

OESG Scientific Report

This document is the draft scientific report of the Open-Ended Scientific Group (OESG) of the Minamata Convention, as of 26 January 2026. The executive summary of this draft has been made available to the Effectiveness Evaluation Group as UNEP/MC/EEG.8/L.7. The OESG plans to further work on the draft and make it available for Party input in February 2026.

0 Executive Summary

Article 22 of the Minamata Convention commits the Conference of the Parties (COP) to evaluate the effectiveness of the Convention, beginning no later than six years after the Convention's entry into force and periodically thereafter at intervals to be decided by the COP.¹ The effectiveness evaluation is to be conducted on the basis of, among other things, available information on the presence and movement of mercury and mercury compounds in the environment, as well as trends in levels of mercury and mercury compounds observed in biotic media and vulnerable populations.

The main purpose of the effectiveness evaluation framework is to assess whether the Parties' collective implementation of the commitments in the Convention is, or is likely to, achieve the objectives of the Convention. To make this assessment, the COP must address the following questions:

1. Have the Parties taken actions to implement the Minamata Convention?
2. Have the actions taken resulted in changes in mercury supply, use, emissions and releases into the environment?
3. Have those changes resulted in changes in levels of mercury in the environment, biotic media and vulnerable populations that can be attributed to the Minamata Convention?
4. To what extent are existing measures under the Minamata Convention meeting the objective of protecting human health and the environment from mercury?

To consider these questions, the COP has created two bodies: the Open-Ended Science Group (OESG) and the Effectiveness Evaluation Group (EEG). The OESG is charged with drafting a scientific report that assesses how changes in emissions and releases attributed to the Minamata Convention have contributed to observed changes in the levels of mercury in the environment and in exposures to biota and human populations (i.e., answering question 3 above). The OESG is also charged with identifying existing data gaps as well as potential scientific actions to address the identified gaps in information and lessons learned for future effectiveness evaluations. The OESG is to submit its scientific report and a summary of lessons learned to the EEG. In addition to the OESG's reports, the EEG will receive other information developed by the Secretariat concerning implementation of the Convention (based on Article 21 reporting) and changes in the global supply, use, and trade of mercury. The EEG is charged with synthesizing the available information into a final report and presenting recommendations for improvement, lessons learned and best practices to the Conference of the Parties. This document is a draft of the scientific report of the OESG.

The OESG is composed of scientific experts nominated by Parties. It is "open-ended" in that the number of Parties who can participate is not limited; each Party may nominate one relevant expert to participate. Furthermore, Parties and stakeholder organizations may nominate additional individuals with relevant expertise to serve on a Roster of Experts, who may contribute to the technical work and discussions of the OESG. Currently, the OESG has 41 members nominated by Parties and a roster of 146 experts. In contrast, the EEG is a regionally representative body with 25 members, comprised of 5 individuals nominated from each of the 5 UN regions.

In accordance with its terms of reference, the OESG conducted its work in two stages:

¹ The Minamata Convention entered into force on 16 August 2017.

- *Stage 1: Planning*
 - o Plan for the compilation and summary of available monitoring data
 - o Plan for the summary of available emissions and releases data
 - o Plan for data analysis consistent with the Monitoring Guidance²

- *Stage 2: Implementation*
 - o Summary of the compiled monitoring data
 - o Summary of the available emissions and releases data
 - o Data analysis addressing the guiding questions outlined in the Monitoring Guidance

OESG's plans for data collection and analysis were circulated to Parties prior to COP-5 and documented in [UNEP/MC/COP.5/INF/24](#).

The OESG plan for collecting available monitoring data and estimates of emissions and releases provided

- Descriptions of the types of data and information that the OESG was seeking to inform their analyses. The data dictionary included in the plan evolved into the recommended data formats for each type of data to be submitted.
- List of known mercury monitoring programs and other sources of relevant environmental and human data, which provided a starting point for OESG outreach to data providers
- Principles that would be used by the OESG to guide its collection and analysis of data, which led to the development of data use agreements and practices.

Following this plan, the OESG began collecting data in September 2023 by inviting national focal points and observer organizations to identify relevant data. The Secretariat, with the support of the Biodiversity Research Institute, established an online data repository where data submissions could be collected for use by the OESG. Some data was submitted by Parties or research organizations, and some data was actively acquired by OESG and rostered experts. The OESG submitted an overview of monitoring, emission and release data to COP-6 ([UNEP/MC/COP.6/INF/23](#))

The amount of data collected and analyzed has been limited by the time and resources available to support the data collection process, the lack of harmonized data reporting formats and existing infrastructure for data management, and limited awareness of Convention activities in the relevant national and international scientific communities.

The OESG's terms of reference specify that the OESG shall analyze the available information to address the "guiding questions" outlined in the Monitoring Guidance.³ The Monitoring Guidance outlines a series of six monitoring objectives:

- Estimation of mercury concentrations for areas without (i.e., background sites) or with (i.e., affected sites) local anthropogenic sources
- Identification of temporal trends
- Characterization of spatial patterns
- Estimation of source attribution of anthropogenic mercury
- Estimation of exposure and adverse impacts

The OESG developed a series of analysis questions to systematically address each of these themes. The OESG found that some of these analysis questions could be addressed by the available information, but many of the questions will require additional monitoring and model development. The main findings from this systematic analysis are presented below.

Key Findings from the Analysis Questions

² The "Monitoring Guidance" referenced in decision MC-4/11 is the "Guidance on monitoring of mercury and mercury compounds to support evaluation of the effectiveness of the Minamata Convention" available as document [UNEP/MC/COP.4/18/Add.2](#).

³ Available as document [UNEP/MC/COP.4/INF/12](#)

Emissions

Available estimates of Hg air emissions show an increase in global emissions between 2010 and 2022, driven largely by an increase in emissions from artisanal scale gold mining (ASGM), which are believed to be under reported (EDGAR; Cheng et al., 2023; supported by GMA based on the 2019 National Inventory reports). Estimates from EDGAR (Emission Database for Global Atmospheric Research) suggest that global Hg emissions peaked in 2018 at 1.98 Gg/year. 2022 annual global anthropogenic mercury emissions are estimated to be 1.86 Gg/year, with gold mining accounting for 50% of annual emissions.

The regional distribution of emissions has shifted over time. Emissions have declined in Europe, North America, Japan, Australia, New Zealand, Eastern Europe, and West-Central Asia. Emissions have increased in Africa, Eastern Asia, Latin America and the Caribbean, South-East Asia and Pacific, and Southern Asia. Sixty eight percent of current emissions now emanate from Africa, Eastern Asia, and Latin America, where ASGM is dominant.

Global air emissions from non-ASGM sources have generally decreased since 2008 (or 2010), with individual sectors showing slight decreases or almost constant levels over the decades, especially in recent years. This result may indicate that the level of mitigation that has been achieved in specific sectors is different in different regions.

Globally, total Hg (THg) air emissions from chlor alkali production, which is addressed by Article 5 of the Convention, have decreased dramatically, declining 95% since 1970 and 49% since 2010. Coal extraction and combustion accounted for 31% of emissions in 1970 and has declined to 18% of emissions in 2022.

Between 2010 and 2022, major air emission sources (under Article 8) have changed little globally, declining only 2%. While emissions from coal combustion decreased by 21%, emissions from metal production and cement production increased by 18% and 26% respectively.

Gold mining sources under Article 7 have increased by 49%. Chlor Alkali production emissions (under Article 5) have declined by 49%. Sources associated with mercury use in products (Article 4) and other manufacturing processes (Article 5) appear to have increased moderately.

Releases

There are far more data available on Hg emissions (THg and Hg speciation) to air than for Hg releases to land and water (as noted by the pollutant release and transfer registers, PRTR). Furthermore, there are significant data gaps over the reporting period for both releases to water and land. Modelling of anthropogenic releases to land and water can be of help in narrowing or addressing this gap.

Releases to water and land are estimated using several different methods, including estimating releases from source category-specific air emissions estimates, source-category specific activity rates, and regional Hg consumption patterns. It should be noted that release estimates are highly uncertain.

ASGM is a major global source of mercury, but current data on water releases are too uncertain to be included in existing estimates. ASGM activity is highest in Africa, Latin America, and Southeast Asia.

For source categories for which global estimates are available, the highest releases to water are associated with municipal wastewater (40 to 460 tonnes per year), coal fired power plants (10-130 t/yr), coal washing (10-100 t/yr), and mercury containing products (~ 100 t/yr). The estimated releases from these sources are highest in Asia.

Air

3 Current levels and geographical coverage:

The levels of mercury observed in the ambient air for total gaseous mercury (TGM) and gaseous elemental mercury (GEM) vary around the world and over time. Globally, TGM (active sampling) levels are 1.51 (0.49-12.2) ng m⁻³. There are distinct hemispherical differences whereby the northern hemispheric (NH) average concentration is 1.54 (0.50 – 12.8) ng m⁻³ and the southern hemispheric (SH) average concentration is 0.99 (0.33-4.51) ng m⁻³. For GEM (passive sampling⁴) the global average concentration is 1.46 (0.94-2.62) ng m⁻³. The NH average concentration is 1.48 (1.06-2.60) ng m⁻³, the SH average is 1.34 (0.94-2.62)-2.56) ng m⁻³.

⁴ Several sites submitted for the effectiveness evaluation reported GEM concentrations significantly higher than the global average. To avoid skewing the results, any site found to have a site average 5 times greater than standard deviation of the global average was classified as outliers and excluded from subsequent statistical analysis. Applying this condition resulted in the removal of eight sites from the hemispheric and regional analyses.

For active air sampling of TGM, most of the levels of mercury fall within the northern or southern hemispheric averages except for East Asia where levels are higher. The passive Hg data has more spatial coverage than the active data and thus shows a similar pattern but different levels. For the passive air sampling data, areas in South America, Africa and East Asia report above average concentration levels in comparison to the hemispherical averages. The West/Central African levels are very high (mean = 14.48 ng m⁻³).

The overall “typical” concentration of mercury in precipitation is between 7.5 and 11.5 ng Hg L⁻¹ based upon the submitted long-term measurements in North America, Japan and Europe. Dry, western North America has the highest typical concentration of 11-12 ng Hg L⁻¹. Eastern North America, Europe, and Japan have similarly wetter climates and have a typical concentration of 7.5 ng Hg L⁻¹. These values have been consistent since 2010, varying by only 2 to 3 ng Hg L⁻¹ over the years. Some of this variability is due to precipitation amount variability between the years; wetter (drier) conditions lead to lower (higher) concentrations when the mercury content does not change. Measurements suggest there is a slight (not significant) tendency for decreasing concentrations over all sites. Despite regional similarities, individual sites themselves can exhibit considerable variability.

Wet deposition measurements of Hg are missing from Asia, Africa and South America and there are no consistent measurements available over the oceans. However, given what we know about the behavior of Hg in the atmosphere and limited measurements reported in the scientific literature, it is reasonable to assume the concentration and deposition rates in the drier regions of Asia and Africa are similar to western North American rates. This is also likely true in South America. The moister regions of the landmasses are likely to have concentration and deposition rates similar to those measured in Europe, Japan, and Eastern North America. Local areas with large and local sources of mercury, with abundant rain will likely have the highest deposition rates (e.g., eastern China, Indonesia).

4 Time trend and gaps:

Temporal trends for active TGM measurements were estimated for 77 of the sites with at least 5 years of data. The trends calculated are for all years of data that were collected and thus variability is expected. Of the trends reported, 49% show a decreasing trend, 49% show no trend and 1% show an increasing trend. It should be noted that the data from these types of measurements are mostly from northern hemispherical locations. Temporal trends were not calculated with the passive GEM data as there is not yet sufficient data for a statistically significant analysis.

For precipitation, temporal trends were estimated for precipitation weighted mean concentrations at 132 sites. 80% of sites show no statistically significant change, about 20% of sites show a decrease in concentration and two sites show an increase in concentration. Trends in overall wet deposition of Hg may be driven by changes in both Hg concentration and precipitation amounts. Approximately 87% of sites show no significant temporal trend in wet deposition of mercury. In Asia, no trends were found in the 2 sites in Japan for wet deposition. Five sites situated along the US/Canada border and one site in western North America show significant increases in wet deposition of Hg. Several of these sites have increasing precipitation rates, suggesting more precipitation is driving some of these higher deposition rates.

In general, most of the data presented are from background locations. However, over the years, there are some areas that stand out with elevated levels including parts of Asia, South America, Western Africa and the general tropics areas. At times at some of the locations, ambient levels of mercury exceeded established health guidance levels. These locations are in areas of known ASGM activities.

While the data presented in this report appear, at times, to be a fulsome data set, there are significant gaps in global air monitoring especially in areas of significantly high levels. Active air and wet deposition monitoring information reflects primarily the northern hemisphere. The passive air monitoring has enabled a preliminary look at levels in other areas of the world that are elevated above global/northern hemispherical averages.

Biota

5 What are current levels of Hg observed in biota?

- Current levels were defined as total mercury measured between 2014 and 2024. Only a small set of the submitted data (31%) represents samples collected during the last decade.
- The majority of the submitted biota data did not reliably support differentiation between sites directly impacted by mercury sources and those that were not. This finding is due to limited information from the submissions.
- The most frequently sampled matrices were fish (muscle), followed by birds (feathers, blood and eggs), invertebrates (soft tissues), and mammals (liver and muscle). Mercury

concentrations varied widely, often spanning several orders of magnitude depending on tissues and species.

- ✧ Across regions and species, THg concentrations in bivalve soft tissues ranged over two orders of magnitude from 10.2 to 2,070 ng g⁻¹ d.w.
- ✧ Across regions and species, THg concentrations in fish muscle samples ranged over four orders of magnitude from <0.1 to 16,400 ng g⁻¹ w.w.
- ✧ Across regions and species, THg ranged from <0.1-6,000 ng g⁻¹ d.w and 2-72,790 ng g⁻¹ d.w in bird eggs and feathers, respectively.
- ✧ Across regions and species, THg concentrations in marine mammal individuals ranged over several orders of magnitude: from 16.5 to 5,962,000 ng g⁻¹ d.w. in liver and 3-472,000 ng g⁻¹ d.w. in muscle.
- The biota data are not suitable for global-scale spatial pattern analysis due to limited geographic coverage and broad variability on the species and trophic position among different regions. Spatial comparisons may be more feasible at regional levels where the same species are well sampled such as in North America, Europe or the Arctic.
- Data coverage is uneven: intensive monitoring exists in North America, Europe and East Asia, but large gaps remain in Africa (AF), Latin America and the Caribbean (LAC), South Asia, and parts of the Pacific. Monitoring in artisanal and small-scale gold mining (ASGM) regions is critical, as these are major mercury emission hotspots.

