From: mariamoran@lowerchurchillproject.ca
Sent: Monday, August 27, 2018 3:13 PM
To: johannes.lampe@nunatsiavut.com
Cc: jim.goudie@nunatsiavut.com "Bert Pomeroy"; Kevin Burt
Subject: Reports relating to methylmercury accumulation and transport from the Muskrat Falls reservoir
Attachments: Harris tech memo on Muskrat Falls Reservoir modeling Aug 3 2018 final.002.pdf; Baird (2018) 12985.101.M3.RevB.Lake Melville Modelling Summary.002.pdf; Predicted Increases in Fish Methylmercury Muscle Tissue Concentrations in Goose Bay and Lake Melville_Appendix A_Appendix B.002.pdf; Azimuth HHRA Technical Memo_July 23 2018.002.pdf; Lake Melville MeHg Mass Balance Perspective_Feb 25 _Final.002.002.pdf; James McCarthy short bio_Nalcor_August 7 2018.pdf; Randy Bio Blurb_August 2018.pdf; Reed Harris biosketch May 2017.pdf; Rob Willis bio for MeHg workshops and other purposes_August 2018.pdf

Hello President Lampe

Attached are four reports that detail the recent research completed by expert consultants in relation to methylmercury accumulation and transport from the Muskrat Falls reservoir into the downstream environment. These topics were discussed through the IEAC and I am pleased to provide you with copies of these reports. They are:

1) Modelling to predict changes to MeHg concentrations generated within the reservoir. This work was led by Reed Harris of Reed Harris and Associates.

Harris tech memo on Muskrat Falls Reservoir modeling Aug 3 2018 final.pdf

2) Modelling to predict MeHg transport from the reservoir downstream through Goose Bay and Lake Melville. This study was led by Alex Brunton of Baird and Associates.

Baird (2018) 12985.101.M3.RevB.Lake Melville Modelling Summary.pdf

3) Research led by James McCarthy of Wood Group (formerly AMEC Foster-Wheeler) to predict MeHg bioaccumulation in potentially affected aquatic species downstream of Muskrat Falls.

Predicted Increases in Fish Methylmercury Muscle Tissue Concentrations in Goose Bay and Lake Melville_Appendix A_Appendix B.pdf

4) Evaluation of human health risk led by Randy Baker of Azimuth Consulting Group.

Azimuth HHRA Technical Memo_July 23 2018.pdf

We will be posting these reports on our Muskrat Falls Project website. As the Azimuth report references a report previously completed in February 2018 considering a methylmercury mass-balance discussion, I have also included a copy of that report for your convenience.

Lake Melville MeHg Mass Balance Perspective_Feb 25_Final.pdf

I encourage you to review the various reports to gather the full context of all of the material, but I would like to point out that Azimuth stated "In summary, there is an extremely low likelihood of risk to human health from consumption of seafood from Goose Bay or Lake Melville at peak mercury levels in a post-impoundment scenario."

We would appreciate the opportunity to provide more information on this topic. The consultants are available next week to discuss this information and answer any questions you may have. If you are not available during that week, let me know a date that works for you and we'll work to accommodate that timing.

Attached are the bios of the expert consultants who conducted the research.

James McCarthy short bio_Nalcor_August 7 2018.pdf

Randy Bio Blurb_August 2018.pdf

Reed Harris biosketch May 2017.pdf

Rob Willis bio for MeHg workshops and other purposes_August 2018.pdf

If you have any questions, please contact me.

Maria

Maria Moran
Manager, Industrial Benefits and Stakeholder Relations
PROJECT DELIVERY TEAM
Lower Churchill Project
t. **709 737-1942**

c. **709 693-5397**

f. **709 737-1985**
e. MariaMoran@lowerchurchillproject.ca
w. muskratfalls.nalcorenergy.com

You owe it to yourself, and your family, to make it home safely every day. What have you done today so that nobody gets hurt?

Azimuth Consulting Group Partnership

218-2902 West Broadway
Vancouver, BC
Canada V6K 2G8

Phone: 604-730-1220
Fax: 604-739-8511
www.azimuthgroup.ca

Technical Memorandum

Date: July 19, 2018
To: Peter Madden, Nalcor Energy
From: Randy Baker
Our File: NE 18-01

RE: Summary of Post-Exposure Human Health Risk Assessment from Methylmercury in Seafood in Goose Bay and Lake Melville, Labrador

---

Objective

The objective of this Technical Memo is to summarize the effects of updated methylmercury (MeHg) modelling and predicted increases in key aquatic species on the Human Health Risk Assessment (HHRA). Dillon Consulting Ltd. completed the initial HHRA on MeHg prepared for Nalcor Energy (Nalcor) in 2016 (Dillon 2016). This provisional update incorporates the most recent science on the manufacture and release of MeHg from the Muskrat Falls Reservoir downstream of the lower Churchill River to Goose Bay and Lake Melville. Based on this, we discuss the implications for human health based on predicted changes in MeHg concentrations in fish and marine mammals. A complete review and update of the 2016 HHRA will be completed by Dillon later in 2018.

Background and Assumptions

In 2016, Dillon completed a full HHRA to determine baseline human health risks from exposure to MeHg in country and store-bought foods consumed by residents of four study area communities prior to creation of the Muskrat Falls Reservoir on the lower Churchill River (the Project). These were Happy Valley-Goose Bay, North West River, Mud Lake and Sheshatshiu.

Various food ingestion-based exposure pathways were assessed for males and females of all age classes in each of the above communities. In all, 293 community members underwent a dietary survey (DS), food consumption rate questionnaire and a human biomonitoring program (HBP), involving human hair sampling and analysis for MeHg.

The Dillon (2016) HHRA used standard methods developed and endorsed by Health Canada (2007, 2010a, 2010b), incorporating reasonable levels of conservatism in its various approaches, models and assumptions to overestimate exposure to MeHg and quantify potential risk to humans. Several lines of evidence were followed that comprised the main outcomes of the HHRA including:

- Calculated human health risk estimates for each of the assessed human receptors (e.g., toddlers, teenagers, adults), in the communities above, expressed as hazard quotients (HQs; i.e., the estimated exposure to MeHg divided by the toxicological reference value(s) (TRVs) for MeHg);
- The relative proportion of MeHg exposure that is attributed to country food vs store-bought food (e.g., tinned tuna);
- Appropriate consideration of safety factors and assumptions with respect to data variability and uncertainty;
- Comparison of local aquatic biota tissue concentration against regulatory standards; and
- Evaluation and consideration of measured human biomonitoring data such as hair MeHg values relative to Health Canada guidance.

The reader is encouraged to consult the Dillon (2016) HHRA for a full explanation of the approach, procedures, assumptions and conclusions. These have not substantially changed since the initial assessment was made and their assessment of current risk is still valid. Note also that while the health benefits of consuming country foods (e.g., fish, seals, game) was recognized by the HHRA, this was not balanced against exposure risk; health benefits were considered as part of forthcoming consumption advice.

The Dillon (2016) HHRA concluded that "t*here is a low to negligible potential for human health risk resulting from MeHg exposure, and a negligible potential for human health risk resulting from inorganic Hg*(mercury) *exposure*. *The calculated MeHg and inorganic Hg exposures and risks are similar to what would be expected in numerous communities in North America where food consumption patterns comprise the ingestion of both store-bought foods and country food items that are of aquatic origin*". Note that the Dillon (2016) HHRA also considered inorganic mercury (Hg); given that virtually all environmental exposure to MeHg is via dietary sources of fish, shellfish and marine mammals (Health Canada 2007, 2010a, 2010b) this update focuses only on MeHg. The exposure scenario for inorganic Hg is unchanged since 2016.

It is noteworthy that the Dillon (2016) HHRA considered *current* health risks, prior to Project-related changes, while recognizing the concern expressed by the downstream communities with respect to increased potential for health risks due to higher MeHg exposure. As a result, on behalf of Nalcor, there was a commitment to comprehensive monitoring and a science-based risk management program, if warranted. Concerns expressed by Indigenous communities were further heightened following publication of the Calder et al. (2016) study, purporting that risks to human health from MeHg exposure were significantly higher than forecast.

As noted above, this Memorandum acknowledges that two of the key baseline HHRA program components consisted of a dietary survey and hair sampling in all study area communities except Rigolet. Results of these two components provided critical information to the HHRA regarding current patterns and level of risk from MeHg exposure. Work by Calder et al. (2016) also incorporated dietary surveys and hair MeHg monitoring at Rigolet and showed only slightly higher reliance on country foods and hair mercury concentrations (see the July 2018 Human Health PPT Presentation for further details). This information is still current and accurate and helps to inform perspective on both current and predicted risk.

Disclaimer

Note that due to circumstances beyond the control of Mr. R. Willis of Dillon Consulting, the main author of the 2016 HHRA, was unable to update the HHRA at this time. Thus, Azimuth was engaged to provide this expertise in the interim. Azimuth has considerable experience addressing human health risks related to MeHg from previous projects and this document follows on from the Dillon (2016) HHRA to incorporate more recent findings related to predicted changes in MeHg concentrations in key species in Goose Bay and Lake Melville. The Dillon (2016) HHRA will be revised at a later date.

2018 Update – Background Information

As noted above, this Technical Memorandum updates the key 2016 HHRA findings by incorporating the most recent investigations from the documents listed below. In this update, we have also predicted the change in tissue MeHg concentration in the most frequently consumed species by local residents in Goose Bay and Lake Melville. Risk predictions have been updated relative to baseline using the most recent tissue MeHg data from monitoring programs, bringing to bear the best available science to predict changes in MeHg in downstream species. With respect to risk predictions, rather than calculating a HQ (i.e., by comparing the ratio of what *could be* eaten relative to what is eaten based on local surveys), we have determined the incremental risk using Health Canada's total daily intake (TDI; Health Canada 2010a) value for MeHg. We calculated the difference in number of seafood meals that can be consumed at post-inundation peak MeHg values relative to baseline while remaining within Health Canada guidelines. This format provides similar but more relevant and meaningful information that Indigenous community members and other stakeholders can understand and base decisions on. There are four main documents from which these predictions are based:

1. **Harris and Hutchinson July 2018** – Predicted changes to MeHg concentrations generated within Muskrat Falls were made using a combination of mechanistic modeling predictions based on results of ResMerc and empirical data from the FLUDEX experiment at the Experimental Lakes Area (ELA) in Ontario. These results were used to predict MeHg concentrations in both surface (i.e., upper 20 m epilimnion and mixing zone) and deeper (>20 m hypolimnion) water in Goose Bay and Lake Melville. Predicted relative increases above existing concentrations were used as an indication of the potential relative increase in biota. For example, if water concentrations increased 50%, so would biota concentrations. Because fish exposure to MeHg is integrated over time, a brief increase in water concentration would not translate into the same relative increase

in biota. Predicted increases in water were therefore averaged over three years in order to predict increases in biota MeHg.

2. **Baird 3-D Hydrodynamic Modeling July 2018** – Baird used high-resolution Delft3D hydrodynamic modeling in Goose Bay and Lake Melville to make conservative predictions regarding the export of MeHg from the reservoir over time, accounting for a variety of factors including dilution as well as losses of MeHg due to photodegradation and settling. The models used five-year predictions of excess (i.e., over and above baseline) MeHg leaving the lower Churchill River to be dispersed in the marine environment. Estimates of excess MeHg concentrations in Goose Bay and Lake Melville were then used in calculations of the relative increases in water column MeHg concentrations above baseline.
3. **AMEC Foster-Wheeler (AMEC) July 2018** – This document summarizes life history and habitat use for key species identified as being important in the diet of local communities harvesting wildlife species in Goose Bay and Lake Melville. This summary also recognizes local Traditional Ecological Knowledge. Application of life history and diet information is used to determine the relative degree to which biota may be exposed to water where MeHg has increased in Lake Melville, post-impoundment. The degree of both spatial and temporal overlap of each species within the zone of exposure was used to prorate the magnitude of exposure, relative to baseline. That is, where, when and for how long a species resides and feeds in either Goose Bay and western and eastern Lake Melville determines exposure and the magnitude of increase in tissue MeHg concentration. The peak increase in MeHg concentrations in water over three years was used to determine the maximum degree to which biota may theoretically respond. The conservative assumption was made that biota tissue concentrations would increase in relative proportion to the increase in water MeHg concentrations for species 'fully exposed' to water with higher MeHg. There is a reasonable amount of conservatism built into this assumption, so as to not underestimate the potential magnitude of change.
4. **Azimuth Consulting Group February 25, 2018** – Evaluation of MeHg production by Muskrat Falls Reservoir and implications for Lake Melville – A top-down, mass-balance approach. This document, makes the case that the biomass of the estuarine/marine environment is considerably greater than the freshwater environment and will nearly entirely dampen any downstream changes in water or biota due to reservoir creation. While this argument has not been brought to bear in hydrodynamic or export models (ResMerc or FLUDEX), or on risk predictions, this should be considered.

2018 Update – Risk Characterization

Dillon (2016) concluded that HQ values suggest a negligible to low potential for human health risk among residents of the study area communities under baseline conditions, particularly given the conservative assumptions used in the HHRA. However, the HHRA did not predict future risk because an estimate of the magnitude of change in tissue MeHg was not available at the time.

Tissue MeHg concentrations of aquatic organisms will only increase if an individual spends a meaningful amount of time feeding within the zone of exposure in Goose Bay and Lake Melville.

Thus, as noted above, life history features of aquatic biota reportedly consumed by local residents must be considered. Although some species were reported as being consumed by some residents (AMEC 2018), several of these were not considered further in the risk analysis for the following reasons:

- lake trout (*Salvelinus namaycush*) - are not present outside of the river mouth and are very rare within the lower portion of the river (i.e. Muskrat Falls reservoir area)
- Atlantic salmon (*Salmo salar* - both anadromous and land-locked) - the landlocked form is very rare within the lower portion of the river (i.e., Muskrat Falls reservoir area) and anadromous returning salmon from the Labrador Sea cease feeding as they enter freshwater of Lake Melville
- Atlantic cod (*Gadus morhua*) - this species has not been documented within Lake Melville
- Capelin (*Mallotus villosus*) - this species has only rarely been observed in Lake Melville since the early 1970s
- Arctic char (*Salvelinus alpinus*) - This species is not found in the lower Churchill River below the Labrador Plateau and only rarely observed in Lake Melville and typically found beyond the Narrows at the eastern end of Lake Melville.

Only three species are abundant, frequently consumed and exposed to changes in MeHg in water in Goose Bay and Lake Melville. These are brook trout (*S. fontinalis*), rainbow smelt (*Osmerus mordax*) and ringed seal (*Phoca hispida*) and are the only species considered in this provisional update to the HHRA. **Table 1** depicts the mean baseline tissue MeHg concentration for each of the three species, the increase factor predicted for Goose Bay and Lake Melville (AMEC 2018) and the predicted post-impoundment increase in MeHg.

Table 1. Mean baseline[1] MeHg concentration (mg/kg) in Goose Bay (GB) and Lake Melville (LM) and increase factor for post-impoundment peak MeHg concentration for key species.

Species	Baseline [MeHg] mg/kg in GB / LM	Increase Factor in GB / LM	Post-Impoundment Peak MeHg (mg/kg)
Brook trout	0.07 / 0.04	1.8 x / 1.2 x	0.13
Rainbow smelt	0.04 / 0.02	2.1 x / 1.5 x	0.06
Ringed seal pup	0.09 LM only	1.2 x	0.11
Ringed seal adult	0.62 LM only	1.2 x	0.74

1 From AMEC (2018)

Note that these values represent the peak three-year increase in water MeHg concentration in Goose Bay and Lake Melville to acknowledge life history features (AMEC 2018) and that time is required to reach equilibrium in key downstream species.

The Health Canada provisional tolerable daily intake (pTDI) for MeHg is a benchmark of acceptable exposure for chronic oral exposure from all sources over a lifetime without harmful effect. Two TDIs for MeHg have been issued by Health Canada, one TDI for the general population and a second to protect 'sensitive' receptors such as women of child-bearing age and children less than 12 years of age. These are stated as a dose of 0.47 µg and 0.20 µg MeHg/kg body weight/day (µg/kg bw/d) respectively (Health Canada 2010a).

To determine the number of weekly seafood servings that can be safely consumed over a lifetime for different receptor groups this simple equation is used:

Servings/Week = (pTDI x body weight (kg) x 7 days) / (Fish [MeHg] x Serving Size gm)

The Dillon (2016) HHRA used locally derived body weights for toddlers, children, teens and adults, by gender (male, female) from the communities. For example, on average, adult males weighed 83 kg, while adult females weighed 70 kg. The assumed average serving size for adults was approximately 170 g, less than for seal meat (193 g). Gender and age-specific serving sizes were used to more accurately estimate dose.

Retaining these site-specific values and using the above equation, we determined the total number of servings per week, by species, by age and gender, that is permissible under Health Canada guidance for a lifetime exposure. **Table 2** presents the total number of weekly servings under current baseline conditions and for the peak three-year period in the years following full impoundment of the Muskrat Falls Reservoir.

Based on results presented in **Table 2** (below) a considerable number of seafood meals are permissible per week under both baseline and post-impoundment scenarios without an effect to humans. For example, an adult can currently safely consume 23 meals per week of brook trout, diminishing to 13 meals per week post-impoundment from Goose Bay. In Lake Melville, this is currently 40 weekly meals, diminishing to 32 meals post-impoundment. Relatively fewer meals are permissible for 'sensitive' receptors such as children and women of child-bearing age. Diet of adult seal muscle is more restrictive, with 3 weekly meals pre-impoundment, diminishing to 2 at peak post-impoundment. To provide perspective using tinned tuna as an example, the number of weekly servings of tuna is more restrictive, at about half what can be consumed from locally harvested species.

The large number of permissible weekly servings of fish and seal pups is due to the very low baseline MeHg concentrations in Goose Bay and Lake Melville and the relatively low increase in tissue MeHg predicted for all key receptors, based on the work by Harris and Hutchinson (2018), Baird (2018) and AMEC (2018).

Summary and Conclusions

In summary, there is an extremely low likelihood of risk to human health from consumption of seafood from Goose Bay or Lake Melville at peak mercury levels in a post-impoundment scenario. Note that the projected increase of tissue MeHg by ~20% does <u>not</u> account for the effect of the large downstream biomass (i.e., the 'biomass effect') of Lake Melville (Azimuth 2018) which would further considerably diminish the predicted downstream increase. Thus, the original conclusions of the Dillon (2016) HHRA remain valid and are further strengthened by this quantitative assessment and provisional update to examine post-inundation MeHg concentrations in seafood and their implications for human consumption.

Literature Cited

Amec Foster Wheeler. 2017. Aquatic environmental effects monitoring program 1998 to 2016 baseline conditions Muskrat Falls. Prepared for Nalcor Energy by AMEC, St. John's NL. August 2017.

Amec Foster Wheeler. 2018. Predicted increases in fish MeHg muscle tissue concentrations in Goose Bay and Lake Melville. Prepared for Nalcor Energy by AMEC, St. John's NL. July 2018.

Brunton. A. 2018. Model analysis of Goose Bay and Lake Melville hydrodynamics and transport of methylmercury from Muskrat Falls Reservoir. Baird Technical Memorandum to R. Harris. July 2018.

Calder, R.S.D., A.T. Schartup, M. Li, A.P. Valberg, P.H. Balcom, and E.M. Sunderland. 2016. Future Impacts of Hydroelectric Power Developments on Methylmercury Exposures of Canadian Indigenous Communities. Environ. Sci. Technol. 50(23): 13115-13122.

Harris, R.C. and D. Hutchinson. 2018. Updated analysis of methylmercury increases in Muskrat Falls Reservoir. Technical Memorandum to P. Madden (Nalcor), July 2018.

Health Canada. 2007. Human Health Risk Assessment of Mercury in Fish and Health Benefits of Fish Consumption. Ottawa, ON, Health Canada, Health Products and Food Branch, Food Directorate, Bureau of Chemical Safety.

Health Canada. 2010a. Federal Contaminated Site Risk Assessment In Canada. Part II: Health Canada Toxicological Reference Values (TRVs) and Chemical-Specific Factors, Version 2.0. Safe Environments, Contaminated Sites Division, Ottawa, ON.

Health Canada. 2010b. Federal Contaminated Site Risk Assessment In Canada. Supplemental Guidance on Human Health Risk Assessment for Country Foods ($HHRA_{Foods}$). Safe Environments, Contaminated Sites Division, Ottawa, ON.

Schartup, A.T., P. H. Balcom, A.L. Sorensen, K.J. Gosnell, R.S.D. Calder., R.P. Mason and E.M. Sunderland. 2016. Freshwater discharges drive high levels of methylmercury in Arctic marine biota. PNAS 112(38): 11789-11794.

AZIMUTH

Table 2. Weekly seafood servings that can be consumed without exceeding Health Canada's pTDI for MeHg.								
		Average[1] peak MeHg concentration	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
Species	**Location**	Serving Size mg MeHg/kg wet wt	75	100	150	163	150	170
Brook Trout	**Goose Bay - Baseline**	**0.07**	**4**	**7**	**8**	**10**	**19**	**23**
	Goose Bay Post-Impoundment	**0.12**	**2**	**4**	**5**	**6**	**11**	**13**
Brook Trout	**Lake Melville - Baseline**	**0.04**	**8**	**12**	**14**	**18**	**33**	**40**
	Lake Melville- Post-Impoundment	**0.05**	**6**	**10**	**11**	**14**	**27**	**32**
Rainbow Smelt	**Goose Bay - Baseline**	**0.02**	**15**	**25**	**28**	**36**	**67**	**80**
	Goose Bay - Post-Impoundment	**0.04**	**7**	**12**	**14**	**17**	**32**	**38**
Rainbow Smelt	**Lake Melville - Baseline**	**0.04**	**8**	**12**	**14**	**18**	**33**	**40**
	Lake Melville - Post-Impoundment	**0.06**	**5**	**8**	**9**	**12**	**22**	**27**
Ringed seal (pup)	**Lake Melville - Baseline**	**0.09**	**4**	**6**	**6**	**8**	**15**	**18**
	Lake Melville - Post-Impoundment	**0.11**	**3**	**5**	**5**	**7**	**13**	**15**
Ringed seal (adult)	**Lake Melville - Baseline**	**0.62**	**0.5**	**1**	**1**	**1**	**2**	**3**
	Lake Melville - Post-Impoundment	**0.74**	**0.4**	**1**	**1**	**1**	**2**	**2**

Notes

1 Arithmetic means of target size fish (AMEC 2018) after 3-year exposure; Seal serving size is 193 gm

Reference # 12985.101

Status: Correspondence

2nd August 2018

Attention: **Reed Harris**

CC: Rob Nairn

From: Alex Brunton

RE: Lake Melville Model Setup and Results

Summary

Two numerical simulation models were used to predict the downstream fate of methylmercury (MeHg) generated in the Muskrat Falls Reservoir flood zone. A high-resolution hydrodynamic model of Goose Bay and Lake Melville was applied using the Delft3D model to examine the effects of downstream mixing and dilution. An associated box model was created to further account for losses due to photodegradation and settling. The models used five-year predictions of excess MeHg entering the lower Churchill River following reservoir filling. These predictions were based on two estimates of the reservoir flood zone signal: (1) Predictions from a mechanistic model (ResMerc) and (2) estimates derived using field data from the FLUDEX experiment in the Experimental Lakes Area in Ontario. Both estimates were developed by Reed Harris Environmental Ltd (Harris and Hutchinson, 2018). Results of simulations using the two loading estimates were averaged for the final analysis.

Area-weighted changes in MeHg concentrations over time were critical to understand the nature and duration of exposure by aquatic biota over and above baseline concentrations to estimate exposure to MeHg in a post-inundation situation. Predicted increases in water column MeHg concentrations declined with distance from the reservoir, due to the effects of dilution, photodegradation and settling. The creation of Muskrat Falls Reservoir was predicted to increase MeHg concentrations in the top 20 m of the water column by 0.019 ng/L in Goose Bay (maximum 3-year average increase in concentration), and 0.005-0.006 ng/L in Lake Melville. MeHg concentrations were predicted to increase less at depths below 20 m: by up to 0.013 ng/L in Goose Bay, and 0.002-0.003 ng/L in Lake Melville. In the top 20 m of the water column, the predicted relative increase over the baseline concentration of 0.017 MeHg was approximately 2x in Goose Bay and 1.3-1.4x in Lake Melville.

Outline

The memorandum includes the following sections:

1. Hydrodynamic Model Overview
2. Summary of Hydrodynamic Model Setup and Calibration
3. Summary of Hydrodynamic Model Results
4. Box Model Overview
5. Box Model Setup and Calibration
6. Box Model Results

Hydrodynamic Model Overview

Delft3D is a three-dimensional hydrodynamic model with wave, sediment transport (cohesive and non-cohesive) and water quality modules. Delft3D was developed by Delft Hydraulics in the Netherlands, and it is a non-commercial, open-source model, which is an important consideration for public agencies. Delft3D is widely considered to be one of the best available models for the prediction of flow and particle fate, particularly in estuarine conditions.

The three-dimensional version of Delft3D model uses a curvilinear grid system, which fits the shoreline boundary conditions in the Lake. Delft3D-FLOW is the hydrodynamic component of the Delft3D model suite, and it can be applied to a wide range of applications, including:

- Tide and wind-driven flow resulting from space and time-varying wind and atmospheric pressure
- Density driven flow and salinity intrusion
- Horizontal and vertical transport of matter on large and small scales
- Stratification in seas, lakes and reservoirs
- Small scale current patterns near harbor entrances

The primary purpose of Delft3D-FLOW is to solve various time-dependent, non-linear differential equations related to hydrostatic and non-hydrostatic free-surface flow problems on a structured orthogonal grid. The equations solved are mathematical descriptions of physical conservation laws for:

- Water volume (continuity equation)
- Linear momentum (Reynolds-Averaged Navier-Stokes (RANS) equations)
- Tracer mass (transport equation), e.g., for salt, heat (temperature) and suspended sediments or passive pollutants

Several different datasets are necessary to set up the hydrodynamic model:

- Coastline and bathymetric data for the Lake and the adjacent areas
- High resolution aerial imagery – used to delineate the lake boundary
- Tide elevations and river flow data in the study area
- Temperature and salinity conditions at the model boundaries
- Amount of excess MeHg entering the Lake system

These datasets are discussed in the subsequent sections of this memorandum.

Summary of Hydrodynamic Model Setup and Calibration

Where available, empirical, site-specific information as possible was used (e.g., river discharge, bathymetry, shoreline data, etc) to construct the Delft3D model, so that it represents the site-specific conditions in the study area as much as possible.

Coastline Data

Coastline data are necessary to define the Delft-3D model domain. The coastline in the study area was digitized from Google Earth and converted for import to the Delft3D model.

Model Grid

The model computational grid was set up to achieve a balance between having sufficient resolution in the narrow sections of the domain, while avoiding over-refining the grid, which would result in an unacceptably-long computation time for the model runs. Overall, there were 18,500 active grid cells in the horizontal plane, and up to 17 layers deep in the vertical direction. Due to limited data for the lower Churchill River, a simplified representation of the river was developed upstream from Goose Bay. This stretch of river was used to define the main inflow to Goose Bay and it was outside the main area of interest, so this simplification was appropriate for determination of flows within the lake itself. The grid cell resolution in the lower Churchill River was 9 cells across the width of the river (each cell was ~100-300 m wide), and the cells were oriented in the dominant downstream flow direction. The river portion of the domain extended approximately 21 km upstream from the entrance to Goose Bay. The grid cell size in the Narrows was approximately 250 m x 250 m, with a minimum of 4 cells across the Narrows, and the average grid cell size in Lake Melville was 750m x 750m. Figure 1 shows the final model grid.

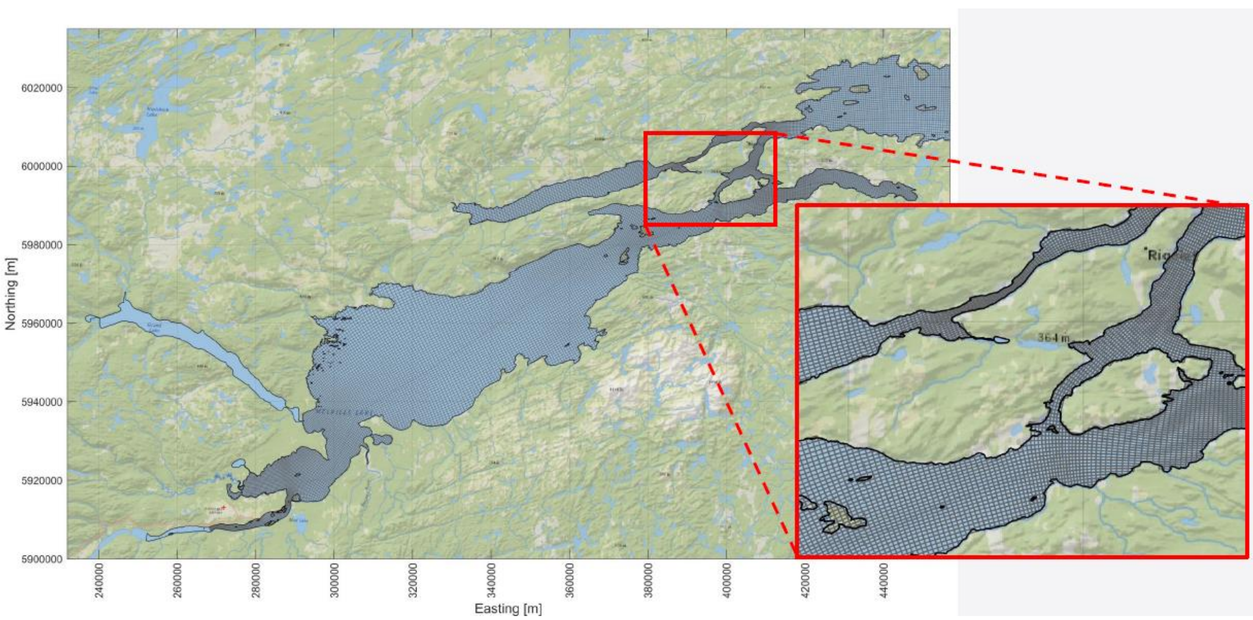


Figure 1 Hydrodynamic Model Grid

Hydrodynamic Model Bathymetry

Model bathymetry was derived from Canadian Hydrographic Survey charts for Lake Melville and then adjusted to mean sea level in the area. Figure 2 shows the final model bathymetry.

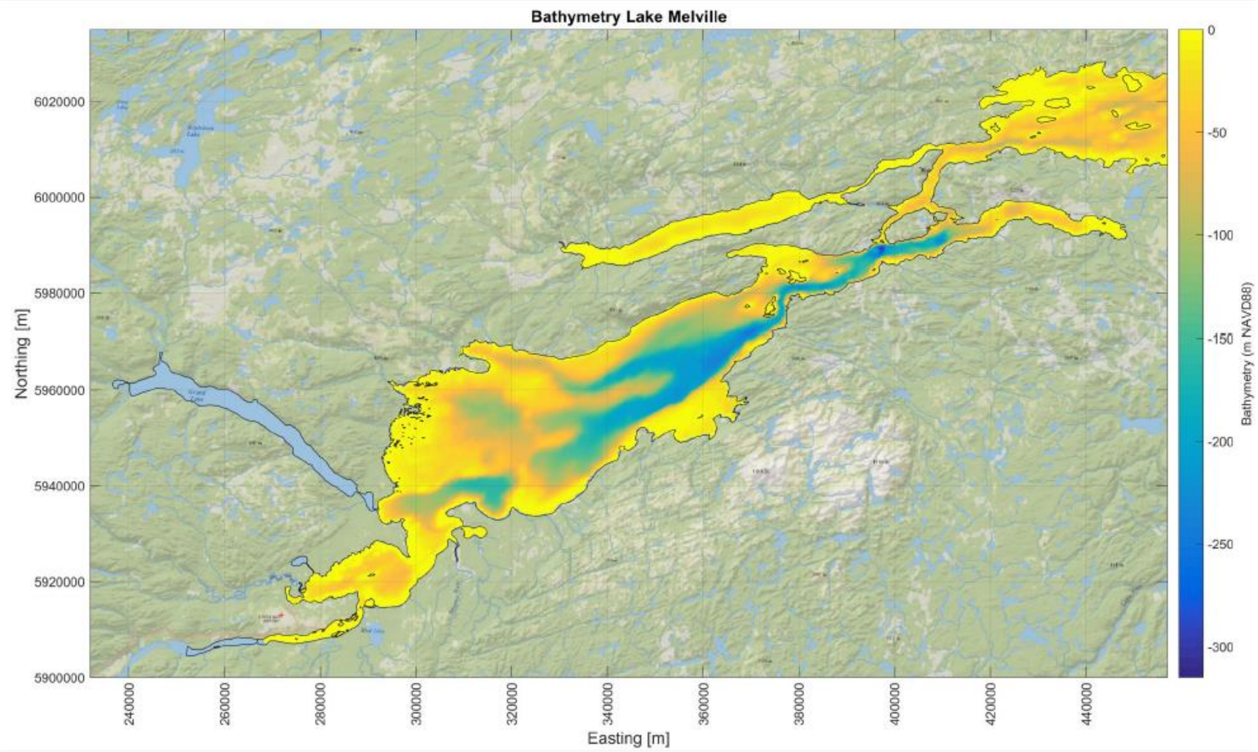


Figure 2 Hydrodynamic Model Bathymetry

Model Atmospheric Conditions

Atmospheric data were downloaded from the Government of Canada Historical Climate Data website[1] for the station at Goose Airport, Goose Bay. This is an hourly dataset, including: temperature (Celsius), relative humidity (%), wind direction (10's deg), wind speed (km/h), visibility (km), atmospheric pressure (kPa) and Cloudiness (%). The data were available for 2010-2017. Gaps in the wind speed and direction data were replaced with zero values.

[1] http://climate.weather.gc.ca/climate_data/

Model Boundary Conditions

Ocean Water level

The water level at the ocean boundary used the astronomic constituents supplied by Wood PLC.[2] Baird compared the water level time series in the model to predicted levels by DFO and determined that the Wood constituents were adequate for use in the model study.

Ocean Temperature and Salinity

Baird retrieved temperature and salinity data for transects at Seal Island, NL, from the DFO Marine Environmental Data Section (MEDS) Portal.[3] A constant temperature of 1 degree Celsius has been used for the Ocean Boundary. This is a reasonable assumption based on data retrieved for Seal Island. Salinity was set to 32 ppt, decreasing to 24 ppt in the top 5 layers. This was based on the data retrieved from Seal Island and the salinity measurements at M2 retrieved from a Memorial University Report.[4]

Lower Churchill River Discharge

Discharge data for the Lower Churchill River were downloaded from the Environment Canada historical hydrometric data portal[5] for station 03OE001 (Churchill River Above Upper Muskrat Falls) for the period 2010-2015. The gauge is located approximately 45 km upstream from where the lower Churchill River enters Goose Bay. This is approximately 20 km upstream from the model boundary in the lower Churchill River. The model assumed no additional inflows in this section and the flows from Muskrat Falls were transposed to the Delft3D upstream boundary. Water temperatures in the lower Churchill River were calculated based on available air temperature data ($T_{water} = 5.0 + 0.75*T_{air}$).

Watershed Inflows

The inflows for rivers other than the lower Churchill River account for approximately 20% of freshwater inflow to Lake Melville, and they are ungauged. The freshwater inflow discharges from the tributary watersheds to Goose Bay and Lake Melville were calculated using an area-weighted method (compared to the lower Churchill River watershed). The watersheds were delineated from the digital terrain model of the area, including a 'burn-in' of the streamlines from the NHD streamline dataset (Figure 3). Water temperatures in the watershed inflows were calculated based on the same air temperature relationship as for the lower Churchill River ($T_{water} = 5.0 + 0.75*T_{air}$). Except for the lower Churchill River, which was modelled explicitly, point sources have been used to represent the watershed inflows to the hydrodynamic model.

[2] Wood PLC, 10 April 2018, pers. comm.
[3] http://www.isdm.gc.ca/isdm-gdsi/azmp-pmza/hydro/index-eng.html
[4] Lu, Z., DeYoung, B. and Banton, S. 2014. Analysis of Physical Oceanographic Data from Lake Melville, Labrador, September 2012 - July 2013. Memorial University Physics and Physical Oceanography Data Report 2014, I.
[5] https://wateroffice.ec.gc.ca/download/index_e.html?results_type=historical

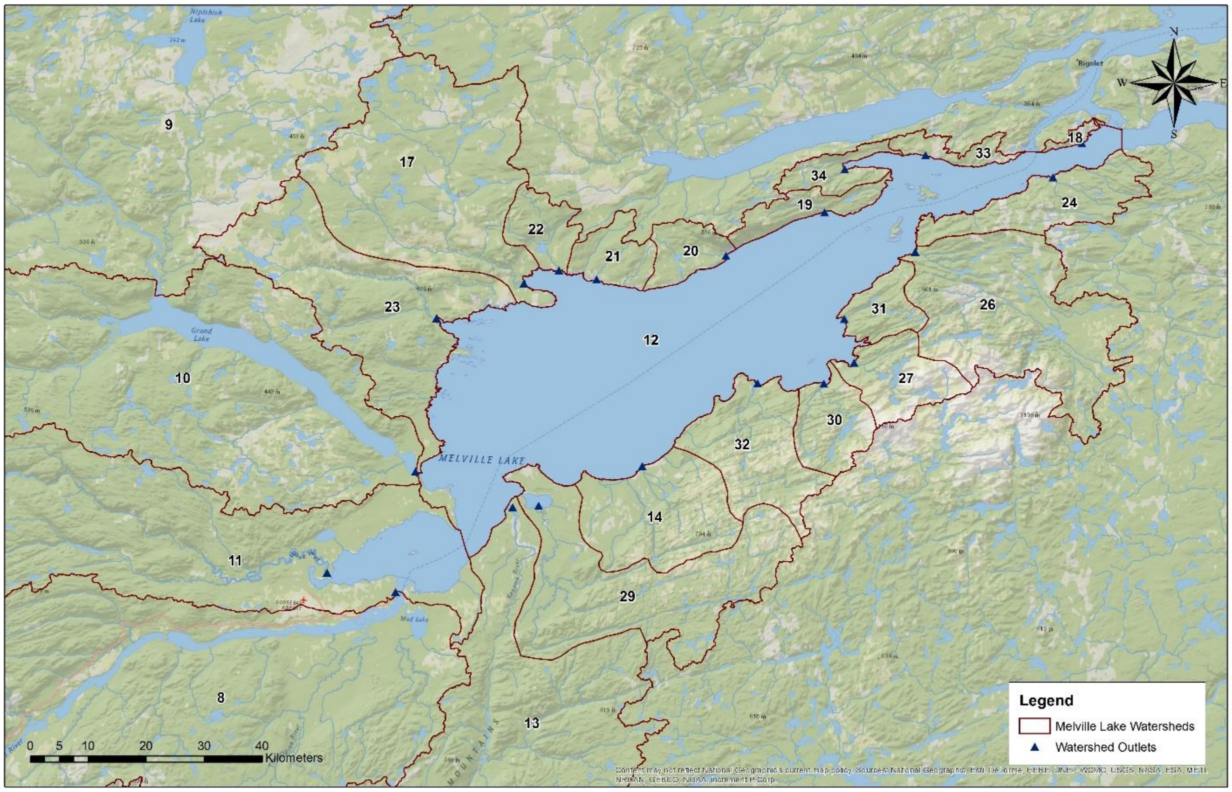


Figure 3 Watersheds discharging to Goose Bay and Lake Melville

Excess MeHg Load from Muskrat Falls Reservoir

Here we define excess MeHg as the increase in load or concentration above baseline, associated with flooding in Muskrat Falls Reservoir. The overall load or concentration is the baseline plus the increase.

Predictions of excess MeHg concentrations and loads exported from Muskrat Falls Reservoir were provided by Reed Harris Environmental Ltd. to Baird (Figure 5). Two sets of excess concentration predictions were provided: one from the output of the ResMerc model, and one based on empirical data from the Experimental Lakes Area FLUDEX experiments (see Harris and Hutchinson, 2018). Both estimates spanned the first 5 years after flooding, a period during which the reservoir MeHg loads and predicted downstream concentrations in water both peaked and began to decline. These predictions were initially applied as a conservative tracer in the Delft3D model. Subsequent simulations using a box model also considered the effects of photodegradation and settling.

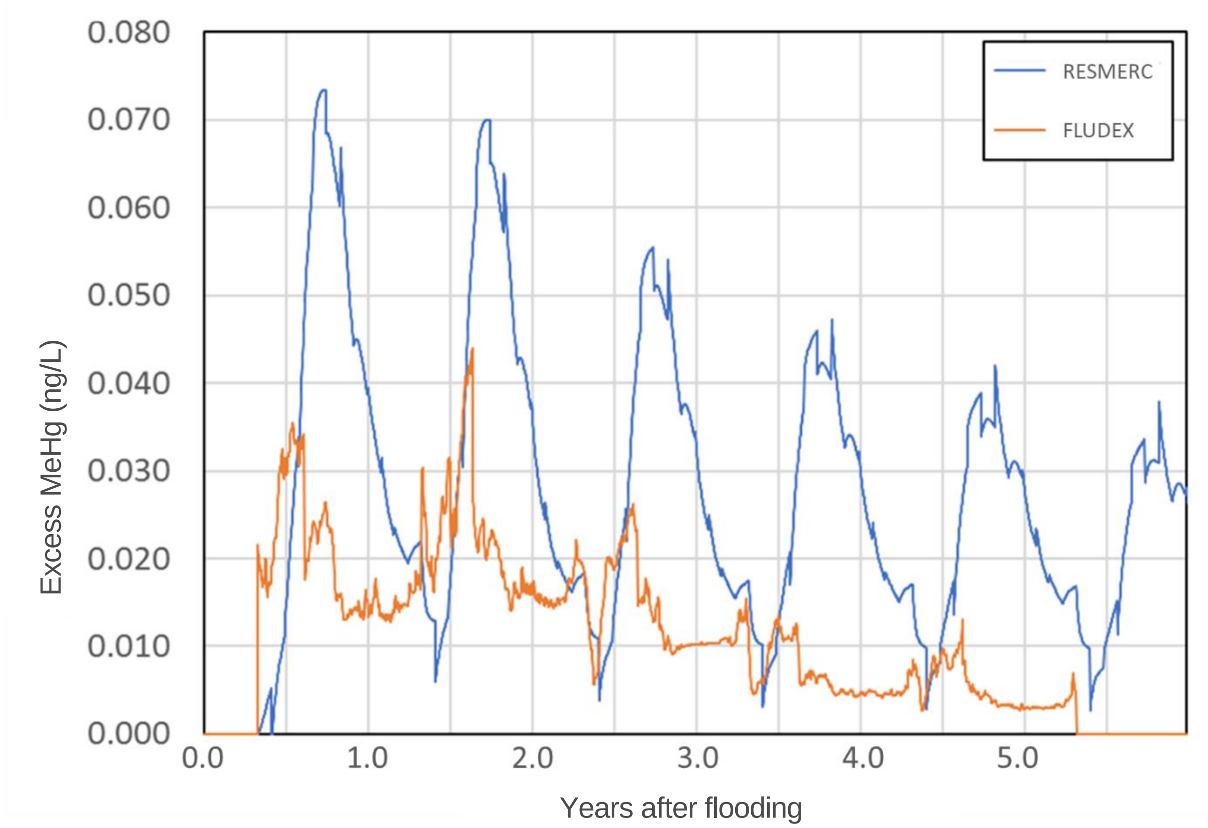


Figure 4 Excess MeHg inflow concentrations (above baseline) based on loads from the ResMerc simulations and the FLUDEX experiments (Harris and Hutchinson, 2018), and flows in the lower Churchill River. Note: Flooding begins in May of first year.

Hydrodynamic Model Calibration

Temperature, Salinity and Density

Comparisons between digitized measurements presented in the Memorial University report for the period in August-October 2012 and model results were made. Overall, modelled temperature, salinity and density profiles showed the same trend as the measurements with respect to the epilimnion, thermocline and the hypolimnion at the Memorial Sonde (conductivity, temperature and pressure (depth) also known as 'CTD probe') locations (Figure 5). Figure 6 to Figure 8 show examples of measured-modelled comparisons on individual days. A good comparison in the upper layers for all 3 parameters is observed. In the deeper layers, temperature is slightly overpredicted, whereas salinity and density are slightly underpredicted.

Model performance statistics are summarized in Table 1 for the final calibrated model. The performance statistics show:

- Water temperature predictions were strongly correlated[6] with observations, and model skill[7] for temperature was high. Modelled values were generally within 3 ^{o}C of observed values, with modelled temperature being warmer than observed.
- Water salinity predictions were strongly correlated with observations, and model skill for salinity was high. Modelled values were generally within 3 ppt of observed values, with modelled salinity being slightly lower than observed.
- Water density predictions were strongly correlated with observations, and model skill for density was high. Modelled values were generally within 2 kg/m^3 of observed values, with modelled density being slightly lower than observed.

The model performance is considered appropriate for determining the distribution and changes in the above parameters. Further calibration of the model may improve the match between measured and predicted parameters, although additional field data would be required to undertake this.

[6] Quantified using the Pearson product-moment coefficient, R

[7] The skill score is an index of agreement between measured and modelled values (Willmott, 1981). A skill score >0.8 is considered excellent.

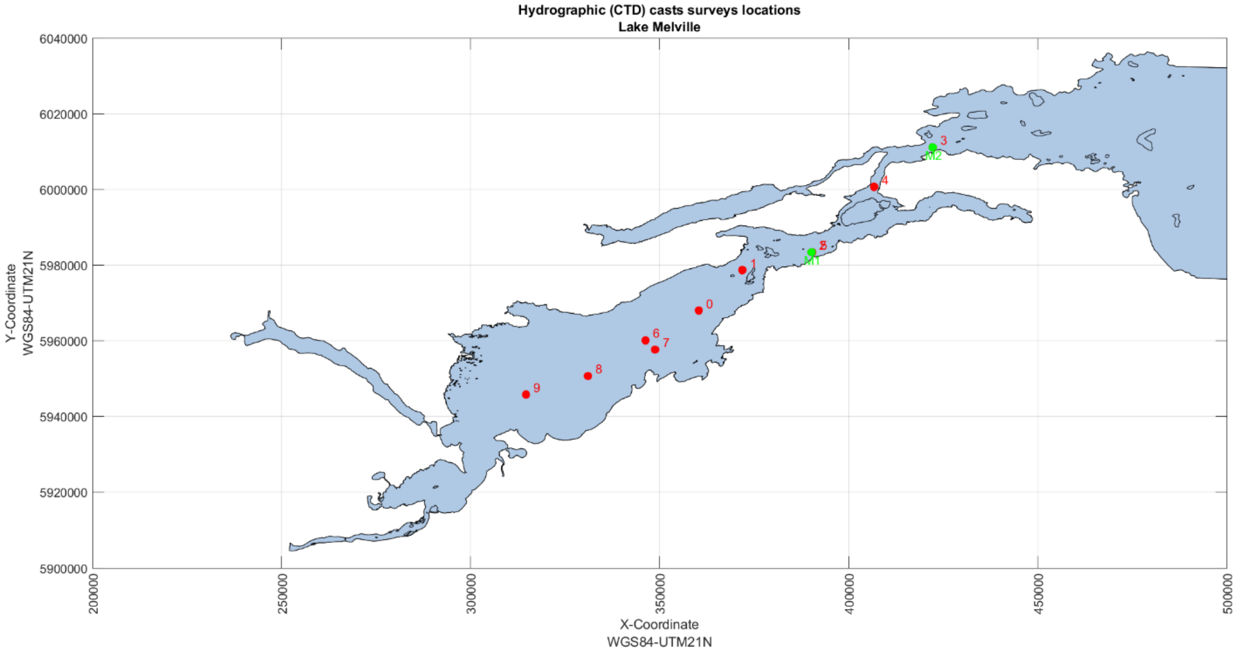


Figure 5 Locations of CTD cast measurements from the Memorial University report. Red dots show the location of CTD probe profiles and Green dots (M1, M2) show the location of Acoustic Doppler Current Profiler (ACDP) deployments

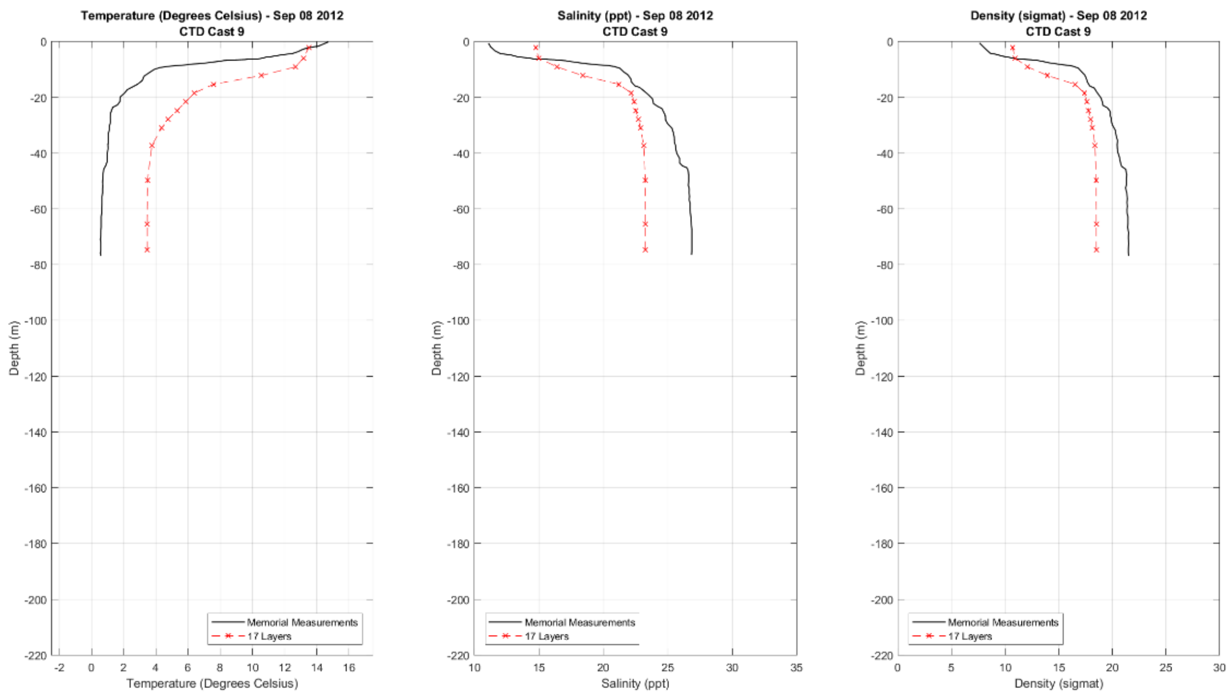


Figure 6 Comparison of measured-modelled temperature, salinity and density in west Lake Melville (CTD9)

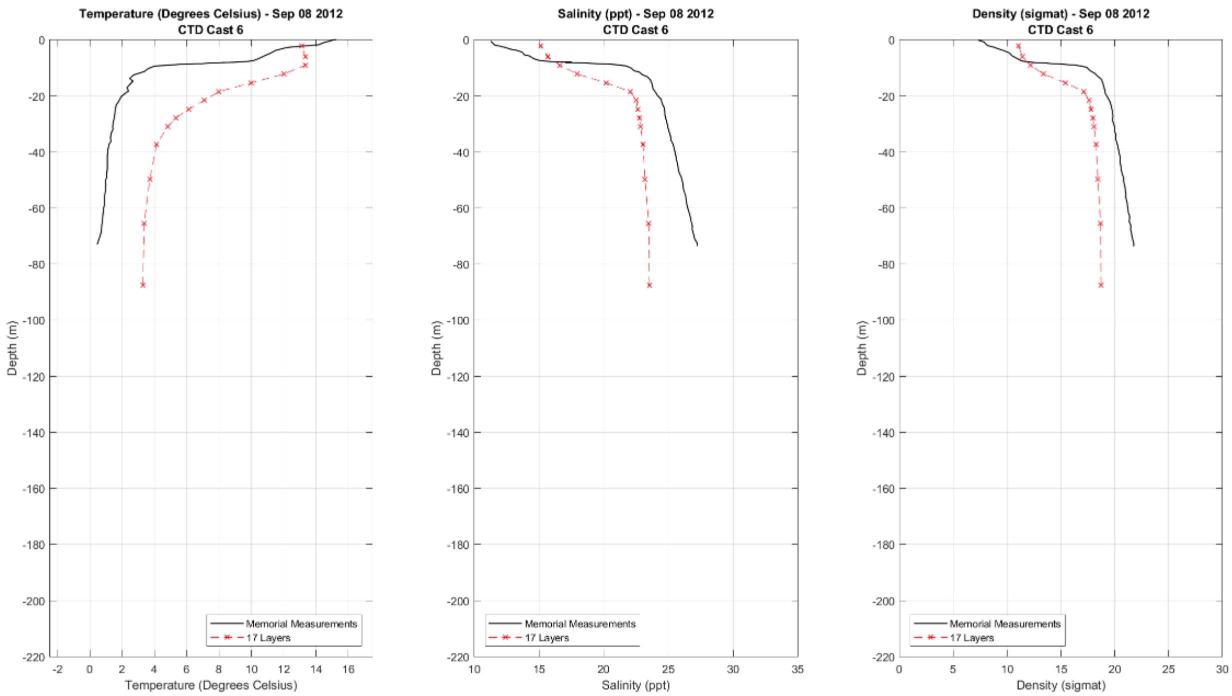


Figure 7 Comparison of measured-modelled temperature, salinity and density in central Lake Melville (CTD6)

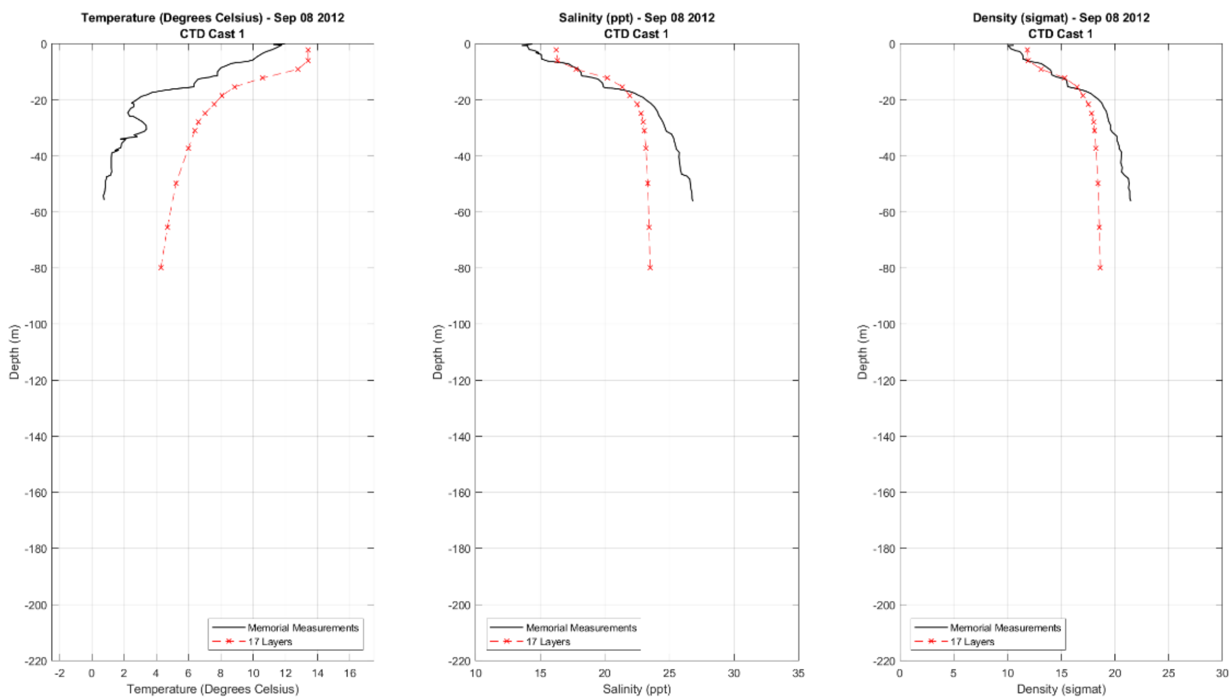


Figure 8 Comparison of measured-modelled temperature, salinity and density in east Lake Melville (CTD1)

Table 1 Model performance statistics for temperature, salinity and density (Means of each parameter)

Temperature (ºC)			
Bias	RMSE	Correlation	Skill
2.03	2.48	0.95	0.87

Salinity (ppt)			
Bias	RMSE	Correlation	Skill
2.29	2.60	0.96	0.85

Density (kg/m³)			
Bias	RMSE	Correlation	Skill
1.67	1.92	0.96	0.89

Comparisons of measured and predicted mean velocity profiles are shown in Figure 9 and Figure 10 for Memorial ADCP locations M1 and M2, respectively. Overall, a reasonable agreement between the profiles is observed, although the deeper layers at site M2 show more bias towards tidal inflow at depth in the Narrows, which could be addressed with for more detailed offshore tide and current information but is not critical to support the conclusions of the analysis. Table 2 shows the summary statistics for velocity. Although the bias and RMSE values are low (which is desirable), correlation between measured and modelled velocities is lower than anticipated due to the lack of available local wind and tide data at the site. Model skill is fair for this parameter.

Table 2 Model performance statistics for flow velocity (Means of each parameter)

Velocity (m/s) u Direction			
Bias	RMSE	Correlation	Skill
-0.01	0.28	0.28	0.52

Velocity (m/s) v Direction			
Bias	RMSE	Correlation	Skill
0.00	0.16	0.19	0.46

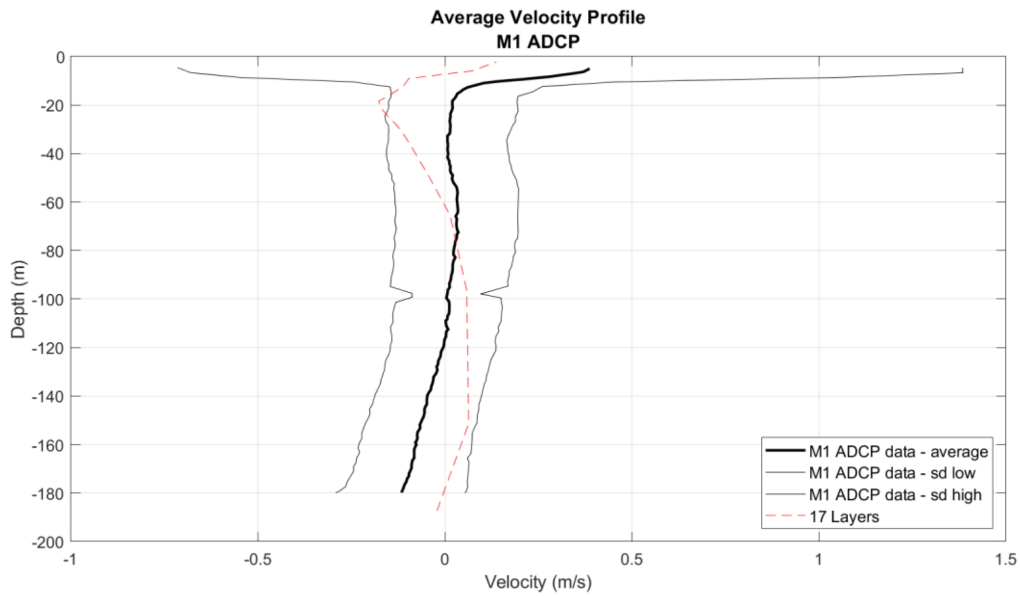


Figure 9 Comparison of measured and modelled mean velocity profiles at ADCP location M1. '17 Layers' = model predictions

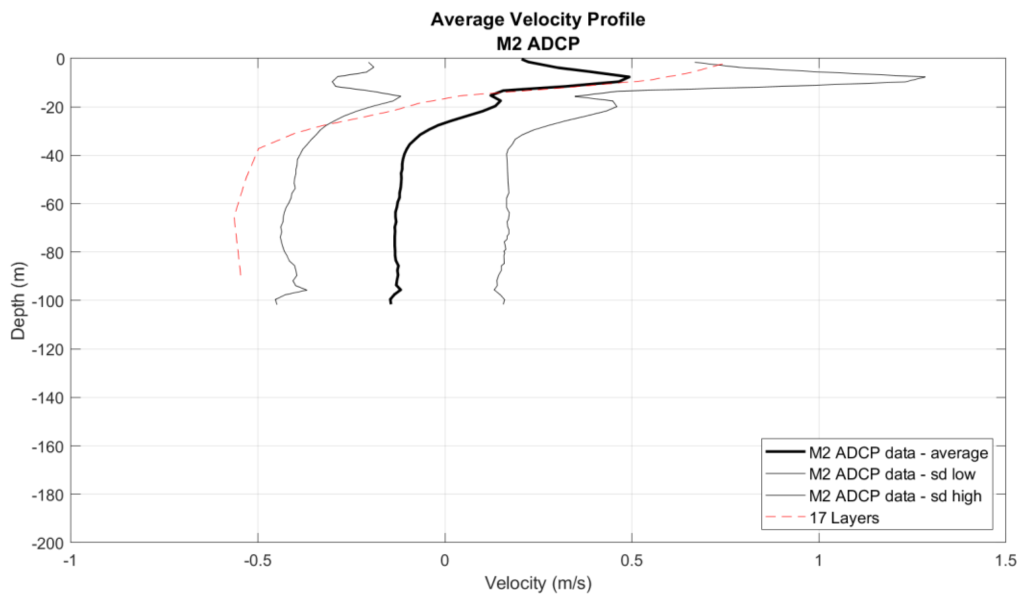


Figure 10 Comparison of measured and modelled mean velocity profiles at ADCP location M2. '17 Layers' = model predictions

Box Model Setup and Results

It was not possible to account for photodegradation and settling losses directly in the high resolution hydrodynamic model. Accordingly, a box model was set up to account for these losses. The box model used flow predictions from the hydrodynamic model to calculate the flux of water between Goose Bay, and the east and west parts of Lake Melville (Figure 11). The box model also considered the flux between vertical layers with depths of 0-3 m, 3-10 m, 10-20 m and >20 m. A schematic of the exchanges in the box model is shown in Figure 12.

