | AR[3], York Landing | 2006 | 387.3 ± 6.4 | 51 | 722.5 ± 36.2 | 51 | 6.5 ± 0.2 | 51 |
| | AR[3], Ilford | 2006 | 397.3 ± 5.1 | 49 | 737.2 ± 33.9 | 49 | 7.6 ± 0.2 | 49 |
| | Assean Lake | 2001 | 389.4 ± 19.6 | 27 | 853.7 ± 117.7 | 27 | 8.2 ± 0.9 | 27 |
| | | 2002 | 399.5 ± 20.4 | 28 | 873.6 ± 120.5 | 28 | 9.2 ± 0.7 | 28 |

1. Number of fish measured.
2. River mainstem between Clark and Gull lakes.
3. Aiken River.

# APPENDIX 7G
# BIOLOGICAL INFORMATION FOR FORAGE FISH FROM STUDY AREA WATERBODIES, 2001–2006

**Table 7G-1: Mean (± standard error [SE]) length, weight, and age of forage fish from study area waterbodies, 2001–2006**

| Species | Waterbody | Year | Length (mm) | n[1] | Weight (g) | n | Age (years) | n |
|---|---|---|---|---|---|---|---|---|
| Rainbow smelt | Split Lake | 2001 | 102.2 ± 2.7 | 37 | 7.0 ± 0.4 | 37 | n.a.[1] | 37 |
| | | 2002 | 97.7 ± 3.1 | 28 | 6.3 ± 0.6 | 28 | 2.3 ± 0.1 | 27 |
| | | 2005 | 111.3 ± 5.9 | 6 | 10.5 ± 0.3 | 6 | 3.5 ± 0.4 | 6 |
| | Clark Lake | 2004 | 95.5 ± 1.4 | 50 | 6.0 ± 0.4 | 50 | n.a. | - |
| | Gull Lake | 2001 | 99.1 ± 2.9 | 29 | 6.5 ± 0.5 | 29 | 2.0 ± 0.1 | 26 |
| | | 2002 | 102.4 ± 3.3 | 32 | 7.3 ± 0.7 | 32 | 2.2 ± 0.1 | 32 |
| | | 2003 | 103.1 ± 1.4 | 40 | 8.3 ± 0.3 | 40 | 2.7 ± 0.2 | 40 |
| | Gull Lake – MSt[2] | 2004 | 87.0 ± 1.8 | 40 | 4.3 ± 0.4 | 40 | 1.1 ± 0.1 | 14 |
| | Gull Lake – BW[2] | 2004 | 91.4 ± 1.3 | 30 | 5.1 ± 0.2 | 30 | 1.8 ± 0.2 | 8 |
| | Gull Lake | 2006 | 92.9 ± 2.3 | 47 | 4.3 ± 0.41 | 47 | 2.0 ± 0.1 | 24 |
| | Stephens Lake | 2001 | 104.0 ± 3.1 | 23 | 7.1 ± 0.7 | 23 | 2.0 ± 0.1 | 21 |
| | | 2002 | 100.8 ± 3.4 | 28 | 6.5 ± 0.7 | 28 | 2.4 ± 0.2 | 28 |
| | | 2003 | 102.4 ± 1.7 | 40 | 7.7 ± 0.3 | 40 | 2.3 ± 0.1 | 33 |
| | | 2005 | 103.6 ± 1.5 | 45 | 7.0 ± 0.3 | 45 | 3.3 ± 0.1 | 14 |
| Emerald shiner | Gull Lake | 2003 | 52.6 ± 5.1 | 18 | 1.6 ± 0.5 | 18 | 1.6 ± 0.3 | 18 |
| | Gull Lake – MSt | 2004 | 85.5 ± 0.8 | 31 | 6.1 ± 0.2 | 31 | 3.4 ± 0.2 | 31 |
| | Gull Lake – BW | 2004 | 85.0 ± 7.0 | 2 | 5.9 ± 1.0 | 2 | 3.5 ± 0.5 | 2 |
| | Stephens Lake | 2003 | 84.6 ± 1.3 | 53 | 7.4 ± 0.3 | 53 | 3.5 ± 0.2 | 24 |
| Spottail shiner | Gull Lake | 2003 | 70.3 ± 3.4 | 32 | 3.5 ± 0.4 | 32 | 2.5 ± 0.4 | 15 |
| | Gull Lake – MSt | 2004 | 67.9 ± 2.4 | 31 | 3.9 ± 0.4 | 31 | 2.4 ± 0.2 | 31 |
| | Gull Lake – BW | 2004 | 82.4 ± 1.5 | 31 | 6.6 ± 0.4 | 31 | 3.3 ± 0.2 | 31 |
| | Stephens Lake | 2003 | 75.9 ± 1.8 | 40 | 6.5 ± 0.3 | 40 | 3.1 ± 0.3 | 17 |
| Trout-perch | Gull Lake | 2003 | 76.3 ± 2.5 | 40 | 6.3 ± 0.7 | 40 | 3.7 ± 0.3 | 18 |
| | Gull Lake – MSt | 2004 | 70.2 ± 2.4 | 20 | 4.7 ± 0.5 | 20 | 2.2 ± 0.2 | 20 |
| | Gull Lake – BW | 2004 | 65.4 ± 2.4 | 16 | 3.5 ± 0.5 | 16 | 1.9 ± 0.2 | 17 |
| | Stephens Lake | 2003 | 78.3 ± 2.3 | 40 | 7.5 ± 0.5 | 40 | 3.1 ± 0.2 | 18 |
| Yellow perch | Gull Lake | 2003 | 78.4 ± 3.0 | 16 | 6.6 ± 0.7 | 16 | 1.8 ± 0.1 | 10 |
| | Gull Lake – MSt | 2004 | 65.5 ± 3.5 | 49 | 5.2 ± 0.8 | 49 | 1.0 ± 0.2 | 13 |
| | Gull Lake – BW | 2004 | 88.6 ± 1.5 | 30 | 8.7 ± 0.5 | 30 | 1.4 ± 0.2 | 10 |
| | Stephens Lake | 2003 | 72.6 ± 2.1 | 44 | 7.0 ± 0.7 | 40 | 1.8 ± 0.3 | 13 |
| White sucker | Gull Lake – MSt | 2004 | 79.2 ± 6.5 | 20 | 6.5 ± 1.2 | 20 | 0.7 ± 0.2 | 10 |
| | Gull Lake – BW | 2004 | 88.3 ± 1.7 | 18 | 8.1 ± 0.5 | 18 | 1.0 ± 0.0 | 12 |