Greater use of knowledge, sciences and practices of Indigenous Peoples will strengthen the relevance, inclusivity, and cultural appropriateness of monitoring programs.

6 How do current levels of Hg observed in biota compare to levels in established guidelines?

• Invertebrates:

- ✧ Observed total mercury concentrations were compared with OSPAR background assessment concentrations for the Northeast Atlantic region and the EU's Maximum Permissible Concentration in food for the protection of public health by the European Commission (1881/2006/EC) to determine the proportion of individuals exceeding these thresholds from 2014–2023.
- ✧ 65% of all oyster observations and 59% of all mussel observations were above the background assessment concentrations defined by OSPAR Commission for the Northeast Atlantic region, but none surpassed food safety limits, indicating low human health risk (500 ng g⁻¹ w.w).

• Fish:

- ✧ Toxicity benchmarks were selected by choosing Toxicity Reference Values (TRVs) that were mathematically derived using a benchmark dose analysis framework when available, as recommended by the U.S. Environmental Protection Agency and European Food Safety Authority.
- ✧ When mathematically derived TRVs were not available, the best available benchmarks from recent peer-reviewed review papers were selected.
- ✧ By comparing observed fish mercury concentrations against these TRVs, we assessed the likelihood of adverse effects on endpoints such as reproductive success, behavior, growth, and survival, thereby providing a consistent framework for evaluating ecological health risks across regions and taxa.
- ✧ Mercury contamination is widespread, with evidence of reproductive, behavioral, growth and survival risks across multiple regions, including regions of Africa, Asia, Australia, Europe, and North and South America.
- ✧ Higher level TRVs for behavioral, growth, and survival effects were exceeded less frequently but remained substantial in hotspots such as the North Atlantic, Pacific Ocean, and southeastern Africa, particularly where shark species dominated samples.

• Birds:

- ✧ Toxicity benchmarks were selected by choosing TRVs that were mathematically derived using a benchmark dose analysis framework when available, as recommended by the U.S. Environmental Protection Agency and European Food Safety Authority. When mathematically derived TRVs were not available, the best available benchmarks were selected from the specific recent review papers.
- ✧ Egg THg concentrations were largely below a toxicity reference value (TRV) of $0.7 \mu\text{g g}^{-1} \text{ w.w.}$
- ✧ Nearly 35% of feather THg concentrations (d.w.) were above a toxicity reference value at no apparent effect category (Chastel et al., 2022). Nearly 23% (2,872 of 12,123) were at low risk, while 6% (731), 1 % (128) and 3.68% (447) were at median, high and severe risk categories, respectively.
- ✧ Across all IPCC regions, more than half of adult bird blood samples (54.4%) exceeded the EC01 threshold for low injury ($0.09 \mu\text{g g}^{-1} \text{ w.w.}$), with 14.6% surpassing the EC05 benchmark for moderate injury ($0.6 \mu\text{g g}^{-1} \text{ w.w.}$). A further 9.6% exceeded the EC10 high injury threshold ($1.3 \mu\text{g g}^{-1} \text{ w.w.}$), and 1.2% of samples were above the EC20 level associated with severe injury ($3.2 \mu\text{g g}^{-1} \text{ w.w.}$).
- **Marine mammals:**
 - To evaluate the risk of mercury toxicity in marine mammals, total mercury concentrations in liver were compared to the four risk categories: $\leq 16 \mu\text{g g}^{-1} \text{ w.w.}$ (No risk), $16\text{--}83 \mu\text{g g}^{-1} \text{ w.w.}$ (Low risk), $83\text{--}126 \mu\text{g g}^{-1} \text{ w.w.}$ (High risk), and $\geq 126 \mu\text{g g}^{-1} \text{ w.w.}$ (Severe risk). Risk categories for adverse effects of mercury on the reproduction, physiology, condition, and behaviour of mammals have been developed by Dietz et al. (2022).
 - Marine mammals showed widespread mercury exposure: 53% of species (8 out of 15 species) in the 2014–2024 dataset had individuals above the $\geq 16 \mu\text{g g}^{-1} \text{ w.w.}$ risk threshold, and 40% (6 out of 15 species) had individuals in the high to severe risk categories. The species most consistently exceeding these benchmarks include beluga, Blainville’s beaked whale, false killer whale, grey seal, long finned pilot whale, polar bear, ringed seal, and strap toothed beaked whale, predators positioned high in marine food webs where mercury strongly bioaccumulates.

7 How have levels of Hg in biota changed over the available record?

Across regional and local case studies (AMAP, Beluga Case Study, OSPAR/HELCOM, NOAA Mussel Watch, Great Lakes, Fennoscandia, Australian Derwent Estuary Program, Common Loon blood mercury study, Japan’s case study), mercury in biota exhibits stable, increasing, and decreasing trends, driven by a complex interplay of local ecology, food-web dynamics, species traits, climate influences, legacy contamination, biogeochemistry, and hydrodynamics. While mercury emission reductions provide benefits, biotic responses are often slow, uneven, and masked by legacy stores and ecological factors. Exceedances of health and environmental thresholds remain common in some systems.

Humans

8 Current exposure:

The first findings from the cross-sectional studies review show that the highest Hg non-occupational exposures were observed for Indigenous freshwater (inland) fish consumers, which are mostly represented by South American riverside communities. Among them, 86 % population groups were reported to live on ASGM contaminated sites, and their average Hg blood or hair levels were 12 times higher than those of the other riverside Indigenous communities and 13 times higher than in the general population.

Occupational exposures to Hg have been observed at levels 6 times higher than in the general population. Some of the highest exposures have been observed in mercury and gold mining, chlorine production, and thermometer and fluorescent lamp manufacturing.

9 Geographical patterns:

Due to ongoing ASGM activities affecting wider regions inhabited by fish consuming communities, current human Hg exposures appear to be highest in South America. Among the nationally representative HBM programs, Japan was recently reported to have 3 times higher exposure in adults than the Republic of Korea, and approximately 6 times higher than other available national programs. Across the general (background) population, the exposure is generally higher in populations residing in coastal regions than inland, which is most plausibly driven by dietary habits, particularly seafood and fish consumption.

10 Time trends and interventions:

Generally, a stable or slightly decreasing trend in Hg exposure can be observed in the last two decades from the available nationally representative data. Regional temporal data is available for pregnant women from the Arctic region of Nunavik and showed marked decrease in Hg exposure from year 2000. However, further analysis of time trends needs to be done using all data from cross-sectional studies to provide a more complete assessment of temporal exposures over different population groups and regions. This will be completed by June 2026.

We have no evidence that the decline in Hg exposure can be directly attributed to the mitigation measures influenced by the Convention. However, there have been case studies conducted regionally or locally that demonstrate decline in exposure following certain measures or interventions, including:

- the phase-down of dental amalgam use has resulted in measurable reductions in children's Hg body burden in Slovenia from 2007 to 2024
- dietary recommendations and interventions to improve access to low-contaminant foods and essential nutrients have resulted in decreasing burdens of Hg and POPs in some Arctic regions
- dietary intervention studies in coastal fish-consuming countries in Europe were successful in lowering the Hg exposures in pregnant women below health-based guidance values.

Other Matrices

Soil

The available data was very sparse except for the USA and Australia, and the available datasets covered different periods, making comparisons and interpretations very difficult. No sites were sampled over time which prevented any evaluation of temporal trends and no mono methyl mercury (MMHg) data was available.

A clear difference between background and anthropogenic levels was difficult to establish with the available data due to the intrinsically high spatial variability of soil THg.

THg levels in soils seem to be regulated by the presence of soil organic matter on the surface and/or the presence of metallic oxides (e.g., iron or manganese oxides) at depth but the relationships observed can strongly vary.

Sediment

Global lake sediment records show that Hg concentrations and fluxes were low and comparable between unimpacted and impacted sites during the pre-industrial period, but diverged thereafter. Unimpacted/remote sites showed steady increases to 2010–2020 while impacted sites collectively peaked around 1990–2000 and declined thereafter, particularly in the IPCC regions of Northern Europe, Eastern North America, Northeast North America, South-Central America, and Northwest-South America.

Flux based enrichment factors (EFs) were used to assess the magnitude of Hg enrichment over various time periods and showed that, as expected and consistent with prior literature, modern (2010-2020) EFs are significantly higher in impacted regions (~7.3) than in unimpacted regions (~3.6). In unimpacted regions, highest modern EFs are observed in Northern Europe, the Tibetan Plateau, and the Arctic while the lowest EFs are observed in South and Southwest-South America.

In unimpacted regions and during some early periods in impacted regions, Hg concentrations were positively correlated with organic matter. This is indicative of the well-established co-transport of Hg and organic matter to lake sediments.

In both impacted and unimpacted regions, Hg fluxes were strongly correlated with aluminum, reflecting the dominant role of sedimentation rates, particularly the delivery of mineral rich catchment material, in controlling sediment derived Hg depositional fluxes.

While catchment:lake area ratios were not a strong predictor of enrichment across the global dataset, watershed development showed a clear influence. Pristine catchments maintained low EFs (<3) over the entire temporal record and minimally to heavily developed catchments exhibited progressively higher and more sustained Hg enrichment (median EFs ~ 5–9 by 2000–2010).

Fresh Water

Freshwater data has limited spatial coverage and is mostly from the Northern Hemisphere. Most data are site-specific or local. Inadequate spatial coverage makes it difficult to assess regional and global patterns. All of the submissions included total Hg and none reported methyl Hg.

The usability of the submitted data for trend analysis is limited due to analytical uncertainties and inconsistent methodologies. Many of the datasets with higher analytical detection limits and data were censored at the reporting limit.

Few multidecadal scientific monitoring studies exist on total Hg and methyl Hg in freshwater systems away from point sources, restricting quantitative assessments of change over time.

A national study of rivers in the Northern Hemisphere highlighted shifting trends in Hg concentrations with most recent increases.

In rivers (lotic systems), Hg variability is driven by flow and particle transport, and its mobility is closely linked to dissolved organic carbon (DOC), especially during high discharge.

In lakes (lentic systems), Hg and methyl Hg are influenced by sediment interactions, seasonal redox conditions, lake water level oscillations, or upstream inflow, with DOC playing a stabilizing role.

Ocean Water

Ocean THg and methyl mercury (MeHg) water column data was available in 62% and 42% of the 55 IPCC open ocean and coastal regions, respectively.

There was no water column Hg monitoring data available and data from the scientific literature is not sufficient to conduct an evaluation of temporal trends.

For both THg and MeHg, the highest concentrations in the surface ocean were found in Polar Regions and the lowest concentrations were found in the South Pacific Ocean.

For both THg and MeHg, the largest difference between coastal and open ocean concentrations were found in midlatitudes and equatorial regions, while little difference was seen for the Polar regions.

The %MeHg in the water varied between 2 and 15% for the surface ocean with a similar range found in the open ocean and coastal ocean. However, there is an indication of lower %MeHg in the southern hemisphere open ocean.

THg concentrations correlated with salinity outside of the Polar regions while no correlation was found between MeHg and salinity. Low salinity (coastal) Hg concentrations varied by several orders of magnitude.

Better data coverage including more coastal regions, sampling method development and more laboratory/method intercomparison studies, better collaboration and use of passive samplers, and an official database extending the current data compilation and including more ancillary data are needed.

Cross Media Comparisons of Observed Trends and Patterns

The Northern Hemispheric average concentration is higher than the Southern Hemispheric average concentration, which is consistent with historical anthropogenic emission patterns. In recent decades, anthropogenic emissions in much of the Northern Hemisphere have declined or stayed constant. This trend in emissions is broadly consistent with the trends observed by active TGM measurements, which are mostly in the Northern Hemisphere. Emissions in the Southern Hemisphere, mostly associated with ASGM, have been increasing. Passive air samplers have been used recently to observe high air concentrations in ASGM areas. However, an increase in the Southern Hemispheric average air concentrations has not been observed at the limited number of sites where long-term observations are available. Additional sustained observations in the Southern Hemisphere are needed.

Given available data, it is unlikely that quantitative conclusions will be drawn from the direct cross-media comparison at broad spatial scales. However, several regional case studies provide information on the linkages between media in the specific contexts of the cases studied. Those studies include:

- Spatial and temporal trends and the distribution observed in tropical tuna in global ocean relate to the spatial/temporal trends of cross media levels and emissions.
- The Mediterranean case study showed holistic datasets in the region, which could be further analysed for cross media comparison.
- The impacts of long-term national efforts on the emission reduction and environmental monitoring around historical contamination sites would be the basis of further analysis of cross media comparison after the Convention.

- Analysis of ASGM sites and observation in East China Sea would also be the basis of further analysis of cross media comparison after the Convention.

Model Evaluation and Process and Source Attribution for Air and Oceans

Quantitative information about the processes and sources driving trends and patterns in global atmosphere and oceans is available from the ongoing Multi-Compartment Mercury Modelling and Analysis Project (MCHgMAP), including comparison of model estimates and observed levels.

Model simulations generally reproduce observed spatial patterns and deposition, but uncertainties in emission inventories – particularly for ASGM, waste, and regional sources – cause discrepancies with observed trends.

