# SITE C CLEAN ENERGY PROJECT

## VOLUME 2 APPENDIX J PART 1

## MERCURY TECHNICAL SYNTHESIS REPORT

***Prepared for:***

BC Hydro Power and Authority
333 Dunsmuir Street
Vancouver, BC
V6B 5R3

***Prepared by:***

Azimuth Consulting Group Partnership
218-2902 West Broadway
Vancouver, BC
V6K 2G8

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AZIMUTH

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***FINAL REPORT***

***Prepared for BC Hydro Power and Authority***
***Prepared by Azimuth Consulting Group Partnership***

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Lead Author: Randy Baker

Section Contributors:

Dr. D. Bodaly – Introduction

Dr. R. Turner – Baseline Terrestrial

Randy Baker – Baseline Aquatic

Dr. W. Jansen – Reservoirs Matrix

## AUTHORSHIP

Randy Baker, M.Sc. ............................................Principal, Azimuth Consulting Group Partnership
Dr. Ralph Turner. ...............................................Associate, Azimuth Consulting Group Partnership
Dr. Wolfgang Jansen. ...............................................................North/South Consultants Winnipeg
Dr. R.A. Bodaly. ........................................................................................Consultant to Azimuth

# EXECUTIVE SUMMARY

Over many hundreds of years, inorganic mercury (Hg) captured from the atmosphere by the leaves and needles of plants falls to the ground and accumulates, to become sequestered and concentrated into organic soils. When reservoirs are created, bacterial decomposition of flooded organic material causes a small amount of the inorganic mercury sequestered in these soils to be converted into organic or methylmercury (MeHg). Methylmercury is easily absorbed by aquatic organisms that feed in sediments and becomes accumulated and concentrated at progressively higher concentrations moving up the aquatic food web. Highest MeHg concentrations occur in large, old fish with piscivorous feeding habits. Methylmercury is accumulated almost exclusively via diet and exposure by humans and wildlife to mercury is primarily through fish consumption. Fish MeHg concentrations are hundreds of times more concentrated than in other herbivorous wildlife (e.g., moose, deer) and domestically consumed species (cattle, chicken, pig, etc.) and many millions of times more concentrated than in water.

The objective of this report is to summarize baseline mercury and methylmercury concentrations in environmental media in the terrestrial (soils, vegetation) and aquatic environments (water, sediment, aquatic invertebrates, fish) within the Peace River technical study area, to provide a basis for determining how creation of the proposed Site C reservoir will alter methylmercury concentrations, with a focus on fish. The technical study area includes the proposed Ste C reservoir and its major tributaries and the Peace River downstream to Many Islands. Many Islands is the furthest downstream location that the majority of fish species routinely move within the Peace River, downstream of Peace Canyon Dam. Some fish downstream of the proposed Site C dam can potentially be exposed to fish with elevated MeHg concentrations entrained out of the reservoir.

To address Hg in soils, Terrestrial Ecosystem Mapping (TEM) was used to stratify the relative spatial abundance (ha) of different forest types that were sampled to define baseline total Hg and organic carbon in soil and vegetation. Dominant tree species (spruce, balsam, willow, alder), shrubs (sarsaparilla, prickly rose, willow, and dogwood) and grasses (horsetail, sedge, reeds, cattail) were measured, including in Watson Slough, a wetland and area with naturally higher mercury concentrations. Total Hg concentration in all plant tissues was very low, in most cases barely above the detection limit of 0.005 mg/kg dw. The most abundant shrub (<0.008 mg/kg dw) and tree species (<0.005 to 0.019 mg/kg) had low and similar mercury concentrations.

The average total Hg content of all organic soils was 0.079 $\pm$ 0.03 mg/kg dw, ranging from 0.023 to 0.173 mg/kg dw. Mercury in soils was positively correlated with high organic content (>30%) with wetland (Watson Slough) having higher concentrations. Soil thickness,

organic content and Hg concentration were integrated across the proposed flood area to develop bulk density estimates for carbon (kg C/m$^2$) and mercury (µg Hg/m$^2$) for use in Hg modeling. On an aerial basis, the Hg density was estimated at 70 µg Hg/m$^2$. Mercury concentrations in soils and vegetation within the Site C reservoir area are low, typical of remote, organic soils removed from natural mineralization.

Key parameters in the aquatic environment that influence Hg methylation within the aquatic food web are hydrology, limnology and specific water and sediment chemistry parameters. These parameters and total and MeHg concentrations were measured in all aquatic environmental media within the Dinosaur Reservoir and the Peace River between Peace Canyon Dam and the proposed Site C dam. The vast majority (>95%) of water within the Peace River downstream of Peace Canyon Dam is discharged from Williston Reservoir and has a considerable influence on water chemistry and ecology of this technical study area. Williston Reservoir water is nutrient poor (ultraoligotrophic), cold (<14$^o$C), well oxygenated, of moderate to slightly basic pH (7.8 – 8.2), low in organic carbon content (<2 mg/L) and has low suspended solids concentrations (<3 mg/L) during nearly all times of the year.

Total Hg concentration in Peace River water downstream of the Peace Canyon Dam to the proposed Site C dam site was consistently near 1.0 ng/L (parts per trillion or 0.001 µg/L). Major tributaries to the Peace River, in particular Halfway and Moberly, contribute more inorganic Hg than is present in the mainstem, adhered to sediment particles during high total suspended solids events during flood or freshet periods; otherwise, these concentrations are very low. Methylmercury concentration in the Peace River was consistently less than the detection limit of 0.05 ng/L. Total mercury in sediment from within Dinosaur Reservoir and the Peace River downstream to Site C was low (0.03 to 0.17 mg/kg dw) when detectable and similar to the range observed upstream, in Williston Reservoir. Methylmercury was 1 – 2% of total Hg ranging from 0.13 to 0.27 µg/kg in Dinosaur Reservoir and 0.57 to 1.8 µg/kg in the Peace River.

In the Peace River technical study area, zooplankton total Hg ranged from 0.004 to 0.009 mg/kg ww, similar to what was observed in Williston Reservoir from earlier studies. Methylmercury concentration in Peace River zooplankton was much lower (0.0001 to 0.0007 mg/kg ww). In Dinosaur Reservoir, total Hg in zooplankton ranged from 0.001 to 0.006 mg/kg ww and MeHg ranged from 0.0003 to 0.001 mg/kg ww, averaging 30% of total Hg. These concentrations are within the low range for plankton from remote lakes unaffected by anthropogenic or natural sources of Hg.

Total Hg from a composite of various taxonomic groups of benthos ranged from 0.010 to 0.023 mg/kg ww. Methylmercury ranged from 0.002 to 0.020 mg/kg ww and averaged 20% of total Hg. In 2011, Peace River mainstem benthos total Hg concentration ranged from 0.046 to 0.082 mg/kg, with MeHg concentrations ranging from 20 – 37% of the total.

Chironomid larvae (0.06 mg/kg total; <0.04 mg/kg methyl and water boatmen (Corixidae) had slightly higher Hg concentrations (0.05 mg/kg total and 0.04 methyl) than did the more common mayflies (Ephemeroptera) and caddisflies (Trichoptera) (0.009 - 0.017 mg/kg total; 0.003 – 0.005 methyl). These concentrations are similar to, or lower than, what has been observed elsewhere in Canadian rivers and lower than what is observed in reservoirs.

The main factors influencing bioaccumulation of MeHg by fish are MeHg concentration of prey, fish size, age, growth rate, genetics and reproduction. Methylmercury is accumulated by fish almost exclusively from dietary sources, thus body burden concentration is highly dependent on prey composition (benthos, small fish) and trophic position. This document summarizes all fish mercury concentrations gathered between the early 1990s and the present time within the Peace River technical study area. Tissue Hg analysis has mainly focused on the dominant species including bull trout, lake trout, Arctic grayling, burbot, lake whitefish, mountain whitefish, rainbow trout, longnose sucker and redside shiner, but also extending downstream of Site C into Alberta (northern pike, walleye, goldeye, burbot).

Bull trout Hg concentration ranged between 0.03 – 0.34 mg/kg with a mean of 0.07 mg/kg, less than from Dinosaur Reservoir (0.12 mg/kg). Dinosaur Reservoir lake trout had a mean tissue mercury concentration of 0.09 mg/kg. Mean Hg concentrations in mountain whitefish and rainbow trout from Peace River and Dinosaur Reservoir were also low (0.03 and 0.04 mg/kg respectively). Similarly, mercury in longnose sucker from Peace River to the Site C dam (0.05 mg/kg) and downstream to Alberta (0.06 mg/kg) were low and similar. Redside shiner, a common prey species, averaged 0.05 mg/kg Hg. Mean Hg concentrations of all fish species in the Peace River technical study area were less than 0.08 mg/kg with nearly all fish less than 0.20 mg/kg. These concentrations are lower than for the same species of a similar size in all other BC lakes and reservoirs.

The final chapter of this document is the Canadian Reservoirs Comparison Matrix. This section compares the main physical, chemical and ecological features and water / biota Hg concentrations from many other Canadian reservoirs to Site C reservoir. Based on this assessment, all of the key physical, chemical or ecological baseline parameters or predicted changes within the proposed Site C reservoir are associated with increases in fish Hg concentrations of less than a 3x increase above baseline concentrations. The proposed Site C reservoir has an oligotrophic upstream reservoir and is projected to have a relatively small increase in reservoir area relative to original area, low water residence time, low nutrients, alkaline pH, low water temperature, high oxygen and low baseline Hg and MeHg in water and biota. Each of these suggests that the magnitude of increase in fish Hg concentrations will be small relative to what has been observed in most other reservoirs elsewhere in Canada.

## ABBREVIATIONS AND ACRONYMS

CCME....Canadian Council for Ministers of the Environment
DOC....dissolved organic carbon
EIS....Environmental Impact Statement
ha....hectares
Hg....mercury
MeHg....methylmercury
mg/kg....milligrams per kilogram (parts per million)
mg/L....milligrams per liter (parts per million)
mg/m2....milligrams per square meter
ng/L....nanograms per litre (parts per trillion)
RESMERC....Reservoir Mercury model
TEM....terrestrial ecosystem mapping
TOC....total organic carbon
TSS....total suspended solids
µg/kg....micrograms per kilogram (parts per billion)
µg/m$^2$....micrograms per square meter

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# 1 PEACE RIVER MERCURY TECHNICAL SYNTHESIS REPORT

## 1.1 Background

BC Hydro has proposed to construct the Site C Clean Energy Project (the Project), a third dam and hydroelectric generating station on the Peace River in northeastern British Columbia (BC). The creation of a new reservoir downstream from the existing Williston and Dinosaur reservoirs would inundate and flood terrestrial soils on river islands and the adjacent forest within the Peace River valley and the mouths of some rivers. Flooding of terrestrial soils and vegetation causes organic material to decompose and break down over time, releasing nutrients and other materials into the water column to be absorbed by plants and animals in the food chain. One of these materials is mercury (Hg).

Over many hundreds of years, inorganic mercury, captured from the atmosphere by the leaves and needles of plant, falls to the ground, accumulates and is sequestered and concentrated into soils. In the 1970s it became known that flooding terrestrial areas to form reservoirs created conditions that were favourable for increased rates of ‘methylation’ of inorganic mercury during soil decomposition by bacteria. That is, a small portion of the inorganic mercury in soil is transformed into an organic form of mercury called methylmercury (MeHg) by a particular group of bacteria called sulfate reducers. Because MeHg is an organic compound, it is much more easily absorbed and accumulated by low food web biota. Over time, MeHg becomes increasingly concentrated at increasing steps up the food chain, with highest concentrations in fish (Potter et al. 1975; Abernathy and Cumbie 1977; Bodaly and Hecky 1979; Bodaly et al. 1984).

Inorganic and methylmercury are naturally occurring elements that are found in low concentrations in all media including water, soil, sediment, plants and at all levels of aquatic and terrestrial food webs. Reservoir creation will temporarily increase mercury and methylmercury concentrations in fish and other aquatic animals. Depending upon baseline conditions and the relative magnitude of increase, there is a potential for increased risk of exposure to methylmercury by wildlife that feed on fish and by humans who may consume fish from the reservoir. This document summarizes aquatic baseline conditions within the Peace River and terrestrial conditions within the proposed Site C Reservoir area which are critical to our understanding and Environmental Assessment of issues surrounding mercury for the proposed Project.

## 1.2 Objective

The objective of this Peace River mercury technical synthesis report is to document baseline mercury and methylmercury concentrations in environmental media in the terrestrial (soils, vegetation) and aquatic environments (water, sediment, aquatic invertebrates and fish) and to provide a basis for determining how creation of the proposed Site C reservoir will alter methylmercury concentrations, with a focus on fish. Methylmercury levels in the aquatic environment and food web are expected to increase temporarily following flooding of the proposed reservoir and to determine the degree to which this may occur relative to baseline, an integrated assessment approach was taken using three lines of evidence including:

- a detailed comparison of the physical, chemical and ecological features of Site C with several other Canadian reservoirs contained within this document,
- regression modeling whereby simple parameters (reservoir area relative to original area; water turnover rate) are found to correlate well with the degree of increase in fish Hg above baseline in many other Canadian reservoirs (Harris and Hutchinson 2011 in Volume 2, Appendix J, Part 3), and
- complex, mechanistic mercury modeling (RESMERC Volume 2, Appendix J, Part 3) that models changes in mercury and methylmercury in a wide variety of environmental media at various time intervals.

Information gathered across these lines of evidence was used to determine the magnitude of change in fish methylmercury concentrations within the proposed Site C reservoir relative to baseline. This value informed the Human Health Risk Assessment to assess potential risk from incrementally higher dietary exposure to methylmercury from fish consumption (Volume 2, Appendix J, Part 2). This risk-based approach quantitatively determines the number of weekly meals of fish, by species that can be safely consumed without exceeding Health Canada's consumption guidelines in a post-Site C Project environment.

This technical synthesis document contains the following information:

- A general introduction to mercury, report objectives and structure, and our study approach
- A general introduction to mercury in the environment, including the following topics: the relationship between mercury, methylmercury and reservoir creation; our understanding of the science in real-life and experimental reservoirs; downstream transport of mercury; management options and possible direct health effects to fish and mercury and public health (i.e., why are we concerned?).

- A summary of baseline terrestrial data on mercury and carbon in soils and vegetation within the inundation area of the proposed Site C reservoir, relative to soils elsewhere in boreal forests of northern Canada. A mass balance accounting of the amount of organic carbon and mercury in soils is also calculated for the within the proposed Site C reservoir that is available for methylation. This is a key input parameter for the RESMERC model (Volume 2, Appendix J, Part 3).
- A summary of baseline aquatic data on mercury and methylmercury and key water quality parameters in water, sediment, zooplankton, benthic invertebrates and fish within the technical study area within the proposed reservoir and downstream. This information establishes the existing conditions from which changes in mercury in aquatic environment media may change.
- Characterization of baseline and physical, chemical and ecological conditions within the proposed Site C reservoir relative to conditions found in a range of hydroelectric reservoirs elsewhere in Canada and implications for mercury methylation within the proposed Site C reservoir. This is described as the Canadian Reservoirs Comparison Matrix and is a key component of the integrated assessment of the change in mercury in Site C reservoir fish.

# 2 INTRODUCTION TO MERCURY DYNAMICS

## 2.1 General Introduction to Mercury

Mercury (Hg) is an element that occurs naturally at low concentrations in a large variety of chemical forms in all environmental compartments, including the atmosphere (as a gas and adhered to small particles) and all terrestrial (soil, vegetation, insects, mammals, etc.) and aquatic (water, sediment, invertebrates, fish) media. Mercury is a rare element of the Earth's crust, but can occur in very concentrated deposits of mercuric sulphide (Hg-S) or cinnabar. Cinnabar has been mined for centuries and processed to acquire the familiar liquid mercury for use in gold and silver mining. In its elemental form, $Hg^0$, mercury has the peculiar properties of occurrence in the liquid state at room temperature, it forms amalgams with other metals such as silver and gold, and because of its unique physical properties has many industrial, chemical and health applications. These include use in electrical switching devices, pressure measuring devices, fluorescent light bulbs, and dental amalgams, as well as use in chlor-alkali chemical plants.

About half of the natural annual global atmospheric contribution of inorganic mercury is from degassing (as $Hg^0$) from weathering of the earth's crust and from volcanoes, forest fires, and evacuation from freshwaters and oceans (Mason et al. 1994; Morel et al. 1998; Boening 2000). The other half is anthropogenic, from burning fossil fuels (especially coal), industrial loss, metal smelting, from crematoria and from small-scale gold mining operations where mercury is widely used to amalgamate gold. In the atmosphere, elemental mercury ($Hg^0$) is oxidized to the mercuric ion (as Hg(II)) and captured by rain and snow to be deposited as wet deposition on land and water (Mason et al. 1994). Mercury also adheres to particles that accumulate on vegetation and in soil.

Mercury is continually cycling in the environment, alternating between the oxidized HgII form and reduced back to $Hg^0$. Ultimately, a portion of the pool of atmospheric mercury is deposited to the earth and accumulates in soils and wetlands, eventually entering the sediment pool of mercury in freshwater and marine systems. Lindberg et al. (2007) estimated that since the Industrial Revolution, global atmospheric concentration and transport has tripled. This may have caused fish mercury concentrations to increase, even in very remote areas far from industrial development.

## 2.2 Reservoir Creation and Methylmercury

The flooding of terrestrial soils and vegetation to create reservoirs during hydroelectric development provides a source of organic nutrients and inorganic mercury that are broken down by bacteria in the flooded environment. Decomposition of this ‘new’ organic material by bacteria results in the creation of a small amount of organic or methylmercury ($CH_3Hg$) that is absorbed by bacteria. Once incorporated into bacterial tissue, it is in the base of the food chain and available to be consumed and accumulated at progressively higher concentrations moving up through the food web. Because methylmercury is absorbed more rapidly than it can be excreted, it accumulates in the body over time. Thus, methylmercury concentrations are higher in larger, longer-lived animals, especially those at the top of the food chain (Hall et al. 1997). Because most fish are carnivorous and feed over multiple levels of the food web, they consume and accumulate more methylmercury than any other animal. Consequently, the vast majority of exposure by humans to mercury is through fish consumption and not through other means. For example, the concentration of methylmercury in fish is typically hundreds of times more concentrated than in other herbivorous animals including domestic (cattle, pigs, sheep) and wild animals such as deer and moose.

The first cases of elevated mercury concentration in fish in new reservoirs were documented in temperate areas of the USA (Potter et al. 1975; Abernathy and Cumbie 1977; Cox et al. 1979) and from Southern Indian Lake, a boreal reservoir in northern Canada (Bodaly and Hecky 1979). Since then, most research on mercury in reservoirs has been conducted in boreal hydroelectric reservoirs in Canada (Bodaly et al. 2007; Schetagne et al. 2003) and Scandinavia (Lodenius et al. 1983). The inundation of organic soils and to a much lesser extent, standing vegetation, introduces inorganic mercury and nutrients to the water, which in turn increases microbial production of methylmercury in the flooded soils (Bodaly et al. 1984; Kelly et al. 1998; Bodaly et al. 2004). This increases the supply of methylmercury to the base of the food chain and to fish that is independent of local atmospheric loading of inorganic mercury (Munthe et al. 2007).

Canadian studies on the evolution of fish mercury concentrations after reservoir creation come mainly from Québec (Schetagne et al. 2003; Schetagne and Verdon 1999), Manitoba (Bodaly et al. 1984; 2007), and Labrador (Bruce and Spencer 1979; Bruce et al. 1979).

Long-term data from studied reservoirs in Québec and Manitoba agree in the general pattern of mercury concentration change over time. Data show that mercury concentrations in adults of large-bodied, relatively long-lived species increase quite rapidly, with peak concentrations three to eight years after reservoir impoundment, after which levels decline relatively slowly to eventually reach pre-impoundment (or baseline) concentrations approximately 20 to 30 years later (Schetagne et al. 2003; Munthe et al. 2007). Predatory species were found to have the highest peak mercury concentrations, take the longest to reach maximum levels, and take longer to return to a baseline level. While these general timelines in newly created reservoirs are well established, substantial variability exists among reservoirs in the number of years for fish to reach peak concentrations, the magnitude of those peaks, and the return time to background levels (Bodaly et al. 1984; 2007; Schetagne et al. 2003). These differences are related to reservoir-specific conditions, such as filling and water residence time (Schetagne et al. 2003), ratio of reservoir area to original wetted area, chemical composition of the newly flooded soil, pH, amount of flooded wetland or peatland, reservoir morphometry and temperature and oxygen regime, and invertebrate and fish community structure, particularly the number of trophic levels. The physical, chemical and ecological factors that contribute to this are fully explored within the Canadian Reservoirs Comparison Matrix, Section 5.

## 2.3 Methylmercury Dynamics

Methylmercury ($Hg\text{-}CH_3$) is one of several organic forms of mercury that is present in very small concentrations in all environmental media and is the most common chemical form of mercury found in fish, typically comprising about 95% of the total concentration (Bloom 1992). It is the production and bioaccumulation of methylmercury that is of particular concern for toxic effects to both fish-eating wildlife and humans (Clarkson et al. 2003; Mergler et al. 2007), because this is the form that is most easily bioaccumulated and biomagnified through food chains. Methylmercury accumulates in food chains so efficiently that fish typically have $10^7$ to $10^8$ (> 10 million times) higher concentrations in their tissue than the water that they live in (Sandheinrich and Wiener 2011). Inorganic forms of mercury do not transfer or bioaccumulate as efficiently in food chains, as methylmercury does (Watras et al. 1998; Pickhardt et al. 2002).

The production of methylmercury from inorganic mercury (mainly Hg(II)) is a key step in the environmental cycling of mercury in all aquatic systems because the supply of methylmercury to the bottom of the food chain is usually an important limiting factor on the concentration of mercury in fish and shellfish (Munthe et al. 2003). As discussed in Section 5, the rate and magnitude of methylmercury production is known to be affected by many factors. These include the bioavailability of inorganic mercury (Orihel et al. 2007; Munthe et al. 2007), temperature (Korthals and Winfrey 1987), pH (Miskimmin et al. 1992), sulphate availability (Gilmour and Henry 1991), oxygen (Gilmour and Henry 1991) and dissolved organic carbon (DOC) concentration (Barkay et al. 1997; Miskimmin et al. 1992). Microbial metabolic rates, as influenced by the availability of organic carbon substrates for bacterial growth influence rates of mercury methylation (Furutani and Rudd 1980; Miskimmin et al. 1992; Pak and Bartha 1998).

Methylmercury can also be degraded or demethylated in aquatic systems, again via microbial processes, (Pak and Bartha 1998; Korthals and Winfrey 1987; Miskimmin et al. 1992) mainly in lake sediments. Demethylation is also carried out by sulphate reducing and methanogenic bacteria (Pak and Bartha 1998), as well as abiotically in surface waters from radiation in sunlight (Sellers et al. 1996). Concentrations of methylmercury in sediments and water are a reflection of the balance between methylation and demethylation rates and this is taken into account in our modeling efforts. In the early life of all new hydroelectric reservoirs, the rate of methylation is far greater than the rate of demethylation.

Fish acquire virtually all of their methylmercury from food (Hall et al. 1997; Rodgers 1994; Harris and Bodaly 1998). The main factors influencing bioaccumulation rates of mercury are mercury concentration in prey, age and size of the fish, growth rate and reproduction. Furthermore, a shift in diet from invertebrates to fish or from small fish to larger fish as a fish gets older and larger will further increase accumulation of mercury by the predator. Changes in growth rate can also influence mercury in fish. Young fish and fish with faster growth rates are more efficient at converting food into biomass and will have a proportionally lower rate of accumulation of mercury than old, slow growing fish, a phenomenon known as ‘growth dilution’ (Simoneau et al. 2005). Similarly, fish with low condition factor (i.e., lower body mass to length) will also have a higher rate of mercury accumulation and is related to reverse growth dilution. For example, in Lake Mead, USA it was found that fish in very poor condition had abnormally high concentrations of mercury (Cizdziel et al. 2002).

Models that simulate the accumulation of mercury in fish are based on bioenergetics models that include rates for feeding, respiration and other energetic outputs, growth, condition and reproduction (Rodgers 1994; Trudel and Rasmussen 1997; Harris and Bodaly 1998). When fish ingest food, methylmercury is very efficiently assimilated during digestion in the alimentary tract (Rodgers 1984; Pickhardt et al. 2006). Methylmercury assimilated from food is taken up by the intestine, transferred to blood and most enters the red blood cells (Oliveira Ribeiro et al. 1999), eventually to become bound to cysteine in proteins in fish muscle (Harris et al. 2003). Fish can excrete or depurate methylmercury during normal respiration (albeit very slowly) or during egg production (Van Walleghem et al. 2009; Trudel and Rasmussen 1997; Johnston et al. 2001).

Baseline mercury concentrations in environmental media and, in particular, fish, are discussed in Section 4.0. Every system is different and unique and will respond differently when flooded. Methylmercury generation rates and bioaccumulation of methylmercury by biota is dependent on many factors and most of these are explored in detail in Section 5.0, the reservoir comparison matrix, to put Site C in context with other Canadian reservoirs.

## 2.4 Experimental Reservoir Studies

There have been two intensively investigated experimental reservoir scenarios carried out at the Experimental Lakes Area in northwestern Ontario. The objective of these investigations was to gain an understanding of the mechanisms, controlling factors and dynamics of methylmercury creation and its movement up the food chain created in different types of reservoirs—a wetland and an upland boreal forest. The first (Experimental Lakes Area Reservoir Project) flooded a 2 hectare (ha) pond and wetland to create a 17 ha reservoir and evaluated impacts on mercury cycling (and greenhouse gas production) for nine years (Kelly et al. 1998; St. Louis et al. 2004). The second (Flooded Uplands Dynamics Experiment) flooded three areas of upland boreal forest of surface area (0.5 to 0.7 ha) and followed mercury and greenhouse gases for five years after flooding (Bodaly et al. 2004; Hall et al. 2005). In these experimental reservoirs, large increases in the production of methylmercury immediately following flooding were observed. Methylmercury production slowed noticeably with time after the initial flooding, especially in the upland reservoirs (Hall et al. 2005; St. Louis et al. 2004). It appeared that the severity of the problem of methylmercury production was similar in the flooded wetland system as compared to the flooded uplands, but that the longevity of elevated methylmercury production may have been greater in the case of flooded wetlands (Bodaly et al. 2004). These studies have greatly enhanced our understanding of methylmercury dynamics in new reservoirs and have been applied to the RESMERC model and ultimately, the Site C Project.

## 2.5 Downstream Transport of Mercury

Newly flooded reservoirs are also known to export inorganic and methylmercury downstream, dissolved in water, adhered to sediment particles and contained within biota (i.e., plankton and fish). Schetagne et al. (2000) determined that a new northern Québec reservoir discharged large amounts of methylmercury in the dissolved and particulate phases of water. Also, it has been found that mercury concentrations in some fish become amplified downstream of some reservoirs, such as in the Churchill River downstream of Smallwood Reservoir in Labrador (Anderson 2011), Southern Indian Lake in Manitoba

(Bodaly et al. 1997) and downstream of Caniapiscau Reservoir, part of the La Grande complex in Quebec (Schetagne et al. 2000). The main reason for the increase in mercury in downstream fish appears to be related to a shift in dietary preference or trophic position. If downstream fish switch from a diet of invertebrates to fish that are stunned, injured or killed from passage through turbines (Brouard et al. 1994), this will increase their mercury intake and consequently, body burden concentrations. The increase does not appear to be related to export of methylmercury dissolved in water as food remains the dominant source of methylmercury in fish (Hall et al. 1997). In upstream reservoirs with a low turnover and large settling capacity the implications for downstream fish appear to be reduced.

The potential issues regarding downstream transport of mercury in a large river system are two-fold. First, in a highly dynamic fluvial environment like the Peace River, sediment and organic particles with adhered mercury exported from the new reservoir may be transported a great distance downstream, before settling out in deposition areas in side channels or terminal lakes and wetlands. Depending on local conditions in depositional areas, this may or may not favour methylation of dissolved and particulate inorganic mercury. Thus, there may be an 'export' of the reservoir phenomenon downstream to discrete depositional areas; however, the magnitude of dilution of this effect cannot be discounted.

Second, depending on the magnitude of fish passage through the turbines from the upstream reservoir, fish and fish-eating wildlife may be exposed to fish with mercury concentrations above background that originated from the new reservoir. Increased mercury concentration in downstream fish will depend on the extent of fish movements in the river downstream of the dam and more importantly, whether a shift in dietary preference occurs (e.g., a shift to fish consumption). An increase in exposure to mercury by wildlife or humans downstream will only occur under certain conditions. That is, fish mercury concentrations must increase within the reservoir and a sufficiently large number of fish must be transported downstream to provide fish with an abundant and consistent food source that would cause them to shift feeding from invertebrates over a long enough time that this would result in elevated tissue concentrations.

In either scenario, the magnitude of increase in mercury, if any, is dependent upon a wide range of site-specific physical/chemical and biological conditions that may or may not favour methylation. These include, but are not limited to, the magnitude and timing of organic material and sediment transported out of the reservoir, extent and location of deposition, local chemistry (temperature, pH, oxygen), local invertebrate population and food chain length and fish population species composition, movement/migration dynamics, growth rates and bioenergetics. These issues will be explored further as part of this environmental assessment.

## 2.6 Strategies to Lower Mercury in New Reservoirs

A number of management strategies to lower the magnitude of mercury increase in fish, and/or reduce the timeframe that fish mercury is elevated above baseline have been considered and explored. Recently, Mailman et al. (2006) reviewed a number of strategies to reduce or manage the undesirable effects of river impoundment on mercury concentrations in fish. Those include:

- *Controlled burning prior to flooding* – Controlled burning would reduce both inorganic mercury and organic carbon and thus reduce stimulation of methylation when burned areas are flooded. Mailman and Bodaly (2005) showed large reductions in organic carbon and mercury (inorganic and methylmercury) in burned plants and soils. However, while reducing water methylmercury concentrations, concentrations in biota were not correlated to methylmercury in water, possibly due to lower bioavailability from higher dissolved organic carbon. Furthermore, burning to reduce mercury must be balanced against greenhouse gas contributions and loss of potential nutrients and fish habitat by leaving vegetation and soils unburned.

- *Removal of vegetation* – Removal of vegetation is the most commonly undertaken measure to reduce mercury. While removal of standing vegetation reduces the amount of organic carbon available to stimulate methylation after flooding, the vast majority of the carbon and in particular, mercury, is actually locked in the organic soils. Only a very small portion of the total biomass of mercury is contained within the living or standing vegetation and any reduction in fish mercury from reservoir clearing would be relatively small. To determine the influence of vegetation removal, two modelling scenarios were run, one with and one without reservoir clearing. This is summarized in the RESMERC mercury modeling report (Volume 2, Appendix J, Part 3).

- *Capping or removal of organic soils* – Capping or removal of highly organic soils such as wetlands, sloughs, marshes and fens prior to flooding would eliminate such soils as contributors of methylmercury generation as part of site preparation. Capping of highly organic soils with a layer of glacial till would effectively isolate the organic-rich materials from ponded water and thus reduce or halt stimulation of methylation with these materials. Similarly, physical removal of wetland, marsh, or peat soils would also eliminate these high carbon, high mercury soil / sediment as a methylation source.

- *Removal of Predatory Fish* – This technique has been suggested as a means of reducing the bioavailable pool of methylmercury in biota. The amount of methylmercury 'tied up' in fish is low relative to other environmental compartments).

## 2.7 Use of Predictive Mercury Models in Hydroelectric Reservoirs

There has been a considerable amount of research and monitoring of mercury in northern Canadian hydroelectric reservoirs over the last few decades. These have focused on measuring methylmercury production, biomagnification and bioaccumulation in food web organisms. An understanding of this issue has evolved to the point where it is possible to construct mechanistic models that simulate the effects of reservoir creation on methylmercury dynamics in the aquatic environment. These models can be used to predict increases in mercury in environmental media, including water, invertebrates and fish in new reservoirs along a broad time-scale. There are currently four models available, all of which have been developed based on data from reservoirs in central and eastern Canada. Two of the models are based on linear regression equations where the input parameters for these models are estimates of flooded area and water flow/residence time (e.g., Harris and Hutchinson 2012). The other two mechanistic models are dynamic, complex and realistic, requiring a large number of input parameters. One was developed from several large reservoir systems in Québec and the other was developed for central Canadian boreal reservoirs. Predictions of mercury in fish that are derived from existing equations and/or dynamic models are desirable because they are objective and quantitative.

As part of a preliminary assessment of the proposed Site C Project, the linear regression model developed by Harris and Hutchinson (2010) was employed to determine how creation of the Site C reservoir may influence mercury in fish. This model predicts the relative increase in fish Hg concentration above baseline for a new reservoir, using only three variables – flooded area, total area ($km^2$) and mean annual flow ($km^3/y$). The outcome is a predicted 'peak increase factor'. That is, a number is generated (e.g. 4x) for which the baseline mean fish Hg concentration is multiplied by to predict a *peak* Hg concentration that may be observed for each species. This simple model does not predict timing of the response nor the return period back to a baseline condition, however. The details of how the model was applied and its outcome, is presented as an appendix within the RESMERC report (Volume 2, Appendix J, Part 3). Despite the simple input parameters, this regression model is based on the outcomes of at least a dozen reservoirs and has proven to be fairly accurate at predicting peak fish mercury concentrations.

Despite the simple although robust prediction of the Harris and Hutchinson (2012) model the RESMERC mechanistic model developed by Reed Harris Environmental was adapted for conditions in BC and applied at the proposed Site C Project (Volume 2, Appendix J, Part 3). RESMERC is a dynamic, state of the art model that includes the latest understanding from scientific studies on the dynamics of mercury in aquatic systems. RESMERC mimics the production, destruction and bioaccumulation of methylmercury in reservoirs using mass balance calculations of elemental, mercuric ion and methylmercury over time. The key outputs of this model are predictions of mercury and methylmercury concentrations in water and biota (e.g., invertebrates, insects and fish) at any point in time within the Site C reservoir. Application of RESMERC to the Site C Project reflects use of the most up-to-date predictive modeling to determine the dynamics of changes in methylmercury concentrations in aquatic environmental media during the period when mercury is elevated above baseline.

A wide variety of baseline data were collected as critical input parameters (Azimuth 2011; Golder 2009a, 2009b, 2012) to run RESMERC. These include: area (ha) of inundated uplands and wetlands; water temperature; hydraulic retention time; thermal stratification; oxygen concentration; pH; dissolved organic carbon concentration; phosphorous and sulphate concentrations; atmospheric deposition and evasion of mercury; concentrations of mercury in inundated soils; inorganic and methylmercury concentrations in biota (plankton, benthos, fish) and actual or predicted fish growth rates and fish bioenergetics.

Results from the RESMERC modeling exercise have been combined with other lines of evidence (regression modeling; Canadian reservoirs comparison matrix, Section 5.0) to predict the likely change in fish mercury concentrations within the proposed reservoir and downstream.

## 2.8 Direct Effects of Mercury on Fish

The objective of this section is to assess the potential sub-lethal, direct effects to fish from exposure and bioaccumulation of methylmercury in fish tissue based on a review of the recent scientific literature. Recent literature suggests that there is the potential for mercury to adversely affect endocrine function, steroid synthesis, gonadal development, behaviour and other factors, at environmentally relevant tissue mercury concentrations (Alvarez et al. 2006; Scheuhammer et al. 2007; Murphy et al. 2008; Crump and Trudeau 2009; Sandheinrich and Weiner 2011, Crump and Trudeau 2009; Depew et al. 2012a, and others).

To research the direct effects of mercury on fish, several recent summary papers were relied on, including a book chapter by Sandheinrich and Wiener (2011) and a subsequent paper by a larger group of authors (Depew et al. 2012a). These authors and others have examined the toxicity of dietary methylmercury and in some cases with the objective of deriving a 'threshold' concentration, above which adverse effects could potentially be observed for a variety of endpoints. This recent research updates a very early review paper by Weiner and Spry (1996) that examined mercury toxicity to fish as an historic starting point. Literature-derived relationships between mercury exposure and effects have examined possible effects on fish growth/condition, reproductive effects and behaviour. Ultimately recent authors (Sandheinrich and Weiner 2011; Depew et al. 2012a) have sought to establish a 'tissue residue guideline' or dietary threshold concentration above which there is the potential for mercury induced effects to fish.

There is consensus within the literature that neurotoxicity may be the most likely mode of action for methylmercury in wild adult fish, with high exposures associated with impairment of the central nervous system, causing a wide range of effects. This damage is the most likely cause of several responses including impaired coordination and feeding ability and diminished responsiveness related to long-term dietary exposure. As these types of responses may be difficult to detect under field conditions, Wiener and Spry (1996) relied on laboratory studies primarily based on water exposures of mercuric chloride to develop critical tissue concentrations related to adverse effects. There were numerous drawbacks to most (e.g., exposure via water and not food; mercuric chloride is not methylmercury, unrealistic water-borne concentrations were used to mimic dietary exposure) that limit the development of a realistic threshold concentration. Subsequently, Scheuhammer et al. (2007) and others pointed out that the use of water-based exposure routes laboratory studies is not realistic and introduces some uncertainty. Unfortunately, the lack of paired studies comparing mercury-related effects to fish between water and food exposures precluded further assessment of this uncertainty until fairly recently.

### 2.8.1 Critical Fish Tissue Concentrations and Effects

Mercury dynamics (e.g., sources, bioavailability and bioaccumulation) have been studied extensively over the past four decades mostly in relation to reservoir creation and increased methylmercury concentrations in fish with subsequent exposure to fish-eating birds (e.g., eagle, merganser, loon) and mammals (e.g., otter, mink). Compared to mammals and avian wildlife species, there is relatively little is known about the toxicological significance of mercury in fish (Scheuhammer et al. 2007).

Wiener and Spry (1996) reviewed laboratory and field studies assessing the potential for direct adverse effects of methylmercury exposure. In laboratory studies, the range of muscle tissue concentrations associated with adverse effects was 5 to 8 mg/kg dry weight in walleye and 10 to 20 mg/kg wet weight for salmonids. In the field, tissue concentrations ranging from 6 to 20 mg/kg wet weight were associated with adverse effects. These concentrations are very rarely observed in the wild. Consequently, they proposed 5 mg/kg wet weight concentration as the threshold below which no direct adverse effects to fish should be observed.

Over the last 5 – 10 years some studies have documented adverse effects to fish with muscle tissue concentrations well below the threshold proposed by Wiener and Spry (1996). We have examined the relevant literature to select and review some of these papers with results summarized below by major endpoint type. Mortality was not considered as an endpoint because fish would be rarely exposed to mercury concentrations (except in the most extreme cases) high enough to result in mortality. Consequently, it is not usually monitored as an endpoint in studies looking for potential effects at environmentally-relevant mercury exposure concentrations.

### 2.8.2 Responses of Fish to Methylmercury Exposure

**Growth** – Very few studies have demonstrated significant negative relationships between mercury concentration and impaired or diminished growth and condition factor. For example, Hammerschmidt et al. (2002) reported that when fathead minnow (*Pimephales promelas*) were exposed to dietary mercury, mean weight was weakly, but positively higher at tissue concentrations of 5.6 mg/kg relative to the control at 0.12 mg/kg. Friedmann et al. (1996) demonstrated that 1.0 mg/kg juvenile walleye (*Sander vitreum*) had slower growth than low mercury walleye (0.25 mg/kg). Others including Houck and Cech (2004) found no difference in growth of juvenile Sacramento blackfish (*Orthodon microlepidotus*) up to 0.75 mg/kg with effects observable only at 15 mg/kg, an unrealistically high concentration. It is difficult to correlate mercury exposure and growth in field studies because of the confounding influences of water temperature, nutrients, genetics and other factors. Furthermore, reductions in growth would first be manifest in many other endpoints (e.g., feeding efficiency, vision, reaction time, histological damage, biochemical impairment) that may be easier to relate to mercury than as gross a measure as growth rate or condition factor.

For example, Adams et al. (2010) examined tissue mercury concentrations (kidney, liver, brain, gonad, muscle) in marine sea trout (*Cynoscion nebulosus*) and correlated these with a range of sublethal parameters including liver function, inflammation, tissue damage indicators and others. They found that at tissue concentrations in excess of 0.5 mg/kg, relative to control fish, there was some evidence of histological changes to liver tissue, pyknosis/necrosis and tubular thickening of kidney tissue, inflammation and bile duct hyperplasia, suggesting sublethal effects at the individual level at relatively low concentrations. It is unknown if these changes were manifest in terms of performance or growth rate.

**Reproduction** – The reproduction endpoint has been one of the best documented parameters with several studies demonstrating potential effects at environmentally relevant concentrations. Three studies examining dietary mercury exposure on spawning success in fathead minnows (Hammerschmidt et al. 2002; Drevnick and Sandheinrich 2003; Sandheinrich and Miller 2006) suggested that mercury exposure can adversely affect reproduction in individual fathead minnow starting at whole body mercury concentrations on the order of 0.5 mg/kg wet weight. The mechanism of toxicity appears to be principally at the cellular level associated with disruption in endocrine regulation (e.g., reduced estradiol and testosterone) and disruption and gene expression. These were manifest as reduced spawning success, less time spent spawning, and fewer eggs laid.

Sandheinrich and Miller (2006) examined the effects of dietary methylmercury on testosterone production and behaviour of male fathead minnow. Fish fed methylmercury contaminated diets spent much less time mating than control fish although there was no difference between courtship and time taken to prepare nests. Total mercury in tissue was not correlated with individual reproductive behaviour but was correlated with hypo activity (i.e., were more inactive). Testosterone production was not correlated to tissue mercury concentration; however, testosterone amount was correlated with reproductive related behaviour.

Crump and Trudeau (2009) reviewed mercury effects on reproduction in male and female fish from a large number of studies, however, they only report on 20 studies that fit their criteria for scientific rigor. In summary, they determined that reproductive effects were manifest in a variety of ways including impairment of steroid synthesis, oocyte development and morphology. It was difficult to determine if effects to the reproductive organs were direct or indirect, mediated via the hypothalamus and pituitary glands, affecting hormone production and release. Maternal diet determines the magnitude of maternal transfer of methylmercury to the eggs during oogenesis, rather than maternal body burden *per se* and is a major determining factor in methylmercury content of eggs (Hammerschmidt and Sandheinrich 2005). Maternal exposure may also affect sensitivity of eggs prior to fertilization. Some studies showed higher tolerance of eggs from Hg exposed females than unexposed, suggesting some form of resistance in wild populations. Among males, data were inconclusive. In addition, in all studies there were differences in sensitivity at different life history stages, inter-species differences and a wide variety of magnitude of response at different dietary mercury exposure. There were also several confounding factors including adaptation or desensitization, antagonistic of other contaminants and the protective effects of selenium.

Crump and Trudeau (2009) concluded that although there were a large number of laboratory based studies that indicated the potential harmful effects of mercury on reproduction there was uncertainty about the specific mechanism and site of action. The most likely cause of reproductive impairment was caused by alterations in factors regulating the hypothalamic – pituitary – gonadal relationship with broad and varied implications. Laboratory studies seem to suggest that there is sufficient evidence to link mercury exposure to reproductive effects but the site of action and mechanisms involved are not known. Furthermore, how this translates to ‘natural’ exposure in the wild is unknown and is subject to a wide variety of confounding influences. The authors did not suggest a ‘threshold’ concentration above which effects might be expected.

**Behaviour –** Behaviour is often used as an endpoint in toxicological studies because it integrates expression over its range of physiological response to its environment. Contaminant-induced behavioural changes can result in a range of indirect effects that may affect many metrics including growth and survival due to reduced feeding efficiency and increased vulnerability to predation. Again, effects observed in the laboratory are extremely difficult to objectively quantify and to extrapolate at the individual level in the wild and then to local or regional population-level effects. Among the variety of studies conducted on behavioural changes, response times to stimuli, predator avoidance and foraging ability were most common.

DePew et al. (2012a) stated that the sensitivity endpoint of behaviour was about an order of magnitude higher than more ‘sensitive’ endpoints related to histology, biochemistry or genotoxicity. They speculated that this is likely a reflection of the ‘difference between the presence of subclinical effects of methylmercury exposure and the accumulated subclinical damage required to elicit behavioural changes at the organism level.’

Regarding predator avoidance, Webber and Haines (2003) was one of the first groups to examine this phenomenon. They observed impaired shoal cohesion and settling time (return to sedentary state) of golden shiners at whole body concentrations of 0.23 and 0.54 mg/kg wet weight respectively, after being fed a mercury-contaminated diet.

Alvarez et al. (2006) found that Atlantic croaker (*Micropogonias undulates*) juveniles with higher maternally transferred mercury body burden demonstrated reduced rate of travel and activity level and delayed response to a vibration startle stimulus. The authors speculated that this impairment could negatively influence survival. On the other hand, there was no difference between larval growth, swimming speed or a visual startle stimulus and methylmercury concentration. They concluded that there was a small probability of a decline in developmental larvae avoiding predation but that there was no threshold in mercury concentration where this would occur.

More recently, Murphy et al. (2008) followed up on this study to express changes in locomotory activity and predator evasion skill in croaker in perspective with an ecological context, by attempting to ‘scale-up’ individual level changes at the population level. Although details of this complex study are not summarized here, they concluded that fish exposed to methylmercury resulted in slowed swimming speed and reduced probability of predator avoidance. Although there are many uncertainties by extrapolating results of individual laboratory experiments to the field and then to the population level, they do establish the possibility between a link between behavioural response to a contaminant exposure and long-term population dynamics and viability.

In a summary paper, Depew et al. (2012a) reviewed 20 studies that examined the effects of dietary methylmercury on a variety of studies examining growth, reproductive, histological, enzymatic and behavioural endpoints. Their conclusions were drawn from laboratory studies (which have limitations and are not necessarily related to field or wild fish) involving 12 species ranging from 25 d – 600 d exposures to dietary methylmercury. Their objective was to derive a dietary threshold (*not* a tissue residue concentration) concentration above which methylmercury would be expected to elicit an ‘ecologically significant adverse effect in fish’. Not surprisingly, adverse effects to fish were observed at dietary methylmercury exposures at ecologically relevant levels at the biochemical, histological, genotoxic and behavioural endpoints. However, there was very wide disparity in response among species due to differences in life history stage (adult, juvenile), species specific differences, genetics, tolerance, experimental technique/design, selenium (as an antagonist to mercury) and many others. In general, endpoints related to growth were least sensitive, followed by reproductive endpoints, behavioural endpoints and then by biochemical and histological endpoints which were most sensitive. Dietary methylmercury concentrations in food proposed as ‘threshold’ concentrations by Depew et al. (2012a) that elicit the lowest observed adverse response in laboratory studies ranged from 1.44 mg/kg ww for growth, 0.50 mg/kg ww for behaviour, 0.04 mg/kg ww for reproductive and 0.06 mg/kg ww for biochemical effects. They acknowledge that these thresholds are conservative and were developed under laboratory conditions with artificially contaminated food in the absence of confounding factors that might be found in the field.

### 2.8.3 Summary of Responses

These results indicate a potential for effects to occur at tissue mercury concentrations far below what was thought only 10 – 15 years ago. Based on this extensive literature review a threshold for possible direct adverse effects to fish is a tissue concentration has been promulgated at approximately 0.5 mg/kg ww. However, this is considered a preliminary screening value that has been associated with low level effects only in some laboratory studies. The data set upon which this value has been based is limited in the number of species/endpoints combinations assessed with relevant exposure scenarios, leading to substantial uncertainty in any identified threshold tissue mercury concentrations. Nevertheless, the weight of evidence of such studies suggests that there is a potential effect on fish at tissue mercury concentrations far below what was previously thought (e.g., Weiner and Spry 1996) only 15 years ago. How this relates to individual performance and health and community or population level effects is unknown.

## 2.9 Mercury and Health

Mercury as an environmental contaminant is well known to the public and regularly receives media attention, mainly related to the consumption of fish and shellfish. People are exposed to methylmercury almost exclusively from the consumption of fish and shellfish (Mergler et al. 2007) and the vast majority of the thousands of fish ‘consumption advisories’ in North America are due to mercury (Wiener et al. 2007; Ontario Ministry of Natural Resources 2011). While all fish contain mercury, increased anthropogenic loading and increased acidification of lakes has caused fish methylmercury concentrations to rise in most lakes and rivers, even in remote, pristine areas.

Long-term, frequent consumption of fish with mercury concentrations in excess of tolerable intake levels may pose a risk to human health (Clarkson 2003), although adverse health effects are difficult to definitively diagnose and are believed to have been rarely if at all observed in Canada (Clarkson 1998; Wheatley and Paradis 1996). Wheatley and Paradis (1996) undertook a 20-year retrospective analysis of methylmercury in fish involving over 38,000 individuals in Canada and did not find any identifiable health problems related to mercury. Nevertheless, concern remains, although Health Canada (2007) advice that fish are a nutritious food source and should be regularly consumed, but that health benefits should be weighed against health risks.

In addition to concerns regarding methylmercury toxicity in people, many studies have shown that dietary methylmercury exposure by many wildlife species has caused health effects to some wildlife species in some areas. These include fish-eating birds such as eagles, egrets, and loons (Finley et al. 1979; Barr 1986; Halbrook et al. 1997; Burgess et al. 1998; Meyer et al. 1998; Scheuhammer and Blancher 2004; Spalding et al. 2000; Kenow et al. 2003; Burgess and Meyer 2008). As well, some mammals (e.g., mink, otter) are especially sensitive with reductions in growth, reproduction, immune function and other ailments (O'Connor and Nielsen 1981; Halbrook et al. 1997; Dansereau et al. 1999; Spalding et al. 2000; Chan et al. 2003; Ben-David et al. 2001) being attributable to methylmercury exposure. Recent studies have also suggested that insectivorous birds such as wren and other songbirds (Cristol et al. 2008; Jackson et al. 2011) may be susceptible to the effects of mercury; however, these studies have been carried out in mercury-contaminated areas.

In Canada, mercury contamination of aquatic environments related to chlor-alkali plants such as the English-Wabigoon system in northwestern Ontario (e.g., Norstrom et al. 1976; Barr 1986 and others) and in northern hydroelectric reservoirs (e.g., Bodaly et al. 1984; Schetagne et al. 2003) have been well publicized. The environmental assessment of mercury is now a universal component of impact studies for hydroelectric projects (e.g., Hydro-Québec 2007; 2008).

# 3 TERRESTRIAL BASELINE CONDITIONS

The objective of this section is to document baseline characteristics of the terrestrial area projected to be impounded and inundated by the proposed Site C dam (EIS Volume 1, Section 4.0). Forest soils with a high organic content are the main reservoir of carbon and inorganic mercury that are subject to methylation in new reservoirs. Information presented here includes results of field investigations on mercury and organic carbon content and thickness of organic soils within the area proposed for inundation. Terrestrial Ecosystem Mapping (TEM) habitat polygons (Keystone 2009) were used to prorate a dedicated sampling effort of soils in 2010 (Azimuth 2011). In addition, total inorganic mercury was directly measured from standing vegetation (trees, shrubs, grasses) to characterize baseline concentrations for a variety of vegetation types (shrubs, grass, trees) within the impoundment area of the proposed Site C dam. These concentrations are compared to values from northern boreal forests elsewhere in Canada and Scandinavia.

## 3.1 Background and Approach

Construction of the proposed Site C dam will inundate a portion of the Peace River valley between the dam site to just below the base of the Peace Canyon Dam (Volume 1, Section 2, Project Description). Increased water levels will inundate terrestrial habitat consisting primarily of forested lands along the north and south banks of the Peace River and on islands. Forests communities are dominated by mixed stands of trembling aspen (*Populous tremuloides*) and white spruce (*Picea glauca*), as well as lodgepole pine (*Pinus contorta*), black spruce (*Picea mariana*), balsam poplar (*Populus balsamifera*) and tamarack (*Laris laricina*). White spruce and cottonwood are more common in the river valley on organically rich soils along river shorelines opposite cutbanks and cliffs. Aspen and poplar are more common on drier, higher elevation slopes while black spruce is common in moister soils. The surface area of wetlands and bogs (e.g., Watson Slough) comprise a relatively small amount (<3%) of the terrestrial land that is forecast to be flooded. There is also some wetland / bog habitat in the vicinity of the mouths of the larger tributary streams, Moberly and Halfway. Boreal grassland and shrub communities characterize steep and south facing slopes along the Peace River.

The most common understory and shrub community species included prickly rose (*Rosa acicularis*), mountain alder (*Alnus incana*), red osier dogwood (*Cornus stolonifera*), Pacific willow (*Salix lucida*), peavine (*Lahyrus* sp.) and western sarsaparilla (*Aralia nudicaulis*). A more complete assessment of vegetation cover within the proposed reservoir areas can be found within the Terrestrial Ecosystem Mapping (TEM) project summarized by Keystone

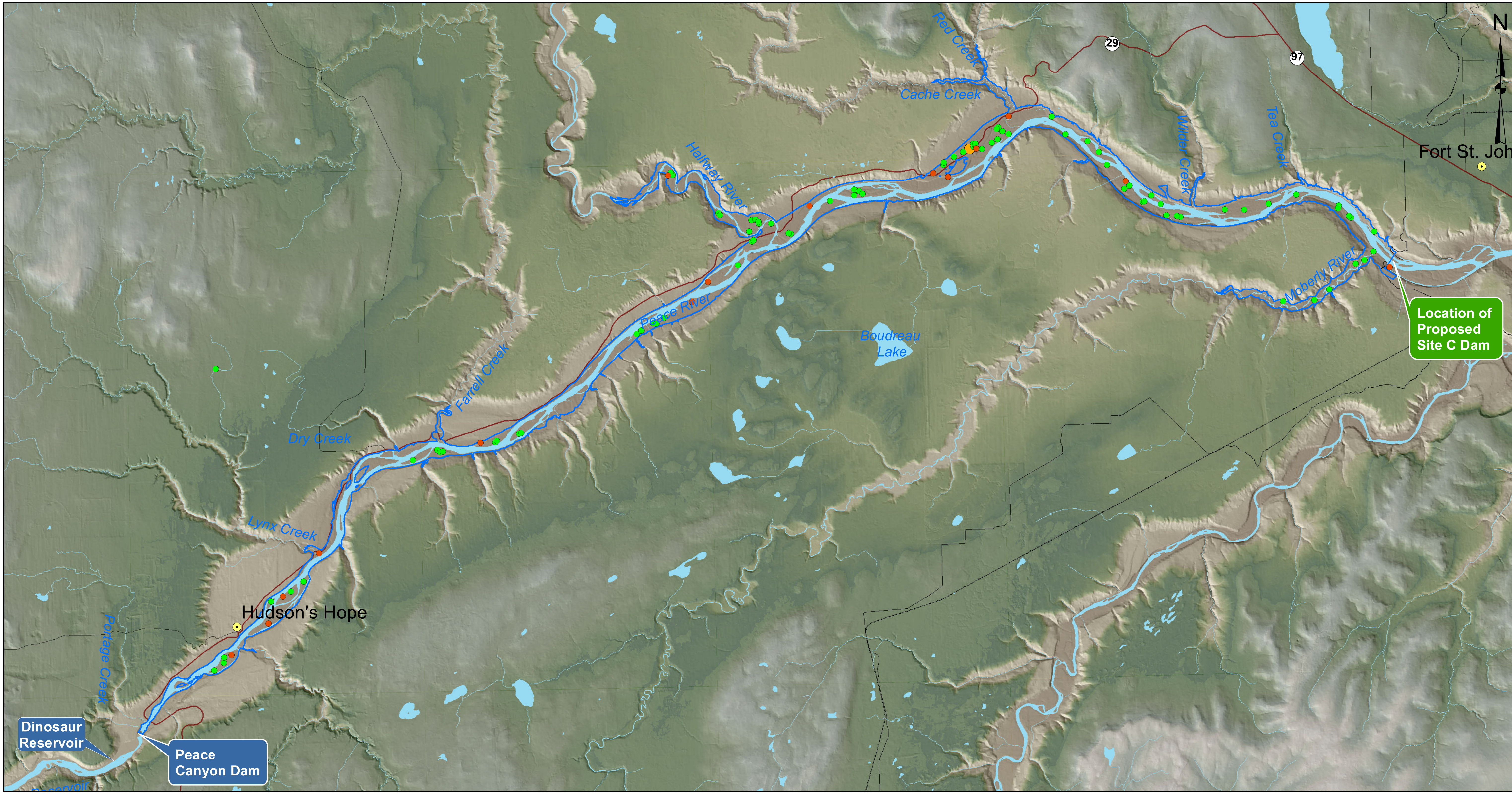
(2009) and as summarized in Volume 2 Chapter 13 Vegetation and Ecological Communities and Appendix R, Terrestrial Vegetation and Wildlife Report for Site C.

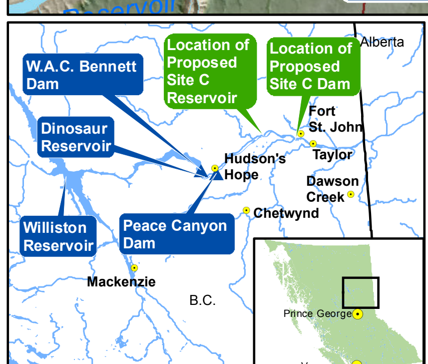
A key objective of the Azimuth (2011) investigation was to characterize the quantity and general quality of the soils (and vegetation) within representative soil/vegetation types throughout the proposed reservoir. This was achieved by measuring organic soil depths, organic carbon content, pH, and total mercury concentration at nearly 100 locations between Hudson Hope and the Site C dam site (**Figure 3.1**). Results of this effort provide the majority of the terrestrial baseline information required to supply key input parameter data to the RESMERC model (Volume 2, Appendix J, Part 3).

The baseline terrestrial assessment describes existing conditions within the terrestrial component within the proposed Site C Project area. Discrete areas not assessed for mercury included roads, bridges, gravel bars, cutbanks and other non-vegetated habitats. Conditions described here are those relevant to the evaluation of the impact of flooding of terrestrial environmental media, predominantly vegetation and soils, on mercury release. In addition to mercury concentrations (mg/kg) in these media it is important to define the inventories of mercury (kg Hg/ha) and carbon (metric tonnes C/ha) in these environmental media. For living vegetation it is especially important to estimate the fraction of carbon that is likely to be rapidly decomposed (e.g., foliage and small stems) following inundation. For soils the most important component is the uppermost organic fraction represented by the litter, fermentation and humus horizons, within several centimeters of the surface. Labile (i.e., easily decomposable, bioavailable) carbon and mercury in these horizons also supports methylation of mercury.

## 3.2 Mercury in Terrestrial Vegetation

The primary exposure pathway of terrestrial plants to most environmental contaminants is via adsorption to root structures, followed by uptake and translocation from roots to shoots and leaves. While this pathway is important for many metals (Chaney 1990), uptake of mercury to plants via the root pathway is generally considered to be low. Instead, mercury adsorption to plants occurs predominantly on plant shoots and leaves (above ground parts) from atmospheric sources (Grigal 2003). Over time, mercury gradually accumulates in organic soils from centuries of litterfall, from both atmospheric and mineral sources.




## Legend

- 2011 Golder Soil Sampling Locations
- 2010 Azimuth Soil Sampling Locations
- Watson's Slough
- Proposed Reservoir
- Road
- Highway
- Railway

Map Notes:
1. Datum: NAD83
2. Projection: UTM Zone 10N
3. Base Data: Province of B.C.
4. Location of Watson's Slough is approximate.
5. Proposed reservoir area (461.8m maximum normal elevation) from Digital Elevation Models (DEM) generated from LiDAR data acquired July/August, 2006.

1:200,000

0 5 km

SITE C CLEAN ENERGY PROJECT

BC hydro

**Figure 3.1:**
**Soil Sampling Locations**

| Date | Oct. 17, 2012 | DWG NO | 1016-C14-B6169 - 1 | R 0 |
|---|---|---|---|---|



Construction of the Site C Clean Energy Project is subject to required regulatory approvals including environmental certification.

Z:\11 GIS Support\Mxd\07 Other\007_GIS_Support\Mercury\Soil_Sampling_Locations_1016_C14_B5753.mxd

### 3.2.1 Sampling Methodology

Two dedicated sampling programs to determine mercury levels in vegetation have recently been undertaken, by Golder (2009) and Azimuth (2011). Golder (2009) collected and composited vegetation (foliage from spruce, birch, rose) from six locations within the inundation zone of the proposed reservoir area in 2008. Vegetation types were composited in unrecorded proportions and thus species-specific results are not available. Azimuth (2011) sampled dominant vegetation species for mercury analysis throughout the proposed area subject to inundation during operation of the proposed Project in 2010. Species included trees (spruce, balsam, willow, alder), shrubs (sarsaparilla, prickly rose, willow, alder and dogwood) and grasses (horsetail, sedge, reeds, cattail). Particular attention was paid to Watson Slough, a wetland and area with naturally higher soil mercury concentrations.

Although methylmercury concentrations were not measured in the 2010 study, the literature reliably indicates that the percentage of methyl relative to total is consistently less than 2% of total (e.g., Grigal 2003).