1. Number of fish measured.
2. Fish caught in mainstem (MSt) and backwaters (BW) areas of the Nelson River at Gull Lake.

# APPENDIX 7H
# MERCURY CONCENTRATIONS OF LARGE-BODIED FISH FROM STUDY AREA WATERBODIES, 1998–2006

**Table 7H-1: Mean arithmetic (± standard error [SE]) and standardized (± 95% CL) mercury concentration (ppm) of large-bodied fish from study area waterbodies, 1998–2006**

| Species | Waterbody | Year | n[1] | Arithmetic | SE | Standard | 95% CL |
|---|---|---|---|---|---|---|---|
| Lake whitefish | Split Lake | 1998 | 24 | 0.117 | 0.007 | 0.111 | 0.075–0.163 |
| | | 2001 | 28 | 0.068 | 0.010 | 0.066 | 0.058–0.076 |
| | | 2002 | 30 | 0.079 | 0.013 | 0.061 | 0.049–0.076 |
| | | 2005 | 37 | 0.075 | 0.004 | 0.030 | 0.021–0.042 |
| | Assean Lake | 2001 | 32 | 0.053 | 0.006 | 0.050 | 0.041–0.060 |
| | | 2002 | 28 | 0.064 | 0.009 | 0.057 | 0.049–0.067 |
| | Gull Lake | 1999 | 22 | 0.098 | 0.016 | 0.075 | 0.055–0.103 |
| | | 2001 | 21 | 0.088 | 0.010 | 0.062 | 0.053–0.073 |
| | | 2002 | 26 | 0.102 | 0.014 | 0.082 | 0.070–0.097 |
| | Stephens Lake | 1999 | 6 | 0.091 | 0.021 | 0.077 | 0.050–0.118 |
| | | 2001 | 15 | 0.153 | 0.014 | 0.104* | 0.037–0.298 |
| | | 2002 | 25 | 0.134 | 0.013 | 0.112 | 0.096–0.131 |
| | | 2003 | 78 | 0.125 | 0.008 | 0.104 | 0.096–0.113 |
| | | 2005 | 25 | 0.108 | 0.009 | 0.029 | 0.020–0.042 |
| Northern pike | Split Lake | 1998 | 38 | 0.304 | 0.027 | 0.265 | 0.086–0.815 |
| | | 2001 | 25 | 0.335 | 0.039 | 0.234 | 0.195–0.281 |
| | | 2002 | 33 | 0.392 | 0.055 | 0.208 | 0.175–0.246 |
| | | 2005 | 51 | 0.237 | 0.023 | 0.182 | 0.164–0.202 |
| | Aiken River[2] | 2002 | 17 | 0.246 | 0.027 | 0.243 | 0.196–0.302 |
| | | 2003 | 18 | 0.268 | 0.038 | 0.327 | 0.252–0.424 |
| | | 2006 | 33 | 0.298 | 0.018 | 0.259 | 0.228–0.293 |
| | Aiken River, Ilford | 2006 | 50 | 0.225 | 0.012 | 0.252 | 0.222–0.285 |
| | Assean Lake | 2001 | 34 | 0.330 | 0.033 | 0.188 | 0.162–0.219 |
| | | 2002 | 35 | 0.355 | 0.032 | 0.194 | 0.168–0.225 |
| | Clark Lake | 2004 | 44 | 0.296 | 0.025 | 0.202 | 0.178–0.228 |
| | | 2006 | 31 | 0.329 | 0.032 | 0.275 | 0.237–0.318 |
| | Nelson River[3] | 2006 | 22 | 0.264 | 0.028 | 0.159 | 0.122–0.207 |
| | Gull Lake | 1999 | 40 | 0.572 | 0.049 | 0.314 | 0.278–0.355 |
| | | 2001 | 33 | 0.447 | 0.060 | 0.220 | 0.181–0.268 |
| | | 2002 | 35 | 0.466 | 0.049 | 0.226 | 0.196–0.260 |
| | | 2006 | 44 | 0.215 | 0.024 | 0.211 | 0.183–0.244 |

**Table 7H-1: Mean arithmetic (± standard error [SE]) and standardized (± 95% CL) mercury concentration (ppm) of large-bodied fish from study area waterbodies, 1998–2006**