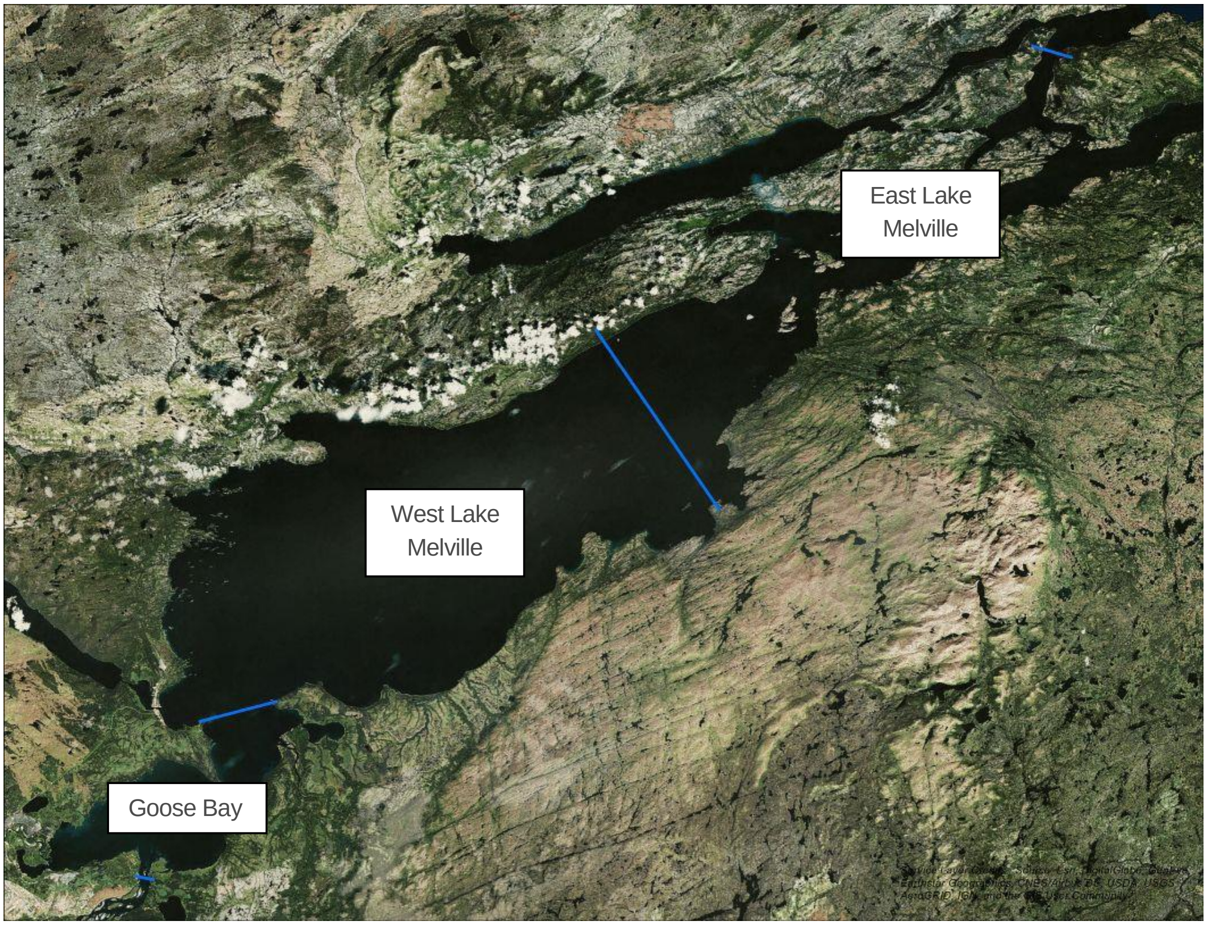


Figure 11 Areas considered in box model

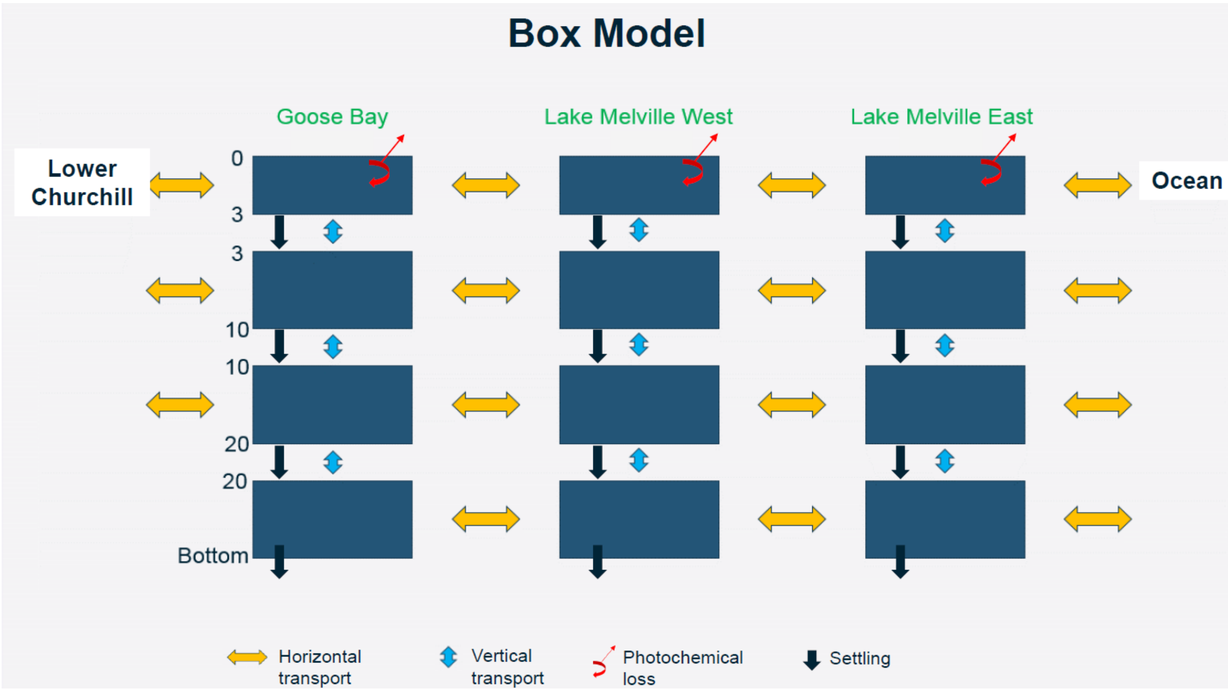


Figure 12 Schematic of box model configuration

Prior to simulating the effects of photodegradation and settling, the box model was calibrated with the goal of matching results of the simulation with the high-resolution model where MeHg was treated as a conservative (non-reactive) substance. Figure 13 shows the comparison of the two models for one of the twelve segments of the box model . Overall, the results in each segment of the box model matched well with the conservative simulation with the hydrodynamic model. The conservative models estimated a water residence time of approximately 11 days in Goose Bay and 125 days in Lake Melville.

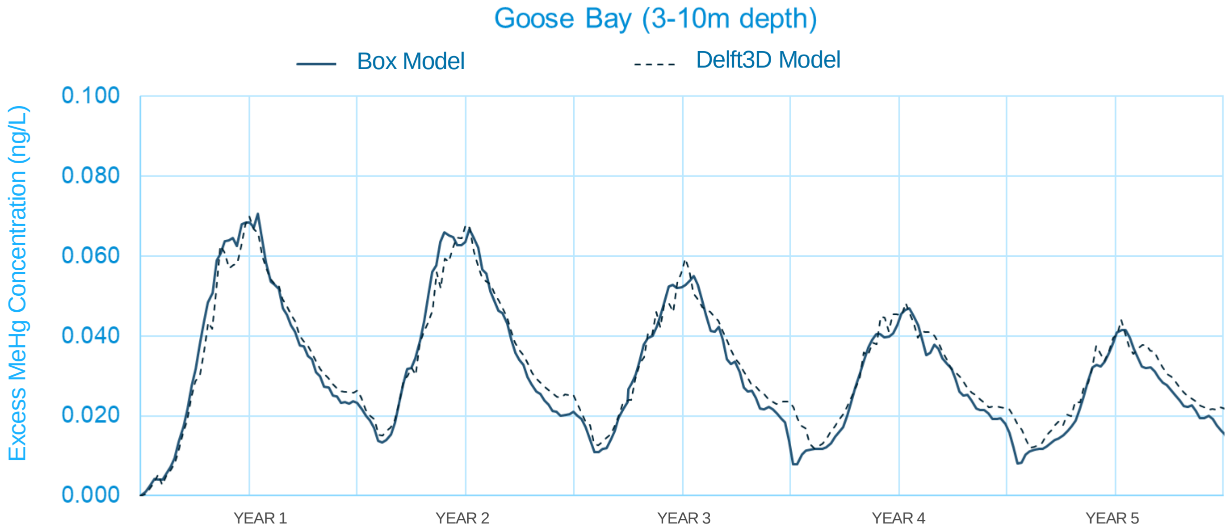


Figure 13 Comparison of Box Model and high resolution hydrodynamic model results for conservative simulation of excess MeHg in water.

Once the conservative mixing and dilution results from the high-resolution model were reasonably represented in the box model, losses due to photodegradation and settling were applied. Photodegradation rate constants (day^{-1}) were provided by Reed Harris using an analysis by Pollman (2018)[8] and were developed as follows:

- Photodegradation included components associated with UVA, UVB and PAR. Each component had a rate constant at the water surface, expressed as the inverse of incident radiation ($m^2\ E^{-1}$). When multiplied by incident radiation rates ($E\ m^{-2}\ day^{-1}$) the result had units of day^{-1}.
- This rate constants for each wavelength were multiplied by the dissolved MeHg concentration to estimate the rate of loss of MeHg in surface waters ($ng\ L^{-1}\ day^{-1}$). The dissolved concentration was assumed to be 70 percent of the unfiltered concentration, based on surface water sampling from 3 stations in Lake Melville from October 2016 – December 2017 .
- Photodegradation occurred during the ice free-season, assumed to occur for Julian days 152-305. Photodegradation was set to zero during the ice cover season.
- Radiation energy decreased with water depth. The extinction coefficients for UVA, UVB and PAR with depth resulted in 95% attenuation at a depth of 1 m for UVA, 0.5 m for UVB and ~3m for PAR.
- Two sets of photodegradation rate constants were used to bracket a range. The first set was based on Black et al (2012)[9] and the second set was based on Lehnherr and St. Louis (2009)[10].
- The average of model results for the two estimates of photodegradation losses was used in the final analysis.

[8] Pollman (2018) Unpublished review of methylmercury photodegradation in surface waters and approaches to modeling the reaction.
[9] Black, F.J., B.A. Poulin, and A.R. Flegal. 2012. Factors controlling the abiotic photo-degradation of monomethylmercury in surface waters. Geochim. Cosmochim. Acta 84: 492-507.
[10] Lehnherr, I. and V.L. St. Louis. 2009. Importance of ultraviolet radiation in the photodemethylation of methylmercury in freshwater ecosystems. Environ. Sci. Technol. 43: 5692-5698

Overall, the photodegradation loss estimate based on Black et al. (2012) ranged from 2-9 percent of the bulk phase per day, integrated over the top 3 m of the water column, during the ice-free season (average = 6.4% per day). The analogous range based on Lehnherr and St. Louis (2009) was approximately 0.5 to 3 percent per day in the ice-free season (average ~2% per day).

Figure 14 shows the effects of photodecomposition in one segment of the box model, using the rates developed using the Black et al. photodegradation constants. The dashed line shows results from the conservative simulation, and the solid line shows predicted concentrations after photodegradation losses. Photodegradation effects were more evident during the ice-free months, while levels returned towards the conservative condition during the months with ice cover as MeHg was replenished in the system. The response of lower depths in the lake was more muted as photodecomposition did not act directly on these layers, and the only effect on concentrations at depth was due to a reduced mass flux from the surface layer of the lake.

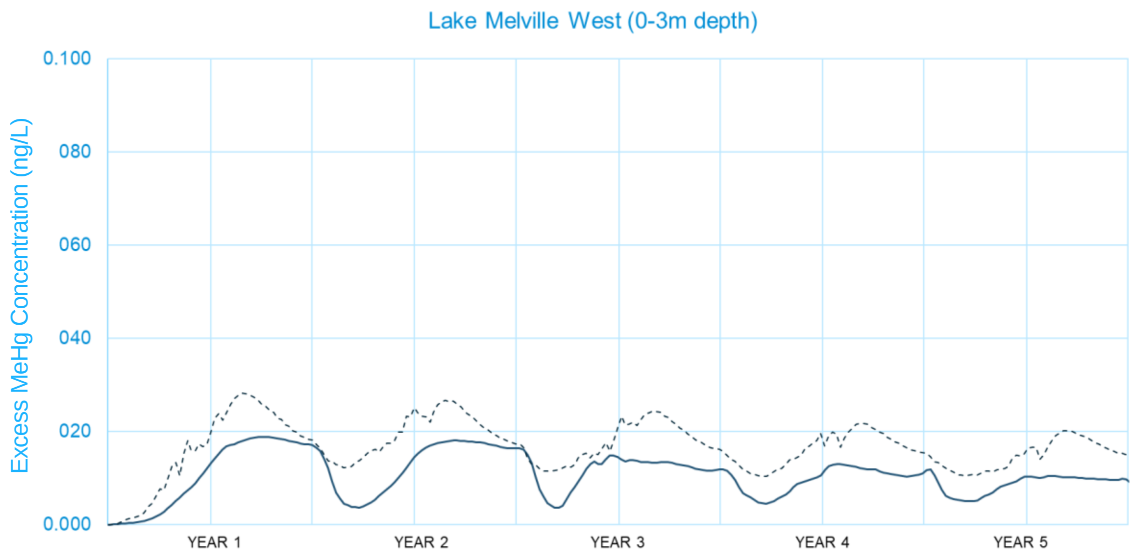


Figure 14 Effect of photodecomposition on predicted excess MeHg concentrations. Dashed line = conservative run; Solid line = with photodegradation. Results based on simulation using photodegradation constants from Black *et al*., (2012)

The effects of settling were also included in the box model. Based on the estimate that 70% of the unfiltered MeHg was dissolved, settling was applied to 30% of the water column MeHg mass in each segment of the box model. Three settling velocities (0.1 m/day, 0.5 m/day and 1.0 m/day) were tested in the box model. Each of these settling velocities is relatively conservative (meaning that higher MeHg concentrations will be predicted), representing the characteristics of the particulate organic carbon and fine clays upon which MeHg typically adsorbs more strongly. A full discussion of settling characteristics and the rationale for selection of settling velocity is beyond the scope of this memorandum, however the range of velocities used herein is considered conservative (i.e. it tends to under-estimate the amount of settling). A value of 0.5 m/day was used in the final analysis.

The effects of settling on predicted MeHg concentrations in surface waters are shown in Figure 15, for the box model compartment representing the top 3 m of Lake Melville West. The orange line shows the effects of settling and photodegradation. The fine dashed line shows photodegradation only, and the dashed line shows the conservative simulation.

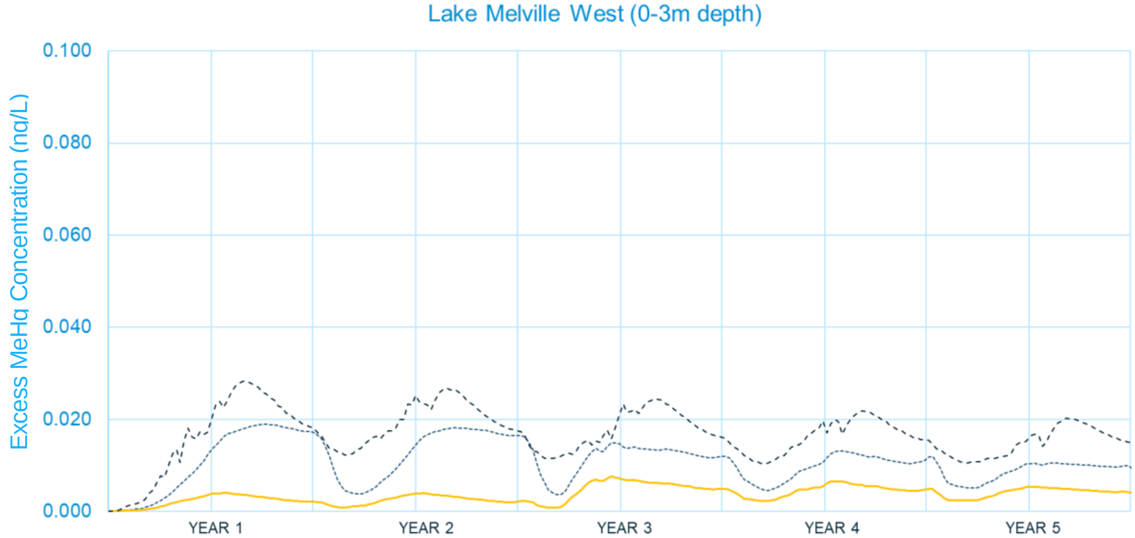


Figure 15 Effects of settling and photodegradation on excess MeHg concentrations in the box model, for the top 3 m of Lake Melville West. Upper dashed line = conservative run; fine dashed line = with photodegradation; orange line = with photodegradation and settling

The results from the overall model analysis are presented in Table 3 to Table 8. These tables summarise the results averaged for simulations using different loading scenarios (based on ResMerc and FLUDEX) and using the two estimates of photodegradation rate constants, along with a 0.5 m/day settling velocity for particulate MeHg. Excess concentrations in the top 20 m were 0.019 ng/L in Goose Bay (maximum 3-year average concentration), and 0.005-0.006 ng/L in Lake Melville (Table 3). Excess concentrations in the hypolimnion were lower, at 0.013 ng/L in Goose Bay, and 0.002-0.003 ng/L in Lake Melville (Table 4). Higher concentrations are to be expected in Goose Bay as it is a smaller waterbody, closer to the source in the lower Churchill River than Lake Melville. The increase in mass of MeHg in each area varied from 0.03-0.21 kg (Table 5 and Table 6). The relative increase over the baseline concentrations of MeHg estimated by Calder et al. (2016) was 1.9-2.1x in Goose Bay and 1.3 -1.4x in Lake Melville (Table 7 and Table 8). These values have been carried forward in the analysis by Wood (2018) to examine the relative degree of exposure by key species to elevated MeHg concentrations in Goose Bay and Lake Melville over time.

Table 3 Excess bulk MeHg water concentrations in the epilimnion (0-20 m)

Location	Excess MeHg Concentration: 3 Year average (ng/L, max)
Goose Bay	0.019
Melville West	0.006
Melville East	0.005

Table 4 Excess bulk MeHg water concentrations in the hypolimnion (Below 20 m)

Location	Excess MeHg Concentration 3 Year average (ng/L, max)
Goose Bay	0.013
Melville West	0.002
Melville East	0.003

Table 5 Increase in MeHg mass over baseline conditions in the epilimnion (0-20 m)

Location	Increase Over Baseline (Year 1-3 Average) (kg)
Goose Bay	0.06
Melville West	0.19
Melville East	0.06

Table 6 Increase in MeHg mass over baseline conditions in the hypolimnion (below 20 m)

Location	Increase Over Baseline (Year 1-3 Average) (kg)
Goose Bay	0.03
Melville West	0.21
Melville East	0.17

Table 7 Relative increase over baseline concentrations in the epilimnion (0-20 m). Baseline concentration: 0.017 ng/L for 0-20 m (from Calder et al., 2016).

Location	Peak/ Baseline
Goose Bay	2.1
Melville West	1.4
Melville East	1.3

Table 8 Relative increase over baseline concentrations (supplied by R. Baker) in the hypolimnion (below 20 m). Baseline concentrations below 20 m depth: 0.015 ng/L (Goose Bay), 0.007 ng/L (Lake Melville).

Location	Peak/Baseline
Goose Bay	1.9
Melville West	1.3
Melville East	1.4

August 3, 2018

Peter Madden
Regulatory Compliance Lead
Project Delivery Team
Lower Churchill Project

re: Updated analysis of predicted increases in methylmercury concentrations and downstream export from Muskrat Falls Reservoir

1 Introduction

In 2017-2018, Nalcor retained Reed Harris Environmental Ltd. and others to update or extend previous studies to predict increases in methylmercury concentrations in water and biota in Muskrat Falls Reservoir and downstream in Goose Bay and Lake Melville. An important component of the analysis was the use of field data collected since 2011, both within the reservoir area and downstream. This technical memorandum describes:

1. updated predictions of increases in methylmercury concentrations in Muskrat Falls Reservoir waters and fish; and
2. predicted methylmercury concentrations and masses exported from Muskrat Falls Reservoir.

Results of this study were used by Baird & Associates (Brunton, 2018) in a model analysis to predict increases in methylmercury concentrations in Goose Bay and Lake Melville waters. The Baird results were then used by Wood (2018) to estimate increases in methylmercury concentrations in biota in Goose Bay and Lake Melville. Finally, predicted increases in methylmercury levels in biota were used by Azimuth (2018) to develop an interim update to the human health risk assessment by Dillon (2016).

2 Methods

The primary goal of this study was to estimate increases in methylmercury concentrations and loads delivered downstream following the creation of Muskrat Falls Reservoir. Two approaches were used (Figure 1). The first approach was based on field data from an experimental reservoir study in Ontario called FLUDEX. This experiment investigated mercury and greenhouse gases intensively in newly flooded uplands and advanced our understanding of mercury in reservoirs (*e.g*. Bodaly *et al.*, 2004, Hall *et al.*, 2005). The second approach was based on a mechanistic model that predicted methylmercury concentrations in water, sediments and biota in the reservoir over time, as well as downstream export rates. The remainder of this section of the document provides additional information on each approach.

While downstream methylmercury export was the focus of the analysis, fish mercury concentrations within Muskrat Falls Reservoir were also predicted and are presented here. Predicted fish mercury concentrations in the reservoir were important in their own right and provided a line of evidence to help assess confidence in concentrations of methylmercury in water predicted by the mechanistic model.

This is because there are long-term data available for fish mercury concentrations from existing reservoirs that can be compared to model predictions, but no analogous data exist for methylmercury concentrations in water over time from full scale reservoirs, to compare directly with models.

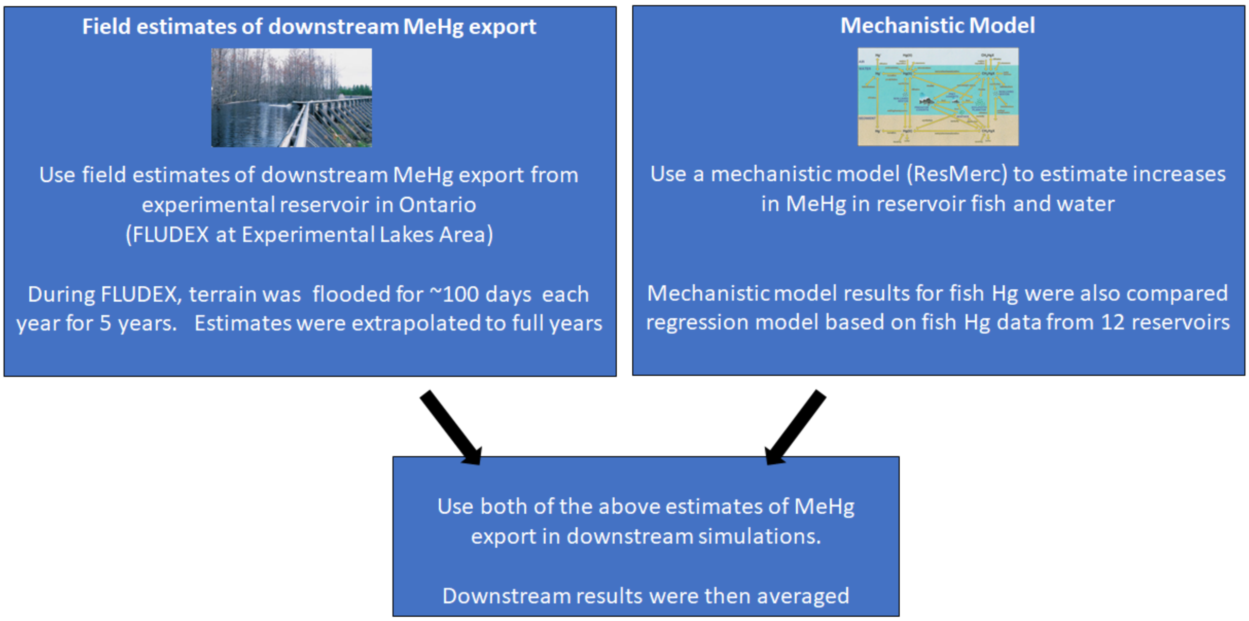


Figure 1. Approach used to estimate downstream export of methylmercury from Muskrat Falls Reservoir

2.1 Approaches to predict increases in fish mercury concentrations in Muskrat Falls Reservoir

Mechanistic and regression models were used to predict increases in fish mercury concentrations in Muskrat Falls Reservoir. Both approaches were previously applied to Muskrat Falls Reservoir in support of the Environmental Assessment (Nalcor, 2009a; Harris *et al.*, 2010).

Mechanistic model description

ResMerc is a process-based simulation model for reservoirs and lakes, originally developed as part of FLUDEX and a companion flooded wetland experiment called ELARP (Harris *et al.*, 2009, Harris and Hutchinson, 2009). In addition to being applied previously to Muskrat Falls Reservoir to support the Environmental Assessment, ResMerc was used for the Site C project in British Columbia (Harris *et al.*, 2012). 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 2). Fish mercury concentrations tend to increase with age and are therefore followed in each year class (up to 20 cohorts). The model predicts concentrations, mercury pools and major fluxes for each mercury form through time.

ResMerc mercury processes include atmospheric deposition, inflows and outflows (surface and groundwater), adsorption/desorption, particulate settling, particle decomposition at the sediment/water interface and within sediments, resuspension, burial, air/water gaseous exchange, industrial point sources, in-situ transformations (*e.g.* methylation, demethylation, methylmercury photodegradation, Hg(II) reduction and Hg(0) oxidation), methylmercury uptake kinetics in plankton and partitioning in benthos, and methylmercury bioaccumulation in fish.

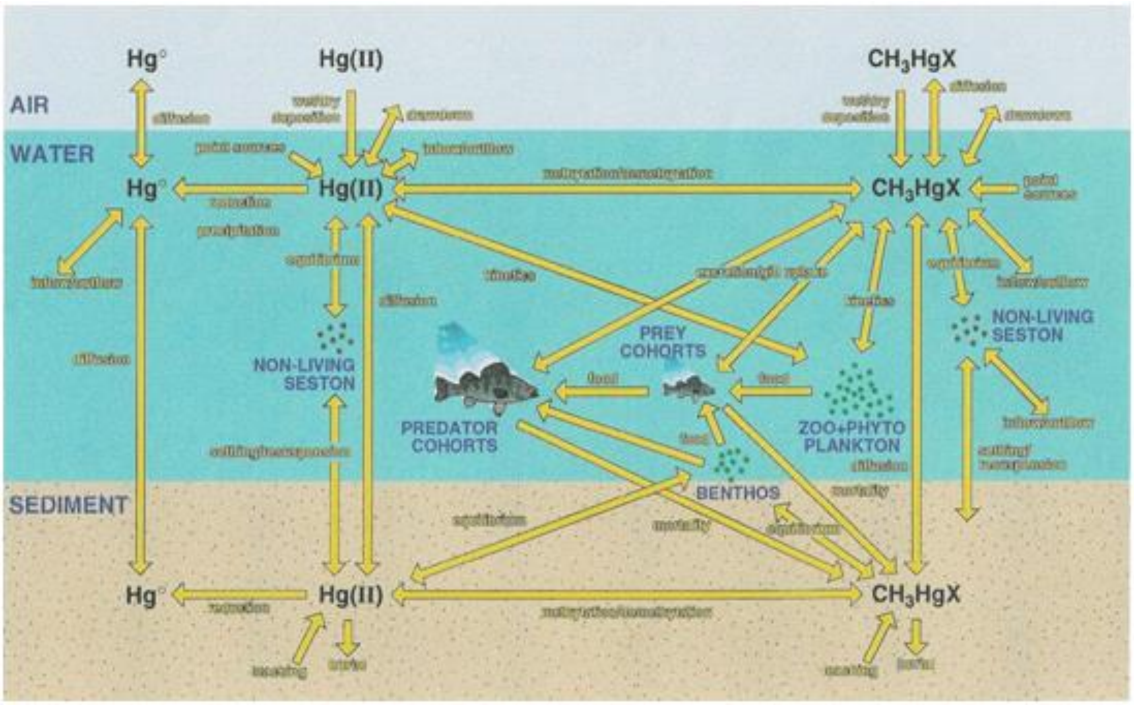


Figure 2. Representation of mercury cycling and bioaccumulation in ResMerc

Methylmercury concentrations in fish are predicted using a bioenergetics approach described by Harris and Bodaly (1998). Methylmercury 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 mercury concentrations in natural lakes, one process takes on special importance in new reservoirs: decomposition. Flooding stimulates decomposition and more activity by microbes that convert inorganic mercury into methylmercury. 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 lake, flooded uplands and flooded wetlands. Each sediment zone has two vertical sediments layers with thicknesses defined by the user. Sediments below the 2nd layer are treated as a boundary condition. Each sediment layer has its own initial conditions, characteristics and inputs. Additional information on ResMerc is available in the model user guide (Harris and Hutchinson 2009) and a report describing the model development (Harris *et al.*, 2009).

The steps involved in the application of ResMerc to Muskrat Falls Reservoir were as follows:

1. The model was calibrated to Robert Bourassa Reservoir in Quebec to estimate the methylmercury loads required from flood zones to support observed increases in fish mercury concentrations. Robert Bourassa reservoir had some of the highest reported mercury concentrations in Canadian Reservoirs, exceeding 3 µg/g in 700 mm northern pike (Schetagne *et al.*, 2003). In lieu of having information characterizing the flood zone at Robert Bourassa Reservoir, it was assumed that the flood zone conditions were the same as estimated during a field survey of Muskrat Falls Reservoir (AMEC Foster Wheeler, 2017).
2. The model calibration from Robert Bourassa Reservoir was applied to Notigi Reservoir, Manitoba, comparing predicted and observed fish mercury concentrations.
3. The model was next calibrated to pre-flood conditions in Muskrat Falls Reservoir.
4. Simulations were carried out to predict mercury concentrations in Muskrat Falls Reservoir after flooding, using the areal flood zone methylmercury loading rates from the Robert Bourassa simulation.

Regression Model

A regression model (Harris *et al.,* 2015) was also used to predict peak increases in mercury concentrations in northern pike in Muskrat Falls Reservoir. The model is derived from a simplified mass balance expression for methylmercury sources and sinks in reservoirs and predicts peak fish mercury concentrations on the basis of three site conditions: flooded area, total area, and mean annual flow. This model does not predict how concentrations change with time. The version of the model applied to Muskrat Falls Reservoir predicts the increase in fish mercury concentration, which is then added to the baseline concentration.

The sites used to develop the regression model had data for peak concentrations but typically did not measure pre-flood concentrations on a site-by-site basis. Fish mercury concentrations in the vicinity of the Muskrat Falls Reservoir site are low, *e.g.* 0.26 µg/g in 700 mm northern pike (additional information is presented below) and possibly outside the range of values bounded by the model development data. To help address this issue, the regression model was tested for a range of assumptions regarding baseline concentrations (0.25 µg/g at all sites or 0.55-0.59 µg/g based on regional data for 12 reservoirs), flooded areas that contribute to methylmercury supply, and whether to allow the regression intercept to float or be forced through the origin. The equation for the base case model for 700 mm northern pike was as follows:

$$\text{Increase in fish Hg } (\mu g/g) = 0.322 * (A_f/(Q + 0.09 * A_t) + 0.202 \qquad (1)$$

Where:

A_f = flooded area (km^2)
A_t = Total reservoir area (km^2)
Q = mean annual flow (km^3/yr)

The overall peak concentration was then calculated as the increase plus the baseline concentration of 0.26 µg/g.

2.2 Approaches to predict increases in methylmercury concentrations exported downstream from Muskrat Falls Reservoir

Two approaches were used to estimate the increase in methylmercury concentrations and loads exported downstream as a result of flooding at Muskrat Falls Reservoir. The first approach was to use estimates from the ResMerc model application to the reservoir. Daily predictions of methylmercury concentrations and loads exported from the reservoir were generated by the model, for 30 years post-flood. The first 5 years after flooding were predicted to have the highest concentrations and export rates and were used in downstream model simulations by Baird (Brunton, 2018).

The second approach was to use data from the FLUDEX experiment as the basis for an estimate. FLUDEX was an upland flooding experiment carried out from 1999-2003 at the Experimental Lakes Area in Ontario (Bodaly *et al.*, 2004, Hall *et al.*, 2005). Three small reservoirs were created, with different carbon pools in the flood zone (per m^2). Each year the experimental reservoirs were flooded at approximately the beginning of June and drained in mid to late September. Methylmercury generation and greenhouse gases were studied intensively. Methylmercury concentrations were measured at the inflow and outflow from each reservoir approximately every two weeks during the flood seasons. This information was used to generate net methylmercury loads and export occurring due to flooding as water passed through the reservoirs. Net loads for the 1st three years of the experiment were published (Bodaly *et al.*, 2004, Hall *et al.*, 2005) and loads for years 4 and 5 were obtained from Britt Hall (Hall, 2018). These loading estimates, per m^2 of flood zone, were scaled up for Muskrat Falls to estimate the downstream methylmercury loads associated with flooding. The FLUDEX site with the highest methylmercury net loads (medium carbon site) was used in the Muskrat Falls Reservoir analysis.

Because flooding occurred from June to September each year during FLUDEX, it was necessary to estimate methylmercury loads for the remainder of the year. One extreme approach would be to assume that no methylmercury would be produced and exported from September through June. That would be unrealistically low. Another option would be to assume that the methylmercury loads produced from June-September would be maintained all year. This would likely be an overestimate, because methylation is temperature dependent. A decision was made to use the average of these two options, effectively using half of the June-September daily average rate for the September to June period. This approach resulted in more than half of the annual estimated methylmercury export occurring from September-June each year.

The two estimates of reservoir methylmercury export were used by Baird (Brunton, 2018) in the downstream modelling analysis.

3 Site Characteristics for Muskrat Falls, Robert Bourassa and Notigi Reservoirs

Muskrat Falls Reservoir

Muskrat Falls Reservoir will have a maximum depth of approximately 27 m at full impoundment, and a mean annual water residence time of approximately 10 days, based a mean annual flow of 1781 m^3/s (average for 2006-2015, Water Survey of Canada, 2017). Based on monitoring at station N1 located at the upstream end of the reservoir from December 2016 - December 2017, river water temperatures ranged from -1 to 18 C, the mean pH was 7.4 and the mean dissolved organic carbon was 4.6 mg/L (derived from Nalcor, 2018). The water column is predicted to remain well mixed and oxygenated after reservoir creation (Nalcor, 2009b).

The total area of the reservoir will be 101.5 km^2. The amount of flooded terrain is 43.9 km^2 (Table 1). Within the flooded area, 6.9 km^2 are gravel bars and 6.6 km^2 are riparian areas with very low carbon content. It was assumed that the flooded area that effectively contributed to elevated methylmercury supply should exclude the gravel bars, and possibly exclude flooded riparian areas. For the purpose of ResMerc and FLUDEX based analyses we conservatively assumed that riparian areas would contribute to methylmercury supply, and the relevant flooded area was 37 km^2. The regression model was applied using scenarios including and excluding flooded riparian areas.

Table 1. Muskrat Falls Reservoir flood zone characterization. Data from AMEC Foster Wheeler, 2018a

ELC type	Area (km^2)	% of Reservoir area	% of Flooded Area
Black Spruce / Feathermoss Forest	8.59	8.5	19.6
Fir - White Spruce Forest	8.14	8.0	18.6
Black Spruce / Lichen Woodland	0.91	0.9	2.1
Hardwood Forest	2.20	2.2	5.0
Mixedwood Forest	6.96	6.9	15.9
Spruce Fir / Feathermoss Forest	1.16	1.1	2.6
Bl. Spruce/Sphagnum Woodland	0.20	0.2	0.5
Unvegetated	0.04	0.04	0.1
Wetland	2.18	2.2	5.0
Riparian	6.56	6.5	15.0
Gravel Bar	6.92	6.8	15.8
All flooded forest	28.18	27.8	64.2
All flooded forest + wetland	30.38	29.9	69.2
Total flooded terrain	43.91	43.3	100.0
Total flooded terrain minus gravel bar	36.98	36.4	84.2
Total flooded terrain minus gravel bar and riparian	30.42	30.0	69.3
Water	57.59	56.7	
Total	101.51	100.0	

Approximately 5% of the flood zone is wetland, representing ~2% of the overall reservoir area (Table 1). Carbon pools were estimated for the upland flood zone based on the following:

- Humic layer: Field survey by AMEC Foster Wheeler (2017)
- Litter: Literature review by AMEC Foster Wheeler (2018b)
- Foliage: FLUDEX experiment data from Hall *et al.* (2005)

It was assumed that foliage would represent a labile pool of carbon affecting methylation rates while other above-ground vegetation would not contribute to elevated methylmercury supply.

ResMerc has two sediment layers. The top layer was set up with a 2 cm thickness and included carbon from foliage and litter. The lower layer represented the humic layer that averaged about 8 cm in depth (Figure 3).

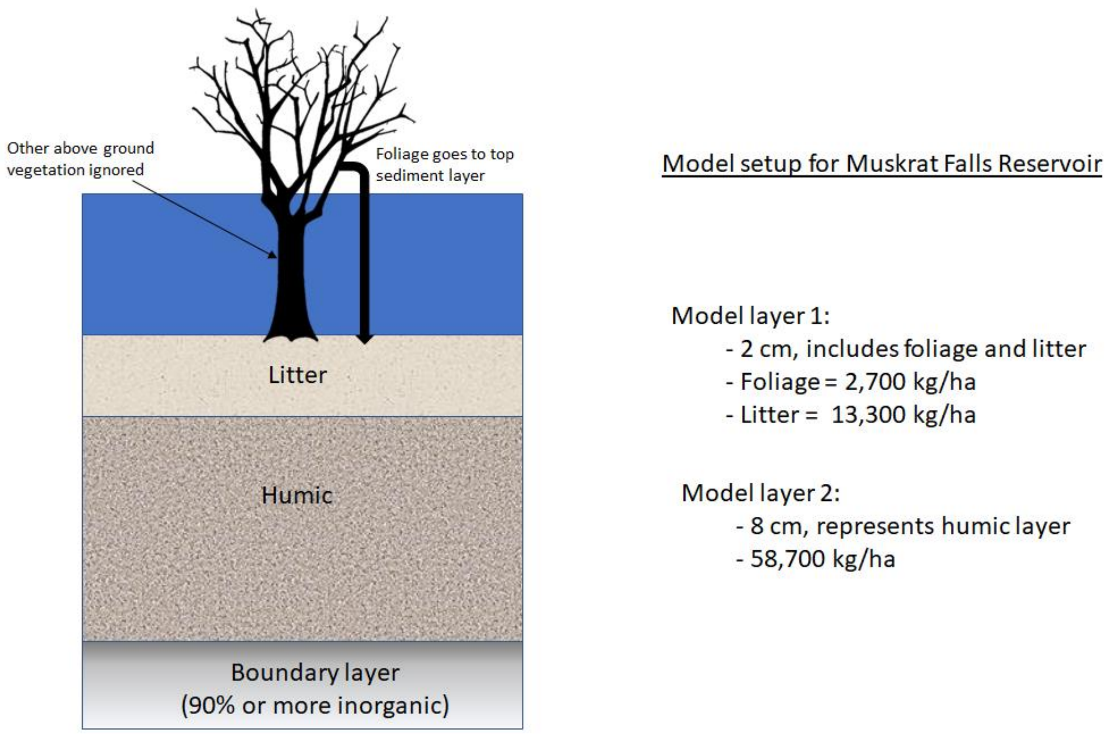


Figure 3. Carbon pools in model soil layers for Muskrat Falls Reservoir flood zone.

The estimated baseline mercury concentration in a 700 mm northern pike in the reservoir area was 0.26 µg/g (Figure 4), derived from McCarthy (2017). The estimated baseline concentrations for 400 mm longnose suckers and lake whitefish were 0.17 and 0.12 µg/g respectively.

The food web related to bioaccumulation by northern pike was set up for ResMerc as shown in Figure 5. Macroinvertebrates are an important component at the base of the northern pike food web in the freshwater system. It was assumed for based case simulations that most of the methylmercury in macroinvertebrates is derived from methylmercury in the water column. Alternative scenarios were also simulated where macroinvertebrates had a greater connection to the pool of methylmercury in sediments.

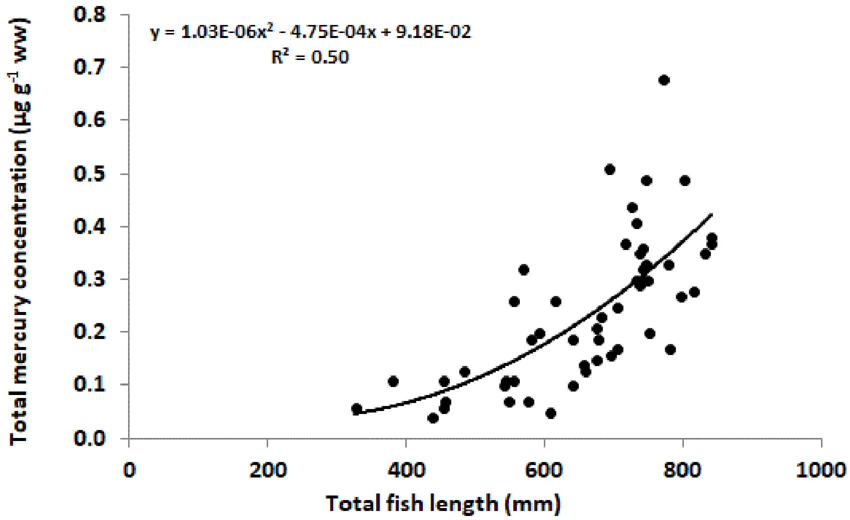


Figure 4. Observed mercury concentrations in northern pike from River Section 2 in Lower Churchill River. Data from 2012-2016, n=52. Data from McCarthy (2017)

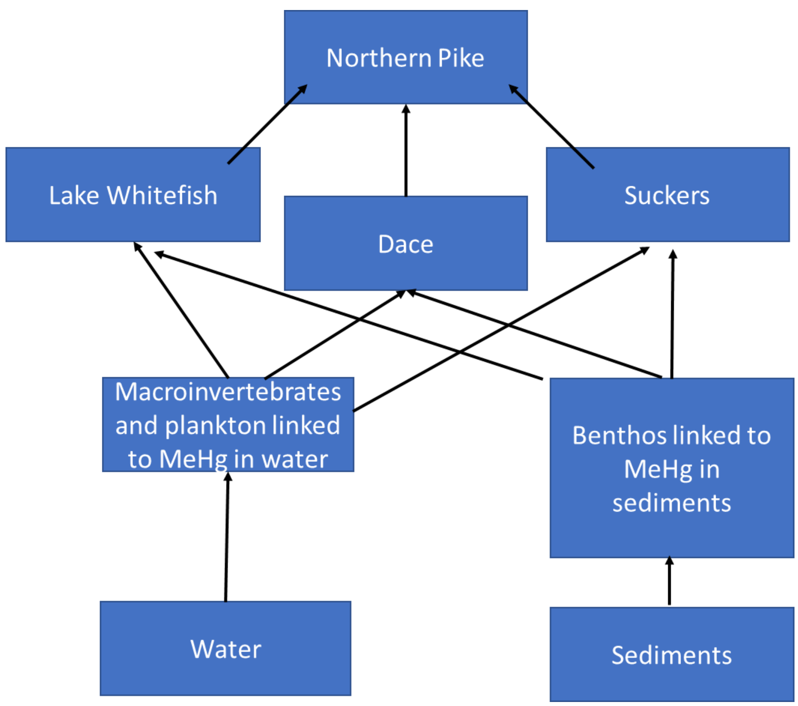


Figure 5. Major food web compartments and links used in simulations for northern pike. Information from J. McCarthy, unpublished.

Robert Bourassa Reservoir

Robert Bourassa Reservoir was developed as part of the La Grande complex in Quebec. Reservoir filling was completed in December 1979. Flooding increased the water surface area from 205 km^2 to 2,835 km^2 (Schetagne *et al.*, 2003). The flood zone represented 92% of the total reservoir. With a mean annual flow of 3,374 m^3/s, the estimated mean hydraulic residence time was 7 months. Mercury levels in 700 northern pike reached 3.3 µg/g 11 years after flooding and then declined towards background levels (Hydro Québec, 2013; Schetagne *et al.*, 2003). Upstream reservoirs in the system, including La Grande 3 immediately upstream, which began flooding in 1981, may also have influenced fish mercury levels in Robert Bourassa Reservoir.

Notigi Reservoir

Notigi Reservoir, Manitoba was created when water was diverted south from the Churchill River through the Burntwood/Nelson River system to boost the water supply to several generating stations on the Nelson River. Reservoir filling was completed in December 1976. Flooding increased the water surface area from 198 km^2 to 785 km^2 from the South Bay diversion channel to Notigi Dam (Manitoba Hydro, 2006a). The flood zone represented 75% of the total reservoir. With a mean annual outflow of 764 m^3/s from Notigi dam (1978-2005, estimated by R. Harris from Manitoba Hydro 2006b), the estimated mean hydraulic residence time was 110 days. Mercury levels in 550 mm northern pike rose to approximately 2 µg/g within 5-7 years, and then declined towards background levels (Bodaly, 2005).

4 Results

4.1 Mechanistic model results

ResMerc was calibrated to estimate the methylmercury loads required from the flood zone to support observed mercury concentrations in northern pike and lake whitefish in Robert Bourassa Reservoir. Annual averaged methylmercury diffusion loads predicted for flooded uplands in Robert Bourassa Reservoir ranged from approximately 80-120 $ng/m^2/day$ for years 2-6 after flooding (filling occurred during the first year). The resulting modeled fish mercury concentrations matched observations well (Figure 6). These methylmercury loads from the flood zone produced peak methylmercury concentrations in water of nearly 1 ng/L (Figure 7). No water column methylmercury data were available from existing full-scale reservoirs for comparison.

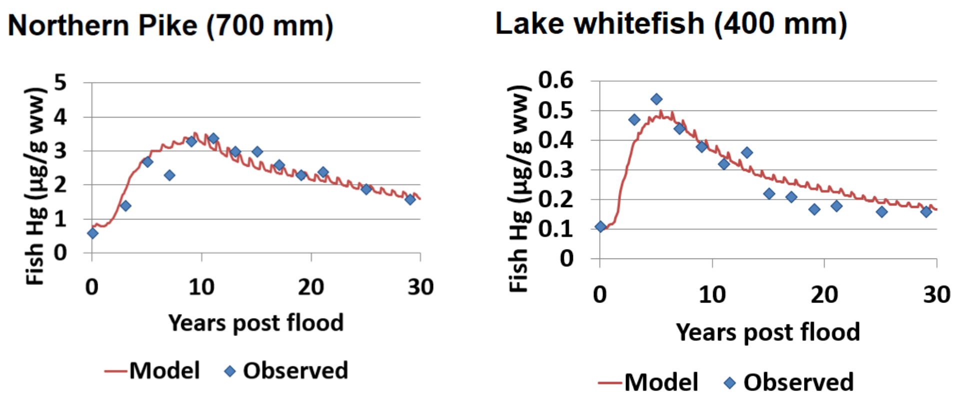


Figure 6. Observations and ResMerc results for methylmercury concentrations in northern plke (700 mm) and lake whitefish (400 mm) in Robert Bourassa Reservoir, QC. Observations from Hydro Québec (2013)

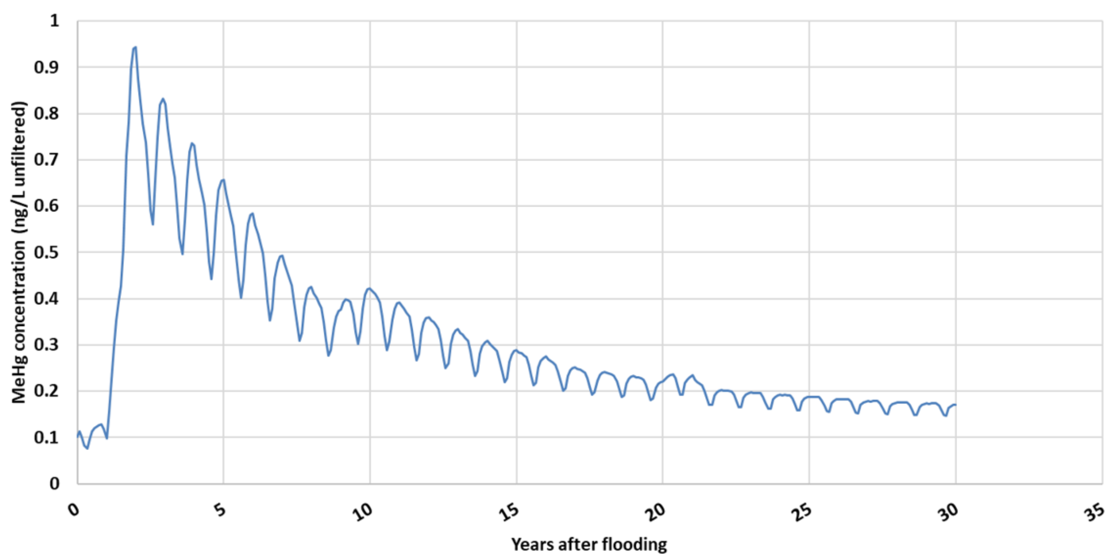
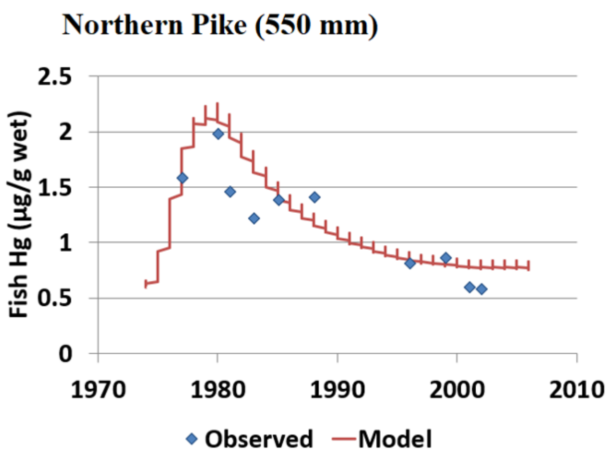


Figure 7. Predicted methylmercury concentration in surface waters in Robert Bourassa Reservoir, QC.

The calibrated model was then applied to Notigi Reservoir, MB, again assuming the same flood zone characteristics and areal carbon pools in the flood zone as were estimated for Muskrat Falls Reservoir. ResMerc predictions of mercury concentrations in northern pike and lake whitefish reasonably matched observations, with a slight tendency to overpredict (Figure 8).



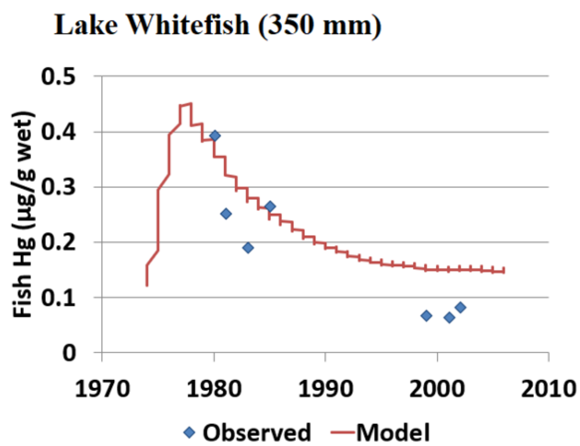


Figure 8. Observations and ResMerc results for mercury concentrations in northern pIke (700 mm) and Lake whitefish (350 mm) in Notigi Reservoir. Observations derived from Bodaly (2005)

ResMerc was next applied to pre-flood conditions in the lower Churchill River at the Muskrat Falls site. Rate constants for mercury cycling and carbon turnover were the same as used for Robert Bourassa and Notigi Reservoirs. The simulation was "warmed up" for 100 years to allow conditions to stabilize, and results were examined for the 101st year. Simulated and concentrations reasonably matched observations of methylmercury in water (Figure 9) and fish (Figure 10). Minor adjustments were made to rate constants for fish methylmercury dynamics to improve the model fit.

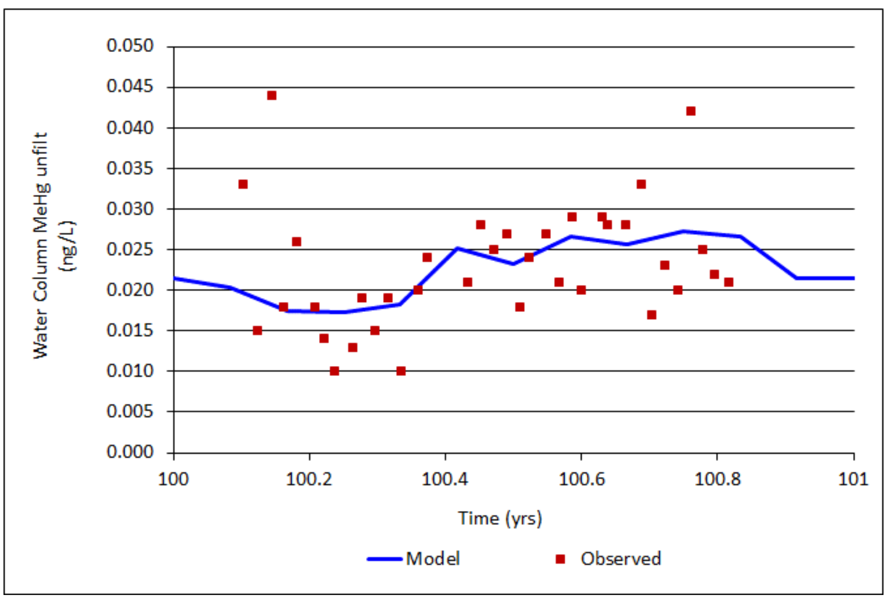


Figure 9. Observed and simulated methylmercury concentrations in surface waters for pre-flood conditions at the Muskrat Falls Reservoir site. Data from Station N1 (Nalcor, 2018)

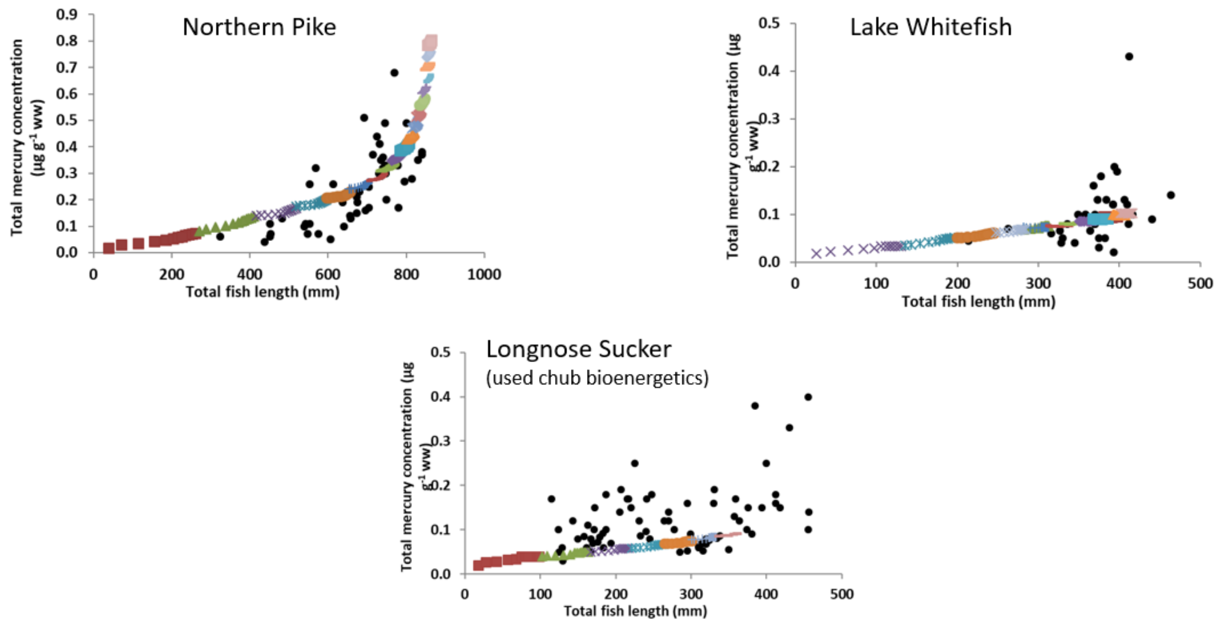


Figure 10. Observed and simulated fish mercury concentrations for pre-flood conditions at the Muskrat Falls Reservoir site. Black dots are observations. Coloured points represent predicted concentrations in different year classes. Data from 2010 to 2016 in River Section 2. Data from J. McCarthy (2017).

The model was then applied to post-flood conditions for Muskrat Falls Reservoir. Due to limitations with the model's ability to simulate the filling period, post-flood simulations started with the reservoir at full elevation (39 m asl). Predicted average annual methylmercury diffusion loads from flooded soils to overlying water ranged from approximately 80-145 ng/m^2/day from flooded uplands during the first 6 years after flooding. Methylmercury concentrations were predicted to increase briefly to approximately 0.1 ng/L in surface waters of Muskrat Falls Reservoir, about 5X the baseline concentration, and the contribution from flooding briefly reached a peak of 0.07 ng/L (Figure 11). The peak export rate for methylmercury briefly reached a peak of 10 g/day (Figure 12). Peak predicted fish mercury concentrations were 0.64 µg/g in 700 mm northern pike and 0.24 µg/g in 400 mm lake whitefish (Figure 13). These values are roughly 2.0 – 2.5X the baseline concentrations. An alternative scenario was simulated assuming that 50% of the base of the food web derived methylmercury from sediments post-flood. The peak predicted mercury concentration for 700 mm northern pike was 0.80 µg/g, approximately 3X the baseline. Overall, peak concentrations predicted in northern pike were 2-3X the baseline concentrations for the scenarios tested.

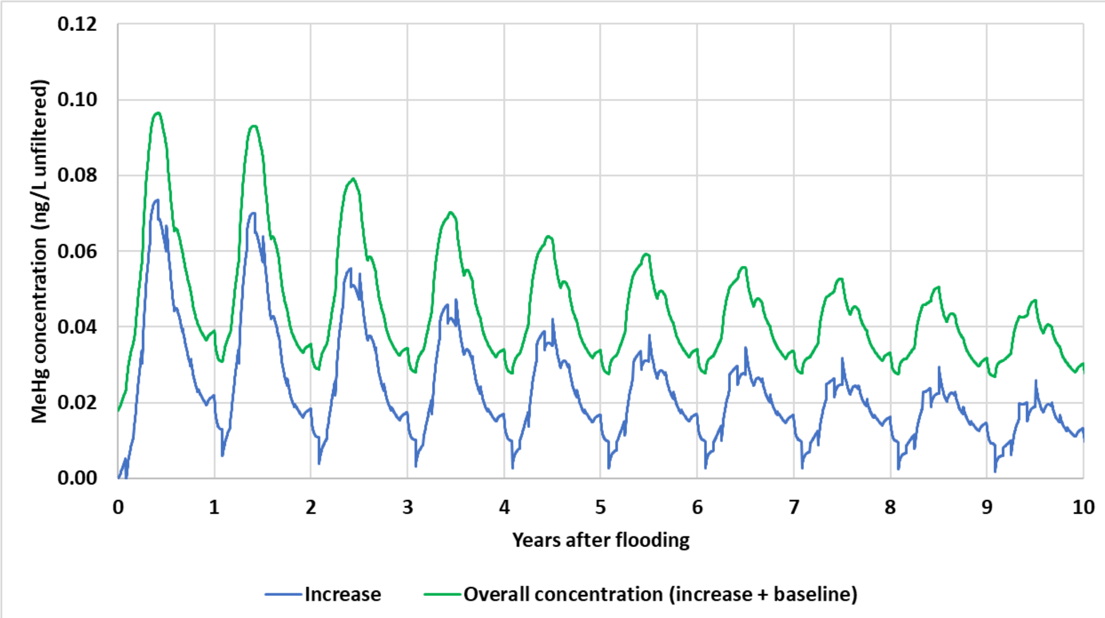


Figure 11. Predicted methylmercury concentrations in Muskrat Falls Reservoir surface waters (and exported downstream) based on ResMerc model.

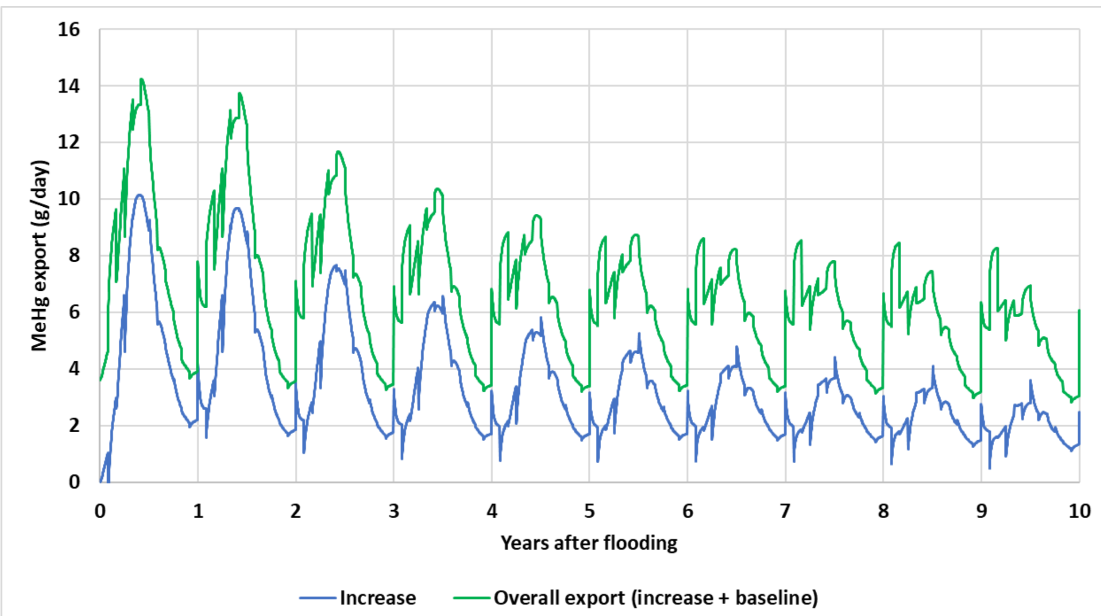


Figure 12. Predicted methylmercury export from Muskrat Falls Reservoir based on ResMerc model.