### 3.2.2 Mercury in Vegetation Summary

According to the literature (e.g., Moore et al. 1995; Grigal 2003), shrubs and trees typically have lower mercury concentrations than other vegetation types such as aquatic macrophytes, mosses, lichen and fungi. In the leaves and needles of terrestrial plants (spruce, balsam, maple, and tamarack) collected in southern Ontario total mercury concentrations ranged from 0.004 to 0.047 parts per million (mg/kg) dry weight (dw) (Rasmussen et al. 1991). Although the concentration of mercury in living or standing vegetation is not expected to be very high (typically <0.02 mg/kg dw), some characterization of the dominant vegetation types is necessary to confirm mercury concentrations and to estimate the inventory of carbon and mercury in this easily decomposed biomass. It is noteworthy that the vast majority of the mercury pool exists in the soils and not in vegetation.

Mercury concentration in composited vegetation in 2008 by Golder (2009) were low for all vegetation samples, ranging from <0.005 to 0.019 mg/kg (dw). These data are comparable to data from northwestern Ontario vegetation (Grigal 2003).

Total mercury concentration in all plant tissues collected by Azimuth (2011) was also low, in most cases barely above the detection limit of 0.005 mg/kg dw. The most common shrubs (sarsaparilla, prickly rose and alder) were low (<0.008 mg/kg dw) and of very similar concentration. Trees were also low in mercury (<0.005 to 0.019 mg/kg). The sedge species, reed and cattail from Watson Slough had only slightly higher mercury concentrations (0.013 to 0.015 mg/kg dw).

Based on these studies, plant tissue mercury data from the Peace River region were on the low end of the scale for equally remote, pristine areas of boreal forest in Canada (Bodaly et al., 1987; Moore et al. 1995; Rasmussen et al. 1991, Rasmussen, 1995; Zhang et al. 1995), Europe (Grigal 2003) and Scandinavia (Jensen and Jensen 1991; Steinnes and Anderson 1991). The standing stock, or biomass, of vegetation proposed to be inundated was calculated based on soil depth and mercury and organic carbon concentration in representative habitat polygons based on Terrain Ecosystem Mapping (TEM) throughout the reservoir area. This information was collected because carbon and mercury inventories are important input parameters used in the RESMERC model (Volume 2, Appendix J, Part 3). There are also published estimates of average non-stem carbon biomass for forest lands in British Columbia that were used to verify our estimate. For example, the BIOCAP Canada Foundation (2003) published a value of 69.5 metric tonnes (mT) of carbon per hectare (C/ha) in forest soils. This is an average value for all productive forest lands in BC and thus only provides an estimate of the living carbon density within the forests at Site C. Non-stem biomass consists of branches, bark and leaves but not shrubs or herbaceous plants. Campbell et al. (2000) published a detailed review of carbon biomass for shrub and herb layers in fens, 1.2 $\pm$ 1.6 mT C/ha, i.e., much lower than that for non-stem biomass as defined by BIOCAP. A preliminary estimate of the pool size of mercury in living vegetation within the Site C project was calculated by multiplying the living biomass (~70 mT C/ha) by the average measured mercury concentration in vegetation (0.01 mg/kg dw, 10 mg/mT), or 70 µg/m$^2$. The latter value compares with those reviewed by Grigal (2003) that ranged from 20 to 680 µg/m$^2$ for forests in North America and Europe. The highest value (680 µg/m$^2$) was for a Norway spruce forest in southern Sweden (Munthe et al. 1998) while the lowest (20 µg/m$^2$) was for an aspen forest in Minnesota.

Overall, mercury concentrations (mg/kg) and density (µg/m$^2$) in vegetation within Site C are comparatively low, in keeping with its remoteness from anthropogenic atmospheric sources of mercury. The current estimate of the vegetation mercury density on an aerial basis is estimated at 70 µg/m$^2$.

## 3.3 Mercury in Terrestrial Soils

Andersson (1979) reviewed the extensive scientific literature available through 1978 on mercury in soils. Many additional publications (see reviews by Adriano 1986; Schuster 1991; Lodenius 1994) have appeared since this review, but our understanding of mercury in soils has not changed much. Mercury concentrations from background non-mineralized areas range from 0.01 to 0.2 mg/kg (e.g., Rasmussen 1994; Lodenius 1994; McKeague and Kloosterman 1974), whereas values for soils from mercury-mineralized areas, such as near the Pinchi fault in BC (Plouffe 1995), range up to several mg/kg. Where soils have developed on uniform parent material, vegetation, cover type and cover age are reported to be very important variables affecting concentration of mercury in soils (Grigal et al. 1994).

The total Hg content of soil is most frequently correlated with organic matter content and less frequently with clay and iron content. Atmospherically-deposited mercury is effectively fixed in the uppermost layer (humus) of forest soils developed on glacial till and granitic bedrock in Sweden (Lindqvist et al. 1991; Aastrup et al. 1991). This fixation in humus is often manifested as sharp decreases (10x within a few centimeters) in Hg concentration as a function of depth in the soil profile. Soils that have developed under deciduous forest canopies or over carbonate bedrock (e.g., limestone, dolomite) generally do not exhibit such distinct vertical gradients in mercury concentrations because these soils experience rapid degradation of particulate organic matter and downward transport of Hg carried by dissolved organic carbon and by activity by insects and worms. This apparent dichotomy in the behaviour of mercury in which organic matter can serve as both an immobilizing and a mobilizing agent is important to recognize. Thus, for example, where many other metals will tend to be mobilized under acidic soil conditions, such as may exist under a coniferous forest canopy, mercury will tend to be immobilized because degradation of organic matter is inhibited and condensation of humic acids is favored under acidic soil conditions.

Azimuth (2011) collected more than 100 samples in 2010 within the area to be inundated by the reservoir (**Figure 3.1**). These were collected in relative proportion to the areal extent of all vegetated habitats. Sampling was also stratified by soil horizon (i.e., vertically) and according to soil and habitat type (**Table 3.1**) based on the TEM (TEM; Keystone 2009). The sampling strategy focused on areas with well-developed organic soil horizons, as well as in discrete areas known for contributing a disproportionately high amount of carbon and mercury such as wetlands (e.g., Watson Slough) and some isolated backwater areas that accounted for only a small percentage within the reservoir area. Note that some seasonally flooded soils consisting of sand and gravels with little to no organic material will contribute very little if any mercury to the Site C reservoir.

**Table 3.1 Site C Habitats of potential importance to mercury cycling**

| Class | Area (ha) | % of Total | # of polygons | Cover descriptions |
|---|---|---|---|---|
| AM | 208 | 5.8 | 16 | Step moss-Peavine |
| BL | 10.6 | 0.3 | 1 | Labrador tea-Lingonberry |
| BT | 19.7 | 0.5 | 2 | Labrador tea-*Sphagnum* |
| CF | 538 | 15.0 | 50 | Cultivated field |
| Fm02 | 1096 | 30.5 | 166 | Cottonwood-Spruce-Red osier Dogwood |
| SE | 56.1 | 1.6 | 3 | Sedge Wetland |
| SH | 1068 | 29.7 | 109 | Currant-Horsetail |
| SW | 230 | 6.4 | 85 | Wildrye-Peavine |
| WH | 365 | 10.2 | 78 | Willow-Horsetail-Sedge Riparian Wetland |
| **Totals** | **3591** | **100** | **510** | |

**Figure 3.2** illustrates the distribution of soil mercury concentrations among major habitat types. The average total Hg content of all samples (N=100) of organic soil horizons, including those with some mineral soil within the upper 5 cm of the soil profile was 0.079 ± 0.031 mg/kg dw. The range of Hg concentration from all soils was 0.023 to 0.173 mg/kg dw, with more organic soils (>30%; SH, AM, SW) having slightly higher Hg concentrations.

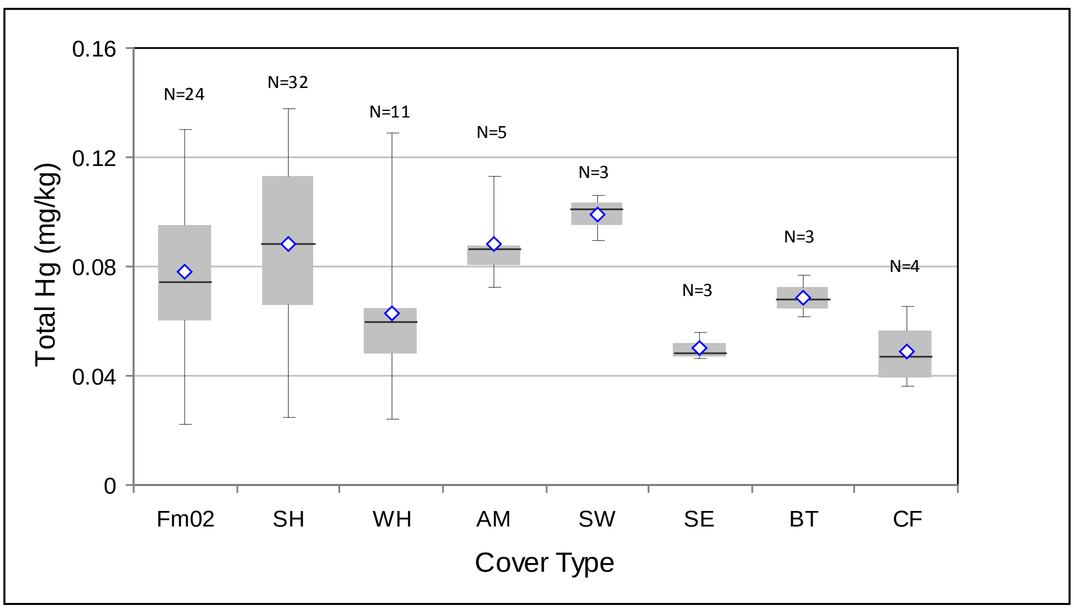



**Figure 3.2. Total Hg in organic soils stratified by TEM habitat**

Notes: N = number of samples, diamond = average, line = median, box = 25 and 75 quartiles, whisker = range.

With respect to methylmercury concentrations, these varied over two orders of magnitude (0.00007 to 0.0071 mg/kg dw) with one of the two highest values (0.0071 mg/kg) measured in Watson Slough. Methylmercury concentration in both of these samples comprised 6% to 7% as a proportion of total mercury, whereas all other samples were <1% methylmercury relative to total, which is typical for non-wetland soils (Grigal 2003).

Total soil mercury concentrations within the proposed Site C Reservoir area are similar to results (<0.05 to 0.13 mg/kg dw) reported for background soils near Ft. St. John, BC (SoilCon 1996; BCMOE 2005). For the two cover types (Fm02, SH) accounting for more than half of the Site C project area, mean total mercury concentrations were similar, although the range in soil mercury values varied within and between cover types (**Figure 3.2**). Given this variation, there was no statistically significant difference in mean mercury content between these cover types (t-test, $p<0.05$). Mercury concentrations in Site C soils fall within the range that is typical for remote, organic soils away from natural mineralization (**Table 3.2**).

**Table 3.2 Organic soil thickness, mercury concentrations and mercury pool sizes**

| Area | Forest Floor Thickness (cm) | Average Total Hg (mg/kg) (N) | Range | Pool Size ($mg/m^2$) | Source |
|---|---|---|---|---|---|
| Site C | 7 | 0.079 (100) | 0.023 - 0.17 | 0.02 - 0.045 | Azimuth 2011 |
| BC | | | | | SoilCon 1996 |
| | - | 0.08 | <0.05 - 0.13 | - | BCMOE 2005 |
| Sweden | - | 0.24 | 0.07-1.00 | 0.17 - 13.5 | Lindqvist et al. 1991 |
| | 12 | 0.25 | - | 3.6 | Aastrup et al. 1991 |
| Norway | - | 0.17 (165) | - | - | Steinnes and Andersson 1991 |
| Saskatchewan | | | | | |
| Young Stand | 8 ± 1.47 | 0.096 ± 0.09 | - | 1.01 + 0.28 | |
| Old Stand | 10.0 ± 1.86 | 0.201 ± 0.12 | - | 2.92 + 0.87 | Friedli et al. 2006 |

## 3.4 Carbon and Mercury Loading Estimates

The spatial distribution and thickness of the organic soils within the reservoir area will influence methylation potential of the flooded soils. Data for thickness of the organic soil horizon (**Figure 3.3**) and bulk density can be converted to estimates of the pool size (kg $C/m^2$) of organic carbon stored in the forest floor (the organic litter, fermentation, humic horizon). For example, bulk densities of 40 to 90 $kg/m^3$ yields a carbon pool size of 2 to 4.5 kg $C/m^2$, assuming that the uppermost 5 cm of the organic horizon is available for methylation (Chojnacky et al. 2009). Most of this carbon is not quickly available to microorganisms. For example, McLauchlan and Hobbie (2004) showed that less than 15% of soil carbon was decomposed in 12 days, while Pokharel and Obrist (2011) observed carbon losses from tree litter was at most only 31% after 18 months. Multiplication of the Site C carbon pool sizes by the total mercury concentration (e.g., 0.1 mg/kg dw) yields estimates of the mercury pool size for the upper 5 cm of organic soil ranging from 0.02 to 0.045 mg/$m^2$. Other published estimates of pool sizes suggest higher values of about 1 to 3 mg/$m^2$ for total mercury in forest floor samples from a comparable habitat (e.g., Norway spruce forest in Germany [Grigal 2003] and spruce-pine forest in Sweden [Aastrup et al. 1991]). The much higher European and Scandinavian pool sizes are likely due to higher atmospheric mercury concentrations and deposition in Europe, as well as the use of thicker organic soil horizons to calculate pool size. The Canadian reservoirs comparison matrix in **Section 5** provides physical and biogeochemical comparisons to other BC and Canadian reservoirs and also gives carbon and Hg pool sizes for planned or completed reservoirs.

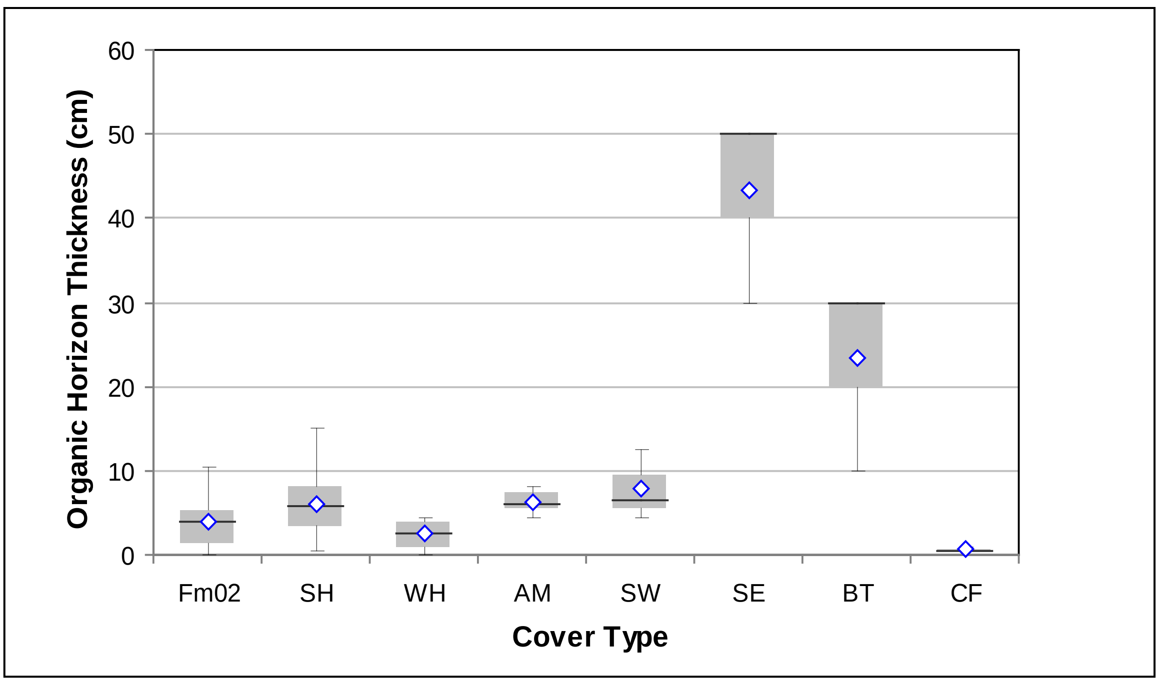



**Figure 3.3. Thickness of organic horizons as a function of cover type**

**Notes:** Diamond = average, line = median, box = 25 and 75 quartiles, whisker = range

## 3.5 Summary

The terrestrial baseline characterization provides a good dataset to support mercury modeling (RESMERC; Volume 2, Appendix J, Part 3) of potential mercury methylation following inundation of terrestrial habitats within the Site C reservoir. The information summarized here for the Site C Project indicates that vegetation and soils have mercury concentrations equivalent to, or lower than, vegetation and soils from other areas throughout North America and Europe without nearby or regional anthropogenic sources of mercury. Accordingly, the estimated vegetation and organic soil inventory of mercury ($\mu g/m^2$) that could potentially be released and methylated is also equivalent to or lower than soils from comparable areas.

# 4 AQUATIC BASELINE CONDITIONS

The objective of this section is to document baseline characteristics of the aquatic environment related to mercury in the Peace River within the proposed Site C technical study area of the Peace River, as well as upstream in Williston and Dinosaur reservoirs. Information presented in this section includes historic and recent results of low level mercury concentrations in water and mercury and methylmercury concentrations in sediment, zooplankton, benthic invertebrates and fish tissue. A wide variety of information has been collected since the 1980s within the Peace River including Williston Reservoir, but mostly related to fish and to evolving limnological conditions within Williston (Stockner et al. 2005).

Williston Reservoir has a considerable influence on downstream water temperature, oxygen, discharge regime, nutrient supply, suspended solids inputs and biota such as plankton and fish in the Peace River. Understanding the implications of the current baseline conditions is important in understanding conditions forecast for the Site C reservoir and especially how these conditions might affect methylation of mercury in this new reservoir and ultimately, accumulation of methylmercury by aquatic biota. As a result, studies focused on mercury and methylmercury in all environmental media (water, sediment, biota) and key water quality parameters that influence mercury concentrations including pH, oxygen, organic carbon, sulphate and total suspended solids concentrations. This information is essential to defining baseline conditions within the Peace River prior to inundation of the proposed Site C reservoir and as critical input parameters to RESMEC to predict mercury concentrations in biota, especially fish (Volume 2, Appendix J, Part 3).

More information on general water chemistry and water quality of the Peace River system can be found in Volume 2 Water Quality (Section 11.4) and Volume 2, Appendix E, Water Quality Technical Data Report.

## 4.1 Technical Study Area

For the purposes of investigations into the relationship between the Site C Project and mercury in aquatic biota, the technical study area boundary includes the Site C reservoir area, as well as the Peace River downstream from the proposed dam location to Many Islands, Alberta. The Site C reservoir extends between the tailrace area below Peace Canyon Dam, 83 km downstream to the Site C dam site.

As explained in Section 2.5 above, there is the potential for mercury to be transported downstream of the reservoir, adhered to sediment particles and organic material, as well as directly in the tissue of plankton and fish that are discharged or entrained out of the reservoir. Based on numerous fisheries investigations by Mainstream Aquatics (2010a; 2011; 2012) and as outlined in Volume 2, Appendix O (Fish and Fish Habitat Technical Report), 30 of 32 fish species migrate downstream of the Site C dam location to at least as far as Many Islands Alberta, about 100 km downstream. Fish within this reach of river downstream of Site C can potentially be exposed to plankton, insects and fish with elevated MeHg discharged or entrained from the reservoir. Because of the availability of injured or stunned fish in the tailrace region of dams, downstream fish may preferentially feed or switch diet to feed on this source of easy prey. When they move back downstream they may carry this higher load of Hg potentially as far as Many Islands. See Volume 2 Appendix (Fish and Fish Habitat) and Appendix Q (Fish Habitat and Migrations) as well as the Sections 12.2.2.2 (Fish Ecology) and 12.1.6.1 (Spatial Boundaries) for further detail.

Consequently, the spatial boundaries of this investigation are defined as being within the Ste C reservoir and its major tributaries and in the Peace River, downstream to Many Islands, which is the downstream extent to which fish routinely move. Although it is possible that some fish of select species (e.g., walleye, goldeye) have been known to extend farther downstream than this area (Mainstream Aquatics 2010b), their temporal interaction or overlap within the tailrace area of the Site C dam is expected to be of limited duration and insufficient time to accumulate as much mercury as reservoir dwelling fish.

## 4.2 Background and Approach

Understanding the potential for changes in mercury concentrations in environmental media is driven by baseline conditions within the Peace River within the proposed reservoir, but also upstream, in Williston and Dinosaur Reservoirs, where water, nutrients and some biota are delivered downstream. Baseline conditions within the Peace River were derived from a variety of sources especially recent efforts by Golder (2009, 2010), Azimuth (2011) with a particular focus on mercury and summaries presented in the Surface Water Regime (Volume 2, Appendix D), Reservoir Water Temperature (Appendix H), Aquatic Productivity Part 1 Biological Assemblages and Part 2 Water Quality modeling and Fish and Fish Habitat Technical Report (Appendix O). Baseline data are described according to the following topics:

- Physical (depth, slope, water residence time, particle settling velocity, etc.) limnological and chemical conditions of Williston Reservoir and Dinosaur Reservoir which is a major driver for loading of nutrients, suspended particles, metals and mercury to the downstream environment based on historic information. Further detail is provided within the documents listed above and summarized in Volume 2 Sections 11.4 Water Quality, 11.7 Thermal Regime and 11.8 Fluvial Geomorphology and Sediment Transport.
- Seasonal water temperature, oxygen regime and chemistry (pH, DOC, hardness, anions, etc.) and mercury concentrations within the Peace River and major tributaries (Halfway, Moberly, Pine), based on historic information.
- Sediment chemistry (grain size, TOC, pH, metals, mercury, methylmercury) of Peace River and major tributary streams - the Halfway and Moberly rivers - within the proposed reservoir area.
- Lower trophic level structure (species composition, abundance and diversity) of phytoplankton, benthic community and zooplankton within the Peace River (Volume 2, Appendix P Biological Assemblages, Part 1).
- Species composition, growth, and diet of fish within major groups (planktivore, benthivore, insectivore, piscivore).

## 4.3 Baseline Hydrology and Limnology

As noted above, baseline mercury concentrations in environmental media within Dinosaur Reservoir and Peace River downstream to the proposed Site C dam are driven by hydrological, limnological and ecological conditions within Williston Reservoir. Williston Reservoir acts as a sink for suspended solids and nutrients and limits downstream drift of invertebrates and fish to the Peace River. As such it has a considerable influence on physical / chemical and ecological parameters within the Peace River. This section describes the baseline hydrological and limnological features of the system as they relate to current, baseline conditions and implications for mercury methylation potential within the proposed Site C reservoir.

### 4.3.1 Hydrology

The Peace River system is regulated, with daily and seasonal control of water released from Williston Reservoir, created by the W.A.C. Bennett Dam, located at the head of Peace Canyon near Hudson's Hope, BC. The reservoir is T-shaped, with the Finlay Reach to the north, Parsnip Reach to the south and the Peace Reach to the east, extending to the Bennett Dam. Water residence or turnover time in the reservoir is approximately 19 months. Just upstream of the dam the maximum water depth is 166 m with a maximum annual drawdown of up to 30 m; however the typical drawdown range is 18 m.

Discharge from the W.A.C. Bennett Dam via the G.M. Shrum Generating Station flows into Dinosaur Reservoir. The intakes drawing water from the Williston Reservoir are 41 m and 72 m below the maximum normal operating level of the reservoir. Thus, for much of the year, water is withdrawn from at least 30 m below surface within the hypolimnion of the reservoir, below the warmer surface waters during mid- to late summer. The depth of the intake is sufficiently deep that cold, nutrient-poor water is withdrawn from the Peace Reach forebay area of Williston Reservoir and discharged downstream. This water has a low abundance of phytoplankton and zooplankton, as these groups are more abundant in warmer surface waters where there is sufficient light penetration to support phytoplankton growth.

Dinosaur Reservoir is a run-of-the-river reservoir with a residence time of two to three days. Water from Dinosaur Reservoir is released to the Peace River through the Peace Canyon Dam, 23 km downstream of Bennett Dam. Given the very short residence time of water within Dinosaur, the physical, chemical and biological features of water in Dinosaur Reservoir and Peace River downstream of Peace Canyon Dam are similar to deep water conditions in the Peace Reach of Williston Reservoir. Having such a large upstream reservoir as Williston has a large influence on the physical, chemical and biological nature on water in the Peace River at least as far downstream as the Site C dam. Further information on the hydrologic features of the system can be found in Volume 2, Chapter 11.4 Surface Water.

Implications of this hydraulic regime on mercury methylation are considerable. The large volume of very low mercury water (see Section 4.4) and the high turnover or replacement rate within the proposed Site C reservoir (approximately 22 – 23 d based on average long-term discharge data from Williston Reservoir) will have a large ameliorating effect on the potential for methylmercury generation. Mercury concentrations in fish from reservoirs with long-retention times will increase to a greater degree and persist for a longer period than mercury in fish from run-of-the-river or short-retention (< months) reservoirs (Schetagne et al. 2003; Bodaly et al 2007) and Section 5.0 below.

### 4.3.2 Trophic Status

Limnological studies of Williston Reservoir have classified the system as ultra-oligotrophic, characterized by having slightly basic or alkaline pH (7.5), low concentrations of carbon, nitrogen and phosphorus nutrients, low total (~1 mg/L) and dissolved solids concentrations and a depauperate plankton community (BC Research 1976; Stockner et al. 2005). Oligotrophication of reservoirs is caused by increasing nutrient deprivation and lower productivity that is mainly driven by the higher drawdown magnitude in BC reservoirs (>10 m) than other Canadian reservoirs. This results in increased turbidity and less productive epilimnetic and littoral zone habitat. Over the last 20 to 30 years, Williston Reservoir has become increasingly oligotrophic due to increased turbidity, reduced light penetration, cold water and ultimately, lost biogenic productive capacity at all trophic levels (Stockner et al. 2005).

Given the low productivity, decomposition of biogenic material in Williston Reservoir is insufficient to lower oxygen concentrations, even in the hypolimnion (the deep, cold layer of water beneath the shallow, warm epilimnion). Mercury methylation is favored in low oxygen conditions, which are not present in Williston Reservoir. Furthermore, discharge of water downstream to Dinosaur and the Peace River from Peace Canyon Dam increases oxygen concentrations to near maximum (Golder 2009a). Given the relative lack of nutrients and planktonic food resources, primary productivity of the Peace River downstream of Peace Canyon is dominated by periphyton growth (Volume 2, Appendix P, Part 1, Biological Assemblages), with diminishing importance going downstream as nutrients and insects are introduced to the mainstem by tributary streams.

Although a large number of studies were conducted within the reservoir and its watershed as part of the Peace Williston Fish and Wildlife Compensation Program (PWFWCP), a few focused on water quality/limnology (e.g., Stockner and Langston 2000), zooplankton biomass (e.g., Wilson and Langston 2000) and assessments of fisheries resources and their evolution within the reservoir based on hydroacoustic and net surveys (e.g., Pillipow and Langston 2002).

Methylmercury generation is generally not favored in riverine environments that are nutrient poor, have low productivity and ecological diversity and a simple trophic structure (Schetagne and Verdon 1999b; Schetagne et al. 2003). These conditions are expected to persist to some degree within the Site C reservoir because of the strong hydrological influence of water discharged from Williston Reservoir that will strongly influence conditions within the evolving reservoir downstream.

### 4.3.3 Water Temperature

Thermal conditions within Dinosaur Reservoir and the Peace River are very well documented seasonally, as well as spatially. The following discussion is based on detailed records of water temperature acquired from the upper and lower penstocks of Williston Reservoir from 2000 to 2011 (Volume 2, Appendix H Reservoir Water Temperature and Ice Regime Technical Data Report).

The thermal regime of Dinosaur Reservoir and Peace River downstream of the Peace Canyon Dam is primarily a function of the temperature of water withdrawn from Williston Reservoir, apportioned between the upper and lower penstocks. Water withdrawn from the upper penstock, situated near the bottom of the epilimnion is warmer during the summer stratification period (July-September) in Williston, relative to colder water withdrawn from the hypolimnion from the lower penstock. This water mixes in the tailrace to a uniform water temperature, which persists with little change through Dinosaur Reservoir, which has a short residence time of three days and is unstratified. Based on data gathered since 2000, most water (about 70%) is withdrawn from the upper penstock, yielding slightly warmer temperatures in summer, than the average temperature of the hypolimnion of Peace Reach. Given the geographic location of the Peace River system, the area experiences long, cold winters, with ice cover on Williston Reservoir beginning in November and extending until the first week of May (Stockner et al. 2005). Thus, water temperature is also cold, averaging about 2°C in Dinosaur Reservoir and downstream between December and May. Water temperature increases to a maximum of about 14°C (average of 12°C during July to September) in August, before declining to 10°C in October and 2°C by December. These uniformly cool conditions in summer and cold water during the remainder of the year are weakly related to lower rates of mercury methylation in sediments and the water column (Rudd, 1995).

### 4.3.4 Oxygen

Williston Reservoir is thermally stratified during summer, with the shallow epilimnion being several degrees warmer than the deep hypolimnion that lies beneath. However, thermal turnover occurs during spring and fall and thus the hypolimnion is well oxygenated year-round, despite the huge reservoir of organic material that still exists within the flooded terrain. Water is further oxygenated as it is discharged into Dinosaur Reservoir and again into the Peace River from the Peace Canyon Dam. Thus, oxygen content of the Peace River is high and fully saturated. Mercury methylation is generally favored under low oxygen conditions (Ullrich et al. 2001) especially in the hypolimnia of some lakes. These conditions are not currently found in the Peace River and are not predicted for the Site C reservoir.

## 4.4 Water Chemistry

### 4.4.1 Background

There are a number of water quality / chemistry parameters that have been documented to influence (positively or negatively) mercury methylation potential, the most important of these are pH (negative), carbon nutrients (positive), total suspended solids (positive; as transport media for mercury) and sulphate (positive). Field studies conducted by Golder (2009a, 2009b) in the Peace River downstream of Peace Canyon Dam in 2007 and 2008 and in 2010/2011 (Azimuth 2011) targeted these parameters to gain a good understanding of seasonal baseline conditions in the river and their likely influence on methylation potential, Total alkalinity and hardness were also measured from the Peace River between 2006 and 2011. Baseline conditions of important chemical parameters that influence mercury methylation are briefly discussed below and in more detail in Section 5.0, the Canadian Reservoirs Comparison Matrix.

- Water pH is one of the most important determining factors in the mercury methylation process, although the mechanism by which this works is still not entirely clear. Slightly acidic water (i.e., lower pH; <6.5) appears to result in elevated fish mercury concentrations compared to fish from circumneutral lakes and reservoirs (Grieb et al. 1990; Wiener et al. 1990; Greenfield et al. 2001). Water pH in excess of 7.0 is associated with lower magnitude increase in methylmercury concentrations in environmental media, including fish.
- Elevated dissolved organic carbon (DOC) concentration is known to stimulate or facilitate the production of methylmercury. Krabbenhoft et al. (2003) found that additions of DOC alone stimulated the production of additional methylmercury from

"old" mercury (i.e., existing mercury in the environment and not newly introduced mercury from atmospheric or other point sources). Additions of DOC and mercury in mesocosm experiments also caused increased methylation than when mercury alone was added. These results suggest that DOC is directly involved in the methylation process, rather than the common assumption that DOC is simply an attractive ligand for mercury in aqueous solution. However, some researchers have shown that increasing DOC concentrations can lower methylation, possibly by complexing with inorganic mercury and sequestering it, making it less available (Miskimmin et al. 1992). Thus, baseline DOC concentrations and forecast elevations in DOC after flooding are positively correlated with the magnitude of elevation in mercury concentrations in aquatic biota.

- Inputs of total suspended solids (TSS) also play an important role in transporting inorganic, particulate bound mercury to lakes and reservoirs where it can be deposited and some of the inorganic mercury can be transformed into methylmercury by bacteria. During freshet this is particularly important as large amounts of TSS can transport nutrients, as well as mercury in concentrations well above guidelines for the protection of aquatic life. However, because the mercury is in the particulate phase and not dissolved, it is not absorbed by biota.
- Sulphate may be an important nutrient for sulphate-reducing bacteria that are widely acknowledged as being primarily responsible for mediating the conversion of inorganic to methylmercury. There is a positive correlation between environmentally relevant concentrations (5 to 30 mg/L) and methylation (Gilmour et al. 1992).
- Finally, concentrations of inorganic and methylmercury in Peace River within Williston and Dinosaur reservoirs are also important factors determining baseline methylation potential and empirical mercury concentrations in water, lower trophic level biota and fish. Low baseline mercury concentrations in water, sediment and ecological compartments will result in smaller absolute changes in methylmercury concentration in ecological media within new reservoirs.

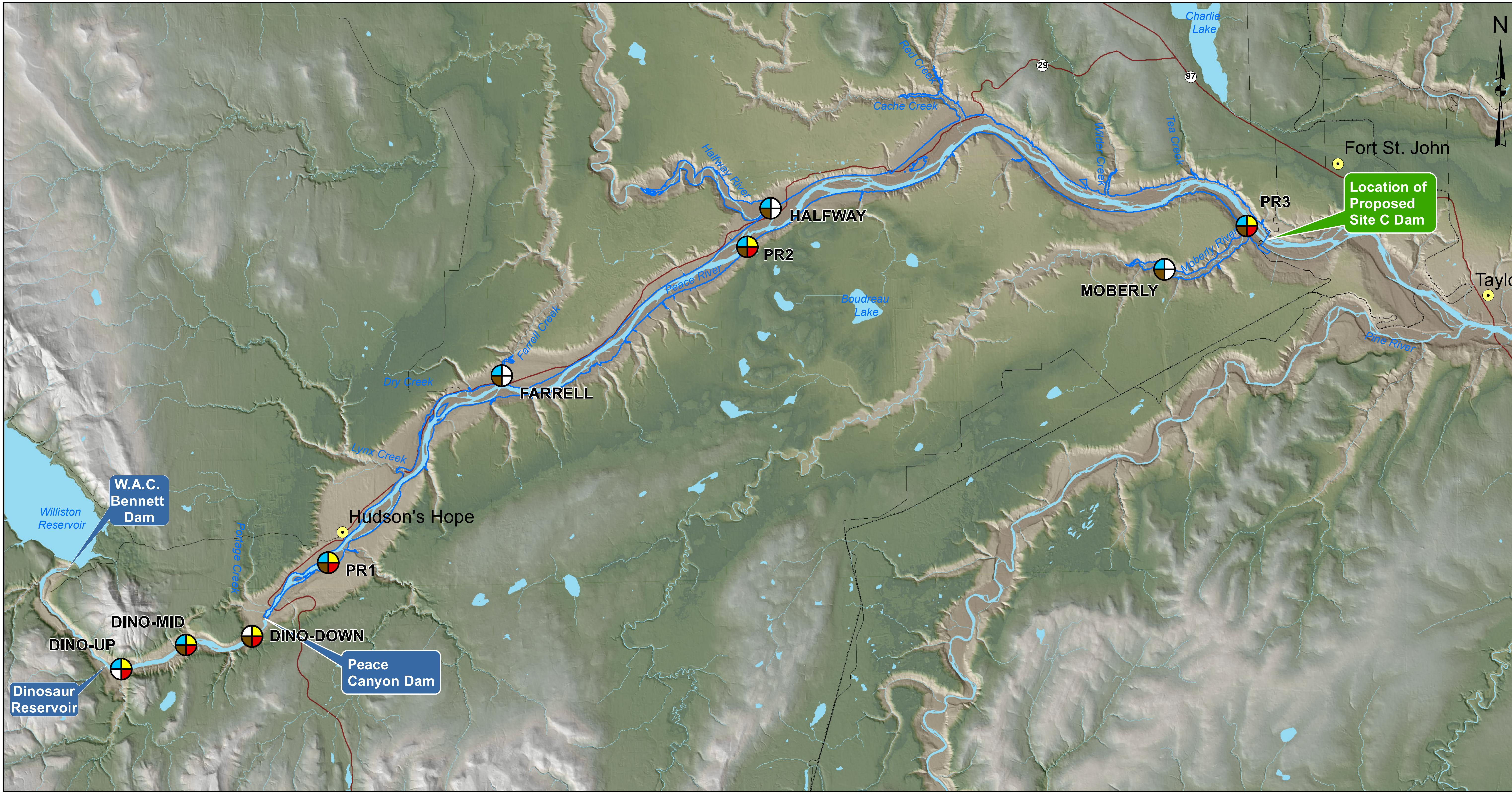
### 4.4.2 Baseline Chemical Conditions

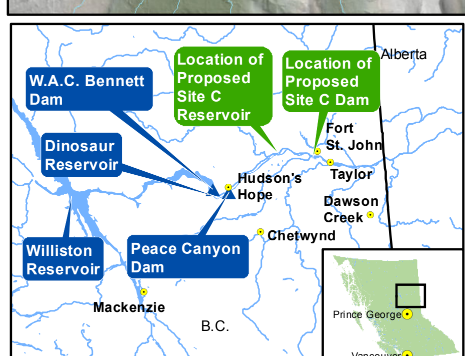
Seasonal and annual trends in key water chemistry parameters summarized in **Table 4.1** are derived primarily from several sources including Pattenden et al. (1990, 1991), Golder (2009a, 2009b, 2012) and Azimuth (2011). Most of the parameters discussed were measured seasonally at various locations in Williston Reservoir in the early 2000s and in

Dinosaur Reservoir and the Peace River and its major tributaries (Farrell, Halfway, Moberly) in 2006 and 2007 (Golder 2009a, 2009b) and again in 2010 and 2011 (Golder 2012).

Further information on baseline water chemistry of the Peace River system can be found in Appendix E, Water Quality Technical Data Report.

**Figure 4.1** depicts the locations of water, sediment and lower trophic level biota (zooplankton and benthic invertebrates) from Dinosaur Reservoir and Peace River above the Site C dam location for analysis of key supporting chemical variables and total and methylmercury concentrations. Water was collected in 2010, 2011 and 2012 for low level mercury and in 2010 and 2011 for benthic invertebrates at the locations indicated in **Figure 4.1**.




Map Notes:
1. Datum: NAD83
2. Projection: UTM Zone 10N
3. Base Data: Province of B.C.
4. Proposed reservoir area (461.8m maximum normal elevation) from Digital Elevation Models (DEM) generated from LiDAR data acquired July/August, 2006.

**Legend**

Parameters Collected

Water — Zooplankton — Sediment — Benthos

Proposed Reservoir
Road
Highway
Railway

1:250,000

0 — 10 km

BC hydro

SITE C CLEAN ENERGY PROJECT

**Figure 4.1: Water, Sediment, Zooplankton & Benthic Invertebrate Sampling Locations, Peace River**

| Date | Oct 17, 2012 | DWG NO | 1016-C14-B6169 - 2 | R 0 |
|---|---|---|---|---|

Construction of the Site C Clean Energy Project is subject to required regulatory approvals including environmental certification.



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**Table 4.1 Key water chemistry parameters used in mercury modeling**

| | | | | | | | | | | | TOTAL | | DISSOLVED |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Year | Date or Range[1] | Area | Station ID | pH unitless | Total Alkalinity mg/L | Hardness mg/L | TSS mg/L | TDS mg/L | TOC mg/L | DOC mg/L | Total Mercury ng/L | Methyl ng/L | Total Mercury ng/L |
| 1989 | 10-Oct-89 | Peace River | DS of Peace Canyon | - | - | - | <1 | - | - | - | <50 | - | - |
| 1989 | 10-Oct-89 | Tributary | Maurice Creek | - | - | - | <1 | - | - | - | <50 | - | - |
| 1989 | 10-Oct-89 | Tributary | Lynx Creek | - | - | - | 241 | - | - | - | 80 | - | - |
| 1989 | 10-Oct-89 | Tributary | Farrell Creek | - | - | - | 7.0 | - | - | - | <50 | - | - |
| 1989 | 10-Oct-89 | Tributary | Halfway River | - | - | - | 9.0 | - | - | - | <50 | - | - |
| 1989 | 10-Oct-89 | Tributary | Cache Creek | - | - | - | 7.0 | - | - | - | <50 | - | - |
| 1989 | 10-Oct-89 | Tributary | Moberly River | - | - | - | 14 | - | - | - | <50 | - | - |
| 1989 | 10-Oct-89 | Peace River | near Site C | - | - | - | 14 | - | - | - | <50 | - | - |
| 1990 | 4-Jun-90 | Peace River | DS of Peace Canyon | - | - | - | 3.0 | - | - | - | 50 | - | - |
| 1990 | 13-Aug-90 | Peace River | DS of Peace Canyon | - | - | - | <1 | - | - | - | 140 | - | - |
| 1990 | 14-Oct-90 | Peace River | DS of Peace Canyon | - | - | - | <1 | - | - | - | <50 | - | - |
| 1990 | 4-Jun-90 | Tributary | Maurice Creek | - | - | - | 152 | - | - | - | 60 | - | - |
| 1990 | 13-Aug-90 | Tributary | Maurice Creek | - | - | - | <1 | - | - | - | 70 | - | - |
| 1990 | 14-Oct-90 | Tributary | Maurice Creek | - | - | - | <1 | - | - | - | <50 | - | - |
| 1990 | 4-Jun-90 | Tributary | Lynx Creek | - | - | - | 543 | - | - | - | 70 | - | - |
| 1990 | 13-Aug-90 | Tributary | Lynx Creek | - | - | - | 89 | - | - | - | <50 | - | - |
| 1990 | 14-Oct-90 | Tributary | Lynx Creek | - | - | - | 86 | - | - | - | 70 | - | - |
| 1990 | 4-Jun-90 | Tributary | Farrell Creek | - | - | - | 722 | - | - | - | 120 | - | - |
| 1990 | 13-Aug-90 | Tributary | Farrell Creek | - | - | - | <1 | - | - | - | 120 | - | - |
| 1990 | 14-Oct-90 | Tributary | Farrell Creek | - | - | - | <1 | - | - | - | <50 | - | - |
| 1990 | 4-Jun-90 | Tributary | Halfway River | - | - | - | 1713 | - | - | - | 120 | - | - |
| 1990 | 13-Aug-90 | Tributary | Halfway River | - | - | - | 7.0 | - | - | - | 190 | - | - |
| 1990 | 14-Oct-90 | Tributary | Halfway River | - | - | - | 1.7 | - | - | - | 70 | - | - |
| 1990 | 4-Jun-90 | Tributary | Cache Creek | - | - | - | 451 | - | - | - | 120 | - | - |
| 1990 | 13-Aug-90 | Tributary | Cache Creek | - | - | - | 4.0 | - | - | - | 160 | - | - |
| 1990 | 14-Oct-90 | Tributary | Cache Creek | - | - | - | 1.4 | - | - | - | <50 | - | - |
| 1990 | 4-Jun-90 | Tributary | Moberly River | - | - | - | 528 | - | - | - | 70 | - | - |
| 1990 | 13-Aug-90 | Tributary | Moberly River | - | - | - | 5.0 | - | - | - | 160 | - | - |
| 1990 | 14-Oct-90 | Tributary | Moberly River | - | - | - | <1 | - | - | - | <50 | - | - |
| 1990 | 4-Jun-90 | Peace River | near Site C | - | - | - | 1936 | - | - | - | 210 | - | - |
| 1990 | 13-Aug-90 | Peace River | near Site C | - | - | - | 1.0 | - | - | - | 140 | - | - |
| 1990 | 14-Oct-90 | Peace River | near Site C | - | - | - | 2.2 | - | - | - | <50 | - | - |
| 2000 | 14-Aug-00 | Williston | Junction | - | - | - | <3 | - | - | 3.3 | 0.68 | 0.041 | 0.63 |
| 2000 | 15-Aug-00 | Williston | Finlay Profundal | - | - | - | <3 | - | - | 2.3 | 0.51 | 0.019 | 0.47 |
| 2000 | 16-Aug-00 | Williston | Finlay Littoral | - | - | - | <3 | - | - | 1.8 | 1.46 | 0.055 | 0.53 |
| 2000 | 17-Aug-00 | Williston | Finlay River | - | - | - | 8.0 | - | - | 1.6 | 0.39 | 0.033 | 0.38 |
| 2000 | 17-Aug-00 | Williston | Davis River | - | - | - | <3 | - | - | 1.2 | 0.36 | 0.056 | 0.37 |
| 2000 | 20-Aug-00 | Williston | Omineca River | - | - | - | <3 | - | - | 1.5 | 1.10 | 0.058 | 0.89 |
| 2000 | 20-Aug-00 | Williston | Omineca Arm | - | - | - | <3 | - | - | 2.0 | 1.22 | 0.036 | 0.67 |
| 2000 | 17-Aug-00 | Williston | Ingenika River | - | - | - | <3 | - | - | 1.8 | 0.44 | 0.022 | 0.41 |
| 2001 | 13-Jun-01 | Williston | Finlay Profundal | - | - | - | <3 | - | - | 2.0 | 1.06 | 0.083 | 0.44 |
| 2001 | 14-Jun-01 | Williston | Finlay Littoral | - | - | - | <3 | - | - | 2.0 | 0.93 | 0.041 | 0.73 |
| 2001 | 16-Jun-01 | Williston | Finlay River | - | - | - | 266 | - | - | 2.4 | 12.5 | 0.073 | 1.39 |
| 2001 | 13-Jun-01 | Williston | Davis River | - | - | - | 124 | - | - | 3.0 | 4.02 | 0.137 | 1.75 |
| 2001 | 13-Jun-01 | Williston | Omineca River | - | - | - | 48 | - | - | 5.7 | 6.63 | 0.067 | 3.17 |
| 2001 | 13-Jun-01 | Williston | Mesilinka River | - | - | - | 70 | - | - | 5.8 | 4.82 | 0.025 | 2.25 |
| 2001 | 13-Jun-01 | Williston | Ospika River | - | - | - | 1180 | - | - | 2.6 | 27.9 | 0.122 | 1.48 |
| 2001 | 16-Jun-01 | Williston | Ingenika River | - | - | - | 90 | - | - | 2.8 | 2.53 | 0.048 | 1.29 |
| 2001 | 16-Jun-01 | Williston | Swannell River | - | - | - | 117 | - | - | 3.8 | 5.45 | 0.191 | 2.44 |
| 2006-07 | Nov-Aug (n=7) | Peace River | Peace 1 | 7.88 - 8.22 | 72 - 91 | 91 - 105 | <3.0 - 14.8 | 98 - 111 | 2.1 - 3.2 | 2.04 - 2.75 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=6) | Tributary | Lynx-10 | 8.16 - 8.40 | 163 - 466 | 207 - 471 | 8.5 - 1960 | 251 - 457 | 2.6 - 57 | 1.93 - 12.6 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=6) | Tributary | Farrell-11 | 8.10 - 8.48 | 100 - 293 | 124 - 428 | 6.7 - 178 | 179 - 582 | 4.2 - 46 | 3.87 - 15.3 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=6) | Peace River | Peace 2 | 8.07 - 8.20 | <2.0 - 85 | 92 - 106 | <3.0 - 19.2 | 104 - 109 | 2.4 - 3.6 | 2.29 - 2.86 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=5) | Tributary | Halfway-9 | 8.19 - 8.47 | 130 - 232 | 166 - 260 | <3.0 - 686 | 202 - 299 | 1.5 - 9.6 | 1.30 - 4.13 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=4) | Tributary | Boudreau-13 | 8.00 - 8.12 | 123 - 286 | 189 - 775 | <3.0 - 23.6 | 310 - 964 | 12 - 30 | 11.1 - 29.2 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=6) | Tributary | Cache-12 | 7.91 - 8.32 | 96 - 355 | 140 - 652 | 4.5 - 2760 | 335 - 1030 | 8.2 - 30 | 6.78 - 22.5 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=8) | Peace River | Peace 3 | 8.03 - 8.24 | 78 - 117 | 91 - 130 | 3.0 - 1020 | 102 - 165 | 2.5 - 5.9 | 2.29 - 3.71 | <50 | - | <50 |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 2006-07 | Mar-Aug (n=4) | Tributary | Moberly-7 | 8.10 - 8.28 | 76 - 108 | 97 - 123 | 7.5 - 651 | 114 - 142 | 4.8 - 11 | 4.72 - 7.06 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=11) | Peace River | Peace 4 | 7.47 - 8.23 | 77 - 116 | 89 - 131 | 3.5 - 1010 | 98 - 162 | <0.5 - 4.8 | <0.5 - 3.56 | <50 | - | <50 |
| 2006-07 | Mar-Aug (n=8) | Peace River | Peace 5 | 7.86 - 8.20 | 69 - 97 | 90 - 117 | 4.5 - 1570 | 103 - 280 | 2.6 - 13 | 2.21 - 8.00 | <50 | - | <50 |
| | | | | | | | | | | | | | |
| 2008 | Mar-Oct (n=9) | Peace River | Peace 1 | 7.92 - 8.15 | 76 - 85 | 83 - 98 | <3.0 - 5.2 | 94 - 107 | 2.4 - 3.3 | 2.17 - 2.91 | <50 | - | <50 |
| 2008 | Mar-Oct (n=6) | Tributary | Lynx-10 | 8.09 - 8.36 | 162 - 458 | 172 - 509 | 14.5 - 1950 | 209 - 522 | 4.0 - 32 | 2.29 - 16.8 | <50 | - | <50 |
| 2008 | Mar-Oct (n=5) | Tributary | Farrell-11 | 7.99 - 8.49 | 83 - 273 | 90 - 310 | <3.0 - 1520 | 181 - 357 | 5.8 - 40 | 5.39 - 22.4 | <50 | - | <50 |
| 2008 | Mar-Oct (n=6) | Peace River | Peace 2 | 7.90 - 8.21 | 80 - 86 | 86 - 99 | 4.2 - 62 | 97 - 127 | 2.6 - 4.5 | 2.47 - 3.38 | <50 | - | <50 |
| 2008 | Mar-Oct (n=6) | Tributary | Halfway-9 | 7.90 - 8.37 | 82 - 213 | 105 - 259 | <3.0 - 497 | 174 - 282 | 1.4 - 34 | 1.27 - 23.6 | <50 | - | <50 |
| 2008 | Mar-Oct (n=5) | Tributary | Boudreau-13 | 7.96 - 8.20 | 103 - 305 | 120 - 528 | 11.3 - 409 | 209 - 676 | 12 - 33 | 10.8 - 30.8 | <50 | - | <50 |
| 2008 | Mar-Oct (n=6) | Tributary | Cache-12 | 7.74 - 8.19 | 88 - 350 | 106 - 756 | 3.0 - 1620 | 125 - 2420 | 6.8 - 54 | 5.79 - 30.1 | <50 | - | <50 |
| 2008 | Mar-Oct (n=6) | Peace River | Peace 3 | 8.01 - 8.25 | 78 - 99 | 89 - 119 | 6.2 - 162 | 101 - 126 | 2.8 - 11 | 2.43 - 6.40 | <50 | - | <50 |
| 2008 | Mar-Oct (n=5) | Tributary | Moberly-7 | 8.06 - 8.39 | 102 - 237 | 95 - 148 | <3.0 - 407 | 124 - 637 | 4.2 - 18 | 3.65 - 16.6 | <50 | - | <50 |
| 2008 | Mar-Oct (n=6) | Peace River | Peace 4 | 8.01 - 8.26 | 79 - 103 | 88 - 126 | 4.7 - 189 | 101 - 128 | 2.4 - 7.8 | 2.17 - 5.20 | <50 | - | <50 |
| 2008 | Mar-Oct (n=7) | Peace River | Peace 5 | 7.86 - 8.30 | 51 - 96 | 63 - 112 | 7.5 - 1640 | 104 - 156 | 2.5 - 35 | 2.21 - 18.5 | 130 | - | <50 |
| | | | | | | | | | | | | | |
| 2010 | 02-Jul-10 | Dinosaur | Upper-reservoir | 8.22 | 81.1 | 91.3 | 1.3 | - | 2.08 | 2.0 | <1.0 | <0.050 | |
| 2010 | 30-Aug-10 | Dinosaur | Upper-reservoir | 7.91 | 83.2 | 90.7 | 0.7 | - | 2.60 | 2.6 | 0.60 | <0.020 | 0.54 |
| 2010 | 02-Jul-10 | Dinosaur | Mid-reservoir | 8.09 | 81.5 | 90.5 | 1.4 | - | 2.14 | 2.2 | <1.0 | <0.050 | <1.0 |
| 2010 | 30-Aug-10 | Dinosaur | Mid-reservoir | - | 86.0 | - | 1.1 | - | 2.40 | 2.3 | 0.64 | <0.020 | 0.62 |
| 2010 | 29-Jun-10 | Peace River | PR-1 | 8.64 | 82.6 | 100 | 0.8 | - | 1.94 | 2.0 | <1.0 | <0.050 | <1.0 |
| 2010 | 29-Aug-10 | Peace River | PR-1 | 7.92 | 84.4 | 90.1 | 1.2 | - | 2.90 | 2.8 | 0.63 | <0.020 | 0.70 |
| 2010 | 03-Jul-10 | Tributary | Farrell Creek | 8.51 | 200 | 251 | 4.4 | - | 7.46 | 7.1 | 1.40 | 0.101 | <1.0 |
| 2010 | 31-Aug-10 | Tributary | Farrell Creek | 7.98 | 178 | 270 | 2.1 | - | 6.00 | 5.9 | 1.31 | 0.030 | 1.40 |
| 2010 | 29-Jun-10 | Peace River | PR-2 | 7.93 | 83.4 | 102 | 1.2 | - | 2.00 | 2.1 | <1.0 | <0.050 | <1.0 |
| 2010 | 29-Aug-10 | Peace River | PR-2 | 7.89 | 84.3 | 91.4 | 1.6 | - | 2.70 | 2.7 | 0.70 | <0.020 | 0.77 |
| 2010 | 03-Jul-10 | Tributary | Halfway River | 8.43 | 171 | 220 | 15 | - | 1.89 | 1.9 | 1.50 | <0.050 | <1.0 |
| 2010 | 27-Aug-10 | Tributary | Halfway River | 8.23 | 173 | 219 | 16 | - | 2.00 | 1.9 | 3.44 | <0.020 | 1.40 |
| 2010 | 30-Jun-10 | Peace River | PR-3 | 8.20 | 99.2 | 126 | 4.7 | - | 1.03 | 2.1 | <1.0 | <0.050 | <1.0 |
| 2010 | 28-Aug-10 | Peace River | PR-3 | 8.23 | 90.5 | 100 | 2.3 | - | 2.60 | 2.5 | 0.85 | <0.020 | 1.24 |
| 2010 | 03-Jul-10 | Tributary | Moberly River | 8.38 | 98.7 | 112 | 20 | - | 5.04 | 5.1 | 2.00 | 0.093 | <1.0 |
| 2010 | 07-Oct-10 | Tributary | Moberly River | 7.49 | 124 | 137 | 4.0 | - | 4.37 | 4.2 | <1.0 | 0.064 | <1.0 |
| | | | | | | | | | | | | | |
| 2011 | 10-May-11 | Peace River | Peace 1 | 7.2 | 84.5 | 99.6 | <3 | 110.1 | - | 3.2 | <1.0 | <0.050 | <10 |
| 2011 | 18-Jul-11 | Peace River | Peace 1 | 7.74 | 83.6 | 96.3 | 18 | 108.7 | - | 3.9 | 1.9 | <0.050 | <10 |
| 2011 | 13-Sep-11 | Peace River | Peace 1 | 8.05 | 75.8 | 89.3 | <3 | 101.2 | - | 3.5 | <1.0 | <0.050 | <10 |
| 2011 | 10-May-11 | Peace River | Peace 2 | 7.48 | 84.5 | 99 | 22 | 109.6 | - | 4.9 | 1.6 | <0.050 | <10 |
| 2011 | 19-Jul-11 | Peace River | Peace 2 | 7.65 | 87.2 | 100 | 74 | 113.5 | - | 5.4 | 4.0 | <0.050 | <10 |
| 2011 | 13-Sep-11 | Peace River | Peace 2 | 8.15 | 76.1 | 88.9 | <3 | 101.5 | - | 3.7 | <1.0 | <0.050 | <10 |
| 2011 | 19-May-11 | Tributary | Halfway (lower) | 7.81 | 89.6 | 114 | 1960 | 122.3 | - | 15.7 | 110 | 0.337 | <10 |
| 2011 | 27-Jul-11 | Tributary | Halfway (lower) | - | - | - | - | - | - | - | 11.8 | <0.050 | - |
| 2011 | 15-Sep-11 | Tributary | Halfway (lower) | 8.43 | 200 | 261 | 6 | 275.1 | - | 2.9 | <1.0 | <0.050 | <10 |
| 2011 | 12-May-11 | Peace River | Peace 3 | 7.78 | 82.3 | 100 | 128 | 109.8 | - | 6.1 | 14.1 | 0.079 | <10 |
| 2011 | 10-Sep-11 | Peace River | Peace 3 | 7.97 | 80.5 | 84.7 | 4 | 105.4 | - | 3.3 | 1.2 | <0.050 | <10 |
| 2011 | 26-Jul-11 | Tributary | Moberly (lower) | 7.88 | 95.1 | 111 | 332 | 120.4 | - | 11.5 | 32.9 | 0.128 | <10 |
| 2011 | 23-Sep-11 | Tributary | Moberly (lower) | 8.13 | 155 | 174 | 22 | 197.2 | - | 5.6 | 1.9 | <0.050 | <10 |
| 2011 | 12-May-11 | Peace River | Peace 4 | 7.85 | 82.6 | 99.9 | 193 | 110.3 | - | 7.4 | 17 | - | <10 |
| 2011 | 2-Jun-11 | Peace River | Peace 4 | 7.91 | 104 | 124 | 336 | 132.2 | - | 5.3 | 15 | - | <10 |
| 2011 | 26-Jul-11 | Tributary | Pine-16 | 7.99 | 128 | 153 | 68 | 166.5 | - | 5.5 | 7.6 | <0.05 | <10 |
| 2011 | 10-Sep-11 | Tributary | Pine-16 | 8.29 | 140 | 150 | <3 | 180.5 | - | 2.2 | <1.0 | <0.05 | <10 |
| 2011 | 27-May-11 | Peace River | Peace 14 | 8.03 | 99.5 | 114 | 503 | 126.3 | - | 6.6 | 19 | - | <10 |
| 2011 | 27-May-11 | Peace River | Peace 15 | 8.07 | 97.1 | 112 | 530 | 120.2 | - | 6.7 | 22 | - | <10 |

**Notes:** [1] n = # samples when > 1. **Data Sources:** *Pattenden et al. 1990* - Peace River and Tributary 1989 data. *Pattenden et al. 1991* - Peace River and Tributary 1990 data. *Baker 2002* - Williston 2000 and 2001 data. *Golder 2009a* - Peace River and Tributary 2007 data. *Golder 2009b* - Peace River and Tributary 2008 data. *Azimuth 2011* - Peace River, Dinosaur and Tributary 2010 data. *Golder 2012* - Peace River, Dinosaur and Tributary 2011 data.

A summary of results for key parameters influencing mercury methylation dynamics for the Peace River (including Williston Reservoir) are presented below and in **Table 4.1**.

**pH** – The Peace River system is slightly alkaline (pH >7.0) and relatively well characterized with a pH ranging from 7.5 to 8.4 and a mean of about 8.1, which is slightly basic (Stockner et al 2005; Golder 2009a, 2009b). Tributary stream pH is slightly higher than mainstem pH values. Dinosaur Reservoir pH in 2010 was 7.9 to 8.2 and is a direct reflection of the pH of Williston Reservoir that dominates chemistry of the Peace River downstream and particularly within the proposed reservoir area of Site C. There were no differences in pH of the river between Dinosaur Reservoir and downstream of Moberly River. As noted above, slightly acidic water pH favors mercury methylation. Reservoirs with pH values that range from 5.8 to 6.7 contain fish with higher mercury than reservoirs and lakes that are circumneutral (Wiener et al. 1990; Miskimmin et al. 1992; Greenfield et al. 2001).

**Hardness** – Hardness (a measure of the mineral content of water; soft water contains less dissolved minerals, particularly calcium and magnesium) in the Peace River downstream of Peace Canyon Dam is low. In 2011 Peace River mainstem hardness was typically <100 mg/L while tributaries (Halfway, Moberly, Pine) ranged up to 260 mg/L. Water from tributaries will almost always be higher in most parameters because of the higher sediment load that they carry whereas solids tend to settle out upstream of WAC Bennett Dam in Williston Reservoir.

**Total suspended solids** – TSS concentrations vary considerably seasonally, episodically and annually depending on rainfall events and snowmelt and freshet flow volume that contribute to tributary discharge and TSS load to the Peace River mainstem. Relative influence of TSS on the Peace River is directly related to watershed size and water volume and to distance downstream of the Peace Canyon Dam. The further downstream the greater the relative influence of tributary inflow on TSS (and other parameters, including metals) on the Peace. In springtime, TSS concentrations should increase from upstream to downstream as more sediment is carried into the river.