| Species | Waterbody | Year | n[1] | Arithmetic | SE | Standard | 95% CL |
|---|---|---|---|---|---|---|---|
| Northern pike (Continued) | Stephens Lake | 1999 | 14 | 0.369 | 0.069 | 0.432 | 0.316–0.591 |
| | | 2001 | 27 | 0.573 | 0.099 | 0.316 | 0.276–0.361 |
| | | 2002 | 35 | 0.663 | 0.083 | 0.332 | 0.280–0.395 |
| | | 2003 | 76 | 0.448 | 0.038 | 0.272 | 0.246–0.301 |
| | | 2005 | 52 | 0.250 | 0.030 | 0.180 | 0.165–0.196 |
| Walleye | Split Lake | 1998 | 25[3] | 0.315 | 0.029 | 0.311 | 0.106–0.914 |
| | | 2001 | 27 | 0.216 | 0.028 | 0.191 | 0.168–0.218 |
| | | 2002 | 30 | 0.238 | 0.025 | 0.210 | 0.180–0.245 |
| | | 2005 | 53 | 0.099 | 0.007 | 0.118 | 0.108–0.128 |
| | Aiken River[2] | 2002 | 41 | 0.224 | 0.014 | 0.221 | 0.197–0.248 |
| | | 2003 | 16 | 0.209 | 0.020 | 0.199* | 0.164–0.242 |
| | | 2006 | 51 | 0.187 | 0.007 | 0.190 | 0.179–0.202 |
| | Aiken River, Ilford | 2006 | 49 | 0.249 | 0.011 | 0.244 | 0.228–0.261 |
| | Assean Lake | 2001 | 27 | 0.180 | 0.017 | 0.176 | 0.158–0.196 |
| | | 2002 | 28 | 0.213 | 0.018 | 0.203 | 0.182–0.227 |
| | Clark Lake | 2004 | 44 | 0.173 | 0.018 | 0.154 | 0.141–0.167 |
| | | 2006 | 48 | 0.281 | 0.022 | 0.229 | 0.203–0.259 |
| | Nelson River[3] | 2006 | 10 | 0.524 | 0.086 | 0.167 | 0.057–0.485 |
| | Gull Lake | 1999 | 22 | 0.414 | 0.042 | 0.293 | 0.244–0.353 |
| | | 2001 | 26 | 0.273 | 0.045 | 0.190 | 0.167–0.217 |
| | | 2002 | 32 | 0.371 | 0.051 | 0.263 | 0.227–0.304 |
| | | 2006 | 34 | 0.405 | 0.051 | 0.216 | 0.182–0.255 |
| | Stephens Lake | 1999 | 24 | 0.444 | 0.058 | 0.425 | 0.356–0.508 |
| | | 2001 | 29 | 0.373 | 0.050 | 0.277 | 0.243–0.316 |
| | | 2002 | 34 | 0.469 | 0.035 | 0.405 | 0.378–0.434 |
| | | 2003 | 70 | 0.418 | 0.027 | 0.329 | 0.298–0.364 |
| | | 2005 | 69 | 0.249 | 0.022 | 0.204 | 0.183–0.227 |

* The relationship between mercury concentration and fish length was not significant.
1. Number of fish measured.
2 At York Landing.
3. River mainstem between Clark and Gull lakes.
4. One outlier mercury value excluded.

# APPENDIX 7I
# MERCURY CONCENTRATIONS OF FORAGE FISH FROM STUDY AREA WATERBODIES, 2001–2006

**Table 7I-1: Mean arithmetic (± standard error [SE]) and standardized (± 95% confidence level [CL]) mercury concentration (ppm) of forage fish from study area waterbodies, 2001–2006**

| Species | Waterbody | Year | n[1] | Arithmetic | SE | Standard | 95% CL |
|---|---|---|---|---|---|---|---|
| Rainbow smelt | Split Lake | 2001 | 37 | 0.039 | 0.003 | 0.036* | 0.032–0.042 |
| | | 2002 | 28 | 0.031 | 0.002 | 0.031 | 0.027–0.034 |
| | | 2005 | 6 | 0.068 | 0.016 | 0.042* | 0.022–0.079 |
| | Clark Lake | 2004 | 51 | 0.017 | 0.001 | 0.017* | 0.015–0.019 |
| | Gull Lake | 2001 | 29 | 0.052 | 0.006 | 0.046* | 0.038–0.055 |
| | | 2002 | 32 | 0.045 | 0.004 | 0.040 | 0.034–0.047 |
| | | 2003 | 40 | 0.024 | 0.001 | 0.023* | 0.021–0.026 |
| | Gull – MSt[2] | 2004 | 40 | 0.016 | 0.002 | 0.010 | 0.008–0.014 |
| | Gull – BW[2] | 2004 | 30 | 0.015 | 0.001 | 0.016* | 0.012–0.021 |
| | | 2006 | 47 | 0.052 | 0.006 | 0.044* | 0.035–0.055 |
| | Stephens Lake | 2001 | 23 | 0.060 | 0.008 | 0.052* | 0.041–0.066 |
| | | 2002 | 28 | 0.053 | 0.009 | 0.039 | 0.030–0.050 |
| | | 2003 | 40 | 0.041 | 0.003 | 0.036* | 0.031–0.042 |
| | | 2005 | 45 | 0.044 | 0.005 | 0.036* | 0.030–0.045 |
| Emerald shiner | Gull Lake | 2003 | 18 | 0.049 | 0.015 | 0.069 | 0.048–0.097 |
| | Gull – MSt | 2004 | 31 | 0.104 | 0.006 | 0.073* | 0.049–0.109 |
| | Gull – BW | 2004 | 2 | 0.110 | 0.029 | 0.071* | - |
| | Stephens Lake | 2003 | 53 | 0.148 | 0.009 | 0.117* | 0.097–0.142 |
| Spottail shiner | Gull Lake | 2003 | 32 | 0.060 | 0.007 | 0.058 | 0.051–0.066 |
| | Gull – MSt | 2004 | 31 | 0.053 | 0.007 | 0.056 | 0.048–0.066 |
| | Gull – BW | 2004 | 31 | 0.074 | 0.006 | 0.047 | 0.041–0.055 |
| | Stephens Lake | 2003 | 40 | 0.155 | 0.011 | 0.138* | 0.117–0.162 |
| Trout-perch | Gull Lake | 2003 | 40 | 0.039 | 0.003 | 0.036 | 0.032–0.041 |
| | Gull – MSt | 2004 | 20 | 0.025 | 0.003 | 0.024* | 0.020–0.030 |
| | Gull – BW | 2004 | 17 | 0.018 | 0.001 | 0.018* | 0.015–0.021 |
| | Stephens Lake | 2003 | 40 | 0.060 | 0.004 | 0.053 | 0.047–0.059 |
| Yellow perch | Gull Lake | 2003 | 16 | 0.023 | 0.001 | 0.022* | 0.019–0.025 |
| | Gull – MSt | 2004 | 49 | 0.029 | 0.002 | 0.027* | 0.024–0.030 |
| | Gull – BW | 2004 | 30 | 0.038 | 0.002 | 0.043* | 0.033–0.055 |
| | Stephens Lake | 2003 | 44 | 0.052 | 0.004 | 0.046* | 0.040–0.055 |
| White sucker | Gull – MSt | 2004 | 20 | 0.022 | 0.002 | 0.021* | 0.019–0.024 |
| | Gull – BW | 2004 | 18 | 0.017 | 0.001 | 0.015* | 0.012–0.020 |