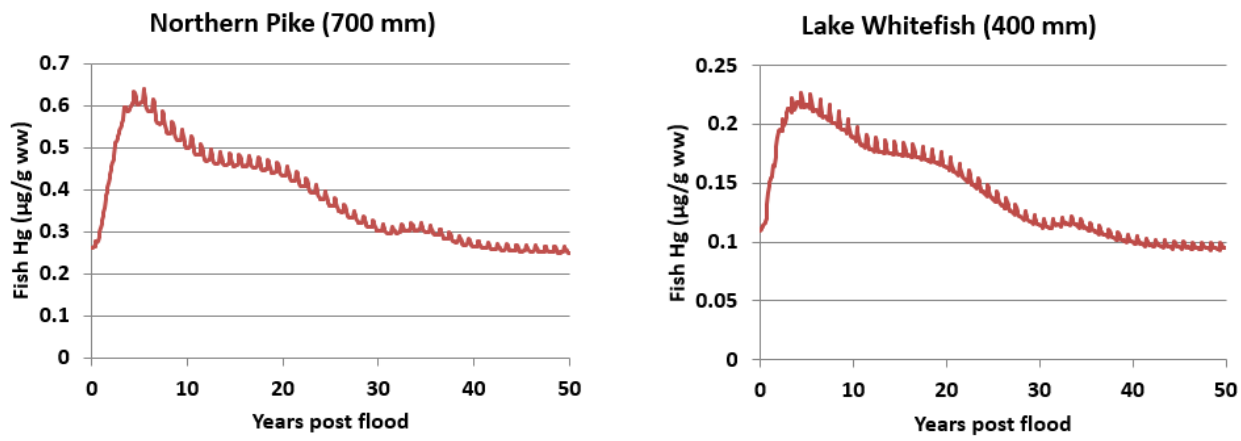


Figure 13. Predicted methylmercury concentrations in Muskrat Falls Reservoir northern pike and lake whitefish, based on the ResMerc Model base case simulation.

4.2 Regression model results

Regression model estimates of peak increases in mercury concentration for 700 mm northern pike in Muskrat Falls Reservoir are shown in Figure 14. The y axis in the figure shows the predicted increase, which must be added to the baseline to estimate the overall peak concentration. The base case model, shown in the figure, allowed the regression intercept to float, although in reality no flooding would produce no increase. Muskrat Falls Reservoir is predicted to have a peak concentration between 0.61 and 0.64 µg/g, about 2.4X baseline concentrations. A range of model outcomes based on different assumptions about baseline fish mercury concentrations, effective flooded area, and whether to allow the model intercept to float or be forced through the origin, is shown in the red shaded area in Figure 14. These results were very consistent with ResMerc predictions.

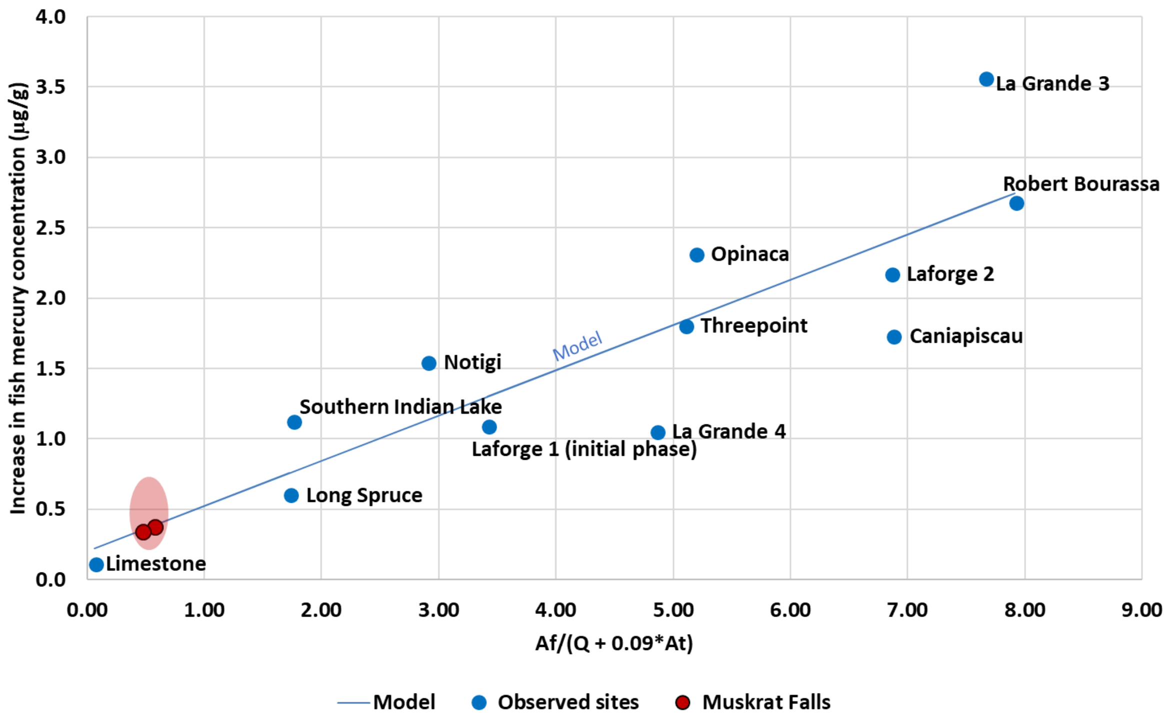


Figure 14. Regression model results for 700 mm northern pike. Blue dots are estimated increases in mercury concentrations in 12 Canadian reservoirs, based on observed peak concentrations and estimated baseline concentrations of 0.55-0.59 µg/g. Blue line is model version with floating intercept. Red dots are predictions for Muskrat Falls Reservoir based on flooded area of 30-37 km^2. Shaded red area includes predictions for a range of assumptions related to the intercept (floating or forced through origin, flooded area (30-37 km^2), and baseline concentrations (0.25 at all sites or 0.55-59 µg/g). Predicted increase must be added to the baseline concentration to estimate the overall peak concentration. Additional information on field data for existing reservoirs available in Harris *et al*. (2015).

4.3 Methylmercury export estimates based on FLUDEX

Methylmercury export rates observed during the FLUDEX experiment are shown in Table 2. These data represent the net export rates each year, based on outflow minus inflow fluxes during the flood period from approximately June – September. The medium carbon site (#2) had the highest methylmercury export rates and was used to estimate methylmercury export for Muskrat Falls Reservoir, shown in Table 3. Results from FLUDEX were applied to Muskrat Falls Reservoir as follows:

- Areal loading rates from FLUDEX from the June -September flood season each year were extended to annual estimates by assuming that if flooding had continued each year, the daily loads would have been half the average rate during the June-September period (see earlier discussion).
- FLUDEX net export rates, per m^2, were multiplied by the portion of the Muskrat Falls flood zone assumed to contribute to excess methylmercury supply (37 km^2), to estimate the mass of methylmercury exported each year from Muskrat Falls Reservoir. This area included about 6.5 km^2 of flooded riparian terrain, which may not contribute as much methylmercury (per m^2) as flooded forest soils.

Year after flooding	Methylmercury export (mg/ha)		
	Reservoir 1	Reservoir 2	Reservoir 3
1	66	126	60
2	77	131	61
3	44	88	32
4	71	42	31
5	38	29	11

Table 2. Net methylmercury export (outflow minus inflow) for the three FLUDEX reservoirs from 1999-2003. Values are mg/ha for flood season each year (approximately June – September). Site 2 data were used for Muskrat Falls Reservoir analysis. Data from Hall (2018).

Year after flooding	Estimated annual methylmercury export (kg)
1	1.08
2	1.13
3	0.76
4	0.36
5	0.25

Table 3. Estimated annual methylmercury export from Muskrat Falls Reservoir for first 5 years after flooding (excess above baseline, associated with flooding). Estimates are based on FLUDEX data, scaled up to flooded area at Muskrat Falls that contributes to excess methylmercury supply (37 km^2). Annual values are based on FLUDEX Reservoir 2 data for June-September each year, plus contribution for remainder of year assuming half the average daily rate for June-September period.

Predicted average annual increases in methylmercury concentrations in water exported from the reservoir are presented in Figure 15 for the FLUDEX and ResMerc based analyses. The FLUDEX-based estimates had largely declined after 5 years. The concentrations based on the model calibration from Robert Bourassa Reservoir were higher and declined more slowly that the estimates based on FLUDEX. Both estimates are much lower than the increase in concentration in the reservoir predicted by Calder et al (2016), also shown in Figure 15. The predicted increase by Calder *et al.* (2016) was 0.16 ng/L, sustained for an undefined period long enough for fish mercury concentrations to respond in proportion. This concentration is 4.6 to 8X greater than the maximum one-year average increases predicted using FLUDEX or ResMerc based estimates, and 5-9X greater than the maximum 3 year average increases from FLUDEX and ResMerc.

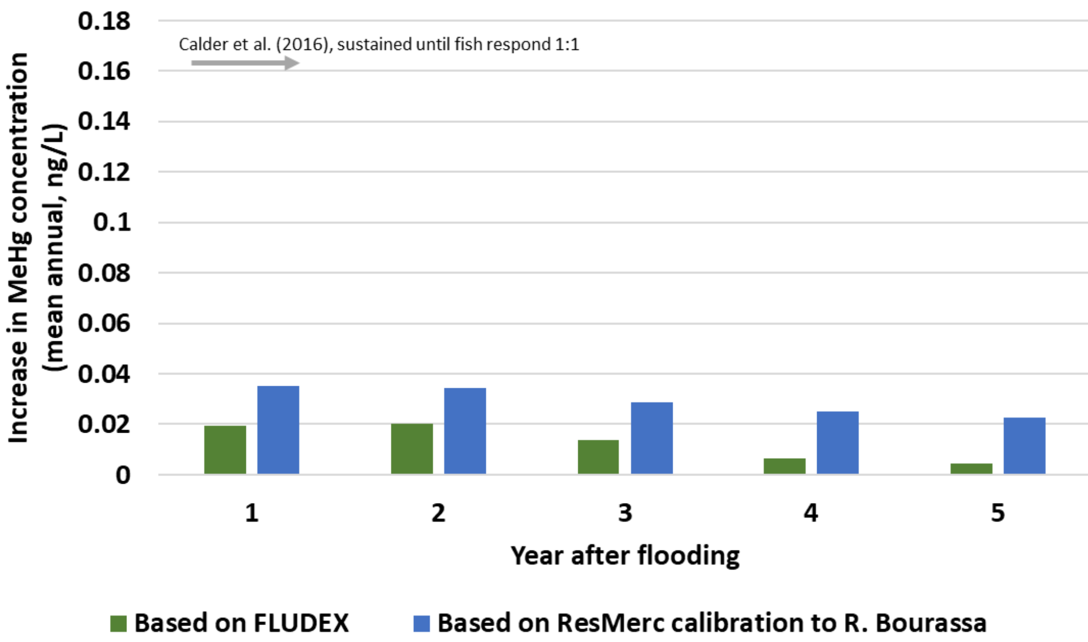


Figure 15. Estimated increases in methylmercury concentrations in waters exported from Muskrat Falls Reservoir. Overall concentration = increase + baseline. Also shown is predicted increase from Calder *et al.* (2016).

5 Discussion

Two approaches were used to estimate the magnitude and timing of downstream export of methylmercury in water from Muskrat Falls Reservoir to Goose Bay and Lake Melville. The first approach used observations from FLUDEX, while the 2nd approach used the ResMerc model to back-estimate flood zone methylmercury loading rates that would produce fish mercury concentrations observed from two existing reservoirs, and then applied those loading rates to predict methylmercury concentrations in water and fish in Muskrat Falls Reservoir. The use of these two approaches was influenced by the absence of measured concentrations of methylmercury in waters from full scale reservoirs. The only known datasets are from the FLUDEX upland and ELARP wetland reservoir experiments at the Experimental Lakes Area, and the FLUDEX data formed an important component of the analysis. The ResMerc model analysis provided a second means to gain insights into water column methylmercury concentrations that occur in new reservoirs.

The FLUDEX-based analysis predicted an increase of 0.02 ng/L in water exported from Muskrat Falls Reservoir (maximum one-year average). The ResMerc analysis predicted an increase of 0.035 ng/L. These estimates are within a factor of 2 of each other in magnitude, which is encouraging in the absence of being able to develop confidence limits associated with predicted increases in water column methylmercury, which would require observations of methylmercury concentrations in water from multiple reservoirs. It is also possible that the actual methylmercury loads from Robert Bourassa Reservoir, per m^2, were greater than occurred during FLUDEX, given that many factors affect the production of methylmercury in reservoirs. The FLUDEX site for example experienced a fire roughly 20 years prior to the experiment. Whether this reduced carbon pools relevant to methylmercury production that was available for bioaccumulation during FLUDEX is not clear. Conversely, Robert Bourassa Reservoir was downstream of other reservoirs that could have contributed to higher fish mercury levels, including La Grande 3. The ResMerc analysis did not explicitly simulate upstream reservoir contributions and allocated any methylmercury supply needed to produce observed fish mercury levels to in-situ production in Robert Bourassa Reservoir. These considerations guided the decision to use the average of the two estimates of downstream export in simulations in Goose Bay and Lake Melville by Baird (2018).

The predicted increases in water methylmercury concentrations exported from Muskrat Falls Reservoir were 4.6 to 8x lower than the 0.16 ng/L increase predicted by Calder *et al.* (2016), which predicted much higher loads of methylmercury from the flood zone than were observed from FLUDEX or predicted from the ResMerc model analysis.

The mechanistic and regression models produced very similar predictions of peak fish mercury concentrations in Muskrat Falls Reservoir, providing consistency among the various lines of evidence used in the analysis. Both models predicted that concentrations in 700 mm northern pike would increase roughly 2.5-3X from a baseline concentration of 0.26 µg/g to a peak in the range of 0.6 to 0.8 µg/g.

Flow dilution was predicted to be an important moderating factor in the mechanistic and regression model predictions for Muskrat Falls Reservoir. For example, the flood zone methylmercury loading rates (per m^2) applied to Robert Bourassa Reservoir in ResMerc simulations produced much lower peak concentrations in water and fish when approximately the same rates were applied to Muskrat Falls Reservoir. This was related to the shorter water residence time (~10 days vs 7 months) and greater flow dilution associated with Muskrat Falls Reservoir, and the fraction of the reservoirs consisting of flooded terrain: about 92% Robert Bourassa Reservoir versus ~40% for Muskrat Falls.

Overall, methylmercury concentrations (baseline + increase) in waters exported downstream from Muskrat Falls Reservoir were predicted to peak at roughly 2-3X baseline values when averaged over time periods relevant to bioaccumulation in adult fish (*e.g.* one-year average concentration up to 0.04 to 0.055 ng/L), based on the two approaches used. The results of this analysis were used in downstream analyses by Brunton (2018) and Wood (2018) to estimate potential increases in methylmercury concentrations in water and biota in Goose Bay and Lake Melville.

6 References

AMEC Foster Wheeler (2018a) Email from J. Abbott to R. Harris, February 21, 2018. Spreadsheet attachment titled: Ecotype Areas by Contour.xlsx

AMEC Foster Wheeler (2018b) Letter from T. Praamsma to R. Harris., titled "RE: TF13104119.5600 Soil Sampling Carbon Estimates in Boreal Forests". January 29, 2018

AMEC Foster Wheeler (2017) Muskrat Falls Soil Sampling Program – 2016 – Final Report submitted to Nalcor Energy. October 23, 2017. Amec Foster Wheeler Project #: TF13104119.5600.

Azimuth 2018. Summary of post-exposure Human Health Risk Assessment from methylmercury in seafood in Goose Bay and Lake Melville, Labrador. A technical memo prepared for Nalcor Energy, St. John's NL by Azimuth Consulting Group Partnership, Vancouver BC. July 23, 2018

Bodaly, R.A. (2005) Email to R. Harris with attached spreadsheet: "Notigi Hg I fish 1977-02.xls. September 22, 2005.

Bodaly, R.A., K.G. Beaty, L.H. Hendzel, A.R. Majewski, M.J. Paterson, K.R. Rolfhus, A.F. Penn, B.D. Hall, C.J. Mathews, K.A. Cherewyk. M. Mailman, J.P. Hurley, S.L. Schiff and J.J. Venkiteswaran (2004) Experimenting with Hydroelectric Reservoirs. Environ. Sci. Technol. 38(18): 346A-352A

Brunton. A (2018) Model analysis of Goose Bay and Lake Melville hydrodynamics and transport of methylmercury from Muskrat Falls Reservoir. Baird Technical Memorandum to R. Harris. July 2018.

Calder, R. S. D., A. T. Schartup, M. Li, A. P. Valberg, P. H. Balcom and E. M. Sunderland (2016) "Future Impacts of Hydroelectric Power Development on Methylmercury Exposures of Canadian Indigenous Communities." Environ Sci Technol 50(23): 13115-22.

Dillon (2016) Final Baseline Human Health Risk Assessment: Lower Churchill Hydroelectric Generation Project. Prepared for: Nalcor Energy Lower Churchill Hydroelectric Generation Project. Submitted by: Dillon Consulting Limited. October, 2016.

Hall, B. (2018) Email to R. Harris April 14, 2018. File attachment: "MeHg and THg yields all years FLUDEX.pdf"

Hall, B.D., V.L. St. Louis, K.R. Rolfhus, R.A. Bodaly, K.G. Beaty, M.J. Paterson, and K.A. Peech Cherewyk (2005) Impacts of Reservoir Creation on the Biogeochemical Cycling of Methyl Mercury and Total Mercury in Boreal Upland Forests Ecosystems 8: 248–266

Harris, R., C. Beals and J. Therrien (2015) Simulations of Peak Fish Mercury Concentrations in Hydroelectric Reservoirs: Modeling and Data Analysis Update. Prepared for: Paul Norris. Ontario Waterpower Association. Peterborough, Ontario. March 5, 2015

Harris, R.C., D.H. Hutchinson and D. Beals (2012). Mercury Reservoir Modeling. Site C Clean Energy Project. Volume 2 Appendix J, Part 3. Prepared for Azimuth Consulting Group. December 2012

Harris, R.C., D.H. Hutchinson and D. Beals (2010) Application of a Mechanistic Mercury Model to the Proposed Lower Churchill Reservoirs. Technical Memorandum in support of the Nalcor response to IR#JRP.166. Prepared for Nalcor. December 2010.

Harris, R., and D. Hutchinson (2009) Reservoir Mercury Cycling Model (ResMerc) for Windows. ResMerc Version 2.0. User's Guide and Technical Reference. Prepared for Manitoba Hydro. April 2009

Harris, R.C., D. Hutchinson and D. Beals (2009) Predicting Mercury Cycling and Bioaccumulation in Reservoirs: Development and Application of the RESMERC Simulation Model Final Report, March 2009. Final Report prepared for Manitoba Hydro, March 2009

Harris, R.C. and R.A. Bodaly (1998) Temperature, growth and dietary effects on fish mercury dynamics in two Ontario Lakes. Biogeochemistry 40: 175-187

Hydro Québec (2013) Évolution des teneurs en mercury dans les poissons. Rapport synthèse 1978-2012. Suivi environnemental du complexe La Grande. Octobre 2013

McCarthy, J. (2017). Email to R. Harris. Spreadsheet attachment titled Mercury Database (rev2) Reed.xlsx. November 8, 2017

Manitoba Hydro (2006a) p162 , Table 8 from 060327-Notigi Info.pdf

Manitoba Hydro (2006b) Flows derived from spreadsheet sent from M. Drouin to Reed Harris, August 29, 2006: reed 20060829.xls

Nalcor (2018) Spreadsheet titled TF13104119.5500 - Lab Results (January 15, 2018).xlsx

Nalcor (2009a) Lower Churchill Hydroelectric Generation Project, Response IR #156

Nalcor (2009b) Lower Churchill Hydroelectric Generation Project – Environmental Impact Statement – Volume II, Part 1, Biophysical Assessment. Section 4.7.2. February 2009.

Schetagne, R., J. Therrien and R. Lalumière (2003) Environmental Monitoring at the La Grande Complex. Evolution of Fish Mercury Levels. Summary Report 1978–2000. Direction Barrages et Environnement, Hydro-Québec Production and Groupe conseil GENIVAR Unc..

Water Survey of Canada (2017) https://wateroffice.ec.gc.ca/report/historical_e.html?stn=03OE001&mode=Table

Wood (2018) Predicted Increases in Fish Methylmercury Muscle Tissue Concentrations in Goose Bay and Lake Melville. Submitted to Nalcor Energy, St. John's, NL. 22pp + appendices

Jim McCarthy

Jim is an associate biologist with Wood Environment and Infrastructure Solutions in St. John's, Newfoundland. He received his B.Sc. and M.Sc. in Biology from Memorial University of Newfoundland in 1991 and 1996, respectively. He is currently a part-time PhD candidate at the Canadian Rivers Institute at the University of New Brunswick where he is investigating various aspects of environmental change and fish adaptation including downstream mercury transport from the Muskrat Falls Hydroelectric Facility, fish adaptation to rapid habitat changes caused by reservoir formations, and adaptive management. He has been involved in a wide range of projects in Newfoundland and Labrador, Nunavut, Alaska, British Columbia and Nova Scotia for private organizations and government agencies. Projects have generally entailed the design and implementation of baseline studies, aquatic offset plans, environmental effects monitoring programs, and impact assessments related to various human activities such as hydroelectric developments, oil and gas, mining/construction, and forest harvesting.

His involvement with the Lower Churchill Hydroelectric Development has spanned 20 years including baseline data collection since 1998, authoring the Aquatic Ecosystem component of the EIS, Fish and Fish Habitat Expert Witness for Nalcor at the Panel Hearings, developed the Fish Habitat Compensation Plan, and continues to monitor baseline conditions as part of Nalcor's Environmental Effects Monitoring commitments. He was recently a member of the Independent Expert Committee (IEC) formed to advise on the potential downstream effects of Muskrat Falls on mercury concentrations. Other large ongoing projects include Aquatic Ecosystem support for Vale's Long Harbour Processing Facility, CFI's St. Lawrence Fluorspar Mine, and the Government of Nunavut Fisheries and Sealing Division. He is a Certified Fisheries Professional with the American Fisheries Society with over twenty years of experience, and a Licensed Hunting and Fishing Guide.

Azimuth Consulting Group Partnership

218-2902 West Broadway
Vancouver, BC
Canada V6K 2G8

Phone: 604-730-1220
www.azimuthgroup.ca

Technical Memorandum

Date: **February 25, 2018**
To: **Peter Madden, Nalcor Energy**
From: **Randy Baker with G. Mann (L. Melville Mass Balance)**
Our File: **NE 18-01**

RE: **Evaluation of MeHg Production by Muskrat Falls Reservoir and Implications for Lake Melville – A Top-Down, Mass-Balance Approach**

---

1 Summary

This Technical Memorandum examines the Calder et al. (2016) assumptions that are crucial to support their key conclusions regarding the rate and duration of methylmercury (MeHg) flux from the flooded soils of Muskrat Falls Reservoir (MFR) and the associated potential to increase MeHg in the downstream food web of Lake Melville. We rely on two primary lines of evidence, grounded in empirical data, to demonstrate that:

1. The baseline physical and chemical conditions at MFR are characteristic of a system that has a weak mercury (Hg) methylation potential. Support for this argument comes from the 'Canadian Reservoirs Comparison Matrix' (CRCM; Azimuth 2012). The CRCM, originally developed for the Site C Hydroelectric Project in BC, used a weight-of-evidence approach to compare key physical, chemical and ecological data from Site C with empirical data from 14 reservoirs. Site C fell into the category of 'low methylating', defined as <3x increase in peak fish Hg relative to baseline. When the same key parameters from MFR are plugged into the CRCM, the two reservoirs overlap nearly completely. Given their great similarity, Calder et al. do not provide sufficient justification to place them at opposite ends of the spectrum of possibilities, especially as Calder et al. *agreed* with the findings related to the Site C project.

2. We have determined that the available mass of inorganic Hg in humic soils in MFR that is available for Hg methylation and transfer to the food web is limited.

Based on empirical data and evidence from the scientific literature, MFR and can potentially only generate a *total mass* of **2.35 kg** of MeHg that is amortized over a period of at least 10 years. Calder et al. assumed al MeHg flux rate of 664 ng/m^2/y, or **7.5 kg** of MeHg *per year* from the 30 km^2 forested area of MFR – every year extending over a period of up to 10 years. We conclude there is an insufficient mass of raw ingredients (carbon and inorganic Hg) within MFR soil to support the magnitude and duration of MeHg flux predicted by Calder et al., MFR cannot generate the mass of MeHg necessary to support the increase in MeHg in the biotic food web of Lake Melville as predicted by Calder et al. (2016).

To pursue this further, we used a literature based (Bundy et al. 2000) Ecopath model and estimated the steady-state biomass across all trophic levels in Lake Melville as 272 tonnes/km^2. Then, using empirical and literature-based biota MeHg concentration data, we determined that the total mass of MeHg within the biotic food web of Lake Melville is approximately **20 kg**. This far exceeds the maximum mass of MeHg that can be generated from the MFR (**2.35 kg**) amortized over a decade. We then used the Calder et al. Bioaccumulation Factor (BAF) approach to estimate the mass of MeHg that is required to load this biomass according to the rate predicted by Calder et al. (2016). This amounted to >50 kg of MeHg. To achieve the predicted increase in MeHg in upper trophic level biota (fish, seals), MFR would have to generate *hundreds of kg of MeHg* amortized over a period of at least a decade.

When viewed from a top-down, mass-balance perspective, the assumptions and findings of Calder et al. (2016) are not supported. The MFR cannot generate a portion of the mass of MeHg predicted by Calder et al. in a single year, let alone over a decade. We argue that Calder et al. have *greatly* overestimated the potential for the MFR to generate MeHg and by extension cannot burden the aquatic food web of Lake Melville with MeHg.

2 Background

2.1 Calder et al. Predictions

Calder et al. (2016) predicted that when the MFR (101 km^2) is fully inundated, decomposition of organic matter by mercury methylating microbes will generate and sustain a *mean flux rate of 664 ng/m^2/day* (nanograms or parts per trillion per m^2) of dissolved MeHg to the overlying water column of the reservoir. According to Calder et al., this flux will cause *"the annual flow-weighted mean MeHg concentration in the Churchill River to increase 10-fold … relative to baseline"* and that "*these changes represent substantial increase in the freshwater environment that will be magnified in local food webs*". Calder et al. further state that "*Modeled MeHg concentrations in the top 20 local foods contributing to Inuit MeHg exposure range from 1.3 to 10 times measured baseline concentrations*". That is, a maximum of 10x in fully obligate freshwater organisms (consistent with the bioaccumulation factor [BAF] approach used by Calder et al.), and proportionately less in organisms that consume relatively more food from the marine food web of Lake Melville, at least as far as Rigolet.

For this magnitude of change to occur, especially in higher trophic levels, the flux rate of dissolved MeHg from the sediment to surface water of the Lower Churchill River (LCR) and Lake Melville must be sustained for a period of many years. In reservoirs, this has been well documented, where peak fish MeHg concentrations are realized between 6 and 10 years after inundation (Schetagne et al. 2003, Bodaly et al., 2007 and others). Calder has also acknowledged this stating "*This analysis assumes* ***steady-state*** *biological MeHg concentrations with peak MeHg fluxes from the reservoir. Data from previously flooded environments indicates that up to* ***ten years*** *are required for biota to reach maximum MeHg levels'*. Clearly, a one or two-year pulse of MeHg in water from Muskrat Falls reservoir would be insufficient to produce the food-web mediated downstream effect predicted in Table 4.1 of Calder et al. Lake Melville

The Science Document (Durkalec et al. 2016) states that L Melville is "*a dynamic environment that supports notably high productivity and species diversity, and has been identified as an Ecologically and Biologically Significant Area by the Canadian Science Advisory Secretariat (2013). This diversity includes freshwater fish species such as lake whitefish, longnose and white suckers and diadromous fish … such as brook trout and rainbow smelt… The lake supports the largest concentrations of surf scoter, a large sea duck; is an important ring seal overwintering and breeding area and harbour seal habitat; and is a feeding area for marine mammals such as dolphins, humpback whales, minke whales, and harp seals."* Clearly, Goose Bay and Lake Melville is an important habitat area to many species – and to local community residents who harvest country foods.

Using the BAF approach Calder et al. predict that "*mean MeHg concentrations in L. Melville surface waters will increase 2.6-fold following flooding*" with greater amounts in some animals (e.g., 5x baseline in seals) and less in others (2.6x in cod), mediated via the food web. Calder et al. further assume that the magnitude of increase in body burden MeHg is prorated, based on the relative amount of time a particular species or group of species spends feeding or acquiring energy (and MeHg) in the estuary.

However, it is important to note that a portion of the MeHg delivered to the estuary is eventually dispersed to all marine biota of Lake Melville; you can't cherry pick where it will end up. As well, another and perhaps very large portion will be demethylated, sequestered by particles or be lost in tidal exchange. These factors were not quantitatively addressed by Calder et al. and we do not address them either. Partitioning of MeHg into the marine environment is necessary to support the increase in MeHg of obligate marine organisms such as Arctic cod and rock cod as predicted. These animals are, in turn, preyed upon by seabirds, seals and many other marine organisms (Scott and Scott 1988).

Given the implications of MFR to Lake Melville and its biota – understanding how MeHg generated within the reservoir is delivered to and becomes accumulated within the complex food web of Lake Melville is important.

2.2 Biota MeHg Bioaccumulation

It is well known that MeHg is accumulated and concentrated into biota over time via a dietary pathway (e.g., Hall et al. 1997 and many others). Methylmercury generated within bacterial tissue and in pore water as a by-product of decomposition of organic matter (Heyes et al. 2000) is incorporated into the lowest rung of the food web, available to be accumulated in tissues of higher consumers. Elevated concentrations of MeHg in pore water of flooded sediments are absorbed by benthic infauna and/or fluxed to the overlying water column. This Hg methylation process is especially important during early stages of reservoir creation, when MeHg is initially present at much higher concentrations in porewater than overlying surface water and is the mechanism driving MeHg flux. This process diminishes over time as carbon as the fuel source, becomes exhausted (Kelly et al. 1997, Ravichandran 2004, Hall et al. 2005).

Once in surface water, MeHg partitions to abiotic media (adsorbed to sediment particles and organic matter; OM; Mierle and Ingram 1991, Choi et al. 1998) and biotic media (absorbed by phytoplankton and very small plankters; Mason et al. 1995, Pickhardt et al. 2003). Higher trophic-level organisms such as insects and fish absorb relatively little MeHg directly from water (~10%; Hall et al. 1997, Mason et al. 1995), given that MeHg is at least 1 billion times more concentrated in fish (e.g., 0.1 mg/kg) than water (<0.1 ng/L).

Thus, abiotic and biotic media leaving MFR, enriched in MeHg, travel 40 km downstream reaching the near-shore estuarine environment of Goose Bay and then Lake Melville. Lake Melville is permanently stratified with an approximately 10 m thick ‘lens’ of fresh/brackish water (Schartup et al. 2015, Durkalec et al. 2017), mostly from the Churchill River (about 70% - 75% of all freshwater inputs; Bobbit and Aikinhead 1982, Kamula 2015) that extends across the estuary. This lens is thickest and most consolidated near the river mouth at Goose Bay and becomes thinner and more laterally dispersed with increasing salinity (and diminishing MeHg concentrations) as the surface water is mixed and diluted into deeper marine waters moving eastwards. Calder et al. state that “*freshwater inputs from the Churchill River … concentrates riverine inputs within a relatively small volume … that is most important for biological productivity, facilitating uptake at the base of the estuarine food web*”.

3 Line of Evidence 1 – Comparison of Physical and Chemical Conditions to Other Reservoirs.

In this Section, we argue that Hg and MeHg concentrations and ancillary parameters in environmental media of the Lower Churchill River are low, and that MFR shares the same chemical, physical and ecological features of other existing reservoirs where within-reservoir peak Hg concentrations have been low (<3x baseline).

3.1 Baseline Water Data

In water, Nalcor has collected more than one-year (October 2016 – October 2017) of near weekly data on key parameters, total Hg, MeHg (total and filtered), TOC/DOC, TSS, pH and nutrients. Water quality data were recently summarized by Azimuth (2017a). Key findings are:

- No particular patterns for total Hg were evident given the relatively high MDL of 1.9 ng/L prior to May 2017. However, since Flett Research Ltd., Winnipeg, MB took over this analysis, total Hg averaged 1.6 ng/L from June to October 2017 at N1, the station upstream of the ponded area of the MFR at full supply. This concentration is low and typical of pristine systems (St. Louis et al. 2004, Driscoll et al. 2007, Bodaly et al. 2004, Krabbenhoft et al. 2007) and no different from values reported by Schartup et al. (2015).

- At N1, total and dissolved MeHg were higher in summer (0.020 – 0.025 ng/L) than winter (0.013 / 0.018 ng/L). Again, these are low values, typical of remote, pristine systems (Watras et al. 1995, Driscoll et al. 2007). The ratio of methyl to total Hg was 1 – 2%, a ratio characteristic of weak net methylation (in Ullrich et al. 2011).
- Total and dissolved organic carbon (TOC/DOC) concentrations were also higher in summer months (6.2 / 5.7 mg/L) than winter (5.1 / 4.6 mg/L), with the ratio of DOC to TOC being >90%. These are low values, typical of oligotrophic conditions (Wetzel 2001).
- Ancillary parameters in river water were characteristic of nutrient poor, highly oligotrophic conditions (Wetzel 2001) including conductivity (20 µS/cm), nitrogen nutrients and phosphorus (below MDLs) and total dissolved solids (<10 mg/L).
- Water pH was circum-neutral year-round (7.0 in winter, 7.14 in summer). Much research confirms that lower pH (≤ 6.5) is positively correlated with Hg methylation potential (Miskimmin et al. 1992, Branfierun et al. 1999, Kelly et al. 2003). Water pH of the LCR is not associated with strong methylating conditions.
- Downstream in Goose Bay (N8) total and dissolved MeHg concentrations were low and just above the MDL in winter (November to May; 0.013 / 0.011 ng/L). and summer (0.020 / 0.015 ng/L). TOC/DOC concentrations were also low year-round (4.0 / 3.7 mg/L) with a mean pH of 7.6.
- In Lake Melville at N12 and N13 (the most easterly station), total Hg averaged 0.83 and 0.66 ng/L respectively. MeHg concentrations were almost always below the MDL of 0.01 ng/L in both surface and deeper waters. TOC / DOC concentrations were always low (3.4 / 3.0). The pH was typical of marine waters at 7.9.

3.2 Baseline Soil Data

Forty-one soil samples collected by AMEC (2017a) from across the MRF, stratified by habitat type, were analysed for TOC, total Hg and a subsample for MeHg. These data were reviewed by Azimuth (2017b) with the following conclusions:

- Six of the soil samples were classified as ‘wetlands’, although two of these did not have ‘wetland’ soil characteristics (i.e., shallow depth, low TOC). The remaining four samples had an average soil dept of 15 cm, mean TOC of 38% and total Hg concentration of 0.05 mg/kg – which is half as much as all other forested stations, so this ‘wetland’ classification is somewhat doubtful.
- Four stations on ‘gravel bars’ were not sampled, having no vegetation whatsoever. Three samples were classified as being from ‘riparian’ habitat.
- Riparian areas are periodically inundated and may have standing vegetation and may have a litter layer, but no humic soil. Riparian areas had no humic layer, low TOC (0.7 – 7%) and low Hg (<0.010 mg/kg).
- Samples at the remaining 35 stations were comprised of soil from black spruce/feathermoss, black spruce lichen, fir-white spruce, hardwood or mixed

wood forests. These had an area weighted humic soil horizon thickness of 8 cm, mean TOC of 30.1% and mean inorganic Hg concentration of 0.10 mg/kg.

The total area of the MFR at 39 m asl (full capacity) is approximately 100.5 km^2. According to AMEC (2017a; **Table 1**) the majority of this area is original river area (56.9 km^2), with small amounts of gravel bar (6.9 km^2) and riparian area with no organic or humic horizon (6.6 k m^2). This leaves a total area of 30.1 k m^2 of flooded wetland and forested soils with an established humic soil layer – which is an approximately 50% increase in terrestrial habitat flooded, relative to original wetted surface area. Thirty km^2 is a lower value than was conservatively assumed by Calder et al., who indicated that what wasn't water (60 km^2), was forested (41 km^2) and contained humic soils. Thus, from this point forward, all calculations of mass of carbon and Hg in MFR will be based on a 30 km^2 area that has an established humic soil horizon.

3.3 Baseline Sediment Data

A total of 159 sediment samples were gathered from the LCR at stations N1 – N7, Goose Bay (N8) and Lake Melville (N10 – N13) between October 2016 and October 2017. Total Hg concentration measured in 138 sediment samples, including in Lake Melville were all below the MDL of 0.05 mg/kg. Riverine sediments were quite sandy (J. McCarthy, personal communication), which explains this result; however, all estuarine / marine sediments containing silt/clay were also all below detection. Of course, Hg is present, but it is in very small quantities in the river and estuarine / marine sediment. By comparison, total Hg in fine sediments (silt/clay and fine sand) in the Peace River within the Site C floodplain ranged from 0.03 mg/kg to 0.17 mg/kg (Azimuth 2011). It's reasonable to assume that concentrations in the LCR would be similar in fine grain size material – however, fines make up a small fraction in the bedload of the river, by mass.

While TOC was not measured in sediments of the LCR, TOC concentrations are typically much lower in sediment than soil. For example, in the Peace River sediment at Site C, TOC was low (1.4 to 2.1%). We could assume similar values in the LCR.

Methylmercury was analysed from all 159 sediment samples by Flett Research, Winnipeg. All samples were below the low MDL of 0.4 µg/kg, including in Lake Melville where results would not be confounded by coarse grain size.

3.4 Canadian Reservoirs Comparison Matrix

In 2010 – 2012 Azimuth (R. Baker, Dr. R.R. Turner) and co-authors Dr. W. Jansen (North/South Consultants) and Dr. R.A. Bodaly (Department of Fisheries and Oceans, retired) compiled the Canadian Reservoirs Comparison Matrix (CRCM) as part of the Site C Environmental Impact Assessment (Volume 2 Appendix J Mercury Technical Reports, Part 1 Mercury Technical Synthesis Report; Azimuth 2012).

The CRCM reviewed key empirical physical, chemical, and ecological parameters that are positively associated with mercury methylation rates, based on what was observed in 15 Canadian reservoirs. An extensive literature review supported the analyses (in Azimuth 2012 and available upon request). How these parameters ultimately influence fish Hg concentrations were contrasted against baseline and predicted conditions within the Site C reservoir, to provide insight into where Site C 'fits' within the spectrum of reservoir types. An advantage of this approach is that it relies on real, empirical data from a range of reservoir types across Canada, to provide insights into those factors that are most strongly associated with large peak fish Hg concentrations, relative to baseline or reference lakes.

Seven Manitoba reservoirs (Keeyask, Limestone, Long Spruce, Notigi, Southern Indian Lake, Stephens, and Wuskwatim), five Quebec reservoirs (Caniapiscau, LG1, LG2 [Robert Bourassa], LG3, and Opinaca), Williston Reservoir (BC) and Gull Island and Muskrat Falls in Labrador were compared. This exercise was undertaken without knowing anything about MFR except what was available in publications at the time.

In the CRCM, how a reservoir aligned with key physical, chemical and ecological parameters very strongly determined whether fish Hg concentrations would ultimately achieve either 'low' (≤3 x) or 'high' (≥3 x) values relative to baseline or nearby reference lakes. The value of 3x baseline was chosen as a cutoff, which is about half the increase in most 'worst-case' scenario increase reservoirs (i.e., 6–7x baseline). A 3x increase factor is conservative, yet high enough that it is readily distinguishable from baseline, and the return to baseline can be measured with precision (Appendix V2J Part 1).

Based on the literature, the CRCM identified the most important physical factors associated with enhanced mercury methylation as:

- Total reservoir area – Larger reservoirs (>200 km^2) produce higher peak fish Hg concentrations and take longer to return to baseline or background (relative to nearby lakes). This is related to having a large pool of organic soils (and Hg). At MFR, the total reservoir area is 101 km^2 of which 30% is flooded organic soil.
- Ratio of total reservoir area to original wetted surface – Peak fish Hg concentrations were ≤3x baseline in reservoirs with a flooded area <3x greater than original surface area. At MFR, the increase is 1.5x greater than baseline.
- Water residence time – Peak fish Hg increase in reservoirs with short residence time (≤ 30 days) was ≤3x baseline and took less time to return to near baseline or regional levels. Reservoirs with longer residence time (months to 1.5 years) had higher peak fish Hg concentrations that persisted for a longer period of time. At Site C, residence time is 22 days, while at MFR, residence time is only 10.6 days.

The most important chemical factors are:

- Slightly acidic water (pH <6.5) is consistently and positively correlated with higher fish Hg concentrations than reservoirs of pH 7.0 or greater. MFR has a pH of 7.1.
- Total or dissolved organic carbon (TOC/DOC) concentrations in water >5 mg/L are weakly but positively correlated with the magnitude of increase in fish Hg.
- Large stores of labile or easily degradable carbon within the reservoirs has been found to be a key contributor to elevated and prolonged mercury methylation rates.

The most important ecological factors are:

- Lower trophic level Hg concentration – Lakes/rivers with higher baseline MeHg concentrations in benthos (reflecting efficient baseline methylating conditions) result in higher MeHg increases post-flood, which persist for a longer period.

- Reservoir productivity – Larger reservoirs (like lakes) with more *in situ* and nutrient inputs from upstream and/or tributaries, have greater biomass and higher sustained Hg methylation rates and consequently, higher MeHg concentrations in biota. High methylation in large reservoirs overcomes the ‘growth dilution’ phenomenon (e.g., Kidd et al. 1995) because of the high mass of MeHg generated early in reservoir life. Also, lake-like reservoirs have established zooplankton populations, adding a trophic level, that run-of-river reservoirs tend not to have.

When site-specific empirical data for Site C and MFR were compared to each chemical, physical or ecological parameter, ***all*** metrics clearly placed both reservoirs into the ‘low’ increase category at ≤3 x baseline (**Table 1** taken from Azimuth 2012).

Summary – Site C and MFR are very closely related. Physically, both are downstream of two of the world’s largest and old (>45 y) reservoirs (which act as sinks), are run-of-river reservoirs with low amplitude elevation change (<2 m), have a relatively small amount of flooded area relative to reservoir size and short water residence time. Chemically, both have low baseline Hg / MeHg concentrations in abiotic and biotic media, are nutrient poor, circumneutral in pH, have low DOC and limited tributary inputs of allochthonous carbon.

During the course of the Site C 2012 EIA, MFR was firmly placed within the low increase category, similar to Site C. We cannot find a single empirical physical or chemical metric where MFR and Site C substantively differed**.**

It is worth noting that among the 15 Canadian reservoirs examined by Calder et al., as being planned or under construction, they *also* placed Site C into the lowest increase category among all reservoirs examined, based on a forecast peak water MeHg concentration of 0.04 ng/L. Their forecast peak MeHg concentration at MFR is 0.19 ng/L, nearly 5x higher than at Site C. There is no rationale presented for this large difference in water concentration – and by extension, the much higher peak fish Hg concentration forecast at MFR. The data from MFR, weighed by the CRCM clearly place this reservoir into the same low increase category as Site C. In light of this, *we see no reason to place MFR and Site C on the opposite ends of the spectrum of possibilities*.

4 Line of Evidence 2: Top-Down, Mass-Balance Approach

The key premise of the Calder et al. (2016) paper is that the MFR is capable of generating and sustaining a flux rate of 664 ng/m^2/d of MeHg, requiring a sustained load over a period of up to 10 years to achieve this new “*steady state equilibrium*” (Calder et al. 2016) in biota of Lake Melville. This includes a 10x increase above baseline in obligate freshwater fish, up to 5x baseline in seals (that may split their time feeding on freshwater versus marine biota) and up to 2.6x in obligate marine species, such as Arctic cod. This assumes a ‘bottom-up’ approach, using BAFs, where MeHg in higher level trophic biota (invertebrates, fish) will eventually and necessarily equilibrate to reflect higher MeHg concentrations in water.

For this to occur, two critical assumptions must be satisfied: 1) there must exist a sufficient supply of organic carbon and Hg to sustain the Hg methylation flux rate; and 2) the load of MeHg generated and delivered downstream must be significantly greater than the mass of MeHg in biota currently residing in Lake Melville.

Table 1. Summary table from Azimuth (2012) – Canadian Reservoirs Comparison Matrix – Site C.

Reservoir Characteristics	**Low Magnitude Increase Reservoirs (Fish Mercury <3x Baseline)**	**High Magnitude Increase Reservoirs (Fish Mercury >3x 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 (MB) for some fish species	LG-1, LG-2, LG-3, Opinaca, Caniapiscau Quebec; Southern Indian Lake, MB (for some species) Williston, B.C.	
Physical Parameters			
Total Reservoir Area	Less than 200 $km^{2,}$ ranging from 28 (Limestone) – 200 km^2 (Muskrat / Gull Island) for all reservoirs	Very large, with most exceeding 2,000 km^2 except Opinaca (1,040 km^2), Williston (1,779 km^2)	Site C predicted area = 93 km^2 and falls into LOW increase category
Original: Flooded Area	Less than 2 at Muskrat (1.5) and Gull (1.7) Nfld/Lab and Limestone (1.3), Long Spruce (1.9), and Wuskwatim, MB (1.5)	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), and Long Spruce (10 d)	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); 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 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/m^2) 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	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

Reservoir Characteristics	Low Magnitude Increase Reservoirs (Fish Mercury <3x Baseline)	High Magnitude Increase Reservoirs (Fish Mercury >3x Baseline)	Predicted Site C Result
		kg/m^2 in peat soils, 9 – 42 kg/m^2 in wetlands and 7 kg/m^2 in forest soil	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. 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. Percentage 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.

NOTES:

THg = total mercury; MeHg = methylmercury; dw = dry weight; MB = Manitoba, PQ = Quebec; SIL = Southern Indian Lake (MB)

Stepping back, we have turned the problem around and posed the following questions, taking a mass-balance, top-down perspective:

1. *What is the mass of organic carbon and inorganic mercury within the MFR?*
2. *Is this mass of carbon and inorganic Hg within MFR sufficient to sustain the Calder et al. forecast flux rate?*
3. *How does this annual mass load of MeHg compare to the existing pool of MeHg within the Lake Melville food-web?*

In this section we demonstrate that the supply of OM and Hg is quite limited and cannot generate or sustain the flux of MeHg that Calder et al. have forecast. This in turn has important implications on the potential to supply MeHg to the downstream environment.

4.1 Key Assumptions

It has been well established that the most important 'raw materials' in the Hg methylation process are organic carbon as a nutrient source for sulphate-reducing bacteria and the mass of inorganic mercury that has been sequestered by plants and stored in soils (Compeau and Bartha 1985, Hall et al. 2005, Ullrich 2011, Paranjape and Hall 2017). Both are required for mercury methylation. Sustaining elevated rates in new reservoirs also depends on inputs of 'fresh' organic matter (OM) that also contain inorganic Hg.

Organic matter is present in new reservoirs in above ground, living vegetation (leaves, needles) and in decomposing organic material in the litter, fermentation and humic (LFH) layers of forest soils. While the mass of above ground OM may seem high, the concentration of inorganic Hg in living, easily decomposable vegetation (i.e., not bole wood) and the litter/fungal layer is actually quite small (Hall et al. 2005). Azimuth (2017c) recently reviewed the literature on this subject and demonstrated that the combined pool of Hg (kg/ha) of all above ground vegetation components (trunks, branches, leaves and needles) accounted for about 1 – 3% of the **total Hg pool** in all ecosystem components. The organic humic soil horizon and decomposing fermentation layer contained the remainder of the mercury pool (97 – 99%) with most of this in humus (>90%).

It should be noted that fresh litter provides enhanced stimulation of bacteria in the early months of reservoir creation by contributing an easily decomposable, labile carbon source. However, this nutrient source is ephemeral and 'burns out' relatively quickly in the evolution of the reservoir; it is the humic layer that provides the long-term OM supply. Azimuth (2017d) analysed the labile content in humic soil from MFR and found that <1% of the humic soil was 'labile' or easily degradable – which is typical of boreal soils.

It is important to understand that not all of the carbon in soil is easily decomposed, nor is all of the inorganic Hg within the column of flooded organic soils vulnerable to methylation. While there is wide acknowledgment in the literature that continuous cycling of Hg methylation and demethylation occurs within the sediment column, especially in newly flooded soil (Driscoll et al. 1995, Hall et al. 1995, Pak and Bartha 1998), much of what is methylated or demethylated remains sequestered in soil (St Louis et al. 1996, Benoit 2002, Rolfhus 2015). In fact, only a fraction of the inorganic mass of Hg that is methylated in surficial sediment is fluxed away from the sediment and 'escapes' to eventually become incorporated into the aquatic food web and/or discharged downstream (Korthals and Winfrey 1987, Boening 2000, Kainz et al. 2011).

This section briefly examines:

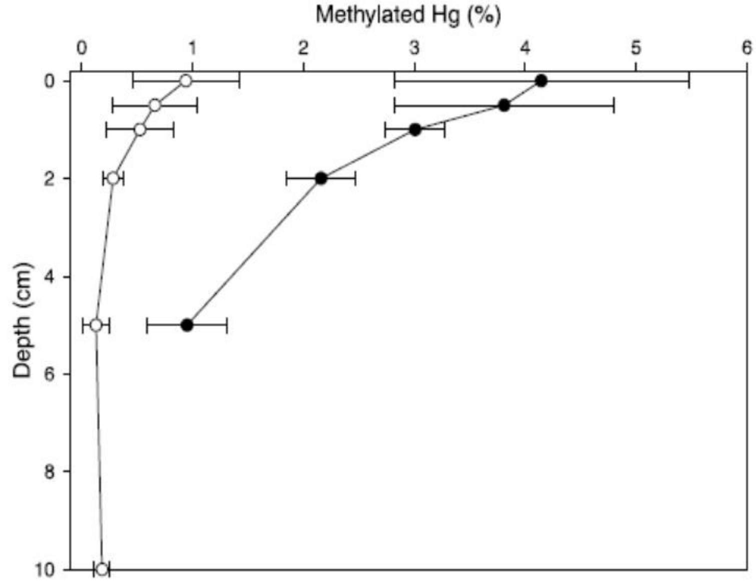
1. The depth (cm) of flooded soil where MeHg is generated and available in porewater, and fluxed to the overlying water column where it is available to aquatic biota; and
2. The proportion (%) of the pool of inorganic Hg in the depth in #1 that is converted to MeHg, and made available for uptake by the aquatic biota.

An understanding of both factors is critical to determining the mass of MeHg that can be generated and made available to the aquatic food web in the reservoir, LCR and eventually, Lake Melville.

Mercury methylation may occur throughout the organic / humic layer of upland forest soil (0 – 15 cm) or deeper in wetlands. A great deal of internal cycling between methylated and demethylated forms occurs, depending on redox conditions, quality of organic material, oxygen, bioturbation, sulphate and other factors. Paranjape and Hall (2017) have recently published an excellent summary paper describing Hg/MeHg production, cycling and dynamics (see p. 92). They review a number of studies that confirm that sediment and porewater are the key sources of MeHg and that elucidate how methylation potential changes with increasing depth within sediment columns. For example, higher MeHg concentrations were consistently observed in *surficial* sediments (i.e., top few cm) in mudflats (Ouddane et al. 2008), lagoons (Monperrus et al. 2007), peatland porewaters (Selvendrian et al. 2008) and estuarine sediment (Liu et al. 2015). These and other studies confirm that MeHg available to be fluxed to surface waters occurs primarily in the upper layers (0 – 3 cm) of sediment, where microbial activity is greatest (Rudd et al. 1983, Korthals and Winfrey 1987, Eckley and Hintlemann 2006).

For example, Kainz et al (2011) found highest MeHg concentrations at the sediment–water interface, with higher concentrations at littoral sites than at offshore sites in Lake Lusignan, Quebec. Littoral sediments contained more terrestrial and bacterial organic matter than offshore sediments, perhaps reflecting nearby allochthonous inputs. The figure below from the Kainz et al. report depicts the logarithmic decline in MeHg from surficial to deeper sediments. Grondin et al. (1995), Korthals and Winfrey (1997), Eckley et al. (2015) and others have observed similar patterns.

Fig. 4. Relative amount (%) of methylated Hg in the offshore (open symbols) and littoral (solid symbols) sediment cores. Data are mean values with standard deviation bars.



While it is generally acknowledged that only a small portion of the MeHg generated from organic Hg in flooded soils is 'lost' from sediments to be bioaccumulated by biota, few studies have examined this directly. This is simply because such a small portion of the pool of Hg in flooded soils is methylated and fluxed away, that pre- and post-flood inorganic Hg concentrations are indistinguishable from one another. The above figure also supports this, as only 1 – 4% of inorganic Hg is present in the methyl form.

The studies that *have* examined this are categorical however, suggesting a small (≤5%) loss of Hg over at least a decade. Grondin et al. (1995) examined Hg profiles in flooded podzols in unflooded lakes and at La-Grande 2 Reservoir in Quebec13 years after flooding. Both lakes and LG-2 had similar lead (Pb) and Hg profiles that were uniform over the entire depth of the core, with "*average concentrations of C and Hg, comparable to those in pristine podzols*." Following impoundment, Grondin et al. (1995) stated that Hg burdens of flooded wetland soils remain almost intact and that "*in the case of flooded peat soils, no significant physical changes in the Hg and Pb profiles could be detected following inundation*." They concluded that "*Upon inundation, soils in reservoirs support intense bacterial activity ... redistribution of nutrients, the production of CH_4 and CO_2 and the methylation of Hg... If direct release of Hg from flooded soils occurs, it is not evidenced by a marked decrease in the initial burden of Hg in the organic horizon*. *This study suggests that the initial reserves of Hg in the LG-2 reservoir have been only slightly depleted after 11 to 13 years of impoundment*".

Mucci et al. (1995) reported a similar result from LG-2. They found that organic carbon and nitrogen content of flooded soils remained high even after 14 years of impoundment, possibly because microbial degradation of terrestrial organic matter is slow at northern latitudes. They also concluded that "*the organic horizon of submerged soil, unaffected by erosion, remains enriched in Hg, indicating that chemical remobilization of the metal to the overlying waters does not deplete its Hg burden significantly*".

At the FLUDEX site at ELA Hall et al. (2005) also showed that while a great deal of Hg methylation occurred in newly flooded soils, most of the MeHg remained sequestered there. They stated that "*The majority of MeHg produced in soils and peat and was not transferred to the water column. Our research indicates that, unless other processes that enhance the movement of MeHg associated with flooded soils and peat particles to the water column are present (for example, erosion see Louchouarn and others 1993), flooding wetlands may not necessarily result in a worse-case scenario for MeHg contamination of reservoir fisheries because the majority of MeHg produced in the soils remains there and does not enter the water column and, thus, the food web*".

In summary, MeHg generated within the top 0 – 3 cm of flooded soils is most vulnerable to being fluxed from the sediment to the overlying water column, where it is available to be accumulated by biota. While Hg methylation may occur at deeper depths in flooded soils, it appears that most of the MeHg is sequestered and/or demethylated there and does not appear to migrate to the surface. Furthermore, a limited proportion (likely ≤5%) of the mass of inorganic mercury in organic soils within the top few cm is methylated, fluxed to surface or upper porewaters and *absorbed into the aquatic food web* and/or transported downstream. This has clear and important implications regarding the mass of inorganic Hg that is ultimately available and accumulated by biota as MeHg.

4.2 Mass Balance Approach for Carbon and Hg/MeHg in MFR

Using empirical soil data from AMEC (2017a), we calculated the total soil and carbon biomass from the forested area of MFR. This was based on the area (ha) of each Ecotype (e.g., black spruce/feathermoss, mixed hardwood, etc.), weighted by mean organic soil horizon depth multiplied by soil density (Pierie and Ouimet 2008). Then, the total mass of inorganic Hg was calculated from the total mass of soil within MFR using the mean Hg concentration prorated by Ecotype.

The calculations are as follows:

- Of the 41 km^2 total flooded area of terrestrial habitat within the MFR, only 30 km^2 consists of forested terrain with an established soil horizon.
- Mean depth of the humic layer is **8.0 cm**. The total mass of humic soil within the MFR (30 km^2) is conservatively estimated at **726,000 tonnes**, prorated by Ecotype (different soil thickness, area) (AMEC 2017a).
- Average TOC content was approximately **35%** of the humic soil layer.
- Although MeHg generated within the top 3 cm is generally considered available to be fluxed and bioaccumulated, we have conservatively assumed that the top 5 cm is 'vulnerable', giving an available mass of **453,000 tonnes** of soil.
- Thus, there is approximately 158,000 tonnes of OM in the upper 5 cm. This is approximately half of the annual load of OM that is transported *annually* by the LCR to Goose Bay / Lake Melville (305,000 tonnes).
- The concentration of inorganic Hg in humic soils averaged **0.10 mg/kg**.
- Assuming a 0.1 mg/kg Hg concentration, the total mass of inorganic mercury is **45 kg** in the top 5 cm.
- According to the available literature, only 2 – 3% of the total inorganic Hg pool is vulnerable to be methylated, fluxed and accumulated by biota. However,

because this has not been well studied, we have conservatively assumed that 5% of the total Hg pool is available over the first 10 y after reservoir creation.

Based on a 5% conversion rate of Hg, a total mass of **2.25 kg of MeHg** can be generated by MFR for a period of up to 10 years. Thus, *no greater mass than this* is ultimately available to the MFR and downstream environment of Goose Bay and Lake Melville. It is also important to note that delivery of this total mass is amortized over a period of at least 5 and possibly 10 years, with higher rates in the first 2-3 years than afterwards (Hall et al. 2005 and others). Thus, the probable maximum annual mass of MeHg delivered to the food web is no more than 0.5 kg/y.

4.3 Implications of Available Hg Mass on Assumed Flux Rates

Calder et al. (2016) assumed a sustained *annual* flux rate of 664 ng/m^2 over the entire 41 km^2 of flooded terrestrial terrain. This amounts to **10.5 kg of MeHg** annually. This is almost 10x the existing load of MeHg (**1.2 kg/y**) carried by the Lower Churchill River – so it is a very significant change from baseline which should easily be detectable in the water quality monitoring program (Azimuth 2017a).

Scaling this back to assume a flooded area of 30 km^2 to be consistent with the actual area with organic soils, equals a mass of **7.7 kg of MeHg** (i.e., 10.5 * (30/41)). This is the mass of MeHg that Calder et al. assume the MFR *must* generate *every year*. This value is perhaps up to an order of magnitude higher than the mass of MeHg that can be generated by MFR within a single year. There is an insufficient supply of available Hg contained within MFR soil to generate a fraction of one year's supply of MeHg at the assumed flux rate of 664 ng/m^2.

Based on this line of evidence, strongly supported by the literature, the MFR *cannot* support the predicted loading rate of 7.7 kg of MeHg within a *single year*, let alone over a period perhaps lasting up to 10 years.

This has critical implications on the ability of the MFR to generate sufficient MeHg to alter the existing load of MeHg contained within the food web of Lake Melville. This is true notwithstanding whatever demethylation, or partitioning of MeHg to a wide variety of media (e.g., periphyton, TOC, DOC, TSS, plankton, etc.) that occurs along the way, which has not been quantified.

4.4 MeHg Mass in Lake Melville Biota

Assuming that there is a mass of **2.25 kg** of MeHg available to be delivered to Lake Melville over a period of up to a decade, the next question to ask is 'how does this mass compare with the mass currently contained in biota that reside in this environment?'

To address this question, we first conducted an extensive literature review to identify regional information on biomass in marine ecosystems. We identified one key study by Bundy et al. (2000; DFO Science Branch and Bedford Institute of Oceanography) entitled '*A mass-balance of the Newfoundland-Labrador Shelf*' that constructed a mass balance using the Ecopath model. Ecopath is a top-down, ecosystem energetics model that looks across extremely wide, ecologically relevant trophic levels to characterize the entire ecosystem. Although the model is for the continental shelf, rather than a nearly-enclosed estuarine embayment like Lake Melville, the latter is acknowledged to support a notably high productivity and species diversity (Schartup et al. 2016, Durkalec et al.

2016). Thus, the biomass estimate derived from the Ecopath model for the Labrador shelf, is considered a reasonable but conservative (i.e., low) estimate for Lake Melville.

Bundy et al. (2000) synthesized information on biomass, consumption, production and diet of major species or species groups spanning the entire ecosystem to estimate a steady-state scenario. This gave us an estimate of the areal biomass (kg/km^2) of marine organisms present in Lake Melville. We accounted for phytoplankton, small and large zooplankters, key benthic organisms (mussels, echinoderms), selected marine fish (smelt, sand lance, plaice, flounder, Atlantic cod, rock cod, etc.), seabirds, and marine mammals (seals, but not whales). We also did not account for freshwater fish (e.g., brook trout) because of their low biomass and ephemeral time spent in the estuary (AMEC 2017b). Then, we took empirically measured Hg data (mg/kg) presented in Schartup et al (2016), Calder et al. (2016), AMEC (2017b), or from the literature and derived an estimate of the total MeHg mass (kg) present in the aquatic food web. Total mass (kg) of MeHg contained within aquatic organisms of Lake Melville was estimated for two scenarios:

1. Current, biomass tonnes/km^2 and mass of MeHg (kg) in Lake Melville under current steady-state, baseline conditions prior to flooding (the “baseline” scenario); and
2. Forecast steady-state mass of MeHg (kg) in Lake Melville under post-flood conditions using Calder’s BAF scenario (the “post-flood” scenario).

The difference between the total MeHg masses for the two scenarios represents the amount of additional (or “new”) burden of MeHg needed to achieve tissue concentrations using the BAF approach as predicted by Calder et al. for Lake Melville biota during future ‘steady state conditions’ following flooding of MFR. All details including methods, assumptions, results and uncertainty analysis are contained in **Appendix A**. Key results are summarized in **Table 1** of this appendix.

The total steady-state mass of biota across all trophic levels in Lake Melville is estimated at 272 tonnes/km^2 for the baseline and post-flood scenarios (i.e., biomass does not change, only the MeHg burden). This biomass estimate is similar to what has been reported from other similar marine coastal shelf and estuarine environments elsewhere. Although biomass estimates ranged from 57 t/km^2 (Hudson Bay) to 3786 t/km^2 (Iceland), the majority of values fell between 200 and 400 t/km^2. These results indicate that the use of the Bundy et al. (2000) biomass estimate for Lake Melville (3000 km^2 x 272 tonnes/km^2 = 816,000 tonnes) is likely conservative and that the actual biomass could be 2-fold to 5-fold higher.