At most times of the year, TSS concentration in the Peace River mainstem is below the laboratory detection limit of 3 mg/L or 1.0 mg/L, depending on the laboratory. In general, the TSS concentration in Peace River downstream of the Peace Canyon Dam ranges from <1 mg/L to about 4 mg/L, due to the very large settling capacity of Williston Reservoir upstream, that contributes very little in the way of suspended solids.

Tributary inputs of TSS vary greatly depending on time of year, strength of freshet flow, rainfall, catchment area and local soil types and erodability. It is not unusual for tributary inputs to have TSS concentrations ranging from hundreds to thousands of milligrams per liter (**Table 4.1**). In 2011 strong freshet and large rainfall caused large increases in TSS with high concentrations in Moberly (332 mg/L), Halfway (1,960 mg/L) and Pine (153 mg/L). These tributary inputs also caused an increase in mainstem TSS concentrations from upstream at Peace 1 (18 mg/L), Peace 2 (74 mg/L), Peace 3 (128 mg/L; **Figure 4.1**), Peace 4 (336 mg/) and Peace 15 (530 mg/L) in May. By September, concentrations had returned to <3 mg/L. Further detail on the influence of spring freshet and flood flow conditions on TSS and water chemistry in general can be found in Golder (2012).

There is a positive correlation between total mercury concentration and TSS, which is evident in **Table 4.1** and described below. Inorganic mercury, like most other metals is adsorbed to fine sediment particles and is transported downstream and ultimately settles out in depositional areas of rivers or lakes. Very little of the mercury is in the dissolved phase that is available to be taken up by biota.

**Total and dissolved organic carbon** – Total (TOC) and dissolved organic carbon (DOC) concentrations in Dinosaur Reservoir and the Peace River varied between 2 and 2.8 mg/L in 2010, which are in the low to moderately low range. TOC in the Halfway (2 mg/L) and Moberly (4.4 and 5 mg/L) were similar. DOC concentrations made up >90% of the TOC indicating that the vast majority of the waterborne carbon was in the dissolved phase. DOC concentrations in 2011 were higher (2.9 to 6.1 mg/L) (**Table 4.1**) because of the greater amount of runoff and contributions from tributary streams (e.g., 15.7 mg/L in Halfway, 11.5 mg/L in Moberly). Based on average flow (2010) and high flow years (2011), organic carbon conditions are well characterized with some variability through the system depending on tributary inputs. DOC concentrations in excess of 5 mg/L are associated with greater methylmercury production.

**Sulphate** – Sulphate is a nutrient for sulphate-reducing bacteria that are responsible for methylmercury production in aquatic systems. There is a positive correlation between methylation rate and sulphate (Rudd 1995) over environmentally relevant concentrations (5 to 30 mg/L). Concentrations in Peace River (12 to 15 mg/L) are in the moderate range (Golder 2009a). Given the relatively remote geography of the area inputs of sulphates from industrial activities are presumed to be low and will not exacerbate methylation.

**Total Mercury** – To provide perspective, total mercury (i.e., all forms, including inorganic and methylmercury) concentrations in remote, pristine areas removed from anthropogenic (e.g., coal-fired generating stations, chlor-alkali facilities) and natural (i.e., mineralized areas, volcanoes) are low and range in concentration from <1 ng/L (i.e., parts per trillion or 1,000 µg/L) to 3 ng/L. Routine analyses by commercial laboratories do not have the analytical capabilities to detect mercury below 50 ng/L or more recently 10 to 20 ng/L. Thus, all historic mercury data on the Peace River were below the laboratory DL of 50 ng/L, except for a specialized study of Williston Reservoir (Baker et al. 2002) and more recently in Dinosaur Reservoir and the Peace River in 2010 and 2011 when low level detection methods were used for mercury in water (**Table 4.1**).

In Finlay Reach of Williston Reservoir in 2000/2001, the average total mercury concentration was less than 1 ng/L (0.5 to 1.4 ng/L) in both surface and deep (profundal) water (Baker et al. 2002). Large tributary streams were also low in mercury (0.4 to 1.2 ng/L). In spring 2001, reservoir concentrations were still low (0.9 to 1.1 ng/L), while tributary streams with high TSS had higher mercury concentrations (4 to 28 ng/L) during freshet, which is typical.

During spring freshet flow from major tributary streams (Halfway, Moberly) total mercury concentration in water is elevated and is always associated with elevated TSS concentrations. However, in historic studies of the Peace River and its tributaries, the detection limit for total mercury was 50 ng/L and mercury could not be detected except during high TSS events, such as during freshet (e.g., Lynx Creek 80 ng/L; 241 mg/L TSS; Pattenden et al. 1990). More recently collected data use lower detection limits and demonstrate that Peace River mainstem total mercury concentrations hover around 1 ng/L and are considered typical of pristine, remote watersheds.

In general, there were no meaningful differences in total mercury concentrations between spring and fall in the Peace River, except during extreme flood events when large amounts of TSS are contributed to the mainstem from tributary runoff. Otherwise, low (~1 ng/L) and consistent concentrations are a reflection of the consistency of water quality within upgradient Williston Reservoir, dating back at least one decade. Recent data confirm that mercury concentrations in Peace River water are very low and typical of pristine systems (Hurley et al. 1995; Krabbenhoft et al. 1999; Krabbenhoft et al. 2007) and similar to what were observed in Williston Reservoir in 2000 and 2001 (Baker et al. 2002).

**Methylmercury** – Methyl or organic mercury concentrations are seldom measured in water because concentrations are typically extremely low (i.e., <0.1 ng/L) and a specialized mercury dedicated laboratory is required to detect these concentrations. Methylmercury was measured in Williston Reservoir in 2000/2001 and from Peace River and tributaries in 2010 and 2011. In Williston, MeHg concentrations ranged from <0.05 ng/L to 0.088 ng/L accounting for 3% to 15% of the total concentration (Baker et al. 2002). In 2010 / 2011, methylmercury in Dinosaur Reservoir and the Peace River technical study area was below the detection limit of 0.05 ng/L on nearly all occasions (**Table 4.1**). The exception was Peace 3 near the proposed Site C dam in May 2011 (0.08 ng/L) with a TSS of 128 mg/L.

Methylmercury concentration exceeded the laboratory detection limit on a few occasions in 2010 and 2011 from tributary streams and usually only when TSS concentrations were relatively high (e.g., 0.13 ng/L in Moberly, 332 mg/L TSS; 0.34 ng/L in Halfway, 1,960 mg/L TSS).

In remote systems removed from potential anthropogenic sources of mercury, the concentration of methylmercury in water typically ranges from 0.04 to 0.8 ng/L. As a percentage of the total mercury concentration in such systems, methylmercury usually comprises between 1% and 10% of the total (Hurley et al. 1995; Krabbenhoft et al. 1999; St. Louis et al. 1995; Bodaly et al. 1997; 2004). These values are very consistent with what has been observed in the Peace River, as methylmercury comprised less than 7% of the total in those instances where DLs were exceeded. Furthermore, methylmercury concentrations in water from the Peace River mainstem are similar to what was observed in Williston Reservoir in 2000/2001 (Baker et al. 2002) indicating that conditions have not changed for at least a decade.

**Mercury Guideline in Water for Aquatic Life Protection** – There are two commonly used guidelines for mercury in water to protect aquatic life: the federal Canadian Council for Ministers of the Environment (CCME 2003) guideline of 26 ng/L, and the BC provincial guideline that varies between 2.0 and 20 ng/L. The BC site-specific guideline is used when methylmercury data are available and is then derived from the ratio or proportion that methylmercury comprises of the total concentration. For example, if the percent of methylmercury relative to total is low (i.e., 0.5% or less), the 30-day guideline concentration is 20 ng/L; if the ratio is greater than 5%, the guideline is lower, at 2.0 ng/L. The guideline was developed to provide a concentration of mercury in water below which (in theory), mercury in the tissue of aquatic life would not exceed a concentration of 0.033 mg/kg ww, which is known as the ‘tissue residue guideline’, developed by CCME. In the Peace River technical study area total and methylmercury concentrations are quite low and based on the ratio of total to methylmercury the site-specific 30-day guideline concentration would range between 2 and 4 ng/L.

## 4.5 Sediment Chemistry

### 4.5.1 Background

Persistent environmental contaminants (natural or human-related) entering aquatic systems usually end up associated with bottom sediments. The mechanism by which this typically occurs is through adsorption to suspended particulate matter in the water column that eventually settles in depositional areas as sediment, especially in deep areas of lakes and reservoirs, and occasionally in backwater areas of large rivers. Sediment, therefore, acts as an integrator of contamination in aquatic systems and can become both a sink and potential source for contaminants within a system. The degrees to which sediments function this way depend on the contaminant and the physical conditions of the environment (temperature, oxygen, redox, pH, grain size, etc.). As discussed above in the water chemistry section, sediment particles act as transport vehicles for many metals including mercury as there is a strong positive correlation between water-borne TSS and mercury. Further information on sediment regime, erosion and transport in the Peace River system can be found in Volume 2 Chapter 11.8 Fluvial Geomorphology and Sediment Transport and in Volume 2, Appendix I, Fluvial Geomorphology.

Of the most common forms of mercury in the environment, inorganic mercury is by far the most dominant species, adhered to fine sediment and organic particles in the sediment. A small amount of the total mercury (usually around 1% is methylmercury), liberated from the sediment sink via methylation, although the concentrations are extremely small, especially in rivers that are poor methylating environments. As described earlier, the methylation of inorganic mercury by sulphur-reducing bacteria present in anoxic sediments is the primary source of methylmercury. Given that rivers are dynamic, well oxygenated environments this is one of the reasons why the methylmercury concentration in river sediments is usually quite low. The concentration of inorganic and methylmercury is correlated positively with small grain size (i.e., clay and silt, not sand/gravel) and total organic carbon (TOC) particles. In the Peace River, grain size is dominated by coarse materials so neither form of mercury would be expected to be present in high concentrations. Nevertheless, some mercury data are available in sediment from Williston Reservoir (Baker et al. 2002) and the Peace River from 2007 (Golder 2009a) and 2010 field studies (Azimuth 2011).

### 4.5.2 Baseline Conditions

Available mercury data for the Peace River system are presented in **Table 4.2**. The earliest total mercury concentrations in sediment were measured from Williston Reservoir (Finlay Reach) in 2000 and 2001. Here concentrations were relatively low (0.022 to 0.092 mg/kg dw; parts per million), with methylmercury concentration (0.17 to 1.90 µg/kg dw; parts per billion), comprising less than 1% of the total mercury concentration (Baker et al. 2002). These values are typical for lake sediments in remote areas away from anthropogenic inputs. Note that methylmercury concentration in Williston Reservoir sediment was similar to a lake, indicating that elevated concentrations normally found in ‘new’ reservoirs are no longer present.

Total mercury concentrations in sediment from mainstem stations along the Peace River in 2007 were in most cases below the laboratory detection limit (0.05 mg/kg), but were relatively low (0.053 to 0.110 mg/kg dw) when detectable (Golder 2009a). Total mercury concentrations were also non-detectable in the tributaries, except for one sample from Moberly River (0.057 mg/kg dw). These low mercury concentrations are partly due to the coarse grain size of the river sediments (48% to 80% sand; **Table 4.2**). Total organic content of the sediment was also low, averaging less than 1% of sediment biomass.

In 2010, sediment sampling within the mainstem targeted fine sediments (>85% silt/clay) dispersed within and beneath the dominant sand/gravel and cobble substrate (**Figure 4.1**; Azimuth 2011). Total mercury in Dinosaur Reservoir and Peace River ranged from 0.032 to 0.17 mg/kg dw which is similar to the range in concentration within Williston Reservoir 12 years earlier (**Table 4.1**). Methylmercury concentrations in Dinosaur Reservoir (0.12 to 0.29 µg/kg) were also very similar to what was observed in Williston Reservoir again, comprising less than 0.5% of the total.

**Table 4.2 Grain size, TOC and mercury concentration in sediment**

| Year | Date | Area[1] | Station ID | TOC % | Sand % | Silt&Clay % | Total Mercury mg/kg dw | Methyl-mercury ug/kg dw | MeHg/THg[2] % |
|---|---|---|---|---|---|---|---|---|---|
| 2000 | 14-Aug-00 | Williston (n=3) | Junction | - | - | - | 0.080-0.091 | 0.41-0.59 | 0.6 |
| 2000 | 15-Aug-00 | Williston (n=3) | Finlay Profundal | - | - | - | 0.055-0.092 | 0.28-1.90 | 1.2 |
| 2000 | 16-Aug-00 | Williston (n=3) | Finlay Littoral | - | - | - | 0.022-0.066 | 0.17-0.55 | 1.0 |
| 2000 | 20-Aug-00 | Williston (n=2) | Omineca Arm | - | - | - | 0.078-0.086 | 0.25-0.53 | 0.5 |
| | | | | | | | | | |
| 2001 | 13-Jun-01 | Williston (n=2) | Finlay Profundal | - | <1 | 99 | 0.032-0.084 | 0.41-0.47 | 1.0 |
| 2001 | 14-Jun-01 | Williston (n=2) | Finlay Littoral | - | 5.0 | 95 | 0.030-0.040 | 0.14-0.23 | 0.5 |
| 2001 | 13-Jun-01 | Williston (n=2) | Davis Bay | - | <1 | 99 | 0.026-0.027 | 0.31-0.35 | 1.2 |
| 2001 | 17-Jun-01 | Williston (n=2) | Chowika Bay | - | 4.0 | 96 | 0.041-0.048 | 0.11-0.13 | 0.3 |
| 2001 | 17-Jun-01 | Williston (n=2) | Collins Bay | - | 1.0 | 99 | 0.022-0.027 | 0.16-0.22 | 0.8 |
| | | | | | | | | | |
| 2007 | 4-Jul-07 | Peace River | Peace 1 | 1.0 | 58 | 13 | 0.085 | - | - |
| 2007 | 14-Aug-07 | Peace River | Peace 1 | 1.0 | - | - | 0.110 | - | - |
| 2007 | 4-Jul-07 | Peace River | Peace 2 | 0.5 | 48 | 16 | <0.050 | - | - |
| 2007 | 14-Aug-07 | Peace River | Peace 2 | 1.1 | - | - | 0.053 | - | - |
| 2007 | 9-Jul-07 | Tributary | Halfway-9 | 0.7 | 66 | 34 | <0.050 | - | - |
| 2007 | 16-Aug-07 | Tributary | Halfway-9 | 0.8 | - | - | <0.050 | - | - |
| 2007 | 14-Aug-07 | Peace River | Peace 3 | 0.8 | - | - | <0.050 | - | - |
| 2007 | 7-Jul-07 | Tributary | Moberly-6 | 1.4 | 55 | 43 | 0.057 | - | - |
| 2007 | 13-Aug-07 | Tributary | Moberly-6 | 1.5 | - | - | <0.050 | - | - |
| 2007 | 5-Jul-07 | Peace River | Peace 4 | 0.6 | 53 | 22 | <0.050 | - | - |
| 2007 | 14-Aug-07 | Peace River | Peace 4 | 0.6 | - | - | <0.050 | - | - |
| 2007 | 5-Jul-07 | Peace River | Peace 5 | 0.5 | 80 | 19 | <0.050 | - | - |
| 2007 | 15-Aug-07 | Peace River | Peace 5 | 1.5 | - | - | <0.050 | - | - |
| | | | | | | | | | |
| 2010 | 19-Oct-10 | Dinosaur | Mid-reservoir 5m | 1.4 | 10 | 90 | 0.060 | 0.29 | 0.5 |
| 2010 | 19-Oct-10 | Dinosaur | Mid-reservoir 10m | 2.1 | 16 | 84 | 0.172 | 0.13 | 0.1 |
| 2010 | 19-Oct-10 | Dinosaur | Lower-reservoir 5m | 1.7 | 15 | 85 | 0.102 | 0.27 | 0.3 |
| 2010 | 19-Oct-10 | Dinosaur | Lower-reservoir 10m | 1.5 | 6 | 94 | 0.069 | 0.12 | 0.2 |
| 2010 | 19-Oct-10 | Dinosaur | Lower-reservoir 15m | 1.4 | 6 | 94 | 0.074 | 0.17 | 0.2 |
| 2010 | 18-Sep-10 | Peace River | PR-1 | 2.5 | 12 | 88 | 0.061 | 0.57 | 0.9 |
| 2010 | 16-Sep-10 | Tributary | Farrell | 6.1 | 10 | 90 | 0.059 | 2.4 | 4.0 |
| 2010 | 18-Sep-10 | Peace River | PR-2 | 2.1 | 11 | 89 | 0.060 | 1.8 | 3.0 |
| 2010 | 16-Sep-10 | Tributary | Halfway | 2.1 | 11 | 89 | 0.054 | 0.64 | 1.2 |
| 2010 | 18-Sep-10 | Peace River | PR-3 | 0.8 | 69 | 31 | 0.032 | 0.51 | 1.6 |
| 2010 | 15-Sep-10 | Tributary | Moberly | 0.8 | 38 | 62 | 0.049 | 1.8 | 3.6 |

**Notes:** [1] n = # samples when > 1. [2] The mean percentage of 2 or 3 samples is reported for 2000 and 2001 data. **Data Sources:** *Baker 2002* - Williston 2000 and 2001 data. *Golder 2009a* - Peace River and Tributary 2007 data. *Azimuth 2011* - Peace River, Dinosaur and Tributary 2010 data.

Methylmercury concentration in the Peace River technical study area (0.94 to 3.0 µg/kg) was somewhat higher and comprised up to 3% of the total. TOC concentrations in Dinosaur (1.4% to 2.1%) and Peace River (0.8% to 2.5%) were similar and higher than in 2007, because of finer grain size. However, because fine grain particles were targeted, they do not represent the most abundant type of material in the river and these concentrations would be considered worst case or maximum concentrations in the sediment to be conservative. These data were used as input parameters by the RESMERC mercury model (Volume 2, Appendix J, Part 3) as baseline mercury and methylmercury concentrations in existing sediments of the Site C reservoir.

These data indicate that Hg concentrations in the Peace River technical study area including Williston and Dinosaur reservoirs are fairly consistent and do not differ spatially between Williston, Dinosaur and downstream in the Peace River mainstem. Relative to other rivers and lakes in British Columbia, these concentrations would be considered quite low. Rieberger (1992a) sampled sediment from a large number of uncontaminated or pristine BC lakes. In this study mean sediment mercury concentrations from 51 lakes in the central interior plateau (0.17 mg/kg), 25 lakes in the northern Omineca (0.14 mg/kg) and 22 lakes from the central Omineca (0.21 mg/kg), were higher for mercury than for Williston (<0.08 mg/kg) and Peace River (<0.06 mg/kg) sediment.

The 2010 concentrations are also similar to what has been observed in other studies elsewhere in North America. For example, total mercury in sediment from 15 lakes in northern Wisconsin ranged from 0.063 to 0.289 mg/kg dw and MeHg averaged about 1.5% of total mercury (Watras et al. 1998); a reference lake near La Grande in northern Québec (0.036 to 0.059 mg/kg dw; Tremblay et al. 1996); >100 lakes in southern Ontario and Québec (0.003 to 0.267 mg/kg dw; Tremblay et al. 1995) and a large number of lakes from northeastern North America (0.08 to 0.27 mg/kg dw; Kamman et al. 2005). Data from the technical study area confirm that baseline sediment mercury concentrations are low, stable and typical of what would be expected from a remote area.

## 4.6 Zooplankton

### 4.6.1 Background

Food is the major pathway of Hg uptake by fish (Hall et al. 1997). Although zooplankton comprise a small portion of the biomass of food available to Peace River fish, they are present and are fed upon to some degree by small fish (e.g., minnow species, sculpin) and juveniles of many species especially whitefish. For example, annual mean zooplankton production in 2010 was 2.2 g dry wt/m$^2$/yr, dominated by cladocerans and then Calanoid copepods as being the next most important group (Aquatic Productivity Report, Volume 2, Appendix P, Part 1, Biological Assemblages). By contrast, annual benthic invertebrate production was estimated to range between 61 and 112 g dry wt/m$^2$/yr among littoral habitat and lotic reaches in the Peace River, at least 30x higher than for zooplankton. Thus, under baseline conditions benthic invertebrates will dominate dietary exposure of Hg by fish, although this ratio will diminish after reservoir completion.

The Peace River downstream of Peace Canyon Dam does not have a resident zooplankton community, but receives zooplankton passed out of Williston Reservoir, Dinosaur Reservoir and then downstream. Consequently, mercury concentration in zooplankton in the Peace River is a direct reflection of Williston Reservoir zooplankton. However, it is expected that a zooplankton community will establish itself within the proposed Site C reservoir and provide an increasingly important food resource for some fish species, in particular kokanee (Volume 2, Appendix P, Part 3, Future Conditions Report).

Methylmercury concentrations increase at all levels of the food web after flooding to create a reservoir, ultimately culminating with highest concentrations in fish, especially in piscivorous species. Understanding baseline concentrations of inorganic and methylmercury in zooplankton is important because the magnitude of increase in mercury (e.g., 3 to 4 times) above baseline dictates the mercury concentration that a fish will be exposed to via diet. In zooplankton the proportion of total mercury that is comprised of the methylmercury species usually ranges between 30% and 60%.

### 4.6.2 Baseline Mercury Concentration

Zooplankton have been collected and sampled for Hg concentrations from Williston Reservoir in 2000 and 2001 (Baker et al. 2002) and in 2010 from three locations in the Peace River (Azimuth 2011). In Williston Reservoir total Hg in a zooplankton composite ranged from 0.006 – 0.019 mg/kg ww in 2000 and 0.003 – 0.005 mg/kg ww in 2001. These concentrations are in parts per million wet weight and are the sum of both inorganic and MeHg concentration. Methylmercury concentration ranged from 0.002 to 0.004 µg/g ww over both years and on average, comprised 44% of the total Hg concentration.

In the Peace River technical study area between the Peace Canyon Dam and the proposed Site C dam site in 2010, total mercury in zooplankton ranged from 0.004 to 0.009 mg/kg ww (Azimuth 2011) which is quite similar to what was observed in Williston Reservoir 12 years earlier. This is to be expected given that all zooplankton in the Peace River downstream of Williston Reservoir originate upstream. However, these data also indicate that mercury concentrations in zooplankton are stable and have not changed over time. Zooplankton methylmercury concentration in 2010 ranged from 0.0001 to 0.0007 mg/kg ww and averaged only 5% of the total mercury concentration, which is lower than typical. In Dinosaur Reservoir, total mercury in zooplankton samples ranged from 0.001 to 0.006 mg/kg ww and methylmercury ranged from 0.0003 to 0.001 mg/kg ww, averaging 31% of total mercury, which is a more typical result. Because these concentrations are relatively low and near the laboratory detection limit, some inaccuracy is to be expected. Nevertheless, methylmercury in zooplankton comprises about 30% to 40% of the total concentration in the Peace River technical study area.

Relative to other areas, total mercury concentration in zooplankton from the Peace River technical study area is typical of concentrations in many lakes elsewhere in North America. For example, total mercury in zooplankton from 15 lakes in northern Wisconsin ranged from 0.003 to 0.021 mg/kg ww (Watras et al. 1998); from 0.003 to 0.038 mg/kg ww in 24 lakes in southern Ontario and Québec (Tremblay et al. 1995); and 0.0025 to 0.057 mg/kg and 0.0025 to 0.057 from 13 northern Québec lakes (Schetagne et al. 2003). These are remote, uncontaminated lakes and Peace River technical study area concentrations and these concentrations fall within the lower end of this range.

## 4.7 Benthic Invertebrates

### 4.7.1 Background

Benthic invertebrates are a key food chain component of the aquatic food web in the Peace River technical study area and are an important food group for many fish species. Downstream of Williston and Dinosaur reservoirs, benthic invertebrates are particularly important as a dietary source for fish because of the lack of zooplankton in the Peace River, which is common in all large rivers (Wetzel 2001). There is also an important spatial component to food availability in the Peace River downstream of Peace Canyon Dam. Williston Reservoir acts as a barrier to the downstream movement or drift of invertebrates into the Peace River, thus there is a relatively depauperate community in the Peace River technical study immediately downstream of Dinosaur Reservoir. Abundance and diversity of invertebrates increases moving downstream to the Site C dam site location, as the benthic community becomes established. Furthermore, these parameters also increase downstream of major tributary streams (Halfway, Moberly) as these streams contribute large numbers of organisms directly as food and indirectly as colonizers of the river (Volume 2 Appendix P, Biological Assemblages).

Estimated mean annual production of benthic invertebrates within the Peace River ranged between 61 and 112 g dry wt/m/$^2$/yr among littoral habitat and lotic reaches of the Peace

River between Peace Canyon Dam and the proposed Site C dam (Volume 2, Appendix P Part 1, Aquatic Productivity) Relative abundance of different groups differed between Dinosaur and the Peace River mainstem downstream. Dinosaur Reservoir supported higher production of oligochaetes and chironomids (i.e., infauna) and lower production of caddisflies, stoneflies, and mayflies (i.e., epifauna) than the Peace River downstream to the Site C dam site.

Benthic invertebrates in the littoral zone of Dinosaur Reservoir and the Peace River downstream of Peace Canyon Dam to the Site C dam site included all orders of aquatic insects as well as some terrestrial insects (Collembola, planarids, Odonata, Megaloptera, Corixidae, Coleoptera) and non-chironomid Diptera (flies) as well as mites, nematodes, Hydra, gastropods, fingernail clams and oligochaete worms. In the Peace River study area, lotic insects (stoneflies, mayflies, and caddisflies), gastropods, and Corixids (water boatmen) added to the assemblage. Downstream of the Site C dam to Many Islands, the most abundant taxa were the EPT species, caddisflies, stoneflies and mayflies.

Gaining an understanding of mercury and methylmercury concentrations in representative groups from the Peace River technical study area provides perspective on baseline conditions as a benchmark from which mercury concentrations will increase in a new reservoir scenario. Data on mercury in benthos also provides input parameters to the mercury modeling exercise (Volume 2, Appendix J, RESMERC, Part 3) ultimately to predict mercury concentrations in fish, according to the consumption of different dietary groups in the new reservoir (Volume 2, Appendix P, Future Conditions, Part 3) that may differ from current dietary groups (Volume 2, Appendix P, Aquatic Productivity Report, Part 1).

### 4.7.2 Baseline Mercury Concentration

Benthic invertebrates, comprised almost exclusively of chironomid larvae, were collected from Finlay Reach, Williston Reservoir in 2000 and 2001 (Baker et al. 2002) (**Table 4.3**). Total mercury ranged from 0.020 to 0.057 mg/kg ww in 2000 and 0.015 to 0.028 mg/kg ww in 2001, higher concentrations than for zooplankton, which is typical. Methylmercury concentration in 2001 ranged from 0.004 to 0.009 mg/kg ww and averaged 40% of total mercury concentration which is also fairly typical for benthos. These concentrations are lower than what would be found in benthos from a relatively new reservoir (i.e., <15 years) and suggest that baseline conditions persist in Williston Reservoir.

In 2010 and 2011 benthic invertebrates were collected from Dinosaur Reservoir and riffle habitats (**Figure 4.1**) close to the three Peace River stations downstream of Peace Canyon Dam to the proposed Site C dam location (Azimuth 2011; Volume 2, Appendix P, Aquatic

Productivity, Part 1). In summer 2011 discrete taxa or taxonomic groups (e.g., Trichoptera, Chironomid, Ephemeroptera) that are targeted by fish were analysed for inorganic and methylmercury concentration to provide better resolution among dietary choices by fish (**Table 4.3**). Taxonomic composition is important because it influences the magnitude of inorganic and methylmercury concentration within invertebrate tissue that can vary according to life history. Like fish, the more carnivorous invertebrates will have higher total and methyl concentrations than their omnivorous or herbivorous counterparts.

In 2010, the single composite sample had a concentration of 0.025 mg/kg ww, similar to mercury concentrations of benthos observed in Williston Reservoir 12 years earlier. This suggests that mercury methylation rates in the technical study area are low and this is reflected in low methylmercury concentration in lower trophic level biota.

In Peace River between Peace Canyon dam and Moberly River during 2010, mercury in benthos ranged from 0.010 to 0.023 mg/kg ww (Azimuth 2011), with methylmercury concentrations of 0.002 to 0.020 mg/kg ww, which is an average of 20% of total mercury concentration. In 2011, Peace River mainstem benthos had total mercury concentrations of 0.046 to 0.082 mg/kg, with methylmercury concentrations ranging from 20% to 37% of the total (**Table 4.3**). Chironomid larvae (0.06 mg/kg total and <0.04 mg/kg methylmercury) and water boatmen (Corixidae) had slightly higher mercury concentrations (0.05 mg/kg total and 0.04 methyl) and total to methyl ratios. Water boatmen are highly carnivorous and can accumulate methylmercury in similar concentrations as some fish. In the Peace River, chironomids and Corixids comprise a small portion of the diet of fish (Volume 2, Appendix P, Aquatic Productivity, Part 1) and thus their contribution to fish mercury would be comparatively less than the numerically dominant EPT taxa.

The 2010 and 2011 concentrations are comparable to or slightly lower than concentrations observed in other studies elsewhere in Canada. Total mercury in benthos from reference lakes near La Grande in northern Québec ranged from 0.013 to 0.026 mg/kg ww (Tremblay et al. 1996), 0.018 to 0.14 mg/kg in Lake Paijanne southern Finland (Sarkka 1979), and from 0.02 to 0.21 mg/kg ww in Manitoba lakes (Jackson 1988). Although the taxonomic composition may be different, the magnitude of concentration is within a similar range and illustrates the consistently low mercury concentrations across the Peace River technical study area, including the upstream reservoirs Dinosaur and Williston.

**Table 4.3 Mercury and methylmercury in zooplankton and benthic invertebrates, Peace River**

| Date | | | | | ZOOPLANKTON (mg/kg ww) | | | | | BENTHIC INVERTERATES (mg/kg ww) | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Year | Zooplankton | Benthic Inverts | Area | Station ID | % Moisture[1] | Inorganic Mercury | Methyl-mercury | Total[2] Mercury | % MeHg | % Moisture[1] | Inorganic Mercury | Methyl-mercury | Total[2] Mercury | % MeHg |
| 2000 | 14-Aug-00 | 14-Aug-00 | Williston | Junction | | 0.007 | 0.004 | 0.011 | 37 | | | | 0.030 | |
| 2000 | 15-Aug-00 | 15-Aug-00 | Williston | Finlay Profundal | | 0.004 | 0.002 | 0.006 | 28 | | | | 0.020 | |
| 2000 | 16-Aug-00 | | Williston | Finlay Littoral | | 0.007 | 0.002 | 0.009 | 22 | | | | | |
| 2000 | 20-Aug-00 | 20-Aug-00 | Williston | Omineca | | 0.015 | 0.004 | 0.019 | 19 | | | | 0.057 | |
| | | | | | | | | | | | | | | |
| 2001 | 13-Jun-01 | 13-Jun-01 | Williston | Finlay Profundal | | | 0.003 | 0.003 | 100 | | | 0.009 | 0.016 | 57 |
| 2001 | 14-Jun-01 | 14-Jun-01 | Williston | Finlay Littoral | | | 0.003 | 0.005 | 57 | | | 0.006 | 0.017 | 36 |
| 2001 | | 13-Jun-01 | Williston | Davis Bay | | | | | | | | 0.008 | 0.015 | 50 |
| 2001 | | 17-Jun-01 | Williston | Chowika Bay | | | | | | | | 0.004 | 0.028 | 14 |
| | | | | | | | | | | | | | | |
| 2010 | 02-Sep-10 | 02-Sep-10 | Dinosaur | Upper-reservoir | 87.2 | 0.005 | 0.001 | 0.006 | 24 | | 0.023 | 0.002 | 0.025 | 8 |
| 2010 | 02-Sep-10 | 02-Sep-10 | Dinosaur | Mid-reservoir | 92.5 | 0.002 | 0.0008 | 0.003 | 44 | | | Composite | | |
| 2010 | 02-Sep-10 | 02-Sep-10 | Dinosaur | Lower-reservoir | 98.2 | 0.001 | 0.0003 | 0.001 | 26 | | | | | |
| 2010 | 18-Sep-10 | 29-Aug-10 | Peace River | PR-1 | 87.6 | 0.008 | 0.0007 | 0.009 | 9 | 52.6 | 0.012 | 0.004 | 0.016 | 25 |
| 2010 | 18-Sep-10 | 29-Aug-10 | Peace River | PR-2 | 90.4 | 0.004 | 0.0001 | 0.004 | 3 | 80.3 | 0.009 | 0.002 | 0.010 | 15 |
| 2010 | 18-Sep-10 | 28-Aug-10 | Peace River | PR-3 | 88.5 | 0.007 | 0.0001 | 0.007 | 2 | 66.6 | 0.003 | 0.020 | 0.023 | |
| | | | | | | | | | | | | | | |
| 2011 | | 15-Sep-11 | Peace River | PR-1 | | | | | | | | | | |
| | | | | Trichoptera | | | | | | | | 0.004 | 0.012 | 34 |
| 2011 | | 15-Sep-11 | Peace River | PR-3 | | | | | | | | | | |
| | | | | Trichoptera | | | | | | | | 0.005 | 0.016 | 29 |
| | | | | EPT[3] | | | | | | | | 0.003 | 0.009 | 37 |
| 2011 | | 15-Sep-11 | Peace River | PR-14 | | | | | | | | | | |
| | | | | Trichoptera | | | | | | | | 0.003 | 0.017 | 20 |
| | | | | Corixidae | | | | | | | | 0.032 | 0.042 | 76 |
| 2011 | | 15-Sep-11 | Peace River | PR-1, -3, -14 | | | | | | | | | | |
| | | | | EP[3] | | | | | | | | <0.009 | 0.014 | 63 |
| | | | | Chironomidae | | | | | | | | <0.026 | 0.046 | 57 |

**Notes:** [3] EPT = Ephemeroptera, Plecoptera, Trichoptera. **Data Sources:** *Baker et al. 2002* - Williston 2000 and 2001 data. *Azimuth 2011* - Peace River and Dinosaur 2010 data. *Azimuth 2011* - Peace River 2010, 2011 data.

## 4.8 Fish

### 4.8.1 Background

Much research has been conducted into mercury in fish and the relationship between the inundation of terrestrial soils to create new reservoirs and increases in fish mercury concentrations above baseline. It has been known since the 1970s that flooding terrestrial areas to form reservoirs creates conditions that are favourable to increased rates of mercury methylation, bioaccumulation of mercury and increase mercury concentrations at all levels of the food chain, especially in fish (Potter et al. 1975; Abernathy and Cumbie 1977; Bodaly and Hecky 1979; Bodaly et al. 1984, 1987, 2007).

The main influencing factors of fish methylmercury concentrations are concentrations of mercury in prey food consumed, the age and size of the fish, fish growth rates, bioenergetics and reproduction. Because methylmercury accumulated by fish is primarily from dietary sources, body burden concentration is highly dependent on concentrations in their food, and is a reflection of the trophic status of the fish. That is, the higher the position of the fish (and its prey) along the food chain, the higher the mercury concentration in the muscle of fish (Sandheinrich and Wiener 2011). In general, insectivorous fish like rainbow trout have lower mercury concentrations than omnivorous fish (e.g., sucker, whitefish) or highly carnivorous fish such as bull trout and lake trout.

The portion of the Peace River downstream of Peace Canyon Dam to Many Islands has at least 29 species of fish. These include a variety of forage / minnow species (dace, chub, shiners, sculpin), suckers (longnose, largescale, white), mountain and lake whitefish and a large number of sport species that are important to the local communities including bull trout, lake trout, Arctic grayling, rainbow trout, kokanee, walleye and goldeye (AMEC 2008). Within the Peace River there is a gradient of species composition and relative abundance in fish species moving from upstream to downstream into Alberta. According to Mainstream Aquatics (2009, 2010), the Pine River confluence with the Peace River acts as a rough transition point delineating a coldwater fish community upstream from a cool-water community downstream. For example, coldwater species such as rainbow trout and Arctic grayling tend to be found upstream of the Pine River, while cool-water species goldeye, northern pike and walleye generally found downstream of Pine River. Mountain whitefish and longnose sucker are the most common and abundant species found throughout the Peace River in BC (RL&L 1990, 1991, 2001; Mainstream Aquatics 2009, 2010, 2011).

Rainbow trout are also reasonably common throughout the Peace River technical study area but are more common within the area proposed to be inundated by the Site C dam and are found in greatest densities upstream of Maurice Creek. They diminish in abundance moving downstream towards the Alberta border. Adult Arctic grayling are most abundant between the confluences of the Pine and Halfway Rivers (RL&L 1991) and are particularly abundant in these tributary streams, as well as in Farrell Creek where their spawning habitat lies. Bull trout have been captured in all areas of the Peace River, with greatest abundance in tributary streams to the Peace River. Abundance in the Peace River mainstream is relatively low. Bull trout depend on large tributary streams for spawning, nursery and foraging, especially as juveniles, but they do range at the confluence of streams and in the mainstem as adults. They are more commonly found in the Halfway River system and above, where they spawn and rear in the upper tributaries. Most other sport species although present in the Peace River, occur in relatively low numbers and include burbot, lake trout, kokanee, goldeye and walleye and are more common downstream of the Pine River.

The Peace River fish community within the technical study area as far downstream as Many Islands Alberta is dominated by cool-water species. Longnose sucker and mountain whitefish are most abundant, although their abundance decreases with increasing distance downstream, while other sportfish such as goldeye, burbot, and walleye become increasingly common. Coldwater species, such as Arctic grayling, rainbow trout and bull trout, are rarely encountered in the reaches near the BC-Alberta border (AMEC 2008; Mainstream Aquatics 2009, 2010).

For a more complete list of reports regarding population structure, abundance, movements and distribution of fish in the Peace River consult the BC Hydro Site C website at http://www.bchydro.com/energy_in_bc/projects/site_c/document_centre/Environment_Socio_economic_reports.html as well as Volume 2, Appendix O Fish and Fish Habitat Technical Data Report.

### 4.8.2 Baseline Fish Mercury Concentration

Collections of fish tissue for mercury analysis was mainly focused on the dominant species present within the Peace River downstream of Williston Reservoir to the proposed Site C dam site, but also fish species that extend downstream of Site C into Alberta as far as Many Islands (Volume 2, Appendix O Fish and Fish Habitat Technical Report). These species include bull trout, lake trout, Arctic grayling, burbot, kokanee, lake whitefish, mountain whitefish, northern pike, rainbow trout, walleye, goldeye, longnose sucker and redside shiner (**Tables 4.4** and **4.5**).

Much of the information on fish mercury concentrations has been collected to define baseline conditions and to gather sufficient baseline data as input parameters to the RESMERC model for the Site C Project (Volume 2, Appendix J, Part 3). This model is capable of incorporating fish mercury and life history data (length, weight, age, growth) from four different fish species within the range of trophic levels found in this system. Fish mercury collections in 2010 (Azimuth 2011) focused on the following species:

- Longnose sucker (*Catostomus catostomus*) a benthic forager that consumes algae and benthic invertebrates.
- Redside shiner (*Richardsonius balteatus*) a common forage species that feeds on insect larvae and zooplankton and is consumed by other fish species.
- Mountain whitefish (*Prosopium williamsoni*), a common benthic feeder that is an important intermediate species in the food web.
- Rainbow trout (*Oncorhynchus mykiss*) a common insectivorous species that is an important fish species targeted by local sport fishers.
- Bull trout (*Salvelinus confluentus*) an important piscivorous species that is at the top of the food web and also targeted by humans for food.

Note that these species were also sampled within Dinosaur Reservoir to gain a spatial perspective on fish mercury concentrations in this system.

In 2011 the fish mercury collection program was expanded to include other important sport species including goldeye and walleye that were more abundant downstream of Site C to address potential bioaccumulation of mercury by fish downstream of the Site C reservoir within the technical study area. These data provide a more complete understanding of mercury data in fish within the Peace River downstream of Dinosaur extending as far as Many Islands, defined as the boundary of the study area based on abundance and species composition of the fish community and migration / movement patterns between Many Islands and upstream to the proposed dam site (Mainstream Aquatics 2009, 2010; Volume 2, Appendix O).

Fish mercury data from the Peace River including Williston and Dinosaur Reservoir have been divided into historic (pre-2001) and current (2008 to 2011) data in **Tables 4.4** and **4.5**, respectively. During the late 1980s, there were dedicated surveys to understand fish mercury concentrations a number of BC Hydro reservoirs including Dinosaur, Arrow, Kinbasket and Revelstoke (among others) and these data are summarized in **Table 4.6**. The earliest mercury data for the Peace River downstream of Peace Canyon Dam were collected by Pattenden et al. (1990; 1991) while with RL&L in 1989 and 1990. Greater detail regarding mercury in fish in these and other reservoirs can be found in Baker (2002) as part of a fish mercury database for British Columbia commissioned by BC Hydro.

Finally, a comprehensive study of mercury in Finlay Reach, Williston Reservoir was undertaken in 2000 by Baker et al. (2002) that focused on mercury in lake whitefish and bull trout. These data are useful to compare historic and current fish mercury data from the Peace River system (Williston, Dinosaur, Peace River) with other lakes and reservoirs in BC in order to put Peace River fish mercury within the Site C technical study area in perspective.

#### 4.8.2.1 Historic Data

The only data sets for which there is a reasonably large sample size (i.e., >10 fish) for mercury concentrations across a representative size range are for bull trout, rainbow trout and lake whitefish from Williston Reservoir (1980, 1988 and 2000, respectively) and lake whitefish (1988/89), mountain whitefish (1988/89), northern pike (1989), and rainbow trout (1989) from the Peace River. Few data exist for burbot, kokanee and walleye from the Peace River, because they are rare and captured relatively infrequently in the Peace River mainstem within the Site C technical study area (AMEC 2008).

**Table 4.4 Historic (pre-2001) Peace River fish mercury concentrations**

| Species | Area | Year | Sample Size | Length (mm) Range | Length (mm) Mean | Weight (g) Range | Weight (g) Mean | Hg (mg/kg ww) Range | Hg (mg/kg ww) Mean | Reference[1] |
|---|---|---|---|---|---|---|---|---|---|---|
| **Arctic Grayling** | | | | | | | | | | |
| | Peace River | 1989 | 2 | 361 - 375 | 368 | 682 - 862 | 772 | 0.03 - 0.04 | 0.03 | Pattenden et al. 1990 |
| **Bull trout** | | | | | | | | | | |
| | Williston - Finlay Reach | 2000 | 46 | 243 - 744 | 457 | 150 - 3750 | 1165 | 0.08 - 2.22 | 0.46 | EVS 2002 |
| | Peace River | 1987 | 1 | - | 545 | - | 1590 | - | 0.07 | Pers. comm.[2] |
| | Peace River | 1989 | 8 | 330 - 814 | 500 | 414 - 5700 | 1904 | 0.04 - 0.83 | 0.21 | Pattenden et al. 1990 |
| | Peace River | 1990 | 5 | 342 - 530 | 431 | 408 - 1918 | 985 | 0.01 - 0.10 | 0.06 | Pattenden et al. 1991 |
| **Burbot** | | | | | | | | | | |
| | Peace River | 1989 | 2 | 370 - 435 | 403 | 342 - 542 | 442 | 0.08 - 0.14 | 0.11 | Pattenden et al. 1990 |
| **Kokanee** | | | | | | | | | | |
| | Peace River | 1989 | 2 | 282 - 315 | 299 | 268 - 376 | 322 | 0.03 - 0.038 | 0.04 | Pattenden et al. 1990 |
| **Lake Whitefish** | | | | | | | | | | |
| | Williston - Akie | 1980 | 14 | 360 - 460 | 405 | 200 - 1000 | 511 | 0.12 - 0.37 | 0.18 | Health and Welfare Canada 1980 |
| | Williston - Ingenika | 1980 | 16 | 360 - 520 | 407 | 140 - 1600 | 437 | 0.10 - 0.38 | 0.20 | Health and Welfare Canada 1980 |
| | Williston - Parsnip Reach | 1988 | 23 | 187 - 352 | 289 | 70 - 470 | 260 | 0.05 - 0.38 | 0.19 | Baker 2002 |
| | Williston - Peace Reach | 1988 | 33 | 160 - 390 | 314 | 30 - 660 | 331 | 0.07 - 0.43 | 0.18 | Baker 2002 |
| | Williston - Finlay Reach | 1988 | 22 | 180 - 345 | 301 | 40 - 380 | 280 | 0.07 - 0.40 | 0.23 | Baker 2002 |
| | Williston - Finlay Reach | 2000 | 23 | 148 - 308 | 238 | 25 - 375 | 177 | 0.03 - 0.24 | 0.12 | EVS 2002 |
| | Dinosaur Reservoir | 1988 | 25 | 289 - 395 | 343 | 280 - 925 | 494 | 0.03 - 0.16 | 0.10 | Baker 2002 |
| | Peace River | 1987 | 1 | - | 500 | - | 1280 | - | <0.05 | Pers. comm.[2] |
| | Peace River | 1988 | 20 | 261 - 432 | 338 | 210 - 1200 | 479 | 0.05 - 0.17 | 0.09 | Baker 2002 |
| | Peace River | 1989 | 11 | 320 - 513 | 373 | 380 - 2300 | 721 | 0.04 - 0.13 | 0.07 | Pattenden et al. 1990 |
| **Mountain whitefish** | | | | | | | | | | |
| | Dinosaur Reservoir | 1988 | 6 | 286 - 349 | 316 | 290 - 515 | 372 | 0.03 - 0.14 | 0.07 | Baker 2002 |
| | Peace River | 1989 | 24 | 272 - 487 | 367 | 222 - 1289 | 676 | 0.02 - 0.12 | 0.06 | Pattenden et al. 1990 |
| | Peace River | 1990 | 30 | 247 - 635 | 402 | 178 - 2259 | 682 | <0.001 - 0.08 | 0.03 | Pattenden et al. 1991 |
| **Northern Pike** | | | | | | | | | | |
| | Peace River | 1989 | 21 | 310 - 790 | 547 | 210 - 3650 | 1411 | 0.03 - 0.25 | 0.09 | Pattenden et al. 1990 |
| **Rainbow Trout** | | | | | | | | | | |
| | Williston - Ingenika | 1979 | 14 | 360 - 420 | 398 | 100 - 750 | 361 | 0.05 - 0.35 | 0.19 | Baker 2002 |
| | Williston - Finlay Reach | 1988 | 12 | 275 - 405 | 332 | 250 - 680 | 413 | 0.03 - 0.35 | 0.08 | Baker 2002 |
| | Williston - Parsnip Reach | 1988 | 22 | 225 - 386 | 305 | 120 - 740 | 345 | 0.02 - 0.08 | 0.04 | Baker 2002 |
| | Williston - Peace Reach | 1988 | 16 | 224 - 360 | 302 | 120 - 480 | 306 | 0.02 - 0.09 | 0.05 | Baker 2002 |
| | Dinosaur Reservoir | 1988 | 4 | 322 - 502 | 400 | 380 - 1450 | 835 | 0.07 - 0.46 | 0.18 | Baker 2002 |
| | Peace River | 1988 | 3 | 247 - 343 | 310 | 200 - 485 | 388 | 0.03 - 0.09 | 0.05 | Baker 2002 |
| | Peace River | 1989 | 23 | 266 - 425 | 325 | 194 - 878 | 458 | 0.01 - 0.09 | 0.03 | Pattenden et al. 1990 |
| **Walleye** | | | | | | | | | | |
| | Peace River | 1987 | 4 | 400 - 580 | 457 | 680 - 2000 | 1058 | 0.07 - 0.32 | 0.17 | Pers. comm.[2] |
| | Peace River | 1989 | 8 | 343 - 415 | 389 | 484 - 936 | 723 | 0.01 - 0.24 | 0.15 | Pattenden et al. 1990 |

**Notes:**
[1] Baker 2002 reference contains original, unpublished BC Hydro data for individual fish from cited reservoirs.
[2] Personal communication with Bruce Carmichael, MOE, Prince George, Feb.2012; concentrations are presumed to be wet weight.

Table 4.5 Current (2008 - 2011) Peace River fish mercury concentrations

| Species | Area | Year[1] | Sample Size | Length (mm) Range | Length (mm) Mean | Weight (g) Range | Weight (g) Mean | Hg (mg/kg ww) Range | Hg (mg/kg ww) Mean | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| **Bull trout** | | | | | | | | | | |
| | Peace River - Site C | 2008 | 21 | 248 - 741 | 484 | 166 - 5450 | 1684 | 0.042 - 0.14 | 0.08 | Mainstream 2009a |
| | Peace River - Downstream | 2008 | 4 | 211 - 544 | 336 | 100 - 1798 | 618 | 0.018 - 0.12 | 0.07 | Mainstream 2009a |
| | Dinosaur Reservoir | 2010/2011 | 6 | 285 - 811 | 476 | 262 - 7775 | 2519 | 0.038 - 0.34 | 0.12 | Azimuth 2011, 2012 |
| | Peace River - Site C | 2010/2011 | 19 | 292 - 806 | 470 | 308 - 7160 | 1635 | 0.031 - 0.34 | 0.07 | Azimuth 2011, 2012 |
| | Peace River - Downstream | 2011 | 2 | 500 - 558 | 529 | 1350 - 1822 | 1586 | 0.077 - 0.09 | 0.08 | Azimuth 2012 |
| **Burbot** | | | | | | | | | | |
| | Peace River - Dunvegan | 2008 | 43 | 274 - 790 | 474 | 132 - 2550 | 753 | 0.018 - 0.14 | 0.06 | Mainstream 2009b |
| **Goldeye** | | | | | | | | | | |
| | Peace River - Downstream | 2010/2011 | 10 | 310 - 410 | 379 | 314 - 854 | 600 | 0.136 - 0.31 | 0.24 | Azimuth 2012 |
| **Lake trout** | | | | | | | | | | |
| | Dinosaur Reservoir | 2010/2011 | 28 | 304 - 630 | 414 | 262 - 2676 | 865 | 0.029 - 0.14 | 0.09 | Azimuth 2011, 2012 |
| | Peace River - Site C | 2010 | 1 | - | 391 | - | 570 | - | 0.07 | Azimuth 2011 |
| **Longnose sucker** | | | | | | | | | | |
| | Dinosaur Reservoir | 2010/2011 | 12 | 268 - 434 | 393 | 240 - 1074 | 755 | 0.063 - 0.36 | 0.20 | Azimuth 2011, 2012 |
| | Peace River - Site C | 2010/2011 | 31 | 295 - 442 | 388 | 362 - 1172 | 770 | 0.017 - 0.17 | 0.05 | Azimuth 2011, 2012 |
| | Peace River - Downstream | 2011 | 10 | 373 - 442 | 403 | 654 - 990 | 779 | 0.019 - 0.10 | 0.06 | Azimuth 2012 |
| **Mountain whitefish** | | | | | | | | | | |
| | Peace River - Site C | 2008 | 30 | 209 - 466 | 340 | 94 - 1180 | 483 | 0.018 - 0.09 | 0.04 | Mainstream 2009a |
| | Peace River - Downstream | 2008 | 31 | 202 - 512 | 355 | 74 - 1526 | 570 | 0.014 - 0.09 | 0.04 | Mainstream 2009a |
| | Dinosaur Reservoir | 2010/2011 | 21 | 246 - 395 | 317 | 192 - 692 | 364 | 0.022 - 0.07 | 0.05 | Azimuth 2011, 2012 |
| | Peace River - Site C | 2010/2011 | 39 | 211 - 480 | 345 | 108 - 1252 | 498 | 0.010 - 0.17 | 0.04 | Azimuth 2011, 2012 |
| | Peace River - Downstream | 2010/2011 | 10 | 237 - 396 | 319 | 158 - 622 | 366 | 0.016 - 0.07 | 0.04 | Azimuth 2011, 2012 |
| **Rainbow Trout** | | | | | | | | | | |
| | Dinosaur Reservoir | 2010/2011 | 10 | 265 - 313 | 292 | 178 - 286 | 242 | 0.036 - 0.06 | 0.05 | Azimuth 2011, 2012 |
| | Peace River - Site C | 2011 | 10 | 215 - 440 | 330 | 128 - 984 | 433 | 0.022 - 0.09 | 0.04 | Azimuth 2012 |
| **Redside Shiner** | | | | | | | | | | |
| | Peace River - Downstream | 2011 | 11 | 85 - 119 | 99 | 6 - 26 | 14 | 0.034 - 0.07 | 0.05 | Azimuth 2011, 2012 |
| **Walleye** | | | | | | | | | | |
| | Peace River - Downstream | 2011 | 16 | 399 - 479 | 431 | 630 - 1204 | 885 | 0.085 - 0.33 | 0.18 | Azimuth 2012 |

**Notes:**

[1] Where captured in the same area, fish data from 2010 and 2011 were combined and are summarized together.

**Table 4.6 Fish mercury concentrations in select BC Hydro reservoirs and lakes**

| Species | Area | Year | Sample Size | Length (mm) | | Sample Size | Weight (g) | | Sample Size | Hg (mg/kg ww) | | Reference[1] |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Range | Mean | | Range | Mean | | Range | Mean | |
| **Bull trout** | | | | | | | | | | | | |
| | Arrow Reservoir | 1987 | 23 | 410 - 790 | 628 | 23 | 740 - 7000 | 3163 | 23 | 0.14 - 1.40 | 0.43 | Baker 2002 |
| | Arrow Reservoir | 1995 | 16 | 430 - 760 | 588 | 16 | 800 - 5300 | 2488 | 16 | 0.10 - 0.28 | 0.17 | Foster and Gadbois 1998 |
| | Kinbaset Reservoir | 1987 | 7 | 285 - 530 | 362 | 7 | 200 - 640 | 381 | 7 | 0.23 - 0.92 | 0.41 | Baker 2002 |
| | Kinbaset Reservoir | 1995 | 11 | 580 - 860 | 736 | 11 | 2000 - 7300 | 5509 | 11 | 0.23 - 0.41 | 0.39 | Foster and Gadbois 1998 |
| | Revelstoke Reservoir | 1987 | 25 | 260 - 565 | 365 | 25 | 160 - 2025 | 572 | 25 | 0.14 - 0.82 | 0.41 | Baker 2002 |
| | Revelstoke Reservoir | 1995 | 17 | 510 - 890 | 670 | 17 | 1400 - 10300 | 4282 | 17 | 0.12 - 0.64 | 0.30 | Foster and Gadbois 1998 |
| **Lake trout** | | | | | | | | | | | | |
| | Babine Lake | 1979 | 28 | 480 - 710 | 589 | 28 | 500 - 4200 | 1991 | 28 | 0.10 - 0.50 | 0.25 | Baker 2002 |
| | Stuart Lake | 2000 | 21 | 351 - 829 | 566 | 21 | 500 - 6050 | 2271 | 21 | 0.10 - 1.0 | 0.31 | Baker 2001b |
| | Trembleur Lake | 2000 | 13 | 498 - 765 | 621 | 13 | 1325 - 6000 | 2927 | 13 | 0.11 - 0.72 | 0.32 | Baker 2001b |
| **Lake Whitefish** | | | | | | | | | | | | |
| | Stuart Lake | 2000 | 31 | 161 - 515 | 312 | 31 | 50 - 1450 | 454 | 31 | 0.04 - 0.22 | 0.09 | Baker 2001b |
| | Trembleur Lake | 2000 | 31 | 122 - 450 | 255 | 31 | 25 - 1175 | 286 | 31 | 0.02 - 0.26 | 0.08 | Baker 2001b |
| **Mountain whitefish** | | | | | | | | | | | | |
| | Carpenter Reservoir | 2000 | 11 | 182 - 275 | 228 | 11 | 75 - 275 | 145 | 11 | 0.09 - 0.19 | 0.13 | Baker 2001a |
| **Rainbow Trout** | | | | | | | | | | | | |
| | Arrow Reservoir | 1986 | 13 | 335 - 650 | 442 | 13 | 410 - 4200 | 1187 | 13 | 0.07 - 0.31 | 0.14 | Baker 2002 |
| | Kinbaset Reservoir | 1985-1987 | 13 | 310 - 440 | 395 | 13 | 390 - 830 | 715 | 13 | 0.05 - 0.27 | 0.14 | Baker 2002 |
| | Revelstoke Reservoir | 1987 | 11 | 270 - 500 | 406 | 11 | 270 - 1100 | 754 | 11 | 0.12 - 0.57 | 0.23 | Baker 2002 |

**Notes:** [1] Baker 2002 reference contains original, unpublished BC Hydro data for individual fish from cited reservoirs.

Following is a brief summary of historical mercury concentration data for key species in Williston Reservoir and the Peace River technical study area. Note that all data are presented in terms of mg/kg ww (i.e., parts per million wet weight) and are not adjusted for different fish sizes. That is, larger fish usually have higher mercury concentrations and differences in mercury concentration may partly be due to differences in fish size.

- **Bull Trout** – Mean mercury in bull trout from Finlay Reach, Williston Reservoir averaged 0.46 mg/kg ww at a mean fish size of 457 mm (Baker et al. 2002). Although data for bull trout from Peace River are few, mean mercury concentrations measured between 1987 and 1990 ranged from 0.07 to 0.21 mg/kg (**Table 4.4**).
- **Lake Whitefish** – Mean mercury in lake whitefish from Williston Reservoir (all reaches) ranged from 0.12 to 0.23 in 1988 (Baker et al. 2002), similar to what was observed in 1980 by Health and Welfare Canada (1980). Mean mercury concentration was lower in the Peace River in 1988 (0.09 mg/kg) and 1989 (0.07 mg/kg).
- **Rainbow Trout** – Mean mercury in rainbow trout from Williston Reservoir (1988) ranged from 0.04 to 0.08 mg/kg (Baker et al. 2002). In the Peace River in 1989, mean mercury was 0.03 mg/kg (Pattenden et al. 1990). Mean fish size did not differ markedly among the studies, averaging just over 300 mm.
- **Northern Pike** – Mercury was measured in northern pike from the lower Peace River in 1989 by Pattenden et al. (1990) with a mean concentration of 0.09 mg/kg (547 mm).
- **Other Species** – Mercury concentrations were measured from small numbers of Arctic grayling from the Peace River in 1989 (0.03 mg/kg), burbot (0.11 mg/kg), kokanee (0.03 mg/kg) and walleye (0.15 mg/kg) (**Table 4.4**).

In general, historic mercury concentrations in most fish species from the Peace River within British Columbia were less than 0.10 mg/kg, except for bull trout, which ranged up to 0.2 mg/kg. Mercury concentrations in Dinosaur Reservoir whitefish were slightly higher 0.03 – 0.17 mg/kg, while rainbow trout was low (0.04 mg/kg). Mercury concentrations in Williston Reservoir mountain whitefish (0.03 – 0.43 mg/kg), rainbow trout (0.03 – 0.35 mg/kg) and bull trout (0.03 – 2.2) were higher than in Dinosaur Reservoir or from the Peace River upstream of the Site C dam.

#### 4.8.2.2 Current Data

Targeted mercury studies were undertaken beginning in 2008 (Mainstream Aquatics 2009) to augment and update the fish tissue database for the Peace River as part of Site C planning. This was continued in 2010 as part of a comprehensive mercury study of Peace

River and Dinosaur Reservoir (Azimuth 2011) and supplementary collections of fish in 2011 to gather baseline data on other key species such as rainbow trout, walleye and goldeye, especially downstream of the Site C dam area extending to Many Islands, the downstream boundary of the technical study area. This was to ensure that the geographic scope of work extended far enough downstream to capture fish species that are rarely found within the Peace River upstream of the dam site, but might move upstream far enough to feed on fish passed out of the Site C reservoir.

- **Bull Trout** – Mean mercury concentrations in bull trout from Peace River technical study area ranged between 0.07 and 0.08 mg/kg ww within (**Table 4.5**) and were slightly higher in Dinosaur (0.12 mg/kg). These concentrations are quite low for a large piscivorous species and there was a positive correlation between increasing fish size and mercury (Azimuth 2011).
- **Mountain Whitefish** – Mean mercury concentration in mountain whitefish from Peace River study area was low and fell within a very narrow range (0.03 to 0.04) in 2008 and 2010/2011. The mean concentration in Dinosaur Reservoir also averaged 0.04 mg/kg, similar to the Peace River downstream study area.
- **Rainbow Trout** – Mercury concentration in rainbow trout in 2011 from Peace River study area and Dinosaur Reservoir averaged 0.04 mg/kg (Azimuth 2012) in both areas and did not vary with differences in fish size.
- **Longnose Sucker** – Mean mercury concentration in longnose sucker from Peace River upstream of the proposed Site C dam site (0.05 mg/kg) and downstream into Alberta (0.06 mg/kg) were low and similar. The mean concentration in Dinosaur in 2010 was higher (0.20 mg/kg) because we believe this species has shifted to a diet of fish based on stable isotope signatures (Azimuth 2011).
- **Lake Trout** – Mean mercury concentration of lake trout from Dinosaur Reservoir in 2010 was 0.09 mg/kg (414 mm) while a single fish captured from the Peace River downstream of the proposed Site C dam site in 2011 was 0.07 mg/kg (391 mm). This concentration is relatively low for piscivorous species such as lake trout.
- **Other Species** – Mean mercury concentration in fish downstream of the proposed Site C dam as far as Many Islands included goldeye (0.24 mg/kg) and walleye (0.18 mg/kg). Both of these species are piscivorous and had higher mercury concentrations than fish with a similar dietary preference (bull trout, lake trout) in the upstream reach below Peace Canyon Dam. Redside shiner, a forage species was captured downstream of the Site C reservoir in 2010 and had a mean mercury concentration of 0.05 mg/kg.