* The relationship between mercury concentration and fish length was not significant.
1. Number of fish measured.
2. Fish caught in mainstem (MSt) and backwaters (BW) areas of the Nelson River at Gull Lake.

# APPENDIX 7J
# EXCEEDENCE OF MERCURY CONCENTRATION THRESHOLDS IN LARGE-BODIED FISH FROM STUDY AREA WATERBODIES, 1998–2006

**Table 7J-1: Percentage of large-bodied fish from study area waterbodies with mercury concentrations of ≥0.2 ppm or ≥0.5 ppm, 1998–2006**

| Species | Waterbody | Year | n[1] | ≥ 0.2 ppm | % | ≥ 0.5 ppm | % | Max[2] (ppm) |
|---|---|---|---|---|---|---|---|---|
| Lake whitefish | Split Lake | 1998 | 24 | 1 | 4.2 | 0 | 0.0 | 0.211 |
| | | 2001 | 28 | 0 | 0.0 | 0 | 0.0 | 0.196 |
| | | 2002 | 30 | 3 | 10.0 | 0 | 0.0 | 0.281 |
| | | 2005 | 37 | 0 | 0.0 | 0 | 0.0 | 0.139 |
| | Assean Lake | 2001 | 32 | 0 | 0.0 | 0 | 0.0 | 0.174 |
| | | 2002 | 28 | 1 | 3.6 | 0 | 0.0 | 0.237 |
| | Gull Lake | 1999 | 22 | 1 | 4.5 | 0 | 0.0 | 0.310 |
| | | 2001 | 21 | 0 | 0.0 | 0 | 0.0 | 0.197 |
| | | 2002 | 26 | 2 | 7.7 | 0 | 0.0 | 0.255 |
| | Stephens Lake | 1999 | 6 | 0 | 0.0 | 0 | 0.0 | 0.155 |
| | | 2001 | 15 | 4 | 26.7 | 0 | 0.0 | 0.236 |
| | | 2002 | 25 | 4 | 16.6 | 0 | 0.0 | 0.333 |
| | | 2003 | 78 | 9 | 11.5 | 0 | 0.0 | 0.370 |
| | | 2005 | 25 | 0 | 0.0 | 0 | 0.0 | 0.194 |
| Northern pike | Split Lake | 1998 | 38 | 26 | 68.4 | 4 | 10.5 | 0.723 |
| | | 2001 | 25 | 17 | 68.0 | 7 | 28.0 | 0.721 |
| | | 2002 | 33 | 24 | 72.7 | 7 | 21.2 | 1.428 |
| | | 2005 | 51 | 27 | 52.9 | 5 | 9.8 | 0.846 |
| | Aiken River[3] | 2002 | 17 | 11 | 64.7 | 0 | 0.0 | 0.466 |
| | | 2003 | 18 | 12 | 66.7 | 1 | 5.6 | 0.641 |
| | | 2006 | 33 | 27 | 81.8 | 1 | 3.0 | 0.547 |
| | Aiken River, Ilford | 2006 | 50 | 29 | 58.0 | 0 | 0.0 | 0.477 |
| | Assean Lake | 2001 | 34 | 23 | 67.6 | 5 | 14.7 | 0.874 |
| | | 2002 | 35 | 28 | 80.0 | 7 | 20.0 | 0.862 |
| | Clark Lake | 2004 | 44 | 31 | 70.5 | 6 | 13.6 | 0.743 |
| | | 2006 | 31 | 22 | 71.0 | 5 | 16.1 | 0.806 |
| | Nelson River[4] | 2006 | 22 | 13 | 59.1 | 2 | 9.1 | 0.596 |
| | Gull Lake | 1999 | 40 | 36 | 90.0 | 22 | 55.0 | 1.257 |
| | | 2001 | 33 | 25 | 75.8 | 12 | 36.4 | 1.470 |
| | | 2002 | 35 | 27 | 77.1 | 15 | 42.8 | 1.136 |
| | | 2006 | 44 | 16 | 36.4 | 3 | 6.8 | 0.775 |
| | Stephens Lake | 1999 | 14 | 11 | 78.6 | 2 | 14.3 | 0.939 |
| | | 2001 | 27 | 22 | 81.5 | 9 | 33.3 | 2.099 |
| | | 2002 | 35 | 33 | 94.3 | 15 | 42.9 | 1.954 |
| | | 2003 | 76 | 60 | 78.9 | 25 | 32.9 | 1.529 |
| | | 2005 | 52 | 25 | 48.1 | 4 | 7.7 | 1.288 |