Thus, using empirical and literature-derived Hg concentrations for all food web components, we determined a cumulative mass of **19.8 kg of MeHg** in the biotic component of the Lake Melville ecosystem under the pre-flood, baseline scenario. This mass is several times higher than the maximum mass that the MFR is capable of generating over its life, or at least the 5 to 10-year period when MeHg generation in flooded soil of MFR is elevated.

Then, we posed the question, “what would the post-flood maximum biomass of MeHg become in Lake Melville biota?”, using the BAF approach used by Calder et al. (2016). The answer is **50.8 kg** of MeHg, a difference of **31.1 kg**. Thirty-one kg is the mass of MeHg that would have to be loaded into the biota of Lake Melville to achieve the new, 'post-flood' steady-state concentrations that were forecast by Calder et al. (2016). It is also important to state that the processes of bioaccumulation and biomagnification to the highest trophic levels is not instantaneous. To accumulate 31 kg of 'new' MeHg, the *actual* MeHg production from the MFR would need to be *considerably higher* – given that it may take on the order of a decade of sustained production to reach this higher, steady-state condition in biota. Thus, perhaps up to *several hundred kg* of MeHg would have to be manufactured within MFR and delivered to Lake Melville to achieve the concentrations that were forecast. Obviously, from a mass-balance perspective, this simply cannot happen, as the 'demand' simply far outweighs the 'supply'.

Finally, the scientific literature suggests that much of the MeHg produced by the MFR and released to water, may not end up in biota. There are many partitioning mechanisms by which MeHg will be scavenged from water by a variety of processes after it leaves the reservoir, as spreads across Lake Melville. Much will be demethylated, adsorbed to sediment particles, DOC, or leave Lake Melville through tidal exchange. Although these processes are important, aside from acknowledging some demethylation, Calder et al. did not qualitatively address them. Because Calder et al. did not address them, neither have we, as this is beyond the scope of our lines of argument and as the results show, is not consequential to our findings.

5 Conclusion

Our key findings are as follows:

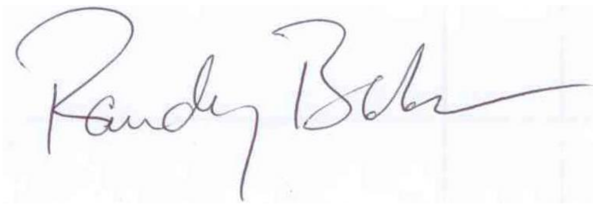
1. When comparing empirical data from MFR to many other Canadian reservoirs, using the CRCM, MFR clearly falls into the low-methylating category where a greater than 3x increase in fish mercury concentration above baseline is not expected;
2. The mass of MeHg that can be manufactured by MFR is on the order of 2 – 3 kg over period of up to 10 years. This mass is less than half a single year's supply of MeHg at the flux rate promulgated by Calder et al. (2016).
3. The mass of MeHg present in Lake Melville biota is conservatively estimated at 20 kg. This is nearly 10x higher than the mass of MeHg that can be generated by MFR over the course of a decade. Finally, in order to achieve the biota concentrations within key species within Lake Melville predicted by Calder et al. (2016) using the BAF approach, perhaps hundreds of kg of MeHg would have to manufactured within MFR and delivered to Lake Melville over time.

When viewed from a top-down, mass-balance perspective, the assumptions and findings of Calder et al. (2016) are not supported. We wish to be very clear that the potential for the MFR to burden the aquatic food web of Lake Melville with MeHg has been greatly over-estimated.

While we are not saying 'no change will occur' in Lake Melville, the evidence presented here strongly suggests that if any increase in MeHg burden were to occur, it would be extremely small and probably difficult to measure, given the lack of a strong pre-flood, baseline dataset of MeHg in lower trophic level biota in Lake Melville, where changes would be first observed (Hall et al. 1997).

Given the clear and unambiguous nature of our findings means that there is an urgent need to clarify the message to resource users and other residents of the local communities, that biota in Lake Melville will not be contaminated with MeHg generated by the MFR.

Randy Baker



M.Sc., R.P.Bio.

Incorporated Partner

References

AMEC 2017a. Muskrat Falls Soil Sampling Program 2016. A report prepared by AMEC Foster Wheeler, St. John's for Nalcor Energy, St. John's NL, June 26, 2017.

AMEC 2017b. Aquatic Effects Monitoring Program 1998 – 2016 Baseline Conditions Muskrat Falls. A report prepared by AMEC Foster Wheeler, St. John's for Nalcor Energy, St. John's NL, August, 2017.

Azimuth 2012. Site C Clean Energy Project – Mercury Technical Synthesis Report Chapter 5: Canadian Reservoirs Comparison Matrix. Volume 2, Appendix J, Part 1. Prepared by Azimuth Consulting Group Partnership, Vancouver BC for BC Hydro, Vancouver BC.

Azimuth 2017a. Relationship between MFR elevation and mercury concentrations, Lower Churchill River, October 2016 to September, 2017. Technical Memo prepared by R. Baker and Dr. R. Turner for Nalcor Energy, St. John's NL. November 6, 2017.

Azimuth 2017b. Review and quality control analysis of empirical soil mercury and organic carbon data from the Lower Churchill River. Technical Memo prepared by Dr. R. Turner and R. Baker for Nalcor Energy, St John's NL. April 9, 2017.

Azimuth 2017c. Contribution by flooded vegetation to mercury methylation in new reservoirs – A Review. Technical Memo prepared by Dr. R. Turner and R. Baker for Nalcor Energy, St John's NL. February 22, 2017.

Azimuth 2017d. Quantitative measurement of labile carbon in organic soils of the Lower Churchill River. Technical Memo prepared by Dr. R. Turner and R. Baker for Nalcor Energy, St John's NL. October 18, 2017.

Benoit, J.M., C.C. Gilmour, A. Heyes. R.P. Mason and C.L. Miller CL. 2002. Geochemical and biological controls over methylmercury production and degradation in aquatic ecosystems. In Biogeochemistry of environmentally important trace elements. Edited by Y Cai and OC Braids. American Chemical Society, Washington, D.C. pp. 262–297.

Bloom N.S., C.J. Watras and J.P. Hurley. 1991. Impact of acidification on the methylmercury cycle of remote seepage lakes. Water, Air, and Soil Pollution 56: 477–491.

Bobbitt, J. and Akenhead, S., 1982, Influence of controlled discharge from the Churchill River on the oceanography of Groswater Bay, Labrador: Research and Resource Services, Department of Fisheries and Oceans.

Bodaly, R. A., Beaty, K.G., Henzel, L.H., Majewski, A.R., Paterson, M.J., Rolfhus, K.R., Penn, A.F., St. Louis, V.L., Hall, B.D., Matthews, C.J.D., Cherewyk, K.A., Mailman, M., Hurley, J.P., Schiff, S.L. and Venkitesvaran, J.J. 2004. Experimenting with hydroelectric reservoirs. Environ. Sci. Tech. 38: 347A-352A.

Bodaly, R. A., W.A. Jansen, A.R. Majewski, R.J.P. Fudge, N.E. Strange, A.J. Derksen, and D.J. Green. 2007. Post-impoundment time course of increased mercury concentrations in fish in hydroelectric reservoirs of northern Manitoba, Canada. Arch. Environ. Contam. Toxicol. 53: 379-389.

Boening, D.W. 2000. Ecological effects, transport, and fate of mercury: a general review**.** Chemosphere. 40: 1335–1351.

Branfireun B.A., N.T. Roulet, C.A. Kelly and J.W.M. Rudd. 1999**.** In situ sulphate stimulation of mercury methylation in a boreal peatland: toward a link between acid rain and methylmercury contamination in remote environments. Global Biogeochemical Cycles. 13: 743–750.

Calder, R.S. A.T. Schartup, L. Miling, A.P. Valberg, P.H. Balcom and E.M. Sunderland. 2016. Future impacts of hydroelectric power development on methylmercury exposures on Canadian indigenous communities. Environ. Sci. Techol. 50: 13115 – 13122.

Compeau GC, and Bartha R. 1985. Sulfate-reducing bacteria: principal methylators of mercury in anoxic estuarine sediment. Applied and Environmental Microbiology. 50: 498–502.

Dmytriw, R. A. Mucci, M. Lucotte and P. Pichet. The partitioning of mercury in the solid component of dry and flooded forest soils and sediments from a hydroelectric reservoir, Quebec (Canada). Water, Air and Soil Poll. 80: 1099 – 1103.

Driscoll C.T., V. Blette, C. Yan, C.L. Schofield, R. Munson R and J. Holsapple. 1995. The role of dissolved organic carbon in the chemistry and bioavailability of mercury in remote Adirondack lakes. Water, Air, and Soil Pollution. 80: 499–508.

Driscoll, C. T., Y. Han, C.Y. Chen, D.C. Evers, K.F. Lambert, T.M. Holsen, N.C. Kamman and R.K. Munson. 2007. Mercury contamination in forest and freshwater ecosystems in the northeastern United States. BioScience. 57: 17-28.

Kelly CA, Rudd JWM, and Holoka MH. 2003. Effect of pH on mercury uptake in an aquatic bacterium: implications for Hg cycling. Environmental Science & Technology. 37: 2941–2946.

Durkalec, A., T. Sheldon, T. Bell (eds). 2016. Lake Melville: Avativut Kanuittailinnivut (Our Environment, Our Health). Scientific Report, Nain, NL. Nunatsiavut Government.

Eckley, C.S. and H. Hintelmann. 2006. Determination of mercury methylation potentials in the water column of lakes across Canada. Science of the Total Environment. 368: 111–125.

Grondin, A. M., M. Lucotte, A. Mucci and B. Fortin. 1995. Mercury and lead profiles and burdens in soils of Quebec (Canada) before and after flooding. Can J. Fish Aquat. Sci: 52: 2493 – 2506.

Hall, B.D., Bodaly, R.A., Fudge, R.J.P., Rudd, J.W.M., and Rosenberg, D.M. 1997. Food as the dominant pathway of methylmercury uptake by fish. Water Air Soil Pollut. 100: 13–24.

Hall, B.D., V.L. St. Louis, K. Rolfhus, R.A. Bodaly, K. Beaty, M.J. Paterson and K. Peech Cherewyk. 2005. Impacts of reservoir creation on the biogeochemical cycling of methylmercury and total mercury in boreal uplands forests. Ecosystems 8: 248 – 266.

Heyes, A., T.R. Moore, J.W. Rudd and J.J. Dugoua. 2000. Methyl mercury in pristine and impounded boreal peatlands, Experimental Lakes Area, Ontario. Can. J. Fish. Aquat. Sci 57: 2211 – 2222.

Hurley, J.P., J.M. Benoit, C.L. Babiarz, M.M. Shafer, A.W. Andren, J.R. Sullivan, R. Hammond and D.A. Webb. 1995. Influences of watershed characteristics on mercury levels in Wisconsin rivers. Environmental Science and Technology 29: 1867-1875.

Kainz, M., M. Lucotte and C.C. Parrish. Relationships between organic matter composition and methyl mercury content of offshore and carbon-rich littoral sediments in an oligotrophic lake. Can. J. Fish. Aquat. Sci. 60: 888 – 896.

Kamula, C.M. 2015. The sediment and particulate organic carbon cycle of Lake Melville, Labrador: a fjord estuary undergoing hydrologic and climatic change. A M.Sc. thesis submitted to the University of Manitoba, Department of Environment and Geography, Winnipeg MB.

Kelly C.A., J.M. Rudd, R.A. Bodaly, N.T. Roulet, V.L. St. Louis, and A. Heyes. 1997. Increases in fluxes of greenhouse gases and methyl mercury following flooding of an experimental reservoir. Environmental Science and Technology. 31: 1334–1344

Kirkwood, A.E., P. Chow-Fraser and G. Mierle. 1999. Seasonal levels in phytoplankton and their relationship with algal biomass in two dystrophic shield lakes. Environ. Tox. Chem.18: 523 – 532.

Korthals, E.T. and M.R. Winfrey. 1987. Seasonal and spatial variations in mercury methylation and demethylation in an oligotrophic lake. App. Environ. Microbiol. 53: 2397 – 2404.

Krabbenhoft, D.P., J.G. Wiener, W.G. Brumbaugh, M.L. Olson, J.F. DeWild and T.J. Sabin. A national pilot study of mercury contamination of aquatic ecosystems along multiple gradients; Morganwalp, D.W., H.T. Buxton, Eds.; 1999, U.S. Geological Survey Toxic Substances Hydrology Program--Proceedings of the Technical Meeting, Charleston, South Carolina, March 8-12, 1999--Volume 2--Contamination of Hydrologic Systems and Related Ecosystems: U.S. Geological Survey Water-Resources Investigations Report 99-4018B, 147-160 pp.

Krabbenhoft, D.P., D. Engstrom, C.C., Gilmour, R., Harris, J., Hurley and R. P. Mason. 2007. Monitoring and evaluating trends in sediment and water indicators. In: Harris, R., D.P. Krabbenhoft, R.P. Mason, M.W. Murray, R. Reash and T. Saltman, Eds.; Ecosystem Responses to Mercury Contamination: Indicators of Change, SETAC, CRC Press, Boca Raton, published Feb. 2007, 47-86 pp.

Lambertsson L, and Nilsson M. 2006. Organic material: the primary control on mercury methylation and ambient methyl mercury concentrations in estuarine sediments. Environmental Science & Technology, 40: 1822–1829.

Liu, B., L.A. Schaider, R.P. Mason, J.P. Shine, N.N. Rabalais and D.B. Senn. 2015. Controls on methylmercury accumulation in northern Gulf of Mexico sediments. Estuarine, Coastal and Shelf Science.159: 50–59.

Louchouarn, P., M. Lucotte, A. Mucci, and P. Pichet. 1993. Geochemistry of mercury in two hydroelectric reservoirs in Quebec, Canada. Can. J. Fish. Aquat. Sci. 50: 269 - 281.

Mason, R.P. J. Reinfelder and F.M. Morel. 1995. Bioaccumulation of mercury and methylmercury. Water, Air and Soil Pollution. 80: 915 – 921.

Mierle, G. and R. Ingram. 1991. The role of humic substances in the mobilization of mercury from watersheds. Water, Air and Soil Poll. 56: 349 – 357.

Miskimmin, B.M., J.W.M. Rudd and C.A. Kelly. 1992. Influence of dissolved organic carbon, pH, and microbial respiration rates of mercury methylation and demethylation in lake water. Can. J. Fish. and Aquatic Sci. 49: 17–22.

Monperrus, M., E. Tessier D. Amouroux, A. Leynaert, P. Huonnie and O. Donald. 2007. Mercury methylation, demethylation and reduction rates in coastal and marine surface waters of the Mediterranean Sea. Marine Chemistry. 107 49–63

Mucci, M. Lucotte, S. Montgomery, Y. Plourde, P. Pichet and J.V. Tra. 1995. Mercury remobilization from flooded soils in a hydroelectric reservoir, La-Grande 2: results of a soil resuspension experiment. Can. J. Fish. Aquat. Sci. 52: 2507 – 2517.

Ouddane, B., N. Mikac, A. Cundy, L. Quillet and J.C. Fischer. 2008. A comparative study of mercury distribution and methylation in mudflats from two macrotidal estuaries: The Seine (France) and the Medway (United Kingdom). Applied Geochemistry. 23: 618–631.

Perie, C. and R. Ouimet. 2008. Organic carbon, organic matter and bulk soil density relationships in boreal forest soils. Can. J. Soil. Sci. 88: 315 – 325.

Pak K-R, and Bartha R. 1998. Mercury methylation and demethylation in anoxic lake sediments and by strictly anaerobic bacteria. Applied and Environmental Microbiology, 64: 1013–1017.

Paranjape AR and Hall BD. 2017. Recent advances in the study of mercury methylation in aquatic systems. FACETS 2: 85–119.

Pickhardt, P. C.L. Folt, C.Y. Chen, B. Klaue and J. Blum. 2002. Alga blooms reduce the uptake of methylmercury in freshwater food webs. Proceedings of the National Academy of Sciences. 99: 4419 – 4423.

Ravichandran M. 2004. Interactions between mercury and dissolved organic matter—a review**.** Chemosphere, 55: 319–331.

Rolfhus, K.R., J. Hurley, R.A. Bodaly and G. Perrine. 2015. Production and retention of methylmercury in inundated boreal forest soils. Environmental Science & Technology, 49: 3482–3489.

Schartup, A.T., P.H. Balcom, A. Sorensen, K.J. Gosnell, R.S. Calder, R.P, Mason and E.M. Sunderland. 2015. Freshwater discharges drive high levels of methylmercury in Arctic marine biota. Proc. Natl. Acad. Sci. 112: 11789 – 11794.

Schetagne, R., J. Therrien, and R. Lalumiere. 2003. Environmental monitoring at the La Grande complex. Evolution of fish mercury levels. Summary report. 1978–2000. Direction Barrages et Environnement, Hydro-Québec Production and Groupe Conseil GENIVAR Inc., 185 pp. and Appendices.

Selvendrian, P., C.T. Driscoll, J.T. Bushey and M.R. Montesdeoca. 2008. Wetland influence on mercury fate and transport in a temperate forested watershed. Environmental Pollution. 154: 46–55.

Scott, W.B. and M.G. Scott. 1988. Atlantic Fishes of Canada. Can. Bull. Aquat. Sci. No. 219. University of Toronto Press, Ontario. 731 p.

St. Louis, V.L., J.W.M. Rudd, C.A. Kelly and L.A. Barrie. 1995. Wet deposition of methylmercury in northwestern Ontario compared to other geographic locations. Water, Air and Soil Pollution. 80: 405-414.

St. Louis V.L., J. M. Rudd JWM, C.A. Kelly, K. Beaty, R.J. Flett and N.T. Roulet. 1996. Production and loss of methylmercury and loss of total mercury from boreal forest catchments containing different types of wetlands. Environmental Science & Technology. 30: 2719–2729.

Ullrich, S.M., T.W. Tanton and S. Abdrashitova. 2001. Mercury in the aquatic environment: A review of factors affecting methylation. Critical reviews in Environ. Sci. Technol. 31: 241 – 293.

Watras, C.J., K.A. Morrison, J.S. Host and N.S. Bloom. 1995**.** Concentration of mercury species in relationship to other site-specific factors in the surface waters of northern Wisconsin lakes. Limnology and Oceanography, 40: 556–565.

Wetzel, R. 2001. Limnology. 3rd Edition; Lake and River Ecosystems. Published by Academic Press, 1006 p. May 2011.

Appendix A

Mass of Methylmercury in Lake Melville Biota – Under Baseline and Calder et al.'s Post-Flood Scenarios

Mass of Methylmercury in Lake Melville Biota –
Under Baseline and Calder et al.'s Post-Flood Scenarios

1. Overview

The purpose of this assessment was to estimate the mass (kg) of methylmercury (MeHg) contained within aquatic organisms of Lake Melville (3100 km^2), for two scenarios: 1) under current, baseline conditions prior to flooding (the "baseline" scenario); and 2) the biomass that must be present under Calder et al.'s (2016a) forecasted post-flood conditions (the "post-flood" scenario). The difference between the total MeHg masses for the two scenarios represents the amount of additional (or "new") burden of MeHg needed to achieve tissue concentrations predicted by Calder et al. (2016) for Lake Melville biota during future 'steady state conditions' following flooding of MFR.

It is important to realize that the *actual* mass of MeHg that must be produced by MFR to achieve the Calder et al. prediction is *considerably* higher than post-flood scenario biomass. This is because it will take on the order of a decade of sustained production to reach this higher steady state. In addition, a substantial amount of the MeHg produced by the MFR will not end up in biota. Much will be buried in sediment, demethylated, or leave Lake Melville through tidal exchange for example; these concepts are addressed in the main document.

Calder et al. (2016) used measured concentrations of MeHg in water (mean annual) and tissues Hg to derive site-specific bioaccumulation factors (BAFs) for each species. They then applied these BAFs to modelled post-flood changes in MeHg concentrations in water to predict concentrations in biota[1]. BAFs are a widely-used, simple empirical tool for estimating steady-state tissue concentrations when direct measurements are impractical or impossible. A key underlying assumption of Calder et al.'s use of BAFs is that there is enough MeHg generation capacity in the MFR to sufficiently elevate MeHg in water throughout the entire study area to reach the new, higher steady-state tissue concentrations predicted for Lake Melville.

Our assessment estimates the mass of MeHg contained within the aquatic food web of Lake Melville. This was done by combining biota tissue concentrations, either measured baseline or Calder et al.'s predicted post-flood, with biomass estimates for biota in Lake Melville across all trophic levels (i.e., from primary producers through top predators) that we derived from the literature. For example, a single 10 kg fish with a MeHg concentration of 0.25 mg/kg ww would contain 2.5 mg of MeHg (i.e., 10 kg x 0.25 mg/kg ww = 2.5 mg). Thus, by pairing MeHg concentrations and population biomass estimates for all the organisms in Lake Melville, we calculated the total mass (kg) of MeHg present and predicted.

2. Methods

Biota Biomass in Lake Melville

A literature search was conducted to identify regional information on biomass in marine ecosystems. One key study (Bundy et al. 2000) was found that constructed a mass balance Ecopath model for the Newfoundland-Labrador Shelf. Ecopath is a top-down, ecosystem energetics model that looks across extremely wide, ecologically relevant trophic levels to characterize the entire ecosystem. A model is considered "balanced" when predator biomass and consumption rates are in line with prey biomass and production rates. While the model is for the continental shelf, rather than a nearly-enclosed estuarine

[1] They also incorporated habitat preferences (i.e., proportion of life history spent in the river, Lake Melville or Groswater Bay) into their predictions to account for potential habitat-related differences in MeHg exposure.

embayment like Lake Melville, the latter is thought to support notably high productivity and species diversity (Schartup et al. 2016, Durkalec et al. 2016). Thus, biomass estimates derived from the Ecopath model for the Labrador shelf are considered a conservative estimate for Lake Melville. Bundy et al. (2000) synthesized information on biomass, consumption, production and diet of major species or species groups spanning the entire ecosystem to estimate a steady-state scenario (i.e., the starting and ending biomass of each species is constant). It is necessary to take this broad approach in order to properly and accurately characterize the marine food web, given that obligate marine biota (e.g., Arctic cod and their prey) are used in the Calder et al. paper. You can't cherry pick where MeHg will end up once in Lake Melville.

We estimated biomass and trophic level estimates using Bundy et al.'s ultimate model (balanced model 2) to estimate the total biomass within the entirety of the Lake Melville ecosystem. The only change we made in the model ecosystem was to use "seals" in general, replacing the named 'harp' and 'hooded' seals, but assuming the same total seal biomass (kg/km^2). The rationale for this is that Lake Melville contains important habitat for ringed and harbour seals (Schartup et al. 2016, Durkalec et al. 2016). While not specifically tailored for Lake Melville, the Bundy et al. (2000) ecosystem and associated biomass estimates are considered conservative for the purposes of this assessment (see results for more discussion on total ecosystem biomass differences between shelf and bay/fjord ecosystems).

Baseline MeHg Concentrations in Lake Melville Biota

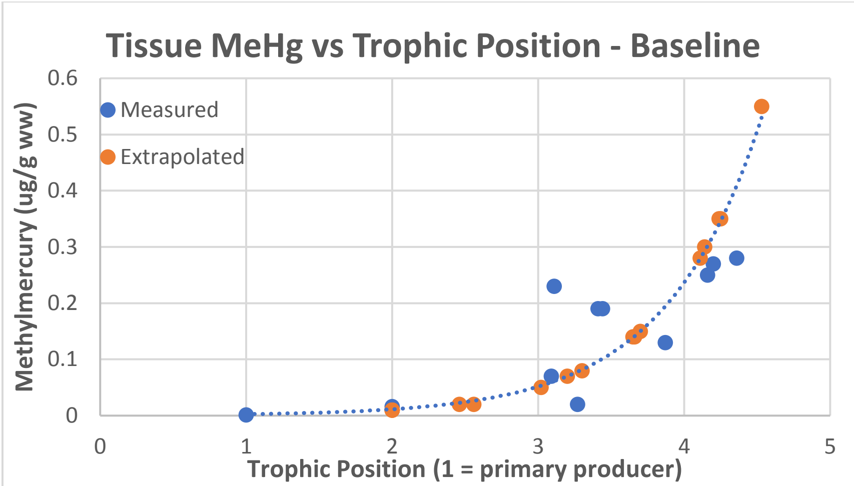
Calder et al. (2016 Supporting Document) report measured concentrations of MeHg in commonly harvested biota from Lake Melville (Table 6a, b); phytoplankton data are reported in Schartup et al. (2015 Supporting Document). Key assumptions were as follows:

- Where more than one tissue type was measured for fish or birds, muscle tissue MeHg concentrations were used;
- Where two size classes were included in the fish biomass estimates (e.g., Atlantic cod), but only one group measured for MeHg - the Calder et al. (2016 Supporting Document) mean -SD was used for the small size class and the mean + SD was used for the larger size class. This accounts for the general MeHg-size relationship in fish;
- Seals – Weighted average MeHg concentrations were derived based on (1) age/size frequency proportion (Chambellant 2010) and (2) tissue proportion (Crile and Quiring 1940, Best 1985, Ryg et al. 1990). The age/size frequency proportion was derived from published age frequency and growth data (i.e., the proportion of the population in an age class was multiplied by the mean weight of that age class, then divided by the total mass across groups). The tissue proportion (i.e., relative proportion of muscle, liver and kidneys) was derived from published data on the weights of various parts of the seal (e.g., muscle, liver, kidneys, blubber, pelt, bones).

Where no measured data were available, tissue MeHg concentrations were estimated using the relationship between measured tissue MeHg concentrations (described above) and trophic position (TP) (from Bundy et al. 2010). MeHg concentrations were log-transformed for the linear regression. The regression equation for the baseline MeHg-TP relationship was as follows:

$$MeHg = 10^{(-3.369 + 0.686*TP)}; \text{ Adjusted } R^2 = 0.81; P<0.001$$

The plotted relationship between tissue Hg (μg/g) and tropic position and extrapolated values are shown below.



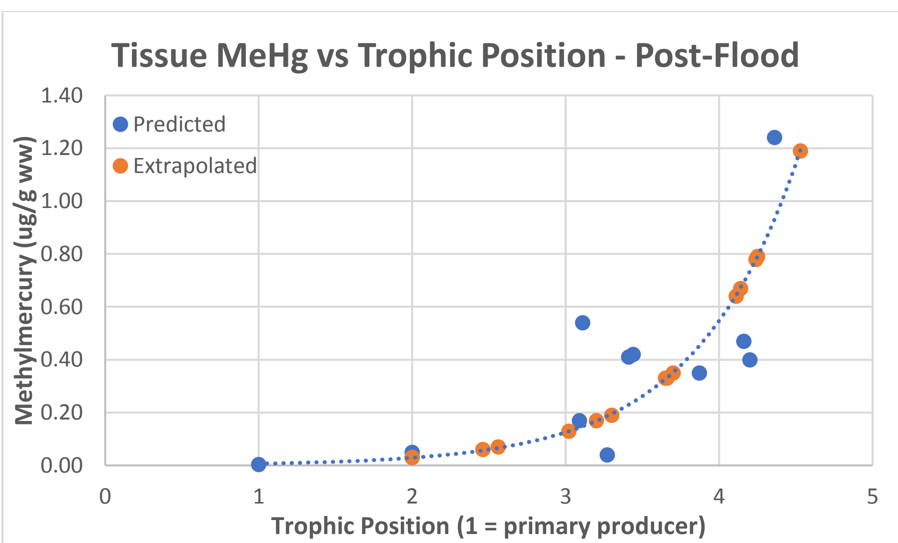
Post-flood MeHg Concentrations in Lake Melville Biota

Calder et al. (2016) report predicted post-flood concentrations for a range of biota, including obligate freshwater fish (lake trout), anadromous fish (Atlantic salmon), marine fish (rock cod), ducks and marine mammals – all of which are assumed to be exposed to MeHg exported from MFR. The approach taken in this assessment for the post-flood scenario was essentially the same as described for the baseline scenario, with the following exception for seals. Calder et al. (2016) report an age-weighted average MeHg for seals based on the preferential harvest by local residents of younger, smaller seals (weights for age classes were: 80% for <1 yrs, 10% for 1 to 4 yrs and 10% for >4 yrs). To obtain weights based on the actual population structure, reported predictions were first unweighted (assuming the same relative proportional differences MeHg concentrations among the age classes as seen in the measured data), then weighted as described above for the baseline scenario.

For organisms that weren't included in the post-flood predictions by Calder et al. (2016), tissue MeHg concentrations were extrapolated from the MeHg-TP relationship using the same approach above for the baseline scenario. The regression equation for the post-flood MeHg-TP relationship was as follows:

*MeHg = 10^(-2.821 + 0.640*TP)*; Adjusted R^2 = 0.76; $P<0.001$

The plotted relationship and extrapolated values are shown below.



3. Results

The estimated cumulative mass of MeHg contained within the biota of Lake Melville is 272 tonnes/km^2 for the baseline and post-flood scenarios is shown in **Table 1**. A sample calculation for mass of MeHg in Lake Melville, using seals in the baseline scenario as an example, is as follows:

- Convert seal biomass to kg: 0.21 t/km^2 = 210 kg/km^2
- Convert seal [MeHg] from mg/kg to g/kg = 0.28/1000 = 0.00028 g/kg
- Multiply seal biomass x [MeHg] = 0.0588 g MeHg/km^2
- Expand to Lake Melville surface area = 0.0588 g MeHg/km^2 * 3100 km^2 = 182.3 g MeHg
- Convert to moles (215.6 g/mol for MeHg) = 182.3g or MeHg/215.6 g/mol = 0.85 mol

The MeHg mass in the biotic component of the Lake Melville ecosystem is **19.8 kg** (91.7 mol). Assuming the changes in biota MeHg concentrations predicted by Calder et al. (2016) using their BAF approach, the mass of MeHg in Lake Melville biota must increase to **50.8 kg** (236 mol) post-flood, to satisfy their predictions. The difference between the two scenarios is **31.1 kg** (144 mol), which is the mass of additional MeHg that would have to be accumulated over time in order to change pre-flood biota concentrations to match the predictions of Calder et al. As stated in the Overview, 31.1 kg of MeHg only represents the mass of "new" MeHg from MFR in the biota. *Actual* MeHg production from the MFR would need to be *considerably higher* given that it will take on the order of a decade of sustained production to reach the higher steady state in biota. Furthermore, as we noted in the main document, a substantial amount of the MeHg produced by the MFR will not end up in biota (e.g., much will be buried in sediment or leave Lake Melville through tidal exchange).

4. Uncertainty Assessment

Biomass estimates are an acknowledged source of uncertainty. For example, we did not include whales in our biomass estimate. Although we know they are present, they are migratory and will not always be present Thus, we conducted a sensitivity analysis to explore the implications of changing biomass estimates. Essentially, any reduction or increase in total biomass will directly affect the estimate of the baseline mass, or the mass of "new" MeHg needed to match the Calder et al. (2016) predictions. Thus, halving or doubling the biomass estimates will do the same to the estimates of the mass of "new" MeHg needed to match the Calder et al. (2016) predictions (i.e., 15.5 kg MeHg and 62.2 kg MeHg for the halving and doubling sensitivity analyses, respectively).

We conducted a literature search to provide context and bound the Bundy et al. (2000) biomass estimate of 276 kg/km^2. We identified 16 other studies that quantified ecosystem biomass in temperate and Arctic marine environments using EcoBase (http://sirs.agrocampusouest.fr/EcoBase/#discoverytools), an online repository of published Ecopath models (**Table 2**, **Figure 1**). The "ecosystem type" field was reported in EcoBase. While biomass estimates ranged from 57 t/km^2 (Hudson Bay) to 3786 t/km^2 (Iceland), the majority of values fell between 200 and 400 t/km^2, similar to our estimate. Interestingly, with the exception of Hudson Bay, the other three bay/fjord ecosystems were considerably higher in biomass than other regional shelf or open ocean ecosystems. In Alaska, Prince William Sound (1078 t/km^2) was nearly 5-fold higher than Southeast Alaska (215 t/km^2) and the Western and Central Aleutian Islands (208 t/km^2). In British Columbia, Western Vancouver Island (236 t/km^2) was approximately 2-fold higher than Haida Gwaii (122 t/km^2) and the Northern BC Coast (129 t/km^2). Finally, Chesapeake Bay biomass (665 t/km^2) was more than double that of the Southern Gulf of St. Lawrence (291 t/km^2) or Newfoundland-Labrador Shelf (273 t/km^2). These results indicate that the use of the Bundy et al. (2000) biomass estimate for Lake Melville is likely conservative and that the actual biomass could be 2-fold to 5-fold higher.

5. References

Ainsworth C., J.J. Heymans, T.J. Pitcher and M. Vasconcellos. 2002. Ecosystem Models of Northern British Columbia for the Time Periods 2000, 1950, 1900 and 1750 Fisheries Centre Research Reports

Araújo J.N. and A. Bundy. 2011. Description of three Ecopath with Ecosim ecosystem models developed for the Bay of Fundy, Western Scotian Shelf and NAFO Division 4X Canadian Technical Report of Fisheries and Aquatic Sciences.

Bundy, A., G.R. Lilly and P.A. Shelton. 2000. A mass balance model of the Newfoundland-Labrador Shelf. Canadian Technical Report of Fisheries and Aquatic Sciences 2310: 1-157 pp.

Calder, R.S. A.T. Schartup, L. Miling, A.P. Valberg, P.H. Balcom and E.M. Sunderland. 2016. Future impacts of hydroelectric power development on merthylmercury exposures on Canadian indigenous communities. Environ. Sci. Techol. 50: 13115 – 13122.

Christensen V., A. Beattie, C. Buchanan, M. Hongguang, S.J. Martell, R.J. Latour, D. Preikshot, M.B. Sigrist, J.H. Uphoff, C.J. Walters, R.J. Wood, and H. Townsend. 2009. Fisheries Ecosystem Model of the Chesapeake Bay: Methodology, Parameterization, and Model Exploration NOAA Technical Memorandum.

Dalsgaard J., D. Pauly and T.A. Okey. 1997. Preliminary mass-balance model of Prince William Sound, Alaska, for the pre-spill period, 1980-1989. Fisheries Centre Research Reports.

Dommasnes A.V. Christensen, B. Ellertsen, C. Kvamme, W. Melle, L. Nottestad, P. Torstein, S. Tjelmeland and D. Zeller. 2001. An Ecopath model for the Norwegian Sea and Barents Sea Fisheries Impacts on North Atlantic Ecosystems: Models and Analyses. pp 213-240. In Guenette S., Christensen V. and D. Pauly (eds) Fisheries Impacts on North Atlantic Ecosystems: Models and Analyses. Fisheries Centre Research Reports 9 (4).

Durkalec, A., T. Sheldon, T. Bell (eds). 2016. Lake Melville: Avativut Kanuittailinnivut (Our Environment, Our Health). Scientific Report, Nain, NL. Nunatsiavut Government.

Espinosa-Romero M.J., E.J. Gregr, C. Walters, V. Christensen and K. Chan. 2011. Representing mediating effects and species reintroductions in Ecopath with Ecosim. Ecological Modelling. 222: 1569-1579.

Guénette S., S.J. Heymans, V. Christensen and A. Trites. 2006. Ecosystem models show combined effects of fishing, predation, competition, and ocean productivity on Steller sea lions (*Eumetopias jubatus*) in Alaska Canadian Journal of Fisheries and Aquatic Sciences. pp. 2495-2517.

Guenette, S. 2005. Models of Southeast Alaska. In Guenette S. & V. Christensen (eds) Food web models and data for studying fisheries and environmental impacts on Eastern Pacific ecosystems. Fisheries Centre Research Reports 13(1):106-178.

Heymans, S. 2005. Ecosystem models of the Western and Central Aleutian Islands in 1963, 1979 and 1991. In Guenette S. & V. Christensen (eds) Food web models and data for studying fisheries and environmental impacts on Eastern Pacific ecosystems. Fisheries Centre Research Reports 13(1): 8-82.

Hoover, C., E. Chmelnitsky, D. Cobb, C. Michel, A. Niemi, P. Ramlal, W. Walkusz, H. Swanson, J. Reist, O. Nielsen, L. Harwood, L., Postma, J. Higdon, S. Ferguson, R. Young, and L. Loseto. 2014. Summary of the Beaufort Sea Shelf Ecosystem Research Initiative. Can. Tech. Rep. Fish. Aquat. Sci. 3078: ix + 87p.

Kumar, R., S. Surma, T. Pitcher, D. Varkey, M. Lam, C. Ainsworth and E. Pakhomov. 2016. An Ecosystem Model of the Ocean Around Haida Gwaii, Northern British Columbia: Ecopath, Ecosim and Ecospace. Fisheries Centre Research Reports 24 (2): 76 pp.

Mendy A.N and E.A. Buchary. 2001. Constructing an Icelandic marine ecosystem model for 1997 using a mass-balance modeling approach Fisheries Impacts on North Atlantic Ecosystems: Models and Analyses. pp 182-197. In Guénette S., Christensen V. and D. Pauly (eds) Fisheries Impacts on North Atlantic Ecosystems: Models and Analyses. Fisheries Centre Research Reports 9 (4).

Pedersen S.A. and D. Zeller D. 2001. A mass balance model for the West Greenland marine ecosystem Fisheries Impacts on North Atlantic Ecosystems: Models and Analyses. pp 111-127. In Guénette S.,Christensen V. and D. Pauly (eds) Fisheries Impacts on North Atlantic Ecosystems: Models and Analyses. Fisheries Centre Research Reports 9 (4).

Savenkoff C., H. Bourdages, D.P. Swain, S. Despatie, M.J. Hanson, R. Méthot, L. Morissette and M.O. Hammill. 2004. Input data and parameter estimates for ecosystem models of the southern Gulf of St. Lawrence (mid-1980s and mid-1990s) Canadian Technical Report of Fisheries and Aquatic Sciences.

Schartup, A.T., P.H. Balcom, A. Sorensen, K.J. Gosnell, R.S. Calder, R.P, Mason and E.M. Sunderland. 2015. Freshwater discharges drive high levels of methylmercury in Arctic marine biota. Proc. Natl. Acad. Sci. 112: 11789 – 11794

Whitehouse A.G., K. Aydin, T.E. Essington and G.L. Hunt Jr. 2014. A trophic mass balance model of the eastern Chukchi Sea with comparisons to other high-latitude systems Polar Biology. NOAA Technical Memorandum NMFS-AFSC-262.

Table 1. Estimated mass of methylmercury in the biota of Lake Melville for baseline and post-flood scenarios.

Group Name	Biomass (t/km^2)	Trophic Position	Baseline: Biota MeHg (mg/kg)	Baseline: g MeHg/ km^2	Baseline: MeHg (mol)	Baseline: Comments	Post-flood Predictions (Calder et al. 2016a,b): Biota MeHg (mg/kg)	Post-flood: g MeHg/ km^2	Post-flood: MeHg (mol)	Post-flood: Comments
Whales	0.25	4.24	--	--	--	Not included in calculations	--	--	--	Not included in calculations
Seals	0.21	4.36	0.28	0.059	0.85	Weighted mean (Calder et al. supp S6a; see text)	1.24	0.261	3.75	Weighted mean (Calder et al. supp S11; see text)
Seabirds	0.01	4.2	0.27	0.003	0.04	Mean of all birds (Calder et al. supp S6)	0.4	0.004	0.06	Mean of all birds (Calder et al. supp S11)
Cod>35cm	2.04	4.16	0.25	0.510	7.33	Mean + SD (Calder et al. supp S6a)	0.47	0.959	13.79	Mean + SD (Calder et al. supp S11)
Cod<=35cm	0.27	3.87	0.13	0.035	0.50	Mean - SD (Calder et al. supp S6a)	0.35	0.095	1.36	Mean - SD (Calder et al. supp S11)
G.halibut>40cm	0.35	4.53	0.55	0.193	2.77	Extrapolated using trophic position (see text)	1.19	0.417	5.99	Extrapolated using trophic position (see text)
G.halibut<=40cm	0.45	4.25	0.35	0.158	2.26	Extrapolated using trophic position (see text)	0.79	0.356	5.11	Extrapolated using trophic position (see text)
Aplaice>35cm	0.97	3.65	0.14	0.136	1.95	Extrapolated using trophic position (see text)	0.33	0.320	4.60	Extrapolated using trophic position (see text)
Aplaice<=35cm	0.78	3.7	0.15	0.117	1.68	Extrapolated using trophic position (see text)	0.35	0.273	3.93	Extrapolated using trophic position (see text)
Flounders	0.89	3.09	0.07	0.062	0.90	Flatfish (Calder et al. supp S6b)	0.17	0.151	2.18	Flatfish (Calder et al. supp S11)
Skates	0.26	4.11	0.28	0.073	1.05	Extrapolated using trophic position (see text)	0.64	0.166	2.39	Extrapolated using trophic position (see text)
Redfish	1.24	3.66	0.14	0.174	2.50	Extrapolated using trophic position (see text)	0.33	0.409	5.88	Extrapolated using trophic position (see text)
L.Dem.Feeders	0.85	3.44	0.19	0.162	2.32	Rock cod used (Calder et al. supp S6b)	0.42	0.357	5.13	Rock cod used (Calder et al. supp S11)
S.Dem.Feeders	2.38	3.11	0.23	0.547	7.87	Sculpin used (Calder et al. supp S6a)	0.54	1.285	18.48	Sculpin used (Calder et al. supp S11)
Capelin	13.61	3.27	0.02	0.272	3.91	Capelin (Calder et al. supp S6b)	0.04	0.544	7.83	Capelin (Calder et al. supp S11)
Sand lance	0.67	3.2	0.07	0.047	0.67	Extrapolated using trophic position (see text)	0.17	0.114	1.64	Extrapolated using trophic position (see text)
Arctic cod	3	3.41	0.19	0.570	8.20	Atlantic cod used (Calder et al. supp S6b)	0.41	1.230	17.69	Atlantic cod used (Calder et al. supp S11)
L.Pel.Feeders	0.03	4.24	0.35	0.011	0.15	Extrapolated using trophic position (see text)	0.78	0.023	0.34	Extrapolated using trophic position (see text)
Pisc.SPF	1.36	4.14	0.3	0.408	5.87	Extrapolated using trophic position (see text)	0.67	0.911	13.10	Extrapolated using trophic position (see text)
Plankt.SPF	2.86	3.3	0.08	0.229	3.29	Extrapolated using trophic position (see text)	0.19	0.543	7.81	Extrapolated using trophic position (see text)
Shrimp	0.82	2.46	0.02	0.016	0.24	Extrapolated using trophic position (see text)	0.06	0.049	0.71	Extrapolated using trophic position (see text)
Large Crustacea	1.73	3.02	0.05	0.087	1.24	Extrapolated using trophic position (see text)	0.13	0.225	3.23	Extrapolated using trophic position (see text)
Echinoderms	112.3	2	0.01	1.123	16.15	Extrapolated using trophic position (see text)	0.03	3.369	48.44	Extrapolated using trophic position (see text)
Molluscs	42.1	2	0.016	0.674	9.69	Mean of molluscs (see note) (Calder et al. supp S6b)	0.05	2.105	30.27	Mean of molluscs (see note) (Calder et al. supp S11)
Polychaetes	10.5	2	0.01	0.105	1.51	Extrapolated using trophic position (see text)	0.03	0.315	4.53	Extrapolated using trophic position (see text)
O.Benthic Inver	7.8	2	0.01	0.078	1.12	Extrapolated using trophic position (see text)	0.03	0.234	3.36	Extrapolated using trophic position (see text)
Lge.Zooplankton	11.23	2.56	0.02	0.225	3.23	Extrapolated using trophic position (see text)	0.07	0.786	11.30	Extrapolated using trophic position (see text)
Sm.Zooplankton	26.94	2	0.01	0.269	3.87	Extrapolated using trophic position (see text)	0.03	0.808	11.62	Extrapolated using trophic position (see text)
Phytoplankton	26.86	1	0.0013	0.035	0.50	Mean of all size classes (Schartup et al. supp S4)	0.0034	0.091	1.31	Proportional change (2.6) to water (Calder et al.)
				Total moles MeHg in biota (baseline)	**91.7**			**Total moles MeHg in biota (post-flood)**	**235.8**	
				Total kg MeHg in biota (baseline)	**19.8**			**Total kg MeHg in biota (post-flood)**	**50.8**	
								Total moles of "new" MeHg in biota (post-flood)	**144.2**	
								Total kg of "new" MeHg in biota (post-flood)	**31.1**	

Notes:
Groups, biomass and trophic position for ecosystem from Bundy et al. (2010).
RED MeHg concentrations were estimated from the MeHg-TP relationship (see text)
Molluscs included clams, scallops, periwinkles, and mussels.

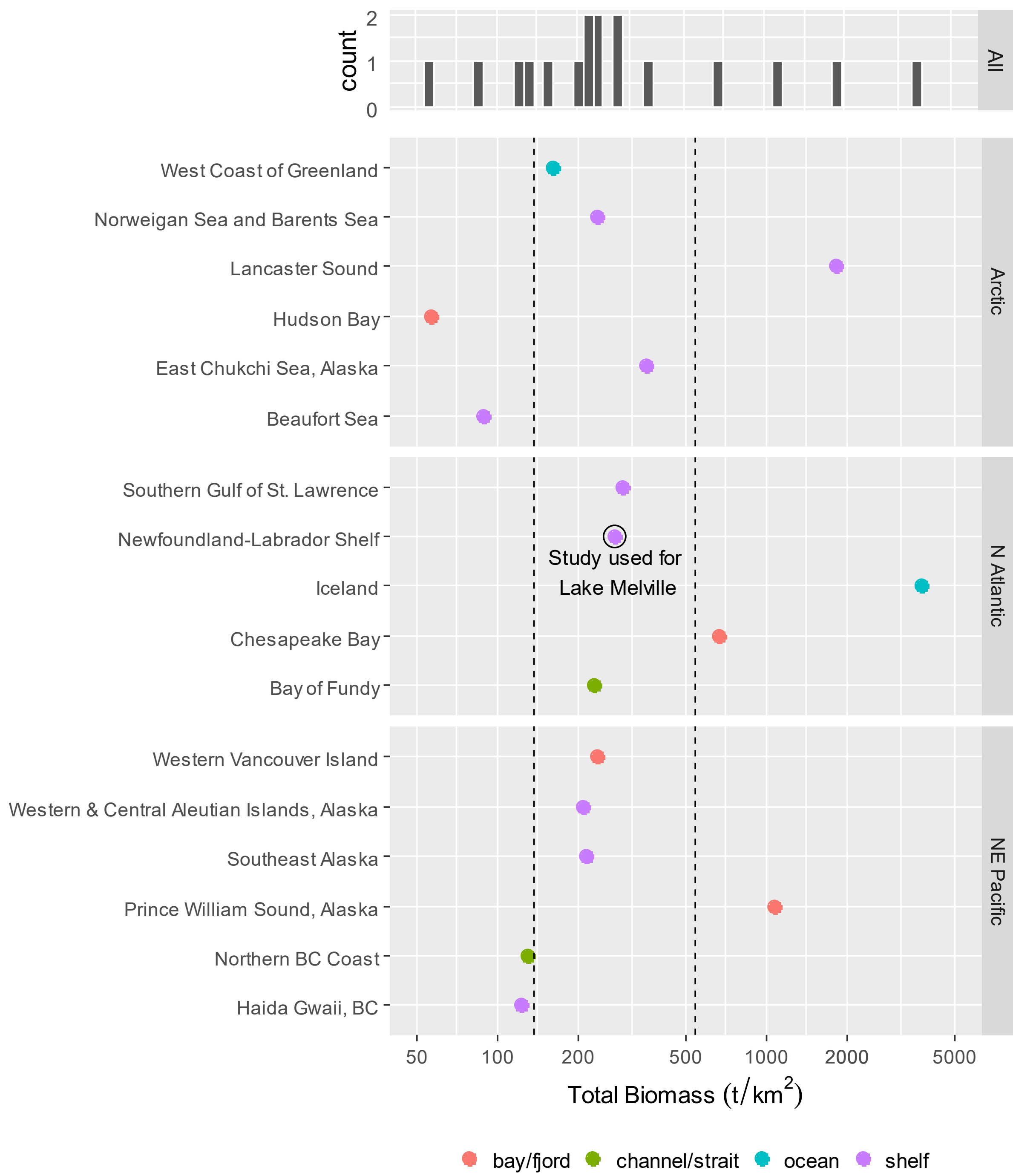
Table 2. Total biomass estimates used in published Ecopath models for northern temperate and Arctic marine ecosystems.

Area	Region	Ecosystem Type[1]	Total Biomass (t/km^2)	Study
Newfoundland-Labrador Shelf[2]	*N Atlantic*	*shelf*	*273*	*Bundy et al. 2000*
West Coast of Greenland	Arctic	ocean	162	Pedersen & Zeller 2001
Prince William Sound, Alaska	NE Pacific	bay/fjord	1078	Dalsgaard et al. 1997
Western & Central Aleutian Islands, Alaska	NE Pacific	shelf	208	Heymans 2005
Southeast Alaska	NE Pacific	shelf	215	Guenette 2005
Northern BC Coast	NE Pacific	channel/strait	129	Ainsworth et al. 2002
Norweigan Sea and Barents Sea	Arctic	shelf	234	Dommasnes et al. 2001
Bay of Fundy	N Atlantic	channel/strait	229	Araujo and Bundy 2011
Iceland	N Atlantic	ocean	3786	Mendy & Buchary 2001
Chesapeake Bay	N Atlantic	bay/fjord	665	Christensen et al. 2009
East Chukchi Sea, Alaska	Arctic	shelf	356	Whitehouse 2014
Haida Gwaii, BC	NE Pacific	shelf	122	Kumar et al. 2016
Hudson Bay	Arctic	bay/fjord	57	Wabnitz & Hoover 2012
Lancaster Sound	Arctic	shelf	1832	Mohamed 2001
Western Vancouver Island	NE Pacific	bay/fjord	236	Espinosa-Romero et al. 2011
Southern Gulf of St. Lawrence	N Atlantic	shelf	291	Savenkoff et al. 2004
Beaufort Sea	Arctic	shelf	89	Hoover et al. 2014

1. Ecosystem type as reported in EcoBase (http://sirs.agrocampus-ouest.fr/EcoBase/#discoverytools)
2. Study used to estimate Lake Melville biomass in this assessment.

Figure 1. Total biomass estimates from published Ecopath models by region (panels) and ecosystem type (point colour) for temperate and Arctic marine ecosystems. Top panel shows histogram of biomass estimates across all ecosystem types and regions.

Note: vertical dashed lines are the 0.5x and 2x biomass estimates used for the sensitivity analysis.



Predicted Increases in Fish Methylmercury Muscle Tissue Concentrations
In Goose Bay and Lake Melville

Submitted to:

Nalcor Energy

Submitted by:

Jim McCarthy
Wood
Environment & Infrastructure Solutions
133 Crosbie Road
St. John's, NL
A1B 4A5

July 2018

Issue No.	Date (d/m/y)	Issue Description	Prepared by	Checked by	Review by
0	July 12, 2018	Draft for review	JH McCarthy	DA Robbins	P Madden / R Baker
1	July 19, 2018	Rev 1 reviewed	JH McCarthy	DA Robbins	P Madden
2					

Table of Contents

List of Figures

List of Tables

APPENDICES

1.0 INTRODUCTION

Nalcor Energy (Nalcor) is developing the remaining hydroelectric potential of the lower Churchill River through hydroelectric generating facilities at Muskrat Falls and Gull Island. The Muskrat Falls portion of the project, which is currently under construction, will create a reservoir with a surface area of $101km^2$ and aquatic residence time of approximately 10 days. The existing river within the proposed footprint of the Muskrat Falls reservoir area has a surface area of ~$60km^2$ therefore the area of additional terrestrial flooding will be approximately $41km^2$, representing a 65-70 percent increase in the existing waterbody surface area. Note that of this 41 km^2 total, approximately 11 km^2 consists of gravel bars and riparian soils with no organic soils, although some deciduous shrubs may be present. Thus, the surface area of forested habitat with an intact organic soil horizon is about 30 km^2.

Many fish species have been predicted to be influenced by the Muskrat Falls project in terms of habitat change and methylmercury as part of the assessment process. Much of the baseline data required for the Environmental Assessment and Environmental Effects Monitoring (EEM) programs describe these species, their potential for interaction with the project, as well as the estimation of potential effects (e.g., Nalcor 2009; Amec 2016a).

One of the most well-known issues surrounding the formation of new reservoirs is the increase in methylmercury (MeHg) concentration in aquatic biota, especially fish resident within the reservoir (e.g., Bodaly et al. 1984; Jackson 1986; French 1997; Anderson et al. 1995). While this has been very well studied within reservoirs, the phenomenon of transport and bioaccumulation of methylmercury in downstream fish populations has been seldom studied (e.g., Anderson 2011; Schetagne et al. 2000). With respect to Muskrat Falls reservoir, possible increases in methylmercury concentration in tissues of fish and marine mammals that are consumed by local human populations downstream in Goose Bay and Lake Melville, has been a very contentious issue since the Calder et al. (2016) publication. While the potential for human health risks to residents due to increased exposure to methylmercury in various fish species from Goose Bay and Lake Melville was modeled and included in Nalcor's Human Health Risk Assessment (HHRA) (Dillon 2016), new information has been developed, especially since the Schartup et al. 2015 and Calder et al. 2016 publications. This relates to the spatial and temporal extent of feeding by key aquatic species within the 'exposure zone' (i.e., downstream area with increased methylmercury above baseline) as well as the amount (e.g., concentration change in water or mass (gm) of methylmercury delivered downstream over time).

Bioaccumulation of methylmercury by biota is almost exclusively via diet (e.g., Hall et al. 1997) which is why prey/food sources of species informs exposure to methylmercury. Therefore, life cycles and feeding habitat used by fish species captured and consumed by local residents is key to understanding and predicting any potential future mercury increases. Of importance is understanding how key species might feed within the zone of exposure in Goose Bay and Lake Melville. For example, if key species do not feed within an area of Lake Melville that is affected by methylmercury exported from the reservoir (i.e., within the zone of exposure), methylmercury concentrations will not change. If organisms are partially exposed, it is reasonable to expect that concentrations will only change in proportion to exposure.

Thus, information on species distribution within and downstream of the Muskrat Fall reservoir, their abundance, trophic position within the food web, and baseline MeHg concentrations is of critical importance. This information, based on data collected since 1998, was presented to the Independent Experts Committee (IEC) on two separate occasions; September 7, 2017 and February 15, 2018. The data collected directly from the lower Churchill River since 1998 clearly shows that the habitat use and exposure of fish species to potential increases in MeHg concentrations in water, and hence the food web, have been inconsistently applied to previous HHRA predictions.

1.1 Purpose

The purpose of this document is to provide summary life history and habitat use by key species identified as being important in local diets that are targeted within Goose Bay and Lake Melville. This data is critical to determining the exposure of these species to any predicted increases in water MeHg concentrations exported downstream from the Muskrat Falls reservoir. New reservoir mercury modelling and detailed hydrodynamic modelling that describes the predicted increase and distribution of water MeHg concentrations in Goose Bay and Lake Melville have been used to inform how fish tissue MeHg concentrations may change over time. The magnitude of change is dictated by the time and space a particular species may forage within Goose Bay and different parts of Lake Melville. It should be noted that to date, most concern by residents is related to fish species captured and consumed within the estuarine environment downstream of the Muskrat Falls reservoir in Goose Bay and Lake Melville. As such, the Muskrat Falls reservoir area and the riverine section of the lower Churchill River are not the focus of this summary as these areas do not contribute to potential human exposure. It has also been conservatively assumed that total mercury concentrations in fish muscle tissue analyzed as part of the baseline program consists entirely of methylmercury based on local comparisons of paired total and methylmercury samples (also see Lasorsa and Allen-Gil 1995; Anderson and Depledge 1997; Marrugo et al. 2007).

2.0 BIOACCUMULATION OF MEHG IN FISH

The primary exposure pathway to methylmercury by all aquatic organisms is almost exclusively via diet (e.g., Hall et al. 1997). Following formation of the Muskrat Falls Reservoir it is predicted that a greater net export of MeHg will be delivered by lower Churchill River to Goose Bay and Lake Melville. Given that this is a very dynamic process and a hydraulically complex environment, it is difficult to predict how changes will occur over time in different parts of Lake Melville. For example, deep water beneath the brackish surface layer will be less influenced and the eastern portion of Lake Melville will be less affected than the western portion, simply because of dilution and loss to photodegradation (e.g., Sellers et al. 1996) and other losses. However, areas of dynamic mixing between the freshwater surface layer and the underlying marine layer where light penetration, and thus productivity, is high is where methylmercury will be most accumulated by bacteria, phytoplankton and nanoplankton. This phenomenon will ultimately distribute methylmercury into the base of the aquatic food web across areas of exposure; however, recent hydrodynamic modelling shows that this occurs disproportionately. Relatively greater water concentrations of methylmercury will be available for accumulation in Goose Bay biota than Lake Melville because of a variety of factors including dilution, photo-demethylation (e.g., Sellers 1992), adsorption to particles, settling and progressive uptake by biota. Thus, where an organism spends its time feeding will dictate its magnitude of exposure. This has important implications in terms of predicted increases in fish MeHg.

For a fish to be exposed, it must occupy and feed in the same space as the contaminant for a sustained period. Therefore, life history is an important factor, informing the magnitude of exposure on both a spatial and temporal scale to produce a change in tissue mercury concentrations. For this analysis, the predicted relative increase in MeHg concentration in water is assumed to be the predicted upper maximum relative increase in fish muscle tissue should a fish be fully exposed to that water concentration. That is, the correlation between the increase in MeHg concentration in water and the increase in prey species concentration is assumed at a 1:1 ratio. This is a conservative assumption but acknowledges uncertainty in attempting to rationalize or justify a lower increase ratio relative to water. This also follows the assumption made in the Calder et al. (2016) paper, therefore comparisons to relative increases between the two can be considered. The predicted relative increase in fish tissue MeHg in Goose Bay and Lake Melville also does not take into account the biomass of MeHg that can be produced by Muskrat Falls reservoir nor the biomass of biota within Goose Bay and Lake Melville for uptake; therefore, they are considered conservative overestimates. Biomass effects on accumulation of MeHg is addressed in Azimuth (2018).

2.1 Predicted Methylmercury Increases in Water

Detailed modelling has been utilized to predict MeHg that will be generated by the Muskrat Falls reservoir using RESMERC and empirical data from the Experimental Lakes Area (ELA) (Harris and Hutchinson 2018). Fludex was an experimentally flooded series of boreal forest systems at the Experimental Lakes Area (ELA) of northwestern Ontario, where net MeHg import, generation and export was measured over a five-year period (Hall et al. 2005). RESMERC is a calibrated model that simulates the magnitude and timing of response, including pulse, of MeHg through the Muskrat Falls reservoir system. The pulse is modelled through the reservoir in various steps in the ecosystem; water, sediments, lower trophic levels, to higher trophic levels (Harris and Hutchinson 2018). At Muskrat Falls Reservoir, a portion of the methylmercury generated will be transported downstream to the lower reaches of the river, Goose Bay and Lake Melville. The quantity of MeHg exported from Muskrat Falls reservoir was estimated using both the results of RESMERC and Fludex. The estimates were used as input parameters for extensive hydrodynamic modelling of Goose Bay and Lake Melville which estimated MeHg increases based on reservoir MeHg outflows (Brunton 2018) and various natural processes that affect MeHg concentrations such as freshwater flows, salinity, currents, flushing, winds, ice, transport, photodegradation, and settling (Brunton 2018). Details of the hydrodynamic model are provided in Brunton (2018).

Hydrodynamic model results indicate that MeHg generated from Muskrat Falls reservoir will be transported downriver via the upper freshwater layer that enters the estuary habitat. Therefore, it has been assumed that the general exposure concentration of prey occurs within the epilimnion, occupying the top 20m of the estuary. This is the depth to which the combined surface freshwater layer and the upper saline water mix, just above the deeper marine layer or hypolimnion. This zone is highly productive and exposed to additional nutrients (Schartup et al. 2015). We have assumed that within this productive zone is where MeHg is accumulated within the lower trophic levels (nano and zooplankton). Thus, it is assumed that larger prey will ultimately derive any increased accumulation of MeHg here. The hydrodynamic modelling also shows that as water from Muskrat Falls reservoir travels downriver and throughout Goose Bay and Lake Melville, predicted concentrations decrease. **Figure 2-1** provides the boundary between three distinct areas where concentrations differ; Goose Bay, West Lake Melville, and East Lake Melville. These three areas are identified as different "zones of exposure" for predicting increases in fish MeHg tissue concentrations. **Table 2-1** provides an estimate of the predicted relative increase in MeHg concentrations within the epilimnion relative to baseline (i.e., 0.017 ng/L) for the three zones of exposure based on the hydrodynamic model results. The predicted relative increase in water is the mean of the consecutive three-year sequence with the highest predicted MeHg concentrations within the upper 20m of the water column, using results from both RESMERC and Fludex. More detailed information can be found in the Technical Memorandum by Harris and Hutchinson (2018). The rationale for using a three-year mean is to realistically estimate the level of exposure throughout the life span of those key fish species in Goose Bay and Lake Melville (see Section 2.2).

2.2 Potential Fish Exposure to Methylmercury

Nalcor has collected baseline data since 1998 on the lower Churchill River, Goose Bay, and Lake Melville. Included in this baseline data are the ongoing results of species distribution and abundance, trophic feeding position, and total mercury concentrations in fish and seals. Detailed summaries of the results are provided in **Appendices A and B**.

In total, the baseline sampling program for the Lower Churchill Hydroelectric Development has sampled over 10,140 fish from over 20 different species between 1998-2017. While many species of fish have been identified as being consumed by residents in previous HHRAs (e.g., Calder et al. 2016 and Dillon 2016), many have either not been captured within or downstream of the Muskrat Falls reservoir (see Amec 2017 and **Appendix A**), have only been captured within the marine environment beyond Lake Melville (e.g., Li et al. 2016; Calder et al. 2016), or do not feed in Lake Melville upon their return to tributaries to spawn. For these species, increases in MeHg exposure are not anticipated and therefore have not been included in further estimates of bioaccumulation increases. **Figure 2-2** provides an overview of the relative abundances of many of the species captured.