Mean mercury concentrations of all fish species in the Peace River within the technical study area of the Site C Project were less than 0.08 mg/kg with nearly all fish less than 0.20 mg/kg. Mercury concentration in forage (redside shiner) and omnivorous species (rainbow trout, whitefish) were similar to bull trout, a large piscivorous species. These data suggest that the Peace River downstream of Peace Canyon dam at least as far as the proposed Site C dam site is a poor methylating environment and that the rate of bioaccumulation and biomagnification of methylmercury by aquatic biota and fish is very low.

The mercury concentrations observed from historic and current studies, especially for large piscivorous species such as bull trout within the Peace River technical study area, including Dinosaur Reservoir are low, lower even than mercury concentrations for similar size fish from other BC lakes and reservoirs (**Table 4.6**). For example, in 1995, mercury in bull trout was 0.17 mg/kg from Arrow Reservoir, 0.30 mg/kg in Revelstoke Reservoir, 0.34 mg/kg in Kinbasket Reservoir (Foster and Gadbois 1998) and 0.36 mg/kg in Carpenter Reservoir (2008; Azimuth 2009). Measured tissue mercury concentrations (all means) from rainbow trout (0.09 mg/kg), lake trout (0.26 mg/kg), Arctic grayling (0.08 mg/kg) and mountain whitefish (0.11 mg/kg) from more than 100 BC lakes (Rieberger 1992b) were higher than for all fish species in the Peace River technical study area.

# 5 CANADIAN RESERVOIRS COMPARISON MATRIX

The aim of this chapter is to summarize the main physical and chemical features, ecological parameters and water / biota mercury concentrations of a wide variety of Canadian reservoirs to relate the importance of these factors to the observed changes in post-impoundment mercury concentrations in fish from new reservoirs. Given the relatively old age of most reservoirs, there were very few if any pre-impoundment lower trophic level and fish mercury concentrations. Until recently, most laboratories did not have the capability of measuring low level mercury concentrations and there is a paucity of accurate data for mercury concentrations in water during the time period when most large hydroelectric facilities in Canada were being constructed (mid1960's to 1990's). Despite these data limitations, the specific baseline physical, chemical and biological conditions of the Peace River technical study area and what is predicted for the proposed Site C reservoir (e.g., Volume 2, Appendices D, Surface Water; H Temperature; O Fish and Fish Habitat; P Future Conditions) can be compared to conditions observed in other Canadian reservoirs. In this way, where the Site C Project fits within the spectrum of reservoir types can be better determined – as it relates to mercury magnification in environmental media. This exercise provides the context and a 'weight of evidence' approach that supports predictions of future fish mercury concentrations following construction and operation of the Site C reservoir.

## 5.1 Introduction

Several physical, chemical, and ecological parameters are known to be key determinants in mercury bioavailability and the rates of mercury methylation/demethylation and biomagnification within the food web. The most important of these are reservoir residence time, pH, the amount, structure and chemical composition of the newly flooded soil and vegetation, and invertebrate and fish community structure (particularly the number of trophic levels) (Ulrich et al. 2001; Schetagne et al. 2003; Bodaly et al. 2007; and many others). Reservoir-specific differences in these factors are likely responsible for the substantial variability in the number of years for fish to reach peak mercury concentrations, the magnitude of those peaks, and the return time to pre-flooding or background that has been observed among reservoirs (Bodaly et al. 2007; Schetagne et al. 2003).

For example, the quantity and quality of the inundated organic material greatly affects methylmercury production. Results from experimental reservoirs at the Experimental Lakes Area (ELA) have shown that flooded wetlands/peatlands have high methylmercury production rates and may sustain production for long periods of time (>8 years; Kelly et al. 1997; St. Louis et al. 2004). Each of the key physical, chemical and ecological factors that contribute to increased rates of mercury methylation will be explored in this chapter as they ultimately relate to mercury concentrations in fish that are of importance to humans and to wildlife.

## 5.2 Key Matrix Parameters

The first step in establishing the Canadian reservoirs comparison matrix was to compile a list of the key physical and biological parameters known to affect mercury methylation (**Table 5.1**). This list also served as a guideline for the collection of pre-impoundment field data in the area of the proposed Site C location (Azimuth 2011) to ensure that no data gaps were left unfilled.

Once the parameter list was established, reservoirs were selected for which information was potentially available on most of the selected parameters. We limited our search to large Canadian hydroelectric generation stations (GS). Most of these are located in the same climate zone as the proposed Site C Project, the associated reservoirs can be expected to provide habitat for some of the same fish species, and unpublished information on desired reservoir parameters may be obtained with reasonable effort. Because Site C is a run-of-the-river project, care was taken to include similar generating stations but also storage facilities with forebays that represent more lake-like conditions.

A list of reservoirs initially considered for an evaluation of parameter information is provided in **Table 5.2**.

**Table 5.1 Physical, chemical, and biological parameters important in determining mercury concentrations at Site C**

| Parameter | Unit | Parameter | Unit |
|---|---|---|---|
| **Physical and chemical parameters** | | | |
| Year of impoundment | Year | pH | pH units |
| Original area | $km^2$ | DOC or TOC | mg/L |
| Reservoir area | $km^2$ | TSS or Turbidity | mg/L or units |
| Ratio: reservoir/original area | n.a. | TDP | µg/L |
| Water residence time | Months/days | $NO_3$ | mg/L |
| Maximum water temperature | º C | Total Hg, water | ng/L |
| Minimum oxygen concentration | % saturation | Methyl Hg, water | ng/L |
| Thermal stratification | yes/no | Total Hg, sediment | µg/kg dry wt |
| Water depth, maximum/mean | Meters | Total Hg, flooded areas | µg/kg dry wt |
| Forebay clearance | % area | Hg pool | $mg/m^2$ |
| Presence of wetlands/peatlands | % area | Carbon pool | $kg/m^2$ |
| **Ecological parameters** | | | |
| Total Hg, zooplankton | µg/g ww | Total Hg*, Bull trout | µg/g ww |
| Methyl Hg, zooplankton | µg/g ww | Total Hg*, Lake trout | µg/g ww |
| Total Hg, benthos | µg/g ww | Total Hg*, other fish species | µg/g ww |
| Methyl Hg, benthos | µg/g ww | Ratio of max Hg/ baseline Hg | Unit |
| Total Hg*, forage fish species | µg/g ww | Return time to baseline Hg | Years |
| Total Hg*, Lake whitefish | µg/g ww | | |

**Note**: Physical and chemical parameters reflect pre-impoundment conditions, except for residence time; temperature, oxygen, stratification, and depth represent post-impoundment conditions. All biological parameters include both baseline information and data for post-impoundment conditions.
ww = wet weight; µg/g = mg/kg or parts per million (ppm); dw (dry weight) is sometimes used.
* maximum mercury concentrations, after reservoir creation.

**Table 5.2 List of Canadian hydroelectric reservoirs evaluated against Site C**

| Reservoir | River | Province | Operator | Year of impoundment |
|---|---|---|---|---|
| Williston | Peace | B.C. | BC Hydro | 1967-1972 |
| Keeyask | Nelson | Manitoba | Manitoba Hydro | 2021 (proposed) |
| Limestone | Nelson | Manitoba | Manitoba Hydro | 1990 |
| Longspruce | Nelson | Manitoba | Manitoba Hydro | 1977 |
| Notigi | Rat | Manitoba | Manitoba Hydro | 1974 |
| Southern Indian Lake | Churchill | Manitoba | Manitoba Hydro | 1976 |
| Stephens | Nelson | Manitoba | Manitoba Hydro | 1970 |
| Wuskwatim | Burntwood | Manitoba | Manitoba Hydro | 1977*; 2012** |
| Caniapiscau | Caniapiscau | Québec | Hydro Québec | 1981-1984 |
| LG 1 | La Grande | Québec | Hydro Québec | 1993 |
| LG 2 (Robert-Bourassa) | La Grande | Québec | Hydro Québec | 1978-1979 |
| LG 3 | La Grande | Québec | Hydro Québec | 1981-1984 |
| Opinaca | Eastmain | Québec | Hydro Québec | 1980 |
| Gull Island | Lower Churchill | Nfld/Lab | N&L Hydro | 2014 (proposed) |
| Muskrat Falls | Lower Churchill | Nfld/Lab | N&L Hydro | 2017 (proposed) |

* Wuskwatim Lake was previously flooded in 1977 as part of the Churchill River Diversion. ** Under construction

During the initial evaluation process it became apparent that information on several of the selected parameters, particularly on mercury concentrations in water, sediment, and lower trophic level biota did not exist because of analytical limitations. This situation mainly existed because the vast majority of existing Canadian hydroelectric generating stations are relatively old (>20 years), which is especially true in British Columbia and is the main reason why no other BC reservoirs were selected for comparison except Williston; there are too few historic data. In BC, the majority of large reservoirs, some with similar physical features as Site C (e.g., Revelstoke, Kinbasket, Arrow lakes), were constructed between 1950 and 1980, and prior to our understanding of the relationship between reservoir creation and increases in mercury in environmental media. Also, this was well before legislation required environmental assessments of large hydro projects, so there are few if any empirical data collected either before or after construction (except perhaps Williston Reservoir under the Peace/Williston Fish and Wildlife compensation program; post-impoundment change only) and especially related to mercury.

Recently, however, there has been a renewed interest by Canadian power utilities in the construction of new hydroelectric stations such as the proposed Site C Clean Energy

Project (BC), Wuskwatim, Keeyask and Conawapa (Manitoba) and Muskrat Falls (Newfoundland and Labrador). Some of these stations that were either proposed or under construction were included in this evaluation, to increase the number of examples with a more complete set of parameter information, at least for pre-impoundment conditions. Including facilities in the planning stage had the disadvantage of missing post-impoundment data, particularly on fish mercury concentrations, although for all reservoirs considered in this category estimates of predicted (modeled) post-project fish mercury levels existed. For the final list of reservoirs included in the comparison table (**Table 5.3**), several of the initially considered reservoirs were excluded primarily because of an inadequate data set for the evaluation parameters. To increase database size, in addition to full-sized reservoirs, two experimental reservoirs for the final evaluation were included, because a great deal of information was available (**Table 5.3**). These were an artificially flooded wetland and an upland reservoir at the Experimental Lakes Area, Ontario. These artificial reservoirs experimentally studied the effects of impoundment on mercury methylation and bioaccumulation up to the trophic level of aquatic macroinvertebrates and forage fish (Kelly et al. 1998; St. Louis et al. 2004; Bodaly et al. 2004).

**Table 5.3: Physical, chemical and biological parameters of Canadian reservoirs relative to the proposed Site C reservoir**

| | Chemical Parameters | | | | | | | | | | | | Physical Parameters | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | Total Hg | | | | | Methyl Hg | | | | | | | | | | | | | |
| Development Project | pH | DOC/TOC (mg/L) | TSS (mg/L); NTU* | Nutrients | Water (ng/L) | Sediment (µg/g d.w.**) | Upland or Flooded Soil (µg/g d.w.) | Mercury Pool (THg mg/m2) | Carbon Pool (kg/m2) | Water (ng/L) | Sediment (ng/g d.w) | Upland or Flooded Soil (µg/kg d.w.) | Year Impounded | Original Area (km2) | Reservoir Area (km2) | Area ratio Reservoir : Original | Mean Residence Time | Max Temp (°C) | Min Oxygen (%/mg/L) | Stratified | Max / Mean Depth (m) | % Cleared | % Wetland / peatland |
| ***A) Full-sized reservoirs*** | | | | | | | | | | | | | | | | | | | | | | | |
| | | | | | Pre- construction | | | | | Pre- construction | | | | | | | Post-construction data | | | | | Area to be flooded | |
| **Muskrat Falls GS (proposed), Lower Churchill River, Newfoundland and Labrador** | ~7.0 | 2.6 - 4.6 | Up to ~45 mg/L | TP: ~10-70 ug/L | ~0.8-1.2 for 5 sites incl 2 ref lakes | <0.01 (DL)-0.14 | 70 soil samples taken for chemical and physical analyses from 22 sites within the potential flood zone area. 53 samples from upland mineral soils: <0.01 - 0.04 mg/kg; 11 samples from upland forest floor horizons: 0.04 - 0.21 mg/kg; 6 samples from wetland organic soils: 0.04 - 0.10 mg/kg | No data | Used General literature estimate: Total carbon: 2.72 in upper 11 cm, labile carbon: 0.25 in upper 11 cm | <0.041 (i.e., DL) | No data | No data | 2017 (proposed) | 71 | 107 | 1.51 | 7 days | 17 °C (monthly mean) | unknown | No | ~36 / ? | 43% of flood zone proposed (under review) | 6% |
| **Gull Island GS (proposed), Lower Churchill River Newfoundland and Labrador** | ~7.0 | 2.6 - 4.6 | <5 mg/L | TP: 6-19 ug/L | ~0.8-1.2 for 5 sites incl 2 ref lakes | <0.01 (DL)-0.14 | 70 soil samples were taken for chemical and physical analyses from 22 sites within the potential flood zone area. 53 samples from upland mineral soils: <0.01 - 0.04 mg/kg; 11 samples from upland forest floor horizons: 0.04 - 0.21 mg/kg; 6 samples from wetland organic soils: 0.04 - 0.10 mg/kg | No data | Used General literature estimate: Total carbon: 2.72 in upper 11 cm, labile carbon: 0.25 in upper 11 cm | <0.041 (i.e., DL) | No data | No data | 2014 (proposed) | 115 | 200 | 1.74 | 26 days | 17 °C (monthly mean) | unknown | No | 95 / ? | 43% of flood zone proposed (under review) | 1% |
| **La Grande 1 (LG1), Québec** | 6.5 - 6.6[a] | TOC : 6.4 - 6.4 | Turbidity : 2.7 - 4.1 (= 4.3 - 6.6 mg/L TSS[b]) | TP: 9-12 µg/L; NO3: <0.1 mg/L of N | 1.51 (range 0.4-2.6) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.04-0.35 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | 0.096 ± 0.024 for unflooded peat soils, 0.189 ± 0.035 for flooded peat soils from northern Québec; Maximum of 0.25 in the top 15 cm of prestine podzols in the La Grande complex | 2.6-3.5 for La Grande peat soils, 2.5-8.4 for wetland soils, 2.1-6.8 for podzols; 0.8-2.6 for La Grande flooded peat soils, 6.9-7.4 for flooded wetland soils, 1.7-3.0 for flooded podzols; depths 4-39 cm (organic layer). | 16.4-23.0 for La Grande area peat soils, 8.9-42.4 for wetland soils, 7.3-18.0 for podzols; 2.0-4.3 for La Grande area flooded peat soils, 6.9-7.4 for flooded wetland soils, 4.1-9.9 for flooded podzols; depths 4-39 cm (organic layer). Labile carbon[c]: 0.53 | 0.049 (range 0.02-0.12) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.14-1.58 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | No data (according to R. Schetagne, Hydro Québec, regional means should be available, but no data could be located) | 1993 1 month (Oct-Nov) | 30 | 70 | 2.33 | 0.15 months; 3 days | unknown | 91-96% | No | 30 / 18.6 | 85% | Probably similar to LG 2 (5% peatland) |
| **La Grande 2 (LG2; Robert-Bourassa), Québec** | 6.2 - 6.7[a] | TOC: 6.1 - 6.3; DOC: 8.9 (4.8-29.4) several stations | Turbidity: 1.0 - 1.9 (= 1.6 -3.0 mg/L TSS[b]) | TP: 10-11 µg/L; NO3: <0.1 mg/L of N | 1.51 (0.4-2.6) from 9 natural lakes (0.45 µm filtered) in northern Québec 1994-1996; 2.39 (0.8-3.7) from 4 natural lakes near LG2 (0.45 µm-filtered) sampled in 1993 | Range of 0.04-0.35 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | 0.096 ± 0.024 for unflooded peat soils, 0.189 ± 0.035 for flooded peat soils from northern Québec ; Maximum of 0.25 in the top 15 cm of pristine podzols in the La Grande complex 0.040-0.275 in the organic layer (~10 cm) of flooded LG-2 podzols and peat soils. 0.104-0.169 in flooded peat soils and forest soils of LG-2 14-16 years after flooding | 2.6-3.5 for La Grande area peat soils, 2.5-8.4 for wetland soils, 2.1-6.8 for podzols ; 0.8-2.6 for La Grande area flooded peat soils, 6.9-7.4 for flooded wetland soils, 1.7-3.0 for flooded podzols; depths 4-39 cm (organic layer). | 16.4-23.0 for La Grande area peat soils, 8.9-42.4 for wetland soils, 7.3-18.0 for podzols; 2.0-4.3 for La Grande area flooded peat soils, 6.9-7.4 for flooded wetland soils, 4.1-9.9 for flooded podzols; depths 4-39 cm (organic layer). Labile carbon[c]: 0.53 | 0.049 (range 0.02-0.12) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.14-1.58 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | 0.003-0.036 in flooded peat soils and forest soils of LG-2 14-16 years after flooding | Nov 1978 to Dec 1979 | 205 | 2835 | 13.83 | 6.9 months (0.58 yrs) | 14 to 18 °C | Winter : 0% summer : 10 to 55% | Usually stratifies from mid July to mid September (top: 12-15°C, bottom: 6-7°C); inverse stratification in winter (January-April) | 145 / 22 | near 0 | 5% peatland |
| **La Grande 3, Québec** | 6.4 - 6.5[a] | TOC: 5.1 - 5.6 | Turbidity: 0.9 - 1.0 (equals 1.4 - 1.6 mg/L of TSS[b]) | TP: 8 µg/L; NO3: <0.1 mg/L of N | 1.51 (range 0.4-2.6) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.04-0.35 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | 0.096 ± 0.024 for unflooded peat soils, 0.189 ± 0.035 for flooded peat soils from northern Québec; Maximum of 0.25 in the top 15 cm of pristine podzols in the La Grande complex | 2.6-3.5 for La Grande area peat soils, 2.5-8.4 for wetland soils, 2.1-6.8 for podzols; 0.8-2.6 for La Grande area flooded peat soils, 6.9-7.4 for flooded wetland soils, 1.7-3.0 for flooded podzols; depths 4-39 cm (organic layer). | 16.4-23.0 for La Grande area peat soils, 8.9-42.4 for wetland soils, 7.3-18.0 for podzols; 2.0-4.3 for La Grande area flooded peat soils, 6.9-7.4 for flooded wetland soils, 4.1-9.9 for flooded podzols; depths 4-39 cm (organic layer). Labile carbon[c]: 0.53 | 0.049 (range 0.02-0.12) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.14-1.58 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | No data (according to R. Schetagne, Hydro Québec, regional means should be available, but no data could be located) | April 1981 to Aug 1984 | 245 | 2420 | 9.88 | 11.0 months (0.92 yrs) | unknown | unknown | Likely similar to LG2 | 80 / 24.4 | near 0 | 10% peatland |
| **Opinaca, Québec** | 5.9 - 6.3[a] | TOC: 7.0 - 9.7 | Turbidity: 0.9 - 1.5 (equals 1.5 - 3.1 mg/L of TSS[b]) | TP: 6-12 µg/L; NO3: <0.1 mg/L of N | 1.51 (range 0.4-2.6) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.04-0.35 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | 0.096 ± 0.024 for unflooded peat soils, 0.189 ± 0.035 for flooded peat soils from northern Québec ; Maximum of 0.25 in the top 15 cm of pristine podzols in the La Grande complex | 2.6-3.5 for La Grande area peat soils, 2.5-8.4 for wetland soils, 2.1-6.8 for podzols; 0.8-2.6 for La Grande area flooded peat soils, 6.9-7.4 for flooded wetland soils, 1.7-3.0 for flooded podzols; depths 4-39 cm (organic layer). | 16.4-23.0 for La Grande area peat soils, 8.9-42.4 for wetland soils, 7.3-18.0 for podzols; 2.0-4.3 for La Grande area flooded peat soils, 6.9-7.4 for flooded wetland soils, 4.1-9.9 for flooded podzols; depths 4-39 cm (organic layer). Labile carbon[c]: 0.53 | 0.049 (range 0.02-0.12) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.14-1.58 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | No data (according to R. Schetagne, Hydro Québec, regional means should be available, but no data could be located) | April 1980 to Sept 1980 | 300 | 1040 | 3.47 | 3.8 months (0.32 yrs) | 17 - 19 | Winter : 0 to 20 % summer : 0 to 40% | Very little stratification during July and August (top: 17-19°C, bottom: 15°C); no stratification in winter . | 36 / 8.2 | near 0 | 16% peatland |
| **Caniapiscau, Québec** | 5.8 - 6.4[a] | TOC: 3.8 - 6.0 | Turbidity: 0.4 - 1.1 (= 0.6 - 1.8 mg/L TSS[b]) | TP: 4-6 µg/L; NO3: <0.1 mg/L of N | 1.51 (range 0.4-2.6) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.04-0.35 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | 0.096 ± 0.024 for unflooded peat soils, 0.189 ± 0.035 for flooded peat soils from northern Québec | available, regional means | Labile carbon[c]: 0.75 | 0.049 (range 0.02-0.12) from 9 natural lakes (0.45 µm filtered) in northern Québec sampled from 1994-1996 | Range of 0.14-1.58 in surfical sediments from 11 natural northern Québec lakes sampled in 1990-95 | No data (according to R. Schetagne, Hydro Québec, regional means should be available, but no data could be located) | Oct 1981 to Sept 1984 | 845 | 4275 (2/3 in drawdown area) | 5.06 | 25.8 months (2.2 yrs) | 14-16 | Winter : 0 to 14 % summer : 60 to 70% | Stratifies mid July to mid September (top: 14-16°C, bottom: 8-9°C); inverse stratification January-April | 45 / 16.8 | near 0 | 7% peatland |

**Table 5.3: Physical, chemical and biological parameters of Canadian reservoirs relative to the proposed Site C reservoir**



| | Chemical Parameters | | | | | | | | | | | | Physical Parameters | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | Total Hg | | | | | Methyl Hg | | | | | | | | | | | | | |
| Development Project | pH | DOC/TOC (mg/L) | TSS (mg/L); NTU* | Nutrients | Water (ng/L) | Sediment (µg/g d.w.**) | Upland or Flooded Soil (µg/g d.w.) | Mercury Pool (THg mg/m2) | Carbon Pool (kg/m2) | Water (ng/L) | Sediment (ng/g d.w) | Upland or Flooded Soil (µg/kg d.w.) | Year Impounded | Original Area (km2) | Reservoir Area (km2) | Area ratio Reservoir : Original | Mean Residence Time | Max Temp (°C) | Min Oxygen (%/mg/L) | Stratified | Max / Mean Depth (m) | % Cleared | % Wetland / peatland |
| ***A) Full-sized reservoirs*** | | | | | | | | | | | | | | | | | | | | | | | |
| | | | | | Pre- construction | | | | | Pre- construction | | | | | | | | Post-construction data | | | | Area to be flooded | |
| **Southern Indian Lake, Manitoba (Fish data are for the South Bay location)** | 7.6 - 7.7 (1972-76); 7.9 - 8.4 (1952, central portion of the lake) | TOC: 7.9-9.3 | 1 - 8 mg/L; 5.3 | Intermediate concentrations; mesotrophic; 15-20 µg/L TP, 0.05-0.1 mg/L NO3 \ NO2 | <20 (i.e., method DL; 1977-84); <5 (i.e., method DL; 1978-81) ; 24 - 27 (1981); 1.2-1.6 (1989) | 0.03 -0.06 (1979-83); Mean: 0.036 (0.01-0.07; 1980); 0.035-0.125 (1981); 0.16-0.37 (1987) | 0.08-0.17 means of 4-10 samples of unflooded soil horizon Ah from 5 sites; 0.07-0.12 means of 4-12 peat/litter/mosses samples from 5 sites; 0.040 (leaf litter), 0.102 (A horizon), 0.056 (B horizon) (n=1; data are presented as ng/g; Means of 3-9 samples of flooded soils from 3 sites (peat/litter/mosses: 0.07-0.16; clay: 0.02-0.08 | No data | 21-176 mg/g d.w. for surfical sediments | <2 (i.e., method DL; 0.3 (0.2-0.4); 0.01-0.05 | 11.3-14.6 | No data | 1976 | 1977 | 2391 | 1.21 | 0.51 (pre), 0.72 (post) yrs | 18.7; up to 24 in shallow bays | 7.2 mg/L at bottom in summer | No | >23 / 8.5 | near 0 | No data, likely >5% |
| **Long Spruce GS, Nelson River Manitoba** | 8.0-8.5 (summer), 7.7-8.0 (winter) | 9.3 (7.1-12.1) | 12.1 (6-20) mg/L | High concentrations; meso-eutrophic; 22.8 (12-31) µg/L TDP plus 11.7 (7-17) µg/L Susp P; 0.023 (0.002-0.044 mg/L NO3 | No data | No data | No data | No data | No data | No data | No data | No data | 1977 | 15.8 | 29.5 | 1.87 | 1.4 (range 0.6-10.7) days | 17 (20 at Station 4) | ~100% | No | 30 / 12.4 | near 0 | No data, likely <3% |
| **Limestone GS, Nelson River, Manitoba** | 8.0-8.5 (summer), 7.7-8.0 (winter) | 9.3 (7.1-12.1) 8 (8-9) TOC | 12.1 (6-20) mg/L (1972-73); 13 (9-17) mg/L (1989) | High concentrations; meso-eutrophic; 22.8 (12-31) µg/L TDP plus 11.7 (7-17) µg/L Susp P; 0.023 (0.002-0.044 mg/L NO3 30 (28-32) ug/L TP | Only post-construction data available: mean of 0.88 ±0.33 for bi-weekly measurements between July 2003 and February 2007 | No data | No data | No data | No data | Only post-construction data available: mean of 0.05 ±0.03 bi-weekly measurements during open-water July 2003 and February 2007 | No data | No data | 1990 | 22.0 25.0 | 28.8 27.1 | 1.31 1.08 | 1.3 (range 0.5-5.5) days | 17 (20 at Station 4) | 10.2 (8.9-11.4) mg/L (year 1991) | No | 33 / 13.5 | near 0 | No data, likely <3% |
| **Wuskwatim GS, Burntwood River Manitoba (2012, under construction [f])** | 7.4 - 7.9 (winter - fall) | 6- 17 DOC 6 -17 TOC | 2 - 24 mg/L | High concentrations; meso-eutrophic; 18-48 µg/L TP, <0.005-0.2 mg/L NO3 \ NO2 | <50 (i.e., method DL) | 0.02-0.03 | No data | No data | No data | No data | No data | No data | 2012 (?) 1977 [f] | 89.7 [e] 53.5 | 94.3 79.3 | 1.05 1.48 | Pre-construct: 3.5 (range 2.8-7.7) days; post-construct: 3.9 (range 2.6-6.9) days | 20 - 23 | 70-80%; locally lower in winter | No | 13.5 / 8.0 | near 0 | No data, likely <1% |
| **Williston Reservoir, Peace River, British Columbia Finlay Reach** (unless stated otherwise) | 8.5-8.6 | 1.7-3.4 DOC | <3 (DL) mg/L | oligotrophic; 2-10 µg/L TP, <0.002 mg/L NO3; whole reservoir (1999-2000): means of 6.2-7.4 µg/L TP, 0.057-0.062 NO3 | 0.4-1.5 unfiltered | 0.035-0.069 (total range from 2 littoral and 2 profundal stations: 0.022-0.092) | The sediments sampled likely represent flooded soils ~30 yrs after flooding) | No data | No data | 0.019-0.108 at the surface | 0.11-0.47; <1% of THg | The sediments sampled likely represent flooded soils ~30 yrs after flooding) | 1967-72 | Estimated at 79 km2 | 1,779 | 22 | 19 months; 727 days (1.99 yrs) using 1183 $m^3/s$ inflow and 7.43 x 100 billion $m^3$ storage | 16.6 | 8.0-10.7 mg/L | Yes Weak | 166 / 41.7 | 0 | 0.5% bogs & wetlands |
| **Site C Reservoir, Peace River, British Columbia** | 7.8 - 8.6 | 2.0-2.8 DOC 2.0-2.9 TOC | <1 mg/L | Low concentrations; oligotrophic | 0.7 - 1.0 | 0.03 - 0.06 | 0.10 (0.022 – 0.139 ) | 0.05 | 5.0 Averaged over the flood zone | <0.02 - 0.07 | 0.5 - 1.8 | 0.071 - 7.1 | 2023 | 3.97 | 9.31 | 2.35 | 23 days | 12 - 14 | ~100% | No | 60 / 20 | Yes, nearly completely | 1-2% |
| ***B) Experimental Reservoirs*** | | | | | | | | | | | | | | | | | | | | | | | |
| **ELARP** | 6.6 (6.3-6.9) | 814 µM/L DOC (range 540-1170) | Not measured | TP 7.3 µg/L (range 5-10) | 3.43 +/- 1.4 and 2.62+/- 1.0 in two pre-flood years | not measured | mean 0.06 (0.019 - 0.123) in top 3 cm of flooded peat | Top 3 cm: mean 60.0 ng/g dw (range 19.0 - 122.8); top 50 cm mean 56.1 ng/g dw (S.D.: 22.73) | 100 | 0.145 +/- 0.08 and 0.08 +/- 0.02 in two pre-flood years | not measured | approx. 0.1 - 1.0 | 1993 | 2.4 ha (Original central pond) | 16.7 ha | 6.96 | Highly variable; days to infinity | 25 | 0 mg/L | Yes | Max: 2.5 | 0 | 100% |
| **FLUDEX High Carbon** | 6.88 - 7.16 (inflowing water from source lake) | 410-510 µM/L DOC (inflowing water from source lake) | Not measured | 220-235 µg/L TDN, 2-4 µg/L TDP, (inflowing water from source lake) | 0.8 - 2.3 (in pumped inflows) | No data (no pre-flooding water body) | 0.089 (upper organic soil horizons) | 1.49 (includes trees, plants, and all soil layers) | 4.6 total carbon; 1.9 labile carbon [g] | 0.01 - 0.15 (in pumped inflows) | No data (no pre-flooding water body) | 1.13 (upper organic soil horizons) | 1999 | 0 (no original water body, 100% terrestrial) | 0.74 ha | No data (all flooded area) | 8-11 days | 24 | 0.4 mg/L | No | Max: 2 | 0 | 0 |
| **FLUDEX Medium Carbon** | 6.88 - 7.16 (inflowing water from source lake) | 410-510 µM/L DOC (inflowing water from source lake) | Not measured | 220-235 µg/L TDN, 2-4 µg/L TDP, (inflowing water from source lake) | 0.8 - 2.3 (in pumped inflows) | No data (no pre-flooding water body) | 0.044 (upper organix soil horizons) | 1.45 mg/ha (includes trees, plants, and all soil layers) | 3.5 total carbon; 0.9 labile carbon [g] | 0.01 - 0.15 (in pumped inflows) | No data (no pre-flooding water body) | 0.20 (upper organic soil horizons) | 1999 | 0 (no original water body, 100% terrestrial) | 0.60 ha | No data | 6-10 days | 24 | 0.3 mg/L | No | Max: 2 | 0 | 0 |
| **FLUDEX Low Carbon** | 6.88 - 7.16 (inflowing water from source lake) | 410-510 µM/L DOC (inflowing water from source lake) | Not measured | 220-235 µg/L TDN, 2-4 µg/L TDP (inflowing water from source lake) | 0.8 - 2.3 (in pumped inflows) | No data (no pre-flooding water body) | 0.039 (upper organic soil horizons) | 0.76 (includes trees, plants, and all soil layers) | 3.1 total carbon; 1.1 labile carbon [g] | 0.01 - 0.15 (in pumped inflows) | No data (no pre-flooding water body) | 0.52 (upper organic soil horizons) | 1999 | 0 (no original water body, 100% terrestrial) | 1.13 ha | No data | 7-8 days | 24 | 3.3 mg/L | No | Max: 2 | 0 | 0 |

All fish lengths are given as fork length (FL), except for Québec (total length)
Chemical parameters are given as ranges (x - x), means, or means +/- standard deviation
NRL = Natural Regional Lakes
N/A = not applicable
* Nephelometric Turbidity Units (NTU).
** d.w.= dry weight; w.w.= wet weight

[a] pH : pre-impondment river values during ice free period
[b] For the La Grande Rivière, a 1.6 average TSS/Turbidity ratio was found. So could multiply the turbidity values given in NTU by 1.6 to get the TSS in mg/L
[c] Carbon in leaves, needles of trees, and the soil litter horizon, x labile soil fraction (US Department of Agriculture; handbook No. 379. 1970). (R.Schetagne, pers. comm. 2011)
[d] Fish from several years were pooled if yearly means were not significantly different (R.Schetagne, pers. comm. 2011). This applies to all maximum fish concentrations from PQ reservoirs
[e] Different from the 79.3 km2 reservoir area for 1977 because flooded areas outside of Wuskwatim Lake proper were considered.
[f] Wuskwatim Lake was previously flooded as a result of the Churchill River Diversion in 1977; all data other than for area are exclusively for the Project under construction.
[g] Carbon in tree foliage, shrubs, herbs, mosses, lichens, and in the litter fungal/humic soil layer (D. Bodaly, pers. comm. 2011)

**Table 5.3: Physical, chemical and biological parameters of Canadian reservoirs relative to the proposed Site C reservoir**



| Development Project | Biological Parameters | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Invertebrate Baseline and Post-Impoundment [µg/g d.w.] | | | | Fish Baseline and Post-Impoundment Hg [µg/g w.w.]; Ratio Max(imum)/Baseline [Hg]; Time to return to Baseline [Hg] (years) | | | | | | |
| | Zooplankton THg (Mean, range) | Zooplankton MeHg (Mean, range) | Benthos THg (Mean, Range) | Benthos MeHg (Mean, Range) | Forage Species | Lake Whitefish (except Site C: Mountain Whitefish) | Northern Pike | Walleye | Other | Comment | References |
| **Muskrat Falls GS (proposed), Lower Churchill River, Newfoundland and Labrador** | Baseline: 0.068-0.260 for 200 µm mesh catch from 5 sites, including 2 NRL; Post-Impound: No data | Baseline: 0.002-0.072 for 200 µm mesh catch from 5 sites, including 2 NR; Post-Impound: No data | No data | No data | No data | Baseline: Mean of 0.19 (range: likely available) for fish standardized to 400 mm FL(?) from 2 mainstem lakes Post-impound (max): Mean of 0.69; Ratio MaxP/BLine: 3.6 | Baseline: Mean of 0.81 (range: likely available) for fish standardized to 700 mm FL(?) from 2 mainstem lakes Post-impound (max): Mean of 1.38; Ratio MaxP/BLine: 1.7 | No data | **L Trout** Base line: Mean of 0.95 for fish standardized to 600 mm FL from 2 NRLs; Post-impound (max): Mean of 1.63 (predicted); **LN Sucker** Base line: Mean of 0.25 for fish standardized to 400 mm FL from 1 NRL Post-impound (max): Mean of 0.61 predicted); Ratio MaxP/BLine: 2.4 **White Sucker** Base line: Mean of 0.26 for fish standardized to 400 mm FL from 1 mainstem lake Post-impound (max): Mean of 0.45 (predicted); Ratio MaxP/BLine: 1.7; Data for **Brook Trout** available | | Harris and Hutchinson (2007)<br>Minaskuat Inc. (2007)<br>Minaskuat Inc. (2008) |
| **Gull Island GS (proposed), Lower Churchill River Newfoundland and Labrador** | Baseline: 0.068-0.260 for 200 µm mesh catch from 5 sites, including 2 NRL Post-Impound: No data | Baseline: 0.002-0.072 for 200 µm mesh catch from 5 sites, including 2 NRL Post-Impound: No data | No data | No data | No data | Baseline: Mean of 0.19 for fish standardized to 400 mm FL(?) from 2 mainstem lakes Post-impound (max): Mean of 0.32 (Predicted); Ratio MaxP/BLine: 1.7 | Baseline: Mean of 0.81 for fish standardized to 700 mm FL(?) from 2 mainstem lakes Post-impound (max): Mean of 1.38 (Predicted); Ratio MaxP/BLine: 1.7 | No data | **L Trout** Base line: Mean of 0.95 for fish standardized to 600 mm FL from 2 NRL; Post-impound (max): Mean 1.63 (predicted); Ratio MaxP/BLine: 1.7; **LN Sucker** Baseline: Mean of 0.25 standardized to 400 mm FL from 2 mainstem lakes; Post-impound (max): Mean of 0.43 (Predicted); **White Sucker** Baseline: Mean of 0.26 standardized to 400 mm FL from 2 mainstem lakes; Post-impound (max): Mean of 0.45 (predicted); Ratio MaxP/BLine: 1.7 Data for Brook trout available | | |
| **La Grande 1 (LG1), Québec** | Base line: Range of 0.025-0.575 for different taxa from 13 natural lakes in northern Quebec, means of 0.085-0.223 from 3 natural lakes in the LG2 area; Post-impound: No data | Base line: Range of 0.025-0.510 for different taxa from 13 natural lakes in northern Quebec, means of 0.018-0.043 from 3 natural lakes in the LG2 area; Post-impound: No data | Base line: Means of 0.031-0.790 for insect larvae from 11 natural lakes in northern Quebec; Post- impound: No data | Base line: Range of 0.025-0.510 for different taxa from 11 natural lakes in northern Quebec Post- impound : No data | No data | Base line: Means of 0.05-0.20 (overall mean: 0.11) for 400 mm long fish from NRL; Mean of 0.09 (range 0.06-0.14) for 10 fish from the LG-1 location of the La Grande River in 1972; Post-impound (max): Mean of 1.18; Ratio Max/BLine: 10.7; Time to return: 21 years (since LG 2 impoundment) | Base line: Means of 0.30-0.93 (overall mean: 0.59) for 700 mm-long fish from NRL Post-impound (max): Mean of 5.14; Ratio Max/BLine: 8.7; Time to return: 21 years (since LG 2 impoundment) | Base line: Means of 0.30-1.02 (overall mean: 0.60) for 400 mm-long fish from NRL; Post-impound (max): No data; Ratio Max/BLine: No data; Time to return: >29 years (since LG 2 impoundment) | Cisco Base line: Means of 0.08-0.30 (overall mean: 0.11) for 150 mm fish from NRL; Post-impound (max): No data; Ratio MaxP/BLine: No data; Time to return: ? **LN Sucker** Base line: Means of 0.12-0.22 (overall mean: 0.12) for 400 mm-long fish from NRL; Post-impound (max): Mean of 1.26; Ratio Max/BLine: 10.5; Time to return: >25 years (since LG 2 impoundment). **Burbot** Base line: Means of 0.60-0.84 (overall mean: 0.82) for 500 mm fish from 1 NRL; Post-impound (max?): Mean of 2.66; Ratio Max/BLine: >3.2; Time to return: 21 y | For all Québec reservoir fish: Longnose sucker (LN sucker): 400 mm standardized total length Lake whitefish: 400 mm standardized total length Northern pike: 700 mm standardized total length Walleye: 400 mm standardized total length Lake trout (L Trout): 600 mm standardized total length | Grondin et al. (1995)<br>Lucotte et al. (1999)<br>Tremblay (1999a)<br>R. Schetagne, pers.comm. 2011<br>Schetagne et al. (2003)<br>Schetagne et al. (2005)<br>Smith et al. (1975)<br>Therrien and Schetagne (2009)<br>Tremblay et al. (1998) |
| **La Grande 2 (LG2; Robert-Bourassa), Québec** | Base line: Range of 0.025-0.575 for different taxa from 13 natural lakes in northern Quebec, means of 0.085-0.223 from 3 natural lakes in LG2 area; Post-impound: 0.45-0.67 at 3 littoral zone stations 14-16 y post-flood | Base line: Range of 0.025-0.510 for different taxa from 13 natural lakes in northern Quebec, means (0.018-0.043 from 3 natural lakes in LG2 area; Post-impound: 0.282-0.450 at 3 littoral zone stations 14-16 y post-flood | Base line: Means of 0.031-0.790 for insect larvae from 11 natural lakes in northern Quebec Post-impound : 0.15-0.40 in Diptera (Benthos taxon with lowest [Hg]) - 0.25-0.80 in Heteroptera (Benthos taxon with highest [Hg]) from 14-16 yr old sediments | Base line: Means of 0.025-0.510 for insect larvae from 11 natural lakes in northern Quebec Post-impound : 0.02-0.15 in Diptera (Benthos taxon with lowest [Hg]) to 0.20-0.60 in Heteroptera (Benthos taxon with highest [Hg]) from 14-16 yr old sediments | **Trout perch** Baseline: No data Post-impound (NOT max): Mean of ~0.52 of fish 120 mm mean TL ; Ratio Max/BLine: N/A; Time to return: unknown **Yellow perch** Baseline: Mean of ~0.08 from Lake Rond-de-Poele ; Post-impound (NOT max): Mean of 0.30 of fish 140 mm mean TL; Ratio Max/BLine: >3.8; Time to return: unknown | Base line: Means of 0.05-0.20 (overall mean: 0.11) for 400 mm long fish from NRL; Post- impound (max): Mean of 0.52 (range: 0.29 (218 mm) - 1.25 (565 mm); Ratio Max/BLine: 4.7; Time to return: 19 years | Base line: Means of 0.30-0.93 (overall mean: 0.59) for 700 mm-long fish from NRL; Post- impound (max): Mean of: 3.28 (range 0.39 (260 mm) - 11.7 (1015 mm); Ratio Max/BLine: 5.6; Time to return: >29 years | Base line: Means of 0.30-1.02 (overall mean: 0.60) for 400 mm-long fish from NRL; Post- impound (max): Mean of 2.76 (range 0.28 (114 mm) - 5.63 (650 mm); Ratio Max/BLine: 4.6; Time to return: >29 years | Cisco Base line: Means of 0.08-0.30 (overall mean: 0.11) for 150 mm-long fish from NRL; Post-impound (max): Mean of 0.50; Ratio Max/BLine: 4.5; Time to return: ? **LN Sucker** Base line: Means of 0.12-0.22 (overall mean: 0.12) for 400 mm-long fish from NRL; Post-impound (max): Mean of 0.63 (range 0.25 (264 mm) - 1.48 (557 mm); Ratio Max/BLine: 5.3; Time to return: 21 years (Schetagne et al. 2003, Fig. 5.7) **Burbot** Base line: Means of 0.60-0.84 (overall mean: 0.82) for 500 mm fish from 1 NRL; Post-impound (max?): Mean of 2.36; Ratio Max/BLine: >2.9; Time to return: 17 y (since LG 2 impoundment) | | R. Schetagne, pers. comm. 2011<br>Montgomery et al. (1995)<br>Lucotte et al. (1999)<br>Tremblay (1999a)<br>Grondin et al. (1995)<br>Tremblay and Lucotte (1997)<br>Schetagne et al. (2003)<br>Schetagne et al. (2005)<br>Tremblay et al. (1998)<br>Tremblay (1999b)<br>Therrien and Schetagne (2009) |
| **La Grande 3, Québec** | Base line: Range of 0.025-0.575 for different taxa from 13 natural lakes in northern Quebec, means of 0.085-0.223 from 3 natural lakes in the LG2 area; Post- impound: No data | Base line: Range of 0.025-0.510 for different taxa from 13 natural lakes in northern Quebec; 0.018-0.043 from 3 natural lakes in the LG2 area; Post-impound: No data | Base line: Means of 0.031-0.790 for insect larvae from 11 natural lakes in northern Quebec; Post- impound: No data | Base line: Means of 0.025-0.510 for insect larvae from 11 natural lakes in northern Quebec Post-impound: No data | **Trout perch** Baseline: No data Post-impound (NOT max): Mean of ~0.45 of fish 110 mm mean TL; Ratio Max/BLine: N/A; Time to return: unknown | Base line: Means of 0.05-0.20 (overall mean: 0.11) for 400 mm long fish from NRL; Post-impound (max): Mean of 0.37 (range 0.17 (220 mm) - 1.08 (585 mm); Ratio Max/BLine: 3.4; Time to return: >27 years | Base line: Means of 0.30-0.93 (overall mean: 0.59) for 700 mm-long fish from NRL; Post-impound (max): Mean of 4.16 (range 0.67 (171 mm) - 8.38 (915 mm); Ratio Max/BLine: 7.1; Time to return: >27 years | Base line: Means of 0.30-1.02 (overall mean: 0.60) for 400 mm-long fish from NRL; Post-impound (max): ?, no data; Ratio Max/BLine: No data; Time to return : 23 years | **Cisco** Base line: Means of 0.08-0.30 (overall mean: 0.11) for 150 mm-long fish from NRL; Post- impound (max?): Mean of 1.10; Ratio Max/BLine: 5.8; Time to return: ? **LN sucker** Base line: Means of 0.12-0.22 (overall mean: 0.12) for 400 mm-long fish from NRL; Post- impound (max): Mean of 0.55 (range 0.20 (121 mm) - 1.10 (616 mm); Ratio Max/BLine: 4.6; Time to return: 19 years | | Schetagne et al. (2003)<br>Lucotte et al. (1999)<br>Tremblay (1999a)<br>Grondin et al. (1995)<br>R. Schetagne, pers. comm. 2011<br>Schetagne et al. (2005)<br>SEBJ (1987)<br>Therrien and Schetagne (2009)<br>Tremblay et al. (1998) |
| **Opinaca, Québec** | Base line: Range of 0.025-0.575 for different taxa from 13 natural lakes in northern Quebec, means of 0.085-0.223 from 3 natural lakes in the LG2 area; Post-impound: 0.148-0.191 12-14 years after flooding | Base line: Range of 0.025-0.510 for different taxa from 13 natural lakes in northern Quebec, 0.018-0.043 from 3 natural lakes in the LG2 area Post-impound: 0.048-0.082 12-14 years after flooding | Base line: Means of 0.031-0.790 for insect larvae from 11 natural lakes in northern Quebec; Post- impound: No data | Base line: Means of 0.025-0.510 for insect larvae from 11 natural lakes in northern Quebec; Post-impound: No data | **Trout perch** Baseline: No data. Post-impound (NOT max): Mean of ~0.32 of fish 110 mm mean TL; Ratio Max/BLine: N/A; Time to return: unknown **Yellow perch** Baseline: Mean of ~0.08 from Lake Rond-de-Poele Post-impound (NOT max): Mean of ~0.33 of fish 150 mm mean TL; Ratio Max/BLine: >4.1; Time to return: unknown | Base line: Means of 0.05-0.20 (overall mean: 0.11) for 400 mm long fish from NRL; Post-impound (max): Mean of 0.44 (range 0.19 (190 mm) - 1.21 (588 mm); Ratio Max/BLine: 4.0; Time to return: 20 years | Base line: Means of 0.30-0.93 (overall mean: 0.59) for 700 mm-long fish from NRL; Post-impound (max): Mean of 2.91 (range 0.22 (167 mm) - 7.80 (1100 mm); Ratio Max/BLine: 4.9; Time to return: >27 years | Base line: Means of 0.30-1.02 (overall mean: 0.60) for 400 mm-long fish from NRL; Post-impound (max): Mean of 2.05 (range: 0.15 (184 mm) - 4.58 (681 mm); Ratio Max/BLine: 3.4; Time to return: 20 years | **Cisco** Base line: Means of 0.08-0.30 (overall mean: 0.11) for 150 mm-long fish from NRL; Post- impound (max): No data **LN Sucker** Base line: Means of 0.12-0.22 (overall mean: 0.12) for 400 mm-long fish from NRL; Post-impound (max?): Mean of 0.72 (range: 0.26 (312 mm) - 1.34 (510 mm); Ratio Max/BLine: 6.0; Time to return: 24 years | | Grondin et al. (1995)<br>Lucotte et al. (1999)<br>Schetagne et al. (2003)<br>Schetagne et al. (2005)<br>SEBJ (1987)<br>R. Schetagne, pers. comm. 2011<br>Therrien and Schetagne (2005)<br>Therrien and Schetagne (2009)<br>Tremblay (1999a)<br>Tremblay et al. (1998) |
| **Caniapiscau, Quebéc** | Base line: Range of 0.025-0.575 for different taxa from 13 natural lakes in northern Quebec Post-impound: 0.274-0.466 11-14 y post-flood | Base line: Range of 0.025-0.510 for different taxa from 13 natural lakes in northern Quebec Post-impound: 0.087-0.168 11-14 y post-flood | Base line: Means of 0.031-0.790 for insect larvae from 11 natural lakes in northern Quebec; Post- impound: No data | Base line: Means of 0.025-0.510 for insect larvae from 11 natural lakes in northern Quebec; Post-impound: No data | **Lake chub** Baseline: Mean of ~0.17 from Lake Sérgny Post-impound (NOT max): Mean of ~0.33 of fish 140 mm mean TL; Ratio Max/BLine: >1.9; Time to return: unknown | Base line: Means of 0.10 - 0.30 (overall mean: 0.17) for 400 mm long fish from NRL; Post- impound (max?): Mean of 0.43 (range: 0.08 (310 mm) - 1.52 (574 mm); Ratio Max/BLine: 2.5; Time to return: 11 years | Base line: Means of 0.36-0.92 (overall mean: 0.55) for 700 mm-long fish from NRL; Post-impound (max): Mean of 2.29 (range: 0.24 (302 mm) - 3.26 (1060 mm); Ratio Max/BLine: 4.2; Time to return: 25 years | No data | Lk Trout Base line: Means of 0.52 to 1.11 (overall mean: 0.74) for 700 mm-long fish from NRL; Post-impound (max): Mean of 2.08 (range: 0.61 (431 mm) - 5.65 (1040 mm); Ratio Max/BLine: 2.8; Time to return: 25; **LN Sucker** Base line: Means of 0.06 - 0.20 (overall mean: 0.12) for 400 mm fish from NRL; Post-impound (max): Mean of 0.52 (range: 0.07 (74 mm) - 0.99 (611 mm); Ratio Max/BLine: 4.3; Time to return: 17 years | | R. Schetagne, pers. comm. 2011<br>Schetagne et al. (2003)<br>Schetagne and Roy (1985)<br>SEBJ (1987)<br>Therrien and Schetagne (2008a)<br>Tremblay (1999a)<br>Tremblay et al. (1998) |

**Table 5.3: Physical, chemical and biological parameters of Canadian reservoirs relative to the proposed Site C reservoir**

## Biological Parameters

| | Invertebrate Baseline and Post-Impoundment [µg/g d.w.] | | | | Fish Baseline and Post-Impoundment Hg [µg/g w.w.]; Ratio Max(imum)/Baseline [Hg]; Time to return to Baseline [Hg] (years) | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Development Project** | **Zooplankton THg (Mean, range)** | **Zooplankton MeHg (Mean, range)** | **Benthos THg (Mean, Range)** | **Benthos MeHg (Mean, Range)** | **Forage Species** | **Lake Whitefish (except Site C: Mountain Whitefish)** | **Northern Pike** | **Walleye** | **Other** | **Comment** | **References** |
| **Southern Indian Lake, Manitoba (Fish data are for the South Bay location)** | Baseline: No data<br>Post-impound: 0.27-0.76, 0.25-1.50, and 0.67-3.0 for zooplankton >73 >153, and >351 µm mesh, respectively, from 4 areas ; 1.06-2.11 for plankton >10 µm mesh from 2 areas | No data | Baseline: No data<br>Post-impound: 0.12-3.49 (converted from 0.023 - 0.698 w.w.) for clams, Nematodes, Oligochetes, chironomid larvae from 2 areas | No data | **Spottail Shiner** Baseline: No data; Post-impound: Means of 0.06-0.38 (total range 0.05-0.46) for fish 47-94 mm TL from 5 sites; Ratio Max/BLine: N/A; Time to return: unknown **Yellow Perch** Baseline: No data; Post-impound: Means of 0.06-0.11 (total range 0.03-0.18) for fish 49-64 mm TL from 3 sites Ratio Max/BLine: N/A; ime to return: unknown | Baseline: Mean of 0.069 (total range 0.03-0.12) for fish length standardized to 350 mm<br>Post-impound (max): Mean of 0.24 for fish length standardized to 350 mm; Ratio Max/BLine: 3.4; Time to return: 10 years | Baseline: Means of 0.26-0.30 (total range: 0.24-0.33) for large fish from lake-wide commercial harvest<br>Post-impound (max): Mean of 1.10 (range 0.26-2.53) for fish from SIL, South Bay length standardized to 550 mm FL; Ratio Max/BLine: 3.9; Time to return: 18 years | Baseline: Means of 0.19-0.30 (total range: 0.16-0.38) for large fish from lake-wide commercial harvest<br>Post-impound (max): Mean of 0.78 (range 0.38-1.49) for fish length standardized to 400 mm FL; Ratio Max/BLine: 3.2; Time to return: 16 years | **LN sucker** Baseline: No data; Post-impound (max): Mean of 0.10 (range 0.04-0.28) for fish length standardized to 400 mm FL from unknown sites; **Cisco** Baseline: No data; Post-impound (max): Mean of 0.20 (range 0.08-0.58) for fish length standardized to 300 mm FL | | Bodaly et al. (2007)<br>Bodaly et al. (1984 ; 1987)<br>Jackson (1988a)<br>Jackson (1988b)<br>Hecky (1984)<br>Hecky and McCullough (1984)<br>McTavish 1952<br>Newbury et al. (1984)<br>Ramsey (1990; 1991; 1988)<br>Williamson (1986)<br>Williamson and Ralley (1993) |
| **Long Spruce GS, Nelson River Manitoba** | No data | No data | No data | No data | No data | Baseline: Means of 0.03-0.06 for 350 mm-long fish from NRL;<br>Post-impound (max?, year 1985): Mean of 0.18 (range: 0.05-0.49) of fish standardized to 350 mm FL; Ratio Max/BLine: 4.0; Time to return: 9 years | Baseline: Means of 0.36-0.47 for 550 mm-long fish from NRL;<br>Post-impound (max?, year 1985): Mean of 0.70 (range: 0.09-2.31) of fish standardized to 550 mm FL; Ratio Max/BLine: >1.7; Time to return: 19 | Baseline: Means of 0.35-0.47 for 400 mm-long fish from NRL;<br>Post-impound (max?, year 1985): Mean of 0.64 (range: 0.07-1.03) of fish standardized to 400 mm FL; Ratio Max/BLine: >1.6; Time to return: 19 | **Mooneye** Baseline: No data; Post-impound (max?, year 1985): Mean of 0.26 (range: 0.07-0.46) of fish standardized to 250 mm FL; **Sauger** Baseline: No data ; Post-impound (max?, year 1986): Mean of 0.61 (range: 0.21-1.31) of fish standardized to 300 mm FL | | Bodaly et al. (2007)<br>Cleugh (1974)<br>Jansen and Strange (2007)<br>A. Zacharias, pers. comm 2011<br>Swanson (1986)<br>Swanson and Kansas (1987) |
| **Limestone GS, Nelson River, Manitoba** | No data | No data | No data | No data | **Spottail Shiner** Baseline: No data<br>Post-impound (NOT max): Mean of 0.06 (range 0.03-0.14) standardized to 75 mm FL **Emerald Shiner** Baseline: No data Post-impound (NOT max): Mean of 0.07 (range 0.03-0.15) standardized to 75 mm FL **Rainbow Smelt** Baseline: No data; Post-impound (NOT max): Mean of 0.02 (range 0.01-0.05)standardized to 100 mm FL **Trout perch** Baseline: No data ; Post-impound (NOT max): Mean of 0.04 (range 0.02-0.08) standardized to 75 mm FL | Baseline: Mean of 0.08 (range 0.01-0.013) for fish standardized to 350 mm FL<br>Post-impound (max): Mean of 0.13 (range: 0.07-0.35) of fish standardized to 350 mm FL;<br>Ratio Max/BLine: 1.6; Time to return: 9 years | Baseline: Mean of 0.32 (range 0.06-0.52) for fish standardized to 550 mm FL<br>Post-impound (max): Mean of 0.45 (range: 0.22-1.02) of fish standardized to 550 mm FL;<br>Ratio Max/BLine: 1.4; Time to return: 9 years | Baseline: No data (1989 data for only 3 fish)<br>Post-impound (max): Mean of 0.64 (range: 0.13-1.30) of fish standardized to 400 mm FL;<br>Ratio Max/BLine: N/A; Time to return: 9-12 years | **LN Sucker**<br>Baseline: Mean of 0.06 (range: 0.02-0.16) of fish length standardized to 400 mm<br>Post-impound (max): Mean of 0.21 (range: 0.02-0.32) of fish standardized to 400 mm FL; Ratio Max/BLine: 3.5 | | Baker (1990)<br>Baker (1992)<br>Cleugh (1974)<br>Horne and MacDonell (1995)<br>Jansen and Strange (2007)<br>Kirk and St. Louis (2009)<br>MacDonell and Horne (1994)<br>A. Zacharias, pers. comm 2011 |
| **Wuskwatim GS, Burntwood River Manitoba (2012, under construction [f])** | No data | No data | No data | No data | No data | Baseline: Means of 0.05-0.12 (total range: 0.02-0.31) for length standardized (350 mm) fish captured in 2002, 05, 07<br>Post-impound: 0.10-0.11 (predicted for length standardized fish); Ratio MaxP/BLine: 1.2; Time to return (predicted): 13-15 years | Baseline: Means of 0.31-0.48 (total range: 0.04-1.52) for length standardized (550 mm) fish captured in 2002, 05, 07<br>Post-impound: 0.38-0.44 (predicted); Ratio MaxP/BLine: 1.04; Time to return (predicted): 13-15 years | Baseline: Means of 0.26-0.39 (total range: 0.10-0.81) for length standardized (400 mm) fish captured in 2002, 05, 07<br>Post-impound: 0.30-0.36 (predicted); Ratio MaxP/BLine: 1.02; Time to return (predicted): 13-15 years | **Cisco:** Baseline: Means of 0.06-0.11 (total range: 0.02-0.29) for length standardized (300 mm) fish captured in 2002, 2005<br>Post-impound: NO predictions available | Under construction; expected in-service date: 2012;<br>All pre-impoundment data collected from 2000-2007 | Jansen and Strange (2007)<br>Jansen and Strange (2009)<br>Jansen (2009)<br>Manitoba Hydro and NCN (2003)<br>A. Zacharias, pers. comm 2011 |
| **Williston Reservoir, Peace River, British Columbia Finlay Reach** (unless stated otherwise) | Baseline: No data<br>Post-impound: 0.061-0.18 in 2000, 0.03-0.05 in 2001 for zooplankton >250 µm mesh | Baseline: No data<br>Post-impound: 19-37% of THg values in 2000, 70% of THg values in 2001 for zooplankton >250 um mesh | Baseline: No data<br>Post-impound: 0.20-0.57 in 2000, 0.15-0.28 in 2001 | Baseline: No data Post-impound: 0.04-0.09 in 2001 | No data | Baseline: No data<br>Post-impound (max?): Means of 0.09 (range: 0.10-0.38) in 1980, 0.21 (range: 0.07-0.40) in 1988, and 0.20 (range: 0.03-0.24) in 2000 for fish standardized to 300 mm FL | No data | No data | **Bull Trout** Baseline: No data<br>Post-impound (max?): Means of 0.68 (range: 0.16-1.69) in 1980, 0.87 (range: 0.14-4.87) in 1988, and 0.56 (range: 0.08-2.22) in 2000 for fish standardized to 550 mm FL | | EVS (2002)<br>Stockner et al. (2005)<br>Tetra Tech (2002)<br>The Peace Water Use Plan Committee (2003) |
| **Site C Reservoir, Peace River, British Columbia** | Baseline: 0.007 (0.004 - 0.009) | Baseline: 0.0004 (0.0001 - 0.0007) | Baseline: 0.016 (0.010 - 0.023) | Baseline: 0.08 (0.0016 - 0.019) | **Redside shiner** Baseline: Mean of 0.05 (range: 0.03 - 0.07) | Baseline: Mean of 0.03 (range: 0.01 - 0.06) | Not Present | Not Present | **Longnose sucker** Baseline: Mean of 0.04; range: 0.02 - 0.12<br>**Bull trout** Baseline: Mean of 0.055; range: 0.03 - 0.08 | All data collected in 2008 and 2010 from 3 locations in the Peace River mainstem within proposed reservoir area | Azimuth (2011)<br>Golder (2009) |
| **ELARP** | Baseline: 0.087 (0.060-0.238)<br>Post-impound: Means of 0.578, 0.619, and 0.502 in three years (total range 171-1173) | Baseline: 0.032 (0.011-0.054)<br>Post-impound: Means of 0.346, 0.319, and 0.300 in three years (total range 29-692) | No data | Baseline: means 0.035-0.060 in collector / shredders, 0.083-0.176 in predators; Post-impound: means 0.047-0.128 in collector / shredders, 0.230-0.344 in predators | Finescale dace (Introduced) Baseline Mean of approximately 0.08 - 0.14 ug/g whole body w.w. (end of summer); post-flooding means approximately 0.2 - 0.3 ug/g whole body w.w. (end of summer); Ratio MaxP/BLine: ~2; Time to return: N/A | No data | No data | No data | | Total Hg in water in 6 post-flood y: 3.2 ng/L +/- 1.9; 2.7 +/- 0.8; 2.6 +/- 1.2; 2.3 +/- 0.7; 2.2 +/- 0.66; 2.2 +/- 0.5. Methyl Hg in 7 post-flooding years: 0.76 +/- 0.74; 0.87 +/- 0.65; 0.62 +/- 0.66; 0.54 +/- 0.27; 0.64 +/- 0.5; 0.46 +/- 0.40; 0.53 +/- 0.43 | Bodaly and Fudge (1999)<br>Hall et al. (2005)<br>Hall et al. (1998)<br>Kelly et al. (1997)<br>Paterson et al. (1998)<br>St. Louis et al. (2004) |
| **FLUDEX High Carbon** | Not measured | Baseline: Not measured<br>Post-impound: 0.285 (0.067-0.571) | Not measured | 206 ng/g dw (range 74-419) post-flooding | Finescale dace (Introduced) Concentrations at end of summer in 5 years: 0.26-0.55 ug/g ww whole body | No data | No data | No data | | | |
| **FLUDEX Medium Carbon** | Not measured | Baseline: Not measured<br>Post-impound: 0.272 (0.075-0.655) | Not measured | 158 ng/g dw (range71-264) post-flooding | Finescale dace (Introduced) Concentrations at end of summer in 5 years: 0.25-0.32 ug/g ww whole body | | | | | | Bodaly et al. (2004)<br>Hall et al. (2005) |
| **FLUDEX Low Carbon** | Not measured | Baseline: Not measured<br>Post-impound: 0.261 (0.079-0.924) | Not measured | 152 ng/g dw (range 70-215) post-flooding | Finescale dace (Introduced) Concentrations at end of summer in 5 years: 0.24-0.42 ug/g ww whole body | | | | | | |

All fish lengths are given as fork l
Chemical parameters are given as
NRL = Natural Regional Lakes
N/A = not applicable
* Nephelometric Turbidity Units (
** d.w.= dry weight; w.w.= wet w

## 5.3 Data limitations

As to be expected for a large data set that was assembled retrospectively, some of the information compiled in **Table 5.3** is incomplete and has limitations related to small sample size, questionable representation of general conditions, or timing issues, among others. These limitations have to be recognized, but they were not considered critical, given our aim of broadly comparing physical, chemical, and biological features of reservoirs to establish patterns indicative of mercury methylation and bioaccumulation potential. This exercise is intended to determine where the Site C Project 'ranks' with respect to its potential to increase fish mercury concentrations within the new reservoir.