Global mercury (Hg) cycling is dominated by legacy re-emissions from soils and oceans (~70% of annual fluxes), with ongoing anthropogenic emissions contributing ~25%. Atmospheric Hg shows elevated concentrations over industrial and artisanal mining regions, with deposition highest in tropical forests and northern mid-latitudes. Observed declines in the Northern Hemisphere largely reflect reductions in anthropogenic emissions, indicating early effectiveness of the Minamata Convention, while ASGM-driven trends dominate the Southern Hemisphere. Model simulations generally reproduce observed spatial patterns and deposition, but uncertainties in emission inventories – particularly for ASGM, waste, and regional sources – cause discrepancies with observed trends. Ocean Hg levels are largely stable, with upwelling regions, polar areas, and the Northern Hemisphere showing highest concentrations; atmospheric inputs strongly control surface ocean Hg. Full Minamata implementation could reduce global anthropogenic Hg emissions by ~18% by 2050.

Key findings on global Hg sources, trends, and future prospects (2010–2020)

- **Primary and secondary sources:** Global natural Hg emissions (~270 Mg/year) come mainly from volcanoes and rock weathering, while biomass burning (~280–600 Mg/year) drives interannual variability. Secondary Hg emissions from soils (~2200 Mg/year) dominate the land–atmosphere exchange and are influenced by soil properties, climate, and vegetation, though uncertainties remain high due to sparse measurements.
- **Atmospheric patterns:** Hg concentrations are elevated over industrial regions and artisanal mining areas, with deposition highest in tropical forests and northern mid-latitudes. Dry deposition dominates over land, wet deposition over oceans. Current monitoring is geographically biased toward the Northern Hemisphere; models fill major gaps, especially in the tropics, Southern Hemisphere, and parts of Asia and Africa.
- **Drivers of trends:** Observed declines in Northern Hemisphere atmospheric elemental mercury (Hg(0)) are largely driven by reductions in anthropogenic emissions, particularly non-ASGM sources, reflecting early Minamata Convention impacts. Environmental drivers had minor effects on Hg(0) trends but modest effects on wet deposition. ASGM emissions dominate trends in the Southern Hemisphere, though attribution is limited by sparse observations and uncertain inventories.
- **Oceanic Hg:** Surface ocean Hg concentrations are highest in upwelling regions, polar areas, and the Northern Hemisphere, strongly influenced by atmospheric inputs. Global levels remained largely stable (2010–2020), with natural variability (e.g., ENSO) driving short-term changes. Ocean Hg acts as a net emitter (300–900 Mg/year) but shows limited temporal trends over the period. Observations are sparse; models provide continuous coverage and reveal features not captured by measurements.
- **Mass balance and re-emissions:** Legacy soil and ocean re-emissions (~70% of annual fluxes) dominate global Hg cycling, while ongoing primary anthropogenic emissions contribute ~25%. Approximately 76–86% of the increase in secondary emissions from soils and oceans (2010–2020) is attributable to changes in anthropogenic emissions. The atmosphere holds <0.5% of total Hg, with soils and the deep ocean as major reservoirs.
- **Future projections:** Under current legislation, global anthropogenic Hg emissions are expected to remain stable, but full implementation of Minamata Convention National Action Plans and co-benefit air pollution controls could reduce emissions by ~18% relative to 2015. Predicted regional trends vary as follows: declines in China, rising emissions in India (non-power sectors), and high uncertainty in ASGM-dominated regions of Africa, South America, and Southeast Asia. Fossil fuel and industrial emissions are relatively well constrained; waste and ASGM emissions remain highly uncertain.

Process and Source Attribution for Biota, Humans, and Other Media

The available observations and models are not sufficient to provide quantitative statements about process and source attribution for biota, humans, and other media.

However, several case studies give information on cross media comparisons in the context of the case studies. Those include:

- A case study on tropical tuna showed non-detectable temporal trends in current decades. Long-term monitoring is necessary to detect the substantial change in tuna levels, in 10 to 25 years.
- Case study on the Great Lakes showed different temporal trends in observations in emission, atmospheric deposition and fish concentrations in monitoring data. Fish levels were rather unchanged in contrast to change of emissions and deposition.

1 Introduction

1.1 Effectiveness Evaluation Framework

Article 22 of the Minamata Convention commits the Conference of the Parties (COP) to evaluate the effectiveness of the Convention, beginning no later than six years after the Convention's entry into force and periodically thereafter at intervals to be decided by the COP.⁵ The effectiveness evaluation is to be conducted on the basis of, among other things, available information on the presence and movement of mercury and mercury compounds in the environment, as well as trends in levels of mercury and mercury compounds observed in biotic media and vulnerable populations.

The COP began discussing the framework for effectiveness evaluation at its first meeting and continued the discussion at each of the following four meetings (through COP-5). A chronological list of the key documents and decisions in the evolution of the framework that emerged is provided in Annex 1. In this section, we present a summary of the key features of the effectiveness evaluation framework that has evolved.

The main purpose of the effectiveness evaluation framework is to assess whether the Parties' collective implementation of the commitments in the Convention is, or is likely to, achieve the objectives of the Convention. To make this assessment, the COP must address the following questions:

1. Have the Parties taken actions to implement the Minamata Convention?
2. Have the actions taken resulted in changes in mercury supply, use, emissions and releases into the environment?
3. Have those changes resulted in changes in levels of mercury in the environment, biotic media and vulnerable populations that can be attributed to the Minamata Convention?
4. To what extent are existing measures under the Minamata Convention meeting the objective of protecting human health and the environment from mercury?

1.2 OESG and EEG Mandates

To consider these questions, the COP has created two bodies: the Open-Ended Science Group (OESG) and the Effectiveness Evaluation Group (EEG). The OESG is charged with drafting a scientific report that assesses how changes in emissions and releases attributed to the Minamata Convention have contributed to observed changes in the levels of mercury in the environment and in exposures to biota and human populations (i.e., answering question 3 above). The OESG is also charged with identifying existing data gaps as well as potential scientific actions to address the identified gaps in information and lessons learned for future effectiveness evaluations. The OESG is to submit its scientific report and a summary of lessons learned to the EEG. In addition to the OESG's reports, the EEG will receive other information developed by the Secretariat concerning implementation of the Convention (based on Article 21 reporting) and changes in the global supply, use, and trade of mercury. The EEG is charged with synthesizing the available information into a final report and presenting recommendations for improvement, lessons learned and best practices to the Conference of the Parties. This document is a draft of the scientific report of the OESG.

The OESG is composed of scientific experts nominated by Parties. It is "open-ended" in that the number of Parties who can participate is not limited; each Party may nominate one relevant expert to participate. Furthermore, Parties and stakeholder organizations may nominate additional individuals with relevant expertise to serve on a Roster of Experts, who may contribute to the technical work and discussions of the OESG. Currently, the OESG has 41 members nominated by Parties and a roster of 146 experts. In contrast, the EEG is a regionally-representative body with 25 members, comprised of 5 individuals nominated from each of the 5 UN regions.

The OESG's work has been conducted primarily through internet meetings and the in-kind contributions of the OESG members and rostered experts. The COP has provided resources for two face-to-face OESG meetings. Of the 41 OESG members nominated by Parties, 5 experts were selected by each UNEP region to lead specific chapters according to their area of expertise. Regionally nominated experts from developing countries and countries with economies in transition receive funding through the budget of the Convention to participate in the face-to-face OESG meetings. In addition, a separate group was contracted to provide support to the secretariat

⁵ The Minamata Convention entered into force on 16 August 2017.

for managing the intake and distribution of the submitted data. The Biodiversity Research Institute (BRI) was contracted for these functions.

The OESG's terms of reference direct the group to conduct its work in two stages: a planning stage and an implementation stage. In each stage, the OESG is expected to circulate draft work products for review and comment by the Parties, providing multiple opportunities for Parties to provide input to the OESG's work. Specifically, the OESG is directed to produce:

- *Stage 1: Planning*
 - Plan for the compilation and summary of available monitoring data
 - Plan for the summary of available emissions and releases data
 - Plan for data analysis consistent with the Monitoring Guidance⁶
- *Stage 2: Implementation*
 - Summary of the compiled monitoring data
 - Summary of the available emissions and releases data
 - Data analysis addressing the guiding questions outlined in the Monitoring Guidance

OESG's plans for data collection and analysis were circulated to Parties prior to COP-5 and documented in [UNEP/MC/COP/5/INF/24](#). The sections below provide overviews of the OESG's processes for data collection and analysis as implemented.

1.3 OESG Internal Structure

Following its terms of reference, the OESG elected two co-chairs, one from a developed country and one from a developing country. To facilitate its work, the OESG organized itself into teams oriented around specific types of data or tasks. Each team is led by an OESG member and composed of OESG members and rostered experts. The first set of teams to be organized focused on the collection and analysis of observational data in different environmental media:

- emissions to air
- releases to land and water
- air concentrations and deposition
- other environmental media (including oceans, freshwaters, sediments, soils, peat, ice, trees, and other vegetation)
- biota (fish, birds, mammals, and invertebrates)
- human populations

Note that methodologies for the collection of observations in air, biota, and humans are addressed in the Monitoring Guidance. The Monitoring Guidance does not address the collection of observations in other media. However, observations in these media provide important information about the fate of mercury in the ambient environment and, particularly, long term trends.

Additional teams were added to focus on:

- integrated analysis of multi-media trends and patterns (looking across media to link trends through focused case studies and global deterministic modeling)
- a vision for future effectiveness evaluation processes (identifying data and capability gaps and opportunities for improvement in future effectiveness evaluation cycles)

The organization of the scientific and lessons learned reports will largely mirror this team structure.

⁶ The "Monitoring Guidance" referenced in decision MC-4/11 is the "Guidance on monitoring of mercury and mercury compounds to support evaluation of the effectiveness of the Minamata Convention" available as document [UNEP/MC/COP.4/18/Add.2](#).

1.4 OESG Data Collection Process

The OESG plan for collecting available monitoring data and estimates of emissions and releases provided

- descriptions of the types of data and information that the OESG was seeking to inform their analyses. The data dictionary included in the plan evolved into the recommended data formats for each type of data to be submitted.
- list of known mercury monitoring programs and other sources of relevant environmental and human data, which provided a starting point for OESG outreach to data providers
- principles that would be used by the OESG to guide its collection and analysis of data, which led to the development of data use agreements and practices.

Following its plan, the OESG began collecting data in September 2023. The Executive Secretary of the Minamata Convention sent a letter to all national focal points and observer organizations inviting them to identify relevant data and express an “intent to submit” data to the OESG process. The letter listed the following types of data: national inventories of mercury emissions to air and releases to land and water and observational data for mercury levels in air, biota (animals and plants), humans, soil, sediments (including sediment cores), fresh water, and ocean water. Potential providers of data were invited to submit a brief description of the dataset, along with data type, time span, geographic scope, and conditions for use of the data. This pre-submission process was automated using on-line forms.

The Secretariat, with the support of the Biodiversity Research Institute, established an online data repository where data submissions could be collected for use by the OESG. Those who submitted an expression of intent were invited to submit data using data format templates developed by the OESG, or alternatively submit data in their own format providing sufficient documentation so that the OESG could compile the data with other submissions. Data providers were also requested to submit a data use authorization form, giving permission to the OESG and participating experts to use the submitted data for purposes of the effectiveness evaluation and indicating any specific data use restrictions. The OESG and participating experts accessing the data submitted to the repository were required to sign a complementary agreement to abide by the data use restrictions identified by data providers. Within the online data repository, a process was established to track contact with potential data providers, record data that was submitted and its associated data use restrictions, identify quality or completeness issues, and track any reformatting or other transformations performed on the submitted data.

The Secretariat received expressions of intent to submit data from 36 individuals from 15 Parties in response to its initial invitation. Expressions of intent were not submitted for many existing data sets that the OESG had identified in their data collection plans. In May 2024, a renewed call for expression of intent was sent to national focal points and observers, and the OESG and participating experts began more actively reaching out to potential data providers and collecting some datasets available through open-access internet sites and published literature. The OESG data collection effort was highlighted at the International Conference on Mercury as a Global Pollutant in Capetown, South Africa, 21-26 July 2024.

The effort to collect available data has been limited by the time and resources available to support the data collection process, the lack of harmonized data reporting formats and existing infrastructure for data management, and limited awareness of Convention activities in the relevant national and international scientific communities. The Convention does not have obligations, procedures, or infrastructure for the reporting or collection of emissions and releases estimates or environmental and human monitoring data. Thus, the OESG had to develop data use policies, standardized data formats, and a data management infrastructure to support the collection process. The Secretariat was able to provide limited resources to support the establishment of a data repository for use by the OESG. However, most of the data sets that have been collected were not submitted or acquired in the recommended format and do not include the full set of recommended variables. Considerable effort has been expended through the in-kind contributions of OESG members and rostered experts to create sufficiently harmonized datasets that are possible to analyze.

The data that has been identified and collected in this process is divided into two categories:

- Received or Acquired: This category includes data that has been submitted to or obtained by the OESG. For each analysis question, the OESG and participating experts will assess each of the datasets in this category in terms of its completeness and quality to determine whether the data is fit for the purpose of addressing the analysis question. The criteria used to determine which data to include in a particular analysis is described for each analysis question.
- Identified: This category comprises data for which the sources have been identified but for which the data is not available for analysis at this time. This category includes data that has been submitted to

or obtained by the OESG, but that is either insufficiently complete or requires reformatting or transformation that is not possible to complete at this time; data that was identified through an expression of intent to submit, but that has not been submitted; or data that has been identified or obtained by the OESG for which OESG has not been granted permission to use the data in their analysis.

Annex X provides links to the online documents developed to implement the data collection process, including the intent to submit form, the data use agreement for the provider, the data use agreement for the analysts, and the data format templates and instructions.

1.5 OESG Data Analysis Process

The OESG's terms of reference specify that the OESG shall analyze the available information to address the "guiding questions" outlined in the Monitoring Guidance.⁷ The Monitoring Guidance, which was drafted in response to Decision 3/10 and presented to COP-4, describes the scientific and technical processes and guiding principles for generating comparable mercury monitoring data with a focus on air, humans and biota. It also suggests methods that can be used to analyse mercury observations in the context of other data to understand the presence, trends, and sources of mercury in the environment and humans to inform the effectiveness evaluation. The Monitoring Guidance outlines a series of six monitoring objectives, each illustrated by one or more questions. The series starts with the most basic objectives and questions and increases in complexity. The six monitoring objectives from the Monitoring Guidance are listed in the first column of Table X.

