Temporal trends (2001–2021) of mercury in the muscle, liver and skin of beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea population in support of the Tarium Niryutait Marine Protected Area indicator evaluation

Marie Noël, Jane Kirk, Gary Stern, Tuktoyaktuk Hunters and Trappers Committee, Frank Pokiak, Nellie Pokiak, Amber Gleason, Debbie Armstrong, Enooyaq Sudlovenick, Fisheries Joint Management Committee, Lisa L. Loseto

Fisheries and Oceans Canada

Freshwater Institute

501 University Avenue

Winnipeg, MB R3T 2N6

2026

Canadian Technical Report of
Fisheries and Aquatic Sciences 3824

Fisheries and Oceans Canada | Pêches et Océans Canada

Canada

Canadian Technical Report of Fisheries and Aquatic Sciences

Technical reports contain scientific and technical information that contributes to existing knowledge but which is not normally appropriate for primary literature. Technical reports are directed primarily toward a worldwide audience and have an international distribution. No restriction is placed on subject matter and the series reflects the broad interests and policies of Fisheries and Oceans Canada, namely, fisheries and aquatic sciences.

Technical reports may be cited as full publications. The correct citation appears above the abstract of each report. Each report is abstracted in the data base *Aquatic Sciences and Fisheries Abstracts*.

Technical reports are produced regionally but are numbered nationally. Requests for individual reports will be filled by the issuing establishment listed on the front cover and title page.

Numbers 1-456 in this series were issued as Technical Reports of the Fisheries Research Board of Canada. Numbers 457-714 were issued as Department of the Environment, Fisheries and Marine Service, Research and Development Directorate Technical Reports. Numbers 715-924 were issued as Department of Fisheries and Environment, Fisheries and Marine Service Technical Reports. The current series name was changed with report number 925.

Rapport technique canadien des sciences halieutiques et aquatiques

Les rapports techniques contiennent des renseignements scientifiques et techniques qui constituent une contribution aux connaissances actuelles, mais qui ne sont pas normalement appropriés pour la publication dans un journal scientifique. Les rapports techniques sont destinés essentiellement à un public international et ils sont distribués à cet échelon. II n'y a aucune restriction quant au sujet; de fait, la série reflète la vaste gamme des intérêts et des politiques de Pêches et Océans Canada, c'est-à-dire les sciences halieutiques et aquatiques.

Les rapports techniques peuvent être cités comme des publications à part entière. Le titre exact figure au-dessus du résumé de chaque rapport. Les rapports techniques sont résumés dans la base de données *Résumés des sciences aquatiques et halieutiques.*

Les rapports techniques sont produits à l'échelon régional, mais numérotés à l'échelon national. Les demandes de rapports seront satisfaites par l'établissement auteur dont le nom figure sur la couverture et la page du titre.

Les numéros 1 à 456 de cette série ont été publiés à titre de Rapports techniques de l'Office des recherches sur les pêcheries du Canada. Les numéros 457 à 714 sont parus à titre de Rapports techniques de la Direction générale de la recherche et du développement, Service des pêches et de la mer, ministère de l'Environnement. Les numéros 715 à 924 ont été publiés à titre de Rapports techniques du Service des pêches et de la mer, ministère des Pêches et de l'Environnement. Le nom actuel de la série a été établi lors de la parution du numéro 925.

Canadian Technical Report
of Fisheries and Aquatic Sciences 3824

2026

Temporal trends (2001–2021) of mercury in the muscle, liver and skin of beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea population in support of the Tarium Niryutait Marine Protected Area indicator evaluation

by

Marie Noël[2], Jane Kirk[3], Gary Stern[4], Tuktoyaktuk Hunters and Trappers[5] Committee, Frank Pokiak[5], Nellie Pokiak[5], Amber Gleason[3], Debbie Armstrong[4], Enooyaq Sudlovenick[6], Fisheries Joint Management Committee[7], Lisa L. Loseto[8]

[1]Fisheries and Oceans Canada
Freshwater Institute
501 University Crescent
Winnipeg, MB, R3T 2N6

[2]Fisheries and Oceans Canada
Pacific Science Enterprise Centre
4160 Marine Drive
West Vancouver, BC, V7V 1N6

[3]Environment and Climate Change Canada
867 Lakeshore Road
Burlington, ON, L7S 1A1

[4]Centre for Earth Observation Science
University of Manitoba
537A Wallace Building
Winnipeg, MB, R3T 2N2

[5]Tuktoyaktuk Hunters and Trappers Committee
P.O. Box 286
Tuktoyaktuk, NT, X0E 1C0

[6]Fisheries and Oceans Canada
Government of Canada
630 Mivvik Street
Iqaluit, NU, X0A 0A1

[7]Fisheries Joint Management Committee
76 Mackenzie Road, P.O. Box 2120
Inuvik, NT, X0E 0T0

[8]Fisheries and Oceans Canada
Institute of Ocean Sciences
9860 West Saanich Road
Sidney, BC, V8L 4B2

© His Majesty the King in Right of Canada, as represented by the Minister of the Department of Fisheries and Oceans, 2026

This work is licensed under the Open Government Licence

Cat. No. Fs97-6/3824E-PDF ISBN: 978-1-100-01147-9 ISSN 1488-5379

https://doi.org/10.60825/x3tx-0707

Correct citation for this publication:

Noël, M., Kirk, J., Stern, G., Tuktoyaktuk Hunters and Trappers Committee, Pokiak, F., Pokiak, N., Gleason, A., Armstrong, D., Sudlovenick, E., Fisheries Joint Management Committee, Loseto, L.L. 2026. Temporal trends (2001–2021) of mercury in the muscle, liver and skin of beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea population in support of the Tarium Niryutait Marine Protected Area indicator evaluation. Can. Tech. Rep. Fish. Aquat. Sci. 3824: vii + 34 p. https://doi.org/10.60825/x3tx-0707

TABLE OF CONTENTS

LIST OF TABLES IV

LIST OF FIGURES V

ABSTRACT VI

RÉSUMÉ VII

PREFACE 1

INTRODUCTION 2

MATERIALS AND METHODS 3

- SAMPLE COLLECTION 3
- MERCURY ANALYSIS 4
- DATA ANALYSIS 5
  - Accounting for age and size as confounding factors of temporal Hg concentrations 5
  - Risk for Hg-mediated health effects 6
  - Statistical analyses 7

RESULTS AND DISCUSSION 7

- DATASETS AND BIOLOGICAL VARIABLES 7
  - Muscle Tissue Data 7
  - Skin Tissue Data 9
  - Liver Tissue Data 11
- TEMPORAL TRENDS 13
  - Muscle Total Mercury (THg) 13
  - Skin Total Mercury (THg) 15
  - Liver Total Mercury (THg) 16
- RISK FOR MERCURY-MEDIATED HEALTH EFFECTS 17

CONCLUSION 18

CREDIT AUTHORSHIP CONTRIBUTION STATEMENT 19

ACKNOWLEDGEMENTS 19

REFERENCES 20

APPENDICES 25

LIST OF TABLES

Table 1. Length and age (mean ± standard deviation) for male beluga whales sampled at each whaling camp between 2001 and 2021 and analyzed for Hg in muscle. Belugas were grouped based on their length into medium (>= 380 cm and <= 420 cm) and large (> 420 cm). Small whales (< 380cm) were not included in the assessment. * Indicates that age was not available for one individual. 8

Table 2. Length and age (mean ± standard deviation) for male beluga whales sampled at each whaling camp between 2001 and 2021 and analyzed for Hg in skin. Belugas were grouped based on their length into medium (>= 380 cm and <= 420 cm) and large (> 420 cm). Small whales (< 380cm) were not included in the assessment. * Indicates that age was not available for one individual. 10

Table 3. Length and age (mean ± standard deviation) for male beluga whales sampled at each whaling camp between 2001 and 2021 and analyzed for Hg in liver. Belugas were grouped based on their age into young (> 15 years and <= 35 years) and old (> 35 years). Immature (<= 15 years) whales were not included in this assessment..... 12

LIST OF FIGURES

Figure 1. Beluga samples were collected during the annual beluga harvest between 2001 and 2021 at four sampling camps within the TN MPA. 4

Figure 2. Total mercury concentrations (log-transformed) in the muscle of individual EBS male beluga whales sampled between 2001 and 2021. Whales were grouped by size into medium (>= 380 cm and <= 420 cm) and large (> 420 cm)..... 15

Figure 3. Total mercury concentrations (log-transformed) in the skin of individual EBS male beluga whales sampled between 2001 and 2021. Whales were grouped into medium (>= 380 cm and <= 420 cm) and large (> 420 cm)..... 16

Figure 4. Total mercury concentrations (log-transformed) in the liver of individual EBS male beluga whales sampled between 2001 and 2021. Whales were grouped by age into young (> 15 years and <= 35 years) and old (> 35 years). 17

Figure 5. Percentage of sampled beluga whales at risk of deleterious health effects related to THg levels measured in liver. 18

ABSTRACT

Noël, M., Kirk, J., Stern, G., Tuktoyaktuk Hunters and Trappers Committee, Pokiak, F., Pokiak, N., Gleason, A., Armstrong, D., Sudlovenick, E., Fisheries Joint Management Committee, Loseto, L.L. 2026. Temporal trends (2001–2021) of mercury in the muscle, liver and skin of beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea population in support of the Tarium Niryutait Marine Protected Area indicator evaluation. Can. Tech. Rep. Fish. Aquat. Sci. 3824: vii + 34 p. https://doi.org/10.60825/x3tx-0707

The Eastern Beaufort Sea (EBS) beluga population, one of the largest beluga populations in Canada, is under increasing anthropogenic pressures including climate-related ecosystem changes and exposure to contaminants. The Tarium Niryutait Marine Protected Area (TN MPA), located in the Mackenzie Estuary where belugas congregate every summer, was designated in 2010 with the objective to "*conserve and protect beluga whales and other marine species, their habitats and their supporting ecosystem*". Mercury, a contaminant of high concern in the Arctic, is accumulated through diet in beluga whales and measuring its level in beluga muscle was selected as one of the key ecological indicators to monitor the effectiveness of the MPA. In the present report, we present levels of THg in muscle (n = 486), skin (n = 478) and liver (n = 474) samples collected from EBS belugas between 2001 and 2021. THg levels in muscle and skin had similar non-linear trends over time. In both medium (380–420 cm in length) and large (> 420 cm) whales, THg levels decreased in the early 2000's for eight to nine consecutive years before increasing until 2013/2014 and declining again until 2017/2018. After that, while THg levels remained stable in medium size whales, they increased in large whales, likely reflecting different changes in the food webs of these two groups of whales as they feed in different habitats. In liver, there was no trend in THg levels measured in young (16–35 years) whales while those measured in older (> 36 years) whales decreased linearly over the study period at a rate of 1.5% per year which may reflect a shift in diet and/or a decrease in the levels of THg in prey. Using THg levels in liver and established thresholds for Hg-mediated health effects, we concluded that the EBS population is currently at low risk of health effects from Hg exposure but more research is needed to develop species-specific risk thresholds. Our results support the use of THg in muscle as an indicator for the TN MPA that can provide insights into diet shifts and/or alterations of Hg transport and fate in the ecosystem but we also recommend the evaluation of THg levels measured in liver against known toxicity thresholds to assess beluga health as it relates to Hg exposure.

RÉSUMÉ

Noël, M., Kirk, J., Stern, G., Tuktoyaktuk Hunters and Trappers Committee, Pokiak, F., Pokiak, N., Gleason, A., Armstrong, D., Sudlovenick, E., Fisheries Joint Management Committee, Loseto, L.L. 2026. Temporal trends (2001–2021) of mercury in the muscle, liver and skin of beluga whales (Delphinapterus leucas) from the Eastern Beaufort Sea population in support of the Tarium Niryutait Marine Protected Area indicator evaluation. Can. Tech. Rep. Fish. Aquat. Sci. 3824: vii + 34 p. https://doi.org/10.60825/x3tx-0707

La population de bélugas de l'est de la mer de Beaufort (EMB), l'une des plus grandes populations de bélugas au Canada, est soumise à des pressions anthropiques croissantes, notamment des changements écosystémiques liés au climat et une exposition aux contaminants. La zone de protection marine Tarium Niryutait (ZPM TN), située dans l'estuaire du Mackenzie où les bélugas se rassemblent chaque été, a été désignée en 2010 dans le but de « conserver et protéger les bélugas et d'autres espèces marines, leurs habitats et leur écosystème de soutien ». Le mercure, un contaminant très préoccupant dans l'Arctique, s'accumule dans l'alimentation des bélugas et la mesure de sa concentration dans le muscle des bélugas a été sélectionnée comme l'un des indicateurs écologiques clés pour surveiller l'efficacité de la ZPM. Dans ce rapport, nous présentons les niveaux de Hg dans les échantillons musculaires (n = 486), cutanés (n = 478) et hépatiques (n = 474) prélevés sur les bélugas de l'EMB entre 2001 et 2021. Les niveaux de Hg dans le muscle et la peau avaient des tendances non-linéaires similaires dans le temps. Pour les baleines de taille moyenne (380 à 420 cm de longueur) et de grande taille (> 420 cm), les niveaux de Hg ont diminué au début des années 2000 pendant huit à neuf années consécutives avant d'augmenter jusqu'en 2013-2014 et de diminuer à nouveau jusqu'en 2017-2018. Par la suite, alors que les niveaux de Hg sont restés stables chez les baleines de taille moyenne, ils ont augmenté chez les baleines de grande taille, reflétant probablement des régimes alimentaires différents entre ces deux groupes de baleines se nourrissant dans des habitats différents. Dans le foie, il n'y avait aucune tendance dans les niveaux de Hg mesurés chez les jeunes baleines (16 à 35 ans), tandis que ceux mesurés chez les baleines plus âgées (> 36 ans) ont diminué de manière linéaire au cours de la période d'étude à un taux de 1,5 % par an, reflétant probablement un changement du régime alimentaire et/ou une diminution des niveaux de Hg dans les proies. En utilisant les niveaux de Hg dans le foie et les seuils établis pour les effets sur la santé liés au Hg, nous avons conclu que la population de l'EMB présente actuellement un faible risque d'effets sur la santé liés à l'exposition au Hg, mais que des recherches supplémentaires sont nécessaires pour développer des seuils de risque spécifiques à l'espèce. Nos résultats soutiennent l'utilisation du Hg dans le muscle comme indicateur écologique pour le suivi de la ZPM TN qui peut fournir des informations sur les changements de régime alimentaire et/ou les altérations du transport et du devenir du Hg dans l'écosystème, mais nous recommandons également l'évaluation des niveaux de Hg mesurés dans le foie par rapport aux seuils de toxicité connues pour évaluer la santé des bélugas en ce qui concerne l'exposition au Hg.