**Table 7J-1: Percentage of large-bodied fish from study area waterbodies with mercury concentrations of ≥0.2 ppm or ≥0.5 ppm, 1998–2006**

| Species | Waterbody | Year | n[1] | ≥ 0.2 ppm | % | ≥ 0.5 ppm | % | Max[2] (ppm) |
|---|---|---|---|---|---|---|---|---|
| Walleye | Split Lake | 1998 | 25[3] | 21 | 84.0 | 3 | 12.0 | 0.602 |
| | | 2001 | 27 | 10 | 37.0 | 1 | 3.7 | 0.533 |
| | | 2002 | 30 | 12 | 40.0 | 2 | 6.7 | 0.652 |
| | | 2005 | 53 | 2 | 3.8 | 0 | 0.0 | 0.222 |
| | Aiken River[3] | 2002 | 41 | 21 | 51.2 | 1 | 2.4 | 0.524 |
| | | 2003 | 16 | 7 | 43.8 | 0 | 0.0 | 0.416 |
| | | 2006 | 51 | 16 | 31.4 | 0 | 0.0 | 0.333 |
| | Aiken River, Ilford | 2006 | 49 | 33 | 67.3 | 1 | 2.0 | 0.553 |
| | Assean Lake | 2001 | 27 | 10 | 37.0 | 0 | 0.0 | 0.423 |
| | | 2002 | 28 | 12 | 42.9 | 0 | 0.0 | 0.449 |
| | Clark Lake | 2004 | 44 | 14 | 31.8 | 1 | 2.3 | 0.609 |
| | | 2006 | 48 | 33 | 68.8 | 7 | 14.6 | 0.806 |
| | Gull Lake | 1999 | 22 | 19 | 86.4 | 7 | 31.8 | 0.791 |
| | | 2001 | 26 | 12 | 46.2 | 5 | 19.2 | 0.864 |
| | | 2002 | 32 | 27 | 77.1 | 15 | 37.5 | 1.074 |
| | | 2006 | 34 | 21 | 61.8 | 15 | 44.1 | 1.184 |
| | Stephens Lake | 1999 | 24 | 15 | 62.5 | 10 | 41.7 | 0.987 |
| | | 2001 | 29 | 16 | 55.2 | 11 | 37.9 | 0.956 |
| | | 2002 | 34 | 34 | 100.0 | 12 | 35.3 | 1.012 |
| | | 2003 | 70 | 59 | 84.3 | 23 | 32.9 | 1.067 |
| | | 2005 | 69 | 34 | 49.3 | 10 | 14.5 | 0.708 |

1. Number of fish measured.
2. Maximum mercury concentration.
3. At York Landing.
4. River mainstem between Clark and Gull lakes.
5. One outlier mercury value excluded.

# APPENDIX 7K
# BIOLOGICAL INFORMATION FOR FISH SPECIES OF COMMERCIAL IMPORTANCE FROM THE AEA OFFSETTING LAKES, 2004–2006

**Table 7K-1: Mean (± standard error [SE]) fork length, round weight, and age of lake whitefish, northern pike, and walleye from the AEA offsetting lakes in 2005–2006**

| Species | Waterbody | Year | Length (mm) | n[1] | Weight (g) | n | Age (years) | n |
|---|---|---|---|---|---|---|---|---|
| Lake whitefish | Caldwell | 2005 | 365.3 ± 18.9 | 25 | 912.0 ± 128.4 | 25 | 8.0 ± 1.0 | 25 |
| | Christie | 2005 | 351.4 ± 16.2 | 23 | 667.4 ± 83.6 | 23 | 7.6 ± 0.8 | 23 |
| | Kiask | 2005 | 321.4 ± 19.3 | 32 | 726.0 ± 82.7 | 25 | 9.4 ± 0.8 | 32 |
| | Limestone | 2005 | 390.0 ± 10.4 | 25 | 840.0 ± 78.1 | 25 | 13.5 ± 1.2 | 25 |
| | Thomas | 2005 | 330.4 ± 22.6 | 27 | 716.7 ± 129.0 | 27 | 7.1 ± 0.9 | 26 |
| | Waskaiowaka | 2005 | 374.0 ± 8.5 | 23 | 828.3 ± 58.3 | 23 | 6.9 ± 0.5 | 23 |
| | Cyril | 2006 | 384.1 ± 15.9 | 51 | 1212.7 ± 117.5 | 51 | 6.9 ± 0.6 | 50 |
| Northern pike | Moose Nose | 2004 | 645.8 ± 22.4 | 39 | 2221.8 ± 260.3 | 39 | 7.0 ± 0.3 | 39 |
| | Christie | 2005 | 602.2 ± 22.7 | 31 | 1838.7 ± 258.3 | 31 | 7.5 ± 0.5 | 31 |
| | Kiask | 2005 | 630.9 ± 29.6 | 19 | 2092.1 ± 357.5 | 19 | 8.2 ± 0.9 | 19 |
| | Thomas | 2005 | 636.4 ± 30.5 | 27 | 2200.0 ± 317.7 | 27 | 8.2 ± 0.7 | 27 |
| | Waskaiowaka | 2005 | 596.7 ± 14.5 | 14 | 1425.0 ± 114.3 | 14 | 6.6 ± 0.5 | 14 |
| | Atkinson | 2006 | 527.4 ± 18.2 | 61 | 1254.1 ± 114.9 | 61 | 5.6 ± 0.4 | 60 |
| | Cyril | 2006 | 519.3 ± 26.1 | 53 | 1509.8 ± 278.0 | 53 | 5.9 ± 0.6 | 53 |
| | War | 2006 | 526.6 ± 18.5 | 50 | 1261.8 ± 188.1 | 50 | 5.8± 0.4 | 50 |
| Walleye | Caldwell | 2005 | 414.4 ± 25.2 | 25 | 1019.0 ± 144.4 | 25 | 9.5 ± 0.9 | 25 |
| | Christie | 2005 | 402.0 ± 17.7 | 25 | 746.0 ± 90.9 | 25 | 10.0 ± 0.8 | 25 |
| | Pelletier | 2005 | 418.2 ± 11.7 | 35 | 776.1 ± 67.3 | 35 | 11.3 ± 0.5 | 32 |
| | Recluse | 2005 | 455.3 ± 11.1 | 26 | 1097.1 ± 89.7 | 26 | 10.7 ± 0.6 | 26 |
| | Thomas | 2005 | 419.3 ± 21.4 | 24 | 1021.9 ± 140.9 | 24 | 8.3 ± 0.9 | 23 |
| | Waskaiowaka | 2005 | 456.5 ± 12.4 | 28 | 1101.8 ± 86.5 | 28 | 12.8 ± 0.6 | 27 |
| | Atkinson | 2006 | 406.7 ± 14.0 | 38 | 858.9 ± 63.1 | 38 | 9.0 ± 0.5 | 37 |
| | War | 2006 | 458.8 ± 11.0 | 45 | 1180.0 ± 69.4 | 44 | 8.2 ± 0.4 | 45 |