Recognizing that Parties have different capacities for investment in monitoring programs, the Monitoring Guidance provides recommended monitoring approaches and methods for each environmental media divided into three tiers. *Tier 1* methods are intended to be cost-effective, practical, and feasible for circumstances where resources are limited, and will help inform monitoring objectives 1, 2, and 3. *Tier 2* is intended to build upon tier 1 methods and create a basis for assessing source attribution at the local, national and global scales (monitoring objective 4 above). *Tier 3* identifies research methods and approaches that may play a vital role in supporting the tier 1 and tier 2 programmes and the effectiveness evaluation, primarily by improving understanding of key processes that link sources to environmental concentrations and exposures (monitoring objectives 5 and 6).

The first step in the development of the OESG's data analysis plan was to break down the monitoring objectives and associated guiding questions of the Monitoring Guidance into themes to support the data analysis. For each data analysis theme, the OESG developed analysis questions for each of the monitoring matrices and for emissions and releases. The second and third columns of Table X present the data analysis themes and summary versions of the analysis questions which capture the scope of the individual questions under each theme.

Table X. Mapping of the data analysis themes and summary-level analysis questions to guide the analysis of data by the OESG in relation to the monitoring objectives in the Monitoring Guidance.

Monitoring objectives (from the Monitoring Guidance)	Data analysis themes	Summary-level operational questions
Estimation of mercury concentrations for areas without (i.e., background sites) or with (i.e., affected sites) local anthropogenic sources	Current Levels	What are current levels of Hg emissions and releases and current levels of Hg observed in air, biota, humans, and other media?
Identification of temporal trends	Temporal Trends	How have levels of Hg emissions and releases and Hg observed in air, biota, humans, and other media changed over the available record? How do those changes compare to the timeline of the Minamata Convention?

⁷ Available as document [UNEP/MC/COP.4/INF/12](https://www.unep.org/mtg/COP.4/INF/12).

		What specific mitigation measures have contributed to changes in emissions and releases? How are levels of Hg emissions and releases and Hg observed in air, biota, humans and other media expected to change in the future?
Characterization of spatial patterns	Spatial Patterns	How do current levels and temporal trends vary geographically at the global scale?
Estimation of source attribution of anthropogenic mercury	Source or Process Attribution	What is the fractional contribution of contemporary anthropogenic emissions and releases to current Hg levels observed in air, biota, humans, and other media? How have these contribution levels changed over the timeline of the Minamata Convention? How do the contribution levels and their trends vary geographically at the global scale? How have drivers other than changes in emissions and releases contributed to the trend in observed Hg levels?
Estimation of exposure and adverse impacts	Health and Ecosystem Impacts	How do current levels of Hg observed in air, biota, humans, and other media compare to levels in established guidelines, as well as to observed and projected thresholds for effects to humans, other living organisms and biodiversity based on recent research and knowledge? How do changes in Hg levels over the timeline of the Minamata Convention compare to those guideline levels and effect thresholds?
Quantification of key environmental processes to improve understanding of cause-effect relationships	Process Understanding	How consistent are current levels, temporal trends, and spatial patterns of Hg emissions and releases and Hg levels in air, biota, humans, and other media with estimates from current mechanistic models?

The OESG and participating experts fleshed out their data analysis plan by considering the following ten categories of information for each media-specific data analysis question:

1. *Relevant versus available data* – to determine the type of “relevant and applicable” data that might be used to answer the operational questions, whether the data is accessible in a usable form, and the sources and formats of data readily accessible to the OESG;
2. *Methodological approach* – to identify an appropriate methodological approach for answering the operational question, taking into account the relevant data that are readily accessible to the OESG and whether the approach involves statistical modelling or mechanistic modelling;
3. *Form of the answer/output* – to identify how the answer to the operational question might be expressed (for example, as a map, a time series, a figure, a table of quantitative metrics or a narrative);
4. *Anticipated result* – to describe, based on published literature, what the answers to the operational questions might look like;
5. *Confidence* – to assess, based on the identified methodological approach, the level of confidence in the answer (qualitative or quantitative) to the operational question; to identify quality assurance measures applicable to the data and the analysis method; and to identify the most important sources of uncertainty and potential sources of bias;

6. *Lead responsibility* – to identify who, within the Open-ended Scientific Group or roster of experts, could take primary responsibility for performing the analysis to arrive at an answer to the operational question;
7. *Contributors* – to identify who, within the Open-ended Scientific Group or roster of experts, could contribute to performing the analysis to arrive at an answer to the operational question;
8. *Identified gaps* – to identify gaps in the existing data, knowledge and tools that might prevent analysis of the operational question;
9. *Capacity needs* – to determine the capacity needs to fill the identified gaps;
10. *Indigenous knowledge or traditional knowledge* – to identify Indigenous knowledge and traditional knowledge that might be used in the analysis.

1.6 Organization of Report

To be completed ...

List of authors and contributors to be added.

2 Current Levels, Temporal Trends, and Spatial Patterns

Currently, this chapter is based on the EDGAR emission database. This chapter is to be updated with the Chinese emission inventory from Cui et al (2025).

2.1 Anthropogenic Emissions to Air

2.1.1 What data are available?

There is more data on anthropogenic emissions of mercury to air compared to releases of mercury to land and water. Data available for Hg emissions to air include: Pollutant Release and Transfer Registries (PRTs), the Minamata Initial Assessments (MIA), the Convention on Long-range Transboundary Air Pollution (CLTRAP) database, the Global Mercury Assessment (GMA), and the Emissions Database for Global Atmospheric Research (EDGAR). Of these, EDGAR provides the most robust/holistic data set for trend analysis. However, some countries have detailed inventories that may be more accurate, particularly for specific sectors or source types⁸. EDGAR is a multipurpose, independent, global database of anthropogenic emissions of greenhouse gases and air pollution on Earth. EDGAR provides independent emission estimates compared to what reported by European Member States or by Parties under the United Nations Framework Convention on Climate Change (UNFCCC), using international statistics and a consistent IPCC methodology.

The EDGAR Global Mercury Emissions v8.1, which provides mercury emissions across all countries for the period from 1970 to 2022, is available at https://edgar.jrc.ec.europa.eu/dataset_tox81. It includes total mercury (Hg) and mercury species, i.e. gaseous elemental mercury (Hg₀), gaseous oxidised mercury (Hg₂⁺) and particle-bound mercury (Hg-P). It provides yearly and monthly time series of emissions by sector and country and gridded emissions maps at a 0.1 x 0.1-degree global resolution. These emissions are aggregated according to IPCC 2006 and 1996 classifications and the Minamata recommendations.

The methodology for mercury emissions estimation in EDGAR aligns primarily with the recommendations from the Long-Range Transboundary Air Pollution Convention and the EMEP/EEA guidebook.^{9,10} The mercury emissions are provided for the key emitting anthropogenic sources.¹¹ The trends by IPCC continental region for the period 2010-2020 are shown in Figure 1.

Regions with increasing mercury emissions, such as Africa, Latin America and the Caribbean, and South-East Asia, are characterized by significant growth in the artisanal and small-scale gold mining (ASGM) sector, which contributes substantially to their total emissions. Conversely, regions like North America and Europe, which have reduced emissions, have managed significant reductions in the power generation sector and other industrial activities. The Asia-Pacific Developed and Middle East IPCC continental regions were not included in this analysis because their contributions to the total global mercury emissions are small (<2%).

⁸ For emission from China, data from the following paper are used for data analysis report. Cui, Y., Wu, Q., Wang, S., Liu, K., Li, S., Shi, Z., Ouyang, D., Li, Z., Chen, Q., Lü, C., Xie, F., Tang, Y., Wang, Y., and Hao, J.: Integrating point sources to map anthropogenic atmospheric mercury emissions in China, 1978–2021, Earth Syst. Sci. Data, 17, 3315–3328, <https://doi.org/10.5194/essd-17-3315-2025>, 2025

⁹ Marilena Muntean, Greet Janssens-Maenhout, Shaojie Song, Amanda Giang, Noelle E. Selin, Hui Zhong, Yu Zhao, Jos G.J. Olivier, Diego Guizzardi, Monica Crippa, Edwin Schaaf, Frank Dentener, Evaluating EDGARv4.tox2 speciated mercury emissions ex-post scenarios and their impacts on modelled global and regional wet deposition patterns, Atmospheric Environment, Volume 184, 2018, Pages 56–68, ISSN 1352-2310, [doi:10.1016/j.atmosenv.2018.04.017](https://doi.org/10.1016/j.atmosenv.2018.04.017).

¹⁰ Muntean, M., Janssens-Maenhout, G., Song, S., Selin, N. E., Olivier, J. G. J., Guizzardi, D., Maas, R., Dentener, F.: Trend analysis from 1970 to 2008 and model evaluation of EDGARv4 global gridded anthropogenic mercury emissions., Science of the Total Environment 494–495 (2014) 337–350, [doi:10.1016/j.scitotenv.2014.06.014](https://doi.org/10.1016/j.scitotenv.2014.06.014)

¹¹ Specifically for the Artisanal Small Scale Gold Mining (ASGM) sector, there is scarce information in scientific literature and publicly available reports, in particular on the achievements related to emissions mitigation in the latest years; in order to provide a more realistic view on global mercury emissions from ASGM, a refinement of these emissions based on the information from official national reporting to the Minamata Convention on Mercury in the framework of the First Effectiveness Evaluation is needed.

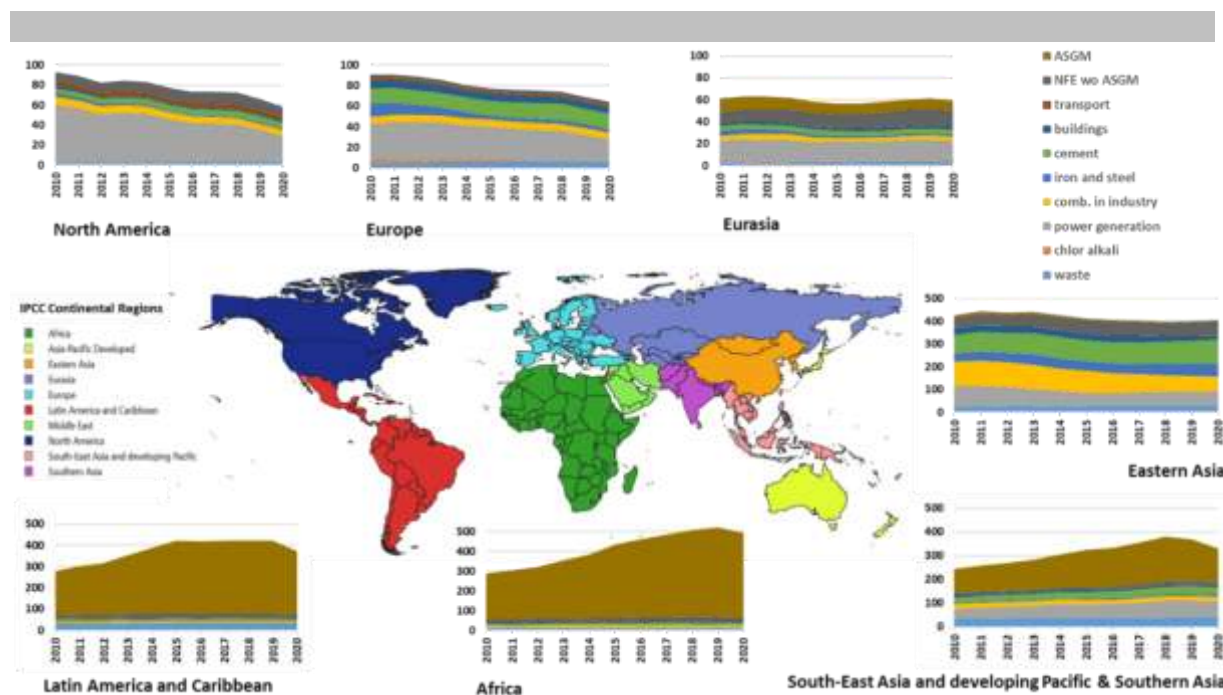


Figure 1. EDGAR mercury emissions in tonnes per year by IPCC continental region for the period 2010-2020.

The purpose of EDGAR inventory is to complete the global picture with emissions time-series for each country and global grid maps. This helps in understanding trends in global emissions by sector and identifying emissions hotspots and top emitters. EDGAR contributes to enhanced transparency and provides an additional source with which national and other global estimates can be compared. The granularity of EDGAR allows meaningful comparisons that can foster a better understanding of the differences in the methodologies, emissions factors used, and sectors/sub-sectors included or missing in different air emissions inventories.

EDGAR datasets and CLRTAP emissions for selected countries were compared to quantify differences. For most countries, the difference is significant and the CLRTAP emissions are higher. The trends between 1990 and 2022 and between 2014 and 2022 are also different for CLRTAP (decreasing) and EDGAR (increasing). E-PRTR shows a decreasing trend in emissions of mercury to air in Europe – 33.87 t/yr in 2007 to 16.05 t/yr in 2022. The trends from the MIA inventories cannot be compared as most countries completed the inventory for a single year during the period 2015-2019.

Sub-Saharan Africa, Latin America and the Caribbean and South-East Asia, are regions with ASGM activities. The increasing Hg emissions from these regions are a result of the significant growth in their ASGM sector. Literature reporting on the contribution of ASGM to Hg emission and releases from the ASGM sector has been very limited prior to the year 2000; but, data availability from this sector increased rapidly from 2001 until recently. ASGM National Action Plans include baseline estimates of the use of mercury in this sector. These estimates are for the base year; however, the implementation of Article 7 of the Convention will be reviewed every three years. Thus, updated estimates are expected to become available in the future.

2.1.2 What are the current levels of Hg (THg, GEM, GOM, PBM) emissions to air, [by country and by world region,] from Each Convention Annex D source category ASGM sources Other anthropogenic sources covered by the Convention (not Annex D, not ASGM) Anthropogenic sources not covered by the Convention

Generally, anthropogenic Hg emissions to the atmosphere are in the form of elemental mercury (Hg^0) and the oxidized form (Hg^{II}).