PREFACE

It is important to recognize the data presented in this study resulted from the generosity of hunters who kindly shared their food to be part of this research. Inuvialuit lead subsistence lifestyles as their ancestors did, and beluga, qilalugak, remain the most revered socio-cultural species (FJMC, 2024). The well-being and health of beluga whales are connected to the health and well-being of Inuvialuit, thus Inuvialuit concerns of contaminants in belugas and supporting ecosystem drives research and contaminants monitoring in the Inuvialuit Settlement Region (ISR).

Our study presents findings of polychlorinated biphenyls (PCBs) measured in beluga blubber for the purpose of tracking temporal trends and examining PCB concentrations in relation to thresholds of physiological impacts to belugas. The temporal trends inform management and policy on the fate and transport of PCBs to Arctic belugas. Currently, there are no consumption advisories in place with regards to consuming beluga whales, and the most recent human health assessment (2007-2008) concluded 'the benefits of eating country foods outweigh the risks from contaminant exposure'. This current report is not intended to supplement or provide guidance for food safety consumption, however these data have been made available for an updated evaluation of risks and benefits of consuming country food.

INTRODUCTION

The Eastern Beaufort Sea (EBS) beluga whale (*Delphinapterus leucas*) population is recognized as one of the largest in Canada, estimated at approximately 32,500 individuals (Harwood et al., 1996; Mayette et al., 2023). Every spring, the EBS beluga migrate from the Bering Sea to the Beaufort Sea where they spend their summer with high site fidelity to the Mackenzie Estuary where large aggregations of nearly 2,000 individuals have been recorded (Harwood et al., 1996, Harwood et al., 2014). It is in these waters where the Inuvialuit have harvested beluga for subsistence for hundreds of years (Alunik et al., 2003; McGhee 1988). Beluga continue to be part of a traditional way of life and provide food and cultural well-being (Usher 2002; Waugh et al., 2018). As part of Inuvialuit being stewards of their resources, they, together with Fisheries and Oceans Canada (DFO) used a Marine Protected Areas (MPAs) area-based management tool to conserve and manage beluga as an important resource. The Tarium Niryutait Marine Protected Area (TN MPA), located in the Mackenzie Estuary was designated in 2010 with the objective to "conserve and protect beluga whales and other marine species, their habitats and their supporting ecosystem" (Gazette, 2010). It is made up of three separate and distinct sub-areas within the Mackenzie River Estuary where belugas aggregate each summer and subsistence harvesting occurs: Niaqunnaq (Shallow Bay), Okeevik (in Beluga Bay, east Mackenzie Bay near Kendall and Pelly Islands) and Kittigaryuit (Kugmallit Bay). To assess the effectiveness of regulations and other management activities in meeting the MPA conservation objectives, a Canadian Science Advisory Secretariat (CSAS) evaluation is set to examine all indicators identified (Fisheries Joint Management Committee (FJMC), 2013, Brewster et al., 2019).

Indicators are variables that are measured or observed repeatedly and consistently over time from which to draw conclusions on existing conditions and measure changes or trends over time (DFO, 2013; FJMC, 2013). They can be used to describe the impacts of various stressors to the state of the ecosystem or to compare actual conditions within the TNMPA with targets for desired ecological conditions. Mercury (Hg), in its methylated, organic form (MeHg), biomagnifies in food webs and accumulates over time in organisms (Wu and Zhang, 2023). This means that levels reported in high-trophic level animals such as belugas provide an integrated signal of Hg levels in the ecosystem which can help provide insights into food web shifts and/or alterations of Hg transport and fate in the ecosystem due to climate change and/or other anthropogenic stressors (DFO, 2013). In this context, the monitoring of Hg in the muscle of EBS belugas has been identified as an important ecological indicator to evaluate one component of beluga health and the supporting food web.

While wildlife of the remote Arctic are generally exposed to moderate levels of contaminants compared to their southern counterparts, there are an increasing number of studies reporting contaminant-associated health effects in Arctic marine mammals, including beluga whales (Letcher et al., 2010; Desforges et al., 2013; Brown et al., 2014; Noël et al., 2014; Krey et al., 2015; Ostertag et al., 2014; Scheuhammer et al., 2015; Dietz et al., 2019). MeHg is efficiently absorbed through the diet and biomagnifies through the food chain and can therefore reach high concentrations in species higher up in the food chain, potentially leading to deleterious health effects. While Hg is well known for its neurotoxicity (Basu et al, 2009; Ostertag et al., 2014; Krey et al., 2015), it has also been associated with endocrine and immune disruption in marine mammals (DeGuise et al., 1996; Lalancette et al., 2003; Dufresne et al., 2010; Frouin et al., 2010, 2012) as well as liver abnormalities (Rawson et al., 1993; Sonne et al., 2007).

With the Arctic warming at an alarming rate and the Beaufort Sea experiencing some of the fastest rates of sea ice cover change in the entire Arctic (Environment and Climate Change Canada (ECCC), 2021), the associated ecosystem changes have a significant impact on Arctic

marine species such as beluga whales. For example, climate-induced impacts such as decreases in Arctic sea ice cover, changes in temperature profiles, food web productivity, and associated changes in beluga distribution and feeding ecology may profoundly change the course of contaminant fate and transport in the Arctic environment (Loseto et al., 2015; Desforges et al., 2021). In addition, climate-related changes may affect the condition and health of beluga whales by reducing the abundance of their preferred prey (Loseto et al., 2018), impacting nutrition and/or increasing energetic expenditures to search for prey potentially leading to additional stress and decrease ability to cope with contaminant-associated health effects. Finally, the increased accessibility to the Arctic and its associated human activities such as shipping and resource exploration are adding pressure to the already fragile Arctic ecosystems (Moore et al., 2012; Reeves et al., 2014).

Here, we report on total mercury (THg) levels and trends measured in beluga whale muscle, liver and skin collected between 2001 and 2021 in collaboration with Inuvialuit hunters harvesting whales within the TN MPA, in the Inuvialuit Settlement Region (ISR) in Canada's Northwest Territories. While THg in muscle was selected as ecological indicator for the TN MPA, investigating levels in liver and skin are also important given that liver is a target tissue for Hg toxicity and skin, along with muscle, is an important part of the local traditional diet. The two objectives of this report are to 1) expand on the previous temporal assessments (1981 – 2012; Loseto et al. (2015); 1981 – 2017; Morris et al., 2022) and 2) evaluate Hg-associated risk to beluga health over time as part of the TN MPA indicator assessment evaluation.

MATERIALS AND METHODS

SAMPLE COLLECTION

In collaboration with the annual beluga harvest by Inuvialuit hunters from Aklavik, Inuvik and Tuktoyaktuk, the FJMC, together with local Hunters and Trappers Committees and DFO have been collecting tissue samples since the 1980s to monitor beluga contaminant levels (Lockhart et al., 2005; Stern et al., 2005; Noël et al., 2018; Loseto et al., 2015). While the majority of samples collected were from Hendrickson Island, a site where harvesters from the community of Tuktoyaktuk flense their whales prior to taking home for food preparation, other whaling locations also collected samples for analyses (see below for details). The liver, muscle and skin were collected by monitors, local hunters trained in scientific collection of the samples, on site and stored in a portable -20°C freezer before shipment to Fisheries and Oceans Canada in Winnipeg where they were stored until analysis. Length and sex were recorded on site for each individual. From the start of the study until 2015, the ages of whales were determined using teeth (typically one tooth per individual, pulled from the right lower mandible, either from position 2 or 5 or the tooth that is least eroded or miss-shaped) and counting individual growth layer groups in the dentine (Stewart et al., 2006); from 2016 onwards, aspartic acid racemization measured in the nucleus of the eye balls were used for aging following methods from Pleskach et al., (2016).

For the purpose of this report, we focused on samples collected at Hendrickson Island between 2001 and 2021. Hendrickson Island is one of the Northern Contaminants Program (NCP) core long term monitoring locations whereby funds are directed to support annual analyses of tissue samples collected. However, on some occasions samples were also collected at nearby camps, namely East Whitefish, Kendall Island, since 2010 (and in 2005 at Kendall Island), and a few samples from Shingle Point (Aklavik), were included in this report as these sites were different locations where hunters bring the belugas for processing and do not imply different beluga populations.

Few females were collected as hunters generally select medium to larger sized males (Harwood et al., 2015). For these reasons, and to avoid additional confounding factors that come with examining THg levels in females, we focused the current data set on male beluga whales. The final datasets comprised a total of 518 muscle, 504 liver and 487 skin samples analyzed for THg in male beluga whales sampled at the four camps between 2001 and 2021.

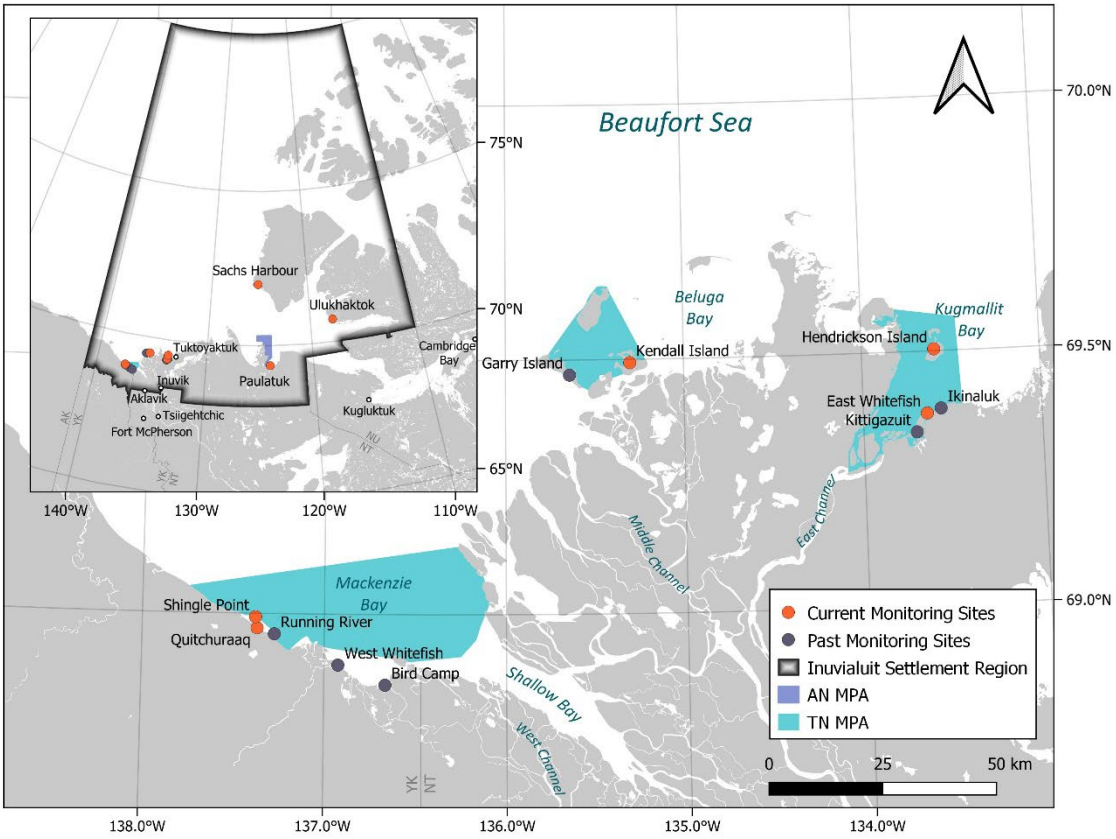


Figure 1. Beluga samples were collected during the annual beluga harvest between 2001 and 2021 at four sampling camps within the TN MPA.

MERCURY ANALYSIS

Beluga muscle, liver and skin were analyzed for total mercury (THg). Subsamples (0.15 g) were taken from partially thawed tissues after removing the outer surface of the sample. Hg levels were expressed in μg/g wet weight (ww).

Samples collected and analyzed from 1984 to 2014 were processed at the FWI using a digest of hydrochloric/nitric acid mixture (Aqua Regia) heated to 90 °C. The digest was analyzed for THg by Cold Vapour Atomic Absorption spectroscopy (CVAAS) (Armstrong and Uthe, 1971). The detection limit was 0.005 μg/g. Certified standard reference materials (CRM 2976, TORT-2, DOLT-2) were analyzed in duplicate in every run. Recovery within 10% of the certified values was used as a batch validation for samples. Duplicates of tissue samples were taken every ten samples with an average difference of ~5%.

From 2015 onwards, THg was analyzed at the Centre for Earth Observation Science (CEOS) at the University of Manitoba as well as the Mercury Analytical Laboratory at the Canada Centre for Inland Waters in Burlington. Both laboratories measured Hg by Combustion Atomic

Absorption Spectroscopy (C-AAS) on a Teledyne Leeman HYDRA IIc Direct Mercury Analyzer (DMA) by following US Environmental Protection Agency (EPA) Method 7473 (US EPA 2007b)). Samples are directly combusted in an oxygen-fed oven, followed by gold trap amalgamation and detection by AAS. Detection limits are 0.04 ng Hg. QA/QC is accomplished using certified reference materials (CRM) from the National Research Council (NRC) Canada, including dogfish muscle (DORM-3, DORM-4), dogfish liver (DOLT-3, DOLT-4, DOLT-5) and lobster hepatopancreas (TORT-2, TORT-3), and Sediment (NIST 2709; MESS-4; PACS-3). Recoveries are 90-110% of established values, and precision is better than 8% Relative Standard Deviation (RSD) for each of the CRM. The laboratory participated successfully in the annual Northern Contaminants Program interlaboratory comparison, which includes Hg in biological tissues.