1. Number of fish measured.

# APPENDIX 7L
# EXCEEDENCE OF MERCURY THRESHOLDS IN LARGE-BODIED FISH FROM THE AEA OFFSETTING LAKES, 2004–2006

**Table 7L-1: Percentage of lake whitefish, northern pike, and walleye from the AEA offsetting lakes with mercury concentrations of ≥0.2 ppm or ≥0.5 ppm, 2004–2006**

| Species | Lake | Year | n[1] | ≥0.2 ppm | % | ≥0.5 ppm | % | Max[2] (ppm) |
|---|---|---|---|---|---|---|---|---|
| Lake whitefish | Caldwell | 2005 | 25 | 0 | 0.0 | 0 | 0.0 | 0.140 |
| | Christie | 2005 | 23 | 0 | 0.0 | 0 | 0.0 | 0.099 |
| | Kiask | 2005 | 32 | 1 | 4.0 | 0 | 0.0 | 0.224 |
| | Limestone | 2005 | 25 | 4 | 16.0 | 0 | 0.0 | 0.345 |
| | Thomas | 2005 | 27 | 0 | 0.0 | 0 | 0.0 | 0.132 |
| | Waskaiowaka | 2005 | 23 | 0 | 0.0 | 0 | 0.0 | 0.098 |
| | Cyril | 2006 | 51 | 0 | 0.0 | 0 | 0.0 | 0.152 |
| Northern pike | Christie | 2005 | 31 | 20 | 64.5 | 4 | 12.9 | 0.905 |
| | Kiask | 2005 | 19 | 13 | 68.4 | 3 | 15.8 | 0.748 |
| | Thomas | 2005 | 27 | 5 | 18.5 | 4 | 14.8 | 0.748 |
| | Waskaiowaka | 2005 | 14 | 11 | 78.6 | 0 | 0.0 | 0.359 |
| | Atkinson | 2006 | 61 | 15 | 24.6 | 1 | 1.6 | 0.555 |
| | Cyril | 2006 | 53 | 10 | 18.9 | 2 | 5.7 | 0.691 |
| | Moose Nose | 2004 | 39 | 10 | 25.6 | 0 | 0.0 | 0.334 |
| | War | 2006 | 50 | 11 | 22.0 | 2 | 4.0 | 0.542 |
| Walleye | Caldwell | 2005 | 25 | 16 | 64.0 | 5 | 20.0 | 0.875 |
| | Christie | 2005 | 25 | 15 | 60.0 | 6 | 24.0 | 1.138 |
| | Pelletier | 2005 | 35 | 32 | 91.4 | 16 | 45.7 | 0.855 |
| | Recluse | 2005 | 26 | 22 | 84.6 | 7 | 26.9 | 0.711 |
| | Thomas | 2005 | 24 | 7 | 29.2 | 0 | 0.0 | 0.477 |
| | Waskaiowaka | 2005 | 28 | 25 | 89.3 | 6 | 21.4 | 0.610 |
| | Atkinson | 2006 | 61 | 6 | 15.8 | 0 | 0.0 | 0.405 |
| | War | 2006 | 44 | 2 | 4.5 | 0 | 0.0 | 0.211 |

1. Number of fish measured.
2. Maximum mercury concentration.

# APPENDIX 7M
# METHODS FOR FISH SAMPLING AND AGEING, TISSUE TRACE ELEMENT ANALYSIS, AND DATA TREATMENT

# 7M.1 FISH COLLECTION

Fish were collected for trace element analysis from one or two sites at each waterbody) over three- to seven-day periods. Gill nets were set at two sites in Split Lake and the Nelson River between 5 and 10 October 2004 and at one site in Stephens Lake between 11 and 13 October 2004 (Map 7M-1). Gillnet gangs consisted of three 22.9 m (25 yards) long, 2.5 m (2.7 yards) deep panels of 76, 95, 108, and 127 mm (3, 3.75, 4.25, and 5 ") twisted nylon stretched mesh. Gill nets were checked approximately every 24 h.

# 7M.2 FISH PROCESSING

Fish captured were enumerated by species, weighed (± 25 g), measured for fork length (± 1 mm), and classified for sex and maturity. Only fish with red gills (*i.e.*, alive or recently dead) were processed for analysis of trace elements. A sample of skeletal muscle was collected just posterior to the dorsal fin from each fish, immediately frozen, and shipped to Maxxam Analytics Inc. (Winnipeg, MB). Ageing structures, including the dorsal spine, otolith, and fin rays/cleithrum from walleye, whitefish, and pike, respectively, were then collected from each fish.

# 7M.3 LABORATORY ANALYSIS

Samples were digested with nitric acid and hydrogen peroxide and analysed for trace elements (except mercury) by Inductively Coupled Plasma Mass Spectroscopy. Total mercury was analyzed by Cold Vapour Atomic Absorption Spectrometry following nitric and sulphuric acid digestion. All concentrations were expressed on a wet weight (w.w.) basis. Laboratory analysis incorporated standard QA/QC methods, including analysis of spiked samples, blanks, and certified reference materials. Laboratory analytical detection limits are presented in Table 7-4 of the main text.