Despite large uncertainties, global THg release to the air from ASGM has been reported to be on the increase between 2010 to date (Cheng et al., 2023). The GMA reported an increase based on national inventories that were reported in 2019. The majority of the emissions are reported to occur in Asia followed by South America and then Sub-Saharan Africa with reported averages of 49%, 18% and 16% respectively. Country emissions vary widely due to specific contributions from the extent of their industrial, mining, energy and other activities.

Although existing data reveals emissions from coal-fired plants, waste incineration (especially medical wastes), industry and mining are amongst the main contributors of mercury emissions to the atmosphere, reemission from the oceans can also play an important role to atmospheric Hg levels despite suggestions that it may even buffer environmental Hg levels (Zhang et al., 2023).

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response: EDGAR datasets and CLRTAP emissions were investigated to assess the differences between these datasets. Selected countries with available data were used. For most countries, the difference between the EDGAR and CLRTAP reporting is significant and the CLRTAP emissions are higher. The trends reported between 1990 and 2022 and 2014 and 2022 are different for CLRTAP than EDGAR. CLRTAP reports decreasing trends and EDGAR reports increasing trends over the same time periods in the air emission data. In case of E-PRTR the emissions to air are decreasing (see table below). The MIA inventories cannot be compared as they were done in 2015-2019 years only and most countries did them only once.

Sub-Saharan Africa including the West Africa coastal belt with countries such as Cote d'Ivoire, Ghana, Benin Republic, Sierra Leone, and moving northwards including Mali and Burkina Faso, Latin America and the Caribbean, and South-East Asia, are regions with ASGM activities. There is an increasing use of Hg from these regions from the significant growth in their ASGM sector and consequently an increase in Hg emissions to the air. Literature reporting on the contribution of ASGM to Hg emission and releases from the ASGM sector has been very low prior to the year 2000, but reporting in this sector increased rapidly from 2001 until recently.

EDGAR reported that the ASGM sector increased from 101 t to 144 t. Total emissions in 2020 were 324 t, with significant contributions from the power generation sector (69 t). Latin America and Caribbean mercury emissions increased by 32%, with their global contribution rising from 18% to 20%. The ASGM sector grew from 208 t to 297 t. Total emissions in 2020 were 369 t. Africa's emissions increased by 71%, raising its global share from 19% to 27%. The ASGM sector increased from 233 t to 429 t. Total emissions in 2020 were 512 t.

Other main anthropogenic sources of Hg covers by the convention include coal, waste, chemical industry, paper and wood processing etc. Global trend from PRTR data sources revealed that THg emissions generally decreased from 2008 to 2023 with individual sectors showing slight decrease or almost constant over the decades.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.3 How have levels of anthropogenic Hg emissions to air from (each Annex D source category, ASGM activities, other covered source categories, or non-covered source categories) by world region changed over the available record? Have the estimated trends changed over the available record, and how do those changes compare to the timeline of the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response: Anthropogenic Hg emissions to air from ASGM activities globally showed an increase mainly due to regions and countries with intensive ASGM activities within sub-Saharan Africa, East and Southeast Asia and South America. In contrast, emission from chlor-alkali production has declined globally.

The increasing emissions from ASGM may not change as ASGM activities are predicted to increase due to the factor that promotes and sustains mining in this sector. For most sub-saharan and developing countries, ASGM is the largest source of mercury use and emission because ASGM is an attractive source of income for the rural poor in developing nations ; hence, it's continued use is expected because of the contribution to national economies as well as meeting the miners socio-economic needs.

Unlike general trends for some sector, because ASGM is limited to selected countries, all data base show similar findings and are consistent with each other despite some variations in reported values.

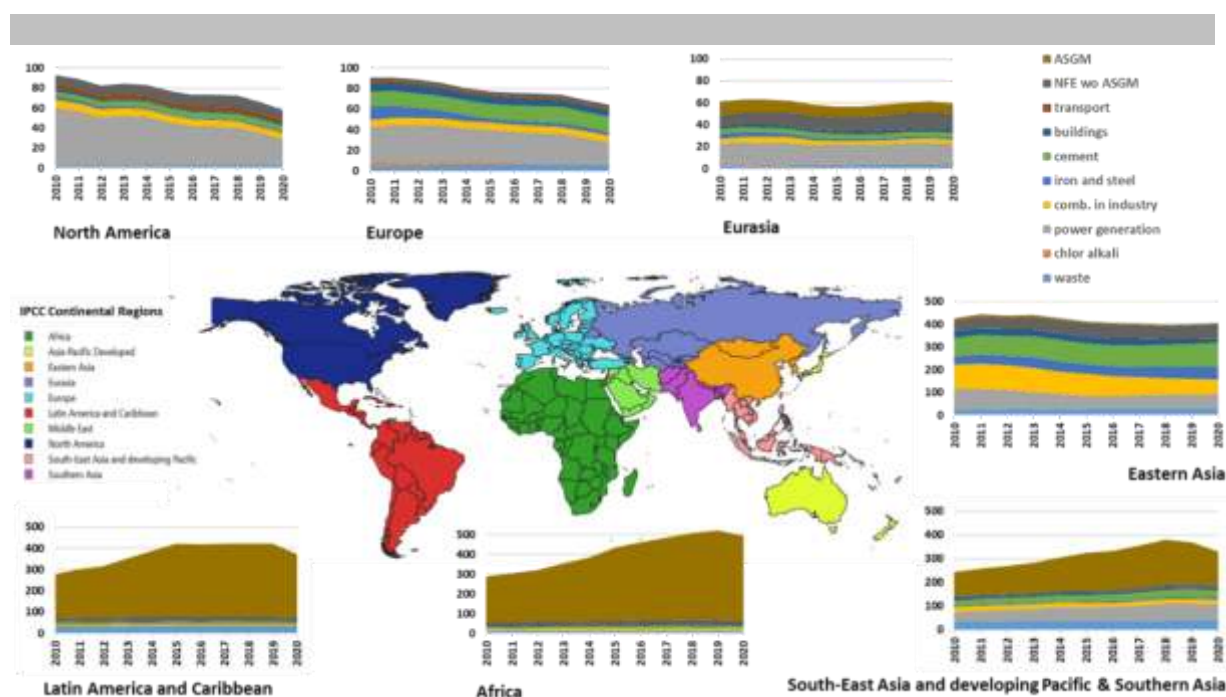


Figure: EDGAR mercury emissions in tonnes per year by IPCC continental region for the period 2010-2020.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.4 What is the contribution of specific mercury mitigation measures to the change in anthropogenic Hg emissions to air from (each Annex D source category, ASGM activities, other covered source categories, or non-covered source categories) by world region over the available record?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.5 What would levels of anthropogenic emissions to air from (each Annex D source category, ASGM activities, other covered source categories, or non-covered source categories) by world region be currently without implementation of obligations related to the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.6 How are levels of anthropogenic Hg emissions to air from (each Annex D source category, ASGM activities, other covered source categories, or non-covered source categories) by world region expected to change by 2030, including consideration of implementation of the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

Anthropogenic Releases to Land and Water

2.1.7 What data are available?

The United Nations Environment Programme (UNEP) Global Mercury Assessment (GMA 2018) provide d estimates of mercury releases to freshwater environments. The GMA 2018 contains information on the distribution of the estimated global anthropogenic Hg releases to aquatic systems in 2015 by sub-continental regions and by sector (Oil refining, Mercury production, Large scale gold, Non-ferrous metal production (Al, Cu, Pb, Zn), Coal-fired power plants, Municipal waste water, Coal washing, Mercury-added products, Chlor-alkali (Hg cell). The methodology used for these estimations was the UNEP toolkit (levels 1&2) to identify and quantify emissions and releases of mercury and its compounds. As no update of global mercury release inventory has been undertaken since this time, the emission and release group of OESG is working to update the information using the same methodology.

2.1.8 What are current levels of THg releases to fresh waters (rivers and lakes) from ASGM activities [by country/world region]? By Hg species?

Hg species from the ASGM is not well studied and very few information is readily available.

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.9 What are current levels of THg releases to coastal waters from ASGM activities [by country/world region]? By Hg species?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.10 What are current levels of THg releases to fresh waters (rivers and lakes) from non-ASGM anthropogenic activities (examples - agriculture, gold plating, dental offices,...) [by country/world region]? By Hg species?

Data from Kocman, 2025, Report to the Convention Secretariat:

Source Category	Tonnes Per Year	Estimation Method
ASGM	uncertain	
Municipal Wastewater	40 - 460	Activity Based Estimate
Coal Washing	10s – 100s	Activity Based Estimate
Mercury Containing Products	~100	Hg Consumption Based Estimate
Coal Fired Power Plants	10 – 130	Activity Based Estimate

Large Scale Gold Production	40	Air Emissions Based Estimate
Aluminum Production	12	Air Emissions Based Estimate
Zinc Production	9	Air Emissions Based Estimate
Lead Production	2	Air Emissions Based Estimate
Oil Refining	0.5	Air Emissions Based Estimate
Chlor-Alkali Production	0.5	Air Emissions Based Estimate
Copper Production	0.5	Air Emissions Based Estimate
Primary Mercury Production	0.5	Air Emissions Based Estimate

2.1.10.1 Using data from GMA 2018

Information Used (including applicable indigenous and traditional knowledge): United Nations Environment Programme (UNEP) Global Mercury Assessment 2018 provides estimates of mercury releases to freshwater environments.

The GMA 2018 contains information on the distribution of the estimated global anthropogenic Hg releases to aquatic systems in 2015 according to **sub-continental regions** and **per sectors** (Oil refining, Mercury production, Large scale gold, Non-ferrous metal production (Al, Cu, Pb, Zn), Coal-fired power plants, Municipal waste water, Coal washing, Mercury-added products, Chlor-alkali (Hg cell).

No specific estimations of releases from agriculture, gold plating, dental offices.

Methodology: The recommended approach is inventory-based estimation using statistical modeling (emission factor approach). The UNEP Mercury Inventory Toolkit (Level 2) and sector-specific emission factors from GMA 2018 can be used to estimate total mercury (THg) releases to freshwater from non-ASGM anthropogenic activities.

Answer/Response: The answer can be expressed as a map, a figure or a table of quantitative metric, along with narrative explanation.

Consistency of Answer with Existing Literature (especially the 2018 GMA): The data are from the 2018 GMA.

Evaluation of Confidence: The confidence of data for the non-ASGM water releases in the GMA 2018 is higher comparing to the ASGM water releases.

Information and Capacity Needs: There is lack of information releases from agriculture, gold plating, dental offices in the GMA 2018.

Excerpts from the GMA report:

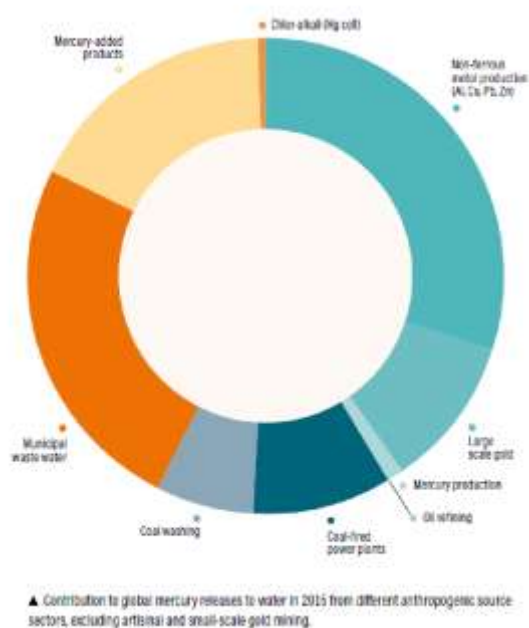
“Global anthropogenic Hg releases to aquatic systems by region in 2015, ASGM not included.

	<i>Annual release, t</i>	
<i>Australia, New Zealand & Oceania</i>	<i>7.22</i>	<i>1.2</i>
<i>Central America and the Caribbean</i>	<i>22.4</i>	<i>3.8</i>
<i>CIS & other European countries</i>	<i>48.7</i>	<i>8.3</i>
<i>East and Southeast Asia</i>	<i>266</i>	<i>46</i>
<i>European Union</i>	<i>33.9</i>	<i>5.8</i>

Middle Eastern States	16.7	2.9
North Africa	10.5	1.8
North America	32.4	5.6
South America	42.8	7.3
South Asia	55.3	9.5
Sub-Saharan Africa	47.1	8.1

Global releases of anthropogenic Hg to freshwater, excluding ASGM, based on revised estimates are **583 t/y** in 2015, compared to 180 t/y in the 2010 GMA inventory. The main sectors identified as sources of anthropogenic Hg releases to water are **non-ferrous metals production** and releases associated with **municipal sewage**, followed by the **disposal of Hg-added products** and releases from the **coal industry**. However, although not directly comparable – because combined releases to land and water are estimated for this sector – **ASGM is still considered the major single anthropogenic source of Hg to aquatic systems**. Some other industries also release substantial amounts of Hg – of the order of tens to hundreds of tonnes per year – either directly to soil/land (such as large-scale Au and primary Hg mining) or to various waste streams (such as for Zn production, VCM production, Cu production and the chlor-alkali industry).

„Releases from selected sectors: The 2015 global inventory of releases to water excludes releases from artisanal and small-scale gold mining, which are treated separately.



Mercury releases from copper, lead, zinc, aluminium, mercury, and large-scale gold production were estimated to be about 240 tonnes or 40% of the inventory. About a quarter comes from large-scale gold production.

Municipal sewage contributes about a further quarter of the global mercury release total. The phase-out under the Minamata Convention of many products that contain mercury is expected to decrease mercury releases in municipal sewage. Anticipated improvements in municipal wastewater treatment around the world are also expected to decrease mercury releases. The 2015 inventory shows that mercury-added products are also a major source of mercury releases. The use of mercury in products, such as batteries, lamps, dental applications, and others, is in decline and so are resulting mercury releases, especially in developed countries.