DATA ANALYSIS

Accounting for age and size as confounding factors of temporal Hg concentrations

Previous studies have shown that THg levels were influenced by both age (liver) and size (muscle and skin) in belugas and that the relationships were tissue specific due to the different forms of Hg found in different tissues (Loseto et al., 2008a, 2008b; Wagemann et al., 1998, 2005; Lockhart et al., 2005). In addition, the age to size relationship in belugas did not follow a linear growth curve, but rather a Gompertz growth curve whereby an asymptote was reached at maturity (Harwood et al., 2002). It was therefore important to consider the appropriate tissue-specific confounding factors when conducting temporal trends.

Size factor and muscle and skin Hg

Mercury in the muscle is largely in its organic form, methylmercury (MeHg), with a half-life of several months in mammals (Petersson et al., 1991). Previous studies on the EBS beluga population showed that the size of a beluga defines the habitat it selects and the prey it feeds on (Loseto et al., 2006; 2009). Size (length) was therefore a key confounding factor to take into account when investigating THg in muscle. Beluga size groups have been defined and used previously and the same groupings was applied here: small (under 380 cm), medium (380–420 cm) and large (over 420 cm) (Loseto et al., 2006; 2008a, b; 2009; 2015). Of note, small belugas were excluded due to the lack of representation within the dataset for all years.

Similar to muscle, the majority of Hg found in skin is in the form of MeHg (Wagemann, 1998). In the present study, THg levels in skin and muscle were indeed significantly correlated ($r = 0.58$ $p < 0.05$; Appendix 1). Beluga size groups were therefore also used in the investigation of THg temporal trends in skin.

Age factor and liver Hg

A large proportion of Hg in the liver is in its inorganic form (88 – 94%), bound to selenium, and has been shown to accumulate with age (Lockhart et al., 2005; Wagemann et al., 1998). Here, there was a significant linear relationship between THg in liver and age with all years and sampling camps combined ($r = 0.18$; $p < 0.05$; not shown). Two approaches have been proposed in the past to ensure age would not impact the evaluation of Hg levels in liver: 1) adjust Hg levels in liver to a specific age (Lockhart et al., 2005) or 2) group beluga by age to capture the range or variability of Hg for each age group (Loseto et al., 2015).

Correcting to a specific age assumes that Hg accumulation with age was consistent and comparable across years. In the present study, there were significant relationships between THg in liver and age for each year of sampling (r^2 ranging from 0.32 to 0.78) except in 2007, 2008 and 2014 (Appendix 2). The slopes for belugas sampled in 2013 and 2019 were

significantly different from those sampled in 2011 and 2020 (Analysis of Covariance (ANCOVA); $p < 0.05$). These results suggested that correcting to a specific age was maybe not the best approach for the current dataset.

We evaluated THg levels in belugas grouped into five-year age interval as described in Loseto et al. (2015). To reduce variability related to sexual maturity, all belugas under the age of 15 (immature) were excluded from the analysis. The results with the present dataset were similar to those in Loseto et al. (2015) so that all age groups under the age of 35 had similar THg levels in liver ($p > 0.05$) and these levels were significantly lower than those reported in all age groups over the age of 35 ($p > 0.05$). We therefore used the same approach as in Loseto et al. (2015) and temporal trend analyses of THg in liver were conducted on two groups of belugas: young (16–35 years) and old whales (over 36 years).

Risk for Hg-mediated health effects

Establishing health thresholds for wildlife species is challenging. Since liver is one of the most widely used tissue for Hg and metal monitoring and is a target tissue for Hg toxicity, there have been attempts to establish Hg toxicity thresholds based on the lowest concentrations associated with significant effects (Dietz et al., 2019). However, such thresholds are highly dependent on the proportion of MeHg, the most toxic form of Hg, which is highly variable and age dependent (Appendix 2; Wagemann et al., 2000). For terrestrial mammals, 25–30 μg/g ww is usually used as a threshold for liver effects related to THg (Thompson, 1996, more recently updated by Shore et al., 2011). For marine mammals, Rawson et al. (1993) found liver damage in bottlenose dolphins (*Tursiops truncatus*) with Hg concentrations of 61 μg/g ww. For polar bears (*Ursus maritimus*), a liver THg concentration of 11 μg/g ww was suggested as a threshold for liver effects although higher liver levels have been reported in apparently healthy animals, which is likely related to the fact that a large portion of Hg in the liver is often found associated with selenium, especially at higher concentrations and this Hg-Se complex is considered to be relatively non-toxic (Sonne, 2010; Dietz et al., 2013).

In recent years, risk categories have been used based on THg concentrations in liver to evaluate risks to Arctic wildlife associated with Hg exposure (Dietz et al., 2019; Arctic Monitoring and Assessment Programme (AMAP), 2021, Dietz et al., 2022) and were used in the present assessment. These categories reflect effects on reproduction, condition and behavior observed in harp seals (*Pagophilus groenlandicus*) exposed to daily dose of MeHg and were as follows (Ronald et al., 1977):

- No risk: < 16 μg/g ww;
- Low Risk: 16–64 μg/g ww;
- Moderate Risk: 64–83 μg/g ww;
- High Risk: 83–126 μg/g ww;
- Severe Risk: > 126 μg/g ww.

In their study, Ronald et al. (1977) reported potential for organ lesions (kidney, liver), anorexia and reduced growth in harp seals in the High Risk Category and severe impacts on organ function such as kidney failure, weight loss and ultimately increased mortality in harp seals in the Severe Risk Category.

It is important to note that, while outliers were removed for the temporal trend assessment (see below), the entire liver dataset was used for the health risk assessment in order to provide a

better representation of the population. We will present number of individuals in each risk category in Appendix.

Statistical analyses

Data for small belugas (muscle analysis) and immature belugas (liver analysis) were not included due to limited data for these categories. THg concentration outliers were determined in all three tissue and within each size or age group separately using the interquartile range method and removed from the data set.

All statistical analyses were performed in R (R Core Team, 2020). Normality and homogeneity of variances were tested for biological variables and Hg data using the Kolmogorov-Smirnov and Levene's test, respectively. If the assumptions of normality and homogeneity of variances were not met, data were log transformed before further statistical analyses. Analysis of variance were used to evaluate differences amongst sampling camps and amongst years. Mercury results are presented as mean ± standard error.

Temporal trends in THg concentrations in muscle, liver and skin were assessed using a method used by AMAP (AMAP, 2020) and recently by Morris et al., 2022. Briefly, temporal trends of THg were analyzed based on log-transformed data with modelling of individual concentration data rather than annual medians. Linear and non-linear models were explored, and best-fit models were selected using the Akaike Information Criterion corrected for small sample size (AIC_c) (Burnham and Anderson, 2002).

Annual percent (%) change in concentration over the entire study period (2001–2021) was only calculated for significant linear trends where the slope of the log-linear trend was used to calculate the annual change (Morris et al., 2022).

RESULTS AND DISCUSSION

DATASETS AND BIOLOGICAL VARIABLES

Muscle Tissue Data

After removing 32 outliers (15 medium and 17 large), the muscle dataset was comprised of 486 male beluga whales sampled between 2001 and 2021 at four different sampling camps (Table 1). Given that length is a key biological factor affecting Hg levels in muscle, we investigated variations in length amongst sampling camps and sampling years to evaluate length as a potential confounding factor in the temporal trend analyses. Within each size group, and for each year considered separately, there were no significant differences in length amongst sampling camps ($p > 0.05$).

When all sampling camps were grouped together, there were no inter-annual differences in length for medium ($p = 0.059$) and large belugas ($p = 0.204$) suggesting that length was not a confounding factor in Hg temporal trend analyses in muscle.

*Table 1. Length and age (mean ± standard deviation) for male beluga whales sampled at each whaling camp between 2001 and 2021 and analyzed for Hg in muscle. Belugas were grouped based on their length into medium (>= 380 cm and <= 420 cm) and large (> 420 cm). Small whales (< 380cm) were not included in the assessment. * Indicates that age was not available for one individual.*

Year	Location	Medium			Large		
		n	Length (cm)	Age (Years)	n	Length (cm)	Age (Years)
2001	Hendrickson Is	14	404 ± 3.4	28 ± 2	3	437 ± 8	41 ± 5
2002	Hendrickson Is	11	400 ± 3.8	24 ± 1	10	436 ± 3	35 ± 2
2003	Hendrickson Is	10	405 ± 3.3	33 ± 4	7	431 ± 4	31 ± 3
2004	Hendrickson Is	8	411 ± 2.2	30 ± 3	10	433 ± 4	26 ± 2
2005	Hendrickson Is	10	398 ± 3.7	24 ± 2	7	438 ± 5	32 ± 4
	Kendall Is	2	399 ± 2.6	17 ± 0	5	461 ± 11	28 ± 1
2006	Hendrickson Is	15	401 ± 2.7	22 ± 1	6	430 ± 2	28 ± 3
2007	Hendrickson Is	8	404 ± 3.7	24 ± 2	8	443 ± 8	26 ± 2
2008	Hendrickson Is	10	406 ± 3.5	34 ± 4	11	439 ± 5	31 ± 2
2009	Hendrickson Is	8	403 ± 4.2	31 ± 6	8	430 ± 3	28 ± 1
2010	Hendrickson Is	8	401 ± 3.3	22 ± 2	8	447 ± 10	30 ± 3
2011	Hendrickson Is	9	395 ± 2.7	23 ± 1	6	427 ± 2	29 ± 2
	East Whitefish				6	436 ± 5	27 ± 3
	Kendall Is	5	400 ± 3.7	24 ± 1	3	428 ± 5	25 ± 0
2012	Hendrickson Is	16	401 ± 2.7	24 ± 2	6	430 ± 4	30 ± 3
	Kendall Is	5	398 ± 6.0	19 ± 1	4	442 ± 8	32 ± 3
2013	Hendrickson Is	14	402 ± 3.2	29 ± 2	9	431 ± 3	31 ± 2
	East Whitefish	4	393 ± 2.6	22 ± 1	1	462	35
	Kendall Is	1	401	Na	5	432 ± 3	29 ± 1
2014	Hendrickson Is	5	403 ± 5.6	24 ± 2	4	435 ± 5	35 ± 6
	East Whitefish	3	412 ± 5.3	23 ± 5	10	443 ± 6	23 ± 1
	Kendall Is				1	442	32

Year	Location	Medium			Large		
		n	Length (cm)	Age (Years)	n	Length (cm)	Age (Years)
2015	Hendrickson Is	9	400 ± 4	23 ± 1	4	433 ± 5	34 ± 1
	East Whitefish				6	439 ± 3	27 ± 4
	Kendall Is	3	407 ± 6	32 ± 6			
2016	Hendrickson Is	4	403 ± 3	Na	4	429 ± 3	Na
2017	Hendrickson Is	9	411 ± 2	23 ± 2	15	435 ± 2	30 ± 4
2018	Hendrickson Is	2	406 ± 5	26 ± 1	7	439 ± 4	38 ± 4
2019	Hendrickson Is	5	413 ± 2	29 ± 3	9	432 ± 4	38 ± 5
	East Whitefish				3	461 ± 6	36 ± 7
2020	Hendrickson Is	10	405 ± 4	24 ± 2	6	433 ± 5	30 ± 6
	East Whitefish	4	406 ± 4	24 ± 2			
	Kendall Is	1	409	14	5	437 ± 4	29 ± 4
	Shingle Pt	1	414	Na	1	467	Na
2021	Hendrickson Is	5	399 ± 5	20 ± 3	5	438 ± 5	28 ± 3
	East Whitefish	3	401 ± 9	33 ± 14	2	450 ± 10	29*
	Kendall Is	4	406 ± 5	22 ± 3	3	444 ± 1	18 ± 1

Skin Tissue Data

After removing 9 outliers (6 medium and 3 large), the skin dataset was comprised of 478 male beluga whales sampled between 2001 and 2021 at four different sampling camps (Table 2). Given that length is a key biological factor influencing Hg levels in skin, we investigated variations in length amongst sampling camps and sampling years to evaluate length as a potential confounding factor in the temporal trend analyses.

Within each group, and for each year considered separately, there were no significant differences in length amongst sampling camps ($p > 0.05$) except in 2019 when large whales sampled at East Whitefish were significantly longer than those sampled at Hendrickson Island ($p < 0.05$). When all sampling locations were grouped together, there were no inter-annual differences in length for medium ($p = 0.098$) and large belugas ($p = 0.546$) suggesting that length was not a confounding factor in the Hg temporal trend analyses in skin.