Of all parameter groups considered in **Table 5.3**, mercury concentrations for large-bodied species, lake whitefish (*Coregonus clupeaformis*), northern pike (*Esox lucius*), and walleye (*Sander vitreus*) had the most complete and statistically sound sub-dataset. Nevertheless, there are several sources of uncertainty associated with the fish mercury levels reported in **Table 5.3**. Uncertainty exists in the exact level of peak concentrations, particularly if records started several years after reservoir creation and the frequency of subsequent yearly measurements was low (>2 year interval). Given that mercury concentration is dependent on size/age of fish, temporal comparisons of means are made based on length standardized data and the standard lengths should be similar for the different data sets. This was not the case for all studies considered in **Table 5.3**.

Uncertainty is also introduced if a group of natural lakes from the general project area is used to represent (missing) baseline data for a specific reservoir. The average mercury concentrations from the group of lakes may only approximate the actual concentrations in fish (and invertebrates) prior to impoundment. Also, if pre-project mercury concentrations are not known but estimated using unimpacted regional lakes, additional uncertainty is introduced to the estimate of the return times of peak concentrations to baseline levels. Furthermore, return times depend heavily on the criterion chosen to establish "return," and may provide different outcomes if a conservative statistical method (Bodaly et al. 2007) or a less stringent approach is used (Schetagne et al. 2003).

Despite these limitations, the weight-of-evidence of data was sufficient to determine whether or not mercury methylating conditions in the Peace River technical study area are favorable or not, and the expected magnitude of increase in mercury concentration in environmental media relative to what has been observed in other Canadian reservoirs.

## 5.4 Reservoir Comparison

### 5.4.1 Physical Factors

Baseline and maximum mercury concentrations, the ratio between peak and baseline concentrations (also referred to as the increase factor), and the return time to baseline or pre-impoundment levels vary greatly among most reservoirs and fish species considered in **Table 5.3**. In particular, most of the Québec reservoir within the La Grande Complex (i.e., excluding Caniapiscau) are characterized by high peak mercury concentrations (piscivores >2 to 5 mg/kg, non-piscivores: 0.5 to 1.3 mg/kg), large increase factors (3.4 to 10.7x baseline), and long return times of 20 to 30 years or more. Notwithstanding some methodological differences in calculations (e.g., different standard lengths compared to the Manitoba studies), Québec reservoir fish had substantially higher baseline mercury concentrations than fish from most Manitoba reservoirs (**Table 5.3**). Schetagne and Verdon (1999a) also concluded that fish from natural lakes in northern Québec have high mercury concentrations compared to other Canadian regions. In contrast, baseline mercury levels of lake whitefish, northern pike, lake trout (*Salvelinus namaycush*), and longnose sucker (*Catostomus catostomus*) from the two Labrador reservoirs (which have one common estimate of baseline values) were slightly higher than those of their conspecifics from Québec reservoirs.

The large increase factors and long return times of mercury concentrations from the Québec reservoirs were not only observed in species generally considered as piscivorous, but also in lake whitefish and longnose sucker, species that normally feed mainly on invertebrates and detritus (Scott and Crossman 1973). At reservoir La Grande 1 (LG1), these two species have increase factors of >10 and return times of 21 years or more, the highest values known from Canadian reservoirs and the largest increase factors for any fish species (**Table 5.3**).

In trying to relate the high peak mercury concentrations and long return times in fish from Québec reservoirs to some of their physical, chemical, and biological characteristics and to evaluate their potential role in promoting high and persistent fish mercury levels, a few parameter stand out that separate the Québec reservoirs from most, if not all other reservoirs listed in **Table 5.3**. Physically, all Québec reservoirs, except for LG1 are spatially large (>1,000 km), have a very high ratio of post-flood reservoir area relative to waterbody area prior to flooding (3.5 to 13.8 times), and have relatively long residence times (4 to 26 months).

Predictive models based on data from northern Manitoba lakes and reservoirs tested on a large set of Canadian reservoirs have shown that the relative proportion of newly flooded reservoir area is a good predictor of mercury concentrations in fish (Johnston et al. 1991). Hydraulic residence time is also a good indicator of the magnitude and temporal extent that newly created methylmercury is available to be accumulated by biota. Reservoirs with longer residence time tend to have higher mercury accumulation rates than reservoirs with a short residence time. Schetagne et al. (2003) postulated that the area of flooded land (A) is the main driver of methylmercury generation while outflow (V) is the primary dilution and export mechanism of mercury in water. These authors found that the A/V (measured as annual mean) ratio was a reasonable predictor of maximum fish mercury concentrations. Harris and Hutchinson (2007) noted that bacterial demethylation, photochemical degradation, and sedimentation are other known processes that remove mercury from waters and suggested that these become more important to reservoir mercury dynamics with increasing water resident time. Using a model approach based on data from four Québec reservoirs and the Smallwood reservoir in Labrador, Harris and Hutchinson (2007, 2011) found that a combination of flooded area, outflow and total reservoir area matched observations for lake whitefish and northern pike, but over-predicted observed increase factors for the Caniapiscau reservoir. Note that this regression approach was one of the lines of evidence used to determine the likely magnitude of increase in fish mercury concentrations within the Site C reservoir (Volume 2, Appendix J, RESMERC Part 3) and ultimately combined to inform the Human Health Risk Assessment (Volume 2, Appendix J, HHRA, Part 2).

In addition, two additional factors not included in **Table 5.3** may be partially responsible for high peak mercury levels and/or prolonged return times of mercury concentrations in Québec reservoir fish. First, compared to most of the other projects listed in **Table 5.3** which, except for Williston Reservoir (see below), have a relative small operating range (e.g., Manitoba reservoirs: <2 m), water levels in the Québec reservoirs fluctuate annually by several meters. Except for LG1, the mean annual drawdown of the other four reservoirs considered in **Table 5.3** ranges from 2.1 (Caniapiscau) to 5.5 m (LG3), with maximum drawdown ranging between 4.0 and 12.9 m (Schetagne et al. 2003). Although the mechanism or the relationship between drawdown and fish mercury is not well understood, it may be that increased erosion caused by water level fluctuations mobilizes organic matter from shoreline areas and promotes mercury methylation. Also, one theory is that drying promotes the oxidation of reduced sulphur species (e.g. sulphides), which upon re-wetting, are available to sulphate reducing bacteria and can help promote methylation.

Several British Columbia reservoirs also have a very high annual drawdown including Williston Reservoir (up to 15 m), Carpenter Reservoir (>40 m), Arrow Lake (20 m) and Mica (>40 m) and it is unknown what effect this has on sustaining elevated fish mercury concentrations. This phenomenon has been implicated in sustaining elevated mercury concentrations in bull trout from Williston Reservoir (Baker et al. 2002) and Carpenter Reservoir BC (Azimuth 2009) as levels are higher in these reservoirs than in lakes and reservoirs with low drawdown.

In summary, reservoir area, ratio of flooded area to original area, and water residence time appear to be the important drivers or determining physical factors dictating the ultimate magnitude of fish mercury concentration increase above baseline and the temporal extent that elevated concentrations persist. Magnitude of reservoir drawdown could also be very important, although there is too little research that has examined this aspect of reservoir management. These physical factors were present in the Québec reservoirs LG-2, LG-3, Opinaca and Caniapiscau, in ELARP (Ontario), and in Williston Reservoir (BC).

### 5.4.2 Reservoir Comparison – Chemical Factors

Several chemical factors have also been demonstrated to be correlated with increases in fish mercury concentrations including pH (negative), DOC/TOC (positive), sulphate (positive) and labile carbon / carbon biomass in soil (positive). Few studies have attempted to determine the fraction of carbon in flooded soils that is labile (i.e., easily available, most amenable to contributing to methylation) relative to refractory (i.e., less available), although a good surrogate for this is the amount (% or ha) of flooded wetland. Relationships between chemical factors in water and soil/sediment and relationships with mercury concentrations in environmental media are explored here.

Water pH is an influential factor that favors mercury methylation (Miskimmin et al. 1992), although the mechanism by which this works is not clear. With a mean annual pH of 5.8 to 6.7, the waters that filled Québec reservoirs are more acidic than those for other full-sized reservoirs that are circumneutral to slightly alkaline in pH (**Table 5.3**). Fish from slightly acidic water, has elevated mercury concentrations compared to fish from circumneutral lakes (Grieb et al. 1990; Wiener et al. 1990; Greenfield et al. 2001). Québec reservoirs also likely had relatively larger pools of carbon and total mercury in the soils of what became flooded areas. Unfortunately, except for the inundation zone proposed for the Site C reservoir, no specific data on these carbon and mercury pools are available for other full-sized reservoirs. The only other comparison data for total mercury come from the three experimental upland reservoirs created as part of the FLUDEX Project. In those three reservoirs total mercury pools prior to flooding were approximately half the size compared to unflooded podzols from the La Grande region (**Table 5.3**). With a range of 7.3 to 23 kg C/$m^2$ for total carbon (including 0.53 kg C/$m^2$ ‘labile’ carbon), the La Grande area soils have high carbon pools, at least when compared with the general literature estimates of 2.7 kg C/$m^2$ total carbon (including 0.25 kg C/$m^2$ ‘labile’ carbon) for the Muskrat Falls and Gull Island reservoirs (**Table 5.3**) and the site-specific estimate of Site C area soils (2 to 4.5 kg C/$m^2$; Azimuth 2011). The total soil carbon pool prior to the flooding of the La Grande area reservoirs also was likely substantially higher than the respective pools for the three FLUDEX upland reservoirs, including the “high” carbon site. Although labile carbon (the fraction that is more readily metabolized and used as an energy source by bacteria) was present in larger quantities in the FLUDEX soils than in the La Grande area soils, this comparison is potentially confounded by differences between the two studies in the organic materials included as ’labile’ carbon (**Table 5.3**). Of all projects listed in **Table 5.3,** only the experimentally flooded ELARP reservoir almost certainly had a higher soil carbon pool prior to impoundment than the Québec reservoirs, partly by design. This is not surprising, because the ELARP reservoir was an existing wetland before it was further inundated. As the Québec and ELA data for different soil types from the same geographical area indicate, wetland and peatland soils store substantially more carbon than podzols (Grondin et al. 1995) and other soil types (Kelly et al. 1997; Bodaly et al. 2004). Just for this reason, the physical measure of percentage wetland/peatland area of total flooded area in **Table 5.3** may be used as an approximate measure of the amount of carbon stored in reservoir soils prior to flooding, and provides an indication of its potential for promoting bacterial methylation, particular if direct measures of carbon (and mercury) pools are unavailable.

Methylmercury concentrations are particularly high in peat and many wetland plants, and equilibrate almost immediately with methylmercury in water (Kelly et al. 1997). More importantly, because of their biogeochemical properties (e.g., low pH, low dissolved oxygen, high dissolved organic carbon) that promote elevated activities of methylating bacteria, peatlands efficiently methylate mercury and produce methylmercury at approximately twice the rate than flooded uplands while providing a larger, longer term supply of carbon for decomposition and methylmercury production (Driscoll et al. 2007; Hall et al. 2005; St. Louis et al. 2004; Bodaly et al. 2004).

### 5.4.3 Reservoir Comparison – Ecological Factors

Several of the more important ecological factors related to reservoir creation that influence mercury accumulation by biota are diet, shift in dietary preference, change in food web structure or length, altered productivity (e.g., initially greater, then lower), change in trophic status, nutrient deprivation by upstream reservoirs and downstream effects.

#### 5.4.3.1 Dietary Exposure

Methylmercury concentrations increase at all levels of the food web after flooding, ultimately culminating with highest concentrations in fish, especially in piscivorous species. The magnitude and duration of increase in fish mercury concentration is directly related to physical/chemical factors that determine the relative degree of methylation (e.g., Herrin et al. 1998), and then by ecological relationships (e.g., Gorski et al. 2003) that may change after reservoir creation.

All fish species considered in **Table 5.3** feed on invertebrates throughout or for at least part of their life, while some species may predominantly consume other fish. Although empirical data were limiting, except from experimental reservoirs (e.g., Paterson et al. 1998), differences in invertebrate and forage fish mercury concentrations between pre- and post-flooding conditions appear to have lower increase factors than are observed in large bodied fish from the same reservoirs. Because of the general lack of baseline data for forage fish from full-sized reservoirs in **Table 5.3**, only lake chub (*Couesius plumbeus*) from the Caniapiscau reservoir provided a comparison between post-impoundment mercury concentrations and estimated (using a natural lake) pre-impoundment levels. The increase factor for lake chub is 1.9, lower than the increase factors of 2.9 and 3.6 for lake trout and northern pike, respectively, from the same reservoir (Schetagne et al. 2003). This is likely related to the fact that chub feed lower in the food web than trout or pike and are ingesting lower mercury food. However, the near doubling in mercury concentrations of lake chub is very similar to the increase factor of finescale dace (*Phoxinus neogaeus*) from the experimental ELARP reservoir (Bodaly and Fudge 1999). Unfortunately, no comparisons with piscivorous fish species were available for the ELARP reservoir.

More data were available for invertebrates than for forage fish. Total and methylmercury concentrations of zooplankton measured 11 to 14 years after flooding of Caniapiscau reservoir fell within the range of baseline concentrations obtained from 13 natural lakes in northern Québec (Tremblay 1999). Likewise, compared to the very low mercury levels in zooplankton from three natural lakes near LG1, LG2, and Opinaca reservoir (Tremblay 1999), total mercury concentrations of zooplankton from Opinaca reservoir 12 to 14 years after impoundment were similar and methylmercury levels were only approximately twice as high (**Table 5.3**). In contrast, post-impoundment (14 to 16 years) zooplankton concentrations from the LG2 reservoir were 3 and 10 times higher for total mercury and methylmercury, respectively, compared to the three natural lakes in the area (**Table 5.3**). Benthic invertebrates from LG2, including data for different feeding groups, also had mercury concentrations that fell into the range of background, in northern Québec (**Table 5.3**). The same pattern was observed for benthos taxa from Caniapiscau Reservoir.

Although invertebrate mercury concentrations from the experimentally flooded ELARP reservoir showed a clear flooding signal, the magnitude of increase in mercury concentrations differed from that of ELARP forage fish. Methylmercury and total mercury concentrations of zooplankton increased ten-fold and five-fold, respectively, above baseline (Paterson et al. 1998). Methylmercury concentrations increased three-fold in predatory insects, and less than two-fold in collectors and shredders (Hall et al. 1998). Total mercury concentrations in fine-scale dace increased two- to three-fold (Bodaly and Fudge 1999), a reflection of dietary preference. The magnitude of increase in invertebrates should not necessarily be expected in fish because of the complexities of bioenergetics, growth, bioaccumulation and other factors.

Lack of congruence in the response of mercury concentrations to reservoir flooding between invertebrates and fish is consistent with diet being the main exposure pathway to mercury. Zooplankton and macroinvertebrates represent several taxonomic groups with various feeding modes and diets and may include two to three different trophic levels (Resh and Rosenberg 1984). Although mercury biomagnification within the pelagic invertebrate community may be limited or masked by growth dilution (Paterson et al. 1998), it is well established that food mercury content (Hall et al. 1997) and a lengthening of the food chain (Cabana et al. 1994; Kidd et al. 1995) increase mercury concentration in fish species at higher trophic levels in addition to dietary plasticity and ontogenetic diet shifts (Jansen et al. 2003; Beaudoin et al. 1999; Jansen and MacKay 1992; Schetagne et al. 2003). All these ecological factors have the potential to result in complex trophic interactions within the invertebrate and fish communities. Therefore, the range in mercury bioconcentration factors observed in zooplankton and/or benthos does not necessarily indicate that a range should be observed at the invertebrate-fish intersection of the food chain.

#### 5.4.3.2 Shift in Dietary Preference

As indicated in the previous section, a potential impact of reservoir creation on fish mercury levels is a dietary shift towards feeding on organisms higher in the food web and/or increasing the length of the food web by creating conditions that are favourable for other intermediate species to flourish (e.g., invertebrate species or a forage fish species). Increasing the length of the food web or feeding at higher trophic levels has been implicated with causing higher body burden mercury concentrations in sport fish such as northern pike, walleye and lake trout.

An example of this is highlighted by the La Grande Complex, where diet shifts and changes in trophic position have occurred in at least three species. Northern pike, particularly from the LG2 reservoir, have developed a more persistent cannibalistic feeding preference than fish in natural lakes in the La Grande Area (Schetagne et al. 2003). Feeding on higher mercury content food has essentially introduced another trophic level into the northern pike food web, a phenomenon that is known to increase mercury concentration in carnivorous species (Cabana et al. 1994; Kidd et al. 1995). Schetagne et al. (2003) speculated that mercury levels in pike from the LG2 reservoir will remain high as long as this so-called “superpredator” behaviour persists. With a mercury concentration of approximately 1.7 ppm (Therrien and Schetagne 2009) concentrations in pike at LG2 have not returned to baseline levels 29 years after first impoundment (**Table 5.3**), one of the longest return times of any fish species from Canadian reservoirs.

Lake whitefish and longnose sucker have also experienced dietary shifts at La Grande. Whitefish, particularly large fish (>450 mm), have been found to feed primarily on small fish below the Robert-Bourassa GS at LG2 (Brouard and Doyon 1991, cited in Schetagne et al. 2003). This dietary switch from a normal diet of mainly invertebrates to one dominated by fish seems to occur only immediately downstream of the GS where mainly cisco (*Coregonus artedii*) have been injured or stunned after turbine passage (Brouard et al. 1994). Mercury concentrations of the large piscivorous whitefish (which were not included when calculating the means listed in **Table 5.3** for LG2) ranged from 2.5 to 5.5 ppm, levels similar to the mean concentration of northern pike when this species reached peak levels within the reservoir, and much higher than the concentrations in whitefish from regional reference lakes (**Table 5.3**). Similarly high mercury concentrations have been observed in lake whitefish from Winokapau Lake, 65 km downstream of the Churchill Falls GS that impounds the Smallwood Reservoir in Labrador and Newfoundland. Six years after reservoir creation, large whitefish (350 mm) in Winokapau Lake had mercury concentrations that occasionally exceeded 2 ppm (Harris and Hutchinson 2007) and the authors hypothesized that these whitefish had fed on turbine-passed fish. Both studies at LG2 and Churchill Falls also found substantially elevated mercury concentrations in longnose sucker downstream of the respective GSs that were interpreted as a result of piscivory in at least some individuals (Schetagne et al. 2003; Harris and Hutchinson 2007).

#### 5.4.3.3 Oligotrophication of Reservoirs

One phenomenon that has been observed in BC that has not been well documented in reservoirs elsewhere in Canada is oligotrophication. Increasing nutrient deprivation and lower productivity appears to be mainly driven by the higher drawdown magnitude in BC reservoirs (>10 m) than other Canadian reservoirs that results in increased turbidity and less productive epilimnetic and littoral zone habitat. For example, Stockner et al. (2005) has placed Williston Reservoir within ultra-oligotrophic range of productivity because of its depauperate microbial, phytoplankton and zooplankton communities and low fish biomass. Very low productivity is caused by a combination of nutrient limitation, low light penetration because of inflow of highly turbid water from tributaries, high winds and sediment introduced from shoreline erosion. Loading models of Williston shortly after impoundment in 1968 revealed that the reservoir was initially moderately productive from nutrient loading as a result of inundation of nearly 1,770 $km^2$ of forest (BC Hydro 2011). The system lost nutrients due to sedimentation and outflow, but also due to scarcity of littoral zone habitat due to drawdown (10 to >15 m annually) and ice scour. Over the last 20 to 30 years, the system has become increasingly oligotrophic due to increased turbidity, reduced light penetration, cold water and ultimately, lost biogenic productive capacity at all trophic levels (Stockner et al. 2005). The steep, V-shaped, mountainous nature of BC reservoirs (e.g., Mica, Arrow, Revelstoke) with high drawdown and large profundal habitat area contributes to reduced productivity relative to eastern reservoirs and is a key difference in mercury dynamics between these very different geographic regions. Central and eastern Canadian reservoirs at similar northern latitudes are shallower and warmer in summer and have overall higher productivity, factors that are more favourable to mercury methylation. Certainly, the influence of a large oligotrophic body of water as Williston directly upstream of the Site C Project will constrict input of nutrients and drifting biota to the new reservoir.

#### 5.4.3.4 Downstream Changes

Increased mercury concentration in fish has been observed downstream of some newly created reservoirs, but not all. The main reason for this is the dietary switch that some individuals make from invertebrates to fish that have been stunned, injured, or killed as they pass through the turbines. Whether this occurs or not at Site C depends on the downstream environment, the degree to which fish are passed downstream, the mercury concentration in these fish and whether individuals make the ‘switch’ to a higher dietary source of mercury. For example, LG1 reservoir was constructed 60 km below a large, lacustrine, long residence reservoir known as LG2 or Robert Bourassa that was constructed only 12 years earlier. Mercury in fish from LG2 reservoir increased at least 5x above baseline. Prior to creation of LG2 mercury levels in LG1 fish were low, but increased 3x post-flood. This was attributed to downstream export of high mercury fish from within LG2 Reservoir (Schetagne et al. 2003), that were fed on preferentially by fish below the LG2 GS in the newly formed LG1 Reservoir. Increased fish mercury concentrations were observed in piscivorous species like burbot and walleye, but also in non-piscivorous species like whitefish and sucker.

According to Schetagne and Verdon (1999b) and Schetagne et al. (2003), the presence of a large, slow-flowing body of water upstream of a reservoir greatly reduces mercury transfer to downstream fish. This is because of sedimentation of suspended particular matter (a main source of water-born methylmercury; Kelly et al. 1997) and local predation of fish food organisms with potentially elevated mercury concentrations, such as zooplankton, macroinvertebrates, and forage fish. The presence of an oligotrophic upstream reservoir such as Williston (and a lesser extent, Dinosaur) acts as sinks for suspended solids and nutrients and limits downstream export of biota (e.g., drifting invertebrates) and fish, which may have a great influence on mercury dynamics in the Site C reservoir.

## 5.5 Implications for Fish Mercury Concentrations at Site C

The inter-reservoir comparisons described above and summarized in **Table 5.3** have identified a number of physical, chemical, and biological parameters that are known to be correlated with the magnitude and duration of elevation in fish mercury concentrations beyond baseline or reference waterbodies in new reservoirs. To place the Site C Project into context with the spectrum of results observed at other reservoirs across Canada, the large matrix was distilled down, to present results of a few key parameters that are summarized in **Table 5.4**.

Based on this review of other Canadian reservoirs, the key physical drivers of mercury methylation potential and magnitude of increase in mercury are:

- Total reservoir area – Larger reservoirs tend to have fish with higher mercury concentrations and take longer to return to baseline or background (relative to nearby lakes).
- Ratio of total reservoir area (original area) – The higher the ratio, the greater the mercury methylation.
- Water residence time – Fish from longer residence time reservoirs have higher mercury concentrations that persist for a longer time period.

The key chemical drivers are:

- Slightly acidic pH (<6.5) is associated with higher mercury concentrations in fish.
- Higher carbon (TOC/DOC) concentrations in water (> 5 mg/L) are weakly positively correlated with the magnitude of increase in fish mercury.
- Labile carbon, best represented by the amount (% of total and/or hectares) of wetland within the reservoir.

The key ecological factors are:

- Lower trophic level mercury concentration – Lakes / rivers with higher baseline mercury and especially methylmercury concentrations in zooplankton and benthos typically see higher increases post-flood and contribute greatly to higher rates of bioaccumulation and biomagnification in fish.
- Reservoir productivity – Generally, larger reservoirs with more *in situ* nutrients, and nutrient inputs from upstream of tributary flow and established lower trophic level and fish communities tend to have greater biomass production and higher mercury methylation potential, and consequently, higher mercury concentrations in biota.

Finally, each of the reservoirs examined in **Table 5.3** were placed into one of two categories, according to the magnitude of increase in fish mercury concentration relative to baseline (provided data are available), or reference data (i.e., nearby waterbodies not influenced by flooding). An increase value of three times above the baseline was used as the cutoff. The value of 3x baseline is approximately half the increase in what is seen in most ‘worst-case’ scenario increase reservoirs (an increase of ~seven times) and higher than many reservoirs where a doubling in concentration was observed. A 3x increase factor is conservative, yet high enough that it is statistically distinguishable from baseline and the return to baseline can be measured with greater precision.

In the simplified matrix (**Table 5.4**) the range of key physical, chemical and ecological parameters are presented and characterized for both low and high magnitude of increase in fish mercury concentration. In the far right column, the proposed Site C Project was determined to fall either within the **LOW** or **HIGH** increase category, relative to other reservoirs across Canada in a weight-of-evidence approach. This judgment was based on existing baseline conditions for the Site C Project area as described within this document and from forecast physical, chemical or ecological conditions where available including temperature (Appendix H), sediment (Appendix I), water quality (Appendix P Part 2, Aquatic Productivity (Appendix P Parts 1 Biological Assemblages and Part 3, Future Conditions), all within Volume 2 of the EIS.

**Table 5.4 Simplified Canadian Reservoirs Comparison Matrix - Low vs High Magnitude Fish Mercury Increase Parameters**

| Reservoir Characteristics | Low Magnitude Increase Reservoirs (Fish Mercury <3 x Baseline) | High Magnitude Increase Reservoirs (Fish Mercury > 3 x Baseline) | Predicted Site C Result |
|---|---|---|---|
| **Magnitude of Fish Mercury Increase above Baseline** | Muskrat Falls, Gull Island (Nfld/Lab); Limestone, Long Spruce, Wuskwatim, Southern Indian Lake (all MB) for some fish species | LG-1, LG-2, LG-3, Opinaca, Caniapiscau Québec; Southern Indian Lake, MB (for some species), Williston BC | |
| **Physical Parameters** | | | |
| *Total Reservoir Area* | Typically less than 200 $km^{2,}$ ranging from 28 (Limestone) - 200 $km^2$ (Gull Island) for all reservoirs | Typically very large, with most exceeding 2000 km2 except Opinaca (1040 km2), Williston (1779 km2) | Site C predicted area = 9.3 $km^2$ and falls into **LOW** increase category |
| *Original:Flooded Area* | The ratio was typically less than 2 at Muskrat (1.5) and Gull (1.7) Nfld/Lab and Limestone (1.3), Long Spruce (1.9) and Wuskwatim (1.5) MB | Typically a ratio well in excess of 2 at LG1 (2.3), LG2 (13.8), LG3 (9.9), Opinaca (3.5), Caniapiscau (5), Williston (22), with a lower ratio at SIL (1.2) | Site C predicted ratio is 2.3 and would fall into the upper end of the **LOW** increase category; Although similar to LG1, the influence of LG2 on Hg in LG1 fish was anomalous |
| *Water Residence Time* | In the order of days and typically less than one month in Muskrat (7d), Gull (26d), Limestone (5d), Long Spruce (10 d) and ELA (<5d) | Residence time much longer, typically greater than 5 months including LG2 (7m), LG3 (11m), Opinaca (3.8m), Caniapiscau (26m) and SIL (8m) | With a water residence time of 23 d, Site C falls into the **LOW** category |
| **Chemical Parameters** | | | |
| *pH* | Usually pH of 7.5 or greater, especially in Manitoba reservoirs (7.5 - 8.5) and Williston (8.5); approximately pH 7 in Gull/Muskrat | A pH of <6.5 for all reservoirs including LG1 (6.5), LG2 (6.2), LG3 (<6.5), Caniapiscau (5.8 - 6.4) and Opinaca (5.9 - 6.3) | Peace River has pH of 7.8 - 8.6 and not predicted to change; clearly placing Site C in the **LOW** increase category |
| *TOC / DOC* | TOC/DOC concentrations are 2.6 - 4.6 mg/L in Muskrat/Gull; 8 - 12 mg/L in MB; 2 - 3 mg/L in Williston | TOC tends to be slightly higher, averaging 6.4 mg/L in LG1, 9-29 mg/L in LG2, 7-10 mg/L in LG3, 4-6 mg/L in Caniapiscau and 7-10 mg/L in Opinaca | TOC/DOC slightly higher in high increase reservoirs. Influence of low TOC water from upstream will likely place Site C in **LOW** increase category, with some uncertainty |

| | | | |
|---|---|---|---|
| *Labile Carbon / %Wetland* | There are few good data for most reservoirs. However, the trend is for % wetland to be 3% or less including Williston (<1%) and Site C (<2%); Few data on labile carbon or biomass except for Nfld/Lab (2.7 kg/m2) and Site C (5 kg/m$^2$) | PQ reservoirs have a high percentage of flooded wetland: LG1 and LG2 (5%), LG3 (10%), Caniapiscau (7%) and Opinaca (16%); No data for Williston; SIL in MB was also high >5% . Carbon pool was also high with 16 - 23 kg/m$^2$ in peat soils, 9 - 42 kg/m$^2$ in wetlands and 7 kg/m$^2$ in forest soil | Site C has a low carbon biomass relative to other reservoirs for which this is known and a low percentage of wetland (<2%), placing Site C in the **LOW** increase category |
| **Ecological Parameters** | | | |
| *THg/MeHg in Lower Trophic Level Biota* | Pre-impoundment THg in Gull/Muskrat Nfld zooplankton 0.07 - 0.26 ppm THg and 0.002 - 0.07 ppm MeHg. At Williston post-impoundment (2000, 2001) THg in zooplankton is 0.06 - 0.18 and 0.03 - 0.05 ppm of which 35% is MeHg; In benthos THg is 0.2 - 0.57 and 0.15 - 0.28 ppm of which 20% is MeHg.<br>Peace River (2011) baseline benthos is 0.07 ppm THg in zooplankton and 0.016 ppm THg in benthos of which approximately 10% is MeHg | The best data sets are for PQ reservoirs; values are on a dw basis. THg in zooplankton (baseline) is 0.03 - 0.57 ppm; 0.03 - 0.51 MeHg; Post-flood range 0.45 - 0.67 THg and 0.45 - 0.82 MeHg. In benthos, baseline THg ranges from 0.28 - 0.45 ppm and 0.25 - 0.8 ppm depending on taxa; MeHg 0.2 - 0.6 and 0.02 - 0.15 ppm post-flood; In SIL post-flood zooplankton was 0.3 - 3.0 and benthos 0.1 - 3.5 depending on taxa and organism size | Peace River baseline THg and MeHg fall into lower range of zooplankton and benthos concentrations. Percent MeHg of THg is also low (<15%). Low baseline lower trophic level Hg concentrations are consistent with a low magnitude increase in fish Hg and place Site C in the **LOW** increase category |
| *Reservoir Productivity Features* | Tend to be run-of-river, have upstream reservoirs that limit nutrient / biota introductions, limited tributary/river inflow, lower carbon biomass and limited connectivity with larger waterbodies. Lack of nutrients and high turnover limit reservoir productivity and thus Hg bioaccumulation | Tend to be spatially large, have higher nutrient inputs, greater connectivity to tributaries and lakes, longer residence time (lower nutrient export), and are more productive, even supporting commercial fisheries (e.g., SIL) | Site C is a run-of-river reservoir receiving very low nutrient water from upstream with limited connectivity and small tributary stream and nutrient inputs. Its low productivity status is consistent with **LOW** magnitude fish Hg increases. |
| THg = total mercury; MeHg = methylmercury; dw = dry weight; MB = Manitoba, PQ = Québec | | | |

Reservoirs where fish mercury concentrations increased by less than 3x above baseline were Muskrat Falls and Gull Island in Newfoundland and Labrador, and Limestone, Long Spruce and Wuskwatim reservoirs in Manitoba. Southern Indian Lake MB straddled this cutoff with some species being more (northern pike, longnose sucker) or less (lake whitefish, chub, lake trout) than a 3x increase. Thus, Southern Indian Lake was included in both categories. Reservoirs where fish mercury concentrations increased by 3x or greater above background, were LG-1, LG-2 (Robert Bourassa), LG-3, Opinaca and Caniapiscau in Québec, Southern Indian Lake MB and Williston Reservoir in BC (**Table 5.3**).

The experimental reservoirs (FLUDEX and ELARP) were not categorized within the simplified matrix because there were no data or insufficient data on mercury in invertebrates and large fish to make use of the results and were excluded from this exercise. Fish were not present in the FLUDEX reservoir.

Note that the relative increase in mercury concentration above baseline or background does not necessarily imply that greater or lesser risks would be posed to people that consume fish. For example, a tripling of a low concentration (e.g., 3 x 0.1 ppm mercury = 0.3 ppm) would be less than a doubling of a moderate mercury concentration (e.g., 2 x 0.5 ppm mercury = 1.0 ppm). Ultimately, it is the product of mercury concentration, meal size and consumption frequency that provides the 'dose' to the consumer and it is this perspective that was taken by the Human Health Risk Assessment (Volume 2, Appendix J Part 2).

Among the physical, chemical and ecological factors primarily responsible for mercury methylation in new reservoirs, the Site C reservoir was clearly classified as having a strong likelihood of producing a less than 3x increase in fish mercury concentrations for all parameters that were considered (**Table 5.4**). In none of the parameters considered, did the Site C Project fall above a 3x increase.

- Physically, the Site C reservoir is considered a run-of-river reservoir that has a relatively low flooded area (9.3 $km^2$) and low-flooded-area to original-area ratio (2.3), much less than high mercury magnitude increase reservoirs that have surface areas in excess of 200 $km^2$ and flood ratios greater than 3.5 and up to 13. Water residence time within Site C is short (23 days), rather than several months to years for high increase reservoirs. All physical parameters indicate that an increase of mercury in fish of no more than 3x baseline is likely.
- Chemically, Peace River water within the Site C reservoir is expected to be slightly alkaline (pH 8) with low TOC (<3 mg/L) and nutrient concentrations. Inflow to the proposed Site C reservoir will continue to be dominated by ultra-oligotrophic water

from upstream (Dinosaur, Williston). This is because of the short residence time of water within the Site C reservoir and the low tributary input volume relative to reservoir volume of Site C. Furthermore, the percent wetland as a generator of DOC and methylmercury within the zone proposed for inundation at Site C is low, less than 2% of the total area flooded and less than the ~10% or more that is typically associated with high magnitude increase reservoirs.

- Ecologically, baseline mercury concentrations in lower trophic levels (zooplankton, benthos), as well as in fish in the Peace River are very low – much lower than in biota from central and eastern Canadian reservoirs. Furthermore, the proportion of mercury in the methyl form in lower trophic level biota is also low, typically less than 20% of the total; whereas, in most other reservoirs, the proportion of methyl relative to total is greater than 30%. Site C reservoir productivity is expected to be low, given the nutrient poor water received from Williston Reservoir, lack of connectivity to upstream nutrient and biota sources (e.g., drifting aquatic invertebrates) and no meaningful tributary input until at least the Halfway River (EIS Sections 11.4, 12.0). Combined, these factors place the Site C reservoir into the low increase category relative to other reservoirs with greater productivity, connectivity and nutrient supply where fish mercury increases have been greater than 3x baseline.

## 5.6 Summary

Based on the above evaluations none of the parameters that are associated with large increases in fish Hg concentrations observed in other Canadian reservoirs are projected to be present within the proposed Site C reservoir. In particular, these include low TOC and nutrients in water, alkaline pH, and presence of an oligotrophic upstream reservoir, low temperature and high oxygen, low baseline mercury concentration in water and biota, small increase in reservoir area relative to river area, small area of flooded wetland and short residence time in this run-of-the-river reservoir.

In summary, given the expected or predicted physical, chemical and ecological conditions for the proposed Site C reservoir, there is a low potential for mercury methylation and bioaccumulation of mercury in all aquatic environmental media.

Authorship:

Senior Author
Randy Baker, M.Sc., R.P.Bio. Principal
Azimuth Consulting Group

Contributors:

Contributor
Dr. Wolfgang Jansen, Ph.D.
North/South Consultants Winnipeg

Contributor
Dr. Ralph Turner, Ph.D.
Associate, Azimuth Consulting Group

Contributor
Dr. Drew Bodaly. Ph.D.
Independent Consultant

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Note that this section also includes all references cited in **Table 5.3**, which may or may not be referenced in the text.

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# SITE C CLEAN ENERGY PROJECT

## VOLUME 2 APPENDIX J, PART 2

## TECHNICAL DATA REPORT:
## HUMAN HEALTH RISK ASSESSMENT OF
## METHYLMERCURY IN FISH

***Prepared for:***

BC Hydro Power and Authority
333 Dunsmuir Street
Vancouver, BC
V6B 5R3

***Prepared by:***

Azimuth Consulting Group
218-2902 West Broadway
Vancouver, BC
V6K 2G8
Tel: (604) 730-1220  Fax: (604) 739-8511

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# SITE C CLEAN ENERGY PROJECT

## VOLUME 2 APPENDIX J, PART 2

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***Prepared for BC Hydro Power and Authority***

***Prepared by Azimuth Consulting Group Partnership***

December 2012

Lead Author: Norm Healey, Toxicologist and Partner, Azimuth Consulting Group

Section Contributors: Randy Baker, Partner, Azimuth Consulting Group

## AUTHORSHIP

Norm Healey. ....................................................... Toxicologist and Partner, Azimuth Consulting Group

Randy Baker. ....................................................................... Partner, Azimuth Consulting Group

## EXECUTIVE SUMMARY

This report presents results of a Human Health Risk Assessment to address potential changes in fish methylmercury (MeHg) concentrations following construction and operation of the proposed Site C Clean Energy Project (the Project). The most popular fish species targeted by First Nations and sport fishers were evaluated including, rainbow trout, bull trout and lake trout within the Peace River upstream of the proposed Site C dam location and walleye, goldeye and northern pike downstream of this location, into Alberta.

When organic soils are inundated with water, some of the naturally occurring inorganic mercury (Hg) is converted by bacteria to a form of organic mercury called methylmercury. Methylmercury behaves differently than other forms of Hg in that it becomes increasingly concentrated by aquatic organisms (bioaccumulation) and becomes increasingly concentrated with progressive steps up the aquatic food web (biomagnification). Predatory fish (e.g., bull trout, lake trout) and some fish-eating birds (loons) and mammals (otters), accumulate the highest concentrations of MeHg in the aquatic food web. In fish, nearly all of the Hg measured in muscle tissue is in the form of MeHg.

Current or baseline concentrations of methylmercury in fish from BC, and the technical study area of the Project in particular, are among the lowest concentrations observed relative to all other Canadian lakes, reservoirs and rivers and are predicted to remain low relative to other hydroelectric reservoirs elsewhere in Canada.

The Site C technical study area is defined as the area of the Peace River impounded by the proposed Site C dam upstream to the Peace Canyon Dam (the Site C reservoir) and downstream from the Site C dam, extending to Many Islands, AB. This is defined as the furthest downstream extent that the vast majority of species and individuals routinely move between and would potentially be exposed to MeHg exported from the proposed Site C reservoir, mostly within the tissue of invertebrates and fish entrained out of the reservoir through the turbines and/or spillway.

Creation of the proposed Site C reservoir would cause MeHg concentrations in fish to temporarily increase, before slowly returning to baseline concentrations. Three lines of evidence were evaluated to determine how fish mercury concentrations may change over time within the proposed Site C reservoir. These were: a detailed comparison between Site C and many other Canadian reservoirs within the Mercury Technical Synthesis Report for Site C (Volume 2, Appendix J, Part 1); linear regression modeling and detailed mechanistic modeling (RESMERC), both within the Mercury Modeling Report (Appendix V2J Part 3). The integrated assessment of these lines of evidence considered by the Site C EIS Section 11.9 Methylmercury, determined that MeHg concentrations in fish within the proposed Site C reservoir would peak at between three and four times baseline concentrations, depending

on the species. This peak would occur between 5 – 8 years after reservoir creation, before slowly returning to baseline over the next 10 – 15 years. Downstream of the proposed Site C reservoir, Section 11.9 predicted a possible doubling of fish MeHg concentrations that would persist for 5 – 8 years, again, depending on the species.

All Canadians are exposed to methylmercury in their environment and the greatest source of exposure to MeHg comes from eating fish. To protect consumers from an excess of dietary methylmercury, Health Canada has defined a ‘provisional tolerable daily intake’ or pTDI for methylmercury. The pTDI is the amount of methylmercury that a person can ingest without risk of adverse health effects. All fish contain methylmercury, with higher concentrations found in large, longer-lived predatory species such as bull trout and lake trout. A person’s methylmercury exposure depends on how frequently fish are consumed, the serving size, species, age and size of fish consumed. Risk is also relative to a person’s age and gender because the developing nervous system of a child is more susceptible to the effects of methylmercury than that of an adult.

The most commonly consumed type of freshwater fish reported by participants in the BC First Nations Food, Nutrition, and Environment Study and First Nations communities in closest proximity to the Project, and participants in the Duncan and Horse Lake First Nation’s Country Food Harvest Consumption Survey, was ‘trout’. Although not specifically broken down, the most commonly consumed species of trout are rainbow trout, bull trout and lake trout. Bull trout are emphasized in this HHRA because, of all trout species in the technical project area, bull trout have the highest baseline mercury concentration.

Based on Health Canada guidance and using current or baseline mercury concentrations in commonly consumed fish species from the technical study area, all species including bull trout, rainbow trout, lake trout, mountain whitefish, and burbot can be consumed regularly (i.e., several times per week), even by the most sensitive age group (i.e., toddlers less than 5 years old) without exceeding Health Canada’s pTDI for methylmercury.

At peak post-impoundment mercury concentrations within the proposed reservoir, bull trout can be consumed by women of child bearing age twice per week, all other adults five times per week, and toddlers once per week without exceeding Health Canada’s pTDI for methylmercury. Mercury levels in lake trout are similar to those in bull trout. Rainbow trout and mountain whitefish have lower mercury concentrations than bull trout and lake trout. At peak post-impoundment mercury concentrations, women of child bearing age can consume at least three servings of rainbow trout per week, other adults eight times per week, and toddlers twice per week without exceeding Health Canada’s pTDI for methylmercury.

Downstream of the proposed Site C dam, fish mercury concentrations may double baseline concentrations (EIS Section 11.9). At peak post-impoundment mercury levels women of child bearing age can consume four servings of bull trout per week, other adults nine servings a week, and toddlers could consume two serving a week without exceeding Health Canada's pTDI for methylmercury. Goldeye and walleye from lower reaches of the Peace River have higher pre-inundation concentrations of mercury than other fish in the technical study area. Assuming a doubling of baseline concentrations consumption frequency ranges from one meal per week (toddler) to 4 meals per week for adults who are not pregnant.

Fish is an excellent source of high quality protein, beneficial omega-3 fatty acids, vitamins, and essential elements. Fish consumption has been shown to protect health and promote healthy development. While MeHg concentrations in fish would temporarily increase within the proposed Site C reservoir, the potential health risks associated with MeHg exposure from fish consumption needs to be carefully weighed against the health benefits of fish consumption. Baseline fish MeHg concentrations in the technical study area are sufficiently low that, even during the period of peak post-inundation mercury levels, the fish consumption rate recommended by Health Canada's Food Guide for Healthy Eating of two servings of fish a week could be met by consuming some popular species of fish, such as rainbow trout, from the Site C reservoir without exceeding Health Canada's pTDI for methylmercury.

The Human Health Risk Assessment focused on the potential incremental health risks associated with methylmercury from fish consumption . Methylmercury concentrations in soil, sediment, water, and terrestrial country foods, such as mushrooms, plants, and game, would not be significantly affected by the Project. While methylmercury levels might increase in fish-eating birds (e.g., loons and mergansers) and mammals (e.g., river otters) in the Technical Study Area, data from the BC First Nations Food, Nutrition, and Environment Study and the Duncan's First Nation and Horse Lake First Nation's Country Food Harvest Consumption Surveys indicate that people do not commonly consume fish-eating wildlife from the Technical Study Area. Therefore, there would not be any appreciable Project-related health risks associated with mercury exposure from sources other than fish consumption. The scope of this Human Health Risk Assessment is consistent with guidance from Health Canada (2011) on assessing potential human health risks associated with hydroelectric projects which recommends that potential methylmercury exposure from the "most commonly consumed aquatic species and tissues" be assessed.

This Human Health Risk Assessment, as with any predictive risk assessment of future exposure scenarios, contains elements of uncertainty. Conservative, or health protective, assumptions were generally used to address these uncertainties and the net effect is that the human health risks associated with exposure to methylmercury from the construction and operation of the Project are unlikely to have been underestimated.

## ABBREVIATIONS AND ACRONYMS

BC MOE ............ BC Ministry of the Environment

CCME ............ Canadian Council of Ministers of the Environment

HHRA ............ Human Health Risk Assessment

Hg ............ Mercury

MeHg ............ Methylmercury ($HgCH_3$)

pTDI ............ provisional Tolerable Daily Intake

pTWI ............ provisional Tolerable Weekly Intake

RA ............ Risk Assessment

TDI ............ Tolerable Daily Intake

U.S. EPA ............ U.S. Environmental Protection Agency

WHO ............ World Health Organization

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# 1 INTRODUCTION

This report presents results of a Human Health Risk Assessment (HHRA) to assess how changes in methylmercury (MeHg) concentrations in fish following construction and operation of the proposed Site C Clean Energy Project (the Project). Changes in fish MeHg concentrations may affect consumption limits of fish in the proposed Site C technical study area. This HHRA discusses the implications of the Project on methylmercury concentrations in other potential exposure media and country foods; however, the vast majority of human exposure to methylmercury is through fish consumption and this is the focus of the HHRA.

## 1.1 Objectives and Scope

The purpose of this HHRA is to assess the potential incremental risk to human health following a predicted increase in methylmercury in fish following inundation of terrestrial soils in a portion of the Peace River valley between the Peace Canyon Dam downstream to the proposed Site C dam site just downstream of Moberly River to create the proposed Site C reservoir (Volume 2 Section 11.9 Methylmercury). The technical study area of this assessment is presented below in Section 1.4.

## 1.2 Approach to Human Health Risk Assessment

Risk assessment was originally defined by the U.S. National Academy of Sciences in 1983 as the use of facts to define the health risk of exposure to hazardous materials and situations (US National Research Council, 1983). Risk assessment was distinguished from *risk management*, which includes social, economic, and technological considerations in the prevention or control of risk.

This report presents scientific data and associated evidence about the potential health risks associated with human exposure to methylmercury in fish. Results of the HHRA identify *relative priorities* for risk management, but the *requirement* for risk management (i.e., deciding which risks are acceptable and which are not) is a risk management decision which, as described above, is made on the basis of wider social, economic, and technological considerations.

The following assumptions and guiding principles were used in this HHRA:

- Inundation of terrestrial soils creates conditions favorable for increased rates of mercury methylation, resulting in bioaccumulation of MeHg within aquatic food web organisms (bacteria, invertebrates, forage fish, piscivorous fish). Methylmercury also increases in concentration with increasing steps up the food web or trophic levels, a process known as biomagnification, causing the highest levels of methylmercury to be found in piscivorous fish species (e.g., lake trout, bull trout, northern pike, walleye) and some fish-eating wildlife (Potter et al. 1975; Abernathy and Cumbie 1977; Bodaly and Hecky 1979; Bodaly et al. 1984; Hall et al. 1997 and others). Methylmercury is more readily absorbed into the body and is more slowly excreted from the body than other forms of mercury. Methylmercury is also capable of crossing the placenta and the blood-brain barrier and a mother's methylmercury exposure can be passed on to her developing fetus and breast-fed infant.
- The vast majority of human exposure to methylmercury in the environment comes from consumption of fish. Of all forms of mercury found in fish (i.e., 'total mercury'), the proportion that occurs as methylmercury is typically about 95% (Bloom 1992). In wildlife, the proportion of mercury that is in the methylmercury form relative to the total mercury concentration is usually less than 5% (Eisler 2006). Furthermore, the total mercury concentration in wildlife is usually much less than that in fish (Gamberg 1999; Eisler 2006). The HHRA is, therefore, focused on exposure to methylmercury from fish consumption. When referring to mercury (Hg) in fish, it is assumed that it is all in the form of methylmercury and the two terms are used interchangeably. More detailed justification for the focus on MeHg in fish is provided in the Problem Formulation for the HHRA presented in **Section 3** of the report.
- At the request of Health Canada, the predicted mercury concentration in water from the Site C Reservoir was compared to the Canadian Drinking Water Quality Guideline for the protection of human health.
- The baseline concentrations of mercury in various fish species used in this report are derived from empirical data from Peace River and Dinosaur Reservoir that are summarized in Azimuth 2011 and presented in context with other Canadian reservoirs in Volume 2, Appendix J, Part 1 Mercury Technical Synthesis Report for the proposed Site C project. Forecast changes to fish MeHg concentrations were determined from three lines of evidence that were integrated together within Volume 2 Section 11.9 Methylmercury. These independent lines of evidence are presented in the Hg Synthesis report (Volume 2 Appendix J Part 1) and the Reservoir Mercury Modeling Report or RESMERC (Volume 2 Appendix J Part 3) to derive an 'increase factor'. This factor is used to multiply baseline fish mercury concentrations to estimate the absolute concentration that is predicted for key fish species that might be targeted by people for consumption, within the proposed Site C reservoir, and downstream.

- Potential human health risks from methylmercury exposure were characterized as the number of servings of fish that can be consumed per week without exceeding Health Canada's provisional Tolerable Daily Intake (pTDI) for methylmercury.
- Fish has a high nutritional value and the health benefits of fish consumption need to be carefully weighed against the potential health risks associated with methylmercury in fish.

## 1.3 Resources

This section lists key resources consulted in the development of the HHRA. These resources include guidance documents from Health Canada on methods for HHRA and reports that contain information on consumption of country foods, including fish, by First Nations and the general Canadian population.

### 1.3.1 Regulatory Guidance on Human Health Risk Assessment

The general framework of this HHRA is consistent with guidance on the practice of HHRA for chemical exposures established by regulatory agencies such as Health Canada, the BC Ministry of the Environment (BC MOE), and the U.S. Environmental Protection Agency (U.S. EPA). The specific methods used in the HHRA are complaint with the most recently available Health Canada guidance, including the following:

- *Health Canada. Draft 2009. Federal Contaminated Site Risk Assessment in Canada, Part I: Guidance on Human Health Preliminary Quantitative Risk Assessment; Version 2.0. Contaminated Sites Division, Safe Environments Directorate, Health Canada, Ottawa, ON.* The purpose of this guidance document is to prescribe, to the degree possible, standard exposure pathways, receptor characteristics, and other parameters required to assess potential chemical exposures and associated human health risks at federal contaminated sites. While this guidance is intended for screening level risk assessments for contaminated sites, the default receptor characteristics and algorithms for calculating exposure estimates are also accepted by Health Canada for use in more complex HHRAs or HHRAs of chemical exposures from sources other than contaminated sites. The receptor characteristics, such as assumed body weights and fish serving sizes, and exposure algorithms used in this HHRA are consistent with the recommend methods provided in the Health Canada guidance document.
- *Health Canada. Draft 2009. Federal Contaminated Site Risk Assessment in Canada, Part II: Health Canada Toxicological Reference Values (TRVs) and Chemical Specific Factors. Contaminated Sites Division, Safe Environments Directorate, Health Canada, Ottawa, ON.* The purpose of this guidance document is to provide Health Canada endorsed toxicological reference values, including

Tolerable Daily Intakes and cancer slope factors. The toxicological reference values used in this HHRA were sourced from this Health Canada guidance document.

- *Health Canada. 2010. Federal Contaminated Site Risk Assessment in Canada, Part V: Guidance on Human Health Detailed Quantitative Risk Assessment for Chemicals (DQRACHEM). Contaminated Sites Division, Safe Environments Directorate, Health Canada, Ottawa, ON.* This guidance document provides recommended methods and considerations for detailed quantitative HHRAs. A detailed quantitative HHRA is more complex than a preliminary quantitative HHRA and may be required when a preliminary assessment is considered too conservative or not adequate to support a risk management plan. The purpose of a detailed quantitative HHRA is to produce a more accurate (i.e., realistic), defensible, and representative estimate of risks than that generated by a preliminary quantitative HHRA. Although the level of detail of such an HHRA can vary considerably, depending on the objectives of the assessment, a detailed HHRA typically uses more comprehensive site characterization data, more representative or site-specific exposure information and in many cases, a higher level of sophistication in the fate, transport, and exposure modeling. Guidance on the use of statistical estimates of exposure point concentrations (i.e., average methylmercury concentrations in fish) was consulted in the development of this HHRA.
- *Health Canada. 2010. Federal Contaminated Site Risk Assessment in Canada: Supplemental Guidance on Human Health Risk Assessment for Country Foods (HHRA Foods). Contaminated Sites Division, Safe Environments Directorate, Health Canada, Ottawa, ON.* This document provides specific advice on HHRA of chemical exposures from consumption of non-commercial foods, such as fish, game, and berries. This document defines country foods as foods that are ‘trapped, fished, hunted, harvested or grown for subsistence, medicinal or traditional purposes, or obtained from recreational activities such as sport fishing or game hunting’.
- *Health Canada. 2011. Hydroelectric Projects. Environmental Assessment Division, Health Canada, Ottawa.* This fact sheet presents an overview of the potential human health risks associated with hydroelectric projects. This document provides a definition of ‘country foods’ and identifies increased methylmercury in fish as a potential health risk associated with the operational phase of a hydroelectric project. The document recommends that potential methylmercury exposure from the “most commonly consumed aquatic species and tissues” be assessed.

### 1.3.2 Information on the Consumption of Country Foods

The HHRA of exposures to methylmercury from consumption of country foods related to the proposed Site C Project requires: (1) information on what country foods are consumed by humans and what the associated serving sizes are; and (2) information on the expected levels of methylmercury in those country food items. The predicted concentrations of methylmercury in fish within the Site C technical study area (Section 1.4) including the proposed reservoir and downstream are described within Appendix V2J Part 3, RESMERC and integrated within Section 11.9 Methylmercury of the EIS. The following documents were consulted for information on what country foods are consumed by First Nations and the general public and what the associated serving sizes are:

- *Country Food Harvest Consumption Survey data for the Horse Lake and Duncan's First Nations.* These questionnaire data provide information from local First Nations on the types of country foods, including fish that are consumed, the frequency of consumption, and the average serving size. These data are summarized in Volume 4, Section 33 (Human Health) of the Site C EIS.

- *Health Canada. 2007. Human Health Risk Assessment of Mercury in Fish and Health Benefits of Fish Consumption. Health Canada, Health Products and Food Branch, Food Directorate, Bureau of Chemical Safety, Ottawa, ON.* This document provides information on mercury concentrations in commercial fish and estimates of fish serving sizes for the general Canadian population.

- *Chan, L., O. Receveur, D. Sharp, H. Schwartz, A. Ing and C. Tikhonov. First Nations Food, Nutrition and Environment Study (FNFNES): Results from British Columbia (2008/2009). University of Northern British Columbia, Prince George, BC.* This document presents the results of a study of traditional food consumption behaviors and contaminant concentrations (including total and methylmercury) in First Nations traditional foods in BC. The study included 1,103 participants aged 19 years and older, living on-reserve, and self-identified as First Nations from 21 randomly selected communities in BC. The study reports questionnaire data on frequency of consumption and serving size of traditional foods. Contaminant concentrations were measured in 429 food samples representing 158 different types of traditional foods, including fish. The study results are reported on a Provence-wide basis or on the basis of regional ecozones. The two regional ecozones with survey data available that are in closest proximity to the proposed Project are ecozone 2, which includes the Doig River and Saulteau First Nations, and ecozone 8, which includes the Profit River and Fort Nelson First Nations.

## 1.4 Study Area Boundaries

The technical study area of this HHRA includes the proposed Site C reservoir and the Peace River downstream from the proposed Site C dam location to Many Islands, AB. The proposed Site C reservoir extends 83 km downstream from the tailrace area below the Peace Canyon dam to the proposed Site C dam site. The proposed Site C dam site is located just downstream of the confluence of the Peace and Moberly rivers. Methylmercury concentrations are predicted to increase within the proposed Site C reservoir (Volume 2 Section 11.9 Methylmercury) which is consistent with methylmercury dynamics observed in many other Canadian reservoirs (Volume 2 Appendix J Part 1 Mercury Technical Synthesis report) and the proposed reservoir is the focus of this assessment.

In addition, it has been observed that fish mercury concentrations have increased far downstream of other boreal reservoirs in Quebec (Schetagne and Verdon 1999a, b), Manitoba (Bodaly et al. 2007) and Labrador (Anderson 2011). The extent and duration of downstream changes to fish Hg levels vary from system to system, depending on specific hydrological and biological features. For example, the extent of dilution from tributaries below the reservoir and the presence of large deep lakes (Schetagne and Verdon 1999b) above or below the reservoir may reduce this effect. The main exposure pathway to downstream fish is when normally non-piscivorous fish (e.g., lake whitefish, longnose sucker) switch their diet to feed on injured or dead fish below the tailrace of large reservoirs.

As explained in the Mercury Technical Synthesis document (Volume 2 Appendix J Part 1), there is the potential for downstream export of Hg adhered to sediment particles and organic material, MeHg dissolved in water and directly in the tissue of plankton and fish that are discharged or entrained out of the Site C reservoir. Based on numerous fisheries investigations by Mainstream Aquatics (2010a; 2011; 2012) and as outlined in Volume 2 Appendix O Fish and Fish Habitat Technical Data Report ), the vast majority of fish species (i.e., 30 of 32) downstream of the proposed Site C dam range between the tailrace and Many Islands, AB, about 100 km downstream of the proposed Site C dam site. Fish within this reach of river downstream of Site C can potentially be exposed to plankton, insects and fish with elevated mercury discharged or entrained from the reservoir. Because of the increased availability of injured or stunned fish in the tailrace region of dams, downstream fish may preferentially feed or switch diet to feed on this source of easy prey. When they move back downstream they may carry this higher load of mercury potentially as far as Many Islands. Although some fish species (e.g., walleye, goldeye, northern pike) have been known to extend farther downstream than Many Islands, AB (Mainstream Aquatics 2010b), their temporal interaction or overlap within the tailrace area of the proposed Site C dam is expected to be of insufficient duration to accumulate methylmercury to the same degree as a fish residing within the reservoir full-time. The implications of this are described in Section 5.0 of this report.