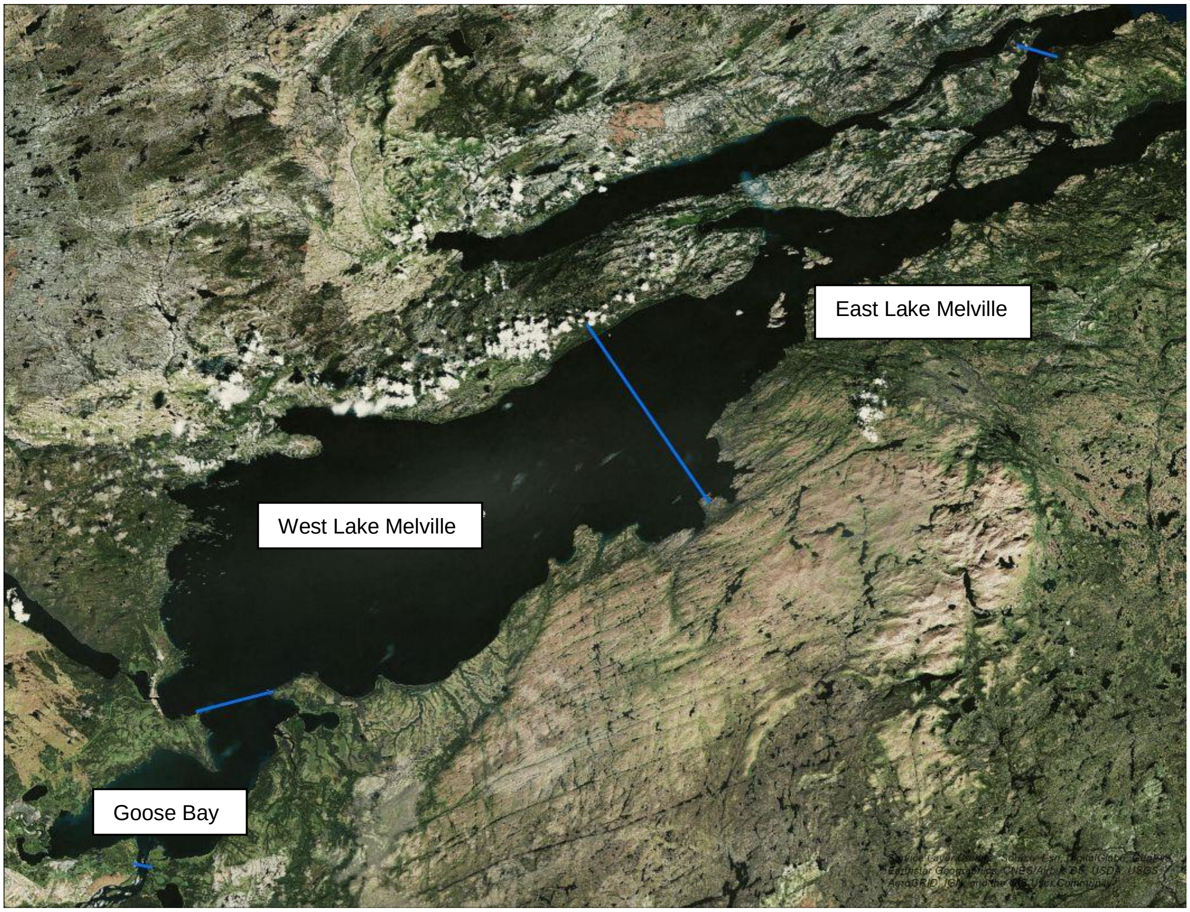


Figure 2-1: General overview of different zones of exposure based on hydrodynamic model (Brunton 2018)

Table 2-1: Hydrodynamic Model Estimates of water MeHg concentration (ng/L) increases (above baseline), Goose Bay, West Lake Melville, East Lake Melville

	Goose Bay	**West Lake Melville**	**East Lake Melville**
Baseline MeHg Water Concentration (ng/L)	0.017	0.017	0.017
Peak Additional Concentration (max 3-yr; ng/L)	0.019	0.006	0.005
Total Predicted Concentration (max 3-yr + baseline; ng/L)	0.036	0.023	0.022
Relative MeHg Increase in Water	**2.12x**	**1.35x**	**1.29x**

		Lower Churchill River	Goose Bay	Lake Melville West Basin	Lake Melville East Basin	Dietary Pathway Freshwater	Estuarine	Marine
Riverine / Estuarine	Longnose sucker	●	●	○	○	X	X	-
	Northern pike	◐	○	-	-	X	-	-
	Lake whitefish	○	○	-	-	X	X	-
	lake chub	◐	◐	◐	-	X	X	-
	Stickleback	●	◐	◐	◐	X	X	-
	Rainbow Smelt	○	◐	●	●	X	X	X
	Brook trout	○	◐	●	●	-	X	X
	Artic char	-	-	-	-	-	-	X
	Lake Trout	-	-	-	-	X	-	-
	Atlantic Cod	-	-	-	-	-	-	X
	Capelin	-	-	-	-	-	-	X
	Atlantic salmon	-	-	-	-	-	-	X
Estuarine / Marine Fish	Sand lance	-	◐	◐	◐	-	X	X
	Tom cod	-	◐	●	◐	-	X	X
	Rock cod	-	○	○	○	-	-	X
	flounder	-	○	○	◐	-	-	X
	Sculpin	-	○	○	○	-	-	X
	Blenny	-	○	○	○	-	-	X
Marine Mammals	Ringed Seals	○	●	●	●	-	X	X

-	Not present or not applicable
○	Present in low or negligible relative aboundance based on catch-per-unit-effort or stomach content analysis
◐	Present or believed to be present in relatively moderate relative aboundance based on catch-per-unit-effort or stor
●	Present or believed to be present in high relative aboundance based on catch-per-unit-effort or stomach content ar
X	Basis of dietary exposure to MeHg based on organism stable C and N isotope signature and on stomach contents

Figure 2-2: Relative abundance summary of key species captured in baseline sampling programs, 1998-2017.

Key species that have been identified in diet surveys include:

- lake trout (*Salvelinus namaycush*) – are not present outside of the river mouth and are very rare within the lower portion of the river (i.e. Muskrat Falls reservoir area)
- Atlantic salmon (*Salmo salar* – both anadromous and land-locked) – the landlocked form is very rare within the lower portion of the river (i.e., Muskrat Falls reservoir area) and anadromous returning salmon from the Labrador Sea cease feeding as they enter freshwater of Lake Melville
- Atlantic cod (*Gadus morhua*) – this species has not been documented within L. Melville
- Capelin (*Mallotus villosus*) – this species has only rarely been observed in Lake Melville since the early 1970s
- Arctic char (*Salvelinus alpinus*) – This species is not found in the lower Churchill River below the Labrador Plateau and only rarely observed in Lake Melville and typically found beyond the Narrows at the eastern end of Lake Melville.

Only three species; brook trout (*Salvelinus fontinalis*), rainbow smelt (*Osmerus mordax*) and ringed seal (*Pusa hispida*) appear to be abundant and widespread in Goose Bay and Lake Melville and perhaps not coincidentally, have also been identified in dietary surveys as preferred food species by local communities (Dillon 2016). These three species are therefore exposed to greater methylmercury concentrations in prey due to the Muskrat Falls reservoir because of their spatial and temporal overlap with the project. For species that do not feed within an area of Goose Bay / Lake Melville that is affected by methylmercury exported from the reservoir (i.e., within the zones of exposure), methylmercury concentrations will not change. Further details on each of these three species is as follows.

2.2.1 Brook Trout (*Salvelinus fontinalis*)

The brook trout is widely distributed throughout Newfoundland and Labrador (Scott and Crossman 1973), at least as far north as the Hebron Fiord (Black et al. 1986), where they have been reported to make extensive use of clear, cool (<20^{o}C) lake habitats (Ryan and Knoechel 1994). Brook trout are known to have both landlocked and anadromous populations throughout Newfoundland and Labrador (Scott and Crossman 1964, 1998). Anadromous populations may spend one or two months feeding at sea in relatively shallow water, close to their natal stream, while others spend their entire life in freshwater (Scott and Crossman 1964; Morrow 1980; Power 1980; Ryan 1980; Scott and Scott 1988).

Brook trout are found throughout the main stem and tributaries of the lower Churchill River between Muskrat Falls and Churchill Falls (Beak 1980; Ryan 1980; AGRA 1999; AMEC 2000, AMEC 2001), being most abundant upriver of Gull Island (above the Muskrat Falls reservoir area) where river and shoreline substrates contain less fine sand and clay substrates (AGRA 1999; AMEC 2000). Brook trout have also been captured below Muskrat Falls within the main stem but at relatively low rates (AMEC 2000; AMEC 2007; AMEC 2009; Amec Foster Wheeler 2015a; Amec Foster Wheeler 2016a).

Based on habitat utilization data, brook trout use stream (i.e. tributary) habitat where spawning and young-of-year occur. Few samples have been collected within the main stem of the lower Churchill River below Muskrat Falls where only 33 have been captured in a combination of fyke nets and gillnets between 1998-2016; however, they are found in relatively higher numbers within the upper habitat of Caroline Brook. Larger numbers have also been sampled within both Goose Bay (191 total) and Lake Melville (535). In both estuarine environments, brook trout have had some of the highest CPUE and biomass of all species sampled (Amec Foster Wheeler 2015a; 2016a). This is most likely the result of the brackish environment of the estuary being a suitable habitat for anadromous brook trout to feed during the summer months. Typically, brook trout will not feed within an estuarine environment beyond several kilometers of its natal stream (Scott and Scott 1988); therefore, most of the brook trout captured in Goose Bay and Lake Melville are likely not far from their home freshwater tributary.

Specimens have been captured from every age-class between one and six (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2015a; 2016a, 2016b). Mean length-at-age data shows they range between 82 mm in length at age one to almost 415 mm at age six. Growth is relatively linear throughout all years.

The diet of brook trout consists of a wide variety of food types including aquatic invertebrates, fish, and terrestrial invertebrates and vertebrates. Stomach content analysis and stable isotope data indicate that brook trout in the estuary feed primarily on marine prey such as sand lance (*Ammodytes americanus*), rainbow smelt, amphipods, and benthic invertebrates (see **Appendix B**). They are one of the top predators within the estuary food chain.

2.2.2 Rainbow Smelt (*Osmerus mordax*)

Rainbow smelt are typically a schooling, pelagic fish, inhabiting mid-water areas of inshore coastal waters (Leim and Scott 1966; Scott and Scott 1988; Scott and Crossman 1998). In Hamilton Inlet and Lake Melville, they are primarily an inshore anadromous species that occur within bays and estuaries, but are rare in the Churchill River freshwater system (Anderson 1985). They are an important species in that they feed on pelagic plankton and are an important food source for most estuarine piscivores such as gadids (e.g., cod species), flatfish (e.g., winter flounder) and salmonids (e.g. brook trout).

Smelt are typically anadromous, moving from estuaries such as Lake Melville and Goose Bay into nearby rivers and streams to spawn in the spring, likely before ice breakup (JWEL 2001). As the hatched larvae grow, they move into areas of higher salinity, such as deeper parts of the estuary or more coastal areas (JWEL 2001). Smelt begin to school at about 19 mm in length, moving into shallow water and returning to deeper channels during the day (Belyanina 1969). They will generally spend the summer feeding on copepods and planktonic larvae and in the fall, juveniles mix with adult schools and move into the upper parts of the estuary (Buckley 1989) where they remain for the winter.

Within Lake Melville, smelt seem to prefer deeper, cooler waters in the summer (JWEL 2001). The JWEL sampling program identified that smelt, which spend the summer in the cooler waters of Lake Melville, move into Goose Bay from August to October (JWEL 2001; AMEC/BAE 2001). There was a slight peak observed in abundance in October in the western portion of Lake Melville and was suggested to be the result of a migration toward the many rivers in the area (JWEL 2001).

Due to physical barriers, this species does not occur above Muskrat Falls in the Churchill River (Ryan 1980) and based on sampling, is very rare upstream of estuarine influences after spawning. Ryan (1980) recorded two specimens (which appeared to be anadromous) downstream of Muskrat Falls and Amec Foster Wheeler captured a lone adult by fyke net just downstream of Muskrat Island in 2016 (Amec Foster Wheeler 2016a). No other known reports occur in the literature for their presence within the freshwater portion of the lower Churchill River (Ryan 1980, Beak 1980, AGRA 1999, AMEC 2000) upstream of the Mud Lake confluence (AMEC 2000). In addition to sampling conducted related to the Project, the main stem between Happy Valley–Goose Bay and Muskrat Falls as well as several tributaries (eg. Birchy Creek and Caroline Brook), were sampled between 2006 and 2008 for the provincial Department of Transportation and Works. Sampling was conducted using fyke nets and tended gillnets through most open water months (i.e. July and October 2006, May and June 2007, April, May, and June 2008, and May 2009) but did not capture rainbow smelt (unpub. data).

Rainbow smelt have been routinely captured during ongoing baseline sampling since 1999 in both Goose Bay and Lake Melville. Sampling by Amec Foster Wheeler has captured approximately 136 and 155 from Goose Bay and Lake Melville, respectively. Baseline work completed by JWEL in 1998 captured a total of 991 rainbow smelt within Goose Bay / Lake Melville which comprised 31 percent of their total catch (JWEL 2001). Rainbow smelt sampled (AGRA 1998) were predominantly between 151-250mm in length with fairly linear growth through all age classes sampled (ages 1-8).

Stomach content analysis and stable isotope data indicate that like brook trout, rainbow smelt are one of the top predators within the estuary food chain and feed primarily on marine prey such as sand lance, other rainbow smelt, and amphipods/decapods (see **Appendix B**).

2.2.3 Ringed Seal (*Phoca hispida*)

The ringed seal is one of the most abundant and widely distributed resident Arctic pinnipeds (Muir et al. 1999). The following general species life history description is from Lowry (2016). As a species, ringed seals are widely distributed in ice-covered waters of the northern hemisphere, and they may presently number about three million animals (Lowry 2016). They prefer annual, landfast ice, but are also found in multi-year ice (Kingsley et al. 1985).

Throughout most of their range they use sea ice exclusively as their breeding, molting, and resting (haul-out) habitat, rarely if ever moving onto land (Frost and Lowry 1981, Reeves 1998). Reported mean age at sexual maturity for female Ringed Seals varies in the literature from 3.5 to 7.1 years (Holst and Stirling 2002, Krafft et al. 2006). Males likely do not participate in breeding before they are 8-10 years old. Ringed seals can be long lived, with ages close to 50 reported (Lydersen and Gjertz 1987). Regional productivity rates are variable; reproductive success depends on many factors including prey availability, the relative stability of the ice, and sufficient snow accumulation prior to the commencement of breeding (Lukin 1980, Smith 1987, Lydersen 1995).

Outside the breeding and molting seasons, Ringed Seal distribution is correlated with food availability (e.g., Simpkins et al. 2003, Freitas et al. 2008). Numerous studies of their diet have been conducted, and although there is considerable regional variation, several patterns emerge. Most Ringed Seal prey are small, and preferred prey tend to be schooling species that form dense aggregations. Fishes are usually in the 5-10 cm length range and crustacean prey in the 2-6 cm range. Typically, a variety of 10-15 prey species are found, with no more than 2-4 dominant prey species for any given area. Fishes are generally more commonly eaten than invertebrates, but diet is determined to some extent by availability of various types of prey during particular seasons as well as by preference, which in part is influenced by energy content of various available prey (Reeves 1998, Wathne et al. 2000). Commonly eaten prey includes cod species redfish, herring, and capelin in marine waters (Lowry et al. 1980, Holst et al. 2001, Labansen et al. 2007). Invertebrate prey species seem to become more important in the open-water season and often dominate the diet of young animals (Lowry et al. 1980, Holst et al. 2001). Large Amphipods, Krill, Mysids, Shrimps, and Cephalopods are all eaten by Ringed Seals and can be very important in some regions at least seasonally (Agafonova et al. 2007).

Ringed seal surveys in Goose Bay and Lake Melville have been completed in 2006 and each year between 2013-2016 (SEM 2007; Amec Foster Wheeler 2016a). During aerial surveys each whelping season, the lower reach of the Churchill River is flown for seal presence and in all years, no ringed seals have been recorded within the river itself (SEM 2007; Amec Foster Wheeler 2016a). Very few seals are observed within Goose Bay (Amec Foster Wheeler 2016a). However, it should be noted that harbour seals (*Phoca vitulina*) have been observed within the river during fisheries surveys during open water; the most observed at any location and time has been three (McCarthy, unpubl data). Using the seal density within the observed area (approximately 517km^2), a relative abundance estimate for the entire EEM zone was generated for each survey year. Relative abundances have ranged between 644 and 2,140 animals with the 2015 survey being the lowest to date (Amec Foster Wheeler 2016a). Seal ages in Goose Bay and Lake Melville, typically range between pups and adults up to 32 years of age. Since seal samples from Goose Bay and Lake Melville are harvested by a local hunter for consumption by the local community, samples are generally biased toward younger animals.

Stomach content analysis has only identified rainbow smelt as prey; however, seals are sampled after whelping and foraging may be more restricted. In addition, pups would only be feeding on milk. Stable isotope data indicate ringed seals are the top predator in the estuary (above brook trout and rainbow smelt) and therefore feed on a variety of marine fish species.

2.2.4 Exposure Summary

Table 2-2 provides a summary of the annual percentage of time spent feeding in the identified estuary zones (Goose Bay, West Lake Melville, and East Lake Melville) for brook trout, rainbow smelt, and ringed seal. This table provides estimates of the temporal overlap for each species within Goose Bay, West and

East Lake Melville, which are expected to have differential exposure to MeHg in water exported from the reservoir.

Table 2-2. Summary of estimated percent annual exposure of key species within the identified estuary zones.

Species	Habitat Not Influenced by Muskrat Falls	Goose Bay	West Lake Melville	East Lake Melville
Brook Trout	30%	70%	70%	70%
Rainbow Smelt	0%	20% - 100%	80%	80%
Ringed Seal	34%	0%	66%	

Brook trout remain very near their home stream but would feed within the estuary environment once reaching the age of three. Discussions with local fishers indicate that brook trout have been captured through the ice and therefore, it has been assumed that up to 70% of the year could be spent within the estuary environment with some (30%) overwintering in tributaries and upstream migration for spawning and feeding where no increases in MeHg exposure would occur. While they would not be anticipated to migrate between each of the estuary zones, the estimated annual exposure within each zone would be similar.

Rainbow smelt that live and are captured in Goose Bay / Lake Melville are assumed to spend their entire lives within this environment; that is, they do not migrate to Hamilton Inlet or further offshore. However, based on surveys of the area, it appears that many rainbow smelt congregate within Goose Bay for a couple of months in the fall. It was therefore assumed that rainbow smelt captured and consumed from the Lake Melville zones could have spent up to 20% of their time feeding within Goose Bay each year and this would increase their exposure to higher MeHg water concentrations. Those fish captured and consumed within Goose Bay are assumed to reside 100% within Goose Bay itself and therefore are predicted to have higher overall exposure than those captured within Lake Melville.

Ringed seals have not been observed within the Churchill River and Chaulk et al. (2013) stated that local residents reported that ringed seals are rarely observed in Lake Melville during the summer, compared to early spring. Chaulk et al. (2013) also noted that DFO (B. Sjare) was tracking seals in the area and the data suggested that ringed seals moved in and out of Lake Melville from other areas of coastal Labrador over the course of the ice-free period. While they are relatively abundant in Lake Melville in the winter, they are uncommon in Goose Bay based on surveys completed since 2006. Based on this available information, it is assumed that ringed seals captured and consumed from Lake Melville spend 66% of their time feeding there. An estimated 34% of their annual feeding would occur outside Lake Melville and therefore outside any exposure to increased water MeHg concentrations.

2.3 Predicted Increases in Fish MeHg Concentrations

Based on the predicted increases in MeHg concentrations in water within the three estuary zones (see **Table 2-1**) and the estimated time of exposure for key species (see **Table 2-2**), increases in fish MeHg muscle tissue were predicted (**Table 2-3**) using the product of the cumulative annual exposure to water predicted to have relative increases in MeHg concentration. Ringed seal liver tissue increases are also provided as this has also been identified as an important diet item (Calder et al. 2016; Dillon 2016). Note that the correlation between the increase in MeHg concentration in water and the increase in prey species concentration is assumed at a 1:1 ratio. This is a conservative assumption but acknowledges uncertainty in attempting to rationalize or justify a lower increase ratio relative to water. This also follows the assumption made in the Calder et al. (2016) paper, therefore comparisons to relative increases between the two can be considered. For species that do not feed within an area of Goose Bay / Lake Melville that is affected by methylmercury exported from the reservoir (i.e., within the zones of exposure), methylmercury concentrations will not change.

As stated previously, brook trout would remain near their home stream but would feed within the estuary environment once reaching the age of three. Since they would not migrate between each of the estuary zones, three separate predicted increases are provided; one for each zone where brook trout may be captured for consumption. As expected, brook trout are predicted to increase more in zones closer to Muskrat Falls. The predicted increases in brook trout tissue during the peak of MeHg in water (three-year max) are 78%, 25%, and 20% in Goose Bay, West Lake Melville, and East Lake Melville respectively.

Based on life history for rainbow smelt as described above and the relative increases in MeHg concentrations in water, the predicted increases during the peak of MeHg in water (three-year max) are 112%, 50%, and 46% in Goose Bay, West Lake Melville, and East Lake Melville respectively. These values are the weighted mean of the portion of time spent feeding in Goose Bay and each of the zones in Lake Melville (see Table 2-2). As noted above, we have assumed that the percent increase in water would ultimately be translated into a similar increase in biota concentrations of key species.

Based on the available life history information, it was assumed that ringed seals captured and consumed from Lake Melville spend 66% of their time feeding there with 34% of their time outside Lake Melville. It was also assumed that seals would freely move between the whole area of Lake Melville, therefore their predicted increase in MeHg would be the weighted mean of the two Lake Melville zones (equal exposure of 33% feeding time in each zone). The predicted increases during the peak of MeHg in water (three-year max) is therefore 21% throughout Lake Melville.

As shown, predicted increases are between 20-112% based on species habitat use and MeHg increases in each of the identified zones. It is critical that the role of life history, habitat, migratory habits, distribution and ecology interact with hydrodynamics to play a critical role in exposure of key species (rainbow smelt, brook trout, ringed seal) to increased MeHg concentrations exported from the Muskrat Falls reservoir. These predicted increases will be incorporated into exposure estimates within the Human Health Risk Assessment (HHRA).

Table 2-3: Summary of predicted increases in MeHg muscle tissue concentration in brook trout, rainbow smelt, and ringed seal

Species	Goose Bay			West Lake Melville			East Lake Melville		
	Predicted MeHg Increase	Baseline MeHg	**Predicted MeHg Conc (mg/kg)**	Predicted MeHg Increase	Baseline MeHg	**Predicted MeHg Conc (mg/kg)**	Predicted MeHg Increase	Baseline MeHg	**Predicted MeHg Conc (mg/kg)**
Brook Trout[a]	1.78x	0.07	**0.125**	1.25x	0.04	**0.050**	1.20x	0.03	**0.036**
Rainbow Smelt[b]	2.12x	0.02	**0.043**	1.50x	0.02	**0.030**	1.46x	0.04	**0.058**
Ringed Seal Tissue[a]	1.32x	-	**-**	1.21x	0.13	**0.157**	1.21x	0.13	**0.157**
Ringed Seal Liver[a]	1.32x	-	**-**	1.21x	13.42	**16.24**	1.21x	13.42	**16.24**

[a] mean MeHg tissue concentrations from 2017 samples.
[b] mean MeHg tissue concentrations from 2016 samples.

3.0 CLOSURE

The biological and habitat use data presented within this report has been compiled using baseline data collected by Wood and others since 1998. The methodologies used to collect and generate the data are generally accepted practices described in detail within the EEM and the Fish Habitat Compensation Plan baseline studies, and have been used for studies within the lower Churchill River, as well as other projects throughout Newfoundland and Labrador.

Yours truly,

Wood Environment & Infrastructure Solutions

Prepared by:

James H. McCarthy, MSc CFP
Associate Biologist and Ecosystem Group Lead

Reviewed by:

David A. Robbins, M.Env.Sci.
Senior Scientist

4.0 REFERENCES

Agafonova, E. V., Verevkin, M. V., Sagitov, R. A., Sipilä, T., Sokolovskay, M. V. and Shahnazarova, V. U.2007. The Ringed Seal in Lake Ladoga and the Valaam Archipelago. Baltic Fund for Nature of Saint- Petersburg Naturalist Society, St. Petersburg State University & Metsähallitus, Natural Heritage Servives, Vammalan Kirjapaino OY., Vammala, Finland.

AGRA Earth & Environmental. 1999. Fish and Fish Habitat, Churchill River Power Project (LHP98-07). Prepared for Labrador Hydro Project, St. John's, NF.

AMEC Earth & Environmental. 2000. Freshwater Fish Mercury Sampling, Churchill River, Labrador (LHP99-07). Prepared for Labrador Hydro Project, St. John's, NF.

AMEC Earth & Environmental. 2001. Churchill River Power Project: A Proposed Framework for HADD Determination (LHP00-07). Prepared for Labrador Hydro Project, St. John's, NF.

AMEC Earth & Environmental Limited. 2007. Lower Churchill Hydroelectric Development Fish and Fish Habitat Baseline Study: catch-based Utilization Index Validation and Additional Habitat Surveys. Prepared for Labrador Hydro Project, St. John's, NF ii + 54 pp. + Appendices

AMEC Earth & Environmental. 2009. Fish Passage Facility (FPF) Assessment of the Lower Churchill River Crossing, Trans Labrador Highway, Goose Bay, NL. Prepared for the department of Transportation and Works, Government of Newfoundland and Labrador, St. John's, NL.

AMEC Earth & Environmental and BAE-Newplan Group Limited. 2001. Aquatic Environment in the Goose Bay Estuary Final Report. Report prepared for Labrador Hydro Project, St. John's NL.

Amec Foster Wheeler. 2015a. Aquatic Environmental Effects Monitoring Program: 1998 to 2014 Baseline Conditions, Muskrat Falls. Report prepared for Nalcor Energy, St. John's NL

Amec Foster Wheeler. 2015b. Fish Habitat Compensation 1998 to 2014 Baseline Conditions, Muskrat Falls Reservoir Area. Report prepared for Nalcor Energy, St. John's NL

Amec Foster Wheeler. 2016a. Aquatic Environmental Effects Monitoring Program: 1998 to 2015 Baseline Conditions, Muskrat Falls. Report prepared for Nalcor Energy, St. John's NL

Amec Foster Wheeler. 2016b. Fish Habitat Compensation 1998 to 2015 Baseline Conditions, Muskrat Falls Reservoir Area. Report prepared for Nalcor Energy, St. John's NL

Anderson, J.L., and M.H. Depledge. 1997. A survey of total mercury and methylmercury in edible fish and invertebrates from Azorean waters. Marine Env. Res. 44(3): 331-350.

Anderson, M.R., D.A. Scruton, U.P. Williams, and J.F. Payne. 1995. Mercury in fish in the Smallwood Reservoir, Labrador, Twenty one years after impoundment. Water, Air, and Soil Pollution 0:927-930.

Anderson, M.R. 2011. Duration and extent of elevated mercury levels in downstream fish following reservoir creation. River Systems 19(3):167-176.

Anderson, T.C. 1985. The Rivers of Labrador. Can. Spec, Publ. Fish. Aquat. Sci. 81:389 p.

Azimuth. 2018. Evaluation of MeHg Production by Muskrat Falls Reservoir and Implications for Lake Melville – A Top-Down, Mass-Balance Approach. Technical Report submitted to Nalcor Energy, St. John's, NL.

Backus, R.H. 1957. The fishes of Labrador. Bulletin of the American Museum of Natural History 113: 277-337.

Beak. 1980. Fisheries Resources of Tributaries of the Lower Churchill River. Prepared by Beak Consultants Limited for the Lower Churchill Development Corporation, St. John's, NF.

Belyanina, T.N. 1969. Synopsis of biological data on smelt, Osmerus eperlanus (Linnaeus) 1758. Food and Agriculture Organization of the United Nations Fisheries Synopsis Number 78, 57p.

Black, G.A., J.B. Dempson, and W.J. Bruce. 1986. Distribution and postglacial dispersal of freshwater fishes of Labrador. Canadian Journal of Zoology 64: 21-31.

Brunton. A. 2018. Model analysis of Goose Bay and Lake Melville hydrodynamics and transport of methylmercury from Muskrat Falls Reservoir. Baird Technical Memorandum to R. Harris. July 2018.

Buckley, J.L. 1989. Species Profiles: Life histories and environmental requirements of coastal fishes and invertebrates (North Atlantic)—rainbow smelt. US Fish and Wildlife Service, Biological Report 82 (11.106). US Army Corps Engineers, TR EL-82-4. 11p.

Calder, R.S.D., A.T. Schartup, M. Li, A.P. Valberg, P.H. Balcom, E.M. Sunderland. 2016. Future Impacts of Hydroelectric Power Developments on Methylmercury Exposures of Canadian Indigenous Communities. Environm. Sci. Technol. 50(23): 13115-13122.

Chaulk, K.G., D. Michelin, M. Williams, T. Wolfrey. 2013. Community-based observations of marine mammal occurrences in Groswater Bay, Labrador. Canadian Field-Naturalist 127(1): 31-37.

Dillon. 2016. Final Baseline Human Health Risk Assessment: Lower Churchill Hydroelectric Generation Project. Submitted to Nalcor Energy, St. John's, NL. 213pp + appendices.

Freitas, C., Kovacs, K.M., Ims, R.A., Fedak, M.A. and Lydersen, C. 2008. Ringed seal post-moulting movement tactics and habitat selection. Oecologia 155: 193-204.

French, K.J. 1997. Mercury Evolution in Freshwater Fish from Reservoirs in Newfoundland, Canada. A thesis submitted to the School of Graduate Studies in partial fulfilment of the requirements for the degree of Master of Science (Biology), Memorial University of Newfoundland, St. John's, 125pp + appendices.

Frost, K. J. and Lowry, L. F. 1981. Ringed, Baikal and Caspian seals *Phoca hispida* Schreber, 1775; *Phoca sibirica* Gmelin, 1788; and *Phoca caspica* Gmelin, 1788. In: S. H. Ridgway and R. Harrison (eds), Handbook of marine mammals, Vol. 2: Seals, pp. 29-53. Academic Press.

Gjertz, I. and Lydersen, C. 1986. Polar bear predation on ringed seals in the fast-ice of Hornsund, Svalbard. Polar Research 4: 65-68.

Hall, B.D, R.A. Bodaly, R.J.P. Fudge, J.W.M. Rudd, D.M. Rosenberg. 1997. Food as the Dominant Pathway of Methylmercury Uptake by Fish. Water, Air, and Soil Pollution. 100: 13-24.

Hall, B.D., V.L. St. Louis, K.R. Rolfhus, R.A. Bodaly, K.G. Beaty, M.J. Paterson, K.A. Peech Cherewyk. 2005. Impacts of Reservoir Creation on the Biogeochemical Cycling of Methylmercury and Total Mercury in Boreal Upland Forests. Ecosystems 8(3): 248-266.

Harris, R.C. and D. Hutchinson. 2018. Updated mercury modeling analysis for Muskrat Falls Reservoir. Technical Memorandum to P. Madden, July 2018

Holst, M., Stirling, I. and Hobson, K. A. 2001. Diet of ringed seals (*Phoca hispida*) on the east and west sides of the North Water Polynya, northern Baffin Bay. Marine Mammal Science 17(4): 888-908.

Holst, M. and Stirling, I. 2002. A comparison of ringed seal (*Phoca hispida*) biology on the east and west sides of the North Water Polynya, Barfin Bay. Aquatic Mammals 28: 221-230.

Jackson, T.A. 1988. The Mercury Project in Recently Formed Reservoirs of Northern Manitoba (Canada): Effects of Impoundment and Other Factors on the Production of Methyl Mercury by Microorganisms in Sediments. Can. J. Fish. Aquat. Sci. 45:97-121.

JWEL. 2001. Biological Study of the Goose Bay Estuary. Prepared by Jacques Whitford Environment Limited, St. John's for Newfoundland and Labrador Hydro, St. John's, NL. JWEL Project No. 1201. 91p + appendices.

Krafft, B. A., Kovacs, K. M., Frie, A. K., Haug, T. and Lydersen, C. 2006. Growth and population parameters of ringed seals (*Pusa hispida*) from Svalbard, Norway, 2002-2004. ICES Journal of Marine Science 63: 1136-1144.

Kingsley, M.C.S., I. Stirling, and W. Calvert. 1985. The distribution and abundance of seals in the Canadian high Arctic, 1980-82. Can J. Fish. Aquat. Sci. 42:1189-1210.

Labansen, A. L., Lydersen, C., Haug, T. and Kovacs, K. M. 2007. Spring diet of ringed seals (*Pusa hispida*) from north-western Spitsbergen, Norway. ICES Journal of Marine Science 64: 1246-1256.

Lasorsa, B. and S. Allen-Gil. The Methylmercury to Total Mercury Ration in Selected Marine, Freshwater, and Terrestrial Organisms. Water, Air, and Soil Pollution 80: 905-913.

Leim, A. H. and W.B. Scott . 1966. Fishes of the Atlantic coast of Canada. Fisheries Research Board of Canada Bulletin Number 155, 485p.

Li, M. A.T. Schartup, A.P. Valberg, J.D. Ewald, D.P. Krabbenhoft, R. Yin, P. H. Balcom, and E.M. Sunderland. 2016. Environmental Origins of Methylmercury Accumulated in Subarctic Estuarine Fish Indicated by Mercury Stable Isotopes. Environ. Sci. Technol. 50: 11559-11568.

Lowry, L. 2016. *Pusa hispida*. The IUCN Red List of Threatened Species 2016: e.T41672A45231341. http://dx.doi.org/10.2305/IUCN.UK.2016-1.RLTS.T41672A45231341.en. Downloaded on 27 January 2017.

Lowry, L. F., Frost, K. J. and Burns, J. J. 1980. Variability in the diet of ringed seals, *Phoca hispida*, in Alaska. Canadian Journal of Fisheries and Aquatic Sciences 37: 2254-2261.

Lukin, L. R. 1980. Habitat of White Sea ringed seal in the initial period of postnatal development. Soviet Journal of Marine Biology 6: 266-269.

Lydersen, C. 1995. Energetics of pregnancy, lactation and neonatal development in ringed seals (*Phoca hispida*). In: A. S. Blix, L. Walløe and Ø. Ulltang (eds), Developments in Marine Biology 4: Whales, seals, fish and man, pp. 319-327. Elseview, Amsterdam, The Netherlands.

Lydersen, C. and Gjertz, I. 1987. Population parameters of ringed seals (*Phoca hispida* Schreber 1775) in Svalbard area. Canadian Journal of Zoology 65: 1021-1027.

Marrugo-Negrete, J., J.O. Verbel, E.L. Caballos, L.N. Benitez. 2007. Total mercury and methylmercury concentrations in fish from Mojana region of Columbia. Environ Gechem Health 30(1): 21-30.

Morrow, J.E. 1980. The freshwater fishes of Alaska. Alaska Northwest Publishing Co., Anchorage, Alaska, 248p.

Muir, D., B. Braune, B. DeMarch, R. Morstrom, R. Wagemann, L. Lockhart, B. Hargrave, D. Bright, R. Addison, J. Payne, K. Reimer. 1999. Spatial and temporal trends and effects of contaminants in the Canadian Arctic marine ecosystem: a review. Science of the Total Environment 230:83-144.

Nalcor, 2009. Lower Churchill Hydroelectric Generation Project Environmental Impact Statement, Volume II Part A: Biophysical Assessment. Nalcor Energy, St. John's, NL.

Power, C. 1980. The brook charr, p. 141-203. In E.K. Balon (ed.), Charrs, Salmonid Fishes of the Genus Salvelinus. Dr. W. Junk Publishers, The Hague, Netherlands.

Reeves, R. R. 1998. Distribution, abundance and biology of ringed seals (*Phoca hispida*): an overview. NAMMCO Scientific Publications 1: 9-45.

Ryan, P.M. 1980. Fishes of the Lower Churchill River, Labrador. Fisheries and Marine Service Technical Report No. 922.

Ryan, P.M., and R. Knoechel. 1994. Lake use by brook trout, Salvelinus fontinalis, in insular Newfoundland, Canada. Verhandlungen der Internationalen Vereinigung fur Theoretische und Angewandte Limnologie 25: 2068-2073.

Schartup, A.T., P. H. Balcom, A.L. Sorensen, K.J. Gosnell, R.S.D. Calder. R.P. Mason, and E.M. Sunderland. 2016. Freshwater discharges drive high levels of methylmercury in Arctic marine biota. PNAS 112(38): 11789-11794.

Schetagne, R., J-F. Doyon, J-J Fournier. 2000. Export of mercury downstream from reservoirs. The Science of the Total Environment 260: 135-145.

Scott W. B. and E.J. Crossman. 1964. Fishes occurring in the fresh waters of insular Newfoundland. Queen's Printer, Ottawa, Ontario. iv + 124 p.

Scott, W.B. and E.J. Crossman. 1998. Freshwater Fishes of Canada. Galt House Publications, Ontario. 966pp. ISBN 0-9690653-9-6.

Scott, W.B., and M.G. Scott. 1988. Atlantic fishes of Canada. Canadian Bulletin of Fisheries and Aquatic Sciences Number 219, 731p.

Sellars, P., C.A. Kelly, J.W.M. Rudd, A.R. MacHutchon. 1992. Photodegredation of Methlymerucry in lakes. Nature 380: 694-697.

SEM. 2007. Lower Churchill Hydroelectric Generation Project Environmental Baseline Report: Seal Abundance and Distribution. Prepared by Sikumiut Environmental Management, St. John's, NL for Labrador Hydro Project, St. John's, NL. 16p + appendices.

Simpkins, M.A., Hiruki-Raring, L.M., Sheffield, G., Grebmeier, J.M. and Bengtson, J.L. 2003. Habitat selection by ice-associated pinnipeds near St. Lawrence Island, Alaska in March 2001. Polar Biology 26:577-586.

Smith, T. G. 1987. The ringed seal, *Phoca hispida*, of the Canadian western Arctic. Canadian Bulletin of Fisheries Research and Aquatic Sciences 216.

Wathne, J. A., Haug, T. and Lydersen, C. 2000. Prey preference and niche overlap of ringed seals *Phoca hispida* and harp seals *P. groenlandica*. Marine Ecology Progress Series 94: 233-239.

Appendix A – Aquatic Species Habitat Overview, Churchill River, Goose Bay and Lake Melville, 1998-2016

Appendix B – Summary of Isotope and Stomach Data, Goose Bay / Lake Melville

Aquatic Species Habitat Utilization Overview

Churchill River, Goose Bay, and Lake Melville
1998-2016

Submitted to:

Nalcor Energy

Submitted by:

Jim McCarthy
Wood
Environment & Infrastructure Solutions
133 Crosbie Road
St. John's, NL
A1B 4A5

July 2018

Issue No.	Date (d/m/y)	Issue Description	Prepared by	Checked by	Review by
0	16/03/2017	Draft for review	J McCarthy	M Gosse	D Robbins / P Madden
1	2/02/2018	Draft submission to IEC for review and Calder incorporation	J McCarthy	D Robbins	
2		Draft submission to Nalcor	J McCarthy	M Gosse	JB Dempson
3	12/07/2018	Revised and appended to Tech memo	J McCarthy	D Robbins	P Madden

Table of Contents

List of Figures

List of Tables

1.0 INTRODUCTION

Nalcor Energy (Nalcor) is developing the remaining hydroelectric potential of the lower Churchill River through hydroelectric generating facilities at Muskrat Falls and Gull Island. The Muskrat Falls portion of the project, which is currently under construction, will result in the creation of a reservoir with a surface area of $101km^2$. The existing river within the proposed footprint of the Muskrat Falls reservoir area has a surface area of ~$60km^2$ therefore the area of additional terrestrial flooding will be approximately $41km^2$, representing a 65-70 percent increase in the existing waterbody surface area.

Many freshwater, estuarine, and marine fish species are within the project's zone of influence and could therefore be affected either directly or indirectly. Much of the baseline data required for the Environmental Assessment and Environmental Effects Monitoring (EEM) program described these species, their potential for interaction with the project, as well as the estimation of potential effects. Interactions between the project and local residents through downstream methylmercury uptake by various species have been modeled and included in Nalcor's Human Health Risk Assessment (HHRA) (Dillon 2016). Simultaneous to this, additional assessments of mercury increase and potential human effects have been published (see Schartup et al. 2016; Calder et al. 2016).

1.1 Purpose

The purpose of this document is to provide additional species summary information related to the species identified within the HHRAs. This information will be helpful in ongoing discussions with local communities and further analysis of potential human risk. The species habitat use information included by this dataset has been used to modify potential species methylmercury exposure related to project effects, both within and downstream of the reservoir.

2.0 STUDY AREA

Figure 2-1 provides a general overview of the Churchill River watershed and the various study regions (e.g., Smallwood Reservoir, Muskrat Falls Reservoir, lower Churchill River, Goose Bay, Lake Melville) where sampling has occurred.

Nalcor has collected baseline data since 1998 on the lower Churchill River, Goose Bay, and Lake Melville. Included in this baseline data are the ongoing results of total mercury concentrations in fish and seal samples. There has also been additional sampling and analysis prior to 1998 as a result of monitoring/research related to the larger Churchill Falls Hydroelectric Development located upriver that was completed in 1974. Fisheries and Oceans Canada (DFO) have also collected data on the Churchill River (e.g., Ryan 1980, Anderson 2011) and this has also been incorporated into this baseline description, where possible.

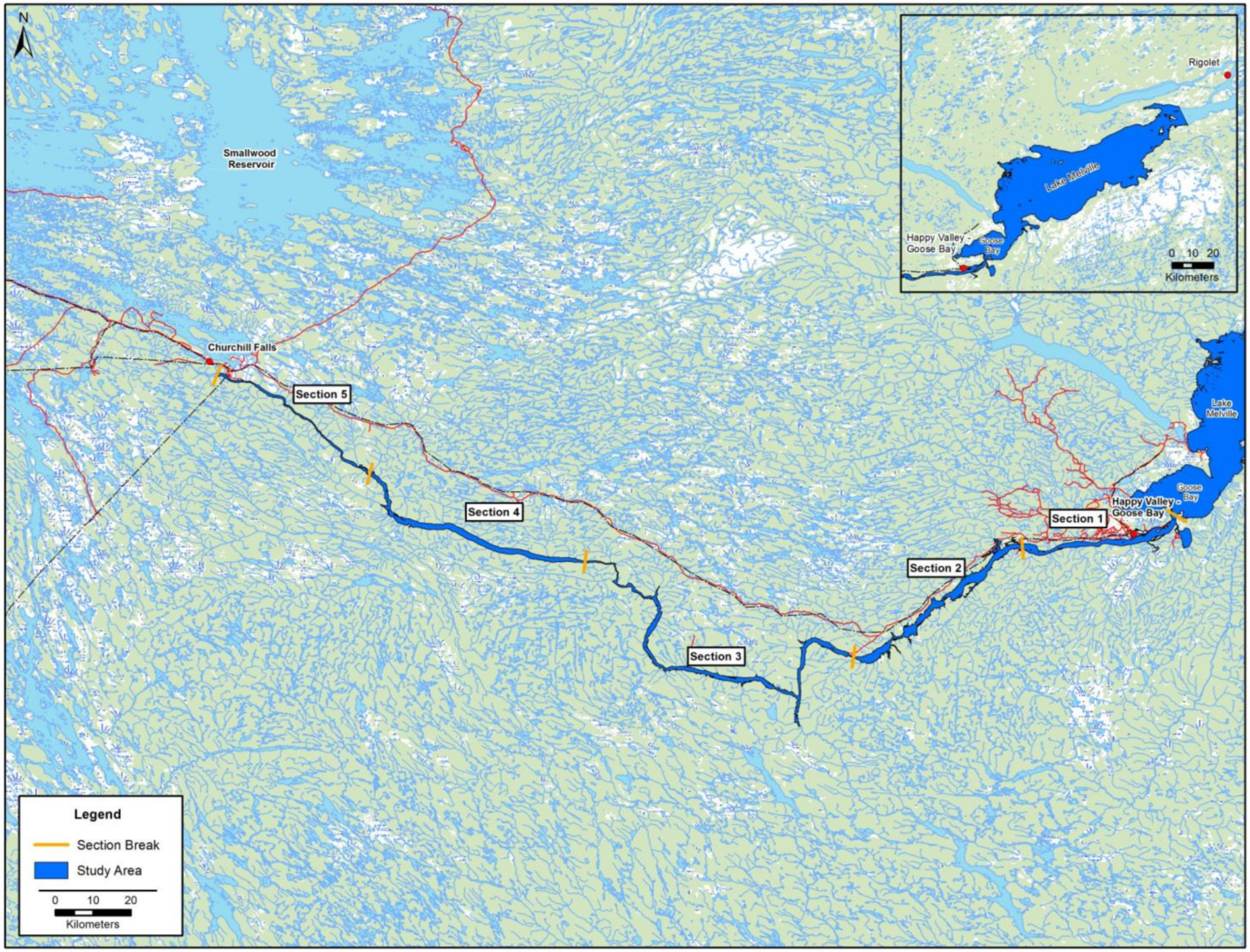


Figure 2-1: Overall baseline study area: mainstem of the lower Churchill River, Goose Bay, and Lake Melville.

Most sampling within the lower Churchill River between the existing Churchill Falls facility and Goose Bay was completed and catalogued by larger river sections with similar habitat conditions. A brief overview of each study section is provided below. Detailed habitat characterization is provided in various reports issued by Nalcor (e.g., AGRA 1999, AMEC 2001, AMEC 2013a). Sampling specifically associated with the Muskrat Falls portion of the Project has been concentrated both within the Muskrat Falls Reservoir Area (Section Two) and downstream of Muskrat Falls. Downstream of Muskrat Falls includes the lower section of the river to English Point at its outflow to Goose Bay Estuary (Section One) as well as Goose Bay Estuary and Lake Melville (Figure 2-1).

The riverine portions of the study area have been sampled much more intensely than the estuarine areas of Goose Bay and Lake Melville; however, these areas have been expanded upon since 2013 and now include fish sample locations just west of Rigolet. The river above the Muskrat Falls Reservoir area (Sections Three, Four and Five) have also been sampled but to a lesser extent.

2.1 Goose Bay and Lake Melville

Lake Melville is a tidal lake/fiord containing brackish waters located at latitude approximately 54° North, along the Labrador coast. Its length is approximately 130km, with a width of 30km near its western end and a maximum depth in excess of 180m. Included within Lake Melville is "The Backway", an arm of the lake extending for approximately 30km from the eastern boundary with depths again over 180m (Bobbitt and Akenhead 1982).

A large portion of the Labrador Plateau (Ungava Peninsula) drains into Lake Melville, with the largest watershed feeding it being the Churchill River, which flows into Lake Melville via Goose Bay Estuary. Goose Bay is a western extension of Lake Melville, situated at its southwest corner and extending for 25km. Goose Bay is approximately 55m deep and connected to Lake Melville by a 2.5km wide, 6m deep channel known as the Goose Bay Narrows (Bobbitt and Akenhead 1982; AMEC- BAE Newplan 2001).

Freshwater input from several rivers, plus the deep basins of Goose Bay and Lake Melville, form a layered saline system with freshwater tending to flow seaward at the surface and saline coastal waters entering the inlets in deeper layers (Bobbitt and Akenhead 1982; Schartup et al. 2016). The thin surface water layer, typically with salinities of less than 10, mixes very slowly in Lake Melville. The salinity changes to approximately 25 below a very sharp halocline at approximately 25m water depth. The mixing and exchange of water will depend on the density (salinity) of the water at the sill depth. Shallow sills at the Lake Melville Narrows (near Rigolet at the mouth of Lake Melville) and at the mouth of Goose Bay, significantly restrict water movement, resulting in a tidal range within Goose Bay of 0.3 to 0.6m, compared to 1.2 to 1.8m along the coast (Bobbitt and Akenhead 1982).

The water in Goose Bay and Lake Melville is warmer than on the Labrador shelf at comparable depths as the sill depth at the Narrows to Lake Melville prevents the colder shelf water from entering the lake (Bobbitt and Akenhead 1982). Temperatures recorded in the thin surface layer of Lake Melville have been up to 15°C, whereas the surface water on the Labrador shelf is typically only slightly above 5°C. Below the sharp thermocline in Lake Melville, the water is close to -0.5°C, whereas on the shelf there is a core of -1.5°C water between 50 and 100m (Vilks and Mudie 1983). Similar temperature patterns have been observed in Goose Bay, as illustrated by the results of conductivity, temperature, depth (CTD) water profiles (AMEC- BAE Newplan 2001).

Prior to development of the Churchill Falls Generating Facility, the Churchill River contributed 50-80% of the total freshwater inflow to Goose Bay. During winter, most of the water in the Labrador Basin (drainage basin feeding the Churchill River) would freeze and cause a drastic seasonal decrease in fresh water inflow (Coachman 1953 in Bobbitt and Akenhead 1982). Since the Churchill Falls Generating Facility development, there has been a notable change in the freshwater inflow into Goose Bay Estuary. The greatest difference occurs in the winter, December to April, where the flow rates have approximately tripled, whereas during June and July, rates have decreased by about a third (Bobbitt and Akenhead 1982).

Glaciomarine mud, comprising clay, silt and some fine sand, is the dominant sediment deposited in the Goose Bay Basin. At the outlet of the Churchill River into Goose Bay, a large semi-submerged delta comprised of sand, silt and clay has formed from the erosion activities upstream. Sieve analysis has demonstrated that the depositional sequence has the heavier sand remaining close to shore with progressive deposition of finer material further out into the basin, with the very fine clays being carried out into Lake Melville (Amec-BAE 2001).

2.2 River Section One

Section One of the river is approximately 43 km long and includes the freshwater main stem between the mouth of the river (English Point) at Goose Bay Estuary and Muskrat Falls (Figure 2-2). The segment is relatively slow flowing (mean water velocity of 0.5m/s), deep (mean water depth of 9.1m), wide (mean width of 1,561m) and a bottom substrate composition almost entirely of mobile sand and smaller material. The surficial geology of the material is fluvial and/or eolian in nature (Minaskuat 2008). The shoreline in some sections is lined/armoured with larger material such as rubble, cobble and boulder, which has been exposed by shoreline erosion (AMEC 2013a).

Acoustic Doppler Current Profiler (ADCP) testing of the river bottom substrate for bed movement near the Trans Labrador Highway's Black Rock Bridge indicates that the substrate is mobile (AMEC 2009). This would make this river section a challenge for benthic macroinvertebrate and fish species that rely on stable, larger substrate particularly for cover and spawning. This river segment is also very rich in suspended sediments compared to those further upriver and currently experiences considerable variation in Total Suspended solids (TSS) concentrations. Suspended sediment concentrations have been recorded from <2 to 1570mg/L within this area, with a mean of approximately 66mg/L. Highest concentrations are typically measured during late winter and spring when runoff from the watershed typically increases (Minaskuat 2007; Amec Foster Wheeler 2016a). The sandy substrate also results in naturally increased turbidity.

Larger tributaries draining into this section include Caroline Brook and the Traverspine, Peter Jackies and McKenzie Rivers.

Figure 2-2: Typical shoreline and bottom substrate, Section One Churchill River.

2.3 River Section Two (future Muskrat Falls Reservoir Area)

Section Two of the river is approximately 58 km long and includes the main stem between Muskrat Falls and Gull Island (i.e. the proposed Muskrat Falls reservoir location). This segment is also relatively slow flowing compared to other river sections (estimated mean water velocity of 1.3m/s), shallow (estimated mean water depth of 6.0m), wide (mean width estimated at 1,030m), and a bottom substrate composition dominated by sand and finer material (85% sand). While ADCP tests for bottom movement have not been conducted within this river section, similar substrates and slightly higher velocities would indicate that a similar substrate dynamic to that in Section One would be present. Similar to Section One, the surficial geology of the material is primarily fluvial and/or eolian in nature (Minaskuat 2008). Figure 2-3 presents typical shoreline and substrate conditions in this river section. Similar to habitat below Muskrat Falls, this section is also very rich in suspended sediments compared to those further upriver. In particular, suspended sediment concentrations have been recorded from <2 to 1170mg/L within this area, with a mean of approximately 42mg/L. Highest concentrations were measured during late winter and spring when runoff from the watershed typically increases (Minaskuat 2007; Amec Foster Wheeler 2016a). These lower reaches of the river are also primarily comprised of sandy substrate, resulting in naturally increased turbidity.

While the majority of the river segment is shallow, Gull Lake is relatively deep (greater than 50m). Gull Lake is also maintained by the same frazil ice process as that described above for the pool below Muskrat

Falls. In this respect, it too contains limited winter refuge for fish as it is filled with ice and velocities greater than that typically found in a large pool.

The most complex ice processes in the Churchill River generally occur between Gull Island and Goose Bay (Hatch 2007). The portion of the Churchill River downstream of Gull Island to Muskrat Falls typically has enough water velocity to prevent an ice cover from forming, except for border ice, and stationary ice covers at the slow-flowing stretches at Sandy Island Lake and Gull Lake (Hatch 2007). The open fast-flowing water generates large amounts of frazil, slush and pan ice, which are then carried downstream. Below Muskrat Falls, the drifting ice becomes trapped under the edge of a stationary ice cover which forms between Muskrat Falls and Goose Bay typically by the end of November. This causes a massive ice jam, backing up the river flow, raising the upstream water level and decreasing velocity. In some years this permits an ice cover to develop and progress upstream (Hatch 2007). During spring breakup, the ice cover upstream of the jam is rapidly eroded by the fast-flowing water, but the jam takes longer to melt away. On average, the ice is completely broken up by the end of May (Hatch 2007).

Larger tributaries emptying into this main stem section include Edward's Brook, Lower Brook, Upper Brook and Pinus River.

Figure 2-3: Typical shoreline and bottom substrate, Section Two Churchill River.

2.4 River Section Three

Section Three is approximately 119 km long and begins to flow through bed material that is primarily upriver of the heavy marine sand deposits found throughout the lower sections. The surficial geology of the river bed and shoreline material is more colluvial and/or glaciofluvial in nature (Minaskuat 2008). This segment is faster flowing (estimated mean water velocity of 1.9m/s) with similar water depths (estimated mean water depth of 8.2m) to previous sections. The estimated mean width is narrower (293m) as a result of less-erodible shoreline material. The bottom substrate composition in this section is dominated by larger material such as boulders, rubble and cobble. Figure 2-4 presents typical shoreline and substrate conditions in this section of river. As expected with a reduced source of finer material, TSS in this river section is much reduced in relation to that measured further downriver. Sample measurements have ranged between <1 and 39mg/L with a mean of approximately 6 mg/L (Minaskuat 2007; Amec Foster Wheeler 2016a).

Open water persists through the winter between Winokapau Lake and Gull Lake. Ice pans are transported as far as Gull Lake where they become trapped at an ice jam formed at a stationary ice cover (Hatch 2007).

Larger tributaries emptying into this main stem section include Bob's Brook, Minipi River, Beaver Brook, Cache River and Shoal River.

Figure 2-4: Typical shoreline and bottom substrate, Section Three Churchill River.

2.5 River Section Four (Winokapau Lake)

Section Four consists of Winokapau Lake which is approximately 46km long and approximately 1,266m wide. The shoreline of the lake is generally very steep and consisting of bedrock. In terms of littoral habitat, most is located at the inflow (near Elizabeth River), outflow and around a small spur of land on the north side of the lake (named Long Point). The littoral material in these areas generally consist of gravel-sized substrate and larger. The maximum water depth of Winokapau Lake is over 200m and hence the flow through the segment is slow. The thermocline within the lake, when one forms, is near 25m water depth. The estimated mean water velocity is 0.03m/s. The bottom substrate composition in this section is predominantly silt with some sand and clay material. Winokapau Lake bottom sediment contains higher concentrations of trace elements, nutrients and carbon compared to the rest of the river (Minaskuat 2007). Overall, sediment quality is good throughout the river, and only nickel concentrations in portions of Winokapau Lake exceeded Sediment Quality Guidelines for the Protection of Aquatic Life probably effect level (PEL), or other relevant benchmark values (Minaskuat 2007). Figure 2-5 presents typical shoreline and substrate conditions in this section of river. As expected with a reduced source of finer material, TSS in this river section is much reduced in relation to that measured further downriver. Sample measurements range between <5 and 7mg/L (Minaskuat 2007).

Figure 2-5: Typical shoreline substrate, Section Four Churchill River.

Lake Winokapau is normally covered by a stationary ice cover from November through to the end of May. This ice cover typically melts in place (Hatch 2007).

The only large tributary that empties into this main stem section is Fig River.

2.6 River Section Five

Section Five is approximately 70 km long and begins at the inflow of Winokapau Lake and extends upriver to the tailrace of the Churchill Falls Generating Facility. The river flows through a single, straight channel, passing through a narrow valley approximately 300m below the surrounding uplands. Similar to Section Three, the river flows over bed material that is primarily upriver of the heavy marine sand deposits found throughout the lower sections. The surficial geology of the river bed and shoreline material is more colluvial and/or glaciofluvial in nature (Minaskuat 2008). This segment is similar in estimated mean water velocity as Section Two (estimated mean water velocity of 1.1m/s) but has similar estimated mean water depths (8.4m) to that of Section Three (i.e. deeper than Section Two). The estimated mean width is 438m, similar to Section Three, as a result of similar shoreline material. The bottom substrate composition in this section is dominated by material such as rubble and cobble. Figure 2-6 presents typical shoreline and substrate conditions in this section of river. As expected with a reduced source of finer material, TSS in this river section is much reduced in relation to that measured further downriver. Sample measurements range between <5 and 9mg/L (Minaskuat 2007).

Figure 2-6: Typical shoreline and bottom substrate, Section Five Churchill River.

Upstream of Winokapau Lake, the river is mostly ice covered from November to April. Open water patches have been observed in the upper end of the reach closest to the Churchill Falls Generating facility, likely due to residual heat in the generating station discharge (Hatch 2007).

Larger tributaries emptying into this main stem section include Elizabeth and Metchin Rivers.

2.7 Churchill Falls Hydroelectric Development (Smallwood Reservoir)

Upriver of Section Five is the Churchill Falls Hydroelectric Development; approximately 240km upriver from Muskrat Falls. The Churchill Falls Hydroelectric Development includes the Smallwood Reservoir system that was flooded between 1971-73. The total area of the Smallwood reservoir is estimated at 5,000 km^2 and includes approximately 2,450 km^2 of unharvested forest, bog, and taiga (Anderson 2011). Approximately 75% of all flow from the lower Churchill River comes from the Smallwood Reservoir (Anderson 2011). Water levels within the Smallwood Reservoir typically fluctuate by three metres annually with an overall range of approximately nine metres (CFLco, unpublished data).

This area has been studied to a lesser extent in recent years; however, sampling of select fish species was completed in 2017 and will be included in the ongoing database when available.

3.0 SAMPLING METHODS

Within most of the study area, sampling for species presence, relative abundance, and population metrics has primarily been completed using a combination of live-capture fyke nets, gillnets, electrofishing, and night snorkeling. Complete descriptions of the methods are available in the Lower Churchill Hydroelectric Generation Project Aquatic Environmental Effects Monitoring (EEM) Program; Muskrat Falls (AMEC 2013a). In addition to methods carried forward during the baseline EEM program, beach seining and otter trawls were completed in 1998 within Goose Bay Estuary and Lake Melville (JWEL 2001). In addition, it is noted that radio telemetry tracking of several species within the lower Churchill River was also completed (JWEL 2000) and relevant movement information has been provided within species overviews.

3.1 Fyke Nets

In the anticipation of a long-term monitoring program associated with the project, a shift in primary sampling method was necessary. During the 2013 sampling program, fyke nets, a live capture sampling method, became the predominant technique employed throughout the riverine habitats included in the Lower Churchill Project's aquatic monitoring programs. Prior to the 2013 program, experimental gillnets were the primary sampling technique (see Section 3.2).

Fyke nets are a form of passive sampling, which is generally non-destructive, meaning the majority of fish captured can be live released following processing. Processing includes the collection of lengths, weights and identification to species. Fyke nets used for this program are the double-bag type that have been manufactured specifically for the program so that are all similar dimensions and sampling gear remains consistent from year to year.

As a means of reducing sampling bias, fyke nets are set in random locations; chosen through GIS. Typically, each is set in relatively shallow water habitats (less than two metres water depth) and secured to shore; however, they have been deployed at variable depths. The lead lines and traps are deployed perpendicular to the shoreline. Depending on the strength of the flow and current, they may be set in the lee of small islands or points within the larger main stem. The lead lines and the traps sit on the bottom and range in height between 0.5-1.5m therefore they sample moving fish both along the bottom and within the water column. Fyke nets are generally set (e.g., a net-night) for at least a 16-hour duration, which will encompass the dusk to dawn period, when fish movement is generally more prevalent. Sampling during these times has been consistent throughout the sampling program since 1999 when this gear type was first included.

3.2 Gillnets

As outlined in the EEM Program (AMEC 2013a), fyke nets have become the primary sample technique to monitor fish within the riverine habitats throughout the Lower Churchill River. Since 2013, gillnets have only been included as a means of augmenting fish collection for mercury analysis. Gillnets remain the primary sampling technique employed in Goose Bay and Lake Melville due to the need for mercury samples and the physical habitat limitations of each sampling area.

Scientific gillnets comprise a series of six separate panels each of different gillnet mesh size ranging from 13mm (0.5 inch) to 127mm (5 inch). As a means of reducing bycatch of non-target species outlined in the EEM Program, the two smaller panels (13mm and 25mm) were removed from gillnet sets prior to deployment since 2015. Similar to fyke nets, gillnets are typically set (e.g., a net-night) for at least 16 hours to cover the dawn and dusk periods. Data collected included those similar to fyke nets, and included the collection of mercury samples and various samples related to fish health monitoring. Gillnets have been used to collect data since 1998 related to the Lower Churchill Hydroelectric Development but have also been used in the Churchill River system since the Smallwood Reservoir was created in 1974.

3.3 Electrofishing

Electrofishing is a standard sampling method that provides data on fish habitat utilization, species presence/absence and standing stocks. The primary limitation with electrofishing is the habitat types where it is most suitable; smaller and shallower streams and deltas where barrier nets can be established and wading with the electrofishing unit is possible. As a result, this method is best suited to tributary deltas and streams.

Standard quantitative electrofishing stations are completed in the lower Churchill River at select sites as outlined in previous surveys. In addition to quantitative stations, index sites (standard 300 second sweeps) are also completed to provide greater overall sample coverage for fish species utilization and presence. Stations are completed during late summer (August-September) as per existing sampling so that values are comparable between sample years. In order to maintain consistency within datasets from year to year, population and biomass estimates are also normalized to one habitat unit ($100m^2$).