*Table 2. Length and age (mean ± standard deviation) for male beluga whales sampled at each whaling camp between 2001 and 2021 and analyzed for Hg in skin. Belugas were grouped based on their length into medium (>= 380 cm and <= 420 cm) and large (> 420 cm). Small whales (< 380cm) were not included in the assessment. * Indicates that age was not available for one individual.*

Year	Location	Medium			Large		
		n	Length (cm)	Age (Years)	n	Length (cm)	Age (Years)
2001	Hendrickson Is	14	404 ± 3	28 ± 2	4	436 ± 6	43 ± 4
2002	Hendrickson Is	11	400 ± 4	24 ± 1	10	436 ± 3	35 ± 2
2003	Hendrickson Is	11	406 ± 3	35 ± 4	7	431 ± 4	31 ± 3
2004	Hendrickson Is	8	411 ± 2	30 ± 3	10	433 ± 4	26 ± 2
2005	Hendrickson Is	11	398 ± 3	24 ± 2	7	438 ± 5	32 ± 4
	Kendall is	2	399 ± 3	17 ± 0	5	460 ± 11	28 ± 1
2007	Hendrickson Is	8	404 ± 4	24 ± 2	8	443 ± 8	26 ± 2
2008	Hendrickson Is	10	406 ± 4	34 ± 4	11	439 ± 5	31 ± 2
2009	Hendrickson Is	8	403 ± 4	31 ± 6	8	430 ± 3	28 ± 1
2010	Hendrickson Is	8	403 ± 4	22 ± 2	8	447 ± 10	30 ± 3
2011	Hendrickson Is	9	395 ± 3	23 ± 1	6	427 ± 2	29 ± 2
2012	Hendrickson Is	17	401 ± 3	25 ± 2	6	430 ± 4	30 ± 3
	Kendall Is	5	398 ± 6	19 ± 1	6	439 ± 6	33 ± 4
2013	Hendrickson Is	15	403 ± 3	29 ± 2	9	431 ± 3	31 ± 2
	East Whitefish	8	399 ± 4	24 ± 2	4	443 ± 9	38 ± 4
	Kendall Is	1	401	Na	5	432 ± 3	29 ± 1
2014	Hendrickson Is	10	405 ± 3	28 ± 2	5	433 ± 4	35 ± 4
	East Whitefish	3	412 ± 5	23 ± 5	9	443 ± 7	30 ± 1
	Kendall Is	1	404	Na	2	437 ± 5	32 ± 1
2015	Hendrickson Is	10	401 ± 4	24 ± 1	5	432 ± 4	33 ± 2
	East Whitefish				9	442 ± 6	29 ± 3
	Kendall Is	6	404 ± 5	28 ± 3	2	441 ± 19	25 ± 3

Year	Location	Medium			Large		
		n	Length (cm)	Age (Years)	n	Length (cm)	Age (Years)
2016	Hendrickson Is	5	405 ± 4	Na	4	429 ± 3	Na
2017	Hendrickson Is	7	410 ± 3	23 ± 2	14	435 ± 3	26 ± 2
2018	Hendrickson Is	2	406 ± 5	26 ± 1	7	439 ± 4	38 ± 4
2019	Hendrickson Is	5	413 ± 2	29 ± 3	9	432 ± 4	38 ± 5
	East Whitefish				5	452 ± 7	34 ± 4
2020	Hendrickson Is	7	401 ± 5	25 ± 3	6	436 ± 5	31 ± 5
	East Whitefish	4	406 ± 4	24 ± 2			
	Kendall Is	1	409	14	5	437 ± 4	29 ± 4
	Shingle Pt	1	414	Na	1	467	Na
2021	Hendrickson Is	5	394 ± 3	18 ± 1	6	439 ± 4	29 ± 3
	East Whitefish	3	401 ± 9	33 ± 14	2	450 ± 10	29*
	Kendall Is	5	407 ± 5	22 ± 3	4	441 ± 3	19 ± 1

Liver Tissue Data

After removing 30 outliers (26 young, 4 old), the liver dataset was comprised of 474 male beluga whales sampled between 2001 and 2021 at three different sampling camps (Table 3; no liver data was available from whales sampled at Shingle Pt). Given that age is a key biological factor influencing Hg levels in liver, we investigated variations in age amongst sampling camps and sampling years to evaluate age as a potential confounding factor in the temporal trend analyses. Within each group, and for each year considered separately, there were no significant differences in age amongst sampling camps ($p > 0.05$). When all sampling locations were grouped together, there were no inter-annual differences in age for young ($p = 0.997$) and old belugas ($p = 0.108$) suggesting that age was not a confounding factor in the Hg temporal trend analyses in liver.

Table 3. Length and age (mean ± standard deviation) for male beluga whales sampled at each whaling camp between 2001 and 2021 and analyzed for Hg in liver. Belugas were grouped based on their age into young (> 15 years and <= 35 years) and old (> 35 years). Immature (<= 15 years) whales were not included in this assessment.

Year	Location	Young			Old		
		n	Length (cm)	Age (Years)	n	Length (cm)	Age (Years)
2001	Hendrickson Is	13	405 ± 5	25 ± 2	3	410 ± 19	45 ± 3
2002	Hendrickson Is	16	406 ± 5	25 ± 1	5	437 ± 6	39 ± 1
2003	Hendrickson Is	9	411 ± 8	24 ± 1	7	416 ± 5	43 ± 2
2004	Hendrickson Is	16	410 ± 8	26 ± 1	3	411 ± 7	42 ± 4
2005	Hendrickson Is	13	406 ± 5	23 ± 1	3	425 ± 6	44 ± 4
	Kendall Is	6	422 ± 18	22 ± 2			
2006	Hendrickson Is	26	404 ± 3	23 ± 1	2	422 ± 20	38 ± 2
2007	Hendrickson Is	13	417 ± 9	26 ± 2			
2008	Hendrickson Is	12	431 ± 7	27 ± 2	7	412 ± 7	41 ± 2
2009	Hendrickson Is	11	417 ± 6	27 ± 1	1	406	54
2010	Hendrickson Is	17	416 ± 8	24 ± 1	1	452	53
2011	Hendrickson Is	14	398 ± 5	23 ± 1	1	427	36
	East Whitefish	4	437 ± 7	25 ± 3	1	424	36
	Kendall Is	10	407 ± 7	24 ± 1	1	366	36
2012	Hendrickson Is	20	403 ± 4	24 ± 1	2	427 ± 13	44 ± 1
	Kendall Is	6	414 ± 10	22 ± 2	3	443 ± 5	41 ± 3
2013	Hendrickson Is	20	406 ± 4	24 ± 1	3	424 ± 11	44 ± 1
	East Whitefish	8	398 ± 11	22 ± 1	3	423 ± 7	42 ± 3
	Kendall Is	2	438 ± 4	29 ± 1			
2014	Hendrickson Is	13	402 ± 8	26 ± 2	2	422 ± 13	42 ± 5
	East Whitefish						
	Kendall Is	1	442	32 ± 1			

Year	Location	Young			Old		
		n	Length (cm)	Age (Years)	n	Length (cm)	Age (Years)
2015	Hendrickson Is	17	396 ± 6	23 ± 1	1	429	39
	East Whitefish	7	418 ± 14	25 ± 3	2	459 ± 24	37 ± 1
	Kendall Is						
2016	Hendrickson Is						
2017	Hendrickson Is	12	426 ± 4	24 ± 2	2	441 ± 1	54 ± 16
2018	Hendrickson Is	6	432 ± 9	29 ± 2	4	432 ± 2	46 ± 3
2019	Hendrickson Is	7	421 ± 4	26 ± 1	6	431 ± 6	47 ± 3
	East Whitefish	3	446 ± 11	31 ± 2	1	467	49
2020	Hendrickson Is	13	412 ± 5	24 ± 2	3	435 ± 10	43 ± 4
	East Whitefish	3	406 ± 6	23 ± 3			
	Kendall Is	1	437	27	1	422	37
2021	Hendrickson Is	11	407 ± 10	24 ± 2	3	411 ± 29	43 ± 3
	East Whitefish	2	434 ± 25	24 ± 5	1	412	47
	Kendall Is	3	428 ± 17	20 ± 2			

TEMPORAL TRENDS

Muscle Total Mercury (THg)

For those years for which belugas were sampled at multiple harvesting camps, no significant differences in muscle THg was observed amongst locations (Appendix 3) except in 2015 when medium whales sampled at Kendall Island had significantly higher levels of Hg in muscle (1.9 ± 0.11 μg/g ww) compared to those sampled at Hendrickson Island (1.3 ± 0.11 μg/g ww; $p < 0.05$) and in 2020 when large whales sampled at Kendall Island had significantly higher levels of Hg in muscle (2.4 ± 0.14 μg/g ww) compared to those sampled at Hendrickson Island (1.6 ± 0.17 μg/g ww; $p < 0.05$). These differences could not be explained by size since there were no significant differences in length between the two locations for those year.

Given that Hendrickson Island is a long-term monitoring site and that the majority of the data for this work is coming from this sampling site, the muscle data collected from medium whales sampled at Kendall Island in 2015 and 2020 was removed prior to temporal trend analyses (however trend analyses were run with and without samples from Kendall Island and trends were not impacted).

Over the entire study period, levels of THg in muscle were significantly lower in medium size whales (1.3 ± 0.04 μg/g ww) compared to larger whales (1.91 ± 0.08 μg/g ww, $p < 0.05$).

Temporally, both medium and large whales showed significant non-linear trends in THg in muscle between 2001 and 2021 ($r = 0.20$ and $r = 0.10$ for medium and large whales, respectively, $p < 0.05$). In both size groups, THg levels declined in the early 2000's to reach the lowest levels in 2007 and 2009 for medium and large whales, respectively (Figure 2). Concentrations then increased in both groups until 2013/2014 before declining again until 2017. In recent years, trends appeared quite different between medium and large whales. While Hg concentrations were fairly stable in medium whales since 2017, levels in large whales were increasing between 2017 and 2021 even though 2021 THg levels (2.2 ± 0.3 μg/g ww) were not significantly different than those reported in 2017 (1.2 ± 0.2 μg/g ww; $p = 0.287$) although the lack of significance may be related to the low sample size. Looking at the start and end of the study period, THg levels in muscle reported in whales sampled in 2021 (0.83 ± 0.06 and 2.2 ± 0.3 μg/g ww for medium and large whales, respectively) were similar to those reported in 2001 for large whales (2.2 ± 0.3 μg/g ww) but they were significantly lower in medium whales (1.5 ± 0.1 μg/g ww; $p < 0.05$).

Because Hg in muscle is driven by dietary uptake and given that medium and large whales feed in different habitats with large males selecting close sea-ice cover in and near the Arctic Archipelago likely feeding benthically and medium-sized males selecting habitats near the ice edge likely feeding on the sea-ice associated Arctic Cod and/or in pelagic food webs (Loseto et al., 2006; 2008), Hg levels in their respective prey may provide insight into the temporal trends observed here, especially since they seemed to differ between the two groups in recent years and might reflect shifts in diet in one group but not the other.

Using data only for whales harvested at Hendrickson Island between 1981 and 2012, Loseto et al. (2015) reported no trend in the medium size group in the early 2000's but, similar to the present study, a significant decline for the larger whales was observed. It is important to note that, in addition to focusing on Hendrickson Island whales only, Loseto et al. (2015) evaluated trends using yearly averages of Hg levels, which may explain the difference in significance for medium whales between their study and the present report.

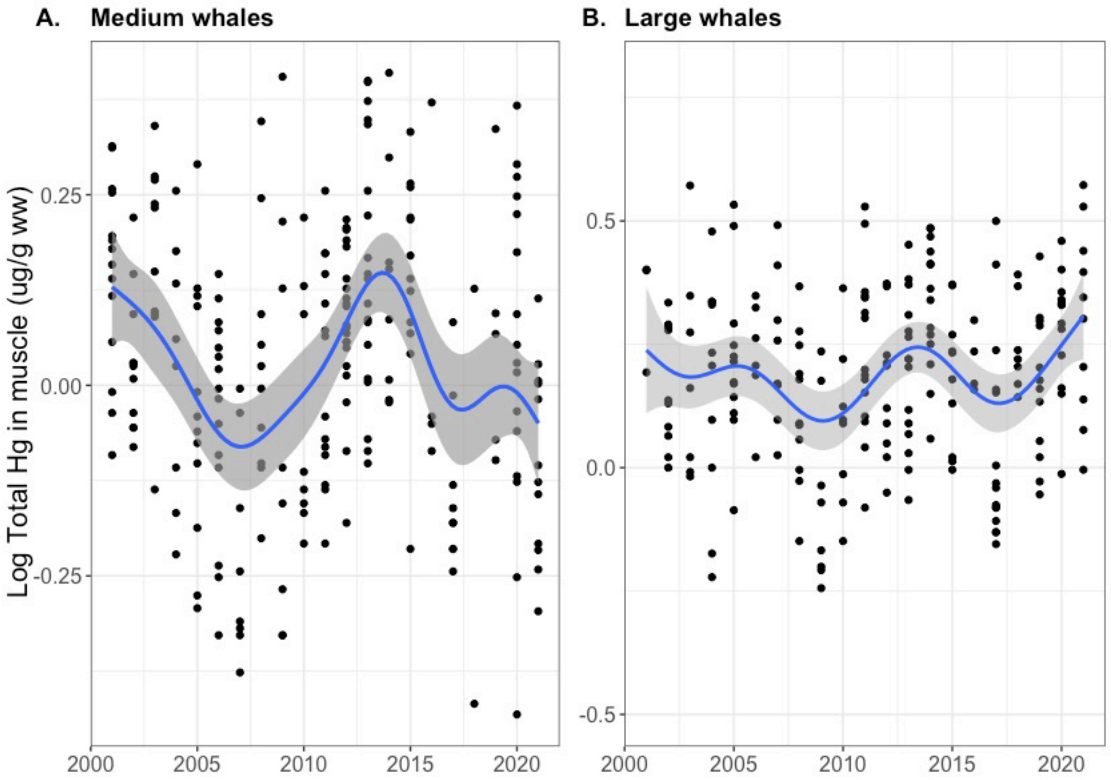


Figure 2. Total mercury concentrations (log-transformed) in the muscle of individual EBS male beluga whales sampled between 2001 and 2021. Whales were grouped by size into medium (>= 380 cm and <= 420 cm) and large (> 420 cm).

Skin Total Mercury (THg)

For years when belugas were sampled at multiple sampling camps, there were no significant differences in THg in skin amongst locations (Appendix 4) except in 2019 when large whales sampled at East Whitefish had significantly higher Hg levels in skin (1.3 ± 0.3 μg/g ww) than those sampled at Hendrickson Island (0.6 ± 0.1 μg/g ww; $p < 0.05$) and in 2020 when large whales sampled at Kendall Island had significantly higher THg in skin (1.1 ± 0.1 μg/g ww) than those sampled at Hendrickson Island (0.4 ± 0.05 μg/g ww; $p < 0.05$). While the 2019 difference could be explained by the large whales sampled at East Whitefish being significantly longer than those sampled at Hendrickson Island, size did not explain the differences observed in 2020. As for muscle, the skin data collected from large whales sampled at East Whitefish in 2019 and Kendall Island in 2020 was removed from the final dataset prior to temporal trend analysis in skin.