## 1.5 Structure of Report

The remainder of this report contains the following sections:

***Section 2*** provides background information on the different forms of mercury in the environment, why methylmercury increases in the aquatic food chain after land is flooded, and why mercury levels associated with the Site C Clean Energy Project are lower than other hydroelectric projects (Volume 2 Appendix J Part 1 Mercury Technical Synthesis Report). **Section 2** also discusses the potential concerns about human consumption of fish that contain elevated levels of methylmercury, defines Health Canada's tolerable level of human exposure to MeHg from all sources and discusses the nutritional benefits from eating fish.

***Section 3*** presents the Problem Formulation for the HHRA. The Problem Formulation defines the scope of the HHRA. **Section 3** also presents the methods used to estimate the levels of MeHg exposure people may experience if they consume fish from areas where there is a temporary increase in MeHg concentration following construction of the Project.

***Section 4*** presents the estimated number of servings of fish per week that people may consume during the period of peak methylmercury levels in fish following construction of the Project, without exceeding Health Canada's tolerable level for exposure to methylmercury. **Section 4** also includes a discussion of the key areas of uncertainty in the HHRA.

***Section 5*** presents a summary and discussion of the results of the HHRA.

# 2 WHY THE CONCERN

All Canadians are exposed to some mercury in their environment and, for the vast majority, eating fish is the primary source of exposure to methylmercury (Abelsohn et al. 2011; Health Canada 2007). The combined amount of exposure to methylmercury from all other environmental sources (e.g., drinking water, commercial foods, and wild game and other country foods), is negligible relative to the amount of methylmercury exposure humans receive from eating fish.

Construction and operation of the proposed Site C reservoir will inundate organic soils and release nutrients and mercury into the aquatic environment. Some of the inorganic mercury bound up in soils is converted to methylmercury by a specific group of bacteria during the decomposition process. Once converted to methylmercury, this form of Hg is easily incorporated into the aquatic food web and is available to be accumulated and concentrated within increasing steps up the food chain. For example, the methylmercury concentration in fish is about 100,000 times more concentrated than in plants and at least 10 million times more concentrated than in water. More detail on this process can be found in Volume 2 Appendix J Part 1 Mercury Technical Synthesis Report.

Volume 2 Section 11.9 Methylmercury predicts that methylmercury concentrations in fish within the proposed reservoir will increase relative to baseline, potentially exposing people to incrementally higher methylmercury concentrations from fish consumption than current exposure. Consequently there is a need to quantify the incremental potential risk so that fish consumers understand the implications and adjust consumption frequencies if necessary.

Many fish samples have been collected from the Peace River technical study area over the last 20 years and analyzed for mercury concentrations (e.g., Pattenden et al. 1991; Baker et al. 2002; Mainstream Aquatics 2009; Azimuth 2011). Results indicate that MeHg concentrations are stable and low. Mercury concentrations in British Columbia fish are generally lower than elsewhere in Canada (Depew et al. 2012), seldom exceeding 0.2 – 0.3 mg/kg wet weight (Baker 2002). Mercury concentrations in most fish in the Peace River are less than 0.1 mg/kg wet weight. Fish from elsewhere in Canada commonly exceed 0.5 – 1.0 mg/kg wet weight for large top level predators such as lake trout, walleye and northern pike (DePew et al. 2012). For example, in Ontario lakes in 2012 there were more than 1650 human consumption advisories related to mercury in fish (Ontario Sport Fishing Guide 2012). In British Columbia, there are only three, two of which are related to contamination from mining sources.

Health agencies, such as Health Canada, define levels of methylmercury exposure that the general population can tolerate without increased risk of adverse health effects. This HHRA provides estimates of how frequently fish can be consumed without exceeding the tolerable levels of methylmercury exposure defined by Health Canada.

Harvesting and consumption of fish and other country foods provide important health benefits. It is important that these activities not be discouraged or discontinued if methylmercury levels in the food are not high enough to present a human health risk. As mentioned above, all Canadians are exposed to mercury in the environment and methylmercury in fish, so the concern is not *if* there will be exposure to methylmercury, but rather *how much* exposure to methylmercury might occur and whether this exposure is within the tolerable levels defined by Health Canada.

## 2.1 Mercury in the Environment

A brief summary of background information on mercury in the environment is provided below. The natural chemistry of mercury, the phenomenon of increased levels of methylmercury in aquatic food chains following flooding of terrestrial environments, and the methods used to forecast the levels of methylmercury in the aquatic food chain following construction of the Project are described in greater detail in the Mercury Technical Synthesis document (Volume 2 Appendix J Part 1).

***Methylmercury Biomagnifies in the Aquatic Food Chain***

Mercury is a naturally occurring element and is commonly found in low concentrations in all environmental media including air, water, soil, sediment, plants and all animals, similar to many other metals. Mercury occurs in several different chemical forms. Most commonly mercury is found in the inorganic form cinnabar, or mercury sulfide (Hg-S), from which elemental mercury is mined. Elemental mercury had been widely used in many industrial processes (e.g., switches, thermometers, pulp and paper, gold mining) and as an ingredient in dental amalgams. The most common organic form of mercury is methylmercury. Naturally occurring bacteria in the environment convert inorganic mercury to methylmercury. The rate of methylmercury production is temporarily increased after organic soils are flooded.

The chemical and physical properties of methylmercury affect how methylmercury behaves in the environment as well as within an organism.

Diet is the main source of human exposure to methylmercury and methylmercury is more easily absorbed into the body than other forms of mercury. Absorbed methylmercury is distributed throughout the body, but the majority ends up bound to proteins in muscle tissue. Methylmercury is excreted from the body very slowly, so under continuous exposure the body burden of methylmercury will continue to increase with time. The same is true in fish as in humans.

Methylmercury biomagnifies in the food chain - this means that the concentration of methylmercury in organisms increases with increasing trophic position (i.e., predators accumulate higher methylmercury concentrations than their prey). The highest methylmercury concentrations in the aquatic food chain are typically found in long-lived predatory fish such as lake trout and bull trout.

***Estimating Post-Construction Methylmercury Levels***

As mentioned above, the rate of methylmercury production increases temporarily following flooding of terrestrial soils, as bacteria transform naturally occurring inorganic mercury to the organic, methylmercury form. Research from Canadian hydroelectric reservoirs shows that peak levels of methylmercury occur in adults of large-bodied, relatively long-lived fish species about 3-8 years after impoundment before returning to baseline levels 10-20 years after peak levels are reached (Bodaly et al. 2004, 2007; Schetagne et al. 2003).

According to results of the RESMERC mechanistic modeling exercise, peak levels of methylmercury in the aquatic food chain following construction of the Site C dam are expected to increase by 3 – 4 times above baseline concentrations in the proposed Site C reservoir. However, given that baseline concentrations are low, the magnitude of increase and the absolute concentrations in fish is also low, and especially low relative to peak methylmercury concentrations that have been reported from other Canadian reservoirs (Volume 2 Appendix J Part 1). This is because (1) the baseline levels of mercury in the Peace River system are low compared to other places in Canada and (2) environmental conditions in the Site C reservoir following construction are expected to result in a relatively low rate of methylmercury production (Volume 2 Appendix J Part 1).

## 2.2 Regulatory Guidance on Methylmercury Exposure

Exposure to methylmercury at sufficiently high doses may cause adverse health effects. International and national health agencies, such as the World Health Organization (WHO), Health Canada, and the U.S. Environmental Protection Agency (U.S. EPA) have defined environmental quality standards and guidelines for acceptable concentrations of mercury in the environment (e.g., mercury concentrations in food, soil, and drinking water) as well as levels of methylmercury exposure that the general population can tolerate without increased risk of adverse health effects. The agency guidelines and standards that define acceptable levels of mercury in the environment and tolerable levels of human intake of methylmercury that are used in this HHRA are listed in **Table 1**. It is emphasized that environmental concentrations or exposures that exceed these benchmarks will not necessarily be associated with adverse health outcomes. These benchmarks are conservative and are derived to be protective of human and ecological health and are not intended to represent a "bright line" demarking "safe" from "unsafe" exposures.

**Table 1. Canadian upper limits for methylmercury ingestion and mercury in the environment**

| Environmental guideline or tolerable daily intake | Value | Units |
|---|---|---|
| CCME Canadian Drinking Water Quality Guideline | 1000 | ng total mercury/L water |
| CCME Canadian Soil Quality Guideline for the protection of human health for agricultural and residential land uses | 6.6 | mg total mercury/kg soil |
| Health Canada provisional Tolerable Daily Intake for methyl mercury for the general population | 0.47 | µg methylmercury/kg body weight/day |
| Health Canada provisional Tolerable Daily Intake for children less than 13 yrs old and women of child bearing age | 0.2 | µg methylmercury/kg body weight/day |

The Canadian Council of Ministers of the Environment (CCME) has established environmental quality guidelines for total mercury in drinking water, soil, and sediments. The most recent CCME environmental quality guidelines can be found at: http://www.ccme.ca/publications/ceqg_rcqe.html.

The CCME Canadian Drinking Water Quality Guideline maximum acceptable concentration of mercury in drinking water is 0.001 mg/L or 1 µg/L, which is one part per billion. The total mercury concentration in Peace River water currently averages 1 ng/L or 1 part per trillion for most of the year - 1,000-fold less than the CCME Canadian Drinking Water Quality Guideline for mercury.

The CCME Canadian Soil Quality Guideline for mercury for the protection of human health in agricultural, residential, and parkland areas is 6.6 mg/kg. The 95% upper confidence limit of the existing mean mercury concentration in Peace River valley organic soils is 0.11 mg/kg, about 60-fold less than the soil quality guideline for the protection of human health.

Health Canada defines a Tolerable Daily Intake (TDI) for various potential contaminants as “the maximum amount of a chemical that can be ingested on a daily basis over a lifetime without increased risk of adverse health effects” (Health Canada 2007). Health Canada’s provisional TDI (pTDI) for methylmercury for the general population is 0.47 µg/kg body weight/day (µg/kg bw/d) (Health Canada 2007, 2009b). A provisional TDI is one that may be updated by Health Canada as new information becomes available – many substances have provisional TDIs.

The developing nervous system can be affected by methylmercury and the developing fetus, infants and young children are considered more susceptible to the potential health effects of methylmercury than people at other life-stages. A mother’s methylmercury exposure can be passed on to her child (methylmercury in a woman’s body can be passed through the placenta to the developing fetus and through breast milk to the infant). Therefore, Health Canada has a separate, lower pTDI for methylmercury of 0.2 µg/kg/d for children less than 13 years of age and women of childbearing age.

## 2.3 Health Benefits of Eating Fish

While fish may contain methylmercury, fish is also an excellent source of high quality protein and is one of the best food sources of omega-3 fatty acids and vitamin D; fish is also a good source of the essential elements selenium, iodine, magnesium, iron and copper (Health Canada 2007). There is evidence that regular fish consumption can benefit cardiovascular health and child development (Health Canada 2007). Health Canada’s Food Guide for Healthy Eating recommends eating at least two 75 g servings of fish per week (i.e., 150 g of fish/week).

Health Canada (2007) states that it is essential that any public risk communication on methylmercury in fish “include information on the health benefits of fish consumption alongside information on the risks of methylmercury exposure so that citizens can consider both the benefits and risks in reaching their own decisions about appropriate fish consumption.” Research has shown that ineffective risk communication can result in decreased fish consumption and the adverse health effects associated with reduced fish consumption can outweigh the potential health benefits associated with avoiding methylmercury exposure from fish (Mozaffarian and Rimm 2006; Teisl et al. 2011). The objective of risk management measures to address methylmercury in fish should not be to reduce fish consumption, but to encourage consumption of fish species with lower levels of methylmercury (Abelsohn et al. 2011; Mozaffarian and Rimm 2006; Teisl et al. 2011).

# 3 EXPOSURE ASSSESSMENT

This section of the report presents the Problem Formulation for the HHRA as well as describes the methods and assumptions used to estimate levels of methylmercury exposure from consuming fish from the Technical Study Area. The Problem Formulation defines the scope of the HHRA.

## 3.1 Problem Formulation

Construction of the Project would not affect human exposure to *inorganic* mercury nor levels of methylmercury in the *terrestrial* environment. Health Canada's Fact Sheet for responsible parties on hydroelectric projects states that there is increased methylation of mercury in the *aquatic environment* after water is impounded behind a dam (Health Canada 2011). No measurable change in mercury in the terrestrial environment and food chain, including air, soil, terrestrial plants and berries, fungi (e.g., mushrooms), or game (e.g., deer, caribou, elk, bear, sheep, mountain goat, porcupine, beaver, rabbit, grouse) is expected. Therefore, potential risks associated with mercury exposure by contact with these environmental media or consumption of these country foods are not included in the HHRA.

As summarized in **Section 2** above and as described in detail in Volume 2 Appendix J Part 1, methylmercury levels in the aquatic environment and food web are expected to increase temporarily following impoundment of the proposed Site C reservoir. An integrated assessment approach was used to determine the most likely degree that fish mercury concentrations would increase above baseline following construction and operation of the proposed new reservoir. This was based on three lines of evidence including:

- a detailed comparison of the physical, chemical and ecological features of Site C with several other Canadian reservoirs (Volume 2 Appendix J Part 1),
- regression modeling whereby simple parameters (reservoir area relative to original area; water turnover rate) are found to correlate well with the degree of increase in fish mercury levels above baseline (Harris and Hutchinson 2012, in Volume 2 Appendix J Part 3), and
- complex, mechanistic mercury modeling (Volume 2 Appendix J Part 3) that models changes in mercury and methylmercury in a wide variety of environmental media at various time intervals.

An integrated approach was taken in Volume 2 Section 11.9 Methylmercury to harmonize and reconcile the three lines of evidence to determine the most likely magnitude of increase in fish Hg concentration with the proposed Site C reservoir. These approaches provide lower and upper bound estimates of increase in fish mercury. Two of the three lines of evidence suggest a low magnitude of increase – 2.3x based on the Harris and Hutchinson (2012) model and less than 3x based on the Canadian reservoirs comparison matrix. Although RESMERC predicts a maximum increase of 4 to 6x above baseline (depending on species), there is inherent conservatism in the model (e.g., assumption of negligible sedimentation during the operation phase) that would suggest a lower increase than what is predicted using this method. Consequently, based on the available information, the most reasonable estimate of the harmonized peak increase factor for all species is considered to be 3x. This value retains some conservatism relative to the results of the strong empirical evidence of the regression and matrix approaches, but also some uncertainty relative to RESMERC. Notwithstanding, for the purposes of assessing the mercury-related changes associated with the proposed Site C on humans, it is recommended that a peak increase factor of 4x be used to reduce the possibility of underestimating of fish Hg concentrations.

With respect to the duration of the increase in fish mercury concentrations within the proposed Site C reservoir, the timing of a return of reservoir fish Hg concentrations to baseline was inferred from the Canadian reservoirs comparison matrix, as well as RESMERC (Volume 2 Appendix J Parts 1 and 3). Reservoirs, with a short hydraulic residence time, small reservoir to original basin ratio, minimal flooded wetland and a large upstream oligotrophic lake or reservoir as exemplified by the proposed Site C Project will have shorter return periods, depending on the species, in the order of 15 – 20 y following impoundment. RESMERC predicts a return time of between 20 and 25 years, depending on the species. In either case forage species such as redside shiner, sucker and rainbow trout that consume lower mercury dietary items will return to a baseline more quickly that omnivorous whitefish and piscivorous bull trout.

Volume 2 Section 11.9 Methylmercury indicates that a return to baseline would occur 20 years after because of the weight of evidence presented by the Canadian reservoirs comparison matrix and particularly the influence of Williston Reservoir upstream.

With respect to downstream fish, it is acknowledged that the return to baseline is much shorter, as has been observed in northern Quebec (2 – 4 y for whitefish and 4 – 8 y for lake trout) (Schetagne and Verdon 1999b) and whitefish (7 – 8 y) downstream of Smallwood Reservoir, Labrador (Anderson 2011). The Site C EIS Section 11.9 predicts a return to baseline in downstream fish on the order of 5 – 8 y after impoundment of the proposed reservoir.

As a result of this temporary increase in methylmercury in the aquatic environment, there may be a temporary, incremental increase in human exposure to methylmercury from consuming fish. However, during the review period of the draft EIS Guidelines, Health Canada requested that the HHRA also address potential methylmercury related health risks from the following: (1) direct contact with and ingestion of methylmercury in water; (2) direct contact with and incidental ingestion of methylmercury in sediments; and (3) the consumption of methylmercury in country foods (besides fish) in the aquatic food chain.

Potential exposure to methylmercury through contact with or consumption of media or food from the aquatic environment is a function of: 1) the MeHg concentration in the exposure media; and 2) the frequency and the amount (mass) of the exposure media that is consumed. The highest human exposure is from fish consumption. However, for completeness and to respond to the request by Health Canada, other potential aquatic exposure sources and pathways were addressed as follows.

### 3.1.1 Potential Exposure to Mercury in Water

The CCME Canadian Drinking Water Quality Guideline for total mercury (i.e., all forms of mercury including methylmercury) for the protection of human health is 1,000 ng/L (parts per trillion). There is no guideline for methylmercury in drinking water. The baseline concentration of total mercury in the mainstem of the Peace River averages 1 ng/L.

The concentration of inorganic Hg in water in the Technical Study Area is not expected to change as a result of the Project, as there will be no ‘new’ source of inorganic Hg to the reservoir. The main source of mercury in water in the Peace River within technical study area is from Williston Reservoir.

Methylmercury generated from inorganic mercury in flooded soils will be released to the overlying water column and the concentration of methyl mercury in water is projected to approximately double from baseline concentrations (Volume 2 Appendix J Part 3). This would increase methylmercury concentrations in water from baseline (the laboratory detection limit of <0.05 ng/L) to <0.1 ng/L in the proposed Site C reservoir. This post-impoundment methylmercury concentration is 10,000 fold lower than the Canadian Drinking Water Quality Guideline for total mercury. Therefore, contact with or consumption of water from the proposed Site C reservoir would not pose a health risk from mercury or methylmercury exposure.

### 3.1.2 Potential Exposure to Mercury in Sediments

Current mercury concentrations in soils that would be impounded by the creation of the Site C reservoir, exclusive of a small area of wetland (Watson Slough), range from 0.02 – 0.14 mg/kg with a mean of 0.08 mg/kg. These concentrations are low compared to other areas of the world at similar latitudes and are typical of pristine, uncontaminated soils elsewhere in Canadian soils beneath boreal forests (Volume 2 Appendix J Part 1).

Humans could potentially come into contact with and accidentally ingest sediments while recreating (e.g., swimming) within the new reservoir. Although there are no Canadian sediment quality guidelines for the protection of human health, the CCME Canadian Soil Quality Guideline for mercury for the protection of human health in agricultural, residential, and parkland areas is 6.6 mg/kg, or about 60-fold higher than the average baseline mercury concentration recorded in the soils surrounding the proposed location of the Site C reservoir. Mercury concentrations in sediments would not increase following construction of the Project. Therefore, the Project is not expected to present any risks to human health from exposure to mercury from contact or incidental ingestion of sediments.

The soil quality guideline is derived to protect against potential human health risks associated with direct contact with mercury contaminated soil. In this case, soils will become sediment immediately after inundation. While rates of dermal absorption and incidental ingestion may be higher for sediments than soils, humans are expected to come into direct contact with sediments much less frequently that they would potentially come into direct contact with soils at agricultural and residential sites. Therefore, the Canadian Soil Quality Guideline for the protection of human health from risks associated with direct contact with mercury contaminated soils at agricultural and residential sites is also protective of potential risks associated with contact with sediments that contain naturally occurring mercury.

### 3.1.3 Potential Exposure to Methylmercury in Country Foods Besides Fish

Health Canada defines country foods as foods that are ‘trapped, fished, hunted, harvested or grown for subsistence, medicinal or traditional purposes, or obtained from recreational activities such as sport fishing or game hunting’ (Health Canada 2011). The focus of the Site C HHRA is on assessing potential risks associated with the consumption of fish muscle tissue. This is because the concentration of methylmercury in fish tissue is at least 100-fold higher than the MeHg concentration in flesh of all other commonly consumed meats and meat products including domesticated animals (cattle, pigs, chickens, etc.) and wild game (moose, deer, caribou, ducks, geese, etc.). The only exception is animals that principally prey on fish such as loons, kingfisher and otter. Wildlife that prey on fish may accumulate methylmercury levels that are similar to predatory fish. The potential human health risks from consumption of wildlife that prey on fish are addressed below.

In a recent survey of contaminant concentrations in 168 First Nations country food items from BC by Chan et al. (2011), exclusive of those results for fish, the methylmercury concentration in all terrestrial game animals tested was below the analytical detection limit (DL) of 0.004 mg/kg ww in *all* samples, except black bear liver and deer liver and heart. The concentrations of methylmercury in terrestrial traditional food items are at least 100-fold less than the methylmercury concentrations in fish. The study did not include measurements of methylmercury in wildlife that prey on fish.

MeHg concentrations may also increase in fish-eating wildlife, such as loon and otter, following impoundment of the Site C reservoir given that accumulation of MeHg is via dietary sources (Hall et al. 1997). Birds that do not eat fish, including ducks and geese, will not be exposed to increased MeHg concentrations in their food, so mercury levels will not change in these species.

There are no data on the human consumption of fish-eating wildlife from the technical study area. The Country Food Harvest Consumption Surveys completed for the Horse Lake and Duncan's First Nations as part of the First Nations Community Assessment include data on consumption of water fowl (ducks, geese, grebes, and crane). While the survey data indicate that water fowl are consumed, there were no data reported on the serving size or consumption frequency of water fowl and no distinction was made between piscivorous and herbivorous water fowl and between the consumption of birds and eggs. The Country Food Harvest Consumption Surveys did not collect data on the human consumption of fish-eating mammals.

Although there are no direct data available on the consumption of fish-eating wildlife from the Peace River technical study area, regional and provincial survey data indicate that fish-eating wildlife from freshwater systems are not commonly consumed by humans. The percentage of subjects in the  First Nations Food Nutrition and Environmental Study from First Nations communities in the region of the proposed Project that responded that they consumed fish-eating birds (e.g., loons, mergansers, or grebes) ranged from 0 – 2%, depending on the community (Chan et al. 2011). This is in contrast to the percentage of respondents that reported consumption of trout ("any type"), which ranged to up to 61% (Chan et al. 2011). Freshwater fish-eating wildlife (e.g., river otter, mink) were not identified as a traditional food item in the First Nations Food Nutrition and Environmental Study identified (Chan et al. 2011). Chan et al. (2011) reported that the traditional food item that contributed the most to First Nations exposure to mercury was 'trout', however the species was not identified. Nine of the top ten traditional food items that contributed the most to First Nations exposure to mercury were fish – the other was pine mushrooms.

Methylmercury exposure risks associated with the consumption of fish-eating wildlife is not assessed quantitatively in this HHRA because the available data indicate that the consumption of fish-eating wildlife would not be common source of MeHg exposure for humans. Loons and mergansers are not abundant in the technical study area and the percentage of subjects in First Nations country foods consumption surveys reporting that they consume fish-eating birds is low and fish-eating mammals were not identified as traditional food items. Finally, regional data indicate that trout is the most significant source of mercury exposure from consumption of country foods (Chan et al. 2011). Therefore, the focus of this HHRA is on potential methylmercury exposure risks associated with the consumption of fish. The consumption of fish eggs or internal organs (liver) is also not addressed as the consumption of non-muscle tissue has not been identified as a dietary item and the mass of non-muscle tissue consumed is assumed to be very low. This is consistent with Health Canada (2011) guidance which recommends that environmental impact assessments for hydroelectric projects include an assessment of potential methylmercury exposure from the "most commonly consumed aquatic species and tissues".

### 3.1.4 Potential Exposure to Methylmercury in Fish

Mercury levels can differ between different species of fish and a person's mercury exposure from fish consumption will depend partly on what species of fish are consumed within the Site C technical study area. This section describes information on local fish consumption patterns based on the most recently available information from local (e.g., consumption surveys) and regional (e.g., Chan et al. 2011) data. These data, combined with information on species composition, distribution and relative abundance of fish species with the technical study area (within the proposed Site C reservoir and downstream) are used to inform fish consumption patterns of local First Nations and sport fishers in the area.

Local and regional survey data from the Horse Lake and Duncan's First Nations Country Food Harvest Consumption Surveys and Chan et al. (2011) provide some information on what fish species are most commonly consumed by First Nations in the vicinity of the Project. It is assumed that these consumption preferences are also representative of those in the general population that may consume fish from the technical study area. Exposure to MeHg is also affected by the availability of different fish species.

Respondents to the Horse Lake and Duncan's First Nations Country Food Harvest Consumption Surveys reported that the most commonly consumed type of fish was "trout", although the species was not identified. In the technical study area there are two common trout species, rainbow trout (*Oncorhynchus mykiss*) and bull trout (*Salvelinus confluentus*), with only small numbers of lake trout present (*S. namaycush*) (Mainstream Aquatics 2009).

The most commonly consumed species of fish reported by participants in the BC First Nations Food, Nutrition, and Environment Study (Chan et al. 2011) from the First Nations communities in closest proximity to the Site C Project (i.e., Cultural Areas 2 and 8) were rainbow trout (22-23%); Dolly Varden trout (19-23%); northern pike (17-20%); lake trout (14-16%); walleye (2-21%); bull trout (5-6%); brook trout (1-3%); Arctic grayling (2-6%); sucker (1-4%); and whitefish (0-2%). Note that Dolly Varden and bull trout are very closely related and for most people, indistinguishable from one another.

According to Mainstream Aquatics (2010a, 2011, 2012; Site C EIS Section 12), bull trout, rainbow trout, lake trout and mountain whitefish are important sport fish species found within the Peace River in the technical study area. These species are expected to persist within the proposed Site C reservoir during the initial period (<8 y) when MeHg concentrations would peak in fish. However, Arctic grayling and mountain whitefish abundance is expected to diminish or be reduced in the reservoir. Species such as walleye, northern pike, goldeye, burbot and yellow perch occur in low abundance in the technical study area downstream of the proposed Site C dam. (Mainstream Aquatics 2010; 2011).

The emphasis of this HHRA is on potential risks associated with consumption of bull trout and rainbow trout because the available data indicate that, of the fish species most commonly consumed by humans, these species would be most abundant during the first decade following construction of the Project. In addition, bull trout have the highest baseline mercury concentrations of all trout species tested in the technical study area and, although bull trout may not be as commonly consumed as other trout species, the results for bull trout represent an upper limit on potential risks associated with consuming trout of any species. While the HHRA includes risk estimates for consumption of goldeye and walleye, these species are not expected to be abundant or resident in the proposed reservoir area of the technical study area following construction of the Project. Therefore, they are unlikely to experience the same degree of increase in mercury as other fish species. It is also unlikely that these species would be captured and consumed as frequently by humans as other species.

### 3.1.5 Summary

This HHRA is focused on potential risks associated with exposure to methylmercury from fish consumption, especially trout, for the following reasons:

- Post-construction water and sediment (soil) concentrations within the proposed Site C reservoir are expected to be well below their respective Canadian environmental quality guidelines for the protection of human health.
- The Site C Project will have no effect on inorganic mercury or methylmercury concentrations in all terrestrial media including air, soil, vegetation, insects, birds and mammals.
- Human exposure to methylmercury in the environment is almost exclusively via fish consumption. Exposure by other means such as dermal absorption or from drinking water is negligible.

- Fish and piscivorous wildlife are expected to be the only country food items that will have increased methylmercury concentrations following creation of the Site C reservoir. Potential risks from consumption of piscivorous wildlife are expected to be much lower than from fish consumption because the available data suggest that humans rarely consume piscivorous wildlife from the technical study area.
- The concentration of methylmercury in non-muscle fish tissues (e.g., eggs and internal organs) is expected to be lower than the concentration of methylmercury in fish muscle tissue. Therefore, results of the HHRA of methylmercury in fish muscle tissue can also be applied to potential risks associated with methylmercury in non-muscle fish tissues.
- Local and regional survey data indicate that trout are the most commonly consumed type of fish and bull trout and rainbow trout are expected to be relatively abundant during peak methyl mercury levels following construction of the Project. Therefore, the focus of the HHRA is on results for trout. Results for other fish species are reported for completeness.

## 3.2 Methods

This section of the report describes the methods and assumptions used to estimate methylmercury exposure from fish consumption in the technical study area, including the formulae used to derive exposure estimates, assumptions regarding the chemical form of mercury, the methylmercury concentrations in fish, and characteristics (e.g., age, body weight, fish serving size) of people that may consume fish.

### 3.2.1 Exposure Estimates

Methylmercury exposure from consumption of fish was estimated as the ingested dose of MeHg per serving of fish consumed. Separate exposure estimates were derived, as per Health Canada (draft 2009; 2010) HHRA guidance, for each of the following life stages:

Toddler: 7 months to 4 years old

Child: 5 to 11 years old

Teen: 12 to 19 years old

Adult : ≥ 20 years old

Exposure estimates for infants (0-6 months) were not derived as data on average serving size of fish were not available for this life stage and it is assumed that infants are either breast- or formula-fed. Risks to infants are expected to be less than those calculated for toddlers as toddlers typically have a higher body weight normalized food ingestion rate than infants. In all cases, the toddler (< 5 y old) is the most sensitive receptor (i.e., has the highest predicted risks for a given concentration of methylmercury in fish).

The formula used to estimate the ingested dose of methylmercury associated with consuming a serving of fish ($I_{serving}$ in mg MeHg per kg body weight per serving) is defined in Equation 1, where

$C_{fish}$ is the concentration of methylmercury in fish muscle tissue in mg MeHg/kg ww

*P* is the serving size of fish in g

*UC* is a constant unit conversion factor of 0.001 kg/g

*BW* is the receptor body weight in kg

**Equation 1**

$$I_{serving} = \frac{(C_{fish} \times P \times UC)}{BW}$$

Input parameters for the variables $C_{fish}$*, P, and BW* are discussed below.

### 3.2.2 The Concentration of Methylmercury in Fish

Methylmercury exposure estimates were calculated for two scenarios: (1) pre-impoundment, or baseline conditions (Volume 2 Appendix J Part 1) and (2) post-impoundment peak methylmercury levels (Volume 2 Section 11.9 Methylmercury). As described in **Section 2**, methylmercury concentrations in fish are expected to peak 5 – 8 years after the proposed Site C reservoir is impounded, with a return to baseline concentrations over a 10-20 year period (Volume 2 Section 11.9 Methylmercury).

It is assumed, for the purposes of the HHRA, that 100% of total mercury measured in fish is methylmercury. This is a slightly conservative (i.e., health protective) assumption since, typically, about 90 – 95% of mercury in fish is in the methylmercury form (Bloom 1992). For example, the BC First Nations Food, Nutrition, and Environment Study reported that the proportion of total mercury that was methylmercury in most fish species ranged from 70-100% (Chan et al. 2011). These data are in general agreement with the percentage of total mercury present as methylmercury in samples of Canadian commercial fish (Health Canada 2007).

Input values for pre-impoundment or baseline methylmercury concentrations in fish are the arithmetic means of the total mercury concentrations measured in adult muscle tissue samples from fish caught in the technical study area since 2009. These data are reported by Azimuth (2011) and summarized in the mercury Technical Synthesis report (Volume 2 Appendix J Part 1). Given the low concentrations and weak positive relationships between total mercury and increasing fish size (length or weight) mean total mercury concentration for adult fish of each species of fish was considered representative of the average mercury concentration in the size of fish that would be retained by anglers and potentially consumed. This approach is consistent with Health Canada's (2010b) guidance on HHRA of contaminants in country foods, which recommends that, for detailed risk assessments, the arithmetic mean be used as a point estimate of contaminant concentrations in exposure media.

As outlined in Volume 2 Section 11.9 Methylmercury, a value of 4x baseline was used to ensure that this assessment is sufficiently conservative to recognize that some larger fish may have higher Hg concentrations. Therefore, the input values for post-impoundment peak MeHg concentrations for the proposed Site C reservoir are the average total mercury concentrations recently measured in fish samples from the technical study area multiplied by four.

As described in Section 1.4 above, there is the potential that some fish downstream of the proposed Site C reservoir may prey on other fish that are entrained through the Site C dam turbines or spillway. The conclusion of the integrated assessment of the EIS Section 11.9 was that Hg concentrations in fish downstream of the proposed Site C dam would double from baseline.

Note that cooking does not reduce the amount of methylmercury in fish (Health Canada, 2007). Therefore, the mercury concentration in raw fish is assumed to be representative of the methylmercury concentration in fish as prepared for consumption (e.g., cooked, smoked, etc.).

The assumed methylmercury concentrations for various fish species used in the exposure estimates for both the reservoir and downstream locations are presented in **Table 3**.

### 3.2.3 Fish Serving Size

Input values for assumed fish serving sizes are presented in **Table 2**. Current Health Canada guidance does not recommend specific fish serving sizes for use in HHRA of country foods. Health Canada's (draft 2009) guidance on preliminary Quantitative Risk Assessment prescribes assumed fish consumption rates (grams per day) for First Nations and these and a number of other sources of data on fish consumption behaviour are presented in **Table 2** for comparison.

**Table 2. Receptor characteristics for the human health RA of methylmercury in fish**

| Receptor | Toddler | Child | Teen | Adult |
|---|---|---|---|---|
| Age | 7 mo. - 4 y | 5 - 11 y | 12 - 19 y | >= 20 y |
| Body weight (kg) | 16.5 | 32.9 | 59.7 | 70.7 |
| Assumed fish meal portion size (g/meal) | 75 | 125 | 150 | 163 |
| Reported fish meal portion size (g/meal) | | | | |
| BC FNFNES | N/A | N/A | N/A | 163† |
| Health Canada, 2007 | 75 | 125 | 150 | 150 |
| Health Canada (draft 2009) recommended assumption for First Nations fish consumption rate (g/day) | 95 | 170 | 200 | 220 |

† Maximum of age and sex-specific mean portion sizes of fish for men and women aged 19-71+; reported range of means was 87-163 g.

Health Canada (2007) conducted a review of information on fish consumption, including serving sizes, and concluded that the best estimate of the average fish serving size for Canadians was 150 g for adults, 125 g for children 5-11 years old and 75 g for children 1-4 years old. Health Canada (2007) considered these values to be conservative (i.e., health protective) estimates of the average serving size of fish.

Ninety-five percent of participants in the BC First Nations Food, Nutrition, and Environment Study reported consuming fish in the year prior to the study and the mean serving size for fish ranged from 87-163 g/serving, depending on age and sex (Chan et al. 2011). The mean fish portion size for women of childbearing age (19-50 years) was 109 g/serving.

Country Food Harvest Consumption Survey data for the Horse Lake First Nation reported that the average number of fish servings per month consumed by participants was 1.4 (range 0-16 servings per month) and the average serving size of fish was 3.6 oz (approximately equal to 102 g). Age or sex-specific serving sizes were not reported.

Country Food Harvest Consumption Survey data for the Duncan's First Nation reported that the average number of fish servings per month consumed by participants was 4.2 (range 0-16 servings per month) and the average serving size of fish was 5.5 oz (approximately equal to 156 g). Again, age or sex-specific serving sizes were not reported.

Based on a review of the above data, the following serving sizes were assumed in the HHRA: toddlers: 75 g/serving; children: 125 g/serving; teenagers: 150 g/serving; and adults: 163 g/serving. These values were, with the exception of the value for adults, based on the conclusions of Health Canada (2007) and used by Health Canada in the national risk assessment of methylmercury in commercially sold fish as conservative (i.e., health protective) estimates of the average serving size of fish. The Health Canada (2007) serving size for adults (150 g/day) was slightly less than the maximum average fish serving sizes recently reported in surveys of local and provincial First Nations populations. Therefore, a higher value of 163 g/serving based on data from these studies was used in this HHRA. For comparison, a 170 g can of light tuna contains approximately 120 g of fish (the rest being water or oil) (Health Canada 2007) and Health Canada's Food Guide for Healthy Eating recommends at least two 75 g servings per week of fish (i.e., 150 g/week of fish).

### 3.2.4 Body Weight

Input values for assumed body weights are consistent with the receptor characteristics prescribed by Health Canada (draft 2009; 2010) guidance on HHRA. Assumed body weights for the HHRA are presented in **Table 2**.

# 4 RISK CHARACTERIZATION

Potential risks from methylmercury exposure associated with the consumption of fish were characterized by estimating the number of servings of fish per week that may be consumed without exceeding Health Canada's provisional tolerable daily intake (pTDI) for methylmercury. Exposures were estimated for two scenarios: 1) pre-impoundment, or baseline, conditions; and 2) post-impoundment peak methylmercury levels.

The formula used to estimate the number of servings of fish that can be consumed per week (*SW*) without exceeding Health Canada's pTDI for methylmercury (in mg MeHg per Kg body weight/day) is defined in Equation 2. Health Canada's pTDIs for methylmercury are defined in **Table 1** and $I_{serving}$ was calculated as per **Equation 1**.

**Equation 2**

$$SW = \frac{(p\ \mathrm{TDI} \times 7)}{I_{serving}}$$

For risk assessors and risk managers that are familiar with the Hazard Quotient approach to risk characterization in HHRAs, the risk characterization metric used here is the inverse of a Hazard Quotient (i.e., a tolerable fish consumption frequency of 0.5 servings per week is equivalent to a Hazard Quotient of 2).

Risk estimates were derived for various different life stages (i.e., age groups), including sex-specific estimates for some life stages. The requirement to derive separate risk estimates is related to: (1) life stage-specific assumptions for body weight and fish serving sizes as well as (2) life-stage and sex-specific pTDIs for methylmercury. Health Canada defines separate pTDIs for (1) the general public and (2) children less than 13 years of age and women of child-bearing age.

Because the relationship between the concentration of methylmercury in fish ($C_{fish}$) and the number of servings of fish that can be consumed per week (*SW*) without exceeding Health Canada's pTDI is linear, the number of servings of fish that can be consumed per week without exceeding Health Canada's pTDI is reduced by the same magnitude that concentrations of methylmercury in fish are expected to increase following impoundment. For example, if methylmercury concentrations in a particular fish species increase by four-fold following impoundment, the number of servings of that species can be consumed per week without exceeding Health Canada's pTDI will similarly be reduced by four-fold.

## 4.1 Results

The risk characterization results in **Table 3** are presented by species for two exposure scenarios: 1) pre-impoundment or baseline conditions and 2) post-impoundment *peak* methylmercury levels for fish caught from two locations: 1) the proposed Site C reservoir and 2) the Peace River downstream of the proposed Site C dam site to Many Islands, AB. Results are also presented by sex for a range of life stages to reflect differences in susceptibility and body weight normalized exposure across these variables.

Mercury concentrations in fish downstream of the dam are not expected to increase as much as mercury concentrations in fish from the reservoir. Given this difference, results for consumption of fish caught within the proposed reservoir are presented separately from results for consumption of fish caught from the Peace River downstream of the reservoir.

Note that for context **Table 3** also contains information on the number of servings a week of commercially available fish such as halibut and tinned tuna, including albacore and light, or skipjack, tuna, that can be consumed without exceeding Health Canada's pTDI for methylmercury.

**Table 3. Maximum number of servings of fish[1] per week that can be consumed without exceeding Health Canada's provisional tolerable daily intake for methylmercury**

| Species | Location | Time | Average[2] peak methylmercury concentration in fish (mg MeHg/kg ww) | | Toddler | Child | Female Teen | Women of Child Bearing Age | Male Teen | Other Adult |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | age | 7 mo. - 4 y | 5 - 11 y | 12 - 19 y | > 20 y | 12 - 19 y | > 20 y |
| | | | | fish serving size (g/serving) | 75 | 125 | 150 | 163 | 150 | 163 |
| **Bull trout** | **Peace River - Site C** | **Pre-impoundment** | **0.072** | | **4** | **5** | **8** | **8** | **18** | **20** |
| | | **Post-impoundment peak** | **0.288** | | **1** | **1** | **2** | **2** | **5** | **5** |
| | **Peace River - Downstream** | **Pre-impoundment** | **0.083** | | **4** | **4** | **7** | **7** | **16** | **17** |
| | | **Post-impoundment peak** | **0.166** | | **2** | **2** | **3** | **4** | **8** | **9** |
| **Rainbow trout** | **Peace River - Site C** | **Pre-impoundment** | **0.044** | | **7** | **8** | **13** | **14** | **30** | **32** |
| | | **Post-impoundment peak** | **0.176** | | **2** | **2** | **3** | **3** | **7** | **8** |
| **Lake trout** | **Peace River - Site C** | **Pre-impoundment** | **0.066** | | **5** | **6** | **8** | **9** | **20** | **22** |
| | | **Post-impoundment peak** | **0.264** | | **1** | **1** | **2** | **2** | **5** | **5** |
| | **Peace River - Downstream** | **Post-impoundment peak** | **0.132** | | **2** | **3** | **4** | **5** | **10** | **11** |
| **Mountain whitefish** | **Peace River - Site C** | **Pre-impoundment** | **0.039** | | **8** | **9** | **14** | **16** | **34** | **37** |
| | | **Post-impoundment peak** | **0.156** | | **2** | **2** | **4** | **4** | **8** | **9** |
| | **Peace River - Downstream** | **Pre-impoundment** | **0.037** | | **8** | **10** | **15** | **16** | **35** | **39** |
| | | **Post-impoundment peak** | **0.074** | | **4** | **5** | **8** | **8** | **18** | **19** |
| **Goldeye** | **Peace River - Downstream** | **Pre-impoundment** | **0.238** | | **1.3** | **2** | **2** | **3** | **6** | **6** |
| | | **Post-impoundment peak** | **0.476** | | **0.6** | **0.8** | **1** | **1** | **3** | **3** |
| **Walleye** | **Peace River - Downstream** | **Pre-impoundment** | **0.182** | | **2** | **2** | **3** | **3** | **7** | **8** |
| | | **Post-impoundment peak** | **0.364** | | **0.8** | **1** | **2** | **2** | **4** | **4** |
| **Commercial canned albacore tuna** | **Health Canada (2007)** | | **0.36** | | **1** | **1** | **2** | **2** | **4** | **4** |
| **Commercial canned light tuna (skipjack)** | **Health Canada (2007)** | | **0.06** | | **5** | **6** | **9** | **10** | **22** | **24** |
| **Commercial halibut** | **Health Canada (2007)** | | **0.31** | | **1** | **1** | **2** | **2** | **4** | **5** |
| **Commercial trout** | **Health Canada (2007)** | | **0.14** | | **2** | **3** | **4** | **4** | **9** | **10** |

Notes

1. Results apply to all fish tissue types: muscle, eggs, internal organs, etc.
2. Arithmetic means of adult fish (Azimuth 2011)

## 4.2 Consumption of Fish from the Site C Reservoir

It is estimated that methylmercury concentrations in fish in the proposed Site C reservoir will, at peak levels, be four times higher than baseline pre-impoundment concentrations (EIS Section 11.9). Correspondingly, the maximum number of servings of fish from the proposed Site C reservoir that can be consumed without exceeding Health Canada's pTDI for methylmercury will decrease by a factor of four.

Bull trout have the highest mercury levels among the fish species in the proposed Site C reservoir reach of the technical study area that are commonly consumed by humans. Bull trout from the reach of the Peace River that will be impounded by the Site C reservoir have an average mercury concentration of 0.072 mg/kg wet weight. While the highest concentration of mercury in the technical study area is found in bull trout, this concentration is lower than for other sport fish species and lower relative to bull trout and other piscivorous species (lake trout) from all other BC lakes (Rieberger 1992), reservoirs (Baker 2002) and in wild fish from other parts of Canada (Depew et al. 2012).

### 4.2.1 Baseline Consumption Frequency

**Table 3** quantifies the number of fish meals, assuming baseline, pre-impoundment concentrations, by species, that various receptors (e.g., children, teens, non-child bearing adults) on a weekly basis. Data for key sport species within the proposed Site C reservoir area of the Peace River include bull trout and rainbow trout. Mountain whitefish are a key food chain species. Walleye, goldeye and northern pike are rare within the Site C reach of the Peace River and are more abundant in the downstream technical study area.

*Bull trout* – Adults could consume between eight (women of child bearing age) and 20 servings per week (all other adults) of bull trout without exceeding Health Canada's pTDI for methylmercury. Toddlers, the most sensitive age group, can consume up to four servings per week of bull trout without exceeding Health Canada's pTDI for methylmercury.

*Rainbow trout* – Adults could consume between nine (women of child bearing age) and 32 servings per week (all other adults) of rainbow trout without exceeding Health Canada's pTDI for methylmercury. Toddlers, the most sensitive age group, can consume up to seven servings per week of rainbow trout without exceeding Health Canada's pTDI for methylmercury.

*Lake trout* – Lake trout are rare within the Peace River technical study area, but are more common in Dinosaur Reservoir, upstream of the Peace Canyon Dam. Baseline consumption frequency of lake trout from this reservoir is also high, up to 22 meals per week by non-childbearing or pregnant adults, yet as many as 5 weekly meals as a toddler.

*Mountain whitefish* – Mountain whitefish are likely consumed very infrequently. Nevertheless, mercury concentrations in this species are very low (0.03 mg/kg) so that consumption is very high for all receptor groups.

### 4.2.2 Proposed Site C Reservoir Consumption Frequency

The peak increase factor of 4x baseline as derived by the Site C EIS Section 11.9 Methylmercury is used to determine the reduction in consumption frequency for key fish species within the proposed Site C reservoir. This increase factor recognizes the uncertainty in making such predictions and was increased from 3x to 4x to provide an extra measure of conservatism to be protective of human health. Furthermore, the increase factor derived in Section 11.9 was derived based on the difference between baseline and *peak* increases in predicted fish mercury concentrations within the proposed reservoir. These peaks are only forecast to persist for a 2 – 3 year window, so there is an additional level of conservatism here as well. Changes to fish consumption frequencies within the proposed Site C reservoir are summarized below, with detailed information in **Table 3**:

*Bull trout* – Adults could consume between two (women of child bearing age) and five servings per week (all other adults) of bull trout without exceeding Health Canada's pTDI for methylmercury. Toddlers and children, the most sensitive age groups, can consume one serving per week of bull trout without exceeding Health Canada's pTDI for methylmercury.

*Rainbow trout* – Adults could consume between three (women of child bearing age) and eight servings per week (all other adults) of rainbow trout without exceeding Health two to three meals per week of rainbow trout without exceeding Health Canada's pTDI for methylmercury.

*Mountain whitefish* – Consumption frequency of mountain whitefish within the proposed reservoir would remain high, ranging between two (toddlers) and nine meals per week (adults) within the proposed reservoir.

## 4.3 Consumption of Fish from Downstream of the Site C Dam

For the purposes of this HHRA, Section 11.9 Methylmercury of the EIS determined that it was possible that fish mercury concentrations downstream of the proposed Site C dam might double baseline concentrations. Given that some fish species such as walleye and goldeye range within the Peace River between Many Islands Alberta and as far upstream as the proposed dam location, it is feasible that, at peak levels, fish may double baseline mercury concentrations and may range as far downstream as Many Islands. Under this assumption, baseline consumption frequencies presented in Table 3 will decrease by a factor of two.

Goldeye (0.24 mg/kg) and walleye (0.18 mg/kg) have the highest mean baseline mercury concentrations among the fish species that have been sampled from the Peace River technical study area downstream of the proposed Site C dam. Goldeye and walleye do migrate within the Peace River below the Site C dam and that mercury concentrations in these species peak at double their pre-construction levels, women of child bearing age could consume one serving a week, while other adults could consume three servings a week. Toddlers could consume about half a serving a week of goldeye without exceeding Health Canada's pTDI for methylmercury. Similar results were found for walleye.

Bull trout are predicted to be present downstream of the proposed Site C dam. At peak post-impoundment mercury levels, women of child bearing age could consume four servings a week, other adults could consume nine servings a week, and toddlers could consume two serving a week of bull trout from the Peace River downstream of the Site C dam without exceeding Health Canada's pTDI for methylmercury.

There are no data on pre-impoundment concentrations of mercury in lake trout or from northern pike from the Peace River downstream of the Site C dam site. However, assuming these species may have similar pre-impoundment mercury concentrations as bull trout, at peak post-inundation mercury levels, women of child bearing age could consume five servings a week. Other adults could consume 11 servings a week, while toddlers could consume two serving a week of lake trout or northern pike from the Peace River downstream of the Site C dam without exceeding Health Canada's pTDI for methylmercury.

### 4.3.1 Duration of Increased Mercury in Fish

Results of the HHRA are based on the forecast *peak* methylmercury concentrations in fish that will be experienced 3 – 8 years after impoundment is achieved (Volume 2 Section 11.9 Methylmercury). Methylmercury concentrations are expected to gradually decline from these peak levels over time, returning to a baseline condition within 10 – 15 years after the peak (20 – 25 y after construction). Therefore, risks estimates in the HHRA will also correspondingly start to diminish 3 – 8 years after impoundment of the proposed reservoir.

## 4.4 Uncertainties

Results of the HHRA for methylmercury are based on exposure estimates calculated by predictive models. While the models are based on the best available scientific data, as with any scientific evaluation, there are uncertainties in how exposures are modeled as well as the data that are used to populate the models. The HHRA for methylmercury is also based on estimates of future concentrations of mercury in the environment and future human behaviors. There are, necessarily, uncertainties in these predictions. Key areas of uncertainty in this HHRA include:

- Quantitative exposure estimates were not derived for potential human exposures to methylmercury in sediments, water, fish-eating wildlife, or non-muscle fish tissues. Post construction methylmercury concentrations in water and sediment are expected to be well below the applicable guidelines. The potential frequency of consumption of fish-eating birds (e.g., loon) is very low and is not considered a meaningful source of methylmercury exposure. Given the available the margin of error before a potential human health risk may occur from exposure to these media, any uncertainty between predicted and actual concentrations of mercury in these exposure media is inconsequential.
- The concentration of methylmercury in the fish size assumed to be most frequently captured and consumed by humans was used to determine meal consumption frequency that is protective of health. Consistent consumption of larger fish (with higher mercury concentrations) may incrementally increase risk, however the relative difference in mercury concentration between moderate and large size fish was small. This uncertainty was also taken into consideration within Section 11.9 of the EIS, as a 4x increase above baseline has been used, rather than a 3x increase.
- The body weight of humans consuming fish was based on Health Canada guidelines. Adults that are larger than the 70 kg male and 60 kg female guideline can consume proportionally more fish without incrementally higher potential risk. The reverse would be true for humans of below average size.
- The average serving size of fish (g) followed Health Canada guidelines. Individuals that consume larger portion sizes would receive an incrementally higher dose of methylmercury. However, body weight is also positively correlated with body size. Since the variability in serving size and body weight are expected to be small and because they will generally cancel each other out, the variability and uncertainty in these parameters is small relative to the overall conclusions of the HHRA.

These uncertainties may result in both under-estimates as well as over-estimates of the potential risks from methylmercury exposure. As per HHRA guidance from Health Canada and best professional practice, conservative, or health protective assumptions were generally made where there was uncertainty in the HHRA. Because the effect of conservative assumptions is often compounded, this provides a high degree of certainty that health risks are not underestimated.

# 5 SUMMARY AND DISCUSSION

All Canadians are exposed to some mercury in the environment and, for the vast majority, the greatest source of environmental methylmercury exposure is from eating fish. All fish contain methylmercury, with higher concentrations found in large, longer-lived predatory species such as bull trout and lake trout. Therefore, a person's methylmercury exposure depends on how frequently they eat fish, the serving size and the species, age and size of fish that they eat. Risk is also relative to a person's age because the developing nervous system of a fetus or a child is more susceptible to the effects of methylmercury than that of an adult.

Health Canada has defined tolerable levels of exposure to methylmercury. Frequent consumption of large servings of some types of sport-caught or retail fish, such as swordfish or fresh tuna, may result in methylmercury exposure in excess of Health Canada's pTDI for methylmercury. However, it must also be recognized that exceedences of guidelines does not necessarily mean that negative effects should be expected.

Currently, baseline mercury concentrations in all popular sport and food species from the Peace River technical study area are low, even for large fish, with average concentrations less than 0.1 mg/kg wet weight (Azimuth 2011; Volume 2 Appendix J Part 1). At these concentrations all species including bull trout, rainbow trout, lake trout, mountain whitefish, northern pike and burbot can be consumed regularly (several times per week), even by the most sensitive age group (i.e. toddlers less than 5 years old) without exceeding Health Canada's pTDI for methylmercury.

Inundation of the Peace River to create the proposed Site C reservoir is expected to cause mercury levels in fish to rise temporarily. Mercury levels in fish are expected to peak at 5 – 8 years following inundation of the reservoir and then slowly return to baseline levels over the following 10 – 20 years. Volume 2 Section 11.9 Methylmercury shows predictions of methylmercury concentrations in fish from the proposed Site C reservoir to peak between three and four times their pre-inundation concentrations, depending on the species. Downstream of the proposed dam, fish mercury concentrations in the Peace River downstream to as far as Many Islands may peak at double pre-inundation concentrations. The number of servings of fish per week that people can consume from the technical study area without exceeding Health Canada's pTDI for methylmercury will correspondingly decrease during the period of post-impoundment elevations in methylmercury levels in fish.

Bull trout have the highest mercury levels among the fish species in the proposed reservoir area of the Peace River technical study area that are commonly consumed by humans. At peak post-impoundment mercury levels, women of child bearing age could consume two servings a week, other adults could consume five servings a week, and toddlers could consume one serving a week of bull trout without exceeding Health Canada's pTDI for methylmercury. Rainbow trout can be consumed very regularly from the proposed Site C reservoir, even during peak post-impoundment mercury levels, up to three times a week by women of child bearing age, eight times a week by other adults and up to twice a week by toddlers without exceeding Health Canada's pTDI for methylmercury.

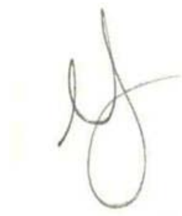
Mercury concentrations in fish from downstream of the proposed Site C dam are not expected to increase as much as in fish from within the proposed reservoir. At peak post-impoundment mercury levels for bull trout, women of child bearing age could consume four servings a week, other adults could consume nine servings a week, and toddlers could consume two serving a week without exceeding Health Canada's pTDI for methylmercury. Goldeye and walleye from lower reaches of the Peace River have higher pre-inundation concentrations of mercury than bull trout in the technical study area. Assuming a doubling of baseline concentrations consumption frequency ranges from one meal per week (toddler) to 4 meals per week for adults who are not pregnant.

Results of the HHRA are based on the forecast *peak* methylmercury concentrations in fish that would be experienced 5 – 8 years after impoundment. Methylmercury concentrations are expected to gradually decline from these peak levels and the number of servings of fish a week that can be consumed within the technical study area without exceeding Health Canada's pTDI for methylmercury will also correspondingly start to increase after peak mercury levels start to decline.

This HHRA, as with any predictive risk assessment of future exposure scenarios, contains elements of uncertainty. Conservative, or health protective, assumptions were generally used to address these uncertainties and the net effect is that the human health risks associated with exposure to methylmercury from the construction and operation of the proposed Project are unlikely to have been underestimated.

While mercury levels in fish would temporarily increase following inundation of the Site C reservoir, it must be kept in mind that fish is high in nutritional value and consumption of fish has been shown to protect health and promote healthy development. Some commonly consumed species of fish, such as rainbow trout, could be consumed, by even the most sensitive age group, at least twice a week without exceeding Health Canada's pTDI for methylmercury. The potential health risks associated with MeHg exposure from fish consumption need to be carefully weighed against the health benefits of fish consumption. The objective of risk management to minimize exposure to methylmercury should not be to reduce fish consumption, but to encourage consumption of fish species with lower levels of methylmercury, if some consumers continue to have concerns about mercury.

Authorship:



Senior Author

Norm Healy, M.Sc., R.P.Bio. Principal
Azimuth Consulting Group

Co- Author

Randy Baker, M.Sc., R.P.Bio. Principal
Azimuth Consulting Group

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# SITE C CLEAN ENERGY PROJECT

## VOLUME 2 APPENDIX J, PART 3

## MERCURY RESERVOIR MODELING

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*Prepared for Azimuth Consulting Group*

*Prepared by*

*Reed Harris Environmental Ltd.*

**December 2012**

| | |
|---|---|
| Lead Author: | Reed Harris |
| Section Contributors: | Dr. David Hutchinson – Simulations |
| | Mr. Don Beals - Simulations |

## Authorship

Reed Harris, B.Sc.(Civ. Eng), M.Sc. (Civ. Eng.), P.Eng. .................................................. Proprietor

## ACRONYMS

Hg ... mercury

MeHg ... methylmercury

Inorganic Hg(II) ... total mercury minus methylmercury

Hg(0) ... elemental mercury

µg ... micrograms

ng ... nanograms

RESMERC ... Reservoir Mercury Model

DOC ... dissolved organic carbon

TOC ... total organic carbon

ww ... wet weight

dw ... dry weight

EIS ... Environmental Impact Statement

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# EXECUTIVE SUMMARY

This report describes the application of a process-based model (RESMERC) to predict mercury (Hg) and methylmercury (MeHg) concentrations, cycling and bioaccumulation in aquatic environmental media within the proposed Site C reservoir, created as part of the proposed Site C Clean Energy Project (the Project) in British Columbia. RESMERC modeling was one of three approaches applied as part of an integrated approach to determine the most likely magnitude of increase in fish Hg concentration with the Site C reservoir and the duration that this phenomenon is expected to occur.

Prior to applying the RESMERC model, a regression model derived from results of 12 Canadian reservoirs was used to predict peak fish Hg concentrations for the Site C reservoir (Harris and Hutchinson 2012). This model predicts the maximum relative increase in fish mercury concentrations for a new reservoir as a multiplier of baseline concentrations, but does not predict timing of the response. This method uses only total flooded area, original area and mean annual hydraulic residence time to predict the peak increase factor (e.g. 5X). Using site-specific Site C factors, the regression model predicted that fish Hg concentrations would increase by 2.3X above baseline, based on long-term discharge rates from Williston Reservoir. When applied to a mean baseline Hg concentration for bull trout, for example (0.09 mg/kg), the predicted peak Hg concentration for a 500 mm bull trout is 0.20 mg/kg.

To provide a more robust and time-sensitive model to Site C, RESMERC was applied. Prior to application at the Site C Project, the model was first calibrated to historic fish Hg observations for northern pike and lake whitefish from the Notigi Reservoir, Manitoba and the Robert Bourassa Reservoir, Quebec; both constructed in the 1970s. This was necessary to 'scale-up' RESMERC from small, experimental reservoirs to full-scale reservoirs. Once calibrated, the updated model was then applied to Site C to predict pre- and post-inundation Hg and MeHg concentrations in environmental media. Pre-inundation simulations were performed first, using baseline information collected in the proposed Site C technical study area. Once it was demonstrated that RESMERC was able to mimic current, baseline conditions, the model was run to predict conditions within Site C reservoir.

Pre-inundation calibrations agreed well with baseline observations, in terms of total Hg and MeHg in water and sediments and MeHg concentrations in epibenthos and fish. Water column MeHg concentrations within the proposed reservoir are predicted to peak at roughly double baseline concentrations, to 0.04 ng/L, with short-term increases up to 0.06 ng/L.

These concentrations are within the range of background concentrations for natural North American boreal waters. Sediment MeHg concentrations are predicted to increase more than the water column in relative terms, into the range of 0.020 to 0.025 mg/kg dw at peak levels.

Simulations of the proposed Site C reservoir predicted peak MeHg concentrations in adult bull trout of approximately 0.45 mg/kg wet muscle, 8 to 10 years after inundation, then declining to regional background concentrations approximately 20 years later. Lower trophic level fish species (suckers, mountain whitefish, redside shiner) were predicted to have peak concentrations in the range of 0.12 to 0.17 mg/kg wet muscle. The short hydraulic residence time for the proposed Site C reservoir and the influence of nutrient-poor, low Hg water from upstream from Williston Reservoir (Volume 2, Appendix J, Part 1, Mercury Technical Synthesis Report) has the potential to reduce the recovery time from peak to background fish Hg concentrations than is predicted by the model.

RESMERC predicted that fish Hg concentrations in the proposed Site C reservoir may peak at concentrations that are 2 – 6 times higher than baseline concentrations, depending on fish species and fish size. Predicted peak fish Hg concentrations are less than 0.5 mg/kg for all species modeled, and are at the low end of the range observed from central and eastern Canadian reservoirs and lakes (Depew et al. 2012). Reasons for predictions being towards the lower end of observations include low baseline fish Hg concentrations in the Peace River, a short hydraulic residence time (23 days) for the proposed reservoir, low inflowing Hg and MeHg concentrations from Williston and Dinosaur reservoirs and several other physical / chemical characteristics predicted for the proposed reservoir that do not favor high mercury methylation rates (Volume 2, Appendix J Mercury Technical Reports, Part 1, Mercury Technical Synthesis Report).

# 1 INTRODUCTION

This report describes application of a process-based model (RESMERC) to predict mercury (Hg) and methylmercury (MeHg) concentrations, cycling and bioaccumulation in aquatic environmental media within the Site C reservoir, created as part of the proposed Site C Clean Energy Project (the Project) in British Columbia.