3.4 Snorkel Surveys

Electrofishing and other passive sampling methods (e.g., fyke nets) generate very useful data in terms of the overall utilization of fish life-cycle stages within various habitat types but they do not provide data on whether each species life-cycle stage is utilizing specific habitat features such as a particular substrate size, velocity or water depth. This may be particularly useful in determining specific habitat use as well as the number of fish observed within that habitat. Snorkel surveys are a useful method to determine species presence and habitat use within specific nearshore habitat types. The method employed has been developed for larger river systems (Hagen et al. 2004) and has been used in other monitoring programs in the province such as Granite Canal (AMEC 2008) and Northeast River (AMEC 2012).

Snorkel surveys are most accurately completed during night (sun down) as fish are startled less by divers and are less likely to move to cover (Hagen et al. 2004). Experienced biologist(s) snorkel slowly along established habitat transects and enumerate the fish species life-cycle stages observed as well as the habitat they are using. Each snorkel location is 350m in total length and is divided into 25m transects.

3.5 Beach Seine

A beach seine is used to sample nearshore habitats and is particularly useful for sampling over slightly cobbled substrates. Beach seining was completed in Goose Bay and Lake Melville by JWEL (2001). Typically, the seine was deployed from a boat, and covered an area of 73.2m^2. Two seines were completed in each identified sampling location. Beach seining has not been completed in subsequent sampling programs.

3.6 Otter Trawl

The otter trawl is a boat-deployed trap that is 5m in diameter and effective over a wide variety of substrates, ranging from clay to small boulders in 3-100m water depth (JWEL 2001). Otter trawls were used in Goose Bay and Lake Melville. Each tow was completed along consistent habitat types, maintaining a particular depth interval. Once the trawl was deployed, the boat maintained a speed of 2 knots and each transect was a total of five minutes duration. Each trawl was capable of sampling an area of 927m^2 during a five-minute tow (JWEL 2001).

4.0 SAMPLING EFFORT

As stated previously, sampling efforts by Nalcor have been ongoing since 1998 and both sample coverage and effort has been extensive. To assist in putting the fish relative abundance numbers in perspective, the overall effort within each habitat is provided in Table 4-1 below.

Table 4-1: Summary of sampling effort by location and dominant gear types, 1998-2016.

Gear Type	Muskrat Falls Reservoir Area	Churchill River below Muskrat Falls Reservoir Area	Goose Bay Estuary	Lake Melville
Fyke Net (net-nights)	453	651	6	14
Gillnet (net-nights)	54	93	29 + 36[1]	21 + 36[1]
Electrofishing (stations)	57	11	na	na
Snorkel Survey (transects)	342	56	na	na
Beach Seine (stations)	na	na	12[1]	24[1]
Otter Trawl (5 min hauls)	na	na	21[1]	21[1]

[1] Sample effort completed in 1998 by JWEL (see JWEL 2001). Provided as separate effort to assist in species overviews within the text.

5.0 SPECIES OVERVIEW

In total, the baseline sampling program for the Lower Churchill Hydroelectric Development, which includes Muskrat Falls, has sampled over 15,600 fish from approximately 29 different species between 1998-2016. By study location;

- 2,285 fish from 15 species have been captured upriver of the Muskrat Falls Reservoir Area (efforts were concentrated in 1998, 1999, 2000, 2006, and 2010 only);
- 3,323 fish from 15 species have been captured in the future Muskrat Falls Reservoir Area;
- 3,212 fish from 13 species have been captured in the lower Churchill River below Muskrat Falls;
- 4,122 fish from 23 species have been captured in Goose Bay Estuary; and
- 2,690 fish from 19 species have been captured in Lake Melville.

Tables 5-1 to 5-3 and Figures 5-1 to 5-3 present relative abundance estimates for each fish species captured within Section Two (what will become the Muskrat Falls reservoir area) and Section One (below Muskrat Falls) of the lower Churchill River as well as Goose Bay, and Lake Melville, respectively (most upstream to downstream). Table 5-4 and Figure 5-4 provide supplemental fyke net results from Goose Bay and Lake Melville completed in 2016. Outer Lake Melville is a sample area near in the eastern portion of the lake near Valley Bight (see Figure 5-5). Estimates of Catch-Per-Unit-Effort (CPUE) for other sections of the river are available in baseline reports (e.g., AGRA 1999, AMEC 2000, 2007). It should be noted that most sampling has occurred during ice-free conditions between June and October and therefore species distribution during the spring ice break up and winter are likely underrepresented. For brevity, the most utilized and effective sampling method for each sample area has been presented below. Summaries of all methods are provided in the 2016 Baseline Report (Amec Foster Wheeler 2016a).

Table 5-1: Summary of mean fyke net CPUE in the mainstem below Muskrat Falls (Section One), 2006 through 2016, Fall sampling

Species	2006-2014		2015		2016	
	Relative abundance CPUE[1]	Biomass CPUE[2]	Relative abundance CPUE[1]	Biomass CPUE[2]	Relative abundance CPUE[1]	Biomass CPUE[2]
Brook Trout	0.08	9.73	0.05	20.37	0.03	5.88
Burbot	0.03	13.84	0.03	40.00	0.03	28.41
Lake Chub	0.72	4.74	1.15	8.10	0.57	6.39
Lake Whitefish	0.02	1.89	0.02	0.03	0.02	0.18
Longnose Dace	0.14	0.32	0.05	0.12	0.05	0.21
Longnose Sucker	0.85	102.76	1.43	60.72	0.48	33.08
Northern Pike	0.09	4.03	0.05	1.76	0.30	28.77
Rainbow smelt	0.00	0.00	0.00	0.00	0.02	0.32
Round Whitefish	0.00	0.00	0.10	10.48	0.00	0.00
Sculpin	0.07	0.18	0.22	0.65	0.13	0.27
Stickleback[3]	3.07	5.87	7.90	16.80	6.18	18.01
White Sucker	0.59	105.14	0.58	68.11	0.48	65.26
Total	5.65	248.48	11.58	227.14	8.30	186.76

1 Relative abundance CPUE expressed as fish/net-night

2 Biomass CPUE expressed as grams/net-night

3 Threespine Stickleback

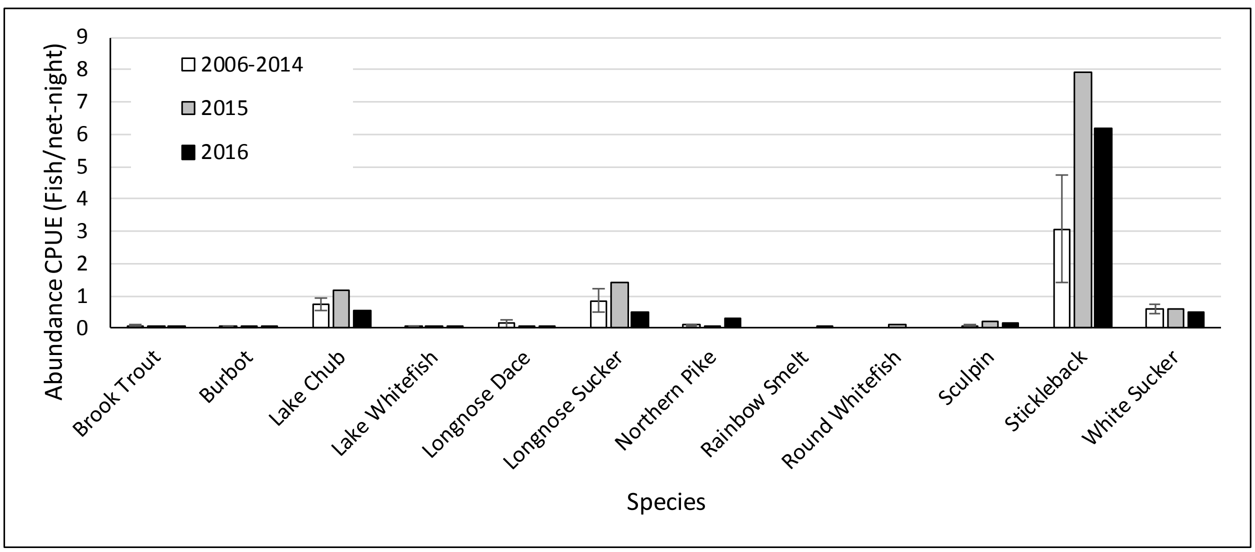


Figure 5-1: Fyke net relative abundance CPUE (fish/net-night) in the mainstem below Muskrat Falls (Section One), 2006 – 2016, Fall sampling (bars present the standard error of the mean CPUE from 2006-2014)

Table 5-2: Summary of mean gillnet CPUE in the Goose Bay, 1999 through 2016

Species	1999-2014		2015		2016	
	Mean Relative abundance CPUE[1]	Mean Biomass CPUE[2]	Mean Relative abundance CPUE[1]	Mean Biomass CPUE[2]	Mean Relative abundance CPUE[1]	Mean Biomass CPUE[2]
Atlantic herring	0.08	23.50	0.00	0.00	0.00	0.00
Brook trout	13.68	3388.07	3.00	487.60	1.33	362.67
Lake chub	5.47	95.29	5.50	86.40	0.00	0.00
Lake whitefish	0.58	197.12	0.00	0.00	0.67	366.57
Longnose sucker	44.38	3812.07	36.00	2325.95	3.67	593.33
Northern pike	0.05	0.90	0.00	0.00	0.00	0.00
Rainbow smelt	7.18	308.04	0.00	0.00	0.00	0.00
Rock cod	0.28	235.96	0.00	0.00	0.00	0.00
Round whitefish	0.13	13.81	0.00	0.00	0.00	0.00
Tomcod	3.28	197.92	1.50	21.10	0.00	0.00
White sucker	9.81	1559.80	15.50	2707.45	3.33	702.57
Winter flounder	0.05	7.28	0.00	0.00	0.00	0.00
Total	84.97	9839.75	61.50	5628.50	9.00	2025.13

1 Relative abundance CPUE expressed as fish/net-night

2 Biomass CPUE expressed as grams/net-night

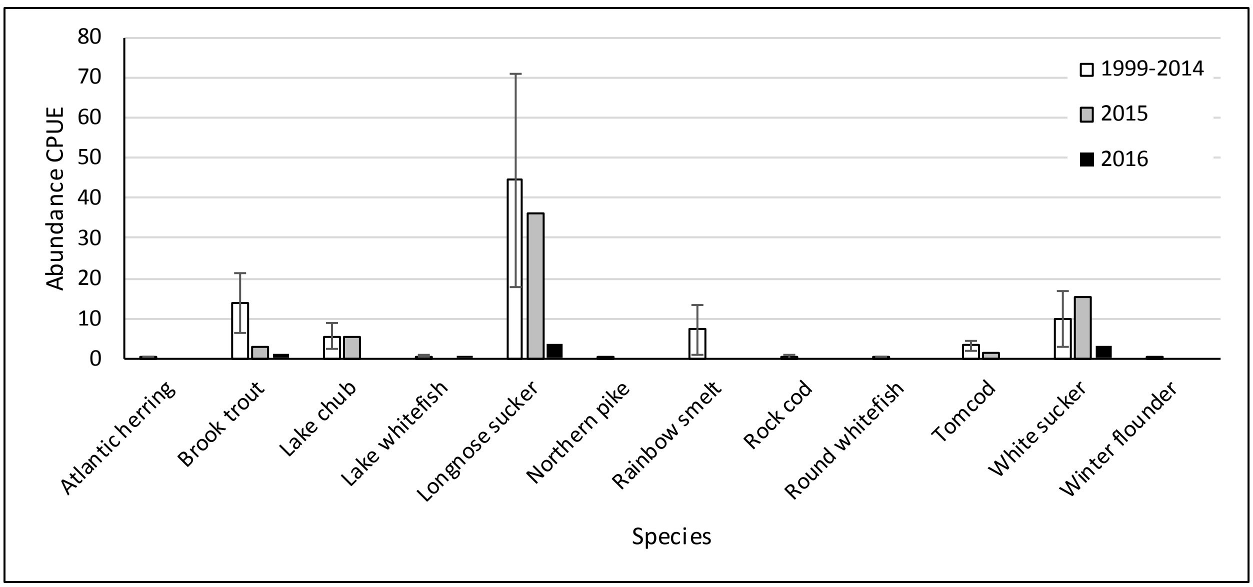


Figure 5-2: Mean gillnet relative abundance CPUE in Goose Bay, 1999 through 2016 (Error bars represent the standard error of the annual mean relative abundance CPUE from 1999-2014).

Table 5-3: Summary of mean gillnet CPUE in the Lake Melville, 2011 through 2016

Species	2011-2014		2015		2016	
	Relative abundance CPUE	Biomass CPUE	Relative abundance CPUE	Biomass CPUE	Relative abundance CPUE	Biomass CPUE
Atlantic salmon	0.00	0.00	0.25	487.50	0.00	0.00
Brook trout	16.90	4573.51	30.25	7,928.30	9.75	2487.78
Lake chub	4.52	76.20	3.00	50.38	0.00	0.00
Lake whitefish	0.58	73.66	0.50	51.18	0.00	0.00
Longhorn sculpin	0.07	0.63	0.00	0.00	0.00	0.00
Longnose sucker	15.83	1264.52	11.50	1,215.70	1.25	190.70
Rainbow smelt	5.35	225.37	4.25	114.70	0.00	0.00
Round whitefish	0.23	21.82	0.00	0.00	0.00	0.00
Tomcod	6.32	424.15	10.25	393.83	0.25	15.33
White sucker	9.43	1201.35	4.50	1,024.70	1.50	514.20
Winter flounder	1.58	87.58	0.00	0.00	0.00	0.00
Total	60.82	0.00	64.50	11,266.28	12.75	3,208.01

1 Relative abundance CPUE expressed as fish/net-night

2 Biomass CPUE expressed as grams/net-night

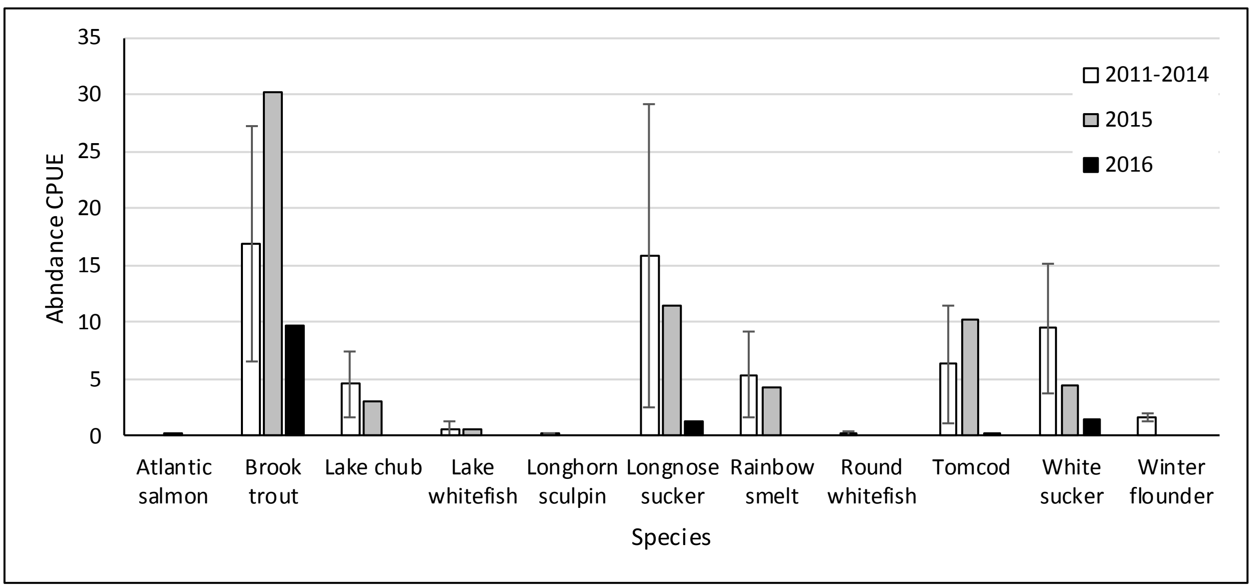


Figure 5-3: Gillnet relative abundance CPUE (fish/net-night) in Lake Melville, 2011 through 2016 (Error bars represent the standard error of the annual mean relative abundance CPUE from 2011-2014).

Table 5-4: Summary of mean fyke net CPUE in estuarine sampling areas, 2016

Species	Goose Bay		Lake Melville		Outer Lake Melville	
	Relative abundance CPUE	Biomass CPUE	Relative abundance CPUE	Biomass CPUE	Relative abundance CPUE	Biomass CPUE
Blenny	0.00	0.00	0.38	2.91	0.00	0.00
Brook trout	0.00	0.00	0.00	0.00	4.50	2344.60
Lake chub	2.33	21.90	5.63	58.93	0.00	0.00
Longnose sucker	2.50	107.08	2.00	245.96	0.50	61.35
Rainbow Smelt	0.17	1.83	2.00	25.80	17.00	194.00
Sculpin	0.00	0.00	0.13	0.86	0.00	0.00
Stickleback	2.67	5.70	6.25	15.61	0.00	0.00
Tomcod	10.50	46.20	29.13	500.34	0.00	0.00
White Sucker	0.33	47.63	2.25	529.64	0.00	0.00
Winter flounder	0.00	0.00	1.75	46.94	2.50	149.05
Total	18.50	230.35	49.50	1,426.99	24.50	2,749.00

1 Relative abundance CPUE expressed as fish/net-night

2 Biomass CPUE expressed as grams/net-night

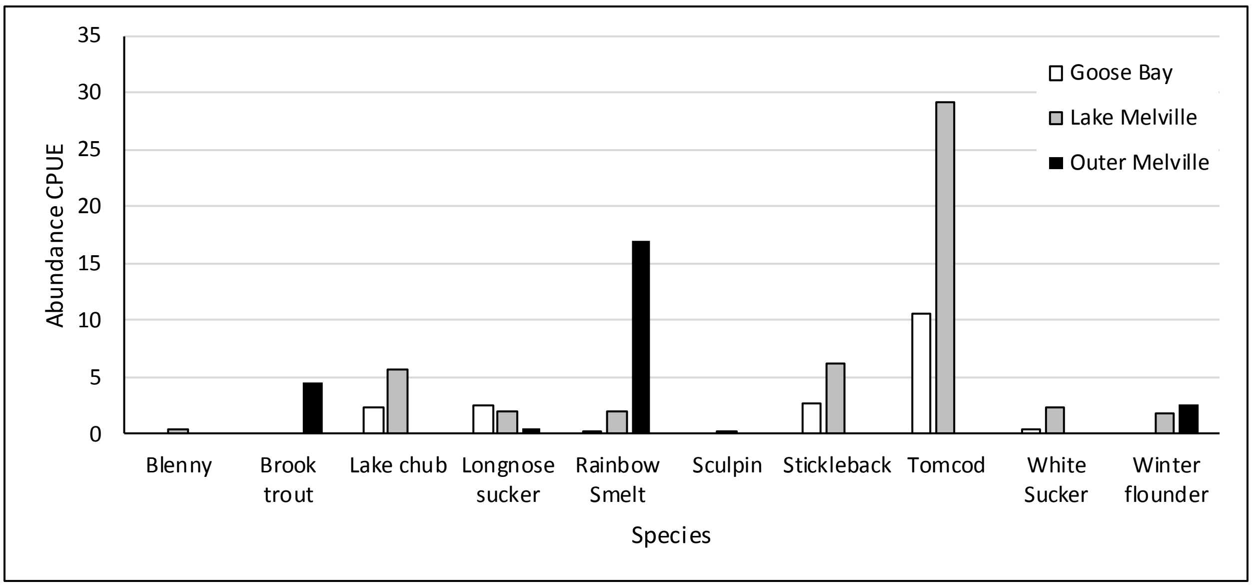


Figure 5-4: Mean fyke net Relative abundance CPUE (fish/net-night) in estuarine sampling areas, 2016

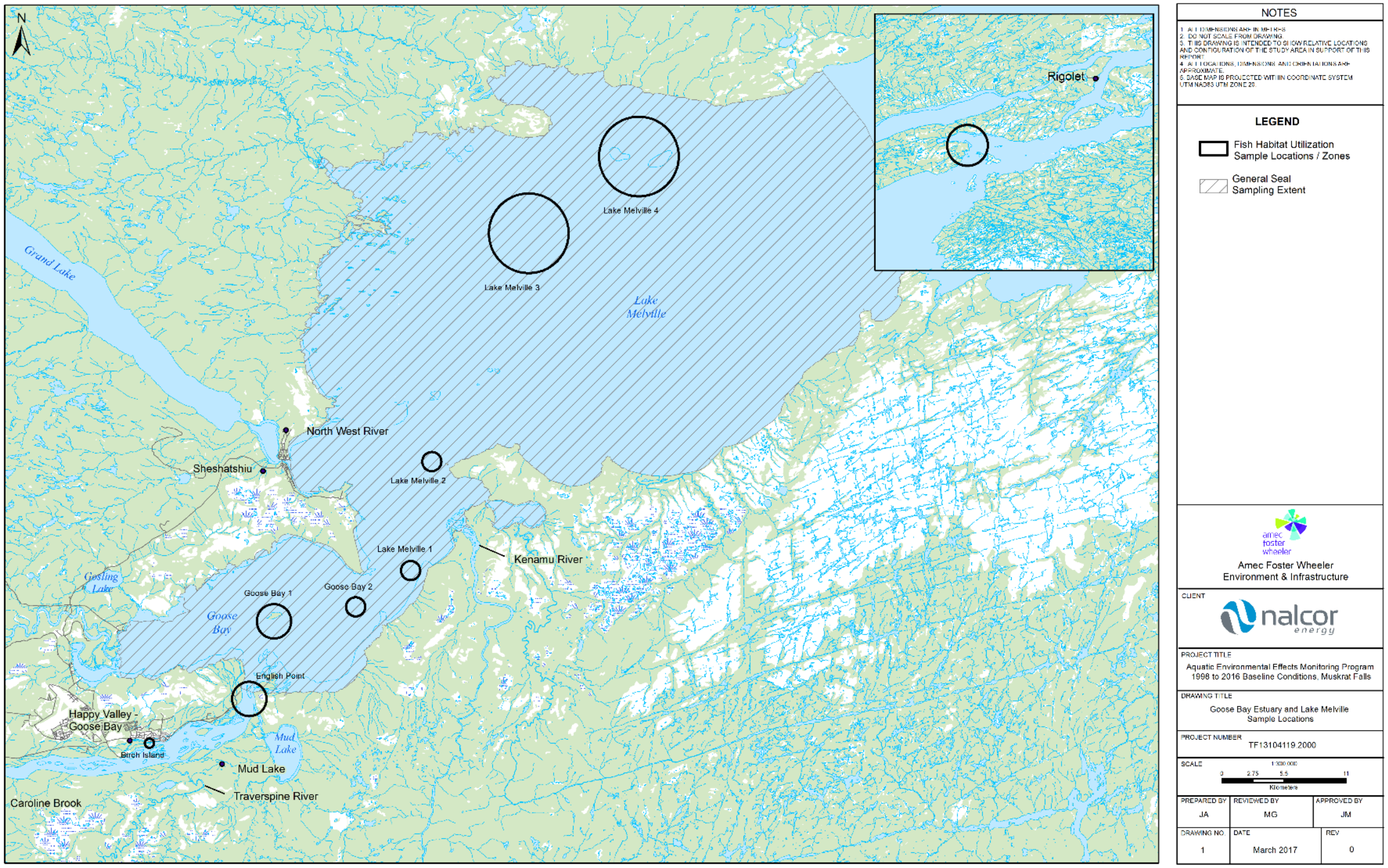


Figure 5-5: Overall EEM study area: Goose Bay estuary and Lake Melville (reproduced from AMEC 2013b).

Calder et al. (2016) recently identified top methylmercury (MeHg) exposure sources/pathways for local resource users downstream of the Muskrat Falls Project that included fish species as well as other animals. These fish species have been described in greater detail below based on Nalcor data collected since 1998. While other species have been captured in the lower Churchill River system (e.g., burbot), the species described below have been based on community fish captures listed in Table S5 (supplemental information in Calder et al. 2016), methylmercury concentrations in aquatic species harvested in the Lake Melville Region (Tables S6a and S6b in Calder et al. 2016), and Biological Accumulation Factor calculations (Tables S7a and S7b in Calder et al. 2016). The same species are input parameters to a revised mercury model being generated by Reed Harris and Associates for use in a re-analysis of the existing Nalcor HHRA.

5.1 Northern Pike (*Esox lucius*)

The northern pike has a circumpolar distribution in the northern hemisphere above 40° North latitude (Toner and Lawler 1969; Scott and Crossman 1998). Its native North American range includes Alaska, most of Canada south of the Arctic Circle, the drainages of the Missouri and Ohio Rivers, and the Great Lakes (Inskip 1982). Pike occur throughout the Churchill River system (Anderson 1985) in relatively low abundance however they occur most in the slower habitat downriver of Gull Island Rapids (i.e. Sections One and Two), with Section One having the greatest relative abundance (Ryan 1980; AGRA 1999; AMEC 2000; AMEC 2009; Amec Foster Wheeler 2016a). Specimens have also been captured at the mouths of tributaries where slower flowing delta-like habitat occurs (eg. Lower Brook, Elizabeth River, Caroline Brook and McKenzie River) (Scruton 1984; AGRA 1999; Amec Foster Wheeler 2016a; 2016b). Many of the pike captured within the mouth of McKenzie River were yearlings and one-year old juveniles (Amec Foster Wheeler 2016a). Beak (1980) also gill netted specimens on Minipi Lake and Dominion Lake and speculated that the species probably occurs on most lakes and ponds in plateau headwater systems of the lower Churchill tributaries. One northern pike has been captured in the Goose Bay estuary (live released) since 1998 and none within Lake Melville (Amec Foster Wheeler 2016a). JWEL did not capture any northern pike in Goose Bay or Lake Melville (JWEL 2001).

Northern pike are not adapted to strong currents and occur most frequently in lakes (Inskip 1982) where they inhabit backwaters and pools (Christenson and Smith 1965; Crossman 1978). In Canada, pike generally inhabit clear, slow, heavily vegetated habitat or weedy bays of lakes (McPhail and Lindsey 1970; Becker 1983; Scott and Crossman 1998) throughout all stages of their life cycle (Ford et al. 1995; Inskip 1982). They have been found over a wide range of water turbidity, although they are much more common in clear and only slightly turbid water (Becker 1983). Based on habitat utilization data and the habitat-types characterized for the lower Churchill River, highest overall utilization for northern pike tends to be within the slower water velocities of the main stem followed by littoral zone habitat of Winokapau Lake. A breakdown of habitat utilization by life-cycle stage shows that highest spawning and young-of-year utilization is within slower water velocity main stem habitat and littoral habitat of Winokapau Lake. Juvenile use is highest throughout the main stem of the lower Churchill River while adults utilize slower velocity tributary habitat and littoral zone habitat of Winokapau Lake. Northern pike were not captured in any deep-water sets within Winokapau Lake (AMEC 2001).

Northern pike have been aged up to eleven years old in the lower Churchill River. Mean length-at-age data shows they range between 104 mm in length at age one to over 930 mm at age eleven. Growth is shown as being relatively linear, although there tends to be a slight reduction in growth after age six. Growth rates determined from baseline sampling are in concurrence with historic rates provided for the lower Churchill River in Anderson (1985).

Northern pike are early spring spawners, with males and females moving into flooded vegetated areas immediately after spring thaw. They generally spawn during daylight hours in shallow, heavily vegetated floodplains of rivers, marshes, and lakes (Clark 1950; Franklin and Smith 1963; McCarraher and Thomas 1972; Scott and Crossman 1998; Bradbury et al. 1999). Adhesive eggs are attached to vegetation where they incubate for only twelve to fourteen days. The newly hatched young (6 to 8 mm in length) remain attached to the vegetation and feed on the yolk sac. After 6 to 10 days, the yolk is absorbed and the free swimming young feed heavily on zooplankton and immature aquatic insects. Within seven to ten days the juveniles begin to feed on small fish and by the time pike reach 50 mm in length, fish have become the primary diet. Baseline aquatic vegetation surveys have identified areas where suitable northern pike spawning habitat occurs. The largest of these include Birchy Creek near Goose Bay, Caroline Brook, the mouth of McKenzie River, the lower sections of Lower Brook and areas near the Metchin River.

The overall sex ratio of specimens within the lower Churchill River favored males (69%). The diet of northern pike sampled consists entirely of fish.

Northern pike were not captured, tagged or recorded below Muskrat Falls (i.e. Section One) during the 1998 migration study (JWEL 2000). Most pike were tagged and tracked in Sections Two and Four (Winokapau Lake), with most activity recorded in Section Two. During the duration of the study, the majority of the tagged northern pike remained sedentary. Primary areas included the confluence with Upper Brook and the lower end of Gull Lake. The main exceptions to this were migrations undertaken during spawning season. Migrations were generally short in nature, the longest recorded was 46.3km, and were concentrated to the mouths or lower reaches of tributaries in Sections Two (i.e. Upper Brook) and Four (i.e. Elizabeth River and small stream west of Long Point). During spawning season, pike were noted in areas consisting of slow habitat; sandy substrate with ample amounts of aquatic vegetation.

During sampling associated with the Smallwood Reservoir, highest levels in 1977-78 were recorded downstream of the reservoir with a peak total mercury level of 1.53 mg/kg for a 600mm standard length northern pike (~4x background) with significant elevated levels downstream to Gull Lake. Concentrations of mercury in northern pike within the lower reaches of the Churchill River were not significantly different from those of other Labrador lakes (Anderson 2011). Mercury concentrations from ongoing baseline data collection associated with the project are provided in **Table** 5-5.

Table 5-5: Summary of total mercury concentrations in northern pike within the baseline study area, 1999-2016

Year	Total Mercury (mg/kg)		
	Sample Size	Mean (SE)	Range
Muskrat Falls reservoir area			
1999	4	0.34 (0.10)	0.15-0.61
2010	0	-	-
2012	16	0.33 (0.03)	0.19-0.68
2013	10	0.15 (0.04)	<0.05-0.41
2014	5	0.21 (0.04)	0.10-0.30
2015	3	0.26 (0.07)	0.14-0.38
2016	23	0.18 (0.03)	<0.02-0.49
Mainstem and Tributaries Below Muskrat Falls			
1999	3	0.13 (0.03)	0.08-0.17
2010	11	0.03 (0.01)	0.01-0.08
2011	5	0.09 (0.02)	0.05-0.15
2012	7	0.08 (0.01)	0.06-0.13
2013	29	0.06 (0.01)	<0.05-0.18
2014	10	0.09 (0.01)	<0.05-0.16
2015	5	0.07 (0.01)	<0.05-0.12
2016	15	0.05 (0.01)	<0.02-0.19
Goose Bay			
2013	1	0.05	-
Lake Melville – none captured			
Eastern Lake Melville – none captured			

Note: Values below detection limits have been incorporated as the detection limit (i.e. 0.02-0.05mg/kg) to produce a conservative estimate of mean concentrations.

5.2 Arctic Charr (*Salvelinus alpinus*)

The Arctic charr has the most northerly distribution of all anadromous and freshwater salmonids. Beak (1980) reported landlocked populations of Arctic charr in both Minipi and Dominion Lakes, where they are believed to be relict from the last glaciation. While they may be present in other larger water bodies on the Churchill plateau, based on all sampling conducted, Arctic charr are not present in the main stem of the Churchill River (Scruton 1984) and have not been collected during any known sampling program in the lower Churchill River, Goose Bay, or Lake Melville (Amec Foster Wheeler 2016a). The Environmental Assessment of the project references the Innu Traditional Knowledge Committee report that indicates that Arctic char have been caught occasionally at North west point (Nalcor 2009).

Although noted in Calder et al. (2016) as being one of the top 20 food sources exposed to MeHg increases downstream of Muskrat Falls, no Arctic charr have been captured during any sampling in Goose Bay or Lake Melville (Amec Foster Wheeler 2016a; JWEL 2001). Samples included in the Calder et al. (2016) analysis were collected 20 miles East of Rigolet (see Table S5 in supplemental information) and would therefore represent a sea-run sample of unconfirmed origin.

Arctic charr were not sampled as part of any post-Smallwood mercury sampling program.

5.3 Atlantic Salmon (*Salmo salar*)

Atlantic salmon are distributed throughout the northern portion of the Atlantic Ocean from Portugal to Norway in the east, throughout southern Iceland and Greenland, and from Hudson Bay to the Connecticut River in the west (Scott and Crossman 1998). In Canada, the anadromous form is distributed throughout eastern Quebec, the Maritimes and Newfoundland and Labrador (Scott and Crossman 1973; Scott and Scott 1988; Black et al. 1986; COSEWIC 2010). Throughout Newfoundland and Labrador, Atlantic salmon occur in both anadromous and landlocked populations (Smith 1988).

Anadromous salmon typically can spend up to one-three years at sea before returning to their home river to spawn in the fall. Upstream migration may occur from July to August in Labrador (see Grant and Lee 2004) with spawning occurring approximately early October – November. During their upstream migration, adult salmon cease feeding (Grant and Lee 2004). Some individuals, usually females, can spawn more than one year. In Labrador, young salmon will typically remain within the freshwater environment for 3-6 years until they reach a length of 12-20 cm (Grant and Lee 2004) before smolting and heading to sea. Adult migration and growth typically occurs in the marine environment. Recent work completed on adult Atlantic salmon in Lake Melville indicates that isotopic signatures of elements within sampled fish (including MeHg) is derived from the marine environment (Li et al. 2016) indicating that adults do not feed extensively within Lake Melville.

During the smolting process, salmon parr move downstream and undergo physiological adaptations for life in a saline environment. Some Atlantic salmon parr in Newfoundland have been shown to use estuaries as rearing habitat as well as during the smolting process (Cunjak et al. 1989; 1990; Cunjak 1992); however, extensive sampling of both the main stem of the lower Churchill River, Goose Bay, and Lake Melville does not indicate any use of these habitats by salmon parr. For example, no juvenile Atlantic salmon have been captured in the main stem, Goose Bay or Lake Melville during any sampling program since 1998 (Amec Foster Wheeler 2015a; 2016a; JWEL 2001) and juveniles have only been captured in low numbers within sampled tributaries (Caroline Brook and McKenzie River) below Muskrat Falls. However, sampling in generally completed in June, August and September and downstream migrations in July might not be adequately documented.

Past reports from both the commercial and recreational fisheries indicate a relatively small salmon migration into the Lake Melville area (Anderson 1985). Two rivers in the region are scheduled Atlantic salmon rivers; Tom Luscombe and Double Mer; however, a large local subsistence fishery for Atlantic salmon and brook trout is conducted on several other larger rivers including Kenamu River. The apparent general under-utilization of rivers in the Lake Melville area by salmon is probably related to lack of good spawning areas, low winter discharges and high turbidity which reduces the quality of parr-rearing habitat and the impact of past fisheries (Anderson 1985). Since 1998, only two adult Atlantic salmon (1998 and 2012) have been captured within the main stem of the lower Churchill River below Muskrat Falls during baseline data collection and one other during the radio telemetry program in 1998 (JWEL 2000). While salmon are using the tributaries directly flowing into the lower Churchill River (both Caroline Brook and McKenzie River have confirmed salmon juveniles), they do not appear to be present in large numbers. Anadromous Atlantic salmon are not found above Muskrat Falls as it is a barrier to upstream migration (Bruce et al. 1975, Ryan 1980, Anderson 1985, AGRA 1999, Nalcor 2009).

5.3.1 Ouananiche

Landlocked Atlantic salmon, commonly called ouananiche, are the dominant species in some Newfoundland lakes where they may exist in either normal or dwarf forms (Smith 1988). Ouananiche are found throughout the main stem of the Churchill River between Muskrat Falls and Churchill Falls (Beak 1980; Ryan 1980; AGRA 1999; Amec Foster Wheeler 2016a), being most abundant in Section Three and Four (Gull Island through Winokapau Lake) (AGRA 1999; AMEC 2000). Sampling since 1998 using gillnets, fyke nets, angling, and snorkeling, has only produced six ouananiche in Section Two of the main stem (i.e., the Muskrat Falls Reservoir area). In Winokapau Lake, most ouananiche have been sampled in the littoral and near-surface habitat of the profundal zone. Although typically a riverine species, ouananiche have only rarely been captured in tributary habitat upstream of Muskrat Falls.

Based on habitat utilization data and the habitat-types characterized for the lower Churchill River, highest overall utilization for ouananiche is intermediate velocity main stem habitat. A breakdown of habitat utilization by life-cycle stage shows that highest spawning and young-of-year utilization is within fast and intermediate velocity main stem habitat types. Juvenile and adult utilization is highest in intermediate velocity main stem habitat. The species has not been captured in any deep-water sampling within Winokapau Lake (AMEC 2001; 2007). While ouananiche have been captured in low abundance within any tributary or stream habitat sampled, the literature does suggest that the habitat types present would be suitable.

Ouananiche may typically live for up to ten years in Newfoundland (Leggett 1965). Specimens have been captured within the upper portions of the Churchill River, above the Muskrat Falls reservoir area, ranging in age from three to eight (AGRA 1999; AMEC 2000). Mean length-at-age data shows they range between 245 mm in length at age three to almost 450 mm at age eight. Growth is shown as being relatively slow between ages three and four with an increase in rate after age four. This may be a reflection of prey selection as many larger, older ouananiche sampled were feeding on a larger proportion of fish and terrestrial mammals. Growth rates determined from baseline sampling are in concurrence with historic rates for the lower Churchill River provided in Anderson (1985).

Ouananiche typically mature at 2-3 years of age (Leggett 1965; Lee 1971; Leggett and Power 1969). Spawning typically occurs in October or November, depending on water temperature, with females ascending tributaries to prepare redds (nests). Lake-spawning has also been observed along shorelines (Leggett 1965) as well as near areas of moving water, usually above outlet streams and near the mouths of inlet streams (Leggett 1965; Harvey and Warner 1970; Einarsson et al. 1990). Typical egg production at spawning is 1,500 eggs per kg of female (Scott and Crossman 1973) but this can be variable.

In the Churchill River watershed ouananiche reach maturity as early as age four (AMEC 2000), however the age-class where 50% of ouananiche mature is six years old. All ouananiche sampled greater than age six were maturing therefore alternate year spawning was not evident (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2015a, 2016a).

Scruton et al. (1995) have shown that ouananiche will overwinter in deep warmer waters of reservoir systems as well as fast-flowing ice-free waters of inlets, outlets and canals.

The diet of ouananiche consists of a wide variety of food types including aquatic invertebrates, fish, and terrestrial vertebrates. Aquatic invertebrates were the most frequent food type consumed within those sampled from the Churchill River. Ouananiche greater than 350 mm in length have a relatively large proportion of their diet consisting of terrestrial mammals (meadow voles, mice and shrews).

The majority of ouananiche movement activity recorded by telemetry was located within Section Five, close to the Churchill Falls Generation facility tailrace (JWEL 2001). It should also be noted, however, that all ouananiche tagged were captured within Section Five. Approximately sixty percent of those tagged underwent long distance migrations (>10km). The longest migration measured was 80km. Most of the long-distance movements occurred in the fall, which coincides with the spawning season of ouananiche. The upper reaches of Section Five (near the Churchill Falls Generating facility) as well as the Unknown River were identified as spawning locations for those fish tagged. The identified areas where ouananiche were recorded spawning are classified as intermediate velocity main stem habitat.

Atlantic salmon (anadromous or ouananiche) were not a target of sampling associated with the formation of the Smallwood Reservoir. Mercury concentrations from ongoing baseline data collection associated with the project are provided in **Table** 5-6.

5.4 Brook trout (*Salvelinus fontinalis*)

The brook trout is widely distributed throughout Newfoundland and Labrador (Scott and Crossman 1973), at least as far north as the Hebron Fiord (Black et al. 1986), where they have been reported to make extensive use of clear, cool (<20°C) lake habitats (Ryan and Knoechel 1994). Brook trout are known to have both landlocked and anadromous populations throughout Newfoundland and Labrador (Scott and Crossman 1964, 1998). Anadromous populations may spend one or two months feeding at sea in relatively shallow water, close to their natal stream, while others spend their entire life in freshwater (Scott and Crossman 1964; Morrow 1980; Power 1980; Ryan 1980; Scott and Scott 1988).

Brook trout are found throughout the main stem and tributaries of the lower Churchill River between Muskrat Falls and Churchill Falls (Beak 1980; Ryan 1980; AGRA 1999; AMEC 2000, AMEC 2001), being most abundant in Section Three and Five (Gull Island to Winokapau Lake and upriver of Winokapau Lake) (AGRA 1999; AMEC 2000). Brook trout have also been captured below Muskrat Falls within the main stem but at relatively low rates (AMEC 2000; AMEC 2007; AMEC 2009; Amec Foster Wheeler 2015a; Amec Foster Wheeler 2016a).

Table 5-6: Summary of total mercury concentrations for Atlantic salmon within the baseline study area, 1999-2016

Year	Total Mercury (mg/kg)		
	Sample Size	Mean (SE)	Range
Muskrat Falls reservoir area – ouananiche			
1999	1	0.12	-
2010	0	-	-
2012	0	-	-
2013	0	-	-
2014	1	0.06	-
2015	2	0.19 (0.10)	0.09-0.29
2016	0	-	-
Mainstem and Tributaries Below Muskrat Falls – no sample sizes sufficient for analysis			
2012	1	0.11	-
Goose Bay – none captured			
Lake Melville – Atlantic salmon			
2011	0	-	-
2013	0	-	-
2014	0	-	-
2015	24	0.09 (0.01)	<0.05-0.16
2016	15	0.04 (<0.01)	0.03-0.08
Eastern Lake Melville – none captured			

Note: Values below detection limits have been incorporated as the detection limit (i.e. 0.02-0.05mg/kg) to produce a conservative estimate of mean concentrations.

Based on habitat utilization data and the habitat-types characterized for the lower Churchill River, highest overall utilization for brook trout is stream (i.e. tributary) habitat followed by areas of intermediate water velocity within the main stem of the lower Churchill River. Use of lake Melville by juvenile and adult brook trout is amongst the highest utilization (Amec Foster Wheeler 2016a). A breakdown of habitat utilization by life-cycle stage shows that highest spawning utilization is within stream and slower velocity tributary habitat types. Young-of-year utilization is also greatest in stream/tributary habitat. Juvenile and adult utilization is highest in stream, tributary, and the slower/intermediate water velocity within the main stem of the lower Churchill River. Brook trout were not captured in any deep-water sampling within Winokapau Lake (AMEC 2001).

Few samples have been collected within the main stem of the lower Churchill River below Muskrat Falls (33 in a combination of fyke nets and gillnets between 1998-2016); however, they are found in relatively higher numbers within the upper habitat of Caroline Brook. Larger numbers have also been sampled within both Goose Bay (191 total) and Lake Melville (535). In both estuarine environments, brook trout have had some of the highest CPUE and biomass of all species sampled (Amec Foster Wheeler 2015a; 2016a). This is most likely the result of the brackish environment of the estuary being a suitable habitat for anadromous brook trout to feed during the summer months. Typically, brook trout will not feed within an estuarine environment beyond several kilometers of its natal stream (Scott and Scott 1988); therefore, most of the brook trout captured are likely from the larger nearby tributaries such as Mud Lake and Kenamu River.

Specimens have been captured from every age-class between one and six (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2015a; 2016a, 2016b). Mean length-at-age data shows they range between 82 mm in length at age one to almost 415 mm at age six. Growth is relatively linear throughout all years. Growth rates determined from baseline sampling are in concurrence with historic rates for the lower Churchill River provided in Anderson (1985).

In the Churchill River watershed, brook trout reach maturity as early as age two (AMEC 2000), however the age-class where 50% of brook trout mature is four years old. All brook trout sampled greater than age four were maturing therefore alternate year spawning is not evident (AGRA 1999; AMEC 2000).

In general, movements of tagged brook trout were relatively short distances, with approximately ten percent exceeding 10km in distance (JWEL 2000). The longest migration recorded was 93.5km. Most migrations were undertaken in late summer to early fall, which coincides with the brook trout spawning season. All movements during this time were to areas of fast and intermediate habitat types, with the majority being focused in Section Three (above the Muskrat Falls Reservoir area).

The diet of brook trout consists of a wide variety of food types including aquatic invertebrates, fish, and terrestrial invertebrates and vertebrates. Aquatic invertebrates were the most frequent food type consumed; however, fish was a large component of brook trout in the 151-250 mm size range (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2015a; 2015b).

Previous sampling in 1977-78 after the Smallwood Reservoir was created showed total mercury concentrations in brook trout from Goose Bay and Lake Melville (standard fish length of 300mm) peaked at 0.15 mg/kg which was similar to other freshwater brook trout samples but approximately four-times greater than sea-run samples from other coastal Labrador locations. By 2005, levels had declined significantly (Anderson 2011). Mercury concentrations from ongoing baseline data collection associated with the project are provided in **Table** 5-7.

5.5 Lake Trout (*Salvelinus namaycush*)

Lake trout are widely distributed in northern North America and are found throughout southern Labrador, except for the southeastern corner (Scott and Crossman 1973; Black et al. 1986). In the south, lake trout prefer cool (<10°C), deep lakes, but in the north where temperatures are lower, they may inhabit shallow lakes and large rivers (McPhail and Lindsey 1970; Ryan 1980). Lake trout occur throughout the Churchill River watershed, but are more prevalent in the upper reaches (Anderson 1985; AGRA 1999; AMEC 2000, 2001). Beak (1980) reported the species as present in the main stem only above Gull Island Rapids (i.e., the upper extent of the Muskrat Falls reservoir area). Sampling since 1998 confirms this as only one lake trout has been captured in Gull Lake within Section Two (the Muskrat Falls Reservoir area) and only one below Muskrat Falls. The only lake trout captured below Muskrat Falls was in 2006 (AMEC 2007) but its condition was poor and seemed as though it had come over the falls in a weakened condition.

Table 5-7: Summary of total mercury concentrations in brook trout within the baseline study area, 1999-2016

Year	Total Mercury (mg/kg)		
	Sample Size	Mean (SE)	Range
Muskrat Falls reservoir area			
1999	26	0.07 (0.01)	0.03-0.16
2010	0	-	-
2012	1	0.11	
2013	7	0.06 (0.01)	<0.05-0.15
2014	2	0.05 (<0.01)	-[1]
2015	2	0.12 (0.04)	0.08-0.16
2016	0	-	-
Mainstem and Tributaries Below Muskrat Falls			
1999	0	-	-
2010	0	-	-
2011	12	0.08 (0.01)	0.04-0.17
2012	18	0.08 (<0.01)	<0.05-0.12
2013	30	0.05 (<0.01)	<0.05-0.09
2014	8	0.06 (<0.01)	<0.05-0.08
2015	13	0.12 (0.03)	<0.05-0.37
2016	35	0.03 (<0.01)	<0.02-0.0.06
Goose Bay			
1999	9	0.06 (0.01)	0.04-0.14
2011	48	0.08 (<0.01)	0.03-0.17
2013	26	0.07 (0.02)	<0.05-0.44
2014	30	0.05 (<0.01)	<0.05-0.10
2015	6	0.05 (<0.01)	<0.05-0.07
2016	6	0.08 (0.01)	0.04-0.13
Lake Melville			
2011	0	-	-
2013	30	0.06 (<0.01)	<0.05-0.10
2014	30	0.05 (<0.01)	<0.05-0.08
2015	31	0.07 (0.01)	<0.05-0.32
2016	30	0.06 (<0.01)	<0.02-0.11
Eastern Lake Melville			
2016	32	0.04 (<0.01)	<0.02-0.11

Note: Values below detection limits have been incorporated as the detection limit (i.e. 0.02-0.05mg/kg) to produce a conservative estimate of mean concentrations.

[1] All fish were below detection limits

Lake trout are primarily located within Winokapau Lake (Section Four) and Section Five of the lower Churchill River (Beak 1980; Ryan 1980; AGRA 1999; AMEC 2000, AMEC 2001; Amec Foster Wheeler 2016a); being most abundant in Winokapau Lake (AGRA 1999; AMEC 2000). Based on habitat utilization data and the habitat-types characterized for the lower Churchill River, highest overall utilization by lake trout is lacustrine habitat of Winokapau Lake (both littoral and profundal) and faster water velocity habitat within the main stem of the lower Churchill River. A breakdown of habitat utilization by life-cycle stage shows that highest spawning utilization is within tributary habitat and Young-of-year utilization is greatest within littoral zone habitat of Winokapau Lake. Juvenile utilization is also highest in littoral zone habitat of Winokapau Lake with adults utilizing profundal habitat within Winokapau Lake and faster water velocity habitat within the main stem of the lower Churchill River.

Although lake trout were noted in Calder et al. (2016) as being one of the top 20 food sources exposed to MeHg increases downstream of Muskrat Falls, no lake trout have been captured during any sampling in Goose Bay or Lake Melville since 1999 (Amec Foster Wheeler 2016a; JWEL 2001). The 13 lake trout samples included in the Calder et al. (2016) analysis were noted as being collected from the Churchill River; however, the location was not provided and was unlikely to be located downstream of Muskrat Falls or within the area of the Muskrat Falls Reservoir (see Table S5 in supplemental information).

Specimens captured within the lower Churchill River, upstream of the Muskrat Falls Reservoir area, ranged in age from five to nine (AGRA 1999; AMEC 2000). Mean length-at-age data shows they range between 272 mm in length at age five to almost 565 mm at age nine. Growth has been shown as being relatively linear throughout years five to eight with an increase in growth apparent at age nine. Growth rates determined from baseline sampling have been in concurrence with historic rates for the lower Churchill River provided in Anderson (1985).

Lake trout usually spawn in shallow inshore areas of lakes, rarely in streams (Machniak 1975; Martin and Olver 1980; Ford et al. 1995). In most areas of Canada, spawning occurs in late summer-early fall (Scott and Crossman 1973; Ford et al. 1995), mainly in September or October in Labrador (Grant and Lee 2004).

Sexual maturity is thought to occur at a relatively old age. When Parsons (1975) sampled the Ossokmanuan Reservoir (part of the Smallwood Reservoir system) they found no sexually mature lake trout under nine years of age, and Ryan (1980) concluded that in the lower Churchill River, they reach maturity at seven years of age. This estimation was confirmed through sampling for the Project, which recorded lake trout maturing at seven years of age. The results also indicate that individuals may not spawn each year as many older fish between seven and nine years of age were not showing signs of maturing for that year (AGRA 1999; AMEC 2000).

The diet of lake trout consists of aquatic invertebrates, fish and terrestrial mammals. Fish was the most frequent food type identified (AGRA 1999; AMEC 2000).

Lake trout were not included in the original scope of work for the telemetry/movement study; however, five were captured and tagged (JWEL 2000). Lake trout activity was generally concentrated within Winokapau Lake and its outflow. Tracking indicates that lake trout used the entirety of Winokapau Lake, with limited movement upstream or downstream. There was a small concentration of activity near the east end of the lake during the late fall as well as downriver from the confluence of Cache River.

Lake trout were sampled in 1978 in Smallwood Reservoir and Winokapau Lake after the formation of the Smallwood Reservoir. Peak total mercury concentration for a standard 600mm fish length reached 1.72 mg/kg (~3x background). Samples from 1999 showed no significant difference from background (Anderson 2011). No lake trout were captured for total mercury analysis during baseline data collection to date.

5.6 Lake Whitefish (*Coregonus clupeaformis*)

Lake whitefish are widely distributed throughout North America from the Atlantic coastal watersheds westward across Canada and the northern United States, to British Columbia, the Yukon Territory, and Alaska (Scott and Crossman 1998). They are distributed throughout southern Labrador (Bruce 1974; Parsons 1975; Beak Consultants Ltd. 1979; Black et al. 1986; Scott and Crossman 1998; LGL Limited 1999). There are two forms of lake whitefish within the lower Churchill River; normal and a dwarf form. The discrimination between both forms is primarily size-at-maturity and length-at-age as per the identification key of Doyon (1998). Besides size-at-maturity and length-at-age, the primary difference between the two forms is the dwarf form tends to be more zooplankivorous and pelagic in nature while the normal form are more benthic feeders (Bruce 1984). Although they are generally found in lakes, they are relatively abundant in the main stem of the Churchill River, as well as the adjoining lakes and ponds within its watershed (Anderson 1985).

They are distributed throughout, from the upper reaches near the existing Churchill Falls Generating facility downstream to the estuary; however, they are most abundant in the upriver segments (Sections Four and Five). Below Muskrat Falls, lake whitefish has been the most abundant salmonid captured; a total of 121 fish between 1998-2016. Lake whitefish have been considerably lower in abundance from Goose Bay (19 total captured by Amec Foster Wheeler) and Lake Melville (10 total captured) in the same time period. While JWEL (2001) does not provide total numbers, they indicate that lake whitefish were captured in Goose Bay during the 2000 summer sampling season; none were captured in October indicating that they may have already ascended rivers to spawn (JWEL 2001). None were captured by JWEL in Lake Melville (JWEL 2001).

While primarily a lacustrine species, based on habitat utilization data and the habitat-types characterized for the lower Churchill River, highest overall utilization for both forms of lake whitefish tends to be faster water velocity habitat within the main stem of the Churchill River, followed by lacustrine habitat types (profundal and littoral). A breakdown of habitat utilization by life-cycle stage shows that highest spawning utilization for both forms is within tributaries. Young-of-year habitat use for both forms appears to be highest within faster velocity main stem habitat as well as profundal habitat of Winokapau Lake. Juvenile utilization is also highest in faster velocity main stem habitat and littoral habitat within Winokapau Lake; with the dwarf form using faster main stem habitat and the normal form using lacustrine. This is most likely associated with their differing feeding preferences. Adults tend to utilize the faster water velocity habitat within the main stem (normal and dwarf) as well as the lacustrine habitat within Winokapau Lake. Within Winokapau Lake, the adult normal form utilizes both littoral and profundal habitats while the dwarf form more heavily utilizes the open-water profundal habitat type. Neither adult form was found to utilize any tributary or stream habitat outside spawning activities.

Specimens have been captured within the Churchill River from every age-class between one and eighteen with additional adults aged as old as twenty-eight (AMEC 2000; Amec Foster Wheeler 2016a). Mean length-at-age data shows they range between 120 mm in length at age one to almost 420 mm at age eighteen. Growth is shown as being relatively linear; however older fish show slightly slower growth. Growth rates determined from baseline sampling do not appear to concur with historic rates for the lower Churchill River provided in Anderson (1985) but more closely resemble those generated for the upper Churchill River watershed/reservoirs (Ryan 1980).

In Labrador, spawning migrations are reported from early September to mid-October (Scruton et al. 1997). In the Churchill River watershed lake whitefish reach maturity over a range of 3-9 years old (Anderson 1985). Sampling conducted for the Project indicates that the age-class where 50% of the lake whitefish were maturing was three years old; however mature individuals were identified at age two. In the extreme northern limits of their range, individuals have been known to only spawn once every two or three years (Scott and Crossman 1998); this may occur within the lower Churchill River as a portion of adults assessed for maturity greater than seven years old were not maturing (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2016a).

Scott and Crossman (1973) note that more northerly populations tend to produce fewer eggs. Egg counts can vary greatly depending on a fish's size, with specimens from the Ossokmanuan Reservoir yielding anywhere from 967 to 20,963 eggs per fish (Bruce and Parsons 1976). The overall sex ratio of specimens sampled was in favor of males (58%).

The diet of lake whitefish consists of a majority of aquatic invertebrates and algae/detritus (AMEC 2001; Amec Foster Wheeler 2015a; 2016a).

Lake whitefish were not captured within Section One (below Muskrat Falls), or Section Three during the telemetry study (JWEL 2000). Movement patterns varied by river section of capture; fish from Sections Two and Four generally stayed within close vicinity to their tagging locations; i.e. Gull Lake and Winokapau Lake. However, fish tagged in Section Five were noted to make migrations downstream of varying distances. Forty-eight percent of the whitefish tagged within the tailrace of the Churchill Falls Generating facility moved downstream in the late fall (mid-September to mid-October), shortly after the typical spawning season. The median migration distance was 5.7km, with a maximum of 240km (one individual traveled from the tailrace region to Gull Lake). In addition, tagged fish from Winokapau Lake remained there for the duration of the study and while there was no identifiable cluster of spawning activity, it can be assumed that spawning did occur within the lake. The identified potential spawning habitats are predominantly found within fast and intermediate velocity habitat types.

Standard length (300mm) lake whitefish were sampled for total mercury in 1977-78 after formation of the Smallwood Reservoir. Total mercury concentrations in the lower Churchill River downstream to

Winokapau Lake peaked at 0.76 mg/kg (~5x background) but by 1987, values were no longer elevated compared to background (Anderson 2011). Mercury concentrations from ongoing baseline data collection associated with the project are provided in **Table** 5-8.

Table 5-8: Summary of total mercury concentrations in lake whitefish within the baseline study area, 1999-2016

Year	Total Mercury (mg/kg)		
	Sample Size	**Mean (SE)**	**Range**
Muskrat Falls reservoir area			
1999	26	0.13 (0.02)	0.05-0.36
2010	11	0.09 (0.01)	0.04-0.16
2012	11	0.14 (0.03)	0.04-0.43
2013	4	0.07 (0.02)	<0.05-0.14
2014	6	0.06 (0.01)	<0.05-0.10
2015	1	0.19	-
2016	12	0.06 (0.01)	<0.02-0.18
Mainstem and Tributaries Below Muskrat Falls			
1999	16	0.10 (0.02)	<0.02-0.23
2010	7	0.03 (0.01)	<0.01-0.05
2011	17	0.08 (0.01	0.04-0.23
2012	5	0.10 (0.02)	0.04-0.15
2013	0	-	-
2014	3	0.05 (<0.01)	-[1]
2015	2	0.09 (0.03)	0.06-0.12
2016	2	0.08 (0.03)	<0.05-0.10
Goose Bay			
1999	7	0.13 (0.02)	0.03-0.21
2011	1	0.10	-
2013	4	0.007 (0.01)	<0.05-0.09
2014	1	0.05	-
2015	0	-	-
2016	2	0.11 (0.02)	0.09-0.13
Lake Melville			
2011	0	-	-
2013	0	-	-
2014	7	0.05 (<0.01)	<0.05-0.06
2015	2	0.06 (0.01)	<0.05-0.07
2016	0	-	-
Eastern Lake Melville – none captured			

Note: Values below detection limits have been incorporated as the detection limit (i.e. 0.05mg/kg) to produce a conservative estimate of mean concentrations.

5.7 Round Whitefish (*Prosopium cylindraceum*)

Round whitefish are widely distributed in lakes and ponds as well as brackish waters throughout North America and into northern Asia (McPhail and Lindsey 1970; Becker 1983; Scott and Crossman 1998). In Canada, they range from northern New Brunswick, Labrador, and Ungava west through parts of Quebec, Ontario, and the Great Lakes and north westward from northern Manitoba through the Northwest Territories and northern British Columbia (Scott and Crossman 1998). Round whitefish have been reported in the Churchill River system (Beak Consultants Ltd 1979; Ryan 1980; AGRA 1999) but appear to be limited in distribution based on sampling; however, they have been captured in the system both above and below Muskrat Falls (Sections One, Two, Three, Four and Five), being most abundant in Winokapau Lake (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2016a). In 2000, they were only captured in sampling conducted in the pelagic (open-water) habitat of Winokapau Lake and they have been captured very infrequently in tributary habitat; primarily juveniles. Below Muskrat Falls, a total of 44 have been captured between 1998-2016 within the main stem of the river. Juveniles have been captured in McKenzie River during electrofishing and fyke netting and both juveniles and adults have been observed during snorkeling surveys (Amec Foster Wheeler 2016a). Adults have been observed making upriver spawning migrations during fall snorkel surveys in 2015 (Amec Foster Wheeler 2016a) and juveniles were identified in 2012, 2014, 2015, and 2016. Very few round whitefish have been captured since 1998 in Goose Bay (two by Amec Foster Wheeler and three by JWEL) or Lake Melville (three by Amec Foster Wheeler).

While primarily a lacustrine species, based on habitat utilization data and the habitat-types characterized for the lower Churchill River, highest overall utilization for round whitefish tends to be within all riverine main stem habitats. A breakdown of habitat utilization by life-cycle stage shows that highest spawning utilization is within streams and littoral habitat of Winokapau Lake. Young-of-year habitat use appears to be highest within slower water velocity main stem habitat. Juvenile and adult utilization is highest in slower and intermediate water velocity main stem habitat. Neither juvenile nor adult life-cycle stages were captured in any deep-water samples within Winokapau Lake (AMEC 2001).

Round whitefish can live for up to 14 years and can reach sizes of 2 kg; however, the average size is much smaller. Ryan (1980) indicates that the growth rates for round whitefish in the Churchill River are at an intermediate level when compared to results from other regions of North America. Specimens have been captured within the Churchill River from every age-class between one and ten (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2016a). Mean length-at-age data shows they range between 84 mm in length at age one to almost 340 mm at age ten. Growth is shown as being relatively linear up to age four or five with a reduction in growth in older fish. Growth rates determined from baseline sampling are in concurrence with historic rates for the lower Churchill River provided in Anderson (1985).

According to Scott and Crossman (1973), round whitefish are fall spawners (October to December) which utilize gravelly shallows of lakes, river mouths and sometimes rivers as spawning substrate. Spawning can take place in the inshore areas of lakes, at river mouths, or occasionally in rivers (McPhail and Lindsey 1970; Scott and Crossman 1998; Bradbury et al. 1999). In the Churchill River watershed round whitefish reach maturity as early as age one (AMEC 2000), however the age-class where 50% of round whitefish were maturing is four years old. As with lake whitefish, all those sampled greater than age four were maturing therefore alternate year spawning was not evident (AGRA 1999; AMEC 2000).

Typical mean egg production at spawning is 12,000 eggs per kg of female (Scott and Crossman 1998). The eggs remain in the spawning substrate until hatching occurs the following April or May. The overall sex ratio of specimens within the lower Churchill River was fairly even between males and females (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2016a).

The diet of round whitefish consists of a majority of aquatic invertebrates and algae/detritus with evidence of limited feeding on other fish.

Round whitefish were not sampled as part of any post-Smallwood mercury sampling program nor are they included in the baseline as a regular target for mercury body burden.

5.8 Rainbow Smelt (*Osmerus mordax*)

Rainbow smelt are typically a schooling, pelagic fish, inhabiting mid-water areas of inshore coastal waters (Leim and Scott 1966; Scott and Scott 1988; Scott and Crossman 1998). In Hamilton Inlet and Lake Melville, they are primarily an inshore anadromous species that occur within bays and estuaries, but are rare in the Churchill River freshwater system (Anderson 1985). They are an important species in that they feed on pelagic plankton and are an important food source for most estuarine piscivores such as gadids (e.g., cod species), flatfish (e.g., winter flounder) and salmonids (e.g. brook trout).