As expected, and since THg levels in skin were correlated with those measured in muscle ($r = 0.58$ $p < 0.05$; Appendix 1), temporal trends observed in skin were similar to those reported in muscle. There were significant non-linear temporal trends in medium and large whales ($r = 0.18$ and $r = 0.15$, respectively; $p < 0.05$). In both size groups, THg levels declined in the early 2000's to reach the lowest levels in 2007/2008 and 2009 for medium and large whales, respectively. Concentrations then increased in both groups until 2013/2014 before declining again until 2017/2018 (Figure 3). In recent years, trends appeared quite different between medium and large whales. While THg concentrations were fairly stable in medium whales since 2017, levels in large whales were increasing between 2017 and 2021 even though 2021 Hg levels (1.1 ± 0.1 μg/g ww) were not significantly different than those reported in 2017 (0.5 ± 0.04 μg/g ww; p =

0.542). Looking at THg levels in skin at the start and end of the study period, whales sampled in 2021 (0.48 ± 0.09 and 1.1 ± 0.1 μg/g ww for medium and large whales, respectively) had similar levels to those reported in 2001 (0.74 ± 0.06 and 1.3 ± 0.1 μg/g ww, respectively).

As explained above for muscle, these divergent trends may be explained by a shift in diet in one group but not the other, since they are feeding in different habitats that may be affected differently by climate change.

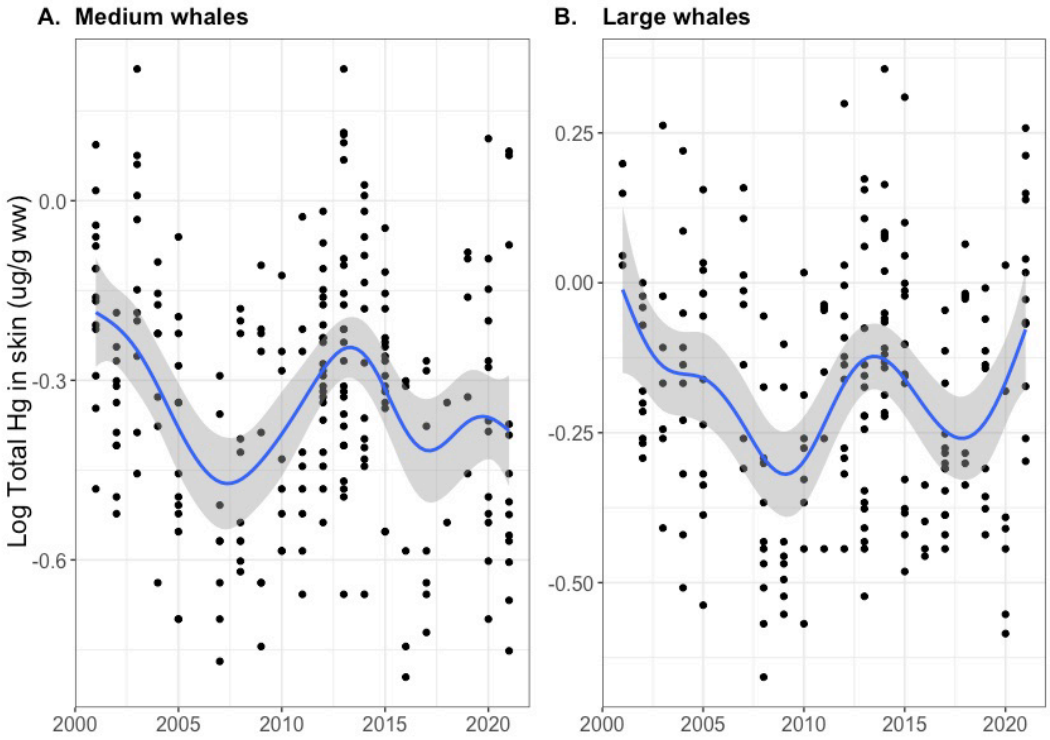


Figure 3. Total mercury concentrations (log-transformed) in the skin of individual EBS male beluga whales sampled between 2001 and 2021. Whales were grouped into medium (>= 380 cm and <= 420 cm) and large (> 420 cm).

Liver Total Mercury (THg)

For years when belugas were sampled at multiple sampling camps, there were no significant differences in THg in liver amongst locations (Appendix 5) except in 2013 when young whales sampled at Kendall Island had significantly higher levels of THg in liver (45.0 ± 4.9 μg/g ww) compared to those sampled at East Whitefish and Hendrickson Island (9.7 ± 3.4 and 18.1 ± 2.1 μg/g ww, respectively; $p < 0.05$). This difference could not be explained by age since there were no significant differences in age between these locations for that year. For the same reasons as for muscle and skin, the liver data collected from young whales sampled at Kendall Island in 2013 were removed from the final dataset before carrying out the temporal trend analysis for liver. Note that in 2014, there was only one young whale sampled at Kendall Island so no statistical analysis was possible, but this whale appeared to have THg levels as high (52.4 μg/g ww) as those reported in whales sampled at Kendall Island in 2013 and this data point was therefore also removed from the final dataset.

Over the entire study period, levels of THg in the liver were on average significantly lower in young whales (22.4 ± 1.2 μg/g ww) compared to old whales (53.4 ± 4.8 μg/g ww, $p < 0.05$). While there was no significant trend in THg in the liver of young whales between 2001 and

2021, there was a significant linear decreasing trend for the old whales ($r = 0.10$; $p < 0.05$) with an annual decrease estimated at 1.5% (Figure 4). For the period 1981 to 2012 (Hendrickson Island only), Loseto et al. (2015) also found no trend in young whales after the year 2000 (with concentrations fluctuating around 20 μg/g ww) and a decreasing trend for the old whales. While they concluded that these trends suggested no change in Hg exposure for the young whales between 2002 and 2012, they suggested that the decline in THg levels measured in older whales could be the result of a shift in dietary exposure and/or a decrease in Hg concentrations in their prey (Loseto et al., 2015).

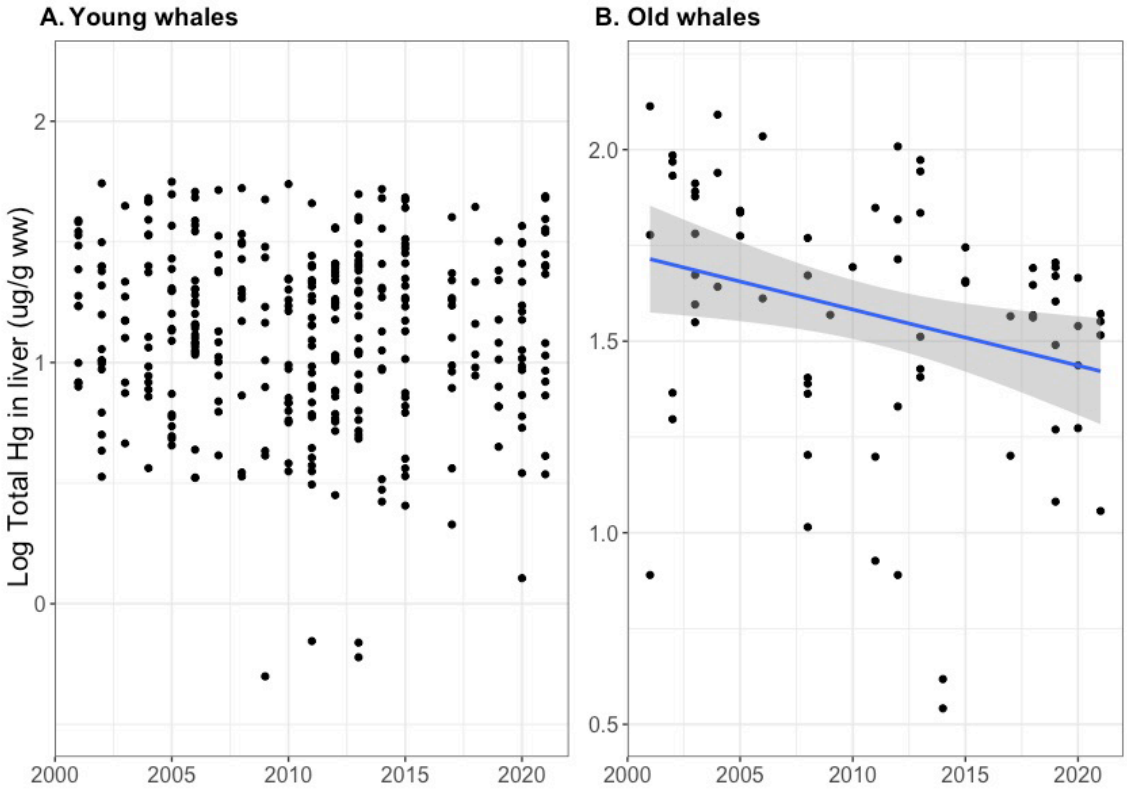


Figure 4. Total mercury concentrations (log-transformed) in the liver of individual EBS male beluga whales sampled between 2001 and 2021. Whales were grouped by age into young (> 15 years and <= 35 years) and old (> 35 years).

RISK FOR MERCURY-MEDIATED HEALTH EFFECTS

As expected, more of the older whales relative to younger whales were in higher risk categories due to the known accumulation of THg with age. The pattern was stable during the study period for young whales, with the majority of individuals in the No Risk and Low Risk categories (50.7 ± 11.8 % and 43.8 ± 11.7% on average over the entire period, respectively) (Figure 5; Appendix 6). For the old whales group, a higher proportion of sampled whales were at higher risk of Hg-mediated health effects in the early 2000's with 60 % of whales being at severe risk in 2001 and between 50 and 66.7 % of sampled whales being at moderate to high risks between 2002 and 2006 (Figure 5; Appendix 7). While 33.3 to 50% of older whales sampled between 2011 and 2013 were at moderate to high risk, most of the older whales sampled in recent years were at low to no risk (Figure 5) in accordance with the decreasing trend in THg levels in the liver of older whales described above. Only one whale recently (2020) was categorized as under 'severe risk' in 2020.

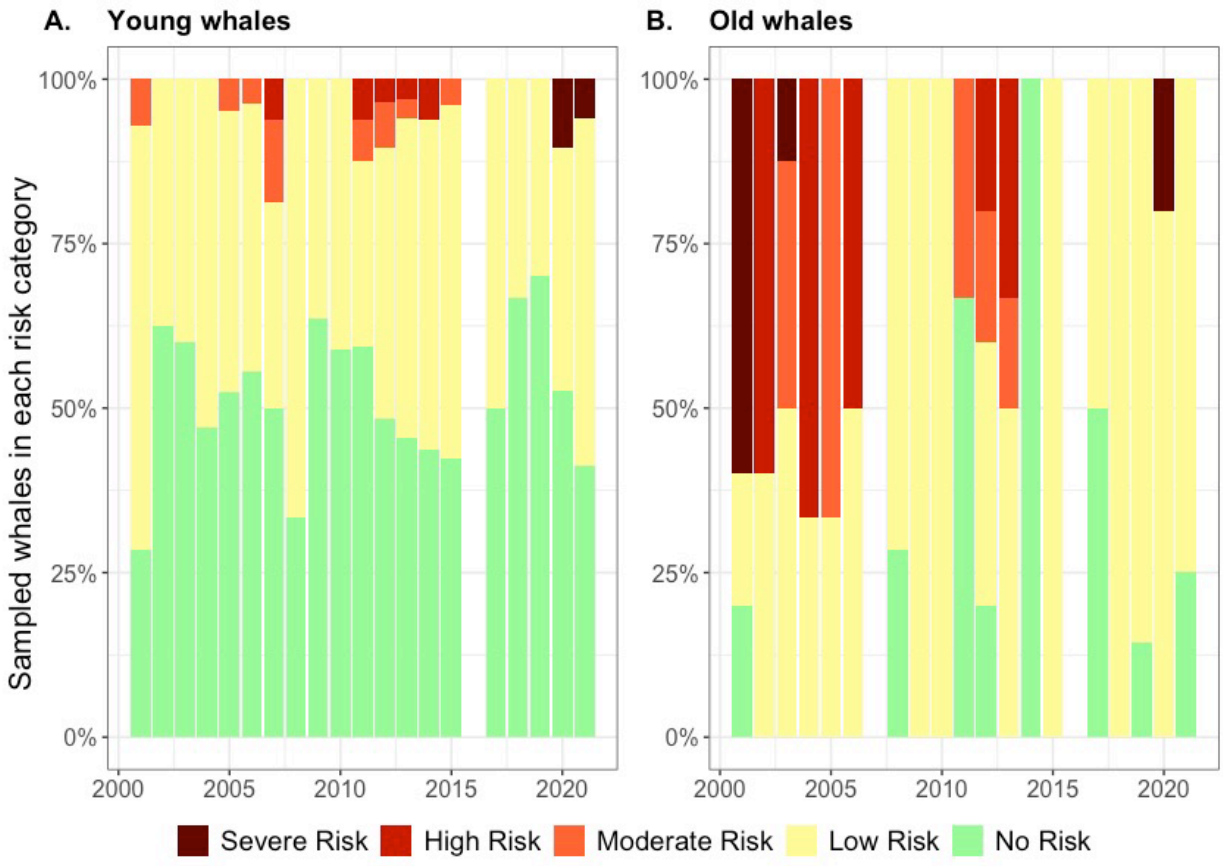


Figure 5. Percentage of sampled beluga whales at risk of deleterious health effects related to THg levels measured in liver.

It is important to note that these results should be interpreted with caution for two main reasons:

- The sample size was very limited for old whales and therefore likely not representative of the entire older beluga population.
- Thresholds used were derived from harp seals and there is potential for inter-species differences in Hg distribution and speciation in the liver as well as sensitivity to Hg.

