**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.<br>Peace River (2011) baseline benthos is 0.07 ppm THg in zooplankton and 0.016 ppm THg in benthos of which approximately 10% is MeHg | The best data sets are for PQ reservoirs; values are on a dw basis. THg in zooplankton (baseline) is 0.03 – 0.57 ppm; 0.03 – 0.51 MeHg; Post-flood range 0.45 – 0.67 THg and 0.45 – 0.82 MeHg. In benthos, baseline THg ranges from 0.28 – 0.45 ppm and 0.25 – 0.8 ppm depending on taxa; MeHg 0.2 – 0.6 and 0.02 – 0.15 ppm post-flood; In SIL post-flood zooplankton was 0.3 – 3.0 and benthos 0.1 – 3.5 depending on taxa and organism size | Peace River baseline THg and MeHg fall into lower range of zooplankton and benthos concentrations. 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:

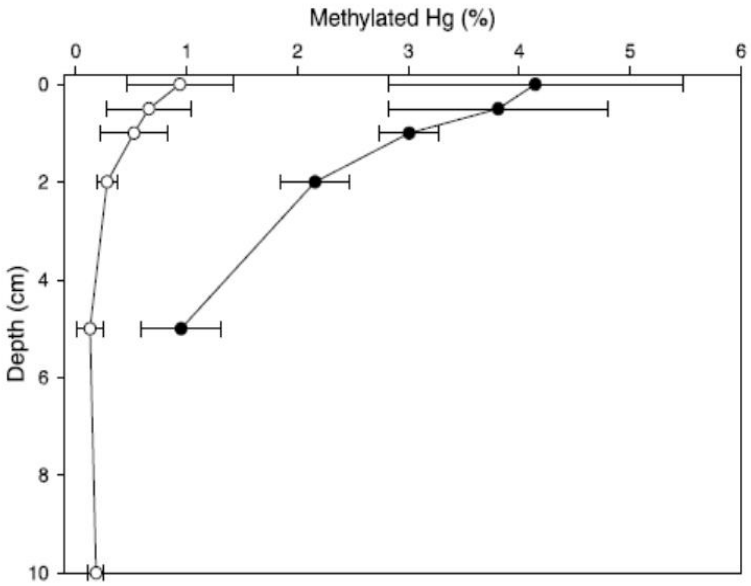
1. The depth (cm) of flooded soil where MeHg is generated and available in porewater, and fluxed to the overlying water column where it is available to aquatic biota; and
2. The proportion (%) of the pool of inorganic Hg in the depth in #1 that is converted to MeHg, and made available for uptake by the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

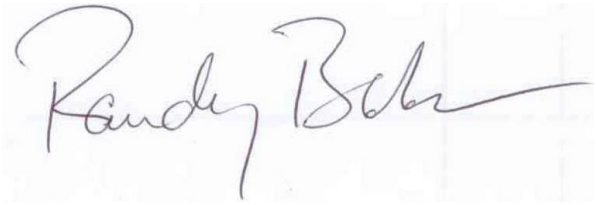
1. When comparing empirical data from MFR to many other Canadian reservoirs, using the CRCM, MFR clearly falls into the low-methylating category where a greater than 3x increase in fish mercury concentration above baseline is not expected;
2. The mass of MeHg that can be manufactured by MFR is on the 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*

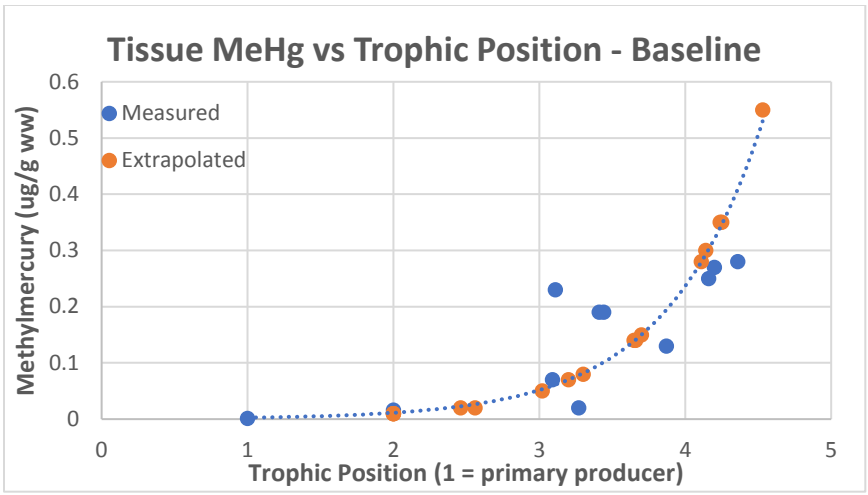
Calder et al. (2016 Supporting Document) report measured concentrations of MeHg in commonly harvested biota from Lake Melville (Table 6a, b); phytoplankton data are reported in Schartup et al. (2015 Supporting Document). Key assumptions were as follows:

- 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.



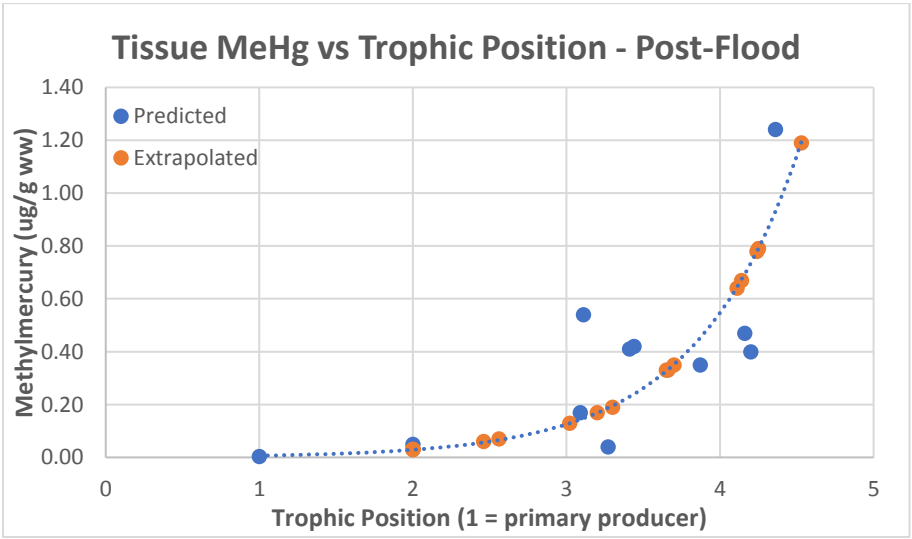

*Post-flood MeHg Concentrations in Lake Melville Biota*

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 | | | | Post-flood Predictions (Calder et al. 2016a,b) | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Biota MeHg (mg/kg) | g MeHg/ $km^2$ | MeHg (mol) | Comments | Biota MeHg (mg/kg) | g MeHg/ $km^2$ | MeHg (mol) | 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.