The 2015 inventory considers mercury releases in wastewater from coal-fired power plants and those resulting from coal washing. Together both releases are estimated to contribute about 15% of the global inventory. In addition, tens of tonnes of mercury per year accumulate in slurry ponds at coal washing sites globally, creating a hazard for local aquatic systems.”

In addition to direct releases to water, some of the sectors discussed here contribute mercury to land, general waste and sector-specific storage, which can be equally if not more important to the global mercury cycle. For example, zinc production is responsible for an estimated 4200 tonnes of mercury per year. Large-scale gold mining is estimated to put 2700 tonnes of mercury into soils each year, 45 times more than it releases directly to water. A further 940 or so tonnes of mercury from mercury-added products enters the solid waste stream, of which only 6% is estimated to be stored safely. A rough estimate of anthropogenic mercury input to soils is 7000-8000 tonnes. All

of this mercury becomes a potential secondary source of releases to water as well as emissions to air. A great deal remains unknown about this pathway of mercury pollution”

Table 6.4 Global anthropogenic Hg releases to aquatic systems by sector. Release data rounded to three significant figures.

Sector	Releases (range), t	% ^a
Production of non-ferrous metals (primary production of copper, lead, zinc and aluminum)	175	30
Production of mercury metal	5.18	0.9
Production of gold from large-scale mining	61.2	10
Mercury releases from oil refining	0.56	0.1
Mercury releases from chlor-alkali industry (Hg-cell technology)	2.70	0.5
Mercury releases with municipal sewage	145 (48.4–242)	25
Mercury releases from coal-fired power plants	55.6 (12.3–123)	10
Mercury releases from coal washing	38.0 (21.3–59.2)	6.5
Mercury releases from Hg-added products use and waste disposal	99.4 (66.5–133)	17
Total (ASGM not included)	583	100
Production of gold from artisanal and small-scale gold mining (releases to both land and water)	1221 (585–1858)	-

2.1.10.2 Using data from MIA Reports

Information Used. MIA reports and inventories of mercury emissions and releases based on the UNEP toolkit, developed by the Parties, to provide data on releases of Hg into water. It should be noted that there is no difference between emissions to freshwater or to coastal waters in this method.

The source categories for which releases are calculated are:

- 5.1 Extraction and use of fuels/energy sources
- 5.2 Primary (virgin) metal production
- 5.3 Production of other minerals and materials with mercury impurities
- 5.4 Intentional use of mercury in industrial processes
- 5.5 Consumer products with intentional use of mercury
- 5.6 Other intentional products/process uses
- 5.7 Production of recycled metals ("secondary" metal production)
- 5.8 Waste incineration
- 5.9 Waste deposition/landfilling and waste water treatment
- 5.10 Crematoria and cemeteries
- 5.11 Identification of potential hot-spots.

No specific estimations of releases from agriculture or gold plating. The releases from Dental amalgam fillings ("silver" fillings) (use/disposal) offices are included in subcategory 5.6.

Methodology. The recommended approach is inventory-based estimation using statistical modeling (emission factor approach).

Answer/Response. The answer can be expressed as a table of quantitative metric, along with narrative explanation.

Consistency of Answer with Existing Literature (especially the 2018 GMA). 2018 GMA contains the data from the national MIAs.

Evaluation of Confidence. According to the UNEP Guidelines „ In most cases, precise data are hard to get or non-existent, or it may be more appropriate to report data as intervals for other reasons, for example due to changes in a relevant time period. Generally, it is recommended to use relevant data intervals, and report them. Alternatively, the "middle ground estimate" or conservative estimate (see above) may be reported accompanied by quantified or estimated uncertainty of the data, for example as "15 kg Hg/year $\pm$ 5 kg""

Information and Capacity Needs. There is lack of information releases from agriculture, gold plating in the MIA reports.

Table below contains the data on mercury releases to waters from Dental amalgam fillings ("silver" fillings) (use/disposal) from the MIA reports

2.1.11 What are current levels of THg releases to coastal waters from non-ASGM anthropogenic activities (examples - agriculture, gold plating, dental offices,...)[by country/world region]? By Hg species?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.12 What are current levels of THg releases to land (separate from water) from ASGM activities, including tailings and sediments [by country/world region]? By Hg species?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.13 What are current levels of THg releases to land (separate from water) from non-ASGM anthropogenic activities (examples - agriculture, gold plating, dental offices,...) [by country/world region]? By Hg species?

The main releases to land come from primary metal production, waste deposition/waste water, use of disposal of mercury containing products.

Information Used (including applicable indigenous and traditional knowledge): From National mercury inventories: Insights and latest trends Nov 2022. 70 National mercury inventories exist and have been reviewed by experts within the framework of the MIAs. The main regions and categories covered are Latin America and the Caribbean, Sub-Saharan Africa, South Asia and South East Asia and the Pacific.

Methodology: The methodology used is the UNEP toolkit (levels 1&2) for the identification and quantification of emissions and releases of mercury and its compounds.

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.14 How have levels of anthropogenic Hg releases to water from (ASGM, non-ASGM activities) by world region changed over the available record? Have the observed trends changed over the available record and how do those changes compare to the timeline of the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.15 What is the contribution of specific mitigation measures to the change in anthropogenic Hg releases to water from (ASGM, non-ASGM activities) by world region over the available record?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.16 What would levels of anthropogenic Hg releases to water from (ASGM, non-ASGM activities) by world region be currently without implementation of the obligations related to the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.17 How are levels of anthropogenic Hg releases to water from (ASGM, non-ASGM activities) by world region expected to change by 2030, including consideration of implementation of the Convention?

Information Used (including applicable indigenous and traditional knowledge): Information was extracted and in some cases adapted from EDGAR, PRTR and GMA.

Methodology:

Answer/Response: Over the past two decades, Hg releases to the environment from ASGM have generally been increasing. This increase may possibly continue to 2030 due to sectors like ASGM, energy and waste. This is because ASGM is driven by communities that primarily depend on it for their livelihood, and more communities are getting involved because the work provides a better source of income at a faster rate than other activities like agriculture and transportation. Furthermore, ASGM is mainly categorized as an informal sector with little regulatory attention or abatement measures being put in place. There is

increasing call to switch technology from Hg use in the ASGM to other technologies. If the switch to other technologies such as the cyanide continues to be successful, then there might be a slowing down of emission increase. The scenario described above for ASGM is consistent throughout the data processors such as EDGAR, PRTR and GMA except that there are differences in absolute mercury emission values.

Consistency of Answer with Existing Literature (especially the 2018 GMA): EDGAR, PRTR, MIA, GMA are all consistent with published literature with regards the general trend on emissions and releases from countries having the ASGM sector. Notwithstanding, there are some notable differences in emissions in a few countries such as Ecuador, Ghana, Nigeria, Tanzania, Sierra Leone, Central African Republic, reported mercury use in some cases are far higher than their National Action Plan reported although in most cases, estimates from countries National Action Plan (NAP) tend to agree with Global Mercury Assessment of 2018 (to illustrate see figure below).

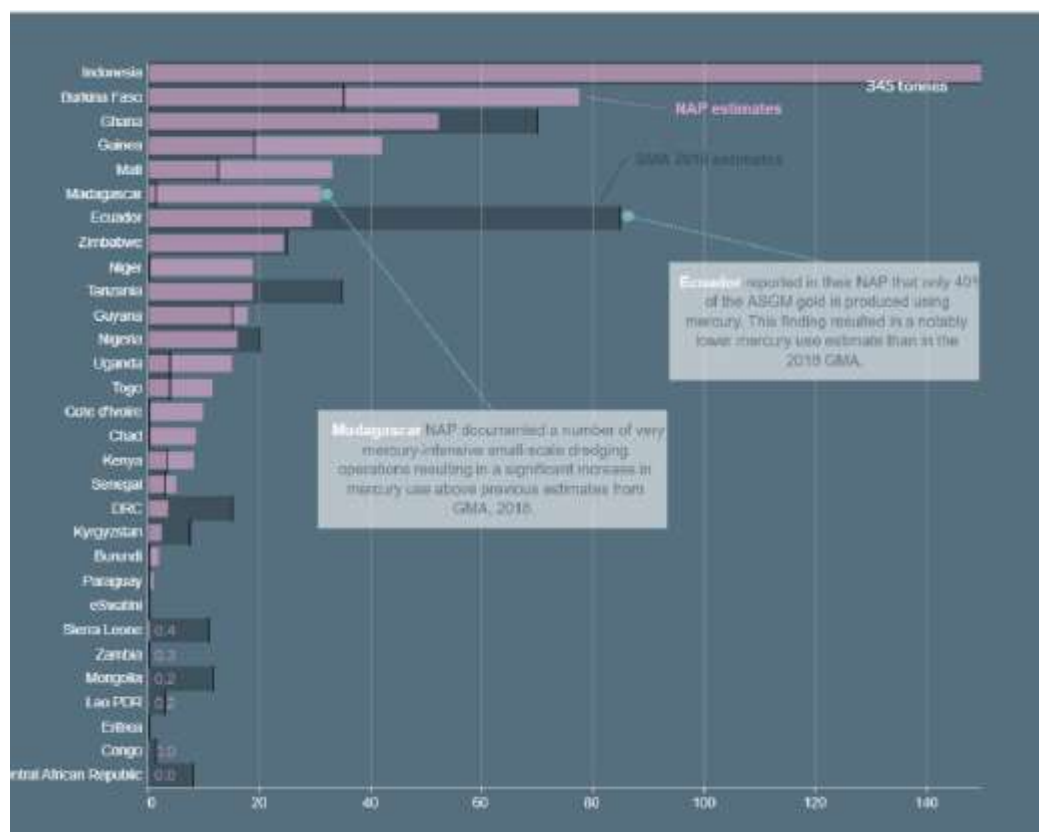


Figure: Comparison of Mercury Use estimates from National Action Plans and Global Mercury Assessment

Evaluation of Confidence:

Information and Capacity Needs: At the moment, there is relatively low availability of data from the ASGM sector. This is because; the sector is in developing countries with little or minimal funding to conduct primary research. This is evident by the fact that data on Hg emissions and releases are mainly from the MIA inventory.

2.1.18 How have levels of anthropogenic Hg releases to land from (ASGM, non-ASGM activities) by world region changed over the available record? Have the observed trends changed over the available record and how do those changes compare to the timeline of the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.19 What is the contribution of specific mitigation measures to the change in anthropogenic Hg releases to land from (ASGM, non-ASGM activities) by world region over the available record?

Information Used (including applicable indigenous and traditional knowledge): No relevant time series or inventories have been provided to the OESG team that would specify which specific mitigation measures have contributed to changes in anthropogenic Hg releases to land.

Methodology: The following is a qualitative expert opinion on potential effects.

Answer/Response: The single largest source of land releases according to the UNEP toolkit-compiled inventories is 'Primary metal production (excl. gold production by amalgamation)' (Category 5.2).

This is followed by emissions from waste deposition, landfilling and wastewater treatment (Categories 5.5, 5.6) and thirdly, consumer products with the intentional use of Hg (considering their whole lifecycle) (Category 5.9).

While land releases from primary metal production can be approximated based on the geology and Hg content of the mined ore, the extraction technology and the (air) pollution control equipment which is used, emissions from waste deposition, landfilling and wastewater treatment are highly uncertain. They depend on the – highly variable – Hg concentration in the deposited waste, its speciation and mobility. The third component, consumer products containing Hg, is addressed effectively through the Minamata convention and a decreasing trend in such emissions is to be expected due to Annex B bans of Hg use in such products or Hg content reductions.

Specific mitigation measures are likely to have mixed effects on Hg land emissions:

E.g. cross-media effects from increasingly efficient air pollution control devices might have adverse effects on emissions to land, e.g. where fly ash is deposited. A recent study looked at the Ultra-low emission retrofits of power plants in China and found that *"Hg content of fly ash increased from 0.16 mg/kg (P50 value) to 0.33 mg/kg after ULE retrofit, while Hg content of gypsum decreased from 0.75 mg/kg to 0.51 mg/kg. Both Hg contents are lower than 25 mg/kg, a proposed threshold to determine mercury-containing wastes. However, approximately 228 t Hg (2017 data) and 36.5 t (2018 data) Hg will be embodied in fly ash and gypsum, respectively, if all coal-combustion installed ULE technology, which implies risk of potential re-emissions and releases. Therefore, policies of treatment and disposal of fly ash and gypsum should be considered in the future."* (Wen et al. 2020, DOI:10.1016/j.jhazmat.2020.122729) However, the actual (leakage) behavior of such embodied Hg emissions to land remains highly dependent on the environmental conditions and the fate of fly ash and gypsum. While there are concerns about the public acceptance of using Hg-containing fly ash and gypsum in e.g. the construction industry, industry experts maintain that this is safe and that leaching is not an issue and the materials pose no risk to the public (e.g. a commentary by Lesley Sloss here: <https://www.sustainable-carbon.org/blogs/fly-ash-let-it-lie/>)

Mitigation measures of banning Hg use in products are very effective. In the longer term, this might have a significant effect on the Hg levels in municipal solid waste and thus on waste emissions.

Mitigation measures for primary metal mining?

- The activity drivers are very unlikely to be influenced by the Minamata convention.
- NFME mining is regulated specifically for emissions to the air under Annex D of the MCM
- The use of pollution control equipment is unlikely to result in more emissions to land as bespoke Hg filters may be used to remove remaining Hg from the valuable side product of sulphuric acid.
- Much of the secondary mining emissions are in the form of byproducts and slag heaps.

To conclude, the MCM is likely to have a direct effect on the total amount of Hg in consumer products, which will have an effect on the Hg levels in municipal solid waste, once those consumer products reach the end of their lifetime.

There is a possibility of a slight positive effect from better pollution control in non-ferrous metal mining, but this is not quantified. Similarly, data on releases from waste is lacking.

Consistency of Answer with Existing Literature (especially the 2018 GMA): The GMA 2018 did not provide data on Hg emissions to land, specifically.

Evaluation of Confidence: A rather low confidence

Information and Capacity Needs: There is a paucity of data on Hg release to land and water. Scientific literature as well as reports on Hg from the ASGM sector are scarce. This is even mentioned by EDGAR. As such, available data is far less compared to emissions to air.