CONCLUSION

THg levels in the muscle of medium and large whales revealed non-linear trends between 2001 and 2021. While trends were similar in both size groups until 2017, recent trends differed with THg levels being fairly constant in medium whales but increasing in large whales. This resulted in 2021 THg levels in muscle being significantly lower than those reported in 2001 in medium whales but similar in large whales. Biological variables such as age and size (length) did not explain any of the trends reported. As Hg in muscle is driven by dietary uptake, analyses of dietary markers such as stable isotopes and fatty acids will help shed some light on the THg trends observed in beluga whales. In addition, given that medium and large whales feed in different habitats with large males selecting close sea-ice cover in and near the Arctic Archipelago likely feeding benthically and medium-sized males selecting habitats near the ice edge likely feeding on the sea-ice associated Arctic Cod and/or in pelagic food webs (Loseto et

al., 2006; 2008), Hg levels in their respective prey may also provide insight into the temporal trends observed, especially since they seemed to differ between the two groups in recent years.

Since THg levels in skin were highly correlated with those in muscle, THg temporal trends in skin were similar to those reported in muscle suggesting that it may not be necessary to evaluate Hg in both tissues when only considering belugas. However, we recognize that both skin (muktuk) and muscle (mipqu) are consumed by Inuvialuit and Hg is a concern for human health (Waugh et al., 2018). Skin therefore remains an important parameter to monitor from a human health perspective.

Hg in liver is not directly linked to Inuit diet unlike the muscle and skin samples, but is important to look at as liver is a target tissue for Hg toxicity in mammals. While no trends were observed in young whales between 2001 and 2021, THg levels in old whales declined over the study period with levels in 2021 being significantly lower than those reported in 2001. THg levels in liver were used to evaluate risks for Hg-mediated health effects in beluga whales. Overall, the EBS population was at low risk of health effects from Hg exposure with most whales being in the no- to low-risk categories in recent years. However, more research is needed in order to develop species-specific appropriate risk thresholds.

These results support the use of Hg levels in beluga muscle as a key ecological indicator for the TN MPA that can provide insights into food web shifts and/or alterations of Hg transport and fate in the ecosystem due to climate change and/or other anthropogenic stressors. However, in order to accurately interpret changes, this indicator needs to be evaluated together with other dietary markers such as stable isotopes and fatty acids along with Hg levels in prey consumed by belugas. Finally, it is important to also consider THg levels in liver to evaluate beluga health as it relates to Hg exposure using established thresholds.

CREDIT AUTHORSHIP CONTRIBUTION STATEMENT

Marie Noel: Writing – original draft, Formal analysis, Visualization, Jane Kirk: Methodology, Writing – review & editing, Gary Stern: Conceptualization, Methodology, Writing – review & editing, Tuktoyaktuk HTC: Conceptualization, Writing – review & editing, Frank Pokiak: Conceptualization, Data curation, Nellie Pokiak: Conceptualization, Data curation, Amber Gleason: Methodology, Writing – review & editing, Debbie Armstrong: Methodology, Writing – review & editing, Enooyaq Sudlovenick: Writing and Reviewing, FJMC: Resources, Data curation, Writing – review & editing, Lisa Loseto: Conceptualization, Funding acquisition, Visualization, Writing – review & editing.

ACKNOWLEDGEMENTS

This study could not have been done without the support from the communities, hunters and Hunters and Trappers Committees of Aklavik, Inuvik and Tuktoyaktuk, NT who kindly shared their food for research and monitoring. We are also grateful for the support of the Fisheries Joint Management Committee, the board and staff who have ensured the beluga harvest program is maintained as well as supported leveraged contaminant monitoring studies. Many community monitors participated in the program to collect the samples and data from harvested whales. Dedicated health monitors at Hendrickson Island, include the co-authors, the late Frank and Nellie Pokiak who sampled tissues from 2001 to 2012, monitors who continued to sample and monitor at Hendrickson island included Verna Pokiak, Rex Noksana, Jonas Lucas, Lionel Kikoak, Dale Panaktalok, Pete Keevik and Ronald Felix. Health tissues were sampled at Kendal Island in 2005 by Jerry Rogers and in following years by the late John Day, Kyle Conley, James Rogers, Donna Rogers, Hang Angasuk, Dale Rogers, Larry Rogers, and Rufus Tingmiak, and

at East Whitefish Bertha Joe, Steve Cockney, Steve Cockney Jr., the late Lawrence (Fraser) Angasuk, Isabel Hendrick, Tori Hendrick, and Rebecca Kaglik. The long term monitoring program was funded by Fisheries and Ocean Canada, the Fisheries Joint Management Committee and the Northern Contaminants Program (CIRNAC). We are thankful for the field support teams from DFO and FJMC including in alphabetical order: Kevin Bill, Connie Blakeston, Jasmine Brewster, Kiyo Campbell, Emily Choy, Larry Dow, Ashley Elliot, Kayla Hansen-Craik, Lois Harwood, Claire Hornby, Andrea Hoyt, Kristin Hynes, Shannon MacPhee, James Malone, Laura Murray, Dana Neumann, Sonja Ostertag, Louie Porta, Oksana Schimnowski, Terry Stein, Erica Wall, Megan Wardekker, Emilie Way-Nee and the laboratory support of Joanne Delaronde, Blair Dunn and Gail Boila at DFO, and Alexis Burt at University of Manitoba.

REFERENCES

Alunik, I., Kolausok, E.D. and Morrison, D.A., 2003. Across time and tundra: the Inuvialuit of the western Arctic. Raincoast Books.

AMAP, 2020. AMAP assessment data portal. Updated January 2020. Accessed November 2023. https://www.amap.no/ahat.

AMAP, 2021. AMAP Assessment 2021: Mercury in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Tromso, Norway, 324pp.

Basu, N., Scheuhammer, A.M., Sonne, C., Letcher, R.J., Born, E.W. and Dietz, R., 2009. Is dietary mercury of neurotoxicological concern to wild polar bears (Ursus maritimus)?. Environ. Toxicol. Chem. 28(1), 133-140.

Brewster, J. D., Hansen-Craik, K., Harwood, L.A., and Blakeston, C. 2021. State of the Tarium Niryutait Marine Protected Areas (TNMPA) Report: Inventory of Monitoring from 2010–2016. Can. Manuscr. Rep. Fish. Aquat. Sci. 3301: x + 84 p

Brown, T.M., Ross, P.S., Reimer, K.J., Veldhoen, N., Dangerfield, N.J., Fisk, A.T. and Helbing, C.C., 2014. PCB related effects thresholds as derived through gene transcript profiles in locally contaminated ringed seals (Pusa hispida). Environ. Sci. Technol. 48(21), 12952-12961.

Burnham, K.P., Anderson, D.R. 2002. Model selection and inference: a practical information theoretic approach. 2nd edition. Springer-Verlag, New York https://doi.org/10.1007/b97636

De Guise, S., Bernier, J., Dufresne, M.M., Martineau, D., Béland, P. and Fournier, M., 1996. Immune functions in beluga whales (Delphinapterus leucas): Evaluation of mitogen-induced blastic transformation of lymphocytes from peripheral blood, spleen and thymus. Vet. Immunol. Immunopathol., 50(1-2),117-126.

Desforges, J.P.W., Ross, P.S., Dangerfield, N., Palace, V.P., Whiticar, M. and Loseto, L.L., 2013. Vitamin A and E profiles as biomarkers of PCB exposure in beluga whales (Delphinapterus leucas) from the western Canadian Arctic. Aquat. Toxicol., 142, 317-328.

Desforges, J. P., Outridge, P., Hobson, K. A., Heide-Jørgensen, M. P., & Dietz, R., 2021. Anthropogenic and climatic drivers of long-term changes of mercury and feeding ecology in Arctic Beluga (Delphinapterus leucas) populations. Environ. Sci. Technol.56(1), 271-281.

DFO, 2013. Tarium Niryutait: Marine Protected Area management plan. Fisheries and Oceans Canada, Winnipeg, MB.

Dietz, R., Letcher, R.J., Desforges, J.P., Eulaers, I., Sonne, C., Wilson, S., Andersen-Ranberg, E., Basu, N., Barst, B.D., Bustnes, J.O. and Bytingsvik, J., 2019. Current state of knowledge on biological effects from contaminants on arctic wildlife and fish. Sci. Tot. Environ., 696, 133792.

Dietz, R., Sonne, C., Basu, N., Braune, B., O'Hara, T., Letcher, R.J., Scheuhammer, T., Andersen, M., Andreasen, C., Andriashek, D. and Asmund, G., 2013. What are the toxicological effects of mercury in Arctic biota?. Sci. Tot. Environ., 443, 775-790.

Dietz, R., Letcher, R.J., Aars, J., Andersen, M., Boltunov, A., Born, E.W., Ciesielski, T.M., Das, K., Dastnai, S., Derocher, A.E. and Desforges, J.P., 2022. A risk assessment review of mercury exposure in Arctic marine and terrestrial mammals. Sci. Tot. Environ., 829,154445.

Dietz, R., Letcher, R.J., Desforges, J.P., Eulaers, I., Sonne, C., Wilson, S., Andersen-Ranberg, E., Basu, N., Barst, B.D., Bustnes, J.O. and Bytingsvik, J., 2019. Current state of knowledge on biological effects from contaminants on arctic wildlife and fish. Sci. Tot. Environ., 696,133792.

Environment and Climate Change Canada 2021. Canadian Environmental Sustainability Indicators: Sea Ice in Canada. Available at https://www.canada.ca/content/dam/eccc/documents/pdf/cesindicators/sea-ice/2021/sea-ice-en.pdf

FJMC, 2013. Beaufort sea beluga management plan. Fisheries Joint Management Committee, Inuvik, NWT.

Frouin, H., Ménard, L., Brousseau, P. and Fournier, M., 2010. T Lymphocyte-Proliferative Responses of a Grey Seal (Halichoerus grypus) Exposed to Heavy Metals and PCBs in Vitro. Aquat. Mam., 36(4).

Frouin, H., Loseto, L.L., Stern, G.A., Haulena, M. and Ross, P.S., 2012. Mercury toxicity in beluga whale lymphocytes: limited effects of selenium protection. Aquat. Toxicol., 109, 185-193.

Gazette, 2010. Tarium Niryutait Marine Protected Area regulations. OCEANS, D.O.F.A (ed.)

Harwood, L.A., Innes, S., Norton, P. and Kingsley, M.C., 1996. Distribution and abundance of beluga whales in the Mackenzie estuary, southeast Beaufort Sea, and west Amundsen Gulf during late July 1992. Can. J. Fish. Aquat. Sci., 53(10), 2262-2273.

Harwood, L.A., Norton, P., Day, B. and Hall, P.A., 2002. The harvest of beluga whales in Canada's Western Arctic: Hunter-based monitoring of the size and composition of the catch. Arctic, pp.10-20.

Harwood, L.A., Iacozza, J., Auld, J.C., Norton, P. and Loseto, L., 2014. Belugas in the Mackenzie River estuary, NT, Canada: habitat use and hot spots in the Tarium Niryutait Marine Protected Area. Ocean Coast. Manag., 100, 128-138.

Harwood, L.A., Kingsley, M.C.S., Pokiak, F. 2015. Monitoring Beluga Harvests in the Mackenzie Delta and Near Paulatuk, NT, Canada: Harvest Efficiency and Trend, Size and Sex of Landed Whales, and Reproduction, 1970-2009. Can. Manuscr. Rep. Fish. Aquat. Sci. 3059: vi + 32p.

Krey, A., Ostertag, S.K. and Chan, H.M., 2015. Assessment of neurotoxic effects of mercury in beluga whales (Delphinapterus leucas), ringed seals (Pusa hispida), and polar bears (Ursus maritimus) from the Canadian Arctic. Sci. Tot. Environ., 509, 237-247.

Lalancette, A., Morin, Y., Measures, L. and Fournier, M., 2003. Contrasting changes of sensitivity by lymphocytes and neutrophils to mercury in developing grey seals. Dev. Comp. Immunol., 27(8), 735-747.

Letcher, R.J., Bustnes, J.O., Dietz, R., Jenssen, B.M., Jørgensen, E.H., Sonne, C., Verreault, J., Vijayan, M.M. and Gabrielsen, G.W., 2010. Exposure and effects assessment of persistent organohalogen contaminants in arctic wildlife and fish. Sci. Tot. Environ., 408(15), 2995-3043.

Lockhart, W.L., Stern, G.A., Wagemann, R., Hunt, R.V., Metner, D.A., DeLaronde, J., Dunn, B., Stewart, R.E.A., Hyatt, C.K., Harwood, L. and Mount, K., 2005. Concentrations of mercury in tissues of beluga whales (Delphinapterus leucas) from several communities in the Canadian Arctic from 1981 to 2002. Sci. Tot. Environ., 351, 391-412.

Loseto, L.L., Richard, P., Stern, G.A., Orr, J. and Ferguson, S.H., 2006. Segregation of Beaufort Sea beluga whales during the open-water season. Can. J. Zool., 84(12),1743-1751.

Loseto, L.L., Stern, G.A., Deibel, D., Connelly, T.L., Prokopowicz, A., Lean, D.R.S., Fortier, L. and Ferguson, S.H., 2008a. Linking mercury exposure to habitat and feeding behaviour in Beaufort Sea beluga whales. J. Mar. Syst., 74(3-4),1012-1024.

Loseto, L.L., Stern, G.A. and Ferguson, S.H., 2008b. Size and biomagnification: how habitat selection explains beluga mercury levels. Environ. Sci. Technol., 42(11), 3982-3988.

Loseto, L.L., Stern, G.A., Connelly, T.L., Deibel, D., Gemmill, B., Prokopowicz, A., Fortier, L. and Ferguson, S.H., 2009. Summer diet of beluga whales inferred by fatty acid analysis of the eastern Beaufort Sea food web. J. Exp. Mar. Biol. Ecol., 374(1), 12-18.

Loseto, L.L., Stern, G.A. and Macdonald, R.W., 2015. Distant drivers or local signals: where do mercury trends in western Arctic belugas originate?. Sci. Tot. Environ., 509, 226-236.