Smelt are typically anadromous, moving from estuaries such as Lake Melville and Goose Bay into nearby rivers and streams to spawn in the spring, likely before ice breakup (JWEL 2001). As the hatched larvae grow, they move into areas of higher salinity, such as deeper parts of the estuary or more coastal areas (JWEL 2001). Smelt begin to school at about 19 mm in length, moving into shallow water and returning to deeper channels during the day (Belyanina 1969). They will generally spend the summer feeding on copepods and planktonic larvae and in the fall, juveniles mix with adult schools and move into the upper parts of the estuary (Buckley 1989) where they remain for the winter.

Within Lake Melville, smelt seem to prefer deeper, cooler waters in the summer (JWEL 2001). The JWEL sampling program identified that smelt, which spend the summer in the cooler waters of Lake Melville, move into Goose Bay from August to October (JWEL 2001; AMEC/BAE 2001); the relative abundances of smelt in Goose Bay estuary nearly quadrupled from July to August and nearly doubled from August to October (JWEL 2001). There was a slight peak observed in abundance in October in the western portion of Lake Melville and was suggested to be the result of a migration toward the many rivers in the area (JWEL 2001).

Due to physical barriers, this species does not occur above Muskrat Falls in the Churchill River (Ryan 1980) and based on sampling, is very rare upstream of estuarine influences after spawning. Ryan (1980) recorded two specimens (which appeared to be anadromous) downstream of Muskrat Falls and Amec Foster Wheeler captured a lone adult by fyke net just downstream of Muskrat Island in 2016 (Amec Foster Wheeler 2016a). No other known reports occur in the literature for their presence within the freshwater portion of the lower Churchill River (Ryan 1980, Beak 1980, AGRA 1999, AMEC 2000) upstream of the Mud

Lake confluence (AMEC 2000). In addition to sampling conducted related to the Project, the main stem between Happy Valley–Goose Bay and Muskrat Falls as well as several tributaries (eg. Birchy Creek and Caroline Brook), were sampled between 2006 and 2008 for the provincial Department of Transportation and Works. Sampling was conducted using fyke nets and tended gillnets through most open water months (i.e. July and October 2006, May and June 2007, April, May, and June 2008, and May 2009) but did not capture rainbow smelt (unpub. data).

Rainbow smelt have been routinely captured during ongoing baseline sampling since 1999 in both Goose Bay and Lake Melville. Sampling by Amec Foster Wheeler has captured approximately 136 and 155 from Goose Bay and Lake Melville, respectively. Baseline work completed by JWEL in 1998 captured a total of 991 rainbow smelt within Goose Bay / Lake Melville which comprised 31 percent of their total catch (JWEL 2001). Rainbow smelt sampled (AGRA 1998) were predominantly between 151-250mm in length with fairly linear growth through all age classes sampled (ages 1-8). The overall sex ratio favored males (63%). Of the 51 rainbow smelt examined for maturity, 36 were maturing for the 2000 spawning season (early spring spawners). The length-class when at least fifty percent were maturing was 151-200mm. The smallest fish which was maturing was 163mm. The age when at least fifty percent were maturing was three.

Previous sampling after the Smallwood Reservoir was created showed rainbow smelt peaked in total mercury (standard fish length of 200mm) at 0.32 mg/kg in 1978, declined in 1999 and by 2008, concentrations were significantly lower at approximately 0.1 mg/kg (Anderson 2011). Mercury concentrations from ongoing baseline data associated with the project are provided in **Table** 5-9.

5.9 Threespine Stickleback (*Gasterosteus aculeatus*)

Threespine stickleback have an almost circumpolar distribution and are widely distributed in the northern hemisphere (Scott and Scott 1988; Scott and Crossman 1998). In Newfoundland and Labrador, it is a euryhaline species and exists as both a freshwater resident and anadromous marine-dwelling form (Scott and Scott 1988; Scott and Crossman 1998). Spawning generally occurs in the summer months, but timing can range from April to September depending in local conditions (Scott and Crossman 1998). Freshwater resident populations spawn in both lakes and rivers, with anadromous populations spawning in brackish or fresh waters (Leim and Scott 1966; Coad and Power 1973; Morrow 1980; Wootton 1984). River-spawning populations undergo a spring migration from lakes or larger rivers into smaller, slower tributaries and backwaters (Scott and Scott 1988; Scott and Crossman 1998). The males build nests over sandy/muddy substrates in areas of low flow and are usually found in the vicinity of submergent vegetation (Hagen 1967; Virgl and McPhail 1994). Lake spawning populations utilize two distinct habitat types; either open-water (Griswold and Smith 1972; Larson 1976; Lewis 1978; Wootton 1984) or in association with aquatic vegetation (McPhail and Lindsey 1970; Larson 1976; Morrow 1980; Sandlund et al. 1987).

Table 5-9: Summary of mean total mercury concentrations, rainbow smelt, baseline study area, 1999-2016

Year	Total Mercury (mg/kg)		
	Sample Size	Mean (SE)	Range
Muskrat Falls reservoir area – none captured			
Mainstem and Tributaries Below Muskrat Falls			
2016	1	0.02	-
Goose Bay			
1999	29	0.18 (0.01)	0.08-0.31
2011	61	0.09 (0.01)	0.03-0.22
2013	21	0.08 (0.01)	<0.05-0.15
2014	2	0.10 (0.04)	0.06-0.13
2015	0	-	-
2016	1	0.02	-
Lake Melville			
2011	0	-	-
2013	21	0.07 (0.01)	<0.05-0.14
2014	26	0.07 (<0.01)	<0.05-0.11
2015	12	0.05 (<0.01)	<0.05-0.06
2016	6	0.02 (<0.01)	<0.02-0.02
Eastern Lake Melville			
2016	16	0.03 (<0.01)	<0.02-0.05

Note: Values below detection limits have been incorporated as the detection limit (i.e. 0.02-0.05mg/kg) to produce a conservative estimate of mean concentrations.

Males construct a nest of small twigs, algae or plant debris typically over a sandy or mud bottom (McPhail and Lindsey 1970; Griswold and Smith 1972; Scott and Crossman 1973; Ryan 1980; Scott and Scott 1988). Females deposit adhesive eggs in clusters in the nest (Morrow 1980). The male subsequently guards and fans the nest (Leim and Scott 1966; McPhail and Lindsey 1970; Scott and Crossman 1973; Scott and Scott 1988), protecting the young for up to 2 weeks after hatching or until they are able to fend for themselves (Wootton 1976; Scott and Scott 1988). Newfoundland populations normally mature in their second or third year (Ryan 1984) and generally do not live past three years (Ryan 1984; Fitzpatrick 1988).

Its presence has been noted through the Churchill River system (Anderson 1985, Scott and Crossman 1973); being found at the mouth of the Elizabeth River (Beak 1980) and Upper Brook (AGRA 1999), and also found in stomach contents of ouananiche, lake trout, burbot, brook trout and northern pike caught in the main stem (Ryan 1980). Since 1998, threespine stickleback have been the most abundant species captured, accounting for almost half of the total catch in the mainstem below Muskrat Falls (Amec Foster Wheeler 2015a; 2016a). They were commonly collected throughout the sampling program in Goose Bay and Lake Melville in the nearshore areas by JWEL (2001) using beach seines. Collections within the estuarine environment comprised mainly juveniles (JWEL 2001).

5.10 Longnose Sucker (*Catostomus catostomus*)

The longnose sucker can be found throughout North America; from Alaska to western Labrador, and from the northern United States to the southern portion of the Northwest Territories (Scott and Crossman 1973). Longnose suckers are primarily bottom dwellers (McPhail and Lindsey 1970; Morrow 1980) and inhabit lakes, rivers and reservoirs. They have also been reported in brackish waters near the vicinity of river mouths (Walters 1955). Longnose suckers are one of the most abundant species within the lower Churchill River. Except for the pelagic and profundal habitat within Winokapau Lake, they are distributed throughout the main stem downriver to the estuary (Ryan 1980; Anderson 1985; AGRA 1999; AMEC 2000, 2001, 2007, 2009; Amec Foster Wheeler 2015a; Amec Foster Wheeler 2016a) as well as adjoining lakes and tributaries (Ryan 1980; Anderson 1985; AGRA 1999). Beak (1980) also reported this species as most abundant in the upper stretches of the lower Churchill watershed tributary systems, where gradients are gentler and where lakes and ponds are more common along main stems. They are the second-most abundant fish species captured below Muskrat Falls (565 fish total between 1998-2016). They are also very abundant within the brackish water of Goose Bay (941 total captured) and Lake Melville (292 captured).

Spawning generally occurs in the spring (mid April or May); however, Ryan (1980) observed spawning in June in the Labrador region. Longnose suckers are broadcast spawners, with adhesive eggs being repeatedly broadcast over a clean substrate comprised of cobble or rubble. As many as 17,000 to 60,000 eggs per female are released during a spawning period of five days (Scott and Crossman 1998). Eggs will typically incubate for two weeks before hatching, although this is temperature dependent.

Based on habitat utilization data and the habitat-types characterized for the lower Churchill River, highest overall utilization tends to be within intermediate water velocity habitat of the lower Churchill River main stem. A breakdown of habitat utilization by life-cycle stage shows that highest spawning utilization is within slower stream and tributary habitat. Young-of-year habitat use also appears to be highest within stream habitat as well as within intermediate water velocity habitat of the lower Churchill River main stem. This would suggest that once hatched, young longnose sucker have greater survival in faster-velocity habitat within the lower Churchill River. Juvenile utilization is highest in streams and slower habitat within the tributaries as well as intermediate water velocity habitat of the lower Churchill River main stem; that is they tend to utilize slightly slower habitat types than those most-utilized by young-of-year. Adults utilize the littoral zone habitat within Winokapau Lake the highest as well as intermediate and faster water velocity habitat of the lower Churchill River main stem.

Specimens have been captured within the Churchill River from every age-class between one and thirteen (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2016a). Mean length-at-age data shows they range between 65mm in length at age one to almost 400mm at age thirteen. Growth is shown as being relatively linear at a rate near the lower limits exhibited by the species as a whole (Ryan 1980). Growth rates determined from baseline sampling are in concurrence with historic rates for the lower Churchill River provided in Anderson (1985).

The diet of longnose suckers consists entirely of invertebrates, mullocs, and algae/detritus (AGRA 1999; AMEC 2000; Amec Foster Wheeler 2016a). The overall sex ratio of specimens sampled was highly in favour of males (77%). Most sampling has not been conducted at a time period to accurately assess the age of sexual maturity; however, literature data from Anderson (1985) indicates that sexual maturity within the Churchill River system occurs at six to seven years of age.

The vast majority of longnose sucker where tagged and tracked within Sections Four and Five of the lower Churchill River (JWEL 2000). There were considerable migrations shown, with one individual migrating upwards of 204km. The median migration measured however was 13.8km. Of the longnose suckers that were tagged, fifty percent migrated during late May to June, presumably to spawning areas, and returned to original locations during August to early September. There was a concentration of activity surrounding Long Point in Winokapau Lake, suggesting this is a possible spawning area for those fish tagged. The upper section of Winokapau Lake, near Fig and Elizabeth Rivers, also had a substantial amount of movement during spawning season. The identified potential spawning habitats are located within main stem fast and intermediate habitats.

Sampling for total mercury in 1977-78 showed significantly elevated levels as far downstream as Winokapau Lake with a peak of 1.43 mg/kg (~11x background) directly below the tailrace for a standard 400mm length fish. Levels were not significantly different from background by 1996 (Anderson 2011). Mercury concentrations from ongoing baseline data collection associated with the project are provided in **Table** 5-10.

5.11 Rock Cod/Greenland Cod (*Gadus ogac*)

Rock cod is a coastal species, tolerant of low salinities and moderate temperatures and exhibits little preference for any particular bottom substrate type (Backus 1957). They typically do not undertake the extensive seasonal migrations of the Atlantic cod (*Gadus morhua*), but there have been reports of rock cod moving from nearshore to offshore in James Bay during summer (see Morin and Dodson 1986 in JWEL 2001).

Rock cod spawn during February and March in brackish waters (Scott and Scott 1988). They are opportunistic feeders and a large portion of their diet within Lake Melville is comprised of sculpin and flounder, along with small quantities of crab, shrimp, and whelk (see Smith et al. 1981 in JWEL 2001). Diet information collected by JWEL (JWEL 2001) showed rock cod eating, in order of frequency in stomachs, rainbow smelt, mysids, tomcod, and benthic invertebrates.

Table 5-10: Summary of total mercury concentrations in longnose sucker within the baseline study area, 1999-2016

Year	Total Mercury (mg/kg)		
	Sample Size	Mean (SE)	Range
Muskrat Falls reservoir area			
1999	0	-	-
2010	30	0.15 (0.01)	<0.05-0.38
2012	31	0.11 (0.01)	<0.05-0.40
2013	8	0.07 (0.01)	<0.05-0.10
2014	3	0.08 (0.02)	<0.05-0.10
2015	1	0.05	-
2016	4	0.05 (0.02)	<0.02-0.09
Mainstem and Tributaries Below Muskrat Falls			
1999	0	-	-
2010	21	0.03 (<0.01)	0.01-0.09
2011	30	0.11 (0.02)	0.01-0.33
2012	31	0.07 (0.01)	0.03-0.22
2013	30	0.10 (0.01)	<005-0.28
2014	9	0.08 (0.01)	<0.05-0.16
2015	27	0.12 (0.02)	<0.05-0.36
2016	37	0.05 (0.01)	<0.02-0.25
Goose Bay			
1999	0	-	-
2011	31	0.03 (<0.01)	0.02-0.11
2013	30	0.05 (<0.01)	<0.05-0.08
2014	30	0.05 (<0.01)	<0.05-0.06
2015	30	0.05 (<0.01)	-
2016	29	0.02 (<0.01)	<0.02-0.09
Lake Melville			
2011	15	0.07 (0.01)	0.03-0.21
2013	26	0.05 (<0.01)	<0.05-0.08
2014	27	0.05 (<0.01)	<0.05-0.05
2015	30	0.05 (<0.01)	<0.05-0.09
2016	21	0.02 (<0.01)	<0.02-0.06
Eastern Lake Melville			
2016	1	0.02	-

Note: Values below detection limits have been incorporated as the detection limit (i.e. 0.05mg/kg) to produce a conservative estimate of mean concentrations.

In a report on the commercial viability of a Greenland cod fishery in the Lake Melville area, Smith et al. (1981 as noted in JWEL 2001) noted that relative abundances were greatest near Northwest River and to a lesser extent Goose Bay, during a fall and winter survey. During July, rock cod were one of the most abundant fish in collections by JWEL (2001) in Lake Melville, but catches were substantially less in August. In Goose Bay, the relative abundance of rock cod was similar in July and August, but nearly doubled in October (JWEL 2001). The relative abundance of rock cod in Goose Bay and Lake Melville coincided well with when rainbow smelt were most abundant (JWEL 2001). There was also good correlation between the depth at which rock cod and smelt were most common. In Goose Bay, both species were most abundant near the bottom during July, August and October, suggesting a predator-prey relationship (JWEL 2001). Given these indications of movement and feeding, it may be suggested that rock cod will remain in Goose Bay throughout the fall and early winter feeding on rainbow smelt and in late winter, will spawn in the estuary. A single rock cod was captured by Amec Foster Wheeler in 2013 in Goose Bay (AMEC 2013a).

Rock cod were not sampled as part of any post-Smallwood mercury sampling program.

5.12 Atlantic Cod (*Gadus morhua*)

Atlantic cod inhabit cool-temperate to subarctic waters from inshore regions to the edge of the continental shelf (Scott and Scott 1988). Atlantic cod occur throughout the Canadian Atlantic area and in each of the different regions there are one or more identifiable cod stocks, each with its own set of characteristics (Scott and Scott 1988). There are at least 12-14 recognized stocks, of which the most important is the southern Labrador-east Newfoundland stock. Others include the northern Labrador stock.

Although noted in the Calder et al. (2016) paper as being one of the top 20 food sources exposed to MeHg increases downstream of Muskrat Falls, no Atlantic cod have been captured during any sampling in Goose Bay or Lake Melville (Amec Foster Wheeler 2016a; JWEL 2001). Samples included in the Calder et al. (2016) analysis were collected from St. Lewis Bay (see Table S5 in supplemental information) located on the coast of Labrador approximately 300km south of Rigolet and the outlet of Lake Melville into Hamilton Inlet.

Atlantic cod were not sampled as part of any post-Smallwood mercury sampling program.

5.13 Longhorn Sculpin (*Myoxocephalus Octodecemspinosus*)

The longhorn sculpin is a year-round resident of coastal waters, moving into deeper waters in winter and returning to shallower water in spring (Scott and Scott 1988). Longhorn sculpin have not been sampled in Goose Bay and Lake Melville since 1999 (AMEC 2000 JWEL 2001; Amec Foster Wheeler 2016a).

They typically feed on other fish and consume a variety of crabs, shrimp, molluscs, squid, sea squirts, and small fishes such as herring, mackerel, smelt, and sand lance.

Longhorn sculpin were not sampled as part of any post-Smallwood mercury sampling program.

5.14 Capelin (*Mallotus villosus*)

The capelin is a marine fish of cold, deep waters, found in the Atlantic Ocean on the offshore banks and in coastal areas, occasionally spending winter and early spring months in deep bays off the east coast of Newfoundland (Scott and Scott 1982). They are pelagic planktonic feeders, primarily feeding on copepods, amphipods, euphasiids and shrimp (JWEL 2001). The largest concentrations in Canadian waters are typically located off Newfoundland and the Labrador coast. An intensive migration inshore by coastal populations takes place prior to spawning activities on beaches. Beach spawning in south-central Labrador occurs during late June to late July. Where capelin are present within the marine ecosystem, they play an important role as a key food source for larger fish, birds and mammals. In the absence of capelin, as is apparently the case in the relatively warm and fresh Goose Bay/Lake Melville ecosystem, rainbow smelt and possibly sand lance to some extent, fill this niche (JWEL 2001).

There are very few reports of the occurrence of capelin in Lake Melville, but JWEL (2001) noted that this may reflect the lack of fisheries research in the area. If capelin exist in Lake Melville, their occurrence may be restricted by the availability of suitable habitat. Backus (1957) reports that capelin are not known to occur in Lake Melville in the summer, and that no spawning beaches are known in the area. In speculating their absence, Backus (1957) suggested that the water in Lake Melville may be too warm in the summer for capelin to spawn and that spawning may occur in Hamilton Inlet. Further evidence of the absence or low relative abundance of capelin in the area, comes from a study of Rock cod in Lake Melville in 1979, which reported no capelin in any of the stomachs examined (Smith et al. 1979 as in JWEL 2001). The number of rock cod stomachs examined was not provided. Additionally, only two capelin were collected in Lake Melville during July 1998 surveys by JWEL (JWEL 2001) and none collected since then by Amec Foster Wheeler (2016a). Capelin have not been identified in any number within Goose Bay or Lake Melville since the early 1970s (M. Clement, pers. Comm.).

Capelin were not sampled as part of any post-Smallwood mercury sampling program.

5.15 Ringed Seal (*Phoca hispida*)

The ringed seal is one of the most abundant and widely distributed resident Arctic pinnipeds (Muir et al. 1999). The following general species life history description is from Lowry (2016). As a species, ringed seals are widely distributed in ice-covered waters of the northern hemisphere, and they may presently number about three million animals (Lowry 2016). They prefer annual, landfast ice, but are also found in multi-year ice (Kingsley et al. 1985).

Throughout most of their range they use sea ice exclusively as their breeding, molting, and resting (haul-out) habitat, rarely if ever moving onto land (Frost and Lowry 1981, Reeves 1998). Their ability to create and maintain breathing holes in ice using well-developed claws on their fore-flippers allows them to thrive in areas where even other ice-associated seals cannot reside (Lowry 2016). Although Ringed Seals are quite small they deal with the thermal challenges posed by the arctic winter by having a very thick blubber layer, and by building lairs (small caves) in the snow on top of sea ice during the winter. The lairs are particularly important for neonatal survival (Lydersen and Smith 1989). Ringed Seals also use natural cracks along pressure ridges and leads in the sea ice for surfacing and breathing.

Reported mean age at sexual maturity for female Ringed Seals varies in the literature from 3.5 to 7.1 years (Holst and Stirling 2002, Krafft et al. 2006). Males likely do not participate in breeding before they are 8-10 years old. Ringed seals can be long lived, with ages close to 50 reported (Lydersen and Gjertz 1987). Regional productivity rates are variable; reproductive success depends on many factors including prey availability, the relative stability of the ice, and sufficient snow accumulation prior to the commencement of breeding (Lukin 1980, Smith 1987, Lydersen 1995).

A single pup is born in late February-early March for the Ladoga, Saimaa, and Baltic subspecies (Sipilä and Hyvärinen 1998) and March-May for the others (Frost and Lowry 1981). Most births occur in subnivean lairs excavated in snow that accumulates near ice ridges or shorelines. Lairs provide thermal protection against cold air temperatures and high wind chill and afford at least some protection from predators (Smith 1976, Smith and Stirling 1975, Gjertz and Lydersen 1986). For Arctic Ringed Seals, lactation lasts an average of 39 days and pups are weaned at approximately 20 kg (Lydersen and Kovacs 1999). Females become receptive for mating towards the end of the lactation period, similar to other phocid seals.

Ringed Seals molt from mid-May to mid-July and during that period they spend quite a bit of time hauled out (Reeves 1998). Feeding intensity is at a minimum during molting (Ryg et al. 1990).

Although they may dive to more than 500 m (Born et al. 2004), in many areas where they feed, the water is not that deep and dives are correspondingly shallower (Gjertz et al. 2000).

Outside the breeding and molting seasons, Ringed Seal distribution is correlated with food availability (e.g., Simpkins et al. 2003, Freitas et al. 2008). Numerous studies of their diet have been conducted, and although there is considerable regional variation, several patterns emerge. Most Ringed Seal prey are small, and preferred prey tend to be schooling species that form dense aggregations. Fishes are usually in the 5-10 cm length range and crustacean prey in the 2-6 cm range. Typically, a variety of 10-15 prey species are found, with no more than 2-4 dominant prey species for any given area. Fishes are generally more commonly eaten than invertebrates, but diet is determined to some extent by availability of various types of prey during particular seasons as well as by preference, which in part is influenced by energy content of various available prey (Reeves 1998, Wathne et al. 2000). Commonly eaten prey includes cod species redfish, herring, and capelin in marine waters (Lowry et al. 1980, Holst et al. 2001, Labansen et al. 2007). Invertebrate prey species seem to become more important in the open-water season and often dominate the diet of young animals (Lowry et al. 1980, Holst et al. 2001). Large Amphipods, Krill, Mysids, Shrimps, and Cephalopods are all eaten by Ringed Seals and can be very important in some regions at least seasonally (Agafonova et al. 2007).

Ringed seal surveys in Goose Bay and Lake Melville have been completed in 2006 and each year between 2013-2016 (SEM 2007; Amec Foster Wheeler 2016a). Using the seal density within the observed area (approximately 517km^2), a relative abundance estimate for the entire EEM zone was generated for each survey year (**Table** 5-11). Relative abundances have ranged between 644 and 2,140 animals with the 2015 survey being the lowest to date (Amec Foster Wheeler 2016a). Seal ages in Goose Bay and Lake Melville, based on 2016 samples, typically range between pups and adults up to eleven years of age. Since seal samples from Goose Bay and Lake Melville are harvested by a local hunter for consumption by the local community, samples are generally biased toward younger animals. Stomach content analysis has only identified rainbow smelt as prey; however, seals are sampled after whelping and foraging may be more restricted. In addition, pups would only be feeding on milk. Mercury concentrations from ongoing baseline data collection associated with the project are provided in **Table** 5-12.

Table 5-11: Summary of seal relative abundance estimates in Goose Bay and Lake Melville, 2006 through 2016

Sample Year	Total Observed	Relative abundance Estimate	95% Confidence Interval
2006	474	1,888	1,746-2,029
2013	535	2,140	2,081-2,199
2014	196	880	858-901
2015	161	644	621-666
2016	393	1,572	1,523-1,620

Note: Relative abundance estimates and confidence intervals are number of individuals within the entire EEM zone

Table 5-12: Summary of total mercury concentrations (muscle and liver) in ringed seal within the baseline study area, 1999-2016. Only captured in Lake Melville.

Life-stage	Year	Total Mercury (mg/kg)		
		Sample Size	Mean (SE)	Range
Muscle				
Pup	2011	9	0.14 (0.03)	<0.05-0.35
	2012	24	0.04 (0.01)	0.01-0.16
	2013	27	0.09 (0.01)	0.07-0.13
	2014	24	0.13 (0.02)	<0.05-0.30
	2015	24	0.09 (0.01)	0.06-0.15
	2016	25	0.07 (<0.01)	<0.05-0.11
Non Pup	2011	5	0.24 (0.05)	0.09-0.39
	2012	6	1.24 (1.01)	0.16-6.30
	2013	3	0.16 (0.03)	0.11-0.20
	2014	4	0.81 (0.34)	0.19-1.43
	2015	3	0.52 (0.18)	0.27-0.87
	2016	5	0.45 (0.20)	0.17-1.25
Liver				
Pup	2012	24	0.32 (0.07)	0.04-1.70
	2013	27	0.33 (0.04)	<0.05-0.9
	2014	24	0.54 (0.10)	0.09-1.81
	2015	24	0.25 (0.02)	0.13-0.44
	2016	25	0.30 (0.05)	0.09-1.23
Non Pup	2012	6	39.66 (16.65)	0.98-110.00
	2013	3	17.67 (7.61)	2.50-26.40
	2014	4	12.91 (2.88)	7.76-18.20
	2015	3	10.86 (0.95)	9.07-12.30
	2016	5	36.19 (12.62)	6.76-78.30

Note: Values below detection limits have been incorporated as the detection limit (i.e. 0.05mg/kg) to produce a conservative estimate of mean concentrations.

Calder et al. (2016) classified Ringed seals as spending up to 25% of their time in riverine habitat; however, during aerial surveys each season, the lower reach of the Churchill River is flown for seal presence and in all years, no ringed seals have been recorded within the river itself (SEM 2007; Amec Foster Wheeler 2016a). Very few seals are observed within Goose Bay (Amec Foster Wheeler 2016a). However, it should be noted that harbour seals (*Phoca vitulina*) have been observed within the river during fisheries surveys during open water; the most observed at any location and time has been three (McCarthy, unpubl data).

6.0 LIFE HISTORY SUMMARY

The bioaccumulation factors (BAFs) calculated by Calder et al. (2016) were the quotient of the methylmercury concentration within each fish species divided by the methylmercury concentration within the water. They were adjusted prior to final incorporation into the risk estimate model based on an estimate of the fraction of lifespan each species spent feeding in each environment (i.e., marine, estuary, freshwater) (see Supplemental Table S7a and S7b in Calder et al. 2016). However, based on the data presented in this report, modifications to the Calder et al. (2016) final BAFs are recommended to better represent actual habitat use.

In total, the baseline sampling program for the Lower Churchill Hydroelectric Development, which includes Muskrat Falls, has sampled over 10,140 fish from 19 different species between 1998-2016. Many fish species that are relied upon by local residents of the Lake Melville area such as lake trout, Arctic charr, Atlantic salmon (both anadromous and landlocked), and Atlantic cod have either not been captured, or captured in extremely low relative abundance both within and downstream of the Muskrat Falls reservoir area. This includes Goose Bay and Lake Melville. These species would not therefore be considered within the zone of influence of the project. These data of species relative abundance and distribution should be considered in the context of any mercury modelling exercises and especially in determining or completing any Human Health Risk Assessments.

7.0 CLOSURE

The biological and habitat suitability data presented within this report has been compiled using baseline data collected by Amec Foster Wheeler and others since 1998. The methodologies used to collect and generate the data are generally accepted practices described in detail within the EEM and the Fish Habitat Compensation Plan baseline studies, and have been used for studies within the lower Churchill River, as well as other projects throughout Newfoundland and Labrador (AMEC 2013b).

Yours truly,

Wood Environment & Infrastructure Solutions

Prepared by:

James H. McCarthy, MSc CFP
Associate Biologist and Ecosystem Group Lead

Reviewed by:

David Robbins, M.Env.Sci
Senior Scientist

8.0 REFERENCES

Agafonova, E. V., Verevkin, M. V., Sagitov, R. A., Sipilä, T., Sokolovskay, M. V. and Shahnazarova, V. U.2007. The Ringed Seal in Lake Ladoga and the Valaam Archipelago. Baltic Fund for Nature of Saint- Petersburg Naturalist Society, St. Petersburg State University & Metsähallitus, Natural Heritage Servives, Vammalan Kirjapaino OY., Vammala, Finland.

AGRA Earth & Environmental. 1999. Fish and Fish Habitat, Churchill River Power Project (LHP98-07). Prepared for Labrador Hydro Project, St. John's, NF.

AMEC Earth & Environmental. 2000. Freshwater Fish Mercury Sampling, Churchill River, Labrador (LHP99-07). Prepared for Labrador Hydro Project, St. John's, NF.

AMEC Earth & Environmental. 2001. Churchill River Power Project: A Proposed Framework for HADD Determination (LHP00-07). Prepared for Labrador Hydro Project, St. John's, NF.

AMEC Earth & Environmental Limited. 2007. Lower Churchill Hydroelectric Development Fish and Fish Habitat Baseline Study: catch-based Utilization Index Validation and Additional Habitat Surveys. Prepared for Labrador Hydro Project, St. John's, NF ii + 54 pp. + Appendices

AMEC Earth & Environmental. 2008. Granite Canal Hydroelectric Development Fish Habitat Compensation Facility (FHCF) 2007 Monitoring Program (Year 5). Prepared for Newfoundland and Labrador Hydro, St. John's, NL.

AMEC Earth & Environmental. 2009. Fish Passage Facility (FPF) Assessment of the Lower Churchill River Crossing, Trans Labrador Highway, Goose Bay, NL. Prepared for the department of Transportation and Works, Government of Newfoundland and Labrador, St. John's, NL.

AMEC Earth & Environmental. 2012. 2011 Freshwater Fish Habitat Compensation Monitoring Northeast River and On-Site Long Harbour. Prepared for Vale NL, St. John's, NL.

AMEC Earth & Environmental and BAE-Newplan Group Limited. 2001. Aquatic Environment in the Goose Bay Estuary Final Report. Report prepared for Labrador Hydro Project, St. John's NL.

AMEC Environment & Infrastructure. 2013a. Lower Churchill Hydroelectric Development Freshwater Fish Habitat Compensation Plan; Muskrat Falls. Report prepared for Nalcor Energy, St. John's, NL.

AMEC Environment & Infrastructure. 2013b. Lower Churchill Hydroelectric Generation Project Aquatic Environmental Effects Monitoring Program; Muskrat Falls. Report prepared for Nalcor Energy, St. John's, NL.

Amec Foster Wheeler. 2015a. Aquatic Environmental Effects Monitoring Program: 1998 to 2014 Baseline Conditions, Muskrat Falls. Report prepared for Nalcor Energy, St. John's NL

Amec Foster Wheeler. 2015b. Fish Habitat Compensation 1998 to 2014 Baseline Conditions, Muskrat Falls Reservoir Area. Report prepared for Nalcor Energy, St. John's NL

Amec Foster Wheeler. 2016a. Aquatic Environmental Effects Monitoring Program: 1998 to 2015 Baseline Conditions, Muskrat Falls. Report prepared for Nalcor Energy, St. John's NL

Amec Foster Wheeler. 2016b. Fish Habitat Compensation 1998 to 2015 Baseline Conditions, Muskrat Falls Reservoir Area. Report prepared for Nalcor Energy, St. John's NL

Anderson, M.R. 2011. Duration and extent of elevated mercury levels in downstream fish following reservoir creation. River Systems 19(3):167-176.

Anderson, T.C. 1985. The Rivers of Labrador. Can. Spec, Publ. Fish. Aquat. Sci. 81:389 p.

Backus, R.H. 1957. The fishes of Labrador. Bulletin of the American Museum of Natural History 113: 277-337.

Beak Consultants Limited. 1979. Fisheries resources of selected tributaries of the Lower Churchill River. Prepared for Newfoundland and Labrador Hydro, St. John’s, Newfoundland, 90p.

Beak. 1980. Fisheries Resources of Tributaries of the Lower Churchill River. Prepared by Beak Consultants Limited for the Lower Churchill Development Corporation, St. John’s, NF.

Becker, G.C. 1983. Fishes of Wisconsin. University of Wisconsin Press, Madison, 1052p.

Belyanina, T.N. 1969. Synopsis of biological data on smelt, Osmerus eperlanus (Linnaeus) 1758. Food and Agriculture Organization of the United Nations Fisheries Synopsis Number 78, 57p.

Black, G.A., J.B. Dempson, and W.J. Bruce. 1986. Distribution and postglacial dispersal of freshwater fishes of Labrador. Canadian Journal of Zoology 64: 21-31.

Bobbitt, J., and S. Aikenhead. 1982. Influence of controlled discharge from the Churchill River on the oceanography of Groswater Bay, Labrador, Can. Tech. Rep. Fish. Aquat. Sci. 1097: iv + 43p.

Born, E.W., Teilmann, J., Acquarone, M. and Rigét, F.F. 2004. Habitat use of ringed seals (*Phoca hispida*) in the North Water Area (North Baffin Bay). Arctic 57: 129-142.

Bradbury, C. M.M. Roberge, and C.K. Minns. 1999. Life History Characteristics of Freshwater Fishes Occurring in Newfoundland and Labrador, with Major Emphasis on Lake Habitat Characteristics. Can. MS Rep. Fish. Aquat. Sci. 2485:vii+150p.

Bruce, W.J. 1974. The limnology and fish populations of Jacopie Lake, west forebay, Smallwood Reservoir, Labrador. Canada Fisheries and Marine Service, Newfoundland Region, Resource Development Branch, St. John's, Newfoundland Technical Report Series Number NEW/T; 74-2, 74p.

Bruce, W.J.; Morry, C.J.; Rowe, L.W.; Wiseman, R.J. 1975. An overview of fisheries problems associated with the proposed Lower Churchill hydroelectric development, Gull Island, Labrador. Fisheries and Marine Service. Newfoundland Region; Lower Churchill Hydroelectric Project. Internal report series (Canada. Fisheries and Marine Service. Newfoundland Region); NEW/I-75-2. 1 v.

Bruce, W.J. and R.F. Parsons. 1976. Age, growth and maturity of lake chub (Couesius plumbeus) in Mile 66 Brook, Ten Mile Lake, Labrador. Canada Fisheries and Marine Service, Research and Development Directorate, Technical Report Number 683, 13p.

Buckley, J.L. 1989. Species Profiles: Life histories and environmental requirements of coastal fishes and invertebrates (North Atlantic)—rainbow smelt. US Fish and Wildlife Service, Biological Report 82 (11.106). US Army Corps Engineers, TR EL-82-4. 11p.

Calder, R.S.D., A.T. Schartup, M. Li, A.P. Valberg, P.H. Balcom, E.M. Sunderland. 2016. Future Impacts of Hydroelectric Power Developments on Methylmercury Exposures of Canadian Indigenous Communities. Environm. Sci. Technol. 50(23): 13115-13122.

Christenson, L.M. and L.L. Smith. 1965. Characteristics of fish populations in upper Mississippi backwater areas United States Fish and Wildlife Service Circular Number 212.

Clark, C.F. 1950. Observations on the spawning habits of the northern pike, Esox lucius, in northwestern Ohio. Copeia 1950: 285-288.

Coad, B.W. and G. Power. 1973. Observations on the ecology and phenotypic variation of the threespine stickleback, Gasterosteus aculeatus L., 1758, and the blackspotted stickleback, G. Wheatlandi Putnam, 1967, (Osteichthyes: Gasterosteidae) in Amory Cove, Quebec. Canadian Field Naturalist 87: 113-122

COSEWIC. 2010. COSEWIC assessment and status report on the Atlantic Salmon Salmo salar (Nunavik population, Labrador population, Northeast Newfoundland population, South Newfoundland population, Southwest Newfoundland population, Northwest Newfoundland population, Quebec Eastern North Shore population, Lake Ontario population, Gaspe-Southern Gulf of St. Lawrence population, Eastern Cape Breton population, Nova Scotia Southern Upland population, Inner Bay of Fundy population, Outer Bay of

Fundy population) in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. Xlvii+136pp. (www.sararegistry.gc.ca/status/status_e.cfm).

Crossman, E.J. 1978. Taxonomy and distribution of North American esocids. American Fisheries Society Special Publication 11: 13-26.

Cunjak, R.A. and E.M.P. Chadwick. 1989. Downstream Movements and Estuarine Residence by Atlantic Salmon Parr (*Salmo salar*). Can J. Fish. Aquat. Sci. 46: 1466-1471.

Cunjak R.A., R.L. Saunders, E.M.P. Chadwick. 1990. Seasonal Variations in the Smolt Characteristics of Juvenile Atlantic Salmon (*Salmo salar*) from Estuarine and Riverine Environments. Can. J. Fish. Aquat. Sci. 47: 813-820.

Cunjak, R.A. 1992. Comparative feeding, growth, and movements of Atlantic salmon (Salmo salar) parr from riverine and estuarine environments. Ecology of Freshwater Fish 1992: 1: 26-34.

Dillon. 2016. Final Baseline Human Health Risk Assessment: Lower Churchill Hydroelectric Generation Project. Report prepared for Nalcor Energy, St. John's NL

Doyon, J.-F., L. Bernatchez, M. Gendron, R. Verdon, and R. Fortin. 1998. Comparison of normal and dwarf populations of lake whitefish (Coregonus clupeaformis) with reference to hydroelectric reservoirs in northern Quebec. Arch. Hydrobiol. Spec. Issues Advanc. Limnol. 50: 97-108.

Einarsson, S.M., D.H. Mills, and V. Johannsson. 1990. Utilization of fluvial and lacustrine habitat by anadromous Atlantic salmon, Salmo salar L., in an Icelandic watershed. Fisheries Research 10: 53-71.

Fitzpatrick, D.G. 1988. A comparative study of aspects of the biology of sympatric sticklebacks (Gasterosteus aculeatus) and (Apeltes quadracus) at Point Verde, Placentia Bay, Newfoundland. B.Sc. Hon. Thesis, Memorial University of Newfoundland, St. John’s, Newfoundland, 49p.

Ford, B.S., P.S. Higgins, A.F. Lewis, K.L. Cooper, T.A. Watson, C.M. Gee, G.L. Ennis, R.L. Sweeting. 1995. Literature reviews of the life history, habitat requirements mitigation/compensation strategies for thirteen sport fish species in the Peace, Liard and Columbia river drainages of British Columbia. Canadian Manuscript Report of Fisheries and Aquatic Sciences Number 2321, xxii+342p.

Franklin, D.R. and L.L. Smith Jr. 1963. Early life history of the Northern pike, Esox lucius L., with special reference to the factors influencing the numerical strength of year classes. Transactions of the American Fisheries Society 92(2): 91-110.

Freitas, C., Kovacs, K.M., Ims, R.A., Fedak, M.A. and Lydersen, C. 2008. Ringed seal post-moulting movement tactics and habitat selection. Oecologia 155: 193-204.

Frost, K. J. and Lowry, L. F. 1981. Ringed, Baikal and Caspian seals *Phoca hispida* Schreber, 1775; *Phoca sibirica* Gmelin, 1788; and *Phoca caspica* Gmelin, 1788. In: S. H. Ridgway and R. Harrison (eds), Handbook of marine mammals, Vol. 2: Seals, pp. 29-53. Academic Press.

Gjertz, I. and Lydersen, C. 1986. Polar bear predation on ringed seals in the fast-ice of Hornsund, Svalbard. Polar Research 4: 65-68.

Gjertz, I., Kovacs, K.M., Lydersen, C. and Wiig, Ø. 2000. Movements and diving of adult ringed seals (*Phoca hispida*) in Svalbard. Polar Biology 23: 651-656.

Grant, C.G.J. and E.M. Lee. 2004. Life History Characteristics of Freshwater Fishes Occurring in Newfoundland and Labrador, with Major Emphasis on Riverine Habitat Requirements. Can. Manuscr. Rep. Fish. Aquat. Sci. 2672: xii+262p.

Griswold, B.L. and L.L. Smith Jr. 1972. Early survival and growth of the ninespine. stickleback, Pungitius pungitius. Transactions of the American Fisheries Society 101: 350-352.

Hagen, D.W. 1967. Isolating mechanisms in threespine stickleback (Gasterosteus). Journal of the Fisheries Research Board of Canada 24: 1637-1692.

Hagen, J., S. Decker and R. Bison. 2004. Calibrated rapid assessment methods for assessments of juvenile steelhead abundance in the Thompson River: 2001-2003 progress report. Unpublished report prepared for B.C. Ministry of Water Land and Air Protection, Fisheries Branch, Kamloops, BC.

Harvey, K.A. and K. Warner. 1970. The land-locked salmon (Salmo salar): Its life history and management in Maine. Maine Department of Inland Fish and Game, Augusta, Maine, 129p.

Hatch. 2007. Ice Dynamics of the Lower Churchill River. Prepared for Newfoundland and Labrador Hydro, St. John's, NL.

Holst, M., Stirling, I. and Hobson, K. A. 2001. Diet of ringed seals (*Phoca hispida*) on the east and west sides of the North Water Polynya, northern Baffin Bay. Marine Mammal Science 17(4): 888-908.

Holst, M. and Stirling, I. 2002. A comparison of ringed seal (*Phoca hispida*) biology on the east and west sides of the North Water Polynya, Barfin Bay. Aquatic Mammals 28: 221-230.

Inskip, P.E. 1982. Habitat suitability index models: Northern pike. United States Department of the Interior, Fish and Wildlife Service FWS/OBS-82/10.17, 40p.

JWEL (Jacques Whitford Environment Limited). 2000. Fish Migration and Habitat Use of the Churchill River (LHP98-03). Prepared for Labrador Hydro Project, St. John's, NF.

JWEL. 2001. Biological Study of the Goose Bay Estuary. Prepared by Jacques Whitford Environment Limited, St. John's for Newfoundland and Labrador Hydro, St. John's, NL. JWEL Project No. 1201. 91p + appendices.

Krafft, B. A., Kovacs, K. M., Frie, A. K., Haug, T. and Lydersen, C. 2006. Growth and population parameters of ringed seals (*Pusa hispida*) from Svalbard, Norway, 2002-2004. ICES Journal of Marine Science 63: 1136-1144.

Kingsley, M.C.S., I. Stirling, and W. Calvert. 1985. The distribution and abundance of seals in the Canadian high Arctic, 1980-82. Can J. Fish. Aquat. Sci. 42:1189-1210.

Labansen, A. L., Lydersen, C., Haug, T. and Kovacs, K. M. 2007. Spring diet of ringed seals (*Pusa hispida*) from north-western Spitsbergen, Norway. ICES Journal of Marine Science 64: 1246-1256.

Larson, G.E. 1976. Social behavior and feeding ability of two phenotypes of Gasterosteus aculeatus in relation to their spatial and trophic segregation in a temperate lake. Canadian Journal of Zoology 54: 107-121.

Lee, S.H. 1971. Fecundity of four species of salmonid fishes in Newfoundland waters. M.Sc. thesis, Memorial University of Newfoundland, St. John's, Newfoundland, 114p.

Leggett, W.C. 1965. The effect of the environment on the food, growth, reproduction and survival of ouananiche in eastern Canada. Fish Culture Development Branch. Newfoundland Region, St. John's, Newfoundland, Progress Report Number 27, 87p.

Leggett, W.C. and G. Power. 1969. Differences between two populations of landlocked Atlantic salmon (Salmo salar) in Newfoundland. Journal of the Fisheries Research Board of Canada 26: 1585-1596.

Leim, A. H. and W.B. Scott . 1966. Fishes of the Atlantic coast of Canada. Fisheries Research Board of Canada Bulletin Number 155, 485p.

Lewis, D.B. 1978. The distribution of sticklebacks (Gasterosteidae) on the Avalon Peninsula of Newfoundland, and the meristic and mensural variations of some selected populations. B.Sc. Honours thesis, Dept. of Biology, Memorial University of Newfoundland. 46p.

LGL Limited. 1999. Churchill River power project – Atikonak Lake ecosystem study: fish and fish habitat of tributaries. Prepared by LGL Limited, St. John's for Newfoundland and Labrador Hydro, St. John's, Newfoundland, 85p + App.

Li, M. A.T. Schartup, A.P. Valberg, J.D. Ewald, D.P. Krabbenhoft, R. Yin, P. H. Balcom, and E.M. Sunderland. 2016. Environmental Origins of Methylmercury Accumulated in Subarctic Estuarine Fish Indicated by Mercury Stable Isotopes. Environ. Sci. Technol. 50: 11559-11568.

Lowry, L. 2016. *Pusa hispida*. The IUCN Red List of Threatened Species 2016: e.T41672A45231341. http://dx.doi.org/10.2305/IUCN.UK.2016-1.RLTS.T41672A45231341.en. Downloaded on 27 January 2017.

Lowry, L. F., Frost, K. J. and Burns, J. J. 1980. Variability in the diet of ringed seals, *Phoca hispida*, in Alaska. Canadian Journal of Fisheries and Aquatic Sciences 37: 2254-2261.

Lukin, L. R. 1980. Habitat of White Sea ringed seal in the initial period of postnatal development. Soviet Journal of Marine Biology 6: 266-269.

Lydersen, C. 1995. Energetics of pregnancy, lactation and neonatal development in ringed seals (*Phoca hispida*). In: A. S. Blix, L. Walløe and Ø. Ulltang (eds), Developments in Marine Biology 4: Whales, seals, fish and man, pp. 319-327. Elseview, Amsterdam, The Netherlands.

Lydersen, C. and Gjertz, I. 1987. Population parameters of ringed seals (*Phoca hispida* Schreber 1775) in Svalbard area. Canadian Journal of Zoology 65: 1021-1027.

Lydersen, C. and Kovacs, K. M. 1999. Behaviour and energetics of ice-breeding, North Atlantic phocid seals during the lactation period. Marine Ecology Progress Series 187: 265-281.

Lydersen, C. and Smith, T.G. 1989. Avian predation on ringed seal *Phoca hispida* pups. Polar Biology 9:489-490.

Machniak, K. 1975. The effects of hydroelectric development on the biology of northern fishes, (reproduction and population dynamics) 1. lake whitefish Coregonus clupeaformis (Mitchill): A literature review and bibliography. Canada. Fisheries and Marine Service. Research and Development Directorate, Winnipeg, MB. Technical Report Number 527, 67p.

Martin, N.V. and C.H. Olver. 1980. The lake charr, Salvelinus namaycush, p. 205-277. In E.K. Balon (ed.), Charrs – salmonid fishes of the genus Salvelinus. Dr. W. Junk Publishers, The Hague, Netherlands.

McCarraher, D.B. and R.E. Thomas. 1972. Ecological significance of vegetation to Northren pike, Esox lucius, spawning. Transactions of the American Fisheries Society 3: 560-563.

McPhail, J D. and C.C. Lindsey. 1970. Freshwater fishes of northwestern Canada and Alaska. Fisheries Research Board of Canada Bulletin Number 173, 381p.

Minaskuat Limited Partnership. 2007. Water and Sediment Quality in the Churchill River, Environmental Baseline Report (LHP 06-04). Prepared for Newfoundland and Labrador Hydro Report No. MIN0271.

Minaskuat, 2008, Project Area Ecological Land Classification Environmental Baseline Report (LCP 535722). Prepared Newfoundland and Labrador Hydro, St. John's, NL. 87pp + appendices.

Morrow, J.E. 1980. The freshwater fishes of Alaska. Alaska Northwest Publishing Co., Anchorage, Alaska, 248p.

Muir, D., B. Braune, B. DeMarch, R. Morstrom, R. Wagemann, L. Lockhart, B. Hargrave, D. Bright, R. Addison, J. Payne, K. Reimer. 1999. Spatial and temporal trends and effects of contaminants in the Canadian Arctic marine ecosystem: a review. Science of the Total Environment 230:83-144.

Nalcor, 2009. Lower Churchill Hydroelectric Generation Project Environmental Impact Statement, Volume II Part A: Biophysical Assessment. Nalcor Energy, St. John's, NL.

Parsons, R. F. 1975. The limnology and fish biology of Ten Mile Lake, Labrador. Fisheries and Marine Service, Resource Development Branch, St. John's, Newfoundland. Technical Report Series. Number NEW/T-75-3, 75p.

Power, C. 1980. The brook charr, p. 141-203. In E.K. Balon (ed.), Charrs, Salmonid Fishes of the Genus Salvelinus. Dr. W. Junk Publishers, The Hague, Netherlands.

Reeves, R. R. 1998. Distribution, abundance and biology of ringed seals (*Phoca hispida*): an overview. NAMMCO Scientific Publications 1: 9-45.

Ryan, P.M. 1980. Fishes of the Lower Churchill River, Labrador. Fisheries and Marine Service Technical Report No. 922.

Ryan, P.M. 1984. Age, growth, and food of threespine stickleback (Gasterosteus aculeatus) in three lakes of central Newfoundland, Canada. Canadian Technical Report of Fisheries and Aquatic Sciences Number 1325, 16p.

Ryan, P.M., and R. Knoechel. 1994. Lake use by brook trout, Salvelinus fontinalis, in insular Newfoundland, Canada. Verhandlungen der Internationalen Vereinigung fur Theoretische und Angewandte Limnologie 25: 2068-2073.

Ryg, M., Smith, T. G. and Øritsland, N. A. 1990. Seasonal changes in body mass and body composition of ringed seals (*Phoca hispida*) on Svalbard. Canadian Journal of Zoology 68: 470-475.

Sandlund, O.T., B. Jonsson, H.J. Malmquist, R. Gydemo, T. Lindem, S. Skulason, S.S. Snorrason, and P.M. Jonasson. 1987. Habitat use of Arctic charr, Salvenlinus alpinus, in Thingvallavatn, Iceland. Environmental Biology of Fishes 20: 263-274.

Schartup, A.T., P. H. Balcom, A.L. Sorensen, K.J. Gosnell, R.S.D. Calder. R.P. Mason, and E.M. Sunderland. 2016. Freshwater discharges drive high levels of methylmercury in Arctic marine biota. PNAS 112(38): 11789-11794.

Scott W. B. and E.J. Crossman. 1964. Fishes occurring in the fresh waters of insular Newfoundland. Queen's Printer, Ottawa, Ontario. iv + 124 p.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Bryant Press Limited, Ottawa, Ontario, xix + 966p.

Scott, W.B. and E.J. Crossman. 1998. Freshwater Fishes of Canada. Galt House Publications, Ontario. 966pp. ISBN 0-9690653-9-6.

Scott, W.B., and M.G. Scott. 1988. Atlantic fishes of Canada. Canadian Bulletin of Fisheries and Aquatic Sciences Number 219, 731p.

Scruton, D.A. 1984. A survey of selected lakes in Labrador, with an assessment of lake status and sensitivity in relation to acid precipitation. Canadian Technical Report of Fisheries and Aquatic Sciences Number 1296, v+115p.

Scruton, D. A., D. R. Sooley, L. Moores, M. A. Barnes, R. A. Buchanan, and R. N. McCubbin 1997. Forestry guidelines for the protection of fish habitat in Newfoundland and Labrador. Department of Fisheries and Oceans, St. John's, Newfoundland, 63p.

Scruton, D.A, L.J. LeDrew, and R.S. McKinley. 1995. Use of Radiotelemetry to evaluate overwintering of landlocked Atlantic salmon (*Salmo salar*) in relation to two hydroelectric developments in Newfoundland, Canada. 8th Workshop on the Hydraulics of Ice Covered Rivers, Kamloops, BC, pp. 143-158.

SEM. 2007. Lower Churchill Hydroelectric Generation Project Environmental Baseline Report: Seal Abundance and Distribution. Prepared by Sikumiut Environmental Management, St. John's, NL for Labrador Hydro Project, St. John's, NL. 16p + appendices.

Simpkins, M.A., Hiruki-Raring, L.M., Sheffield, G., Grebmeier, J.M. and Bengtson, J.L. 2003. Habitat selection by ice-associated pinnipeds near St. Lawrence Island, Alaska in March 2001. Polar Biology 26:577-586.

Sipilä, T. and Hyvärinen, H. 1998. Status and biology of Saimaa (*Phoca hispida* saimensis) and Ladoga (*Phoca hispida* ladogensis) ringed seals. In: M. P. Heide-Jorgensen and C. Lydersen (eds), Ringed seals in the North Atlantic, NAMMCO.

Smith, K.E.H. 1988. The Atlantic salmon. Underwater World Factsheet. Fisheries and Oceans Canada, Ottawa, Ontario. Catalogue Number Fs 41-33/19-1988E, 8p.

Smith, T. G. 1987. The ringed seal, *Phoca hispida*, of the Canadian western Arctic. Canadian Bulletin of Fisheries Research and Aquatic Sciences 216.

Smith, T. G. and Stirling, I. 1975. The breeding habitat of the ringed seal (*Phoca hispida*). The birth lair and associated structures. Canadian Journal of Zoology 53: 1297-1305.

Toner, E.D. and G.H. Lawler. 1969. Synopsis of biological data on the pike, Esox lucius (Linnaeus, 1758). Food and Agriculture Organization Fisheries Synopsys Number 30 (Revision 1).

Vilks, G. and P.J. Mudie. 1983. Evidence for postglacial paleoceanographic and paleoclimatic changes in Lake Melville, Labrador, Canada. Arctic and Alpine Research, 15:307-320.

Virgl, J.A and J.D. McPhail. 1994. Spatio-temoral distribution of anadromous (trachurua) and freshwater (leiurus) threespine sticklebacks, Gasterosteus aculeatus. Canadian Field-Naturalist 108: 355-360.

Walters, V. 1955. Fishes of the western Arctic America and eastern Arctic Siberia: taxonomy and zoogeography. Bulletin of the American Museum of Natural History 106: 255-368.

Wathne, J. A., Haug, T. and Lydersen, C. 2000. Prey preference and niche overlap of ringed seals *Phoca hispida* and harp seals *P. groenlandica*. Marine Ecology Progress Series 94: 233-239.

Wootton, R.J. 1984. A functional biology of sticklebacks. University of California Press. Berkeley and Los Angeles, California.

Memo

To: Peter Madden, Nalcor

From: Jim McCarthy

cc: Reed Harris, Randy Baker

Date: May 10, 2018

Re. Summary of Isotope and Stomach Data, Goose Bay / Lake Melville Estuary

1. Introduction

As part of the ongoing baseline data collection for the Environmental Effects Monitoring (EEM) Program for the Muskrat Falls portion of the lower Churchill Project, fish have been collected for numerous analyses. Presented below is a brief summary of ongoing stable isotope and stomach content data that provides estimates of downstream habitat use and feeding behaviour to support recent modelling of mercury bioaccumulation and exposure risk due to consumption. The data has been separated by location of capture below Muskrat Falls (e.g., riverine below Muskrat Falls, Goose Bay, inner Lake Melville, and outer Lake Melville). Inner Lake Melville includes all sample locations in the western portion of the lake while outer Lake Melville are those fish sampled near Valley Bight at the eastern end of Lake Melville (Figures 1-1 and 1-2). Additional food web analysis is ongoing as part of PhD research.

Fin clips have been collected from subsets of fish and analyzed for stable isotope ($\delta^{13}C$ and $\delta^{15}N$) ratios by the Stable Isotope in Nature Laboratory (SINLab) at UNB. The ratio of stable isotopes of nitrogen can be used to estimate trophic position because the $\delta^{15}N$ of a consumer is typically enriched by 3-4$^{o/oo}$ relative to its diet (DeNiro and Epstein 1981, Post 2002, Jardine et al 2003, Borga et al. 2011). When comparing among ecosystems (eg. Freshwater to estuary), the $\delta^{15}N$ and $\delta^{13}C$ of an organism alone provides little information about its absolute trophic position or ultimate source of carbon. This is because there is considerable variation among ecosystems in the $\delta^{15}N$ and $\delta^{13}C$ or the base of the food web from which organisms draw their nitrogen and carbon (Post 2002). Without suitable estimates of food web base $\delta^{15}N$ and $\delta^{13}C$, there is no way of knowing if variation reflects changes in food web structure and carbon flow, or just variation in the base nitrogen or carbon values. The simplest model for estimating the trophic position of a secondary consumer is: trophic position = $\lambda + (\delta^{15}N_{secondary\ consumer} - \delta^{15}N_{base})/\Delta_n$, where λ is the trophic position of the organism used to estimate $\delta^{15}N_{base}$ (Post 2002, Borga et al. 2011).

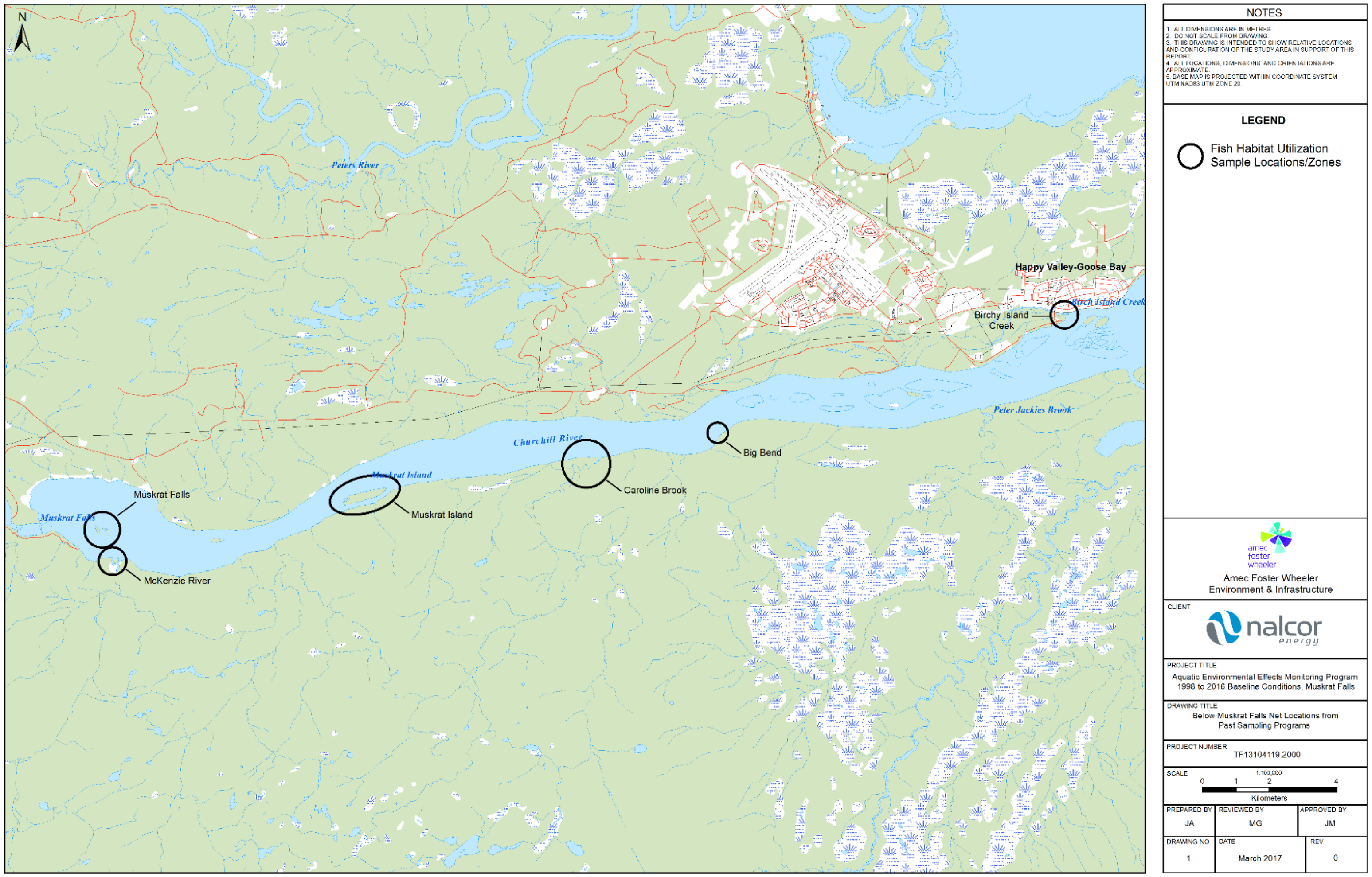


Figure 1-1: Overall EEM study area: mainstem of the lower Churchill River (AMEC 2013b).

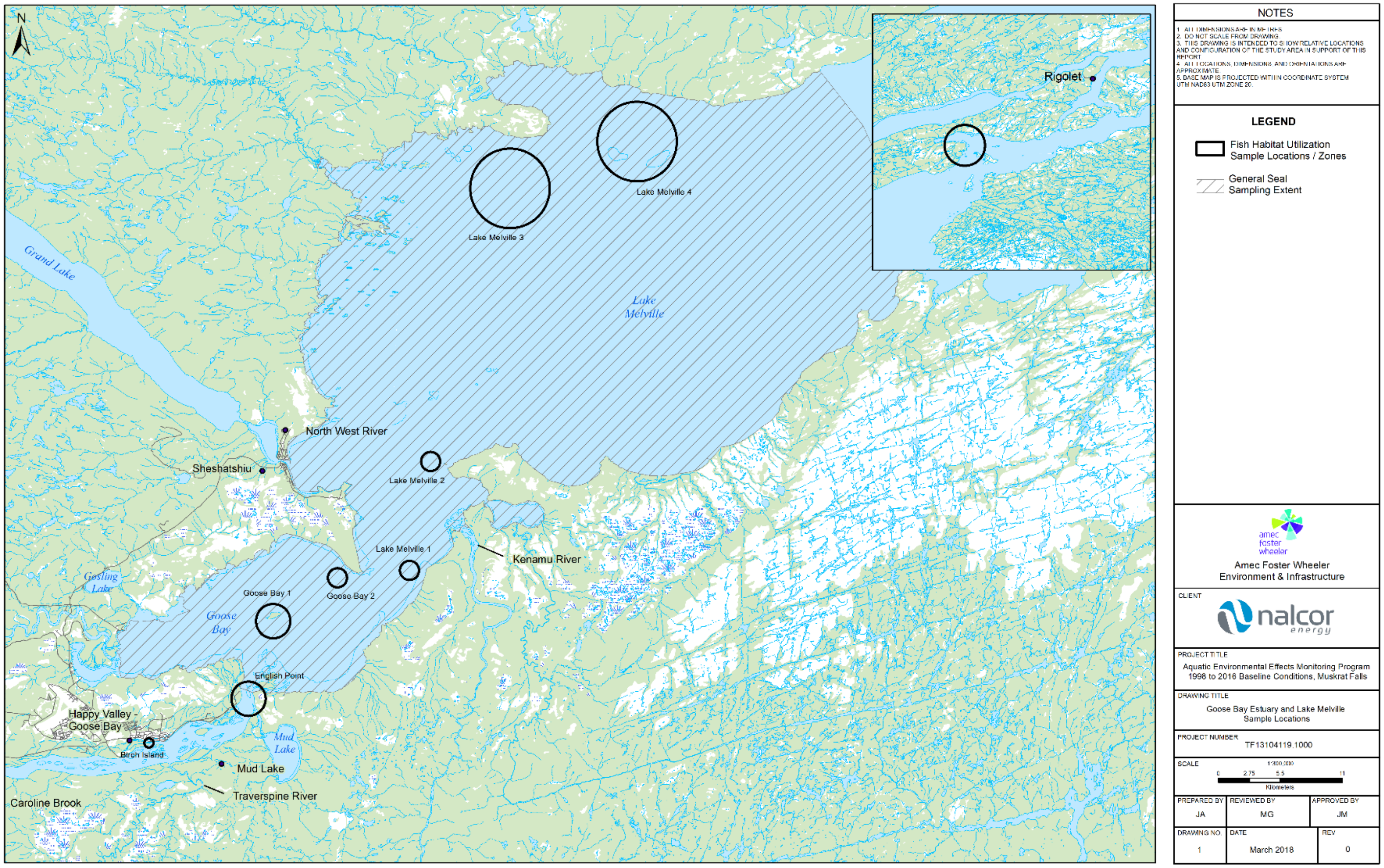


Figure 1-2: Overall EEM study area: Goose Bay estuary and Lake Melville (AMEC 2013b).