Modelling of anthropogenic releases to land is urgently needed to address this question, albeit with the help of more data. The only way to answer it is to model the underlying technology structure of the different emission sources, and the changing technologies due to changing policy.

Additionally, it is slightly questionable how well “releases to land” can be separated from ‘releases to water’...

2.1.20 What would levels of anthropogenic Hg releases to land from (ASGM, non-ASGM activities) by world region be currently without implementation of the obligations related to the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.1.21 How are levels of anthropogenic Hg releases to land from (ASGM, non-ASGM activities) by world region expected to change by 2030, including consideration of implementation of the Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.2 Natural Primary Emissions and Releases

2.2.1 What data are available?

Outcomes from MA-1 to MA-18 are provided in “Model Assessment of Hg Levels, Spatial Patterns and Temporal Trends (Progress Report)” by The Multi-Compartment Mercury Modelling and Analysis Project (MCHgMAP). The report is in Appendix-1.

2.2.2 What are current levels of primary Hg emissions to air from natural sources? (MA-1)

Information Used. This assessment integrates several key data streams to quantify primary natural mercury (Hg) emissions for the 2010-2020 decade:

- Post-2000 measurements of mercury-to-gas ratios (Hg/SO₂ or Hg/CO₂) from the plumes of 28 volcanic systems, representing about 15% of all active volcanoes.
- Continuous sulfur dioxide (SO₂) and carbon dioxide (CO₂) flux data of approximately 100 actively degassing volcanoes from satellite and ground-based observatories.
- Peer-reviewed reports detailing Hg releases from volcanic eruptions between 2010 and 2020.
- A database of more than 600 measurements of Hg flux from soil to air in mercury-enriched geological terrains.
- Geological maps detailing the locations of major mercury deposits.

Methodology. The estimation of primary natural mercury emissions is based on a synthesis of field data, remote sensing, and statistical modeling. Monte-Carlo propagation was used to combine uncertainties by sampling the full distribution of published emission factors.

- **Volcanic Emissions:** Emissions are calculated by scaling continuous SO₂ and CO₂ flux measurements using measured Hg/gas emission factors. Emissions from explosive eruptions are estimated from peer-reviewed reports.
- **Rock Weathering Emissions:** These estimates are derived by merging site-specific flux-chamber measurements with spatial overlays of ore-deposit maps and geological information.

Answer. Total primary mercury emissions from natural geological sources for the 2010-2020 period are estimated to be 271 Mg/year. This total is divided almost equally between volcanic activity and rock weathering, 136 and 135 Mg/year, correspondingly.

Volcanic emissions are dominated by continuous (passive) degassing from about 100 persistently active volcanoes, which accounts for nearly 90% of the volcanic total. Explosive eruptions supply the remaining 10%. The five largest continuous volcanic sources are Ambrym (Vanuatu) at 13.2 Mg/year, Kīlauea (Hawaii) at 6.6 Mg/year, Etna (Italy) at 4.1 Mg/year, Miyake-jima (Japan) at 0.5 Mg/year, and Erta Ale (Ethiopia) at 0.4 Mg/year. Total annual volcanic emissions varied from a high of 157 Mg in 2010 to a low of 93 Mg in 2020, largely reflecting the level of eruptive activity.

Rock weathering emissions result from the breakdown of mercury-bearing minerals, which release gaseous elemental mercury. These emissions are concentrated in well-known mercury mineral belts like the California Coast Range (USA), the Great Basin (USA), and Almadén (Spain).

Regarding temporal trends, natural emissions appear to be nearly constant on decadal time scales. However, due to a lack of long-term, continuous measurements, the temporal trends in natural Hg emissions and their significance to the overall flux remain largely unknown. Many measurement sites are represented by only a single data point in time, which precludes robust trend analysis.

Consistency with Existing Literature. The current estimates represent a substantial improvement in certainty compared to past assessments, including the 2018 Global Mercury Assessment (GMA). Over the last decade, coordinated research has narrowed the uncertainty of the total natural primary emission flux from orders of magnitude to less than a factor of two. This convergence is the result of improved analytical methods, a more critical application of emission factors, and the integration of high-quality satellite data for volcanic monitoring.

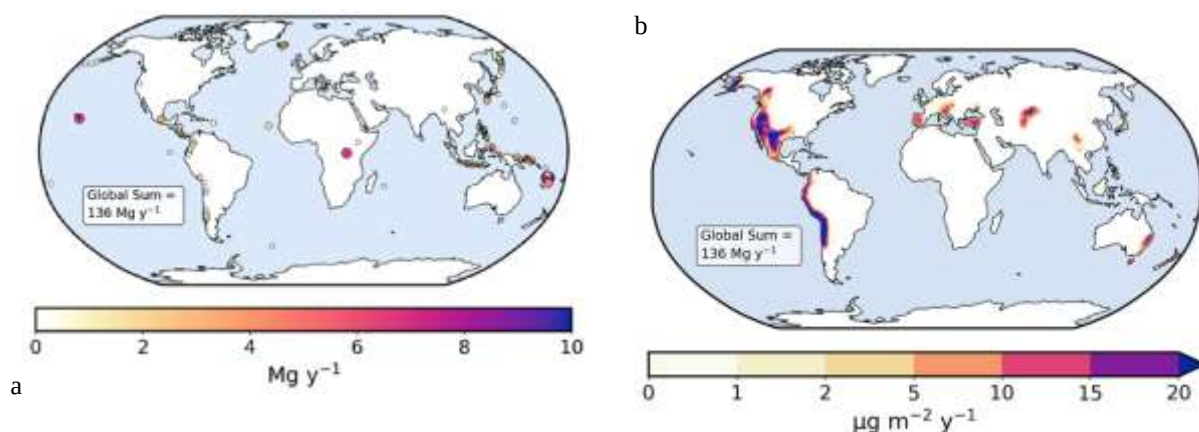


Figure 1. Geographical distribution of natural volcanic (a) and rock weathering (b) emissions.

Confidence. The current understanding of natural mercury emissions is much better than before – emission estimates now vary within one order of magnitude instead of three orders of magnitude in earlier studies, but major uncertainties and data gaps remain, with remote and submarine volcanoes now dominating the error budget.

- **Geographic Coverage:** Direct Hg measurements exist for only about 15% of active subaerial volcanoes and are biased towards a few regions like the Mediterranean and Hawaii. Some remote volcanic areas with high gas emissions, like Papua New Guinea, have no mercury measurements at all.
- **Underwater Volcanoes:** There is almost no data about mercury emissions from underwater volcanoes, although estimates suggest they might emit 100 Mg per year or more.
- **Explosive Eruptions:** There is great uncertainty about Hg fluxes from major explosive eruptions; near-source measurements are extremely sparse, representing only four eruptions since 1980.
- **Temporal Coverage:** The spatial lack of data is magnified temporally. Long, continuous datasets are scarce, which prevents the analysis of how emission factors might change with magmatic or hydrothermal conditions.
- **Mercury Speciation:** While fresh volcanic plumes contain more than 95% gaseous elemental mercury $\text{Hg}(0)$, this transforms into more oxidized and particle-bound forms in aging plumes. Quantifying this evolution is critical for transport models and understanding local deposition.

Information and Capacity Needs. To improve our understanding of natural mercury emissions, we need:

- **Expanded Measurements:** We should measure mercury emissions from volcanoes we have not studied yet, especially in remote regions such as Melanesia, Indonesia, and the Andean Central Volcanic Zone. We should use modern techniques and drones to reach difficult locations.
- **Long-Term Monitoring:** Establish a network of long-term monitoring stations at key volcanoes to understand temporal variability in emissions and to check if our emission factors are accurate.
- **Mercury Transformation Studies:** We need more research on what happens to mercury after it leaves a volcano – how it changes form and where it goes in the environment.
- **Submarine Volcano Research:** Develop and deploy new tools to quantify mercury emissions from submarine vents.
- **Expanded soil/bedrock flux studies** in tropical and high-latitude mineral belts, where data are scant.
- **Historical Records:** Studying ice cores, lake sediments, and other natural archives could tell us about natural emission changes over longer time periods.

2.2.3 What are current levels of primary Hg releases to a) fresh water and b) oceans from natural sources?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.2.4 What are current levels of primary Hg releases to land from natural sources?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.2.5 How have levels of natural primary Hg emissions to air changed over the available record? Have the estimated trends changed over the available record and how do those changes compare to the timeline of the Minamata Convention?

Information Used (including applicable indigenous and traditional knowledge): This assessment uses the same inventories described in detail in section 0.4.2.

Methodology: The methodology used to estimate primary Hg emissions to air is described in detail in section 0.4.2. We define primary emissions to air as volcanic and rock weathering sources. The interannual variability of volcanic emissions are estimated using satellite and surface-based estimates of volcanic activity between 2010 and 2020. We do not have sufficient information to estimate temporal variability of rock weathering emission sources.

Answer/Response: Average loadings and spatial distributions of primary Hg emissions are discussed in depth in section 0.4.2. Since this inventory does not consider the temporal evolution of rock weathering emissions, the temporal trend is driven by trends in volcanic activity data. Volcanic emissions are highest in 2010 at 157 Mg yr⁻¹ and lowest in 2020 at 93 Mg yr⁻¹ (Figure 2). We estimate that total primary emissions vary between a maximum of 292 Mg yr⁻¹ in 2010 and a minimum of 228 Mg yr⁻¹ in 2020.

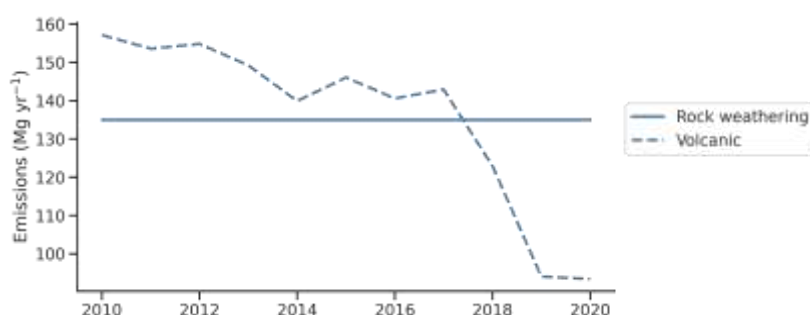


Figure 2. Interannual variation of global primary Hg emissions using inventory developed for MCHgMAP project.

Consistency of Answer with Existing Literature (especially the 2018 GMA): The consistency of this inventory with existing literature is described in depth in section 0.4.2. The 2018 Global Mercury Assessment did not evaluate the temporal variability of primary Hg emissions, preventing a comparison of temporal variability.

Evaluation of Confidence: Recent observational advances have made the evaluation of temporal variability of volcanic sources possible. However, following from the limitations discussed in section 0.4.2., the trend presented in figure 2 remains uncertain. In particular, improved constraints on the spatial and temporal variability of volcanic activity and Hg emission factors are necessary to validate estimated trends.

Information and Capacity Needs: Information and capacity needs for primary emissions of Hg are discussed in section 0.4.2.

2.2.6 How have levels of natural primary Hg releases to water changed over the available record? Have the observed trends changed over the available record and how do those changes compare to the timeline of the Minamata Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.2.7 How have levels of natural primary Hg releases to land changed over the available record? Have the observed trends changed over the available record and how do those changes compare to the timeline of the Minamata Convention?

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.2.8 What are the current levels and temporal trends of secondary Hg emissions from biomass burning across the globe? (MA-2)

Information Used. Emissions of biomass burning require information about the location and timing of the burning event, the amount of dry matter burned over each time interval, the amount of mercury contained within the plant matter that was burned. Several different biomass burning emission inventories have been developed to determine the timing, location, and amount of dry biomass burned using a combination of satellite-based retrievals and vegetation models. Furthermore, a growing number of measurements of Hg emission factors have become available in recent years, making these estimates more robust than in previous assessments.

Here, we use two biomass burning emission inventories:

- FINN2.5, a new inventory that was first published in 2023 and uses newly available satellite retrievals of fire radiative power to more accurately model the intensity of fires. This inventory serves as the default for our ensemble simulations.
- GFED4s, which was originally released in 2017 and remains the default inventory for several chemical transport models including GEOS-Chem.

Hg emission factors were obtained from a meta-analysis compiled by *Andreae* (2019) and refined with estimates from aircraft campaigns led by *Friedli et al.* (2003) and *McLagan et al.* (2021). From these studies, we have emission factors for six different land use types that correspond with those provided by the biomass burning inventories.

Methodology. Both biomass burning inventories separate emissions into 6 different vegetation categories that correspond with our literature based Hg emission factors. The emission factors are multiplied by their respective land use category and re-gridded to the resulting emissions to a 0.25° lat by 0.25° long grid, if necessary. Based on recent literature, it is assumed that 90% of Hg emissions are gaseous elemental Hg and the remaining 10% are in the form of particulate bound Hg.

Answer. Average annual emissions using the FINN2.5 inventory are 603 Mg yr⁻¹ and 282 Mg yr⁻¹ for GFED4s. Both inventories reproduce a similar interannual variation in Hg emissions on a global scale, albeit with the GFED4s inventory remaining approximately 50% of FINN2.5. When compared to other emission sources being used in ensemble simulations, biomass burning acts as a dominant contributor to interannual variability. We do not observe any noteworthy trends in either of the biomass burning emission inventories used here on a global

scale, which is unsurprising given the long-term climatic drivers that could impact biomass burning frequency, location, and intensity.