Loseto, L.L., Hoover, C., Ostertag, S., Whalen, D., Pearce, T., Paulic, J., Iacozza, J. and MacPhee, S., 2018. Beluga whales (Delphinapterus leucas), environmental change and marine protected areas in the Western Canadian Arctic. Estuar. Coast. Shelf Sci., 212, 128-137.

Mcghee, R. 1988. Beluga hunters: An archeological reconstruction of the history and culture of the Mackenzie Delta Kittegaryumiut. Canadian Museum of Civilization, Gatineau, QC.

Mayette, A., Marcoux, M., Iacozza, J., Ferguson, M., Brower, A. and Loseto, L., 2023. Summer distribution and habitat preference of beluga whale social groups in the Eastern Beaufort Sea. Arctic Sci..

Moore, S.E., Reeves, R.R., Southall, B.L., Ragen, T.J., Suydam, R.S. and Clark, C.W., 2012. A new framework for assessing the effects of anthropogenic sound on marine mammals in a rapidly changing Arctic. BioSci., 62(3), pp.289-295.

Morris, A.D., Wilson, S.J., Fryer, R.J., Thomas, P.J., Hudelson, K., Andreasen, B., Blévin, P., Bustamante, P., Chastel, O., Christensen, G. and Dietz, R., 2022. Temporal trends of mercury in Arctic biota: 10 more years of progress in Arctic monitoring. Sci. Tot. Environ., 839, 155803.

Noël, M., Loseto, L.L., Helbing, C.C., Veldhoen, N., Dangerfield, N.J. and Ross, P.S., 2014. PCBs are associated with altered gene transcript profiles in Arctic beluga whales (Delphinapterus leucas). Environ. Sci. Technol., 48(5), 2942-2951.

Noël, M., Loseto, L.L. and Stern, G., 2018. Legacy contaminants in the eastern Beaufort Sea beluga whales (Delphinapterus leucas): are temporal trends reflecting regulations?. Arctic Sci., 4(3), pp.373-387.

Ostertag, S.K., Shaw, A.C., Basu, N. and Chan, H.M., 2014. Molecular and neurochemical biomarkers in Arctic beluga whales (Delphinapterus leucas) were correlated to brain mercury and selenium concentrations. Environ. Sci. Technol., 48(19), 11551-11559.

Petersson, K., Dock, L., Söderling, K. and Vahter, M., 1991. Distribution of mercury in rabbits subchronically exposed to low levels of radiolabeled methyl mercury. Pharmacol. Toxicol., 68(6), 464-468.

Pleskach, K., Hoang, W., Chu, M., Halldorson, T., Loseto, L., Ferguson, S.H. and Tomy, G.T., 2016. Use of mass spectrometry to measure aspartic acid racemization for ageing beluga whales. Mar. Mamm. Sci., 32(4), 1370-1380.

Rawson, A.J., Patton, G.W., Hofmann, S., Pietra, G.G. and Johns, L., 1993. Liver abnormalities associated with chronic mercury accumulation in stranded atlantic bottlenosed dolphins. Ecotoxicol. Environ. Saf., 25(1), 41-47.

R Core Team, 2020. R: a language and environment for statistical computing. URLR Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Reeves, R.R., Ewins, P.J., Agbayani, S., Heide-Jørgensen, M.P., Kovacs, K.M., Lydersen, C., Suydam, R., Elliott, W., Polet, G., van Dijk, Y. and Blijleven, R., 2014. Distribution of endemic cetaceans in relation to hydrocarbon development and commercial shipping in a warming Arctic. Mar. Pol., 44, 375-389.

Ronald, K., Tessaro, S.V., Uthe, J.F., Freeman, H.C., Frank, R., 1977. Methylmercury poisoning in the harp seal (Pagophilus groenlandicus). Sci. Tot. Environ. 8, 1-11.

Scheuhammer, A., Braune, B., Chan, H.M., Frouin, H., Krey, A., Letcher, R., Loseto, L., Noël, M., Ostertag, S., Ross, P. and Wayland, M., 2015. Recent progress on our understanding of the biological effects of mercury in fish and wildlife in the Canadian Arctic. Sci. Tot. Environ.t, 509, 91-103.

Shore, R.F., Pereira, G., Walker, L.., Thompson, D.R. 2011. Mercury in non marine birds and mammals. In: Beyer, W.N., Meador, J. Editors. Environmental contaminants in biota: Interpreting tissue concentrations. Second Edition Boca Raton, FL: CRC Press. P 603-618.

Sonne, C., Dietz, R., Leifsson, P.S., Asmund, G., Born, E.W. and Kirkegaard, M., 2007. Are liver and renal lesions in East Greenland polar bears (Ursus maritimus) associated with high mercury levels?. Environ. Health, 6(1), 1-9.

Sonne, C., 2010. Health effects from long-range transported contaminants in Arctic top predators: an integrated review based on studies of polar bears and relevant model species. Environ. Int., 36(5), 461-491.

Stern, G.A., Macdonald, C.R., Armstrong, D., Dunn, B., Fuchs, C., Harwood, L., Muir, D.C.G. and Rosenberg, B., 2005. Spatial trends and factors affecting variation of organochlorine contaminants levels in Canadian Arctic beluga (Delphinapterus leucas). Sci. Tot. Environ., 351, pp.344-368.

Stewart, R.E.A., Campana, S.E., Jones, C.M. and Stewart, B.E., 2006. Bomb radiocarbon dating calibrates beluga (Delphinapterus leucas) age estimates. Can. J. Zool., 84(12), 1840-1852.

Thompson, D.R., 1996. Mercury in birds and terrestrial mammals. In: Beyer WN, Heinz, GH, Redmon-Norwood AW, editors. Environmental contaminants in wildlife: interpreting tissue concentrations. CRC press; 1996. P 341-56.

Usher, P.J., 2002. Inuvialuit use of the Beaufort Sea and its resources, 1960-2000. Arctic, pp.18-28.

US EPA (2007). SW-846 Test Method 7473: Mercury in Solids and Solutions by Thermal Decomposition, Amalgamation, and Atomic Absorption Spectrophotometry. CAS# 7439-97-6. US Environmental Protection Agency, 17 pp.

Uthe, J.F., Armstrong, F.A.J. and Tam, K.C., 1971. Determination of trace amounts of mercury in fish tissues: Results of a North American check sample study. J. AOAC Int., 54(4), 866-869.

Wagemann, R., Trebacz, E., Boila, G. and Lockhart, W.L., 1998. Methylmercury and total mercury in tissues of arctic marine mammals. Sci. Tot. Environ., 218(1), 19-31.

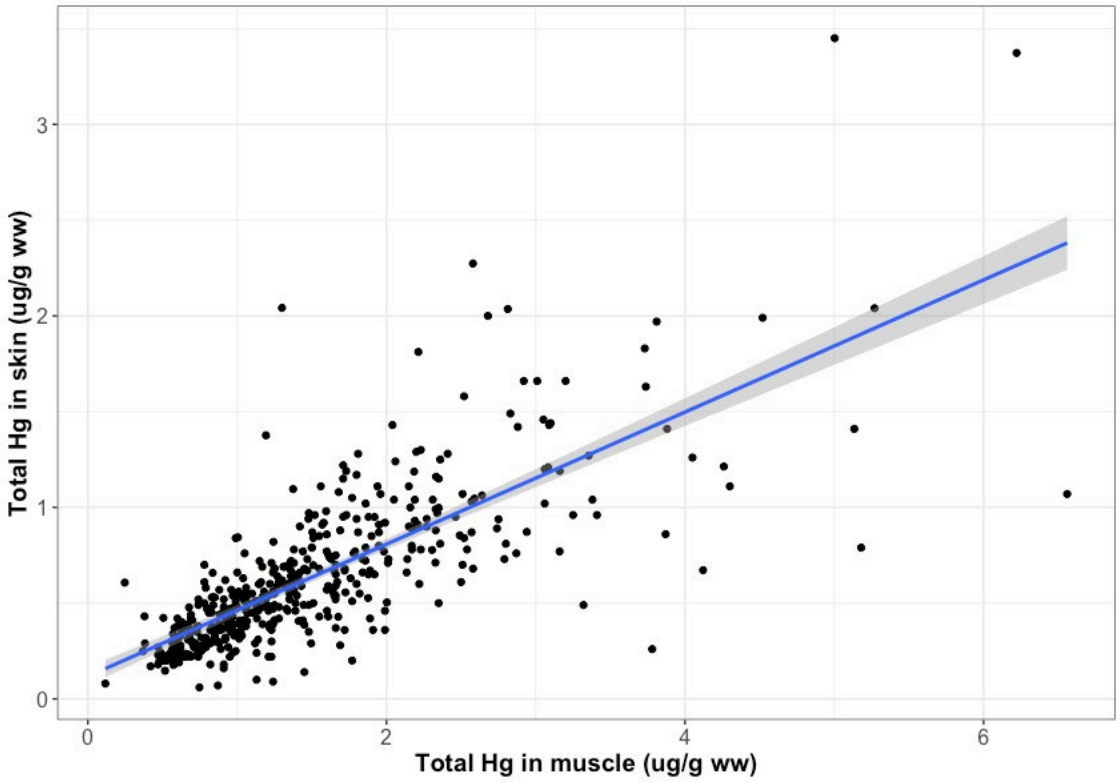
Wagemann, R., Trebacz, E., Boila, G. and Lockhart, W.L., 2000. Mercury species in the liver of ringed seals. Sci. Tot. Environ., 261(1-3), 21-32.

Wagemann, R. and Kozlowska, H., 2005. Mercury distribution in the skin of beluga (Delphinapterus leucas) and narwhal (Monodon monoceros) from the Canadian Arctic and mercury burdens and excretion by moulting. Sci. Tot. Environ., 351, 333-343.

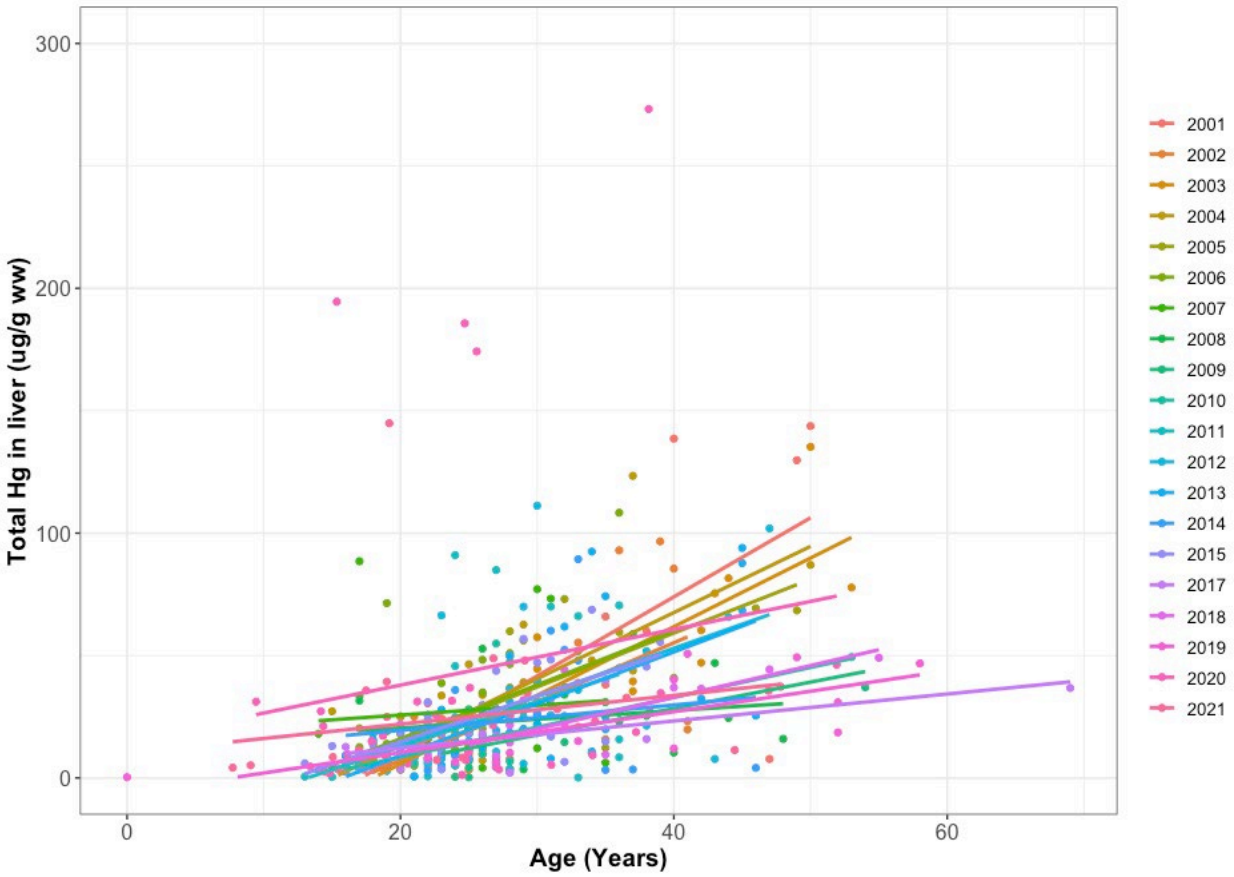
Waugh, D., Pearce, T., Ostertag, S.K., Pokiak, V., Collings, P. and Loseto, L.L., 2018. Inuvialuit traditional ecological knowledge of beluga whale (Delphinapterus leucas) under changing climatic conditions in Tuktoyaktuk, NT. Arctic Sci., 4(3), 242-258.

Wu, P., & Zhang, Y., 2023. Toward a global model of methylmercury biomagnification in marine food webs: Trophic dynamics and implications for human exposure. Environ. Sci.Technol.57(16), 6563-6572.