## 1.1 Background

The RESMERC model has been applied as one of three lines of evidence used to predict changes in MeHg concentrations in water, sediments and biota in the proposed Site C reservoir on the Peace River. The proposed Project would be the third reservoir on the Peace River, located just downstream of Williston Reservoir, created in 1968 and Dinosaur Reservoir, created in 1980. Water discharged from Williston and Dinosaur Reservoirs is currently nutrient poor and has low concentrations of Hg and MeHg (see Volume 2, Appendix J, Part 1, Mercury Technical Synthesis Report).

Considerable knowledge has been gained over the last 30 – 40 years to understand how the creation of new reservoirs influences the dynamics of MeHg production, cycling and bioaccumulation in the aquatic food web. Each reservoir has unique physical, chemical and ecological conditions and there is no single accepted tool or method to forecast what will happen within different reservoirs. For this reason, several lines of evidence were used to determine the most likely magnitude of change in MeHg concentrations in environmental media within the proposed Site C reservoir. Two of these lines are presented here, while the third line of evidence is presented as the Canadian reservoirs comparison matrix, as Part 1 of the Mercury Technical Synthesis Report in Volume 2, Appendix J. Further details on each of these tools employed at Site C are as follows:

- Canadian Reservoir Comparison Matrix – Chapter 5 of the Mercury Technical Synthesis Report (Volume 2, Appendix J, Part 1) undertook a comprehensive review of many key physical, chemical and ecological factors that are associated with creating conditions that enhance mercury methylation in reservoirs. Fifteen large reservoirs from Manitoba, Quebec, BC and Labrador were evaluated. Baseline and predicted values for these parameters from the Site C technical study area were contrasted against what has been observed elsewhere in Canada, to put the Site C Project in perspective with other large Canadian hydroelectric projects, with a focus on changes in fish Hg

- Screening level regression model – This is a linear regression model (Harris and Hutchinson 2012) that used three input parameters: flooded area, total area and mean annual flow to predict the maximum relative increase of fish Hg concentrations above regional background levels, when peak concentrations occur. This model does not predict the timing of the peak nor the return to a baseline condition Results of this exercise are presented as Appendix A in this document.
- RESMERC Model – The RESMERC model (Harris and Hutchinson 2009; Harris et al. 2009) presented in this report, is a process-based model that predicts the effects of inundation on mercury cycling and bioaccumulation in reservoirs, as a function of time. Key outputs include predictions of Hg and MeHg concentrations in water and biota (e.g., invertebrates, fish) at any point in time, in this case, within the proposed Site C reservoir.

Taken together, these documents determine the incremental potential risk to humans from exposure to methylmercury from fish consumption before and after construction and operation of the proposed Site C reservoir. These results are described in the Human Health Risk Assessment, Volume 2, Appendix J, Part 2.

## 1.2 Objectives

The primary objective of this modeling study was to use RESMERC to estimate concentrations of total Hg and MeHg in various environmental media in the proposed Site C reservoir. Environmental media included water column and flooded soils (sediments), lower trophic level biota and fish.

Combined with the Harris and Hutchinson (2012) regression model (Appendix A) and a comparison with reservoir features associated with higher or lower fish Hg levels (Volume 2, Appendix J, Part 1, Mercury Technical Synthesis Report), this modeling study was a key component towards estimating how fish Hg levels would change following construction and operation of the proposed Site C reservoir.

## 1.3 Mercury in New Reservoirs

Reservoir creation has been well documented to result in increased fish mercury (Hg) concentrations (Bodaly et al. 2007; Jacques Whitford 2006; Schetagne et al. 2003; Bodaly et al. 1997; Canada-Manitoba Governments 1987). While most Hg in the environment is inorganic, some is converted in aquatic systems to MeHg. This newly formed MeHg is absorbed at low levels of the food web (i.e., bioaccumulation) and then becomes more concentrated at higher trophic levels in the aquatic food web (i.e., biomagnification), reaching the highest concentrations in fish. More detail on MeHg cycling and methylation and how the understanding of this process is applied to the proposed Site C reservoir can be found in the Mercury Technical Synthesis Report in Volume 2, Appendix J, Part 1.

The primary cause of increased fish MeHg concentrations in new reservoirs is driven by accelerated, bacterially mediated conversion of inorganic Hg into MeHg during the decomposition of organic matter in flooded areas. An indicator of this efficiency is the fraction of total Hg that is MeHg in water or sediments. MeHg typically represents approximately 5 to 10% of total Hg in the surface waters in remote lakes, but can rise to more than 70% in new reservoirs (St. Louis et al. 2004). Because MeHg is much more concentrated in fish than in water (e.g., > 1 million times) fish consumption remains the primary pathway for exposure of MeHg to humans.

The duration of elevated fish MeHg concentrations in boreal reservoirs typically lasts two to three decades, with peak MeHg concentrations in some higher tropic level fish species of between 4 – 7 times greater than background levels (Bodaly et al. 2007; Schetagne et al. 2003) (Figure 1-1 c and d, Figure 1-2 bottom panel). Lower trophic level fish species such as lake whitefish tend to have lower concentrations (compare Figure 1-1a and b, Figure 1-3) and slightly lower relative increases at 2 to 5 times above regional baselines.

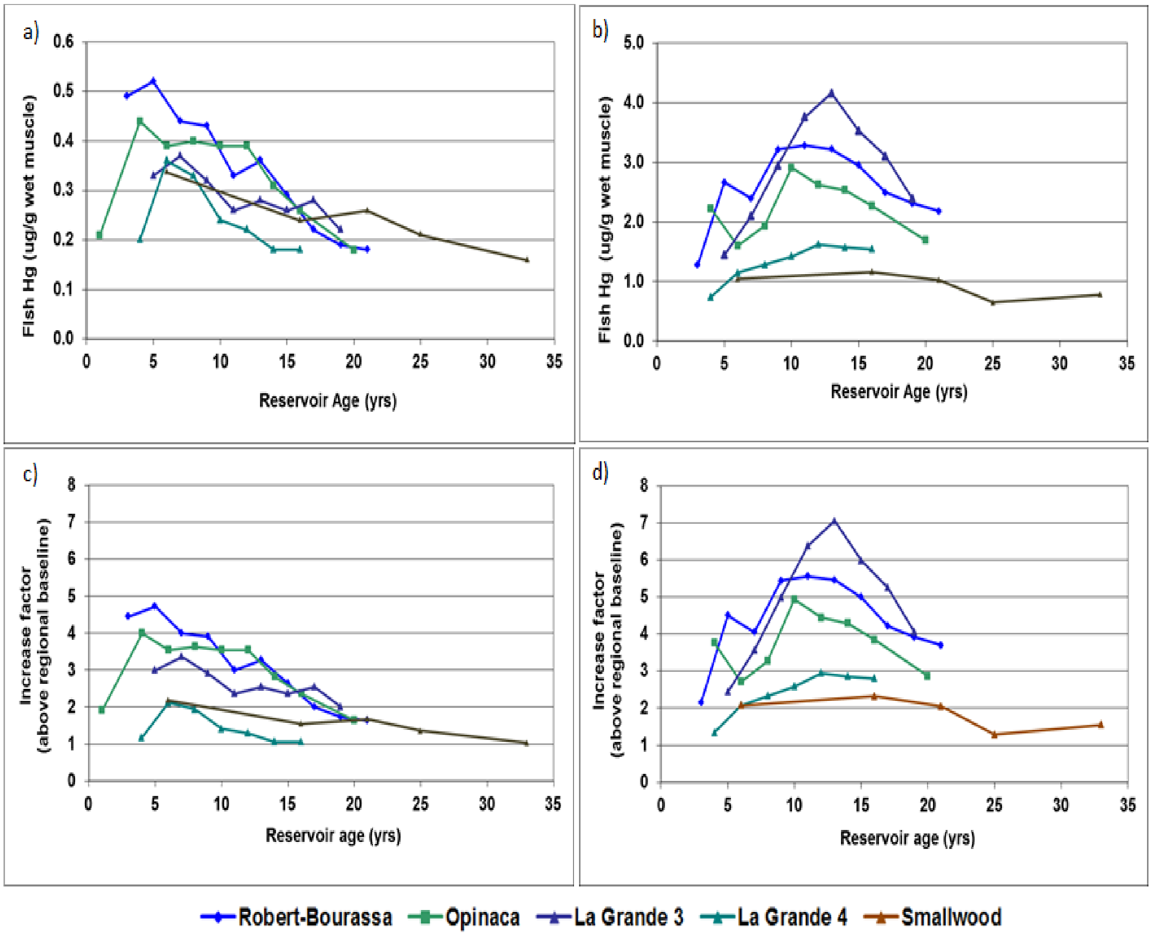



**Figure 1-1. Hg levels (mg/kg) in 400 mm lake whitefish and 700 mm northern pike as a function of reservoir age in Quebec and Labrador.**

Note: a) and c) are lake whitefish, b) and d) are northern pike. Relative increases are with respect to regional baselines: 0.55 - 0.59 µg/g (equivalent to mg/kg) for Quebec, 0.5 µg/g average from Atikonak Lake and Shipiskan Lake for Labrador Quebec data from Schetagne et al. (2003). Smallwood Reservoir data from Jacques Whitford (2006) for Lobstick and Sandgirt lakes.

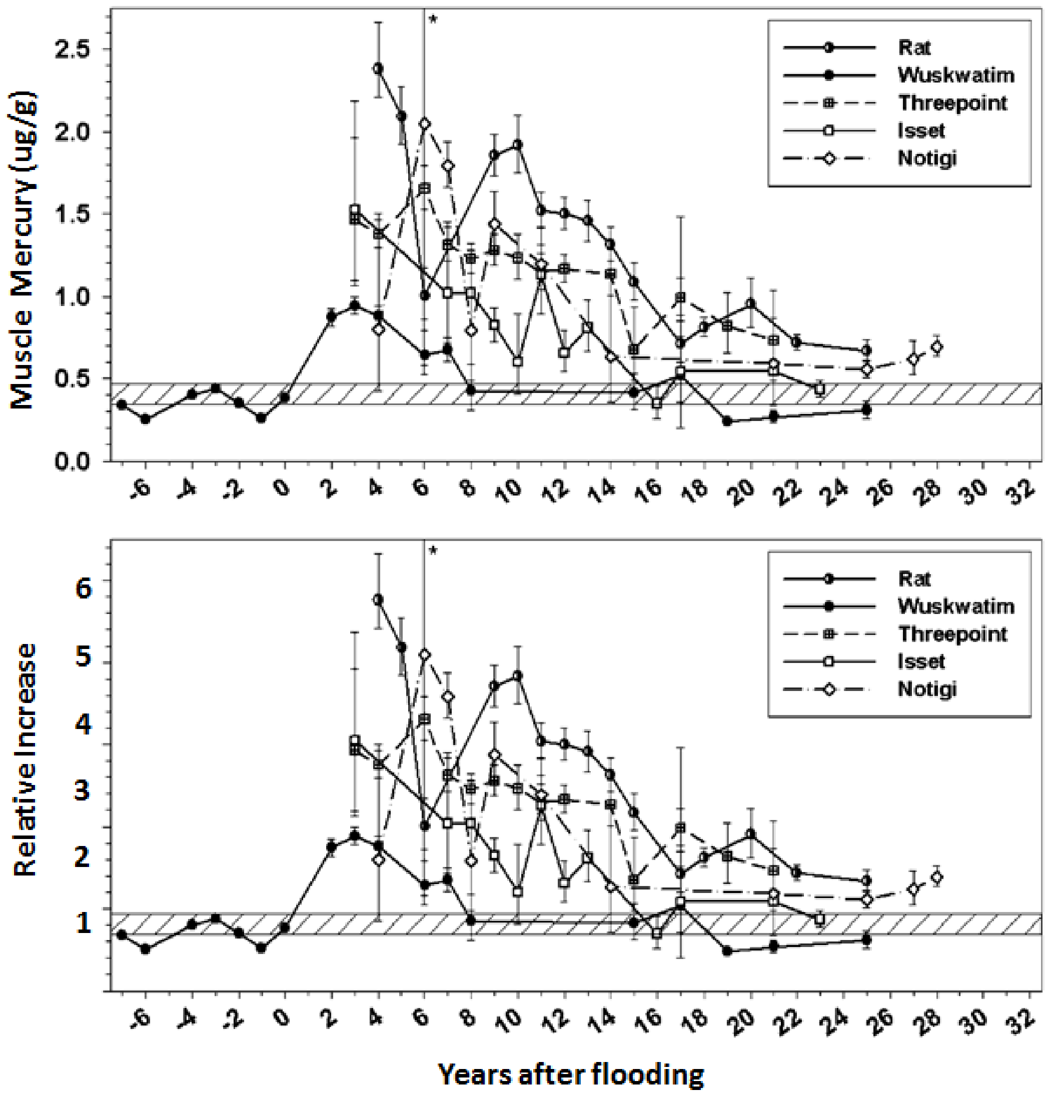



**Figure 1-2. Hg levels in 400 mm walleye in selected reservoirs in the Rat Burntwood system, Manitoba as a function of reservoir age.**

Data are derived from Bodaly et al. (2007). Hatched area is background range for lakes.

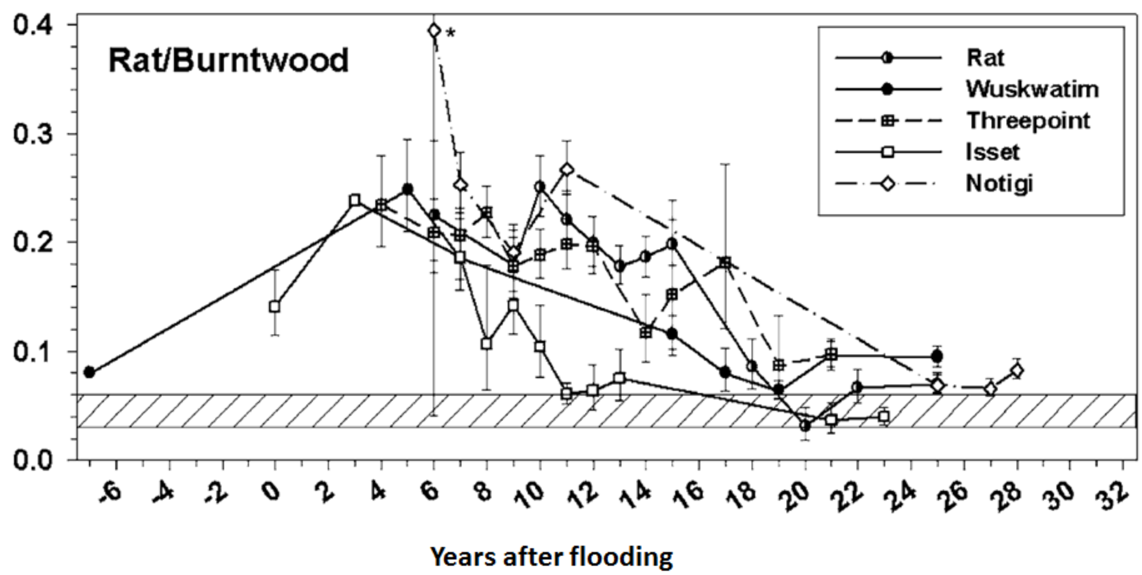



**Figure 1-3. Hg concentrations in 350 mm lake whitefish in selected reservoirs in the Rat Burntwood system, Manitoba as a function of reservoir age.**

Data from Bodaly et al. 2007. Hatched area is background range for lakes.

The response of fish Hg concentrations following inundation is influenced by many reservoir-specific features, especially the type of terrain flooded and hydraulic residence time (Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1). Results from a series of shallow experimental reservoirs in flooded uplands and wetlands at the Experimental Lakes Area in Ontario (Hall et al. 2005; St. Louis et al. 2004; Bodaly et al. 2004) suggested that flooded wetlands produce MeHg at comparable but higher rates than flooded uplands (e.g. ~2X), and provide a longer term supply of carbon for decomposition and increased MeHg production. Short hydraulic residence times should, on the other hand, result in lower MeHg concentrations in surface waters of reservoirs via dilution, but create the potential for greater downstream transport. These factors, and how they contribute to MeHg bioaccumulation, are more fully explored in the Mercury Technical Synthesis Report (Volume 2, Appendix J, Part 1).

There are also differences in the magnitude and timing of the response of MeHg concentrations among ecosystem compartments and trophic levels in the same reservoir. Peak MeHg concentrations occur sooner in the water column, in lower trophic level organisms and young fish, and later in top predators, such as adult walleye and northern pike (Bodaly et al. 2007; Schetagne et al. 2003). These trends are consistent with a pulse in

MeHg production that peaks within a few years after inundation, and then takes time to move through the food web to adult top predators. Because fish obtain most of their MeHg via the diet (Hall et al. 1997; Harris and Bodaly 1998), increases in MeHg levels in adult predatory fish such as northern pike lag until MeHg increases in their diet (e.g. smaller fish). Adult northern pike, for example, may take about 5 to 15 years to reach peak levels (Schetagne et al. 2003), while lake whitefish typically reach peak concentrations and start to decline years sooner (Figure 1-1). For a given species, younger fish also respond faster than adults.

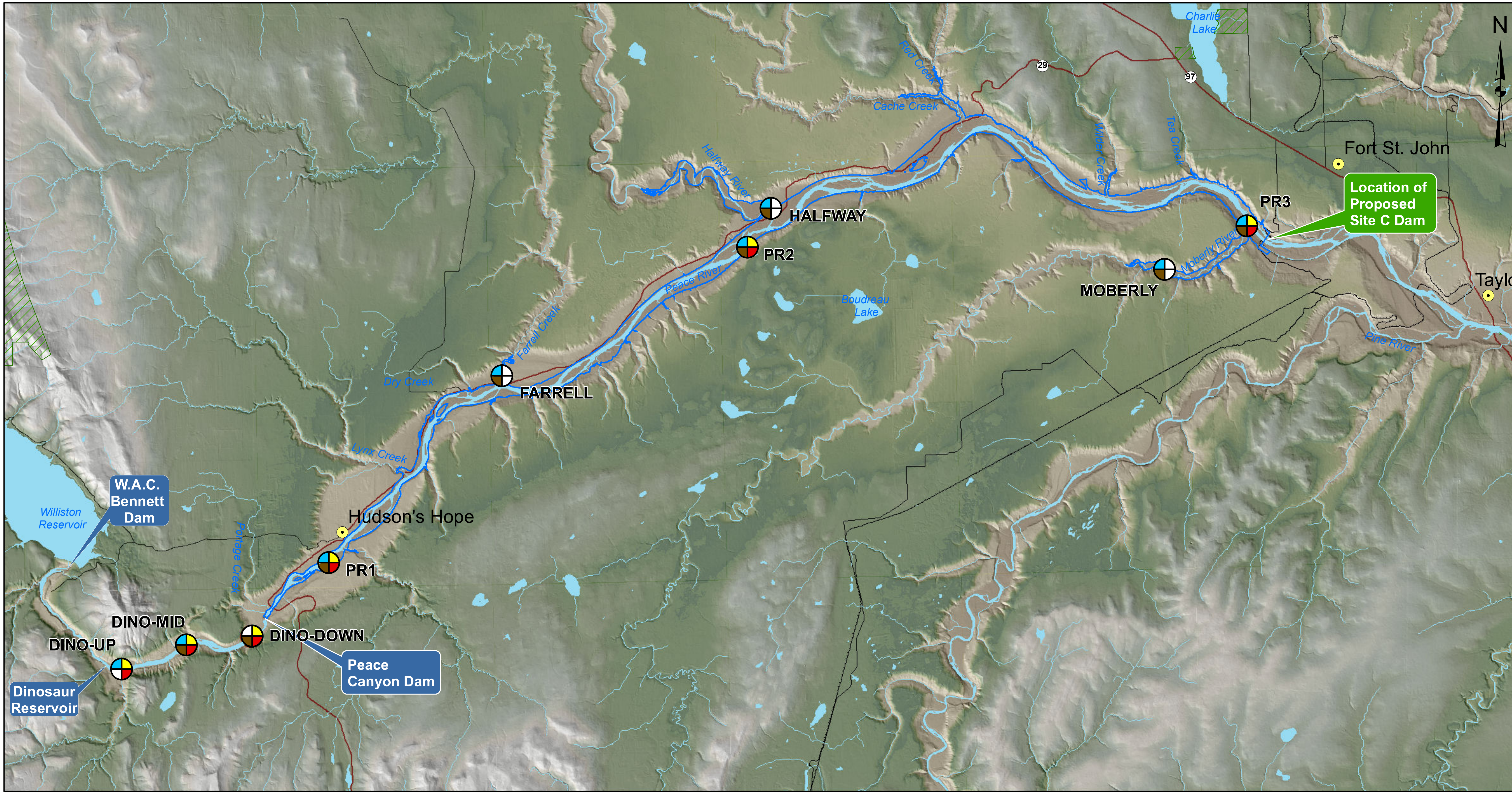
Fish Hg levels downstream of some reservoirs can increase to levels comparable and sometimes greater than in upstream reservoirs (Schetagne et al. 2003; Anderson 2012). Existing observations from boreal reservoirs suggest that increased Hg concentrations can occur in fish downstream of some reservoirs until a large waterbody is encountered, where dilution and natural processes such as sedimentation and photochemical degradation reduce levels of MeHg in waters travelling downstream. Fish migration also has the potential to produce scenarios where fish are exposed to elevated MeHg levels near reservoirs and then travel downstream. Thus, the distance downstream of reservoirs where increased fish Hg levels occur likely depends on system-specific features. The extent to which this may occur at Site C is explored in the Mercury Technical Synthesis Report (Volume 2, Appendix J, Part 1).

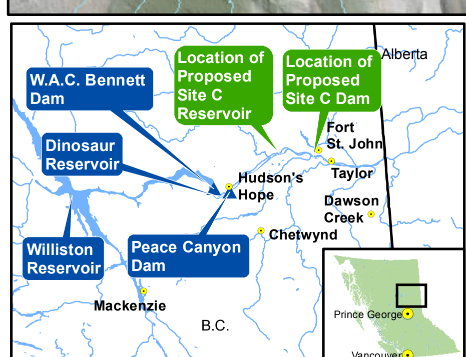
# 2 PROJECT DESCRIPTION

## 2.1 Proposed Reservoir Features

The proposed Site C Project would be the third hydroelectric development on the Peace River system in British Columbia (Figure 2-1). The Site C dam and generating station (GS) would be located approximately seven km southwest of Fort St. John, and one km downstream of the existing Moberly River confluence. Basic characteristics of the Site C reservoir relevant to RESMERC simulations are given in Table 2-1. The proposed reservoir would be 83 km long, with a total area of 93.3 $km^2$ and a flooded area of 53.4 $km^2$ (57% of total area) (EIS Section Volume 1, 2.0 Project Description). Less than 2% of the area proposed to be inundated is wetland habitat (Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1). Mean hydraulic residence time is on average predicted to be 23 days. Thermal stratification is predicted to occur in the summer in the water column in the downstream portion of the reservoir, but not the upper 25 km of the reservoir (Figure 2-2;

Volume 2 Appendix H, Reservoir Water Temperature and Ice Regime Technical Report). Oxygen depletion in the water column is not predicted. The water pH is also assumed to remain in the current range (pH 8) because of the influence of Williston Reservoir upstream. No estimates are available for predicted concentrations of dissolved organic carbon (DOC); however, current baseline DOC levels are low, 2-3 mg/L and future reservoir DOC levels are assumed to be strongly influenced by DOC levels in water discharged from Williston Reservoir. See the Water Quality Technical Data Report, Volume 2, Appendix E for a discussion of baseline water quality.




Map Notes:
1. Datum: NAD83
2. Projection: UTM Zone 10N
3. Base Data: Province of B.C.
4. Proposed reservoir area (461.8 m maximum normal elevation) from Digital Elevation Models (DEM) generated from LiDAR data acquired July/August 2006.

**Legend**

Parameters Collected

Water — Zooplankton — Sediment — Benthos

Maximum Normal Reservoir Level (461.8 m)

Protected Areas, Provincial Parks, Regional Parks, and Ecological Reserves

City/District Municipality

Existing Road

Existing Highway

Existing Railway

1:250,000 — 0 — 10 km

BC hydro

SITE C CLEAN ENERGY PROJECT

**Figure 2.1**
**Water, sediment, zooplankton & benthic invertebrate sampling locations, Peace River to support mercury modeling**

| Date | Dec. 10, 2012 | DWG NO | 1016-C14-B6299 | R 0 |
|---|---|---|---|---|



Construction of the Site C Clean Energy Project is subject to required regulatory approvals including environmental certification

**Table 2-1. Key characteristics and data sources for simulations of the proposed Site C reservoir.**

| Characteristic | Value | | Source |
|---|---|---|---|
| | **Pre Flood** | **Post-Flood** | |
| Length (km) | 83 | 83 | EIS Vol 1 Section 2 |
| Total area ($km^2$) | 39.7 | 93.3 | EIS Vol 1 Section 2 |
| Flooded area ($km^2$) | 0 | 53.4 | EIS Vol 1 Section 2 |
| Percent of reservoir area that is flooded | N/A | 57 | EIS Vol 1 Section 2 |
| Percent of flood zone area that is wetland | N/A | 1.6 | Azimuth (2011) and Appendix V2J Part 1 |
| Maximum annual water level fluctuations (m) | N/A | 1.8 | EIS Vol 1 Section 2 |
| Mean hydraulic residence time (days) | <1 | 23 | EIS Vol 1 Section 2 |
| Annual temperature range in surface waters (°C) | 2 - 14 | 0-18 | Appendix V2H |
| Vertical stratification in water column | No | Yes in summer, lower reach only | Appendix V2H |
| Suspended solids in water column (mg/L) | <3 except freshet or storms | <3 except freshet or storms | Appendix V2E Post flood: Assumed similar to pre-flood |
| Oxygen depletion in water column | No | No | Appendix V2E |
| Predicted pH | 8 | 8 | Pre flood: Appendix V2E Post flood: Assumed similar to current value |
| Initial carbon pool in flood zone (Kg C/$m^2$ in upper 5 cm) | N/A | 0.5-1.2 | Appendix V2J Part 1 |
| Initial Hg concentration in flood zone (mg/kg dw) | | 0.079 | Appendix V2J Part 1 |

Note: EIS Volume 1 Section 2 refers to the Site C Project Description; Appendix V2J Part 1 is the Mercury Technical Synthesis Report; Appendix V2E is the Baseline Water Quality Report Appendix V2H is the Water Temperature and Ice Regime Report.

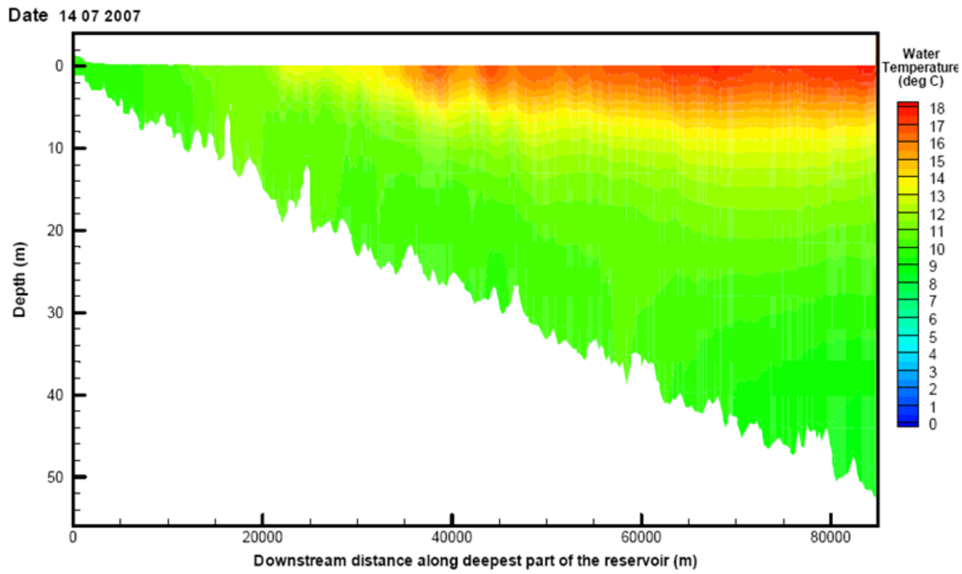



**Figure 2-2. Predicted temperature regime in mid-summer for Site C reservoir.** Source is Volume 2, Appendix H, Water and Ice Regime Technical Data Report

## 2.1 Technical Study Area

Within the Site C reservoir, changes will occur between Peace Canyon Dam to the proposed dam site and in the lower reaches of the larger tributaries (Halfway and Moberly). Changes may also extend downstream of the Site C dam, potentially including changes to Hg and MeHg concentrations in water and biota. Downstream increases in fish Hg concentrations have been described by Schetagne et al. (2000, 2003) in some Quebec reservoirs and in the Churchill River downstream of Smallwood Reservoir, Labrador (Anderson 2011). The distance downstream to which increased fish Hg concentrations may occur depends on the downstream transport of Hg and fish movement patterns. Local fish populations have been shown to migrate upstream to the proposed Site C dam site from as far downstream as the area of Many Islands Alberta (Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1) based on fish tagging studies (EIS Section 12). Modeling presented in this study discusses changes to Hg concentrations within the proposed Site C Reservoir. The potential for increased fish mercury concentrations downstream of the proposed Site C dam is described in the Mercury Technical Synthesis document (Volume 2, Appendix J, Part 1).

# 3 MECHANISTIC MODEL OVERVIEW

RESMERC is a process-based simulation model for reservoirs and lakes (Harris et al. 2009). Model compartments include the water column, sediments, and a simplified food web that consists of several trophic levels (phytoplankton, zooplankton, benthos and up to four fish species) (Figure 3-1). The model predicts concentrations, mercury pools and major fluxes for each mercury form (i.e., inorganic Hg, MeHg) through time.

An overview of the major processes involved in the Hg cycle in reservoirs is shown in Figure 3-1. Hg processes represented in RESMERC include atmospheric deposition, inflows and outflows (surface and groundwater), adsorption/desorption, particulate settling, particle decomposition at the sediment/water interface and within sediments, re-suspension, burial, air/water gaseous exchange, industrial point sources, *in-situ* transformations (e.g. methylation, demethylation, MeHg photodegradation, Hg(II) reduction and oxidation), Hg uptake kinetics in plankton and partitioning in benthos, and MeHg bioaccumulation in fish. As much site-specific, empirical information is used as possible. Where site specific data do not exist or have small influences on model predictions, regional or literature information is used, such as regional atmospheric Hg concentrations.

MeHg concentrations in fish are predicted using a bioenergetics approach described by Harris and Bodaly (1998). Fish Hg concentrations tend to increase with age, and are therefore followed in each year class (up to 20 cohorts). Mercury fluxes are expanded from individual fish to entire fish populations by computing the fluxes for individual fish and then multiplying by the number of fish in each age class.

While many factors affect fish Hg concentrations in natural lakes, one process takes on special importance in reservoirs: decomposition. Inundation of soils stimulates decomposition and more activity by microbes that convert inorganic Hg into MeHg. Special attention is devoted to these processes in RESMERC. Sediments are divided into a maximum of 5 zones in the model, based on terrain type and elevations set by the user. These zones can include littoral and profundal zones in the original waterbody, flooded uplands and flooded wetlands. Each sediment zone has two vertical sediments layers with thicknesses defined by the user (e.g. a surface layer of 1 cm and an underlying layer of 3 cm). Sediments below the 2$^{nd}$ layer are treated as a boundary condition. Each sediment layer has its own initial conditions, characteristics and inputs. Additional information on the RESMERC model is available in the model user guide (Harris and Hutchinson 2009) and a report describing the model development (Harris et al. 2009).

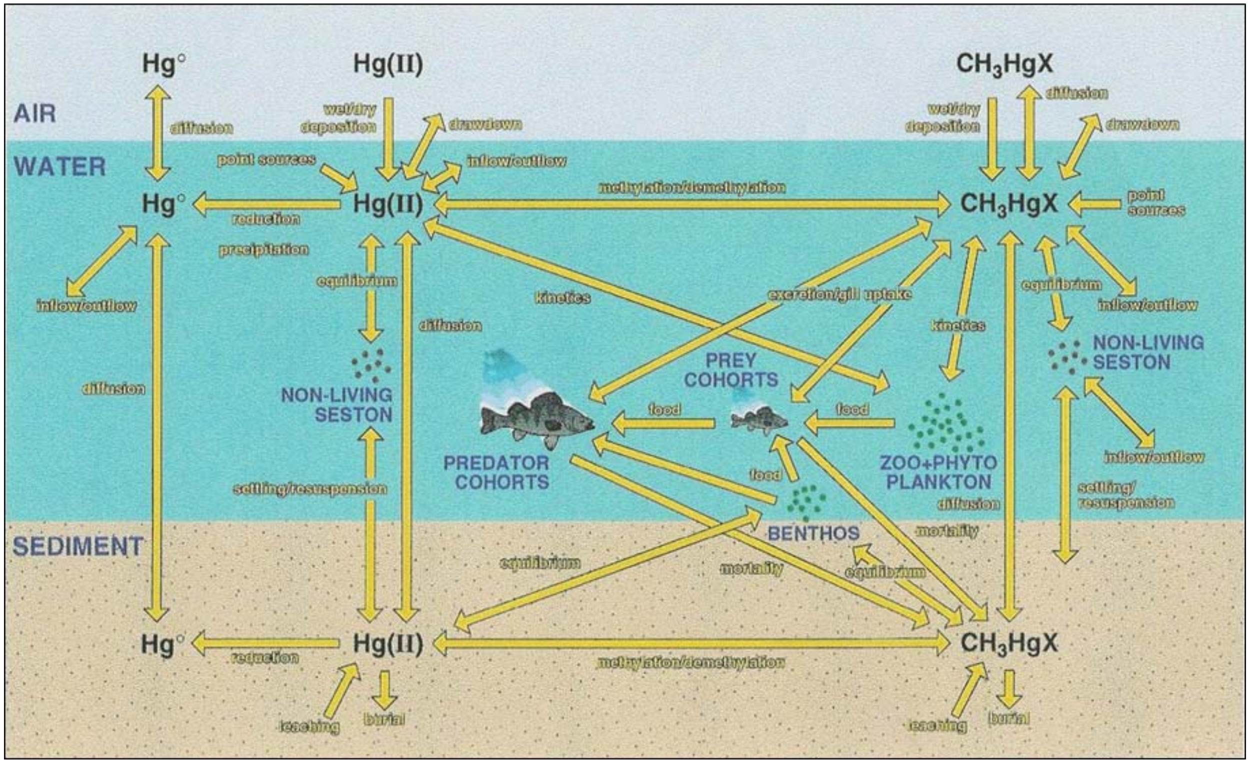



**Figure 3-1. Conceptual diagram of Hg cycling and bioaccumulation in RESMERC**

# 4 OVERALL MODELING APPROACH

The approach used to apply RESMERC to the proposed Site C reservoir was as follows:

1. Update the model calibration by applying it to two full scale reservoirs created in the 1970s with long-term fish Hg datasets: Robert Bourassa Reservoir, Quebec, and Notigi Reservoir in Manitoba.
2. Calibrate the model to pre-inundation conditions in the Peace River using data from baseline studies for the proposed Site C Project.
3. Apply the model to the Site C reservoir to predict the magnitude and duration of changes to fish mercury concentrations in key species in the reservoir.

RESMERC was originally developed and calibrated as part of two experimental reservoir projects carried out in the Experimental Lakes Area, Ontario (ELA). Three small upland reservoirs (FLUDEX project) and one wetland reservoir (ELARP project) were created and studied for Hg cycling and greenhouse gas emissions (Bodaly et al. 2004; St. Louis et al. 2004). A single model calibration was found for RESMERC that agreed well with observations for the upland and wetland (FLUDEX and ELARP) (Harris et al. 2009). These sites were relatively shallow and water levels were drawn down substantially each year, and then refilled in the spring. Prior to applying RESMERC to Site C, it was necessary to test and apply the model on full scale reservoirs with long term monitoring of fish Hg levels, to allow predictions from RESMERC to be calibrated to observations. Long-term datasets of fish Hg concentrations were available only for a limited number of full scale reservoirs, as many reservoirs were built prior to the identification of the Hg issue in reservoirs in the 1970s. There were no candidate reservoirs available in British Columbia for this task as all were constructed before there was an understanding of the relationship between new reservoirs and increases in fish Hg. Long-term fish Hg datasets were however available for Notigi Reservoir in Manitoba and some of the reservoirs associated with the La Grande River projects developed in Quebec. RESMERC simulations were carried specifically for Notigi Reservoir and Robert Bourassa Reservoir, to test the ability of the model calibration developed at ELA.

Once the RESMERC calibration was updated for the two full scale reservoirs (see Section 4-1), it was applied to existing conditions in the reach of the Peace River where the Site C reservoir is proposed (Section 4-2). Baseline data for site conditions and Hg concentrations in water, sediments, the lower food web and fish were used in the application of the model for existing conditions. These data were derived from field investigations in the Peace River and Dinosaur Reservoir specifically to address site-specific data requirements of the model. A full documentation of data is available in Azimuth (2011; and Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1). Simulations of existing pre-inundation fish Hg concentrations in the Peace River were carried out assuming that current conditions in the river have existed for sufficient time for Hg concentrations in water, sediments and fish to reach stable levels.

Finally, simulations were carried out for the proposed Site C reservoir. Concentrations estimated with the pre-flood simulation were used as the starting values for post-flood scenarios. The reservoir water column is predicted to stratify vertically in the summer, but only in the downstream end (Figure 2-2). Because stratification can affect Hg cycling, the reservoir was divided into two reaches. The upper reach included the first 25 km of the reservoir (in the direction of the flow), while the downstream reach included the remaining 58 km. It was assumed that the effects of the filling period for the Site C reservoir were negligible in terms of affecting peak fish mercury concentrations (expected years later), and the reservoir was assumed to be at full capacity when inundation occurred. Simulations were carried out for a post-inundation period of 50 years, long enough for predicted fish Hg concentrations to reach peak values and then decline to background levels. Predicted Site C reservoir simulations did not consider the potential effects of reservoir clearing or other construction phase activities, and only represent the operating phase of the Project.

# 5 MODELING RESULTS

## 5.1 Model Calibration to Large-scale Reservoirs

RESMERC was applied to two full-scale reservoirs (Notigi MB; Robert Bourassa PQ), to test the calibration and make modifications if necessary, prior to applying it to the Site C reservoir.

Hydro Quebec constructed a series of large reservoirs in the 1970s and 1980s in the La Grande River system. Robert Bourassa Reservoir (LG2) was filled in 1978-79 and has a maximum area of 2,835 km$^2$, of which 2,630 km$^2$ is flooded (92%). The mean annual discharge is 3,374 m$^3$/s, and the mean hydraulic residence time is 6.9 months (Schetagne et al. 2003). Mean annual drawdown of water levels is 3.3 m. Monitoring of fish mercury levels in Robert Bourassa Reservoir began in 1978 at multiple stations. Sampling at the LG2 station for example occurred at least every 2 years from 1982 through 2000

(Schetagne et al. 2003). Length-standardized Hg concentrations rose to a peak of 3.28 mg/kg wet muscle in 700 mm northern pike in 1990, 11 years after inundation. Lake whitefish (400 mm) peaked at 0.52 mg/kg five years post flood, and longnose sucker (400 mm) peaked seven years post-flood at 0.63 mg/kg. Site specific pre-flood baseline concentrations were not available, but regional mean concentrations from natural lakes sampled in the project area were 0.59, 0.11 and 0.12 mg/kg respectively for northern pike, lake whitefish, and longnose sucker for the lengths mentioned above (Schetagne et al. 2003).

Notigi Reservoir, Manitoba was created when water was diverted south from the Churchill River through the Burntwood/Nelson River system to increase the water supply to several generating stations on the Nelson River. Notigi Reservoir began filling in April 1974 and was completed in December 1976. Annual water level variations occur on the order of 2 to 4 metres per year. For the purposes of the model application, it was assumed that the entire stretch of water from the South Bay Diversion Channel to Notigi Dam was a single waterbody. Flooding increased the water surface area from 198 km$^2$ to 785 km$^2$ (Manitoba Hydro 2006a). The flood zone represented 75% of the total reservoir and was approximately 20% wetland. The estimated mean hydraulic residence time was 110 days (Harris et al. 2009). Water quality data 15-25 years post-inundation suggest that Notigi Reservoir is naturally productive (e.g. total phosphorus levels 15-33 µg/L (Morrow Environmental Consultants 2001)). For the period 5 - 25 years after inundation, dissolved organic carbon (DOC) levels were ~5-10 mg/L and pH ranged from 6.9 – 8.3. Mercury levels were monitored in several fish species in Notigi Reservoir since the mid 1970s. Typical of reservoirs on the Canadian Shield, Hg concentrations in sport fish (walleye and northern pike) rose significantly after inundation, reaching approximately 2 mg/kg wet muscle within 5-7 years after inundation, before declining towards background levels. By 2002, 26 years after inundation, MeHg concentrations adult walleye and northern pike remained elevated despite low MeHg concentrations in surface waters (<0.06 ng/L), zooplankton (15-20 ng/g dw), and sediments (0.41 ng/g dw).

RESMERC simulations were carried out for Robert Bourassa and Notigi Reservoirs to predict MeHg concentrations in northern pike and lake whitefish. These two species were chosen because data were available from both reservoir, and these species represented two trophic levels, including a top level predator. Adjustments were made to the model calibration until satisfactory results were obtained for both sites with one model calibration. This calibration could then be applied to the conditions for the Site C reservoir. Adjustments to the model calibration primarily involved the following:

- The depth of active sediments/flooded soils was increased to 7 cm, slowing the rate constant for carbon decomposition
- Adjustments to Hg partitioning for abiotic solids,
- Adjustments to MeHg partitioning in the lower food web.

To achieve agreement with observed fish Hg trends in the two full scale reservoirs, it was necessary to increase the depth of the active surface sediment layer to 7 cm, with an areal carbon pool of 1.9 $kg/m^2$. The deeper, larger soil pool produced higher peak MeHg concentrations in simulations and lengthened the recovery period for fish Hg to better match observations.

Predicted and observed Hg concentrations in 700 mm northern pike and 400 mm lake whitefish in Robert Bourassa reservoir are shown in Figure 4-1 a and b. Predicted and observed Hg concentrations in Notigi Reservoir for 550 mm northern pike and 350 mm lake whitefish are shown in Figure 4-1 c and d. The simulations reasonably represented the magnitude and timing of the response of Hg concentrations in northern pike and lake whitefish in both reservoirs.

Overall, the updated model calibration resulted in good agreement with observed increases and subsequent declines of fish Hg concentrations in northern pike and lake whitefish in the two reservoirs simulated. No observations were available for total Hg or MeHg in the water column or lower food web during the early years after inundation when maximum changes would be expected for these environmental media. One caveat is that the revised calibration did not produce adequate results when applied to the FLUDEX reservoirs in Ontario. It is hypothesized that the FLUDEX experiment represented the response of the littoral zone of reservoirs but additional processes may occur in large, deeper reservoirs that influence fish Hg concentrations. Stratification and oxygen depletion may have occurred in deeper waters of Notigi and Robert Bourassa reservoirs, leading to increased rates of MeHg production. The FLUDEX reservoirs were also drawn down heavily, which could accelerate carbon decomposition rates relative to permanently inundated zones in deeper reservoirs. Nevertheless, the successful use of one model calibration for two fish species in two full scale reservoirs provided a sufficient basis to proceed with Site C simulations.

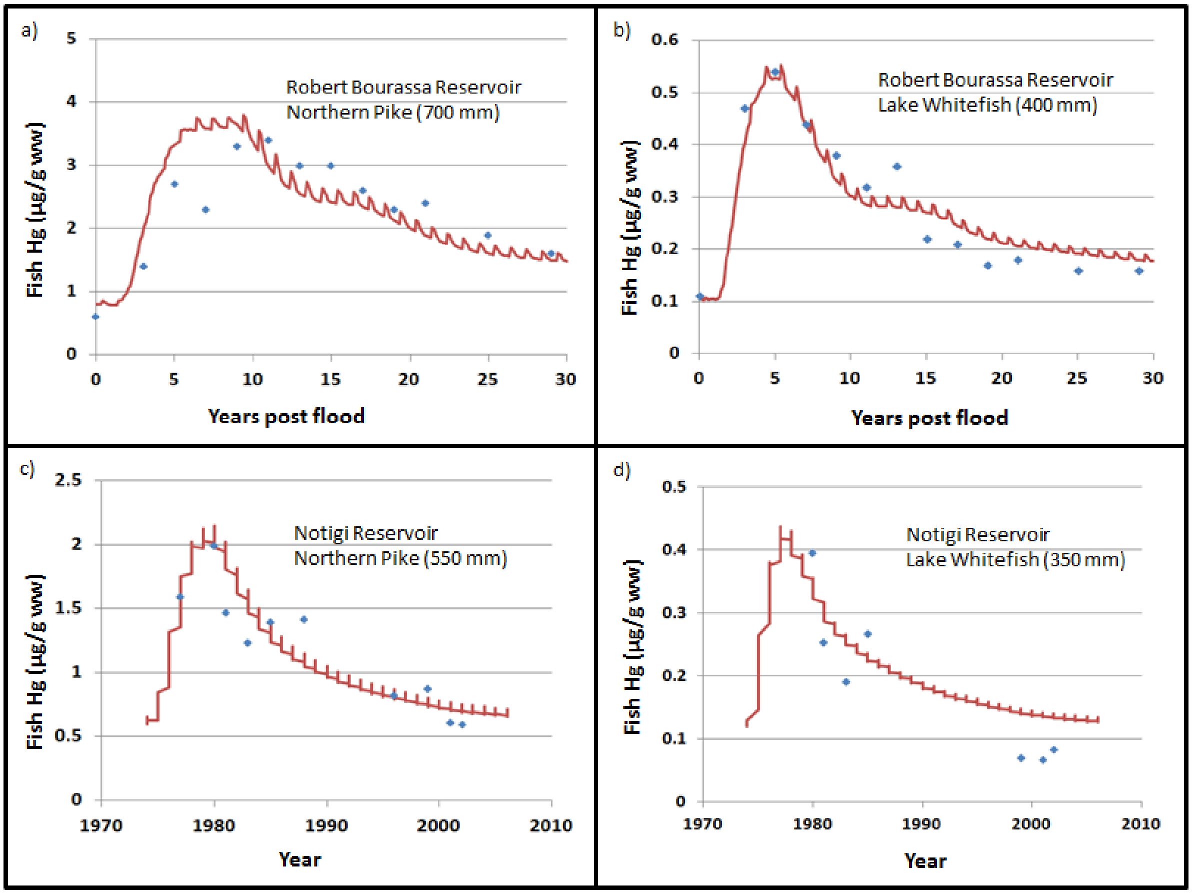



**Figure 5-1. Predicted and observed Hg concentrations in northern pike and lake whitefish in Robert Bourassa and Notigi Reservoirs, using the final model calibration.**

Predictions shown with line. Points are observations. Robert Bourassa data from Therien and Schetagne (2009). Notigi Reservoir data are from Bodaly et al. (2007).

## 5.2 Model Application to Site C

RESMERC was applied to the Site C Project in two stages. First, simulations were carried out for existing baseline conditions in the Peace River where the reservoir portion of the technical study area. Then the model simulated the operating phase of the Site C reservoir, using the Hg concentrations estimated during the pre-inundation simulation as the starting point for post-inundation simulations. This ensured that predicted changes in fish Hg concentrations post-inundation were due solely to reservoir creation. The existing configuration of the Peace River and the proposed reservoir areas are shown in Figure 2-1.

As stated above, simulations applied only to the operating phase of the Project, and did not consider the effects of construction phase activities or reservoir clearing on Hg cycling and bioaccumulation during the operating phase.

### 5.2.1 RESMERC Application to Existing Peace River Conditions

RESMERC pre-inundation simulations were carried out for the existing 83 km reach of the Peace River that would become part of the Site C reservoir. This river was assumed to be well mixed, with the same site conditions throughout. Pre-inundation site conditions in the river relevant to Hg cycling are summarized in Table 2-1.

<u>External Hg Loads</u>

Inflowing Hg loads were based on 2000-2010 flows through this river reach, including outflows from the Peace Canyon Dam and key tributary streams Lynx, Farrell, Cache Creeks and Halfway and Moberly Rivers (Volume 2, Appendix P, Part 3; Future Conditions Report). At all times of the year except during freshet, at least 95% of Peace River flow between Peace Canyon Dam and the Site C dam is received from Dinosaur Reservoir, after flowing through Williston Reservoir. Baseline sampling in 2010 and 2011 in the Peace River and its tributaries, exclusive of high TSS events during freshet, indicated total Hg concentration of approximately 1 ng/L (Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1). Similarly low concentrations were reported in Williston Reservoir in the early 2000s (Baker et al. 2002). Methylmercury concentrations in Peace River and tributary stream water was below the laboratory detection limits of 0.02 to 0.05 ng/L in nearly all samples except during 2011 at Peace River near Moberly River (0.08 ng/L) during a high flow TSS event (128 mg/L). The inflow MeHg concentration was assumed to be 0.02 ng/L for pre-inundation and post-inundation simulations.

<u>Fish Species, Growth and Diets Pre-inundation</u>

Pre-inundation simulations were carried out for bull trout, mountain whitefish, longnose sucker, rainbow trout and redside shiner. These species represent a top-level predator, a fish with an omnivorous diet, a benthivore and a forage species, respectively. Dietary preferences for each species were derived from Ecopath model predictions of post-inundation diets and empirical data from dietary studies of fish conducted in the Peace River and Dinosaur Reservoir (Volume 2, Appendix P Part 3, Future Conditions Report; Pattenden et al. 1990 and Section 12 of the EIS Fish and Fish Habitat). The dominant lower food web organisms in fish diets in the pre-flood reach of the Peach River are epibenthic invertebrates, dominated by caddisflies and mayflies, with some stoneflies, water boatmen, snails, mites, clams and chironomid fly larvae (Volume 2, Appendix O; Fish and Fish Habitat Technical Data Report).

There is uncertainty in the scientific community whether MeHg in epibenthos is derived primarily from exchange with the water column, sediments or a mix of both. This is relevant to post-inundation simulations because MeHg concentrations in sediments were predicted to increase significantly more in relative terms than MeHg in the water column (see later discussions). If epibenthos were linked exclusively to sediment MeHg concentrations, and MeHg concentrations in newly flooded sediments increase as predicted by RESMERC, fish that eat these organisms would be expected to experience increases beyond what is typically observed in the literature (e.g. 2-5X) for whitefish in several Quebec reservoirs (Schetagne et al. 2003). Therefore, a mix of water column and sediment exposure was used for epibenthos in simulations. The pre-inundation calibration was adjusted so that the lower food web compartment in RESMERC that is typically used to simulate zooplankton was instead calibrated to represent epibenthos linked to MeHg in the water column, with the same concentration observed for epibenthos. To the extent that fish diets included epibenthos, equal portions were assigned to model compartments linked to MeHg in the water column and sediments. This approach was considered reasonable in the absence of further information.

Approximately 25% of the rainbow trout diet in simulations consisted of terrestrial insects, whose MeHg concentrations will not change following creation of the Site C reservoir. RESMERC does not have provisions for dietary items with MeHg concentrations that are independent of conditions in the reservoir. Simulations therefore added the terrestrial component of the diet to the component represented by epibenthos linked to the water column. This approach conservatively tends to overestimate the effects of inundation on MeHg exposure for rainbow trout.

Fish growth rates were calibrated for each species using site-specific data collected in 2010 and 2011 (Azimuth 2011 and Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1).

Pre-inundation RESMERC Calibration Results

Modeled and observed total Hg and MeHg concentrations in water, sediments and lower food web compartments are shown in Table 4-1. Due to the short hydraulic retention time of water in the pre-inundation simulation (< 1 day), predicted concentrations of total Hg and MeHg in the water column were similar to inflowing Hg concentrations. Upstream Hg loads to the pre-flood river reach (~ 740 $\mu g/m^2/yr$) far exceeded direct atmospheric Hg deposition to the river surface (assumed to be on the order of 5 $\mu g/m^2/yr$). Direct atmospheric Hg deposition therefore had a negligible effect on fish Hg levels during pre-inundation simulations.

Modeled MeHg concentration in pre-inundation epibenthos was approximately 0.005 mg/kg ww. Observed MeHg concentrations in benthic invertebrates collected from the Peace River (locations shown in Figure 2-1) ranged from 0.003 to 0.030 mg/kg ww.  Zooplankton MeHg concentrations observed in the Peace River were very low: 0.0001 – 0.0007 mg/kg ww (Azimuth 2011) but similar to what were observed in Williston Reservoir (Baker et al. 2002).The Hg Technical Synthesis Report (Volume 2, Appendix J, Part 1) provides an extensive review of total Hg and MeHg concentrations observed in environmental media from the Peace River as well as upstream (Dinosaur, Williston) and several other Canadian reservoirs for comparison. Overall the allocation of dietary preferences used for fish species resulted in good estimates of observed fish MeHg concentrations. Simulated fish Hg concentrations were with the range of observations for all species (Figure 4-2).

**Table 5-1. Observed and model-calibrated Hg concentrations in water, sediment and lower food web for pre-inundation conditions.**

*Zooplankton calibrated to observed epibenthos MeHg concentrations. See discussion in text. All data from Azimuth (2011) and Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1

| Hg form | Compartment | Value | | Units |
|---|---|---|---|---|
| | | Model | Observed | |
| Total Hg | Water column | 0.75 | 0.6 – 0.8 | ng/L unfiltered |
| MeHg | Water column | 0.02 | <0.05 | ng/L unfiltered |
| Total Hg | Sediments | 0.04 – 0.05 | 0.03 – 0.06 | mg/kg dw |
| MeHg | Sediments | 0.001 | 0.0005 – 0.0018 | mg/kg dw |
| MeHg | Benthos | 0.005 | 0.003 – 0.03 | mg/kg ww |
| MeHg | Zooplankton | 0.005* | 0.0001 – 0.0007 | mg/kg ww |

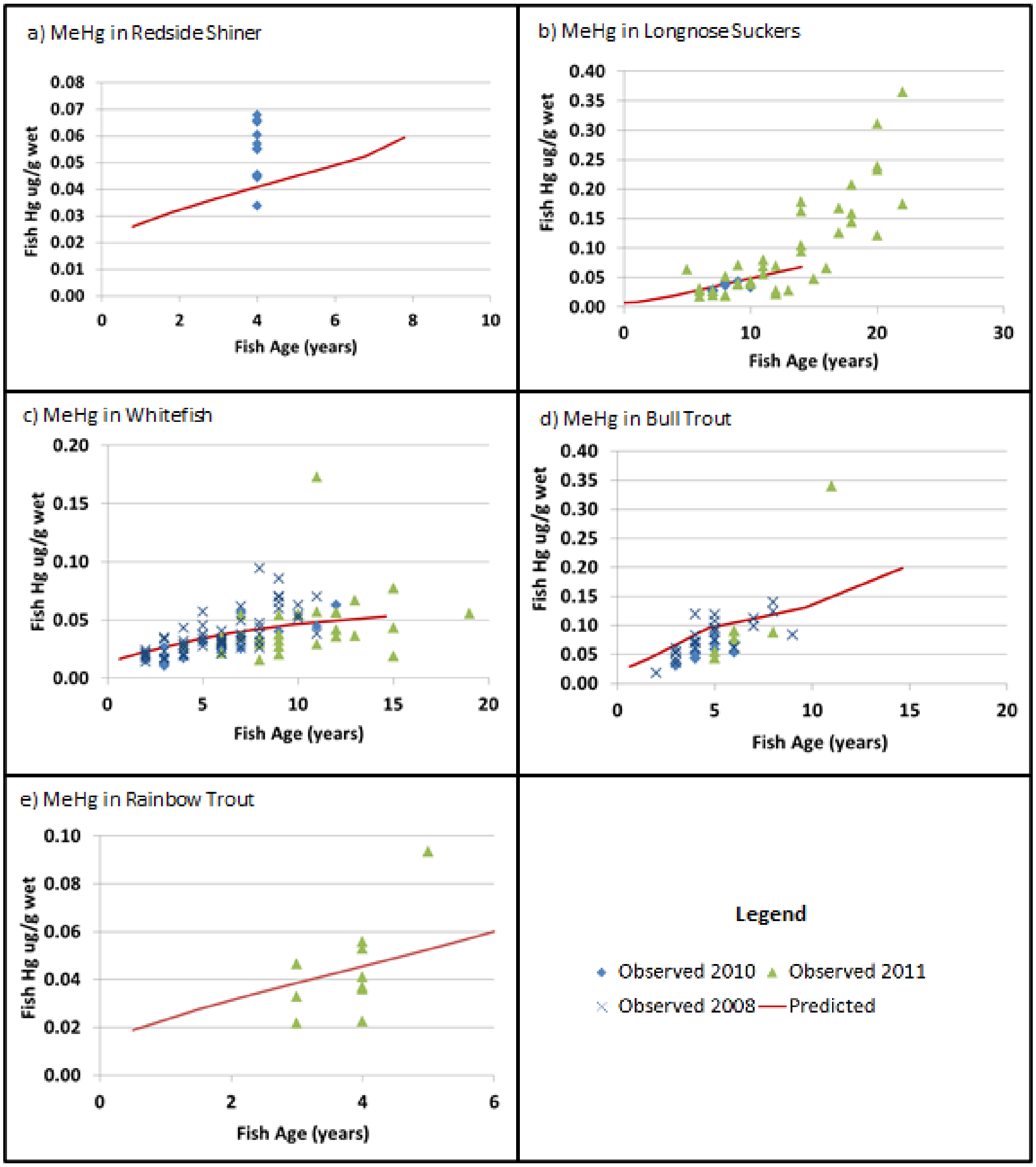



**Figure 5-2. Simulated and observed fish MeHg concentrations for pre-inundation conditions**

### 5.2.2 RESMERC Application to the Site C Reservoir

RESMERC was applied to the Site C reservoir using the calibration updated for the pre-inundation simulation. Features of the post-inundation simulations included:

- The reservoir was assumed to be fully inundated at the beginning of the simulation. Filling would actually occur over a period of several weeks.
- A 50-year duration to allow sufficient time for fish Hg concentrations to rise and decline to long-term stable levels.
- Construction phase and reservoir clearing effects were not included in simulations.
- Simulations assumed small increases in suspended solids concentrations in the water column, while the Fluvial Geomorphology and Sediment Transport Report (Volume 2, Appendix I) estimated high rates of sedimentation in the reservoir during the early years after inundation, ranging from 0 – 25 cm (or more) during the first decade post-flood. Assuming that eroded shoreline material would be primarily inorganic, the potential exists for high sedimentation rates to reduce the production of MeHg in surface sediments and reduce peak fish Hg levels.
- Baseline Hg concentrations simulated in the pre-inundation simulation were used as initial concentrations post-inundation.
- 2000-2010 flows from Williston Reservoir were assumed to continue in the post-inundation scenario.
- External Hg loads were assumed the same as during pre-inundation conditions.
- The reservoir was segmented into two reaches: the upper reach representing the upstream 25 km, and a lower reach representing the remaining 58 km to the Site C Dam. The distinction was made because temperature modeled predicts that the upper 20-25 km would remain well mixed year-round, while the lower reach would stratify during summer.

Selected physical and chemical characteristics of the proposed reservoir are given in Table 2-1. Additional information on trophic conditions and inundation zone characteristics in the proposed Site C reservoir is given below.

Fish Species, Growth and Diets Post-inundation

Post-inundation simulations were carried out for bull trout, mountain whitefish, longnose suckers, redside shiner and rainbow trout. Dietary preferences for each species were derived from Ecopath model predictions of post-flood diets and empirical data from dietary studies of fish conducted in the Peace River and Dinosaur Reservoir (Volume 2, Appendix P, Future Conditions Report, Part 3; Mainstream Aquatics 2010, 2011). Bull trout were more piscivorous (65%) in post-flood simulations after the age of 3 years pre-flood (45%). The Future Conditions Report (Volume 2, Appendix P, Part 3) estimated that 50+ years after inundation, bull trout would be highly piscivorous (85%), mostly targeting kokanee. The time course of this transition to greater piscivory is uncertain, depending partly on the time required for a pelagic food web to become well established. RESMERC does not simulate dietary shifts for adult fish with time. A value of 65% piscivory was therefore used for adult bull trout, the average of the pre-flood long term, post-flood estimates of piscivory. Simulations carried out for the operating phase in this study included redside shiner as an important dietary item for bull trout, based on Pattenden (2012). Similar to pre-inundation simulations, epibenthos consumption by fish was allocated equally to epibenthos whose MeHg was linked to water column and sediments.

Fish growth rates have been documented to accelerate in new reservoirs in connection with trophic surges. Based on the literature, more rapid fish growth has been documented, when using a size-based standard for Hg levels (e.g. Simoneau et al. 2005; Harris and Bodaly 1998), a phenomenon known as growth dilution. Quantitative estimates of changes to fish growth rates were not available for the Site C reservoir, and fish growth rates were assumed to remain the same as used in the pre-inundation simulation.