Dried ageing structures were prepared and analyzed using a variety of methods (Mackay *et al.* 1990). Fin rays or spines were coated in epoxy and sectioned with a Struers microtome saw. Sections were fixed on glass slides with Cytoseal 280 and fish ages were determined by examining the slides under a Wild M3 dissecting microscope. Otoliths were lightly polished with fine sandpaper, immersed in synthetic wintergreen oil, and viewed with a dissecting microscope. Cleithra were cleaned and examined under reflected light to determine ages.

# 7M.4 DATA ANALYSIS

Mean element concentrations were calculated according to species and waterbody. Values reported as less than the analytical detection limit were assigned one-half the detection limit for the purposes of deriving summary statistics. Concentrations were tabulated as means with their standard errors (±SE). The average (±SE) length, weight, and ages of fish sampled were also tabulated by species and waterbody. Differences in mean tissue element concentrations between species and waterbodies (for regional comparisons) were ascertained employing one-way ANOVA. If F-values were significant, differences between individual means were confirmed by Holm-Sidak's pairwise multiple comparison tests. If normality of data distribution or equality of variances could not be achieved by data transformation, Kruskal-Wallis one way ANOVA on ranks was performed. In all cases, significance was established at p less than or equal to 0.05. Statistical analyses were run using Sigma Stat V. 3.01 (SPSS Inc. 2003) software.

# 7M.5 REFERENCES

## 7M.5.1 LITERATURE CITED

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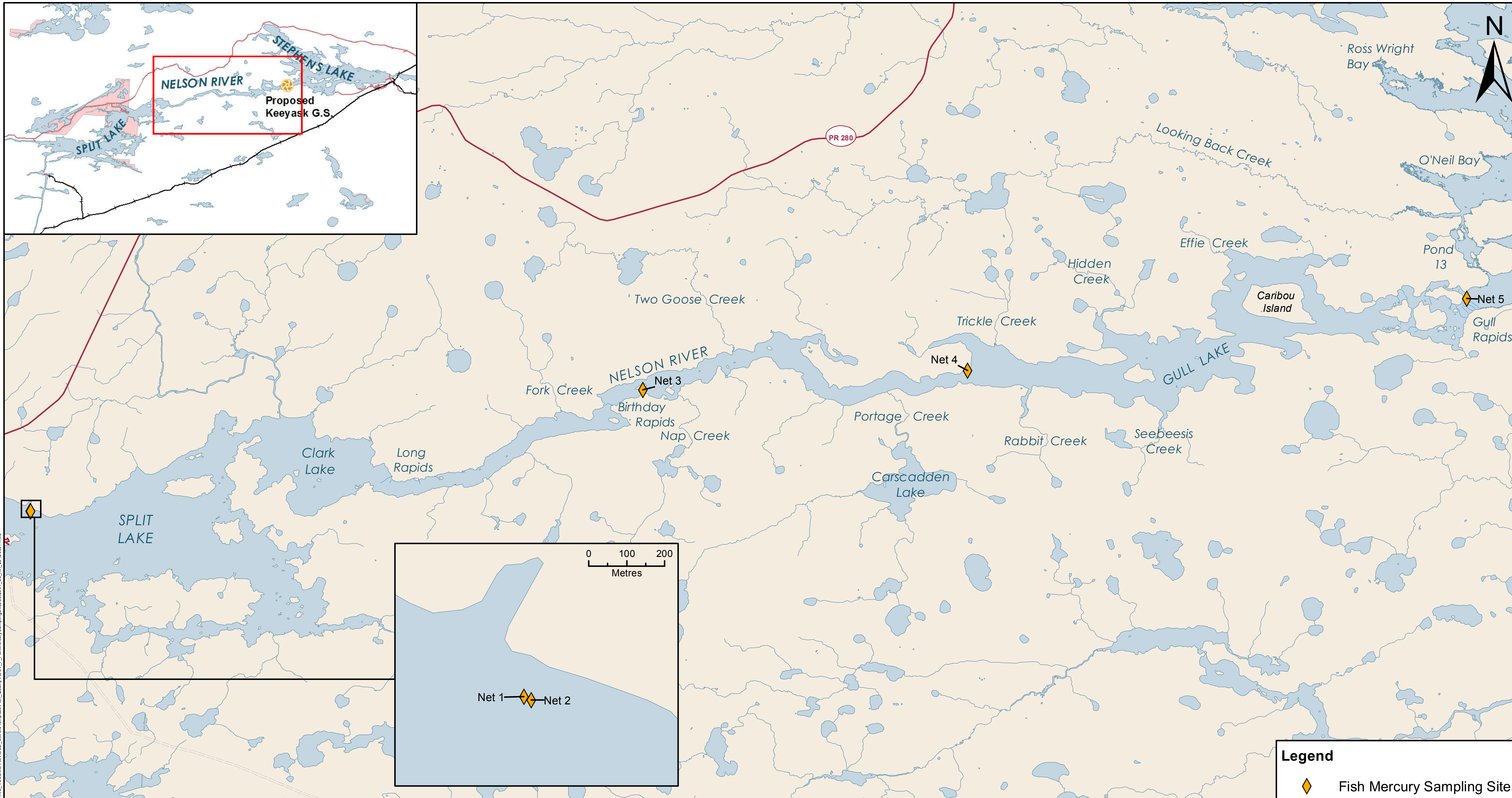


KEEYASK
Hydropower Limited Partnership

0 2.5 5 7.5 10 Kilometres
0 1.25 2.5 3.75 5 Miles

Projection: UTM Zone 15, NAD 83
Data Source: NTS base 1:50 000
Stephens Lake Shoreline - Quickbird@Digitalglobe, 2006
Nelson River Shoreline modelled by Manitoba Hydro

# Fish Mercury Sampling Sites October 2004

Split Lake and Keeyask Areas

# APPENDIX 7N
# WHITEFISH INSPECTION PROTOCOL EMPLOYED BY THE FRESHWATER MARKETING CORPORATION IN WINNIPEG TO DETERMINE THE RATE OF INFESTATION OF LAKE WHITEFISH WITH CYSTS OF *TRIAENOPHORUS CRASSUS* (VERSION OF 9 DECEMBER 1999)

# 7N.1 WHITEFISH INSPECTION PROTOCOL

## 7N.1.1 SCOPE

This protocol defines the policy and procedures for Packers in the Western and Ontario Areas of the Canadian Food Inspection Agency to export lake whitefish to the USA. Whitefish from waters identified in the list of "Lakes with Whitefish Acceptable for Exports to the USA" can be exported by any fisherman, packer, or registered plant without further restrictions. This protocol is available to all fishermen packers or boxers of fresh, whole whitefish.