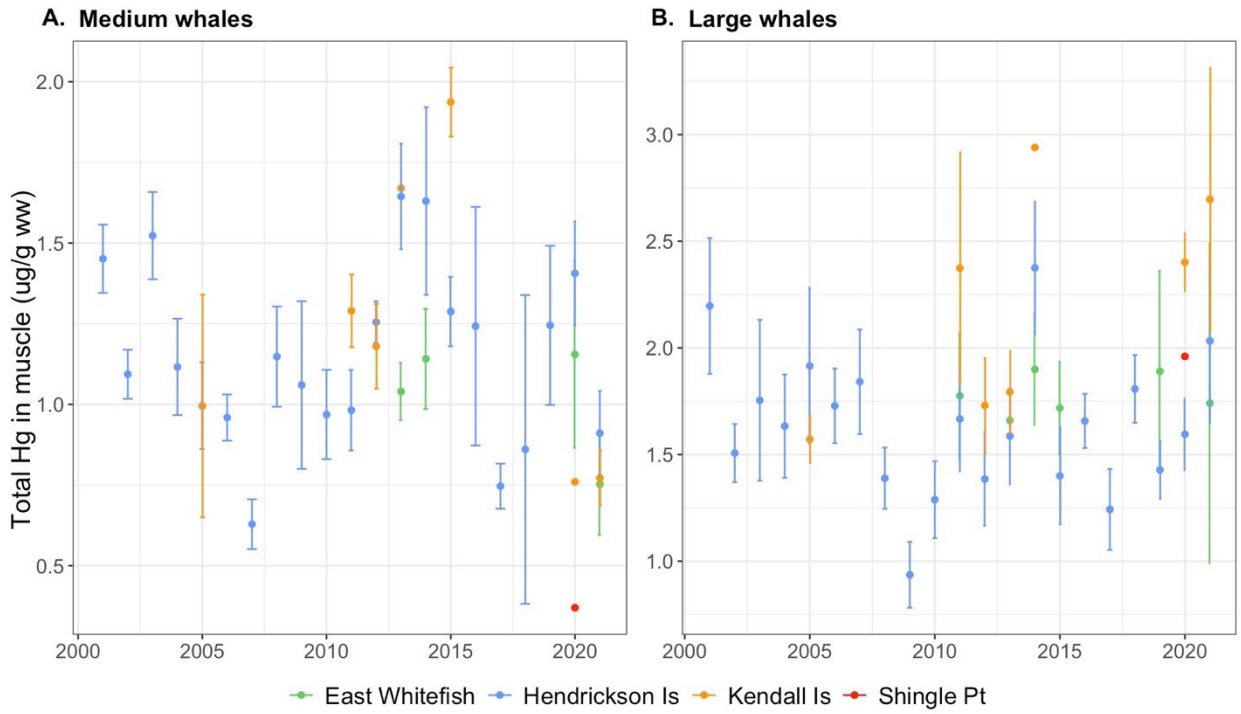
APPENDICES



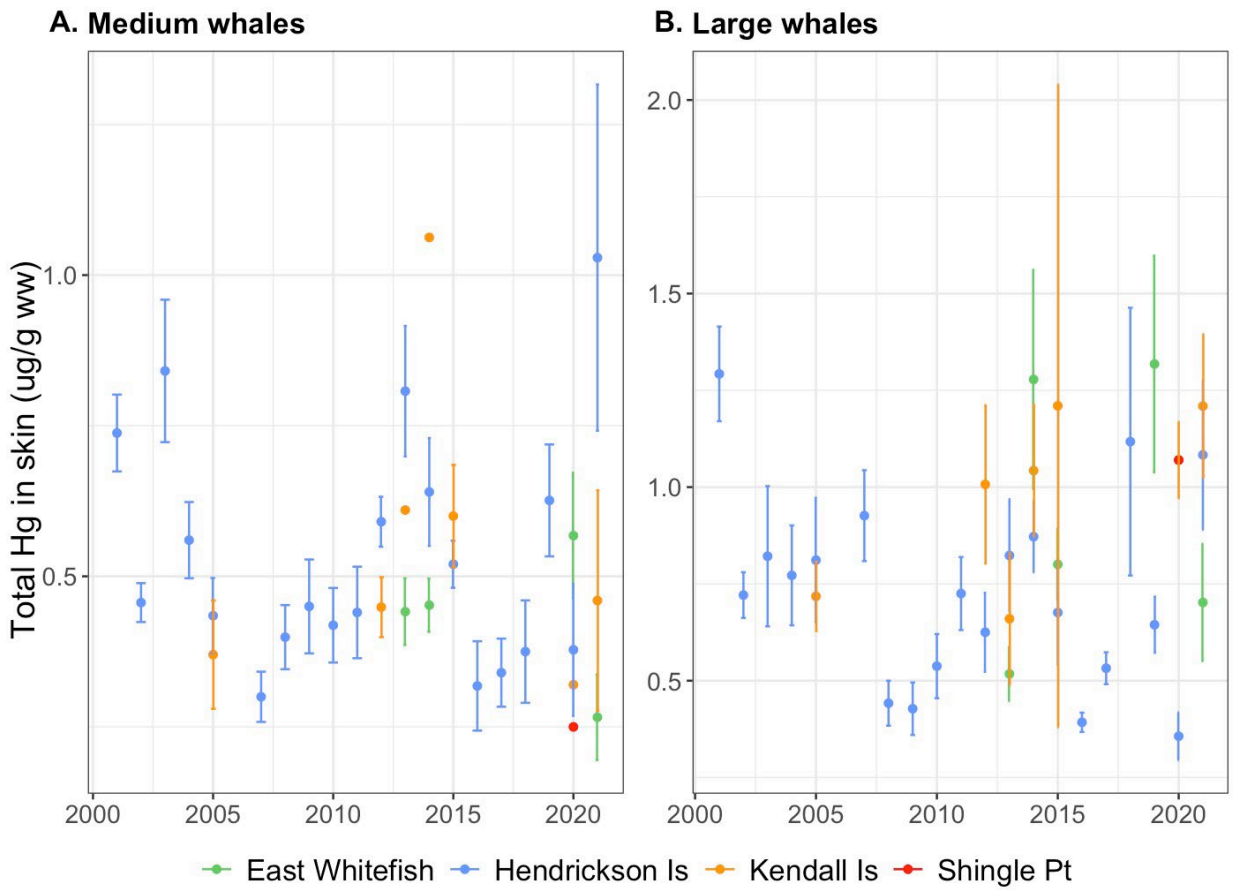
Appendix 1. THg levels in skin and muscle of EBS male beluga whales sampled between 2001 and 2021 at four sampling stations (THg skin = 1.67 THg muscle + 0.44; R = 0.58; p < 0.05).



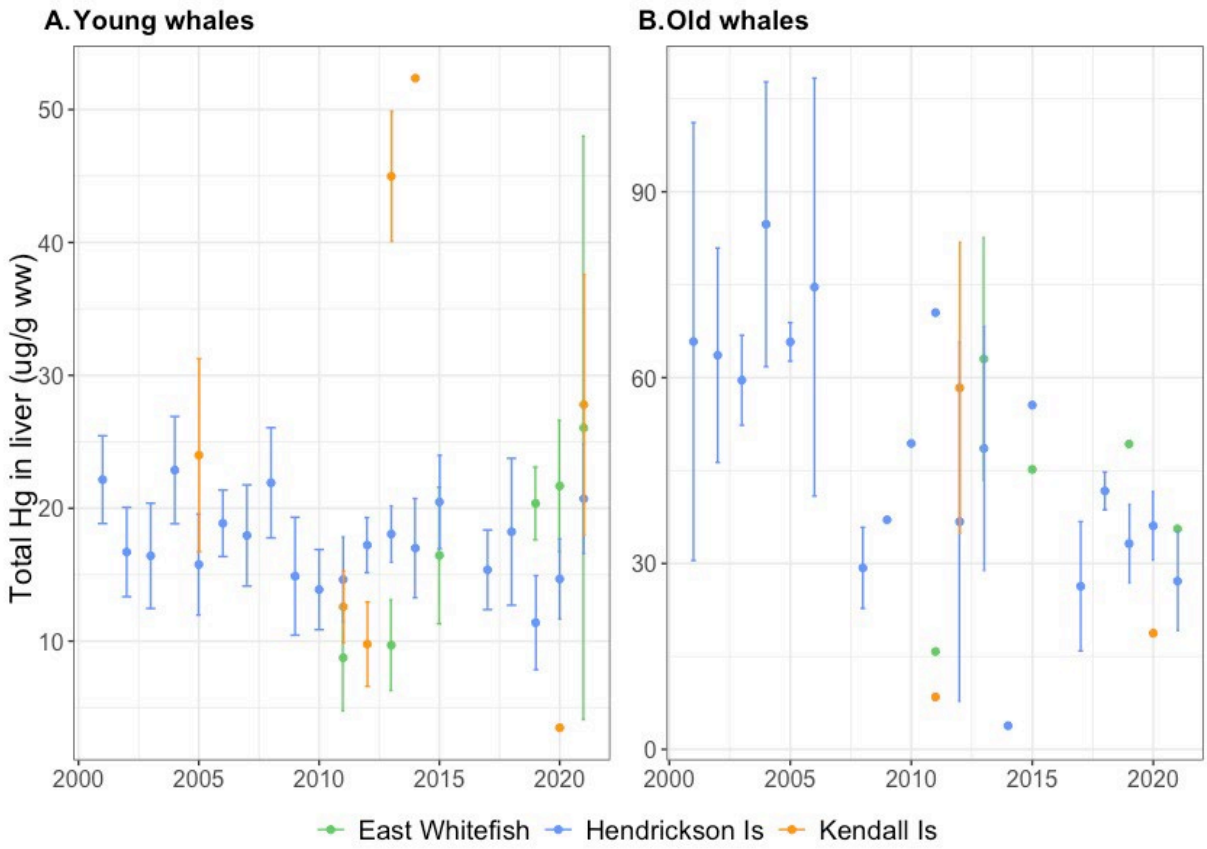
Appendix 2. Relationship between THg levels in the liver of EBS male beluga whales and age for each sampling year separately (all sampling camps combined). Note: No age data was available for whales sampled in 2016.



Appendix 3. THg in the muscle of medium (>= 380 cm and <= 420 cm) (A) and large (>420 cm) (B) EBS male belugas whales collected between 2001 and 2021 at four different sampling camps.



Appendix 4. THg in the skin of medium (>= 380 cm and <= 420 cm) (A) and large (>420 cm) (B) EBS male belugas whales collected between 2001 and 2021 at four different sampling camps.



Appendix 5. THg in the liver of young (> 15 years and <= 35 years) (A) and old (>35 years) (B) EBS male belugas whales collected between 2001 and 2021 at three different sampling camps (There were no whales from the Shingle pt camp as part of this dataset).

Appendix 6. Percentage of young (> 15 years and <= 35 years) EBS male beluga whales sampled between 2001 and 2021 in each risk category associated with Hg exposure.

Year	Risk	n	%
2001	Moderate Risk	1	7.1
	Low Risk	9	64.3
	No Risk	4	28.6
2002	Low Risk	6	37.5
	No Risk	10	62.5
2003	Low Risk	4	40.0
	No Risk	6	60.0
2004	Low Risk	9	52.9
	No Risk	8	47.1
2005	Moderate Risk	1	4.8
	Low Risk	9	42.9
	No Risk	11	52.4
2006	Moderate Risk	1	3.7
	Low Risk	11	40.7
	No Risk	15	55.6
2007	High Risk	1	6.3
	Moderate Risk	2	12.5
	Low Risk	5	31.3
	No Risk	8	50.0
2008	Low Risk	8	66.7
	No Risk	4	33.3
2009	Low Risk	4	36.4

	No Risk	7	63.6
2010	Low Risk	7	41.2
	No Risk	10	58.8
2011	High Risk	2	6.3
	Moderate Risk	2	6.3
	Low Risk	9	28.1
	No Risk	19	59.4
2012	High Risk	1	3.4
	Moderate Risk	2	6.9
	Low Risk	12	41.4
	No Risk	14	48.3
2013	High Risk	1	3.0
	Moderate Risk	1	3.0
	Low Risk	16	48.5
	No Risk	15	45.5
2014	High Risk	1	6.3
	Low Risk	8	50.0
	No Risk	7	43.8
2015	Moderate Risk	1	3.8
	Low Risk	14	53.8
	No Risk	11	42.3
2017	Low Risk	6	50.0
	No Risk	6	50.0
2018	Low Risk	2	33.3
	No Risk	4	66.7
2019	Low Risk	3	30.0

	No Risk	7	70.0
	Severe Risk	2	10.5
2020	Low Risk	7	36.8
	No Risk	10	52.6
	Severe Risk	1	5.9
2021	Low Risk	9	52.9
	No Risk	7	41.2

Appendix 7. Percentage of old (>35 years) EBS male beluga whales sampled between 2001 and 2021 in each health risk category associated with Hg exposure

Year	Risk	n	%
2001	Severe Risk	3	60.0
	Low Risk	1	20.0
	No Risk	1	20.0
2002	High Risk	3	60.0
	Low Risk	2	40.0
2003	Severe Risk	1	12.5
	Moderate Risk	3	37.5
	Low Risk	4	50.0
2004	High Risk	2	66.7
	Low Risk	1	33.3
2005	Moderate Risk	2	66.7
	Low Risk	1	33.3
2006	High Risk	1	50.0
	Low Risk	1	50.0
2008	Low Risk	5	71.4
	No Risk	2	28.6
2009	Low Risk	1	100.0
2010	Low Risk	1	100.0
2011	Moderate Risk	1	33.3
	No Risk	2	66.7
2012	High Risk	1	20.0
	Moderate Risk	1	20.0
	Low Risk	2	40.0

	No Risk	1	20.0
2013	High Risk	2	33.3
	Moderate Risk	1	16.7
	Low Risk	3	50.0
2014	No Risk	2	100.0
2015	Low Risk	3	100.0
2017	Low Risk	1	50.0
	No Risk	1	50.0
2018	Low Risk	4	100.0
2019	Low Risk	6	85.7
	No Risk	1	14.3
2020	Severe Risk	1	20.0
	Low Risk	4	80.0
2021	Low Risk	3	75.0
	No Risk	1	25.0