Inundation Zone Characterization

Inundation zone characteristics are important to consider when estimating the effects of inundation on fish Hg concentrations. Terrestrial studies carried out in the proposed inundation zone to characterize the quantity and general quality of soils and vegetation measured soil depth, organic carbon content, pH, and total Hg and MeHg concentrations (Azimuth 2011, Volume 2, Appendix J, Part 1). Wetland habitat comprised less than 2% of total reservoir area, dominated by Watson Slough (Azimuth 2011). The overall average organic carbon content of all organic horizons sampled was 27%. A limited number of samples from inorganic horizons (n=4) indicated low organic carbon content (<5.0%) as expected, given the sandy nature of the underlying soil. The carbon pool in the upper 5 cm was estimated to range from 0.5 to 1.2 kg $C/m^2$. By comparison, RESMERC simulations for

Notigi Reservoir and Robert Bourassa Reservoir used upland characteristics that would result in 1.4 kg C/m$^2$ over a 5 cm depth. Given the heterogeneity of inundation zones and associated uncertainties associated with estimating carbon pools in these zones, the upland inundation zone properties used in simulations of Notigi Reservoir and Robert Bourassa Reservoir were also used for the Site C simulations. It was also assumed that the active surface sediment layer was 7 cm in Site C reservoir simulations. This approach is slightly conservative, as larger carbon pools in inundation zones lead to greater MeHg production in simulations.

Calibration Results for the Site C Reservoir during Operating Phase

Results for RESMERC simulations of MeHg concentrations in water, sediments and selected biota in Site C reservoir are presented in Figure 4-3 and Figure 4-4. Water column and sediment predictions are presented for both reservoir reaches modeled, and are higher in the lower reach. Total Hg concentrations are predicted to increase only marginally in the water column (not included in Figure 4-3), while MeHg concentrations in surface waters are predicted to roughly double at peak levels compared to existing conditions, using annual averages. Short-term MeHg concentrations in the water column are predicted to reach ~0.06 ng/L. While inundation will cause increases in MeHg levels in the water column, these predicted concentrations are within the range observed for North American boreal waters without point sources of Hg (e.g. Hurley et al. 1995; Scudder et al. 2009). Flow dilution is a contributing factor to the predicted concentrations being lower than would otherwise occur. Sediment MeHg concentrations are predicted to increase more in relative terms than the water column, reaching peak levels of approximately 0.020 – 0.025 mg/kg dw. Because fish movement between the reaches is assumed to occur for the new reservoir, model predictions for fish Hg concentrations were averaged on the basis of the relative areas of the two reaches (roughly 20% in the upper reach and 80% in the lower reach). The predicted increase in fish Hg concentration depended partly on the extent to which its MeHg exposure could be traced back through the food web to MeHg in the water column or sediments. Fish Hg concentrations tended to increase more in relative terms than surface waters but less than sediments (Figure 4-4). The results presented represent mean values for a particular size class at a given time. There would be individuals with higher and lower Hg concentrations. Overall, the predicted peak fish Hg concentrations would be relatively low in the context of peak fish Hg levels observed for top-level predators in eastern/central reservoirs, which commonly exceeded 1 – 2 mg/kg (Schetagne et al. 2003; Bodaly et al. 2007). Bull trout peak Hg is predicted at near 0.45 mg/kg for a 600 mm fish and 0.43 mg/kg for a 500 mm fish. Lower trophic level fish species (suckers, whitefish, shiners) were predicted to have peak concentrations less than 0.2 mg/kg wet muscle for the size classes examined. Additional discussion of predicted fish Hg concentrations is provided below:

**Mountain whitefish** MeHg concentrations are predicted to peak between 0.08 to 0.13 mg/kg for fish ranging from 100 – 300 mm. The baseline Hg concentration from the Peace River (2008 – 2011) was ~0.04 mg/kg (320 – 340 mm), thus the modeled peak concentration represents a ~ 2-4X increase above the baseline, depending on fish size.

**Redside shiner** are predicted to colonize the reservoir in the short-term (<10 y post flood) and are expected to be an important dietary source for bull trout (EIS Section 12). MeHg concentrations in redside shiner are predicted to peak at approximately 0.12 mg/kg for a 100 mm fish. This is the most common size of fish captured during baseline investigations, although younger, smaller fish are likely more abundant they are not captured as easily in conventional fishing programs (Mainstream Aquatics 2010, 2011). The baseline mean Hg concentration of redside shiner from the Peace River in 2011 was 0.05 mg/kg, and the modeled peak concentration represents an increase of ~2.5X above the observed baseline.

**Longnose sucker** have an omnivorous, opportunistic dietary strategy (Scott and Crossman 1979) and are an important forage species for predatory fish. Young and juvenile fish are expected to be successful colonizers of the proposed new reservoir because habitat and dietary preferences are amenable to conditions here (EIS Section 12, Fish and Fish Habitat). Mercury concentrations in 100, 200, 300 and 400 mm longnose sucker are predicted to peak at 0.07, 0.10, 0.15, and 0.23 mg/kg respectively. The baseline mean Hg concentration of longnose sucker from the Peace River (Azimuth 2011, Volume 2, Appendix J, Part 1, Mercury Technical Synthesis Document) is 0.05 mg/kg (380 mm). This peak concentration represents an increase of ~ 4-5X above baseline.

**Bull trout** are expected to persist within the Site C reservoir at least during the period when Hg levels are expected to rise. Predicted Hg concentrations for 400 mm, 500 mm and 600 mm bull trout are predicted to peak at 0.38 mg/kg, 0.44 mg/kg and 0.46 mg/kg respectively. Concentrations of individual fish may be higher or lower than these mean levels for a given length. Although the baseline Hg concentration for bull trout (2008 – 2011) ranged from 0.07 – 0.08 mg/kg for a mean size of 500 mm fish (Azimuth 2011), the maximum observed fish size was >800 mm with an Hg concentration of 0.34 mg/kg. The predicted maximum increase for discrete size fish ranges from approximately 5-7X above the baseline and are predicted to occur approximately eight years after reservoir impoundment. Bull trout sizes between 500 mm and 600 mm are emphasized because this is the size that is most common in fisheries surveys and therefore, is likely the size most targeted for consumption by domestic or sport anglers (Human Health Risk Assessment, Volume 2, Appendix J, Part 2).

**Rainbow trout** are expected to continue to be the dominant insectivorous fish species in the Site C reservoir over the short term (< 10 years). MeHg concentrations in rainbow trout in the Peace River are currently low, with most fish having concentrations less than 0.1 mg/kg with the majority of observed concentrations being less than 0.06 mg/kg. The peak MeHg concentrations for 300 mm and 400 mm rainbow trout are predicted to be 0.15 mg/kg and 0.17 mg/kg, respectively. This represents a 4 – 4.5X increase above baseline. According to the ECOPATH model, a significant fraction of the rainbow trout diet (26%) consists of terrestrial insects whose mercury levels most likely will not be impacted by reservoir creation. The RESMERC model does not have the capability of assigning fish diet items external to the reservoir system. Because mercury levels increase in all rainbow trout diet items in the RESMERC simulation, the modelled peak mercury levels for rainbow trout have been over-predicted.

The predicted time required for fish Hg concentrations to return to background concentrations is approximately 20 – 30 years, depending on individual reservoir characteristics and fish species and size. Lower trophic level species and younger fish for a given species are expected to recover more quickly than older adults for higher trophic level fish. This is because it takes time for MeHg increases to move through the reservoir ecosystem from water and sediments to the lower food web and different trophic levels of fish to top level predators. Bodaly et al. (2007) reported that Hg in several fish species in Manitoba reservoirs returned to background regional concentrations within 10-23 years. Monitoring in Quebec reservoirs showed that lower trophic level fish such as lake whitefish took 10-20 years to return to regional concentrations for lakes, while predatory fish species such as northern pike usually took 20-30 years (Schetagne et al. 2006). Predictions for the Site C reservoir emerge from the calibration of RESMERC to Notigi and Robert Bourassa reservoirs where up to three decades were required for northern pike to return to regional background levels (Figure 4-1). Given that the Site C reservoir has a short predicted hydraulic residence time (mean = 23 days, much less than Notigi and Robert Bourassa reservoirs), this has the potential to shorten the recovery period for the Site C reservoir. This is consistent with predictions made in the Canadian reservoirs comparison matrix in Section 5 of the Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1.

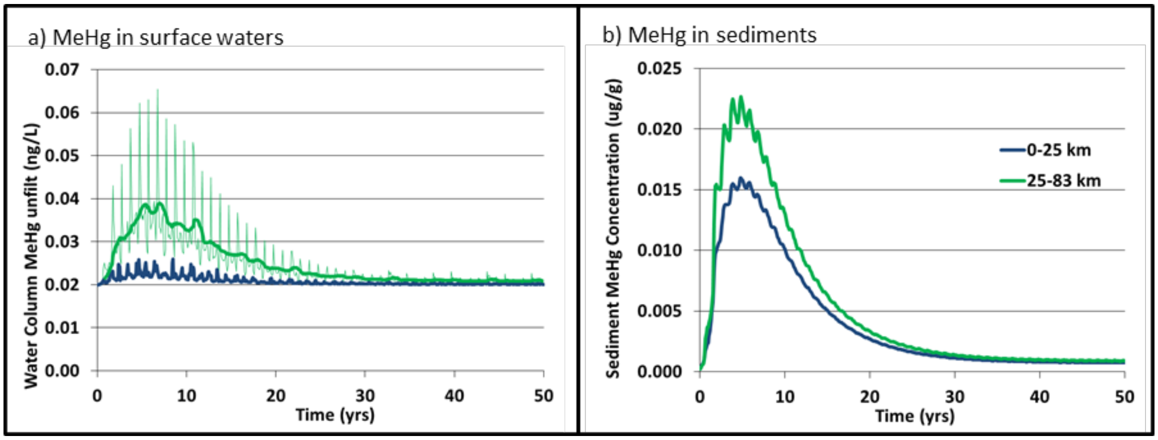



**Figure 5-3. Predicted MeHg concentrations versus time in surface waters and sediments in the Site C reservoir**

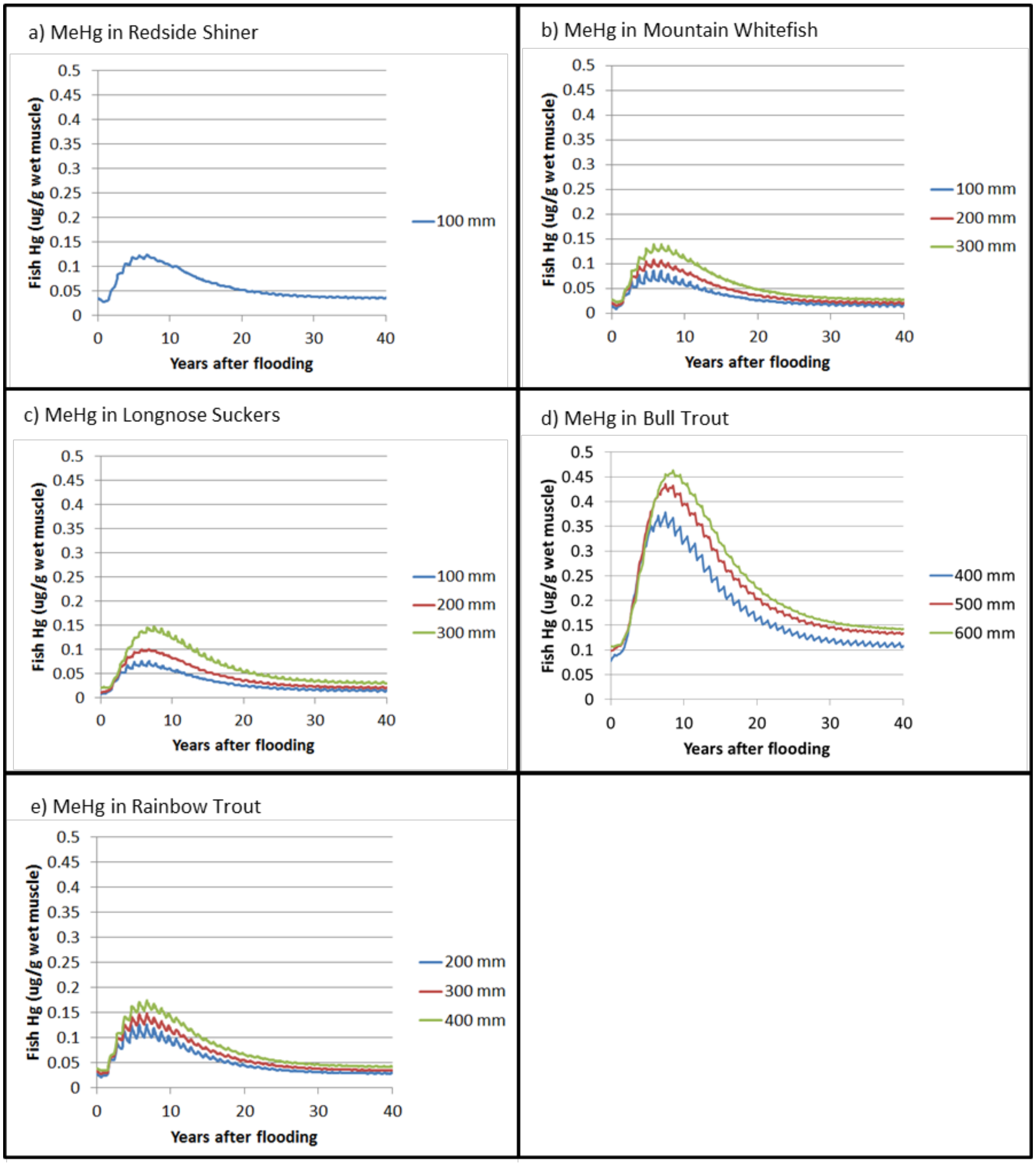



**Figure 5-4. Predicted MeHg concentrations versus time in fish in the Site C reservoir.**

Uncertainty Associated with RESMERC Predictions to Site C

Fish Hg concentrations predicted in this study are best characterized as an indication of whether increases are likely to be modest (e.g., up to 2x increase), moderate (e.g. 3 – 4 x increase), or high (e.g. >4x increase), rather than placing a high degree of numerical accuracy to the numbers. This characterization is based on the range of increases in fish Hg concentrations observed from hydroelectric developments elsewhere in Canada including in Quebec (Schetagne et al. 2003) and Manitoba (Bodaly et al. 2007) and summarized in the Mercury Technical Synthesis Report, Volume 2, Appendix J, Part 1. Factors that preclude more accurate numerical predictions include:

- Gaps in the scientific understanding of mercury cycling in aquatic systems, including the effects of inundation on MeHg production, and MeHg partitioning at the base of the food web in terms of links to sediments and the water column.
- Limited data from full scale reservoirs for environmental media other than fish, including water, sediments, and the lower food web. Furthermore, such long-term datasets to calibrate models against are not available for western Canada.
- Natural spatial variability. The model segmented the reservoir into two reaches spatially. Actual conditions may differ from scenarios tested and local variability is always expected.
- Model predictions were sensitive to rates of sedimentation and food web structure, which are also predicted features of the ecosystem, and have uncertainty.
- Construction phase activities may affect Hg cycling and bioaccumulation during the operating phase. These effects have not been simulated.

Given these uncertainties, some conservative approaches were used when predicting fish Hg concentrations in the Site C reservoir:

- Effects of high rates of mass sedimentation predicted by the Fluvial Geomorphology and Sediment Transport report (Volume 2, Appendix I) were tested with the model and found to have the potential to considerably reduce MeHg production and diminish the predicted increase in fish Hg concentrations if the sedimenting material was highly inorganic. Due to a lack of information on this phenomenon in existing reservoirs, post-inundation sedimentation patterns for the Site C Project were not incorporated into the RESMERC model.
- Simulations did not consider potential effects of the construction phase. It is assumed that some inundation would occur during the construction phase, while RESMERC simulations assumed all inundation would occur during the operating phase. By distributing decomposition associated with inundation over a longer

period that includes the construction phase (~3y), lower peak fish Hg concentrations would be expected, although the duration of increased fish Hg concentrations could be longer. The potential also exists for erosion and downstream transport of organic matter during the construction phase, although this has not been quantified. These processes could also reduce the magnitude of decomposition during the operating phase as well as increases in MeHg production and fish Hg levels.

- Clearing was not included in RESMERC simulations. Clearing of standing trees and other vegetation has been hypothesized to result in lower peak fish Hg concentrations due to the removal of a portion of the carbon pool in the flood zone that would stimulate methylation. However, the vast majority of the carbon and mercury available for methylation are contained within the organic soil layer and not in the standing vegetation. Thus the benefits of clearing from a mercury methylation perspective are small and have been estimated to be on the order of producing a 10% lower peak fish Hg levels (Hydro Quebec 2007, NALCOR 2011).
- Carbon pools in the flood zone were assumed to be the same as used in the calibration of the model to eastern reservoirs. These pool sizes are slightly greater than site-specific estimates for the terrestrial lands proposed to be inundated via the creation of the Site C reservoir.

# 6 CONCLUSIONS

A mechanistic model of mercury cycling and bioaccumulation in reservoirs has been applied to the proposed Site C reservoir in British Columbia. The RESMERC model was first calibrated to fish mercury observations from two reservoirs constructed in the 1970s: Notigi Reservoir in Manitoba and Robert Bourassa Reservoir in Quebec. A single model calibration was developed that adequately simulated the observed rise and decline of Hg concentrations in northern pike and lake whitefish in both eastern reservoirs. This model calibration was then applied to the Site C Project. Pre-inundation simulations using baseline information collected in the Site C technical study area were carried out, followed by simulations of the Site C reservoir. The reservoir was segmented into two reaches for post-flood simulations. The upper reach represented the first 25 km of the reservoir as a well-mixed waterbody, while the lower 58 km stratified in simulations in the summer.

The pre-inundation calibration results agreed well with baseline observations, in terms of total Hg and MeHg concentrations in water and sediments, and MeHg concentrations in epibenthos and fish. Simulations run on the Site C reservoir predicted that water column concentrations of MeHg would peak at roughly double baseline concentrations to 0.04 ng/L, with short-term increases up to 0.06 ng/L. These concentrations are within the range observed for natural boreal lakes with no natural or anthropogenic point sources of mercury. Sediment MeHg concentrations are predicted to increase more than the water column in relative terms, into the range of 0.020 to 0.025 mg/kg dw at peak levels.

Simulations for fish tissue mercury concentrations predicted that adult bull trout Hg would peak at 0.45 mg/kg wet muscle (5 – 7x baseline), 8 to 10 years after inundation, before declining to regional background concentrations approximately 20 years later. Lower trophic level fish species (suckers, whitefish, shiners) were predicted to have peak concentrations in the range of 0.12 to 0.17 mg/kg wet muscle. Note that the short hydraulic residence time for the Site C reservoir and the influence of nutrient-poor, low mercury water from upstream in Williston Reservoir (Volume 2, Appendix J, Part 1, Mercury Technical Synthesis Report) has the potential to decrease the time required to recover from peak to background fish Hg concentrations than is presented here.

Peak fish mercury concentrations predicted by RESMERC are 2 – 6 times above current or baseline concentrations for the Peace River, depending on fish species and fish size. Predicted peak Hg concentrations for all fish species within the Site C reservoir are less than 0.5 mg/kg and are at the low end of the range observed from central and eastern Canadian reservoirs, where peak fish Hg concentrations often exceeded 1-2 mg/kg in sport fish (Bodaly et al 2007; Schetagne et al. 2003). These concentrations are also lower than for fish from all other BC lakes and reservoirs (Rieberger 1992; Baker 2002) and are some of the lowest observed anywhere in Canada (Depew et al. 2012). Possible reasons for predicted fish Hg concentrations being within the lower end of observations include low baseline fish Hg concentrations in the Peace River, a short hydraulic residence time (23 days) for the proposed reservoir, low inflowing Hg and MeHg concentrations from upstream and predictions of well-oxygenated conditions in the proposed reservoir (Volume 2, Appendix J, Part 1, Mercury Technical Synthesis Report).

RESMERC modeling was one of three approaches jointly used to determine the most likely magnitude of increase in fish Hg concentration with the proposed Site C reservoir and the duration that this phenomenon is expected to occur.

Authorship:

Senior Author

Reed Harris
Principal, Reed Harris Environmental Inc.

Contributors:

Simulations:
Don Beals

Simulations:
Dave Hutchinson

Review, Advisor:
Randy Baker
Principal, Azimuth Consulting Group

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# Appendix A

# Screening Level Predictions of Peak Mercury Concentrations in Bull Trout for the Proposed Site C Reservoir, BC.

# Screening Level Predictions of Peak Mercury Concentrations in Bull Trout for the Proposed Site C Reservoir, BC.

**Prepared for**

**Azimuth Consulting Group**
**Vancouver, BC**

Prepared by

Reed Harris and David Hutchinson
Reed Harris Environmental Ltd.
Oakville, ON

Updated December 2012

# 1 INTRODUCTION

Reservoir creation has been well documented to result in increased fish methylmercury (MeHg) concentrations (Bodaly et al. 2007; Jacques Whitford 2006; Schetagne et al. 2003: Bodaly et al. 1997; Canada/Manitoba Governments 1987). Fish MeHg concentrations rise after flooding in new reservoirs, and the duration of elevated fish Hg concentrations in boreal reservoirs lasts up to three decades. Peak concentrations, especially in top predatory fish, can be 2 to 7 times greater than background levels in boreal reservoirs (Bodaly et al. 2007; Schetagne et al. 2003).

While existing bull trout Hg concentrations in the proposed Site C section of the Peace River are low (0.07 – 0.08 mg/kg for a 500 mm standard length; Azimuth 2011) there are examples where bull trout (*Salvelinus confluentus*) have exceeded these concentrations in other reservoirs. Mercury (Hg) concentrations in standardized 550 mm bull trout in Finlay Reach, Williston Reservoir were 0.87 in 1988 (20 years post-flood) and 0.56 mg/kg in 2000 (EVS 2002). The relative increase in fish Hg levels in Williston and other BC reservoirs is not known, because no data are available however for both pre- and post-inundation fish Hg concentrations in western Canadian mountainous regions.

The response of fish Hg concentrations following inundation is affected by reservoir-specific features including the type and amount of terrain flooded, hydraulic residence time, water level fluctuations, erosion and water chemistry. Reservoirs with higher fractions of their total area represented by flooded terrain tend towards higher fish Hg levels. In contrast, short hydraulic residence times should result in lower MeHg concentrations in surface waters of reservoirs via dilution, but create the potential for greater downstream transport.

This technical memorandum describes the application of a regression model (Harris et al. 2008) to carry out a screening level assessment to forecast the magnitude of increase in fish Hg levels in connection with the proposed Site C Dam, a potential third dam and generating station on the Peace River in northeastern B.C. (Kirk and Consulting and Synovate 2009). The Site C reservoir would be 83 km long with a total area of 93.3 $km^2$. The flooded area could include up to 53.4 $km^2$ of floodplain soils, gravel bars, riparian and upland soils, and vegetation (Azimuth 2010, 2011; Volume 1 Section 2 Project Description).

# 2 MODEL DESCRIPTION

The model approach used in this study was originally developed to predict peak mercury concentrations for northern pike (*Esox lucius*) in connection with the proposed Lower Churchill Hydroelectric Development in Labrador (Harris et al. 2008). The model was subsequently applied to proposed reservoirs in Ontario on the White River (Harris and Beals 2009), Kabinakagami River (Reed Harris Environmental Ltd. 2012) and Little Jackfish River (ongoing). The regression approach predicts the maximum relative increase in fish mercury concentrations for a new reservoir, based on the flooded area, total area and mean annual flow (Equation 1). The predicted peak increase factor (e.g. 3X) is then multiplied by existing baseline fish Hg concentrations to predict peak Hg concentrations that will be observed in the reservoir. The model does not predict the timing of the response.

The approach is empirical but has mechanistic underpinnings. It effectively assumes that the primary source of MeHg in a new reservoir is the flooded terrain (numerator in Equation 1), while MeHg removal (denominator in Equation 1) is more efficient in situations with rapid flow rates and dilution. When hydraulic residence times are longer, outflow is less effective at removing MeHg and other mechanisms become more important, including bacterial demethylation, photochemical degradation and sedimentation. The peak increase for fish mercury concentrations should be greater when there is more flooding and flow is less able to remove/dilute MeHg from the reservoir.

$$\text{Peak Increase factor} = k_1\left(\frac{A_{\text{Flooded}}}{(Q + k_2 A_{\text{total}})}\right) + k3 \qquad \text{(Equation 1)}$$

Where:

| | |
|---|---|
| Peak increase factor | = Peak increase factor for fish MeHg concentration |
| $A_{flooded}$ | = Flooded area ($km^2$) |
| $A_{total}$ | = Total reservoir area ($km^2$) |
| Q | = Mean annual flow ($km^3$/yr) |
| $k_1$ and $k_2$ | = Regression coefficients (km/yr) |
| $k_3$ | = Regression coefficient (dimensionless) |

The use of area in the denominator reflects an assumption that MeHg removal mechanisms other than outflow are primarily related to area (e.g. photodegradation, burial and sediment demethylation) rather than volume.

To consider the effects of upstream flooding in river systems with multiple reservoirs, an approach was developed that added areas cumulatively as sites were examined in the downstream direction if certain criteria were met. Reservoirs in series were essentially treated as a single reservoir if inundation occurred less than 10 years apart, the water travel time between reservoirs was 1 week or less, and no large new flows were introduced between reservoirs. Northern pike Hg data from 12 reservoirs in Quebec, Manitoba and Ontario were used to develop and calibrate the model (Harris et al. 2011). Estimates of regional background fish Hg concentrations were adjusted slightly from Harris et al. (2011) and the optimized regression model for the peak relative increase for 700 mm northern pike is as follows:

$$\text{Peak Increase Factor} = 0.427 * \frac{A_{flooded}}{(Q + 0.075 * A_{total})} + 1.77 \qquad \text{(Equation 2)}$$

No long-term monitoring data were available to calibrate the model for conditions in British Columbia, for bull trout or other fish species. Most hydroelectric facilities in British Columbia were constructed before the relationship between inundation and increases in fish mercury was known, so there are no data for at least 20 years post-flood for most reservoirs. The regression developed for northern pike was therefore used as a surrogate for bull trout at the proposed Site C reservoir, as bull trout and northern pike are both predatory fish species.

# 3 MODEL INPUTS

The regression model requires three inputs: flooded area, total area, and mean annual flow (Equation 1). The values used for Site C are shown in Table 1. In addition to a simulation based on the mean annual flow, scenarios were also simulated using low (5$^{th}$ percentile) and high (95$^{th}$ percentile) annual flows to test the potential effects of sustained periods (e.g. years) of high or low flow.

**Table 1. Model inputs for the proposed Site C reservoir.**

| Model Input | Value | Reference |
|---|---|---|
| Flooded area | 53.4 $km^2$ | Volume 1 Section 2 of EIS |
| Total area | 93.3 $km^2$ | Volume 1 Section 2 of EIS |
| Mean annual flow* | 1,160 $m^3$/s mean (36.7 $km^3$/yr) | Appendix V2D |
| Minimum annual flow* | 522 $m^3$/s minimum (16.5 $km^3$/yr) | Appendix V2D |
| Maximum annual flow* | 1,880 $m^3$/s maximum (59.3 $km^3$/yr) | Appendix V2D |
| Baseline [Hg] for 550 mm bull trout | 0.09 mg/kg wet muscle | Azimuth 2011 |

*Peace above Pine exceedences data (1979-95,1997-2007); Appendix V2D is Volume 2, Appendix D Surface Water Regime Technical Memos.

# 4 MODEL RESULTS AND DISCUSSION

Results of the model simulations to predict peak increase factors (relative increases) for the proposed Site C reservoir are shown in Figure 1 and Table 2. Predicted peak concentrations for 550 mm bull trout are shown in Table 2. Predicted peak increase factors for the proposed Site C reservoir ranged from 2.1 to 2.8x baseline, depending on flow through the reservoir (Table 2). Greater flow over the long-term would result in lower predicted increases in fish Hg levels, due to a dilution effect as more water passes through the reservoir. When combined with a baseline concentration of 0.09 µg/g wet muscle (Azimuth 2011), predicted peak Hg concentrations in 550 mm bull trout ranged from 0.20 to 0.25 µg/g wet muscle. Assuming that long-term discharge patterns from Williston Reservoir are similar in the future, the model predicts that mercury concentrations in 550 mm bull trout would peak at slightly more than double baseline the concentration.

These predicted peak concentrations are low in the context of what has been observed in predatory fish in reservoirs in eastern Canada (Schetagne et al. 2003; Bodaly et al. 2007). The low predicted peak values were due to both low to moderate predicted relative increases (2.1 to 2.8X) compared to the observed range up to 7X in eastern Canada (Figure 1), and low baseline concentrations for bull trout in the study area (Table 2).

Results presented here should be viewed in the context of a screening level assessment. This is particularly the case because a calibration of the model to conditions in eastern Canada has been used to predict the relative increase expected for British Columbia conditions; northern pike data were also used as a surrogate for bull trout when predicting relative increases (both are top-level predators). However, the baseline mercury concentration for bull trout is based on empirical data from the Peace River within the technical study area (Azimuth 2011).

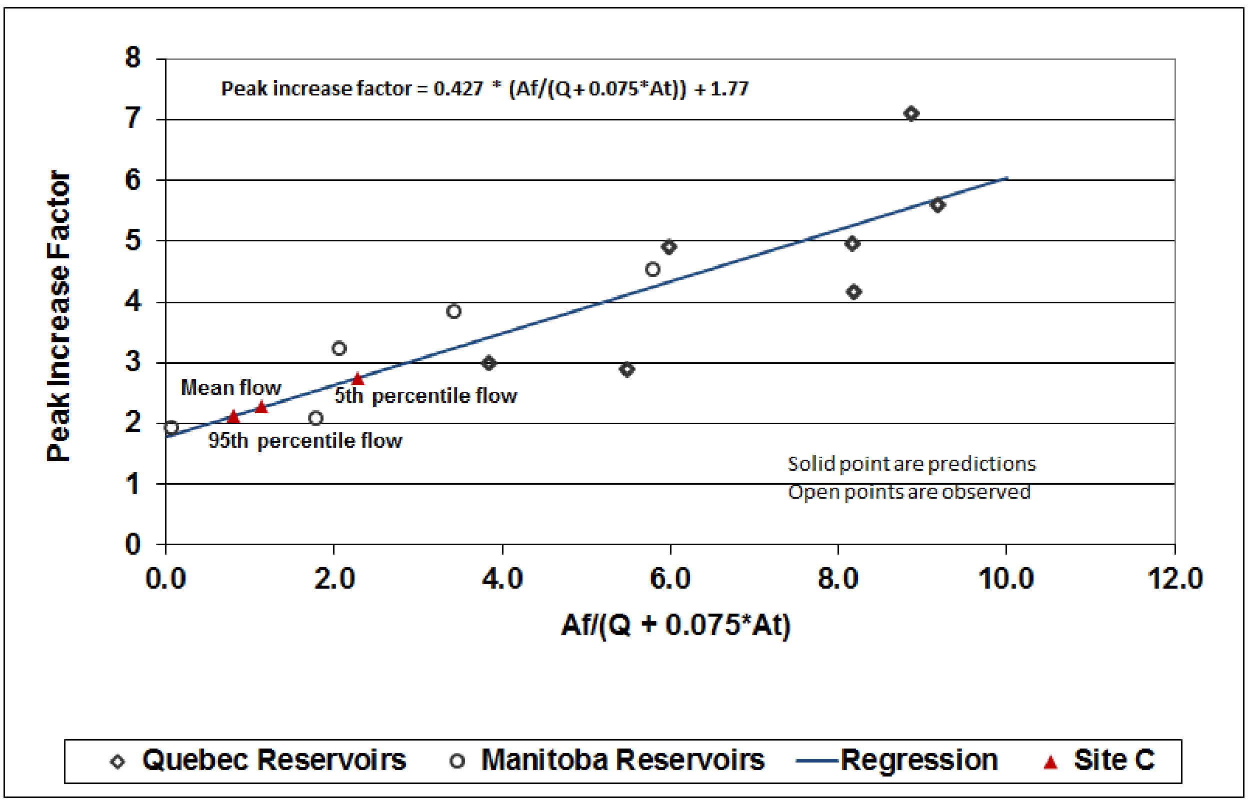


**Figure 1. Predicted peak increase factor for adult predatory fish species (Bull Trout) in the proposed Site C reservoir.**

Results are shown for scenarios with mean, 5$^{th}$ percentile and 95$^{th}$ percentile for annual flow. Data for Manitoba sites derived from Bodaly et al. (2007). Data for Quebec sites derived from Schetagne et al. (2003).

**Table 2. Predicted peak mercury concentrations in bull trout (standardized 550 mm) for the Site C reservoir.**

| Scenario | Predicted peak increase factor (relative increase) | Baseline Hg (mg/kg wet muscle) | Predicted peak concentration (mg/kg wet muscle) |
|---|---|---|---|
| Site C mean flow | 2.3 | 0.09 | 0.20 |
| Site C 5$^{th}$ percentile flow | 2.8 | 0.09 | 0.25 |
| Site C 95$^{th}$ percentile flow | 2.1 | 0.09 | 0.19 |

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## Appendix B

## Calibration of fish growth rates for existing conditions in Peace River (proposed Site C reservoir)

## Calibration of Fish Growth Rates for Existing Conditions in Peace River

## (Proposed Site C reservoir site)

**Fish growth rates for the pre-flood simulation were calibrated using data from Azimuth (2011) and Appendix V2J Part 1. Calibrations of length versus weight, weight versus age, and length versus age are shown below.**

**<u>Bull trout</u>**

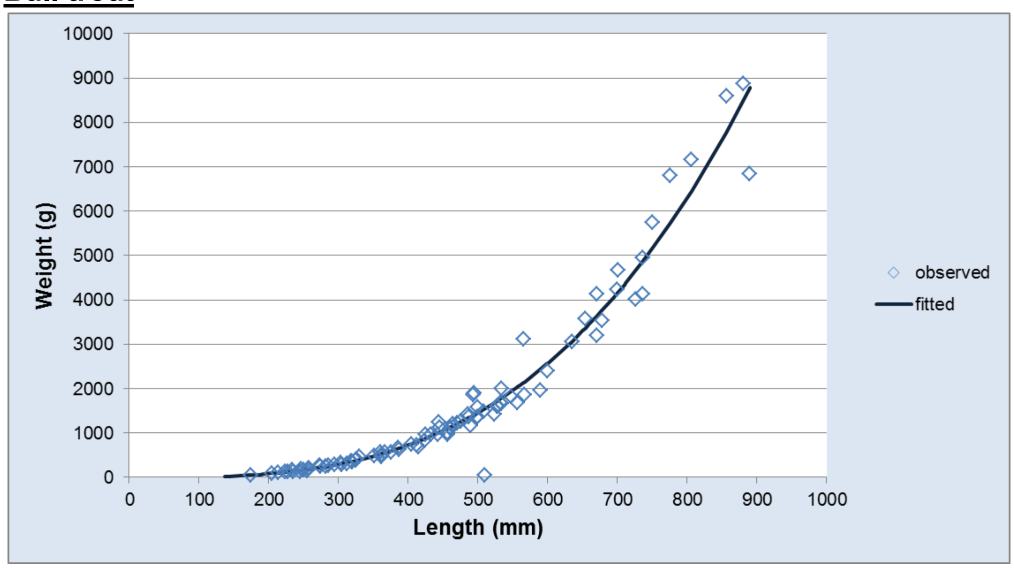


**Figure B-1. Fitted and observed weight vs. length for Peace River bull trout.**

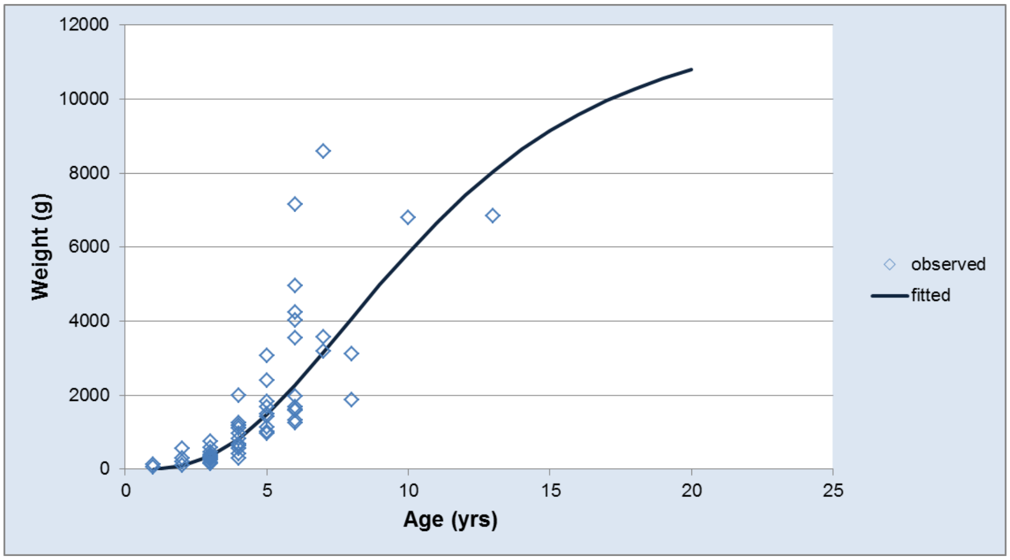


**Figure B-2. Modelled and observed growth for Peace River bull trout (weight vs. age).**

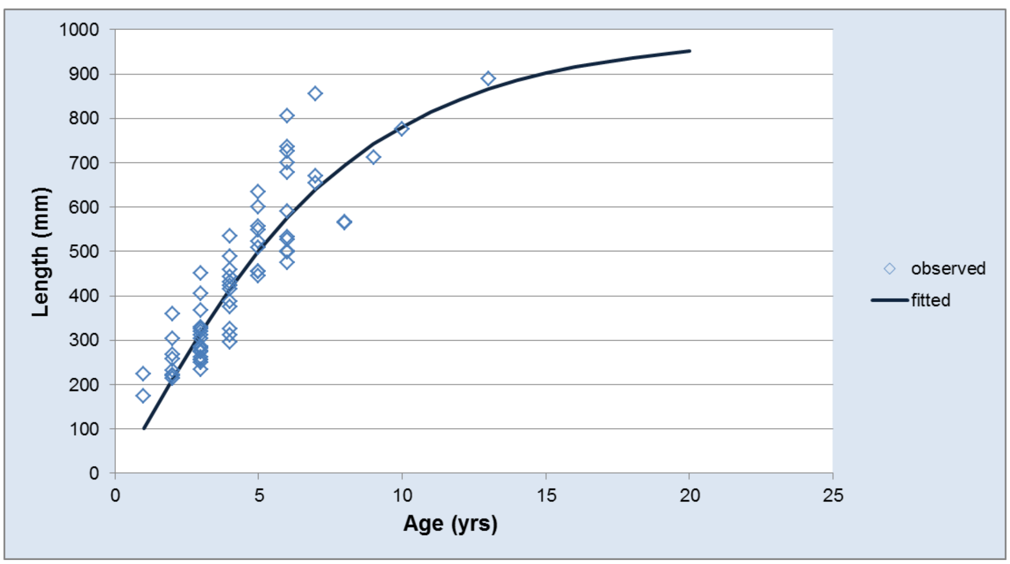


**Figure B-3. Modelled and observed growth for Peace River bull trout (length vs. age).**

**Mountain whitefish**

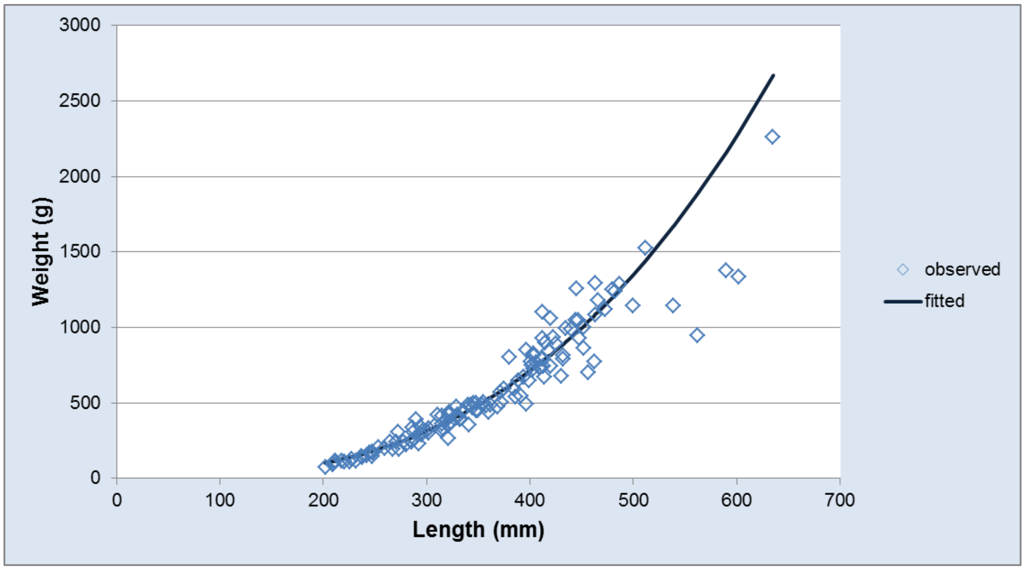


**Figure B-4. Fitted and observed weight vs. length for Peace River mountain whitefish.**

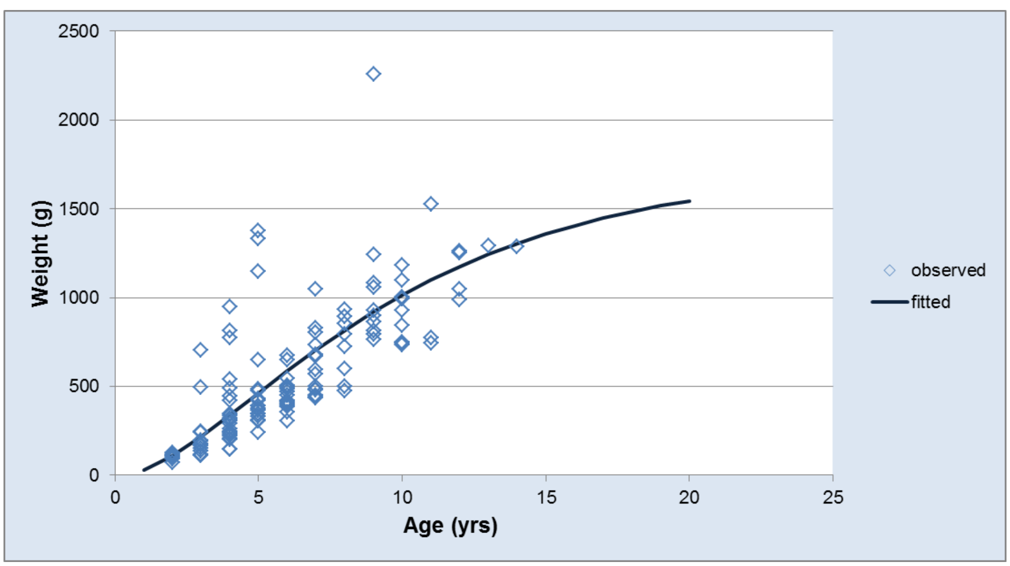


**Figure B-5. Modelled and observed growth for Peace River mountain whitefish (weight vs. age).**

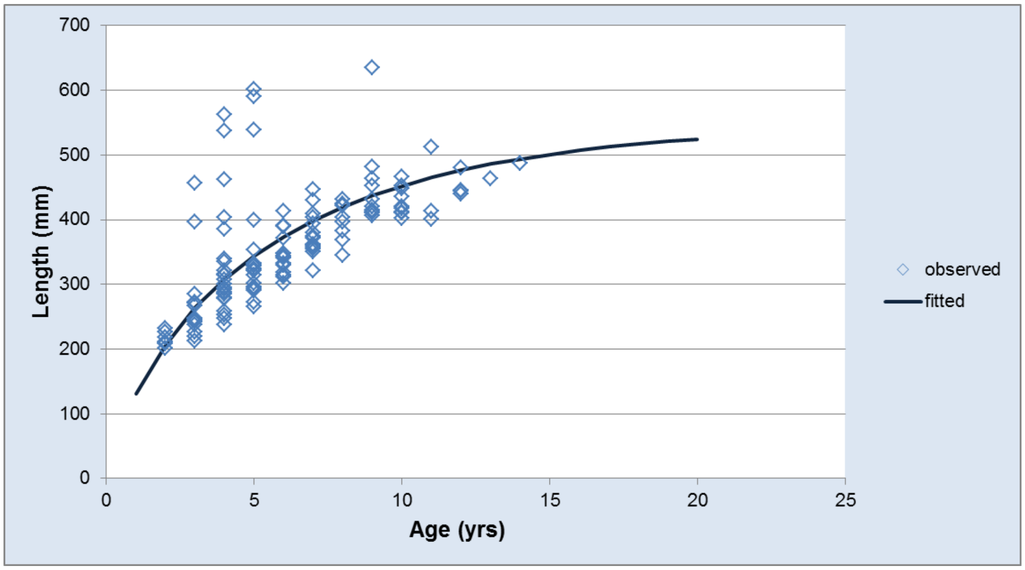


**Figure B-6. Modelled and observed growth for Peace River mountain whitefish (length vs. age).**

<u>**Longnose sucker**</u>

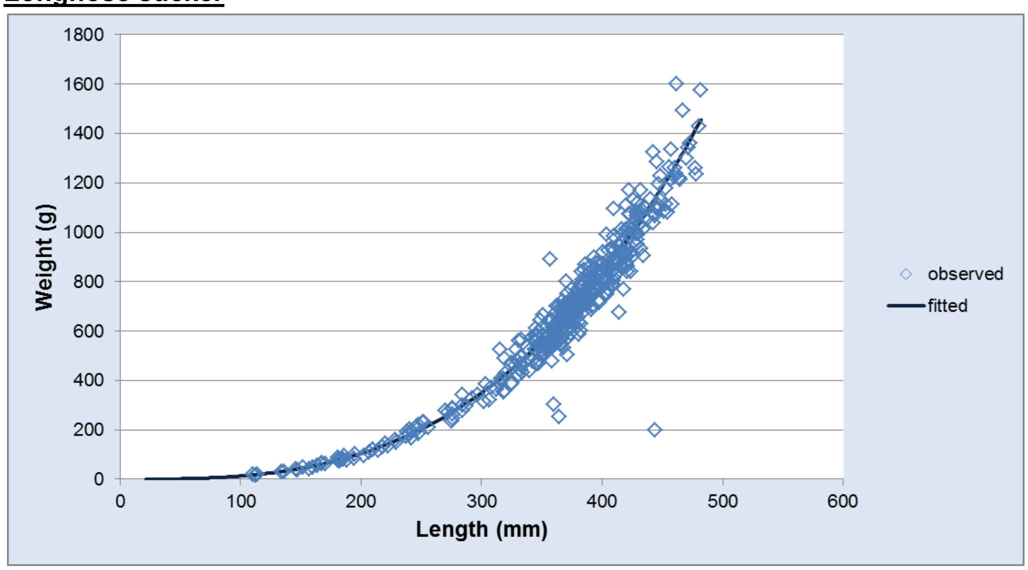


**Figure B-7. Fitted and observed weight vs. length for Peace River longnose sucker.**

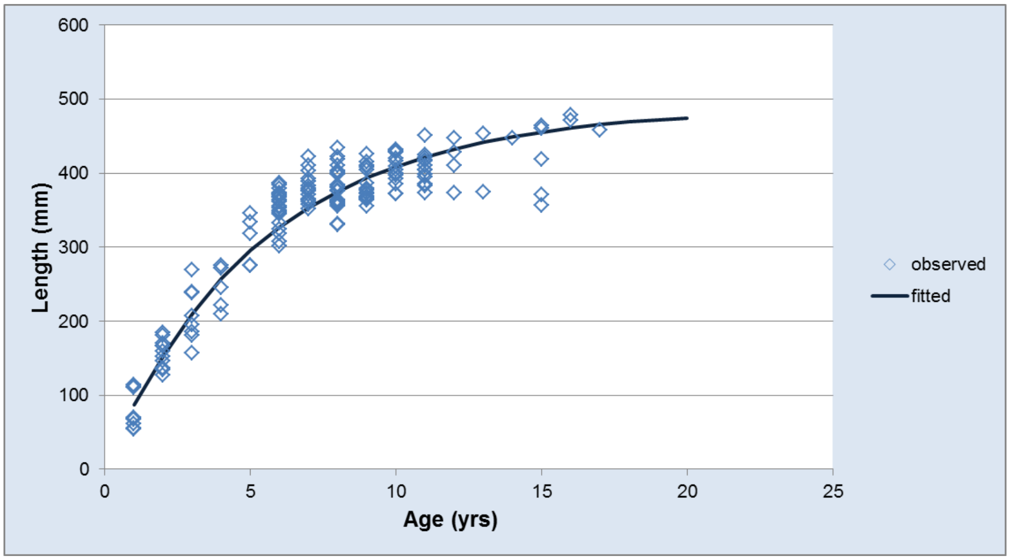


**Figure B-8. Modelled and observed growth for Peace River longnose sucker (weight vs. age).**

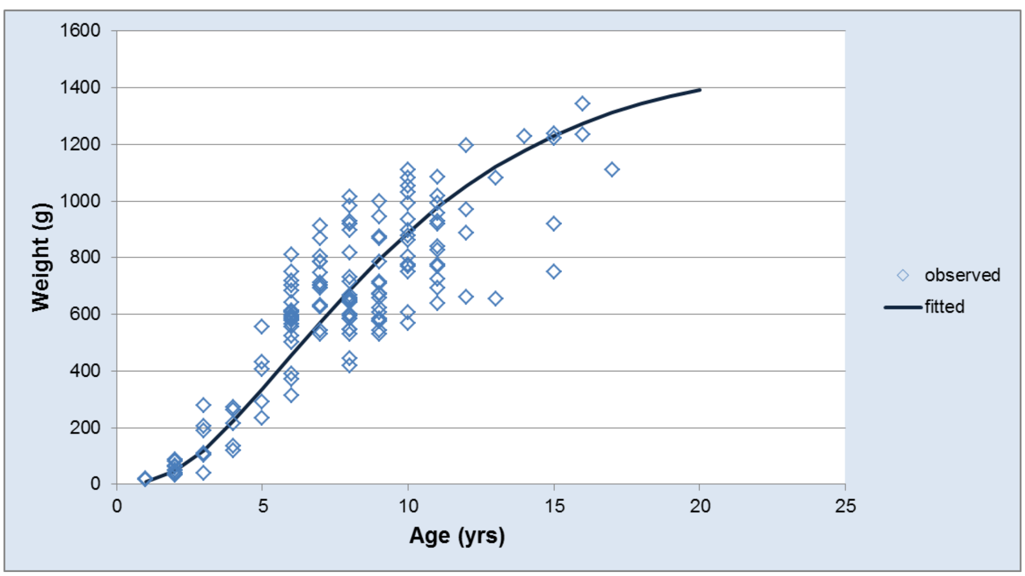


**Figure B-9. Modelled and observed growth for Peace River longnose sucker (length vs. age).**

## Redside shiner

Growth data from Scott and Crossman (1973)

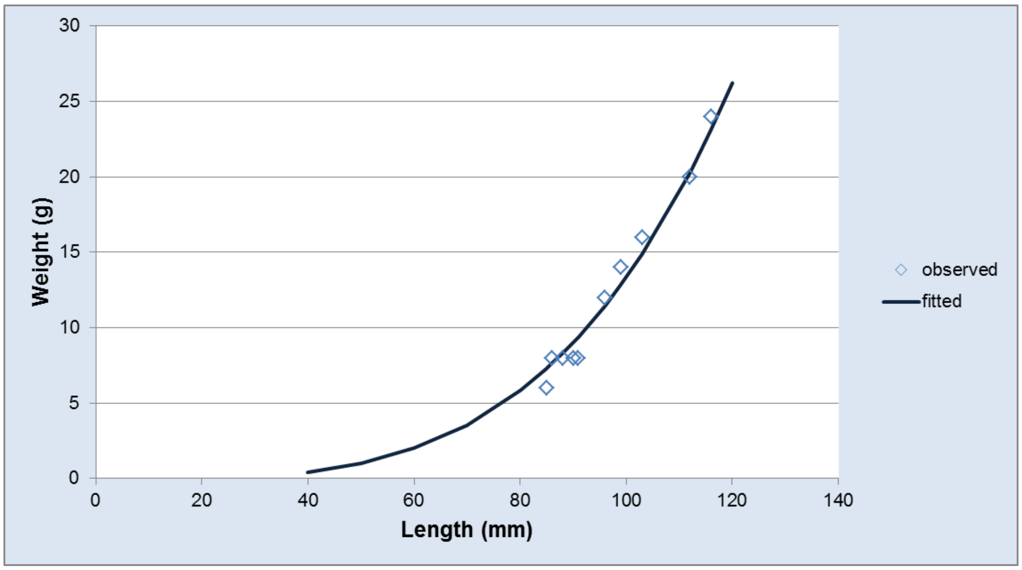


**Figure B-10. Fitted and observed weight vs. length for Peace River redside shiner**

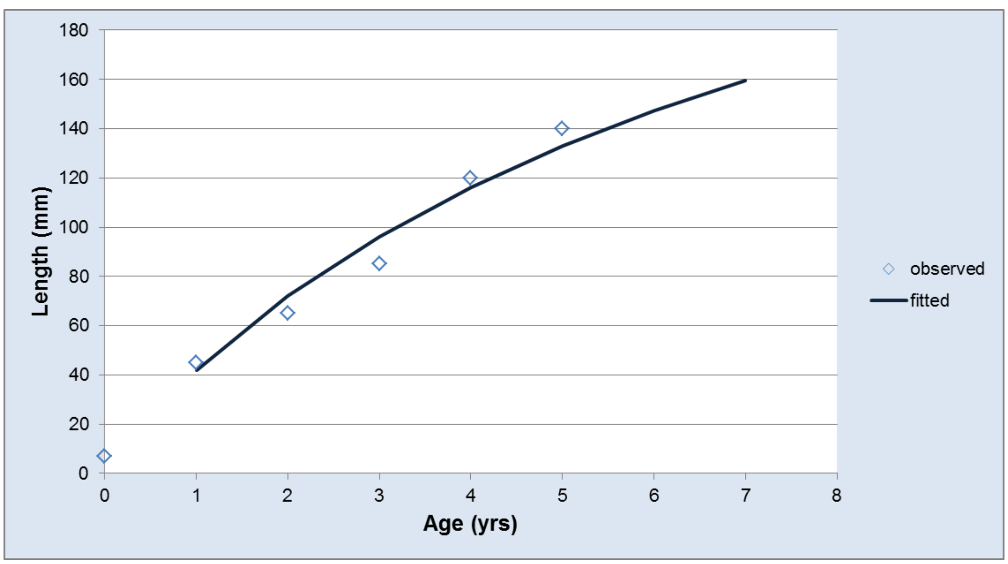


**Figure B-11. Modelled and observed growth for Peace river Redside shiner (length vs. age).**

## Rainbow trout

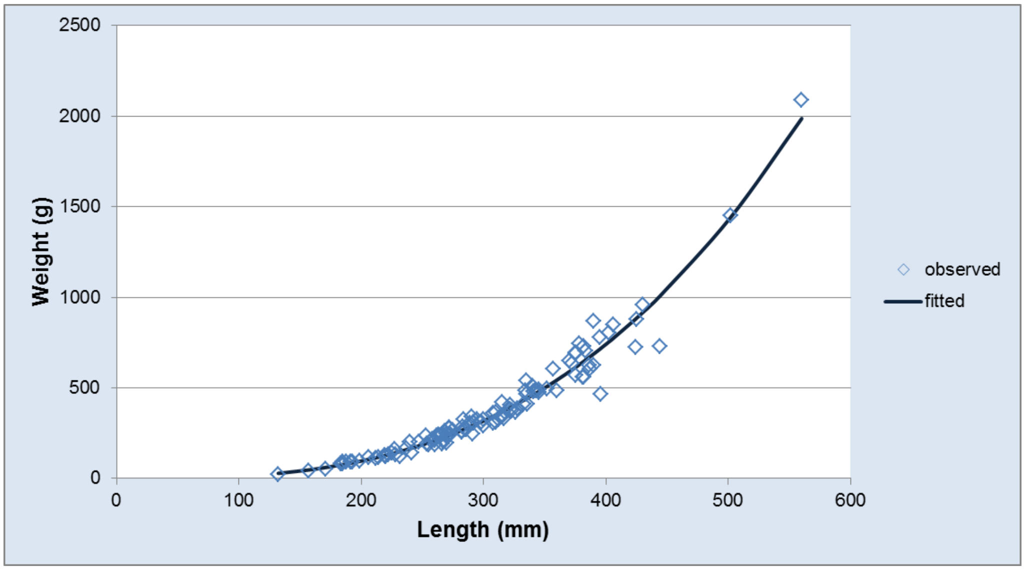


**Figure B-12. Fitted and observed weight vs. length for Peace river rainbow trout.**

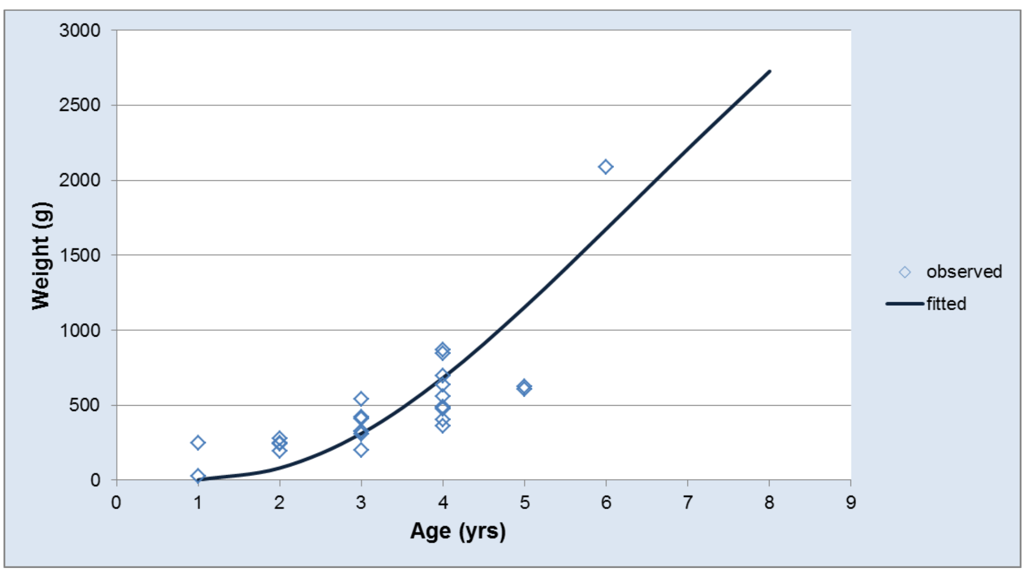


**Figure B-13. Modelled and observed growth for Peace River rainbow trout (weight vs. age).**

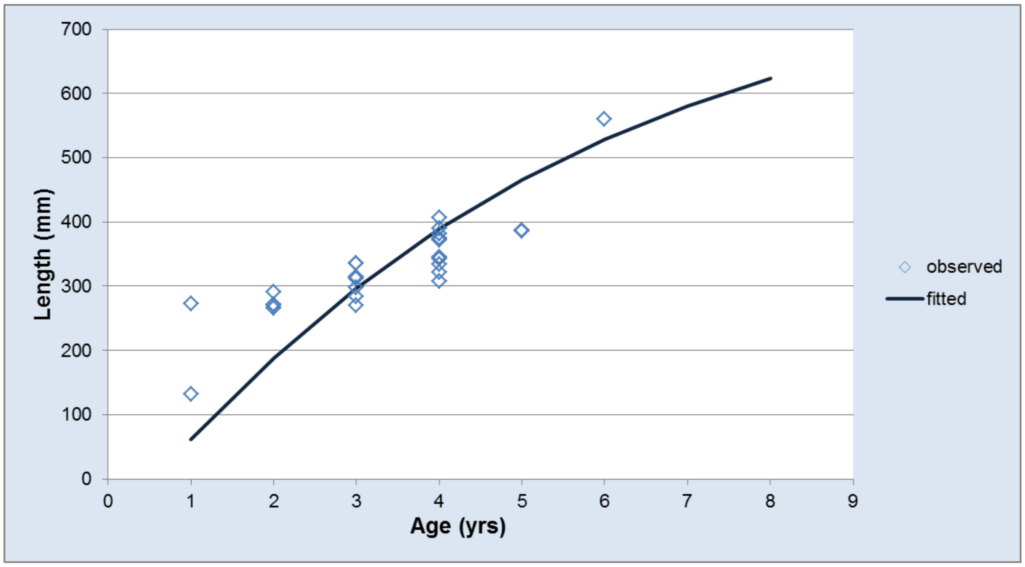


**Figure B-14. Modelled and observed growth for Peace river rainbow trout (length vs. age).**