## 7N.1.2 AUTHORITIES

### 7N.1.2.1 Fish Inspection Regulations

3(1) Subject to subsection (2), these Regulations apply only in respect of fish and containers intended for export and import.

(2) These regulations do not apply to fish that is imported or exported for personal consumption or use.

4. All fish are subject to inspection and an inspector may take samples free of charge for the purpose of inspection

### 7N.1.2.2 U.S. Food and Drug Administration Tolerances for naturally occurring defects

Imported fish shall not contain parasites in excess of 50 cysts per 100 pounds.

## 7N.1.3 POLICY

3.1 Whitefish from lakes not considered "Export" must either originate from a facility listed as "Approved Whitefish Exporter" or must be accompanied by a Certificate of Inspection form the Canadian Food Inspection Agency prior to export of shipment

3.2 Packers that meet the conditions of this protocol will be listed as "Approved Whitefish Exporters". Their inspections for the cestode *Triaenophorus crassus* are sufficient to ensure compliance to standards enforced by the USFDA.

3.3 Packers not on the list will be required to obtain a Certificate of Inspection from the Inspection Branch for each shipment of whitefish from lakes that are not listed as Export.

## 7N.1.4 REQUIREMENTS

4.1 All whitefish must be inspected for acceptable quality and all whitefish not from lakes approved for export to the USA must be inspected for infestation. The Packers must maintain a record of all whitefish inspected and all whitefish exported in an acceptable manner. Those records must identify the following:

- The name of the lake or origin.
- The name of the consignee.
- The date of the shipment.
- The number of fish inspected, the rate of infestation, and the quality.

4.2 Vessels, tubs, transport vehicles and packing/handling/or holding facilities must be maintained in good condition and meet the requirements of the Fish Inspection Regulations.

4.3 The fish must be packed in sufficient, clean ice.

4.4 The fish must be properly graded and culled and not tainted, decomposed or unwholesome.

4.5 The containers must be properly labelled to identify the net contents, species, lake of origin and packer.

## 7N.1.5 MONITORING

5.1 Compliance to the requirements of Section 3 will be routinely monitored by the Fish Inspection Branch of the Canadian Food Inspection Agency. If effective corrective action with problems identified as a result of an inspection is not implemented within 48 h, it will result in the removal of the Packer form the list of Approved Whitefish Exporters.

## 7N.1.6 FORMS AND SAMPLING PROCEDURE

The following forms and sampling schedule can be utilized by the packer to implement this Protocol but they are not mandatory:

- Sampling Schedule.
- Record of lots exported to the USA.
- Whitefish Inspection Reports.
- Facility Inspection Reports and Daily Sanitation Records.

# 7N.2 WHITEFISH INSPECTION PROCEDURES AND SAMPLING SCHEDULE

## 7N.2.1 PROCEDURE

1. Estimate the total number of whitefish in the lot to be inspected.
2. Record the total weight of the fish in sample to the closest tenths of a pound.
3. Fillet the whitefish. Make thin slices (no thicker than ¼ ") at right angles to the length of each fillet.
4. Count the total number of cysts of *Triaenophorus crassus* in the fillets and calculate the RI according to the following formula:

   RI = (number of cysts/pound of fish samples) × 100

The RI must be less than 50 cysts per 100 pounds if the lot is to be exported to the USA.

# APPENDIX 7O
# METHODS AND DATA ANALYSIS OF FISH PALATABILITY STUDIES

# 7O.1 FISH PREPARATION AND TASTE TEST

The fish species sampled for the palatability studies included lake whitefish, walleye, and northern pike of similar size (whitefish: 1,400–2,000 g; pike: 2,700–3,000 g; walleye: 700–1,000 g) that were collected with gill nets set three to five weeks prior to taste-testing. Fillets (northern pike and walleye) or scaled and eviscerated whole fish (whitefish) were prepared on-site and stored frozen until taste sample preparation. The samples consisted of small (less than 10 g) pieces of fried (pike and walleye) or boiled and deboned flesh (whitefish) that were presented to panellists three to four at a time, with each piece representing a coded sample from each lake. Fish species were evaluated on separate days. Between 23 and 48 panellists, stratified by age (16–35, 36–55, greater than55 years) and consumption habit (eat fish every day, three to four times per week, one to two times per week, three times per month) tasted fish at each community. Panellists expressed their acceptability of the fish based on a seven-point category scale from "like very much" (7) to "dislike very much" (1) (for a full list of categories see Ryland and Watts 2002a; 2004a, b).

# 7O.2 DATA ANALYSIS

Analysis of variance followed by, if applicable, Duncan's test for means comparisons was conducted on the numerical categories of fish acceptability using PROC GLM (SAS, PC Version 8.2) to determine if there were significant differences among the acceptability of fish of the same species from different locations or among panellist age categories. The significance level of the palatability studies at Bird was set at p=0.10 and at p=0.05 for York Landing and Split Lake.

# 7O.3 REFERENCES

## 7O.3.1 LITERATURE CITED

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SAS. 1999. SAS for Windows, V.8.2. SAS Institute Inc. Carry, NC.