Using isotope data from base organisms from the main stem of the Churchill River and estuary (e.g., molluscs, phytoplankton and zooplankton), the trophic position of each fish species was estimated.

In addition to stable isotopes, prey selection by key species has been ongoing via stomach content analysis which can augment isotope data. Stomach content analysis of a subset of samples (focusing on salmonids, northern pike, rainbow smelt and tomcod) was completed from 2017 to augment the trophic results determined by stable isotope ratios. The data presented has been characterized as the percent of all non-empty stomachs analyzed that contained that prey type and does not estimate the quantity within each stomach. Since one fish could have been feeding on multiple prey types, a single stomach sample can be included in multiple categories. Because the number of benthic macroinvertebrate families is high, individual families were consolidated into a larger benthic macroinvertebrate category for ease of presentation.

2. General Isotope Trends

To illustrate the general trends in isotope data, Figure 2-1 shows a generalized plot of isotope signatures for fish sampled in the estuarine (Goose Bay and Lake Melville) and freshwater environments of the lower Churchill River and its tributaries below Muskrat Falls in 2017. The graph shows the division of isotope signatures between the two habitats, as shown by variations in the $\delta^{13}C$ values. It also shows that there are fish that have been sampled in the freshwater environment that display isotope signatures similar to estuarine environments; however, the species and numbers are limited. Note that the identification of 'estuarine' and 'freshwater' are not indicative of the life history of the species, rather it identifies the location in which the specimen was captured (i.e. estuarine samples have been collected from Goose Bay and Lake Melville, while freshwater samples are from the mainstem and associated tributaries below Muskrat Falls). For example, species such as brook trout that are captured in the freshwater of the lower Churchill River below Muskrat Falls show an estuarine isotope signature because they are returning from feeding in the estuary and do not spend considerable time in the main stem prior to migrating up tributaries to spawn.

Freshwater

Figure 2-2 presents the isotope signatures for all species sampled within the mainstem of the Churchill River and tributaries below Muskrat Falls during 2017. Brook trout and Atlantic salmon have the highest $\delta^{15}N$ values and therefore make up the highest trophic levels sampled in 2017.

A general $\delta^{13}C$ ratio greater than -23 can indicate estuarine/marine habitat use (B. Graham, pers. comm. 2011). Several species captured in freshwater in 2017 (i.e. brook trout, northern pike and white sucker) showed $\delta^{13}C$ ranges that could potentially include a marine signature (Figure 2-3). Since netting in Goose Bay and Lake Melville began, brook trout and white sucker have been captured in relatively high abundances in these habitats (see Amec Foster Wheeler 2016). There have been very few northern pike captured within the estuary, however isolated captures of juveniles around Rabbit Island in Goose Bay have occurred. Pike could be preying on fish with estuarine influence (i.e., prey may be feeding near/within the estuary environment).

Goose Bay and Lake Melville

Samples collected from Goose Bay and Lake Melville also show within species variability. Figure 2-4 presents isotope ranges for each fish captured in Goose Bay and Lake Melville during 2017. Brook trout, rainbow smelt and tomcod occupied the highest trophic levels in 2017, similar to past sampling programs. Unlike the freshwater habitats, very few fish captured in the estuary environment showed potential freshwater signatures.

Since isotope analysis of ringed seal muscle samples began, they have consistently been shown to occupy the highest trophic level within Goose Bay and Lake Melville (Figure 2-6), indicating that they are likely relying on fish as a primary food source.

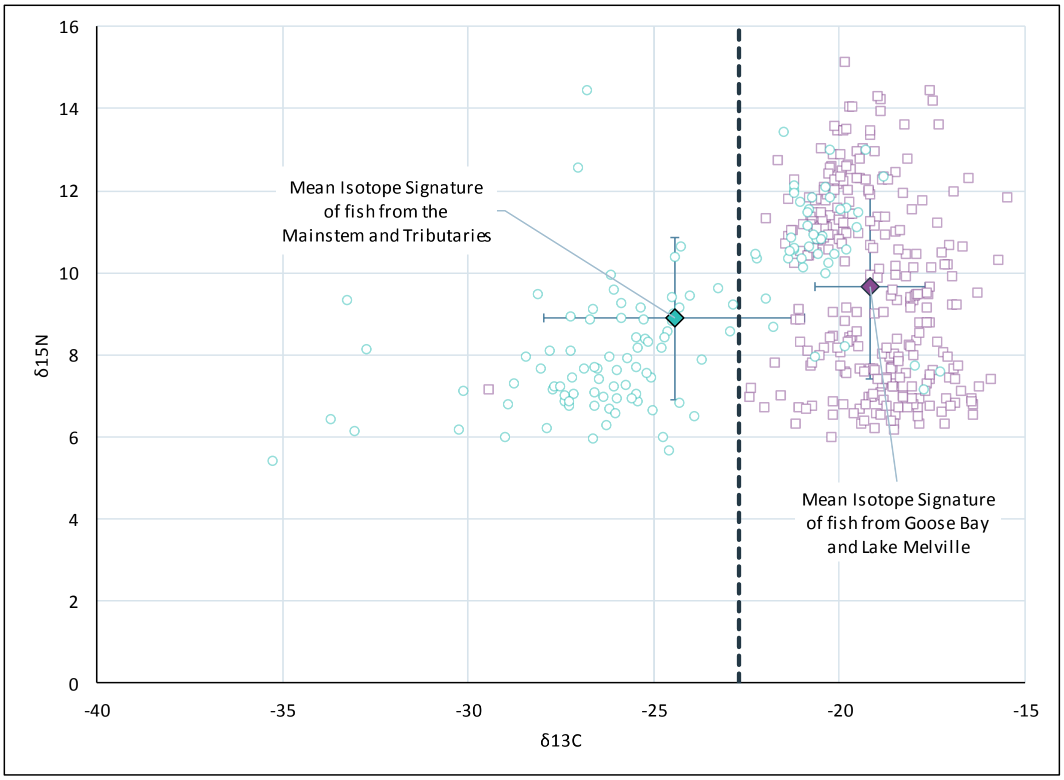


Figure 2-1: Isotope signatures from fish captured within the mainstem and tributaries below Muskrat Falls, Goose Bay and Lake Melville, 2017

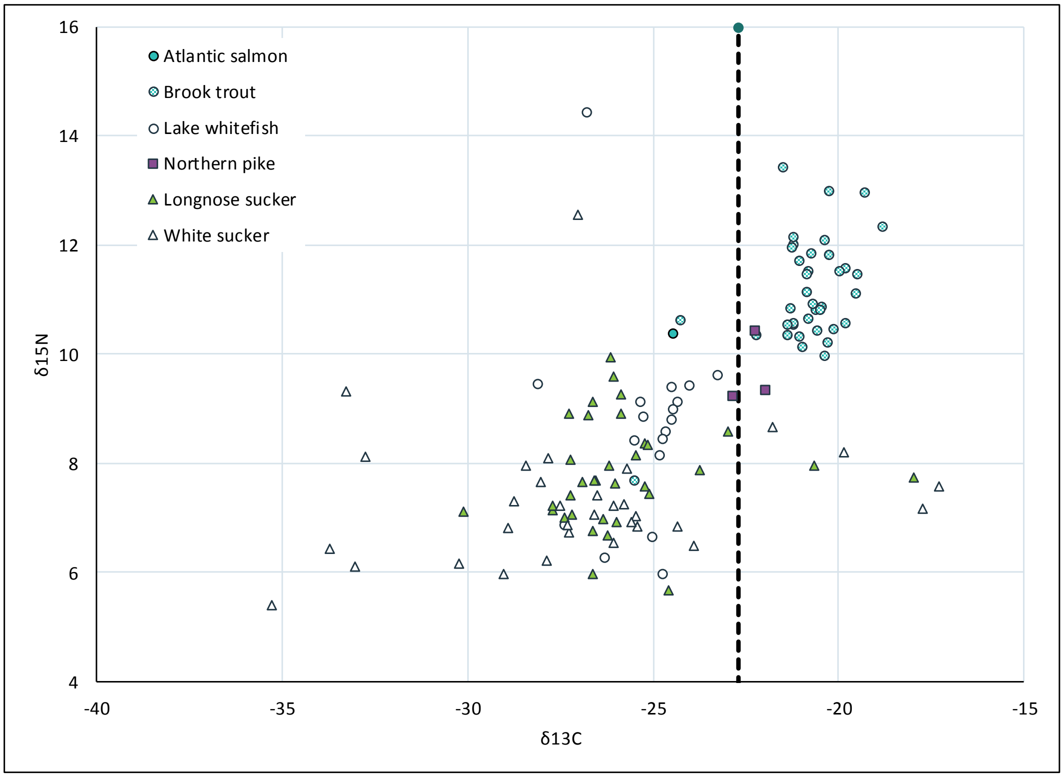


Figure 2-2: Isotope signatures of fish captured the mainstem and tributaries below Muskrat Falls, 2017

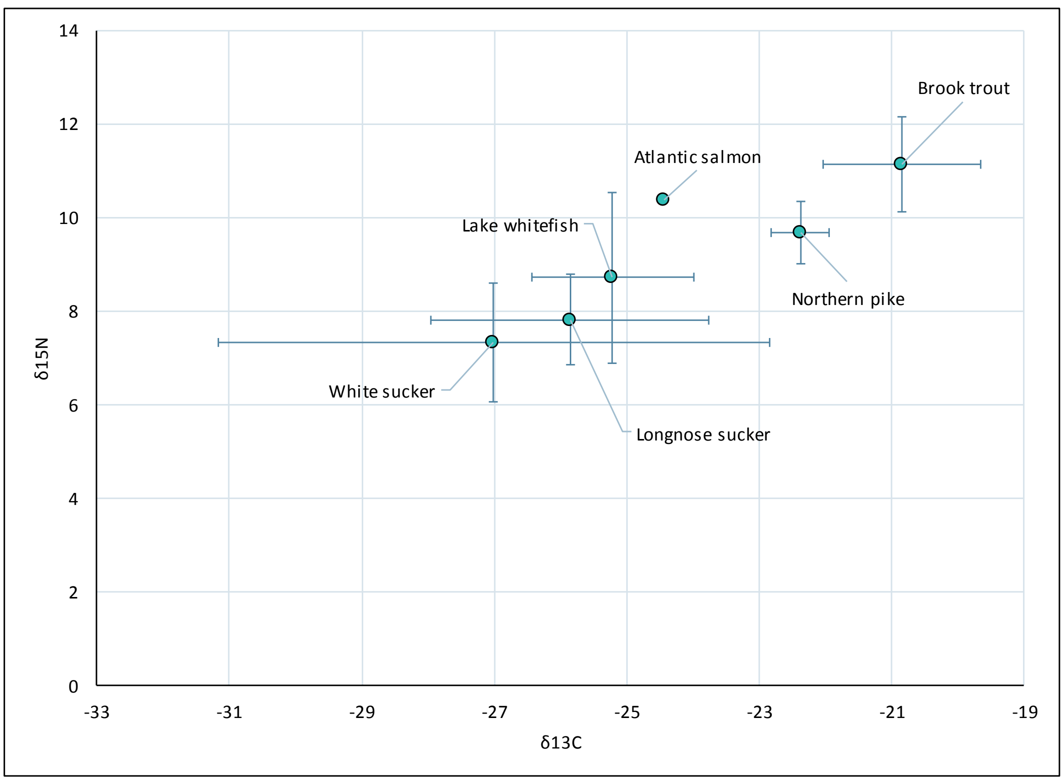


Figure 2-3: Variability in carbon (habitat usage) and nitrogen (trophic level) signatures in fish captured in the mainstem and tributaries below Muskrat Falls, 2017

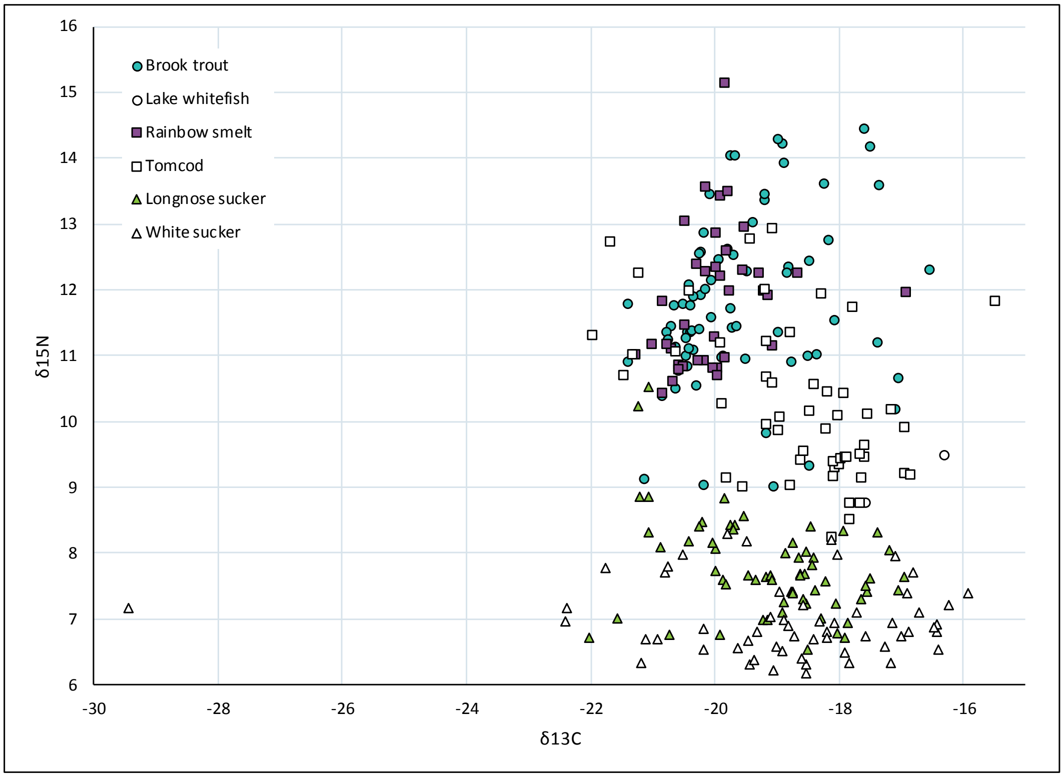


Figure 2-4: Isotope signatures of fish captured in Goose Bay and Lake Melville, 2017

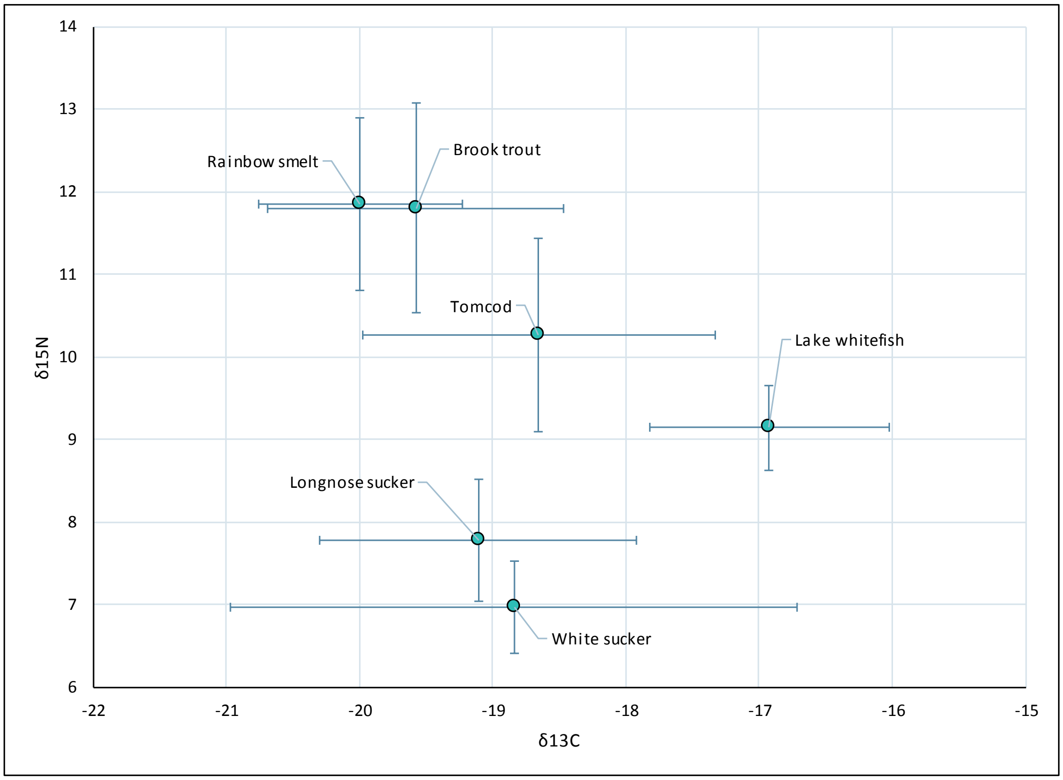


Figure 2-5: Variability in carbon (habitat usage) and nitrogen (trophic level) signatures in fish captured Goose Bay and Lake Melville, 2017

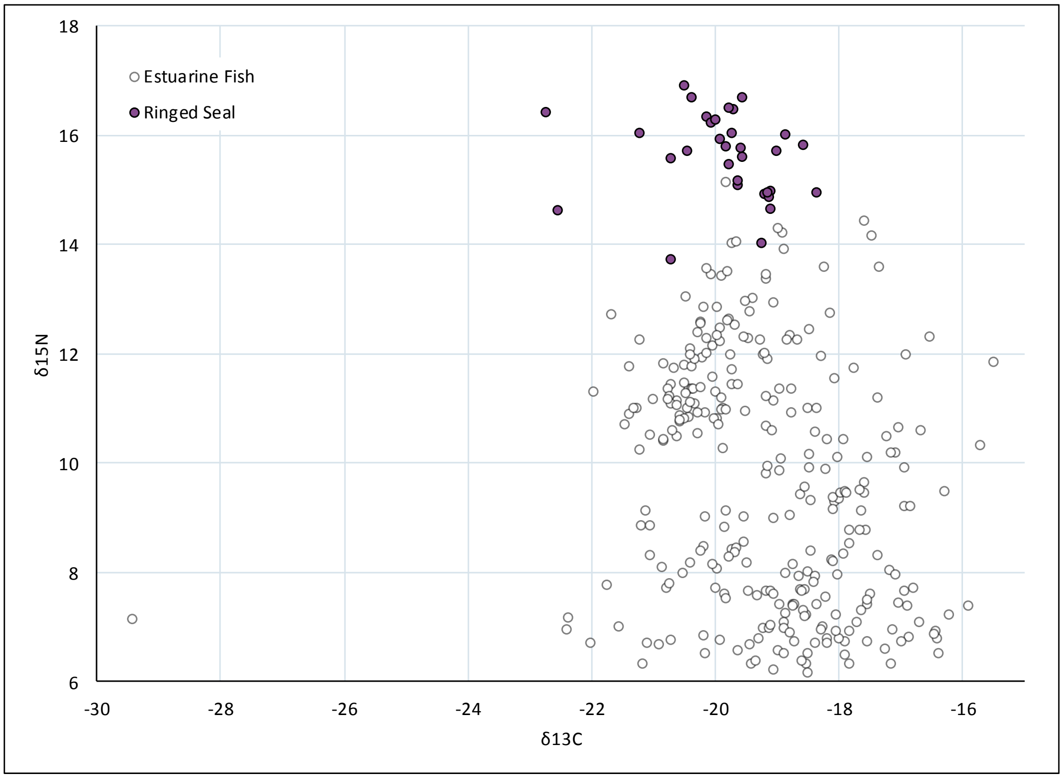


Figure 2-6: Ringed seal isotope signatures, 2017

3. Food Chain / Species Trophic Position

Table 1-1 provides a summary of mean stable isotope data collected from key fish species included in the EEM program since 2011 separated in the above noted locations. This data was used to estimate trophic position and food chain length for each species. Table 1-2 provides a general summary of stomach contents within each species to assist in feeding habitat characterization (e.g., freshwater or saline) and interpretation of isotope values.

As shown in Table 1-1, most piscivorous fish species are at the highest trophic positions, generally between values of 4-5. These species include Atlantic herring, Atlantic salmon, brook trout, rainbow smelt, and tom cod which all primarily feed in the estuarine environment as adults (Atlantic salmon off the Labrador coast). Species such as lake whitefish show a slightly lower trophic position as they typically rely on benthic invertebrates and larger amphipods/zooplankton and do not consume other fish species. It is noteworthy that northern pike also appear to be at a general trophic range of 2-5-3.5 likely due to their major prey being suckers (a fish also at a lower trophic range of ~1.5-2.5). Ringed seals have the greatest trophic values as they likely feed on a variety of fish and are one of the top predators monitored in Lake Melville.

It should be noted that some of the variability within the trophic ranges of fish could be due to variations in sample sizes between years and varying size classes of fish sampled (e.g., the data were not separated by age-class or size) for this exercise.

Table 3-1: Summary of mean annual stable isotope data and trophic level estimates, Churchill River, Labrador.

Species	Location	Year	Sample Size (n)	Mean Carbon δC^{13} ($^{o/oo}$)	Mean Nitrogen δN^{15} ($^{o/oo}$)	Estimated Trophic Level & Food Chain Length[1]
Atlantic herring	Goose Bay	2011	1	-18.4	13.2	4.7
Atlantic salmon	Inner Lake Melville	2010	6	-19.8	10.7	4.0
Atlantic salmon	Inner Lake Melville	2013	2	-18.7	11.9	4.3
Atlantic salmon	Inner Lake Melville	2015	22	-21.0	11.9	4.3
Brook trout	Goose Bay	2011	43	-21.3	10.9	4.0
Brook trout	Goose Bay	2013	26	-19.7	10.7	4.0
Brook trout	Goose Bay	2014	30	-18.9	12.8	4.6
Brook trout	Goose Bay	2015	6	-19.1	10.9	4.0
Brook trout	Goose Bay	2016	6	-19.5	11.1	4.1
Brook trout	Goose Bay	2017	11	-18.5	11.0	4.1
Brook trout	Inner Lake Melville	2013	35	-18.5	10.7	4.0
Brook trout	Inner Lake Melville	2014	30	-19.7	11.9	4.3
Brook trout	Inner Lake Melville	2015	29	-18.9	12.6	4.5
Brook trout	Inner Lake Melville	2016	30	-19.6	11.8	4.3
Brook trout	Inner Lake Melville	2017	32	-19.7	12.2	4.4
Brook trout	Outer Lake Melville	2017	29	-19.8	11.7	4.3
Brook trout	Below Muskrat Falls	2010	2	-26.4	14.3	5.5
Brook trout	Below Muskrat Falls	2011	12	-23.0	11.1	4.6
Brook trout	Below Muskrat Falls	2012	18	-22.3	11.9	4.8
Brook trout	Below Muskrat Falls	2013	30	-22.4	10.2	4.3
Brook trout	Below Muskrat Falls	2014	8	-24.3	9.06	4.0
Brook trout	Below Muskrat Falls	2015	11	-23.9	10.4	4.4
Brook trout	Below Muskrat Falls	2016	35	-22.4	10.6	4.4
Brook trout	Below Muskrat Falls	2017	40	-20.9	11.1	4.6
Lake Whitefish	Goose Bay	2011	1	-19.1	10.2	3.8
Lake Whitefish	Goose Bay	2013	4	-20.7	9.1	3.5
Lake Whitefish	Goose Bay	2014	1	-18.4	11.2	4.1
Lake Whitefish	Goose Bay	2016	2	-21.1	9.5	3.6
Lake Whitefish	Inner Lake Melville	2014	7	-19.1	9.2	3.5
Lake Whitefish	Inner Lake Melville	2015	2	-20.2	9.3	3.6
Lake whitefish	Goose Bay	2017	2	-16.9	9.1	3.5
Lake whitefish	Below Muskrat Falls	2010	6	-24.3	8.6	2.9
Lake whitefish	Below Muskrat Falls	2011	14	-24.4	9.8	3.3
Lake whitefish	Below Muskrat Falls	2012	5	-22.8	9.6	3.2
Lake whitefish	Below Muskrat Falls	2014	3	-24.8	9.2	3.1
Lake whitefish	Below Muskrat Falls	2015	2	-23.2	9.2	3.1
Lake whitefish	Below Muskrat Falls	2016	2	-21.6	10.8	3.6
Lake whitefish	Below Muskrat Falls	2017	19	-25.2	8.7	3.0
Rainbow Smelt	Goose Bay	2011	30	-20.9	12.6	4.5
Rainbow Smelt	Goose Bay	2013	21	-20.4	12.7	4.6
Rainbow Smelt	Goose Bay	2014	2	-18.2	14.2	5.0
Rainbow Smelt	Goose Bay	2016	1	-19.9	9.3	3.6
Rainbow Smelt	Inner Lake Melville	2013	21	-20.3	12.7	4.6
Rainbow Smelt	Inner Lake Melville	2014	25	-19.6	13.3	4.7
Rainbow Smelt	Inner Lake Melville	2015	12	-20.1	12.7	4.6
Rainbow Smelt	Inner Lake Melville	2016	6	-20.2	11.3	4.1
Rainbow Smelt	Outer Lake Melville	2016	16	-18.3	10.6	3.9
Rainbow Smelt	Inner Lake Melville	2017	16	-20.4	11.6	4.2
Rainbow Smelt	Outer Lake Melville	2017	22	-19.7	12.1	4.4
Rainbow Smelt	Below Muskrat Falls	2016	1	-21.6	10.7	3.8
Tom cod	Goose Bay	2011	6	-20.5	12.6	4.1
Tom cod	Goose Bay	2013	8	-20.7	11.0	4.0

Species	Location	Year	Sample Size (n)	Mean Carbon δC^{13} (o/oo)	Mean Nitrogen δN^{15} (o/oo)	Estimated Trophic Level & Food Chain Length[1]
Tom cod	Goose Bay	2014	1	-19.0	10.7	4.0
Tom cod	Goose Bay	2016	6	-20.9	10.9	4.4
Tom cod	Inner Lake Melville	2011	7	-20.4	13.1	4.7
Tom cod	Inner Lake Melville	2013	12	-17.4	10.3	3.8
Tom cod	Inner Lake Melville	2014	30	-18.7	11.5	4.2
Tom cod	Inner Lake Melville	2015	8	-17.9	11.5	4.2
Tom cod	Inner Lake Melville	2016	30	-18.0	10.6	3.8
Tom cod	Goose Bay	2017	3	-19.1	12.1	4.5
Tom cod	Inner Lake Melville	2017	39	-18.6	10.1	3.8
Tom cod	Outer Lake Melville	2017	11	-18.8	10.5	3.9
Winter flounder	Inner Lake Melville	2011	10	-19.6	13.5	4.8
Longnose sucker	Goose Bay	2011	29	-18.8	7.5	1.6
Longnose sucker	Goose Bay	2013	27	-17.8	8.0	1.7
Longnose sucker	Goose Bay	2014	29	-18.4	8.5	1.9
Longnose sucker	Goose Bay	2015	29	-17.4	8.0	1.7
Longnose sucker	Goose Bay	2016	29	-18.6	7.4	1.5
Longnose sucker	Inner Lake Melville	2011	15	-17.7	8.6	1.9
Longnose sucker	Inner Lake Melville	2013	26	-18.5	8.2	1.8
Longnose sucker	Inner Lake Melville	2014	26	-19.1	8.6	1.9
Longnose sucker	Inner Lake Melville	2015	30	-18.3	7.8	1.7
Longnose sucker	Inner Lake Melville	2016	21	-18.9	7.2	1.5
Longnose sucker	Outer Lake Melville	2016	1	-18.4	8.1	1.6
Longnose sucker	Inner Lake Melville	2017	32	-18.9	7.7	1.7
Longnose sucker	Outer Lake Melville	2017	32	-19.3	7.8	1.7
Longnose sucker	Below Muskrat Falls	2011	26	-26.4	8.0	2.8
Longnose sucker	Below Muskrat Falls	2012	29	-26.4	7.3	2.6
Longnose sucker	Below Muskrat Falls	2013	29	-23.4	8.9	3.0
Longnose sucker	Below Muskrat Falls	2014	9	-25.9	9.2	3.1
Longnose sucker	Below Muskrat Falls	2015	27	-26.4	7.8	2.7
Longnose sucker	Below Muskrat Falls	2016	31	-26.6	7.7	2.7
Longnose sucker	Below Muskrat Falls	2017	36	-25.7	7.8	2.7
Northern Pike	Goose Bay	2013	1	-21.0	7.7	2.7
Northern Pike	Below Muskrat Falls	2010	7	-25.5	8.1	2.8
Northern Pike	Below Muskrat Falls	2011	5	-28.2	9.0	3.1
Northern Pike	Below Muskrat Falls	2012	7	-24.6	8.9	3.0
Northern Pike	Below Muskrat Falls	2013	28	-24.6	9.3	3.1
Northern Pike	Below Muskrat Falls	2014	10	-24.5	10.1	3.4
Northern Pike	Below Muskrat Falls	2015	5	-25.9	8.8	3.0
Northern Pike	Below Muskrat Falls	2016	15	-25.8	9.2	3.1
Northern Pike	Below Muskrat Falls	2017	3	-22.4	9.7	3.3
Ringed Seal	Inner Lake Melville	2011	14	-19.5	15.5	5.4
Ringed Seal	Inner Lake Melville	2012	30	-19.1	16.2	5.6
Ringed Seal	Inner Lake Melville	2013	29	-19.2	16.1	5.6
Ringed Seal	Inner Lake Melville	2014	28	-19.5	16.0	5.5
Ringed Seal	Inner Lake Melville	2015	27	-19.6	15.9	5.5
Ringed Seal	Inner Lake Melville	2016	29	-20.1	16.0	5.5
Ringed Seal	Inner Lake Melville	2017	30	-19.9	15.6	5.4

[1] Based on each trophic level accounting for approximately 3.4 o/oo although it is recognized that this can be variable.

Table 3-2: Summary of prey diversity below Muskrat Falls, 2017

Species	Vegetation	Invertebrates	Fish	Plankton
Freshwater				
Brook trout		Odonata, Ephemerelidae, Daphniidae, Plecoptera, Tipulidae, Chironimidae, Coleoptera, Hydroptilidae, Leptophlebidae, Diptera	Rainbow smelt, Lake Chub, Sculpin	
Lake whitefish	Filamentous algae	Daphniidae, Leptoceridae, Chironimidae, Cyelopidae, Chydoridae, Podocopida		
Longnose sucker	Filamentous algae	Chironimid, Hydrachnidia, Bivalves		
Northern pike			3-spine stickleback, Longnose sucker, White sucker	
Rainbow smelt				
Atlantic salmon		Pteronacidae	Unidentified fish	
Goose Bay				
Brook trout		Tricoptera, Chironimidae, Odonata, Formicidae, Hymenoptera	Sculpin, Tomcod, Rainbow smelt, Longnose sucker	
Lake whitefish		Chironimidae, Diptera, Hymenoptera	Sculpin, Unidentified fish	
Longnose sucker				
Northern pike				
Rainbow smelt				Decopod
Tomcod			Lake chub, Sand lance	
Lake Melville				
Brook trout		Diptera, Chironimidae, Hydroptilidae, Ichnumonidae, Cicadellidae, Staphylinidae, Hymenoptera, Bivalve	Tomcod, Sand lance, 3-spine stickleback, Winter flounder, Rainbow smelt, Unidentified fish	Amphipod, Decopod
Lake whitefish				
Longnose sucker				
Northern pike				
Rainbow smelt			Sand lance, Rainbow smelt, Unidentified fish	Decapod, Amphipod
Tomcod		Chironimidae	Sand lance, Sculpin, Rainbow smelt, Lake chub	Amphipod, Decapod, Isopod

Brook trout were collected in all four habitat areas (below Muskrat Falls, Goose Bay, inner Lake Melville, and outer Lake Melville). Below Muskrat Falls, Brook trout displayed generally greater range in tropic level ($\delta^{15}N$), indicating variation in diet (Figure 2-3). In the estuary environments, brook trout showed one of the greatest ranges in $\delta^{15}N$ signatures (Figure 2-5) and suggests they may be opportunistic feeders and are likely preying on various fish and planktonic species.

Stomach content analysis summary is provided in Figure 3-1. As shown, the influence of benthic macroinvertebrates is greatest in those fish captured within or near (i.e. Goose Bay) the lower Churchill River. Similar to other years, most 2017 brook trout in freshwater were captured in September within tributaries of the lower Churchill River such as Caroline Brook and McKenzie River. The presence of marine prey such as sandlance and rainbow smelt in a percentage of the non-empty stomachs indicates a return from the estuary environment. The presence of benthic invertebrates as prey in samples from inner Lake Melville was much lower, possibly indicating lower influence of freshwater. A general increase in prey diversity can also be seen from samples collected from outer Lake Melville (e.g., Valley Bight area). Brook trout from the more eastern portion of the lake preyed on amphipods and decapods which were not identified in freshwater, Goose Bay or inner Lake Melville samples. Sandlance appeared to be a prevalent prey item within Lake Melville while tomcod seemed to play a greater role as prey in Goose Bay but less so further into Lake Melville.

The brook trout stomach content results support the isotope values recorded in both the freshwater and estuary environment. Brook trout captured and sampled in the freshwater environment are feeding on benthic macroinvertebrates and fish with some of the fish being estuary origin. The estuary samples indicate higher numbers of brook trout preying on fish along with zooplankton in outer Lake Melville. This places them near the higher $\delta^{15}N$ values and would explain the high range of $\delta^{15}N$ values measured as they feed at various trophic levels. A similar trophic level and chain length among freshwater and estuary samples indicates the general movement of brook trout into the estuary from freshwater environments to feed (see Table 1-1).

Tomcod were sampled in all estuary environments (i.e., Goose Bay, inner Lake Melville and outer Lake Melville) but not in freshwater. Similar to brook trout and rainbow smelt, tomcod showed one of the greatest ranges in $\delta^{15}N$ signatures (Figure 2-5) and suggest that they may be opportunistic feeders and are likely preying on various fish and planktonic species.

With respect to stomach content analysis, a high proportion of stomachs from Goose Bay were empty in contrast to those from Lake Melville (Figure 3-2). In Goose Bay, fish was the only prey item identified (sandlance and lake chub). In Lake Melville, there seemed to be little freshwater influence in terms of prey items and greater presence of amphipods, isopods and decapods and generally lower predation on fish species in outer Lake Melville. This is also evident in the $\delta^{15}N$ isotope signatures which tended to be lower than those of brook trout and rainbow smelt (see Figure 2-5).

Rainbow smelt were also sampled in all estuary environments, similar to tomcod. Also similar to tomcod and brook trout, they showed some of the largest range in $\delta^{15}N$ signatures (Figure 2-5) and may suggest that they are opportunistic feeders and are likely preying on various fish and planktonic species.

However, those rainbow smelt sampled for stomach contents in Goose Bay appeared to rely heavily on decapods (Figure 3-3). Within inner Lake Melville, fish was the most prevalent prey item with fish and zooplankton (amphipods and decapods) preyed upon in outer Lake Melville.

This trend is similar in some ways to tomcod and likely reflects general prey availability for these species within Lake Melville.

Rainbow smelt showed a similar $\delta^{15}N$ isotope signature range to that of brook trout which indicates that the relative proportions of prey items may be similar among these species diet. They both appear to be the two fish species (of those sampled) highest on the estuary food web (see Figure 1-7).

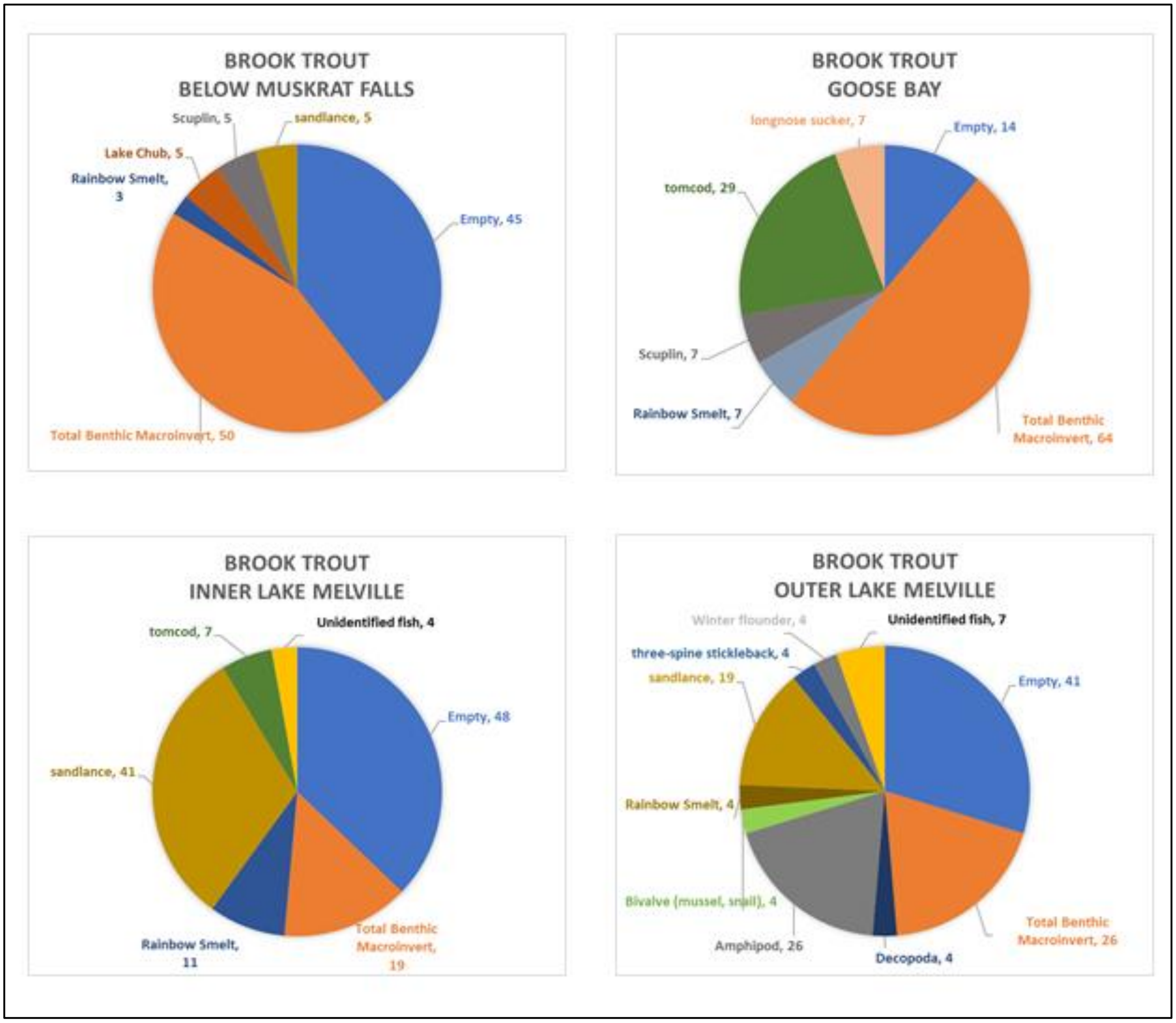


Figure 3-1: Brook trout stomach content analysis. Numbers presented are the percentage of stomachs which contained that prey item.

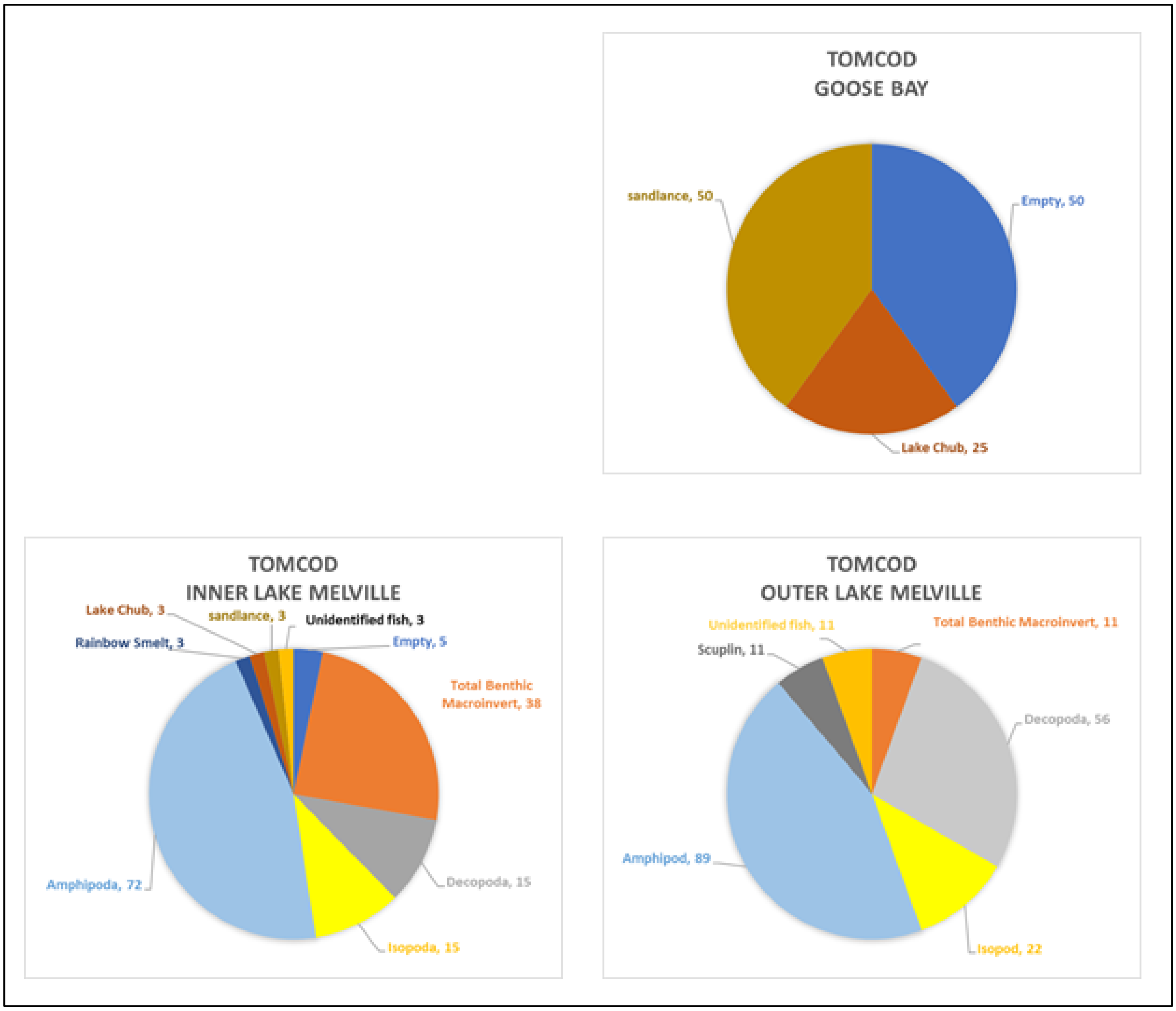


Figure 3-2: Tomcod stomach content analysis. Numbers presented are the percentage of stomachs which contained that prey item.

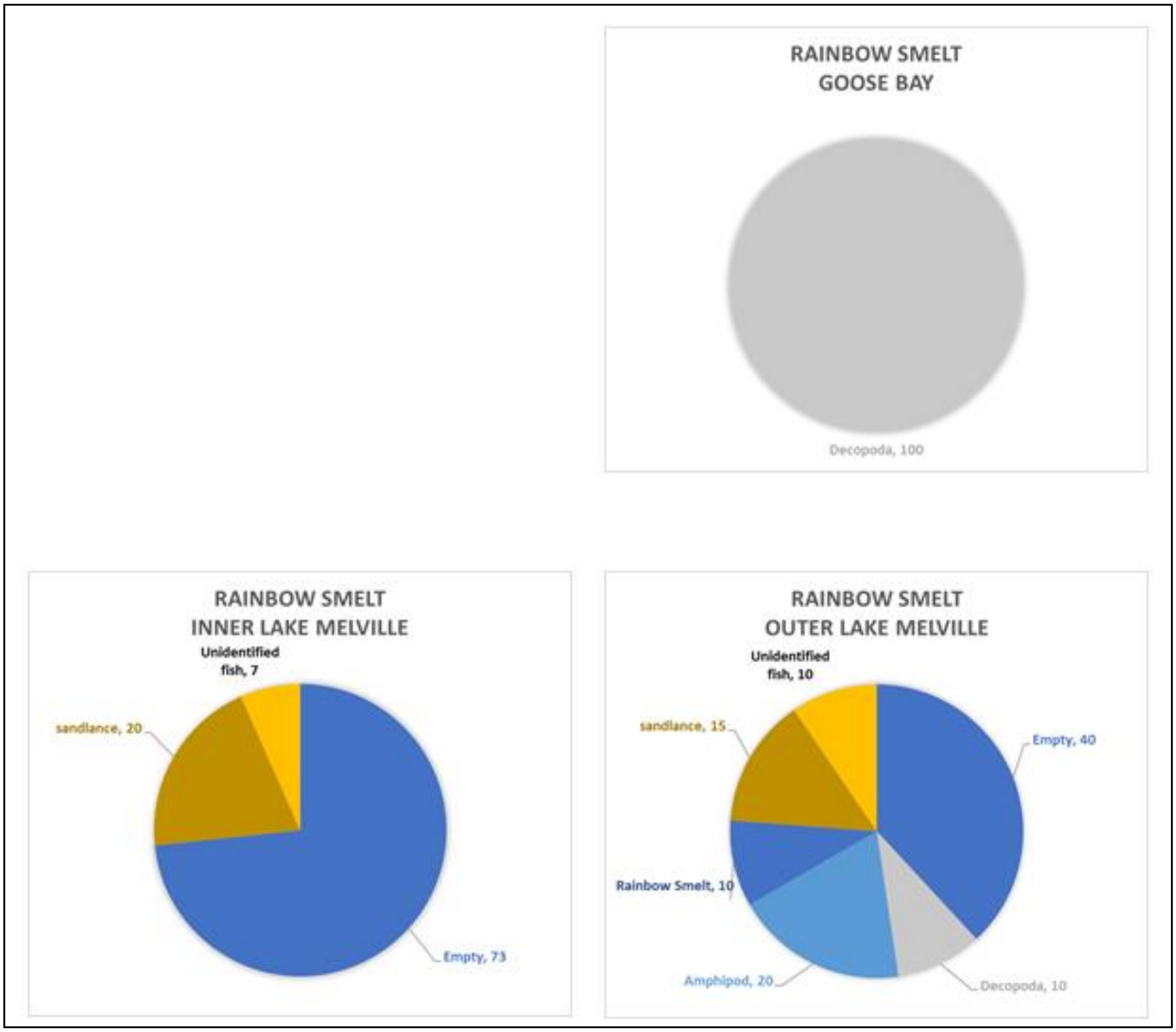


Figure 3-3: Rainbow smelt stomach content analysis. Numbers presented are the percentage of stomachs which contained that prey item.

Lake whitefish were sampled in and near the freshwater environment (Figure 3-4). Similar to brook trout, lake whitefish displayed generally greater range in tropic level ($\delta^{15}N$), indicating variation in diet (Figure 2-3).

As shown via stomach content analysis, there was a large benthic macroinvertebrate prey influence with some fish predation identified within the freshwater environment. The higher benthic invertebrate prey is also reflected in the $\delta^{15}N$ isotope signature range (see Figures 2-3 and 2-5) which places this species, as expected, lower than brook trout, rainbow smelt, tomcod, and northern pike. The species food chain length is also relatively shorter that these other species with the exception of northern pike (see Table 1-1).

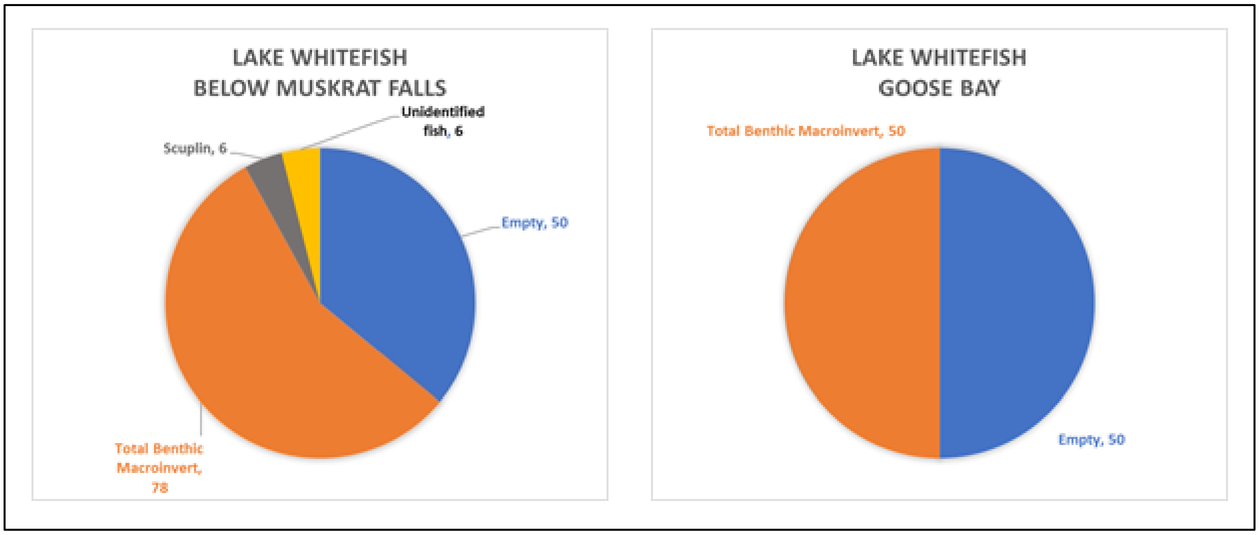


Figure 3-4: Lake whitefish stomach content analysis. Numbers presented are the percentage of stomachs which contained that prey item.

Northern pike were only sampled from the freshwater environment in 2017 (Figure 3-5). Based on isotope signatures, northern pike displayed the lowest variability of $\delta^{15}N$ isotope signature (Figure 2-3), indicating that northern pike are likely relying on other fish as a food primary source and may be keying in on specific species based on abundance or capture success.

Based on stomach content analysis, northern pike appear to be heavily reliant upon fish as a food source within the mainstem and tributaries such as white sucker, longnose sucker, and stickleback. This information tends to confirm that pike are feeding on lower trophic level fish as shown in their $\delta^{15}N$ isotope signature range as shown in Figure 2-3. While they are feeding on other fish, these prey species have relatively short food chain lengths which is reflected in the pike's lower food chain length as well (see Table 1-1).

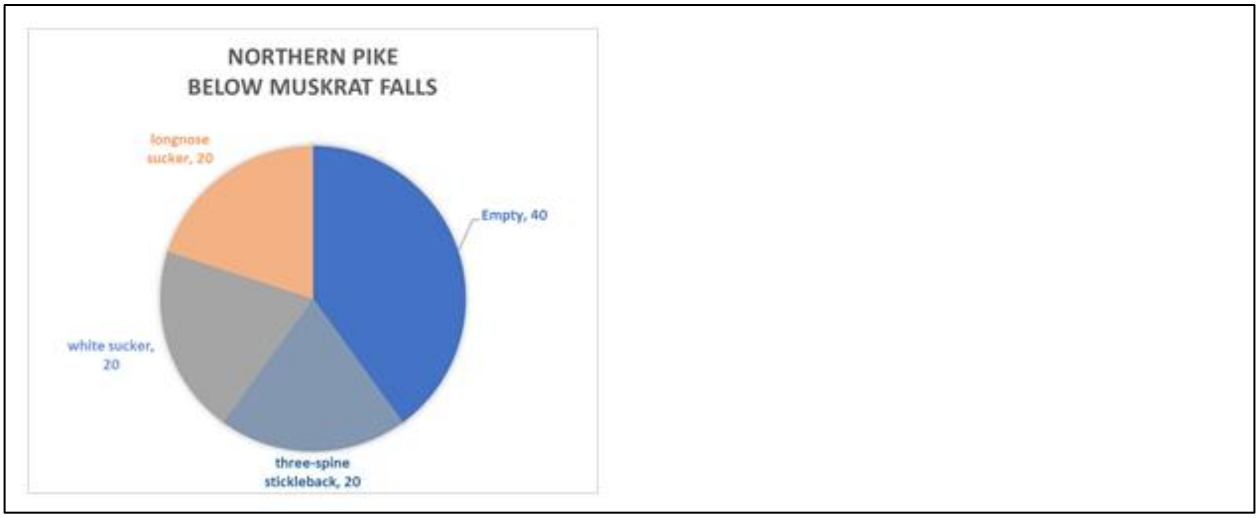


Figure 3-5: Northern pike stomach content analysis. Numbers presented are the percentage of stomachs which contained that prey item.

Longnose sucker showed the greatest range in $\delta^{13}C$ signatures, indicating that they may be feeding on a wide range of terrestrial, benthic, and pelagic carbon sources that have settled to the substrate (Figure 2-5).

Stomach content analysis from Goose Bay (the only location where stomach content analyssi has been completed) confirms that they appear to feed on benthic organisms such as filamentous algae, benthic macroinvertebrates, and bivalves (mussels and snails) (Figure 3-6). Their overall low trophic level and food chain length in all estuarine habitats seems to indicate that they feed at a similar trophic level throughout (see Table 1-1). It is notable however that the estimated trophic level of longnose sucker sampled within the freshwater environment appear to have a slightly higher trophic level and food chain length, possibly related to greater benthic macroinvertebrate diversity in the tributaries (e.g., predacious benthic inverts such as Odonata), bivalve availability, or pelagic contributions to the bottom substrate such as settling zooplankton.

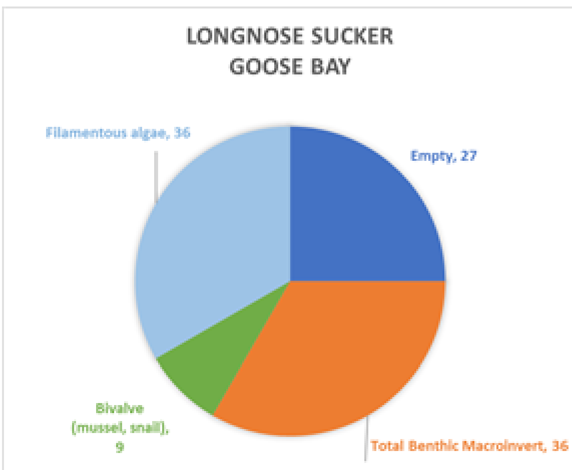


Figure 3-6: Longnose sucker stomach content analysis. Numbers presented are the percentage of stomachs which contained that prey item.

4. Recommendation on Habitat Utilization and Food web influence

The data on stable isotopes and stomach content analysis suggests that many of the fish species that utilize Lake Melville for feeding are preying on other lower trophic fish and zooplankton that are more marine origin. This would suggest that species spend greater time in the lower more-saline layer of Lake Melville to feed. This information should be considered in terms of the pathway for any predicted increase in methylmercury exposure.

Qualifications

Randy is an aquatic ecologist and mercury scientist with nearly 40 years of experience. While I have a broad background of experience in fisheries ecology, limnology, contaminated sites and risk assessment, mercury has been a common theme throughout my career. Over the last 30 years or so this has become the main focus of many of my investigations. My first job in science was in 1979 with the Department of Fisheries and Oceans at Southern Indian Lake, MB studying the effects of reservoir creation on mercury levels in fish. Later, during the 1980's and early 1990's, as a private consultant, I directed many mercury and reservoir related studies on the Nelson River MB related to large existing (Long Spruce, Limestone) and planned reservoirs. For example, I was senior scientist for all ecological and mercury-related studies during a four-year investigation on the Nelson River related to the planned Manitoba Hydro Conawapa Generating Station.

I moved to British Columbia in 1998, but continued my mercury-related investigations here, further broadening their scope. One my first tasks for BC Hydro was assembling a fish mercury database for BC reservoirs and lakes (e.g. Baker 2000) prior to conducting detailed work on methylmercury in environmental media, including fish, in Finlay Reach of Williston Reservoir in 2000/2001 (Baker et al. 2002).

I have also been senior ecologist and mercury scientist for many sophisticated mercury-related investigations related to the remediation of the Pinchi Lake Mercury Mine (on behalf of Teck Metals) between 1998 – 2012. During this time, I also pioneered a technique for non-destructive sampling of fish for methylmercury analysis (Baker et al. 2004) that has since been adopted by Environment Canada's Environmental Effects Monitoring (EEM) program for mines.

I also spent four years (2004 – 2007) working for the Global Mercury Project for the United Nations Industrial Development Organization (UNIDO) under the Global Environment Fund (GEF) as senior mercury scientist studying the effects of mercury release to watersheds as a result of small-scale artisanal gold mining activities. I also had the responsibility of working with local communities and governments to promote health communication regarding risks to exposure to mercury and methylmercury. My countries of responsibility included Indonesia and Laos, but I also participated in studies in Brazil and Venezuela. As a result of this work, I co-authored a book entitled "Protocols for Environmental and Health Assessment of Mercury Released by Artisanal and Small-Scale Gold Miners" by Veiga, M. and R. Baker (2004) Vienna, Austria. 289 p.

Between 2010 and 2013, I led all investigations related to methylmercury in environmental media related to the Site C Clean Energy Project on behalf of BC Hydro. This included collections of soil and vegetation throughout the proposed reservoir area, as well as assembling and analyzing data for mercury and methylmercury in all environmental media, including fish, in the Peace River as far downstream as Many Islands AB. I was senior author for the EIA chapter related to methylmercury, including the Canadian Reservoirs Comparison Matrix and co-authored the Human Health Risk Assessment and Wildlife Risk Assessment.

I am currently project manager and senior mercury scientist for a three-year investigation (2016 – 2018) of methylmercury in fish within the Williston and Dinosaur Reservoir watershed on behalf of the Fish and Wildlife Compensation Program (FWCP) Peace Region. This is a

program supported by Department of Fisheries and Oceans, BC Ministry of Environment and eight First Nations, including West Moberly First Nation. I am extremely familiar with the dynamics of mercury methylation in reservoirs and in particular in Williston Reservoir.

Finally, since 2017 I have been an external expert advisor to Nalcor Corporation, Newfoundland assisting in the investigation to determine the extent and magnitude of effects related to methylmercury as a result of construction and operation of the Muskrat Falls Project. In particular, my investigations have focused on methylmercury concentrations in downstream freshwater and marine biota of Lake Melville, Labrador and potential implications on human health.

Reed Harris, BSc. (Civ Eng), M. Eng., P. Eng., has over 30 years of experience in the environmental engineering field. Since 1988, Mr. Harris has specialized in the behaviour of mercury in aquatic and terrestrial ecosystems. He has developed and applied models of mercury cycling and bioaccumulation in freshwater, marine and terrestrial systems, including reservoirs. Reed was the lead modeler for two reservoir mercury projects at the Experimental Lakes Area (FLUDEX, ELARP) and managed a joint Canadian-US whole-ecosystem mercury addition experiment (METAALICUS) that examined the relationship between atmospheric mercury deposition and fish mercury concentrations. Mr. Harris has done work for the private and public sectors in the United States Canada, and abroad. He has served on expert panels on the mercury issue and has been a witness on mercury at hearings (Lower Churchill River, Penobscot Estuary, Maine, Ontario Hydro Supply-Demand hearings), published peer-reviewed articles in scientific journals, given plenary presentations at international conferences, and was a lead editor of a book outlining a national scale mercury monitoring program in the United States.

Rob Willis, B.Sc., M.E.S., EP, QPRA – Senior Toxicologist & Risk Assessor

Rob Willis is the Senior Toxicologist and Risk Assessor for Dillon Consulting Limited. He has extensive (>20 years) experience and expertise in human health and ecological (terrestrial and aquatic) risk assessment (HHERA), toxicity-based benchmarks development, the development of HHERA guidance and approaches, chemicals management and priority setting, and various aspects of applied human and environmental toxicology and environmental chemistry. He has evaluated mercury and methylmercury exposure and risk in a number of previous human health risk assessment (HHRA) studies in various regions of Canada. He is currently retained by Nalcor Energy as their HHRA subject matter expert for the Lower Churchill Hydroelectric Generation Project HHRA Program.

Rob holds a Masters of Environmental Studies degree from Dalhousie University and a Bachelor of Science degree in biology, with an emphasis on environmental toxicology, from the University of Guelph. He is recognized by the Canadian Environmental Certification Approvals Board as being a Canadian Certified Environmental Practitioner (EP) in the areas of air quality protection, and human and environmental health and safety (since 2004), and by the Ontario Ministry of Environment as a qualified person for risk assessment under O. Reg. 153/04.

Rob also frequently serves as an expert reviewer of risk assessment and toxicological documents prepared by others, is routinely invited to participate in federal and provincial risk assessment and environmental quality benchmark programs to help develop and improve guidance, protocols, methodologies and benchmarks, and serves (or has served) as an invited member on a number of provincial and regional technical committees that pertain to HHERA and risk-based environmental quality benchmarks development programs (including: Atlantic PIRI ecoRBCA Task Group (2006-ongoing), the Atlantic PIRI Chlorinated VOCs Task Group (2013-ongoing), the Nova Scotia Site Assessment and Numerical Standards Working Group (2008 to 2009), and the Ontario Ministry of the Environment and Climate Change Human Toxicological Reference Value Expert Work Group (2015-ongoing).