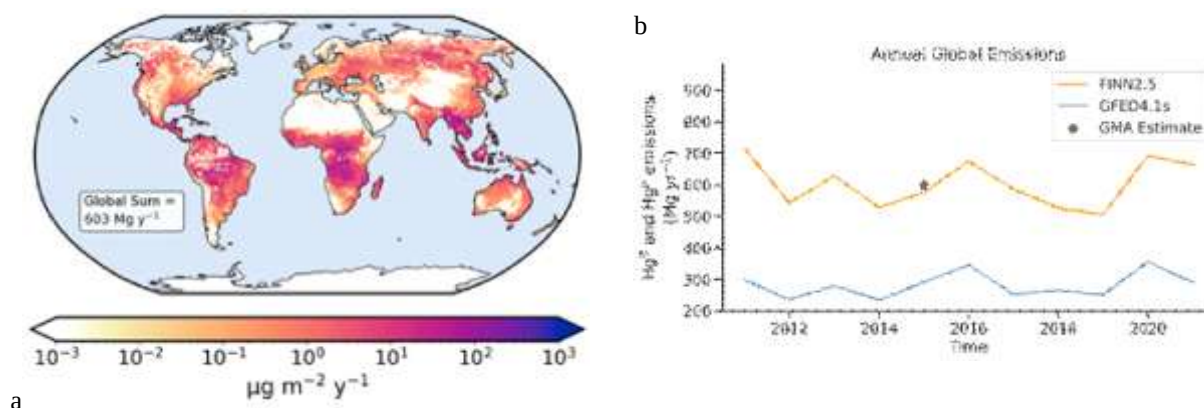


Figure 2. (a) Average spatial distribution of annual Hg biomass burning emissions in the FINN2.5 inventory and (b) interannual variation of Hg biomass burning emissions using both FINN2.5 and GFED4s biomass burning inventories.

Consistency with Existing Literature. The range of emissions estimated from the FINN2.5 and GFED4s inventories is consistent with estimates in the existing literature. The 2018 Global Mercury Assessment estimated annual biomass burning emissions of 600 Mg yr⁻¹ globally. Another recent review of global mercury cycling found that literature-based estimates ranged from 220 Mg yr⁻¹ to 612 Mg yr⁻¹. The emissions used as the default inventory remain on the higher end of existing estimates, albeit within the typically published range. One key driver of this difference is a more accurate representation of ‘before the Southeast Asian monsoon season’, which was poorly captured by previous biomass burning inventories, but more robustly represented by the FINN2.5 inventory.

Confidence. Based on the consistency of these global estimates with existing literature, it is expected that the FINN2.5-based emissions to be generally representative of the global flux annually. However, emissions are expected to be notably different from the estimates on regional and local scales, due to the assumptions required to estimate biomass burning emissions. In particular, it is expected that the emission factors will vary significantly from region to region due to differences in the frequency of fires, the age of standing biomass, and the relative contribution of above ground and soil biomass that is burned by a given fire. Furthermore, a growing body of research has highlighted that trees and sediments can act as effective proxies of previous atmospheric Hg loadings; however, this temporally and spatially varying accumulation of Hg is not captured in the six globally constant emission factors used by current CTMs. Therefore, while global estimates of annual Hg biomass burning emissions are expected to be conservatively robust within +/-50%, emissions from individual fires will be far more variable.

Information and Capacity Needs. Measurements of Hg concentration and speciation within biomass burning plumes over a wide range of vegetation types will be essential for further constraining the regional contribution of biomass burning reemissions to atmospheric and depositional loadings. Alternatively, this effort could be supported by a robust global network where, in addition to background and urban sites, several sites were specifically chosen to lie downwind of consistent seasonal fires (e.g., Howard et al. 2017).

2.2.9 What are the current levels and temporal trends of secondary Hg emissions from soils across the globe? What is the contribution of changes in environmental conditions to changes in secondary Hg emissions from soils over the period circa 2010 to present across the globe? (MA-3)

Secondary emissions of Hg from soils are expected to contribute a sizable fraction of annual inputs to the atmosphere, yet remain uncertain in the literature. Soil emissions are driven by the photoreduction or thermal desorption of previously deposited Hg on the soil surface or within soil matrices. To better understand the current levels and temporal trends from soil across the globe, estimates from models using contemporary understanding of the Hg evasion process is necessary.

Information Used. Soil evasion measurements available within existing literature were used to identify the key environmental factors driving soil emissions globally. This was used to construct a spatially and temporally

resolved model of soil evasion globally. Model-specific soil emission parameterizations were then adjusted by each modeling group to match the global total evasion predicted by this soil evasion model.

Methodology. A 2-level factorial design of experiments is used to estimate secondary Hg emission flux from soils and the potential effects caused by the changes of environmental conditions. This method is statistically robust, and the synergistic and antagonistic interactions among model parameters can be estimated with indications of statistical significance. The global modeling uses a spatial resolution of $0.25^\circ \times 0.25^\circ$ and a temporal resolution of 1 hour using the Hg(0) concentration field generated from the global atmospheric Hg model. The average global emission of Hg from soils was used to tune model-specific soil evasion parameterizations in the atmospheric models.

Answer. To estimate the current level and temporal trend of global secondary Hg emission from soil, model simulations using the means of key parameters were performed from 2010 through 2020 (i.e., the model results represent the simulated flux by running the model at half of the experimental level). In the modeling, the soil Hg concentration and soil bulk density across the globe are assumed without temporal variation, and the LAI, temperature and solar radiation are all temporally dependent. The model results are shown in Table 2. The estimated Hg secondary emission mass ranges from 2185.95 to 2277.54 Mg yr^{-1} from 2010 to 2020. A distinct temporal trend was absent.

In regions that undergo significant landscape changes because of climate change, the vegetation coverage and the biomass density over the past decade may exhibit significant fluctuations. For example, the Leaf Area Index (LAI) in China and India has risen relatively quickly. However, on a global scale, these climate-driven fluctuations are diluted, and changes in vegetation patterns are offset by changes in other regions. For instance, the reduction in tropical rainforests counterbalances the increase in forests in China. Based on the model estimates from 2010 to 2020, climate and vegetation patterns affect soil Hg emissions at limited spatiotemporal scales, but a temporal trend does not appear to be obvious.

To provide flexibility for individual atmospheric models, each ensemble member uses its own soil emission routine to represent emissions but adjusts relevant parameters to approach the annual global total predicted by the soil model described above. The ensemble average of these parameterizations is shown in Figure 3a.

We found the soil bulk density (BD), temperature (T), Leaf Area Index (LAI), soil Hg concentration (Hg_s) and solar irradiance (R_0) above canopy have strong effects on Hg emission from soil (Figure 3b). On average, increasing soil bulk density from 0.1 to 1.5 g cm^{-3} , and Hg content from 10 to 400 ng g^{-1} , and soil temperature from 0 to 30°C , and solar radiation from 0 to 1000 W m^{-2} , enhance the flux by $20\text{--}30 \text{ ng m}^{-2} \text{ h}^{-1}$. Additional $18\text{--}20 \text{ ng m}^{-2} \text{ h}^{-1}$ synergistic effects by the two-way interactions are also predicted. Furthermore, an increase in LAI from 0 to $7 \text{ m}^2 \text{ m}^{-2}$ reduces the flux by $19 \text{ ng m}^{-2} \text{ h}^{-1}$. The LAI could offset the positive effects caused by bulk density, soil Hg concentration, and solar radiation above canopy, leading to a decrease of -19 to $-16 \text{ ng m}^{-2} \text{ h}^{-1}$, indicating that canopy shading substantially reduces soil Hg evasion. Since the typical variation range of the examined environmental parameters is smaller than the range used for the modeling analysis (Table 3), the actual flux change will likely be smaller.

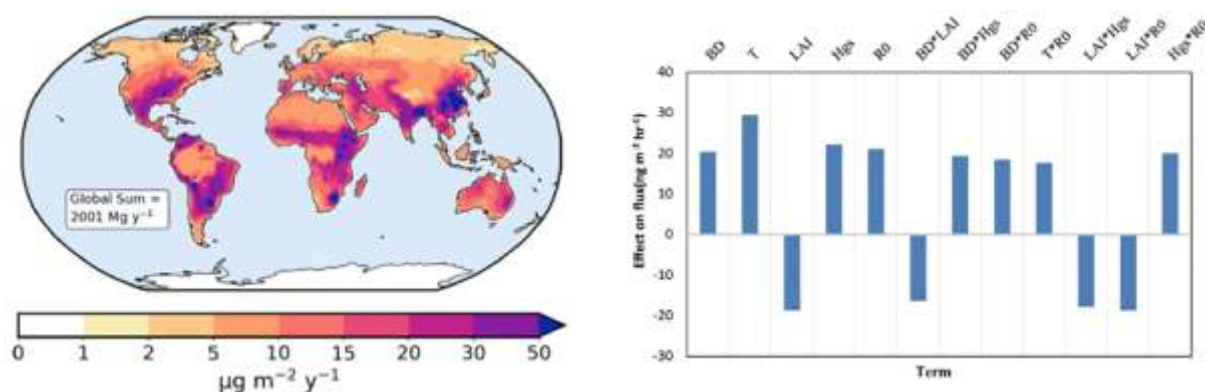


Figure 3. (a) Spatial distribution of average annual Hg emissions from soil used to inform soil emissions in model ensemble and (b) sensitivity analysis on model parameters for air-soil exchange using a 2-level factorial design after pre-screening the model variables shown in Table 1 for the identified significant factors. The effects shown in the figure are based on a significance level of 95% (i.e., $p < 0.05$).

Consistency with Existing Literature. The interannual range of $2186\text{--}2278 \text{ Mg yr}^{-1}$ lies on the higher end of existing literature based estimates. The previous Global Mercury Assessment (2018) used estimated global

emissions from soils and vegetation of approximately 1000 Mg yr⁻¹. Zhang et al. (2023) found that annual soil emissions of 1330 Mg yr⁻¹. Horowitz et al. (2017) used soil emissions of 910 Mg yr⁻¹. However, as exemplified by compilations like Agnan et al. (2016), large uncertainties in our ability to assess the flux of Hg from soils persist, underscoring the wide range of soil emissions that could satisfy globally available constraints.

Evaluation of Confidence. Emission estimates are based on several parameters that remain poorly constrained due to a limited number of direct measurements of soil emissions.

- Recent literature has shown that soils may effectively retain Hg, suggesting that previous emission estimates could be too high.

Information and Capacity Needs. More direct observations of soil emissions over background, rural, urban, and polluted sites are required.

Table 3. Examined model variables and the experimental levels of factorial design for air-soil exchange.

Terms	Description	Low bound	Upper bound
BD	Soil bulk density (g cm ⁻³)	0.1	1.5
pH	Soil pH (dimensionless)	4	9
P	Soil total porosity (%)	0.05	0.5
T	Soil temperature (°C)	0	35
SMOIS	Soil moisture (%)	0.05	0.5
R0	Solar irradiance above canopy (w m ⁻²)	0	1000
LAI	Leaf area index (dimensionless)	0	7
GEM	Atmospheric Hg(0) concentration (ng m ⁻³)	1.5	5
Hgs	Hg concentration in soil (ng g ⁻¹)	10	400
Foc	Soil organic matter content (%)	0.5	30
k1	photo-reduction rates in soil solution (m ² W ⁻¹ s ⁻¹)	3×10 ⁻⁹	9×10 ⁻⁹
k2	photo-reduction rates in soil particles (m ² W ⁻¹ h ⁻¹)	0.7×10 ⁻³	3.0×10 ⁻³
k3	Non-photo-reduction rates (thermal, h ⁻¹)	1.0×10 ⁻³	2.3×10 ⁻³

2.3 Air

Contaminants are usually first detected in the environment in the air. Mercury has a long residence time in the atmosphere (Dastoor et al., 2025) and can travel long distances from emissions sources rendering it a global pollutant. Atmospheric mercury has been monitored and studied for several decades where transport, transformation, deposition and trends have been reported in many areas (Cole et al., 2014; Dastoor et al., 2022; K. MacSween et al., 2022; Roberts et al., 2021; Soerensen et al., 2013; Steffen et al., 2013; Streets et al., 2019; Weiss-Penzias et al., 2016). While much data is available, there are significant gaps of atmospheric mercury information to provide a truly global picture. Further, there have been analyses of atmospheric mercury data from various regions and some attempts for a global snapshot but, until now, little has been done to collect and analyze all the air data from one large data set (AMAP & Environment, 2019; Bencardino et al., 2024; Sprovieri et al., 2010; UNEP: Global Mercury Assessment, 2002).

Mercury is found in the atmosphere in several forms (Ariya et al., 2015). In the gas phase mercury is found as elemental (gaseous elemental mercury (GEM)) or oxidized (gaseous oxidized mercury (GOM)) and is often summed as total gaseous mercury (TGM). Mercury is found on airborne particles primarily in the oxidized form but can be found as elemental as well (particulate mercury (PHg)). Atmospheric mercury can undergo wet and dry deposition. Wet deposition of mercury is the concentration in precipitation, amount of precipitation, and flux of Hg to the land surface. This measurement is used to understand the movement of mercury from the air to the surface via rain and snow (Prestbo & Gay, 2009; Weiss-Penzias et al., 2016). It should be noted that only GOM and PHg are deposited through precipitation, whereas GEM does not deposit to the earth surface through precipitation. Dry deposition of all mercury forms occurs to vegetation and can constitute an important input to terrestrial ecosystems in, for example, forested areas. Dry deposition is not regularly monitored and is not included in this report.

In this analysis report of the data submitted by Parties, a few assumptions have been made so that the data could be looked at together on a global scale. Only submitted data was used in this chapter as the data was pulled together into one data set for analysis. Data results from the scientific literature were not used. The site locations and duration of sampling for each site for the TGM (active), GEM (passive) and wet deposition are shown in Annex Air Tables 1-3. All active atmospheric mercury data is considered TGM even if it was collected as GEM. The passive air data is reported as GEM only. Due to the volume of the data submitted and the requirements to ensure consistency with the data sets, the air data is considered preliminary and subject to change for the final report.

2.3.1 What data are available?

Atmospheric mercury data has been provided by Parties and has been grouped into 3 categories for the purpose of the analysis. The overall air data submitted includes data from: 121 active air monitoring sites, 136 passive air monitoring sites and 198 wet deposition sites as shown in Figure AIR 1. Site locations and period of collection for TGM, GEM and wet deposition are shown in Annex Air Tables 1-3, respectively. Figures Annex Air 1a-f show the Gantt charts for all the TGM data collected and the period in which the data was collected. Figures Annex Air 2a-f show the Gantt charts for all the wet deposition data collected and the period in which the data was collected. Figures Annex Air 3a-h show the Gantt charts for all the passive air sampling data collected and the period in which the data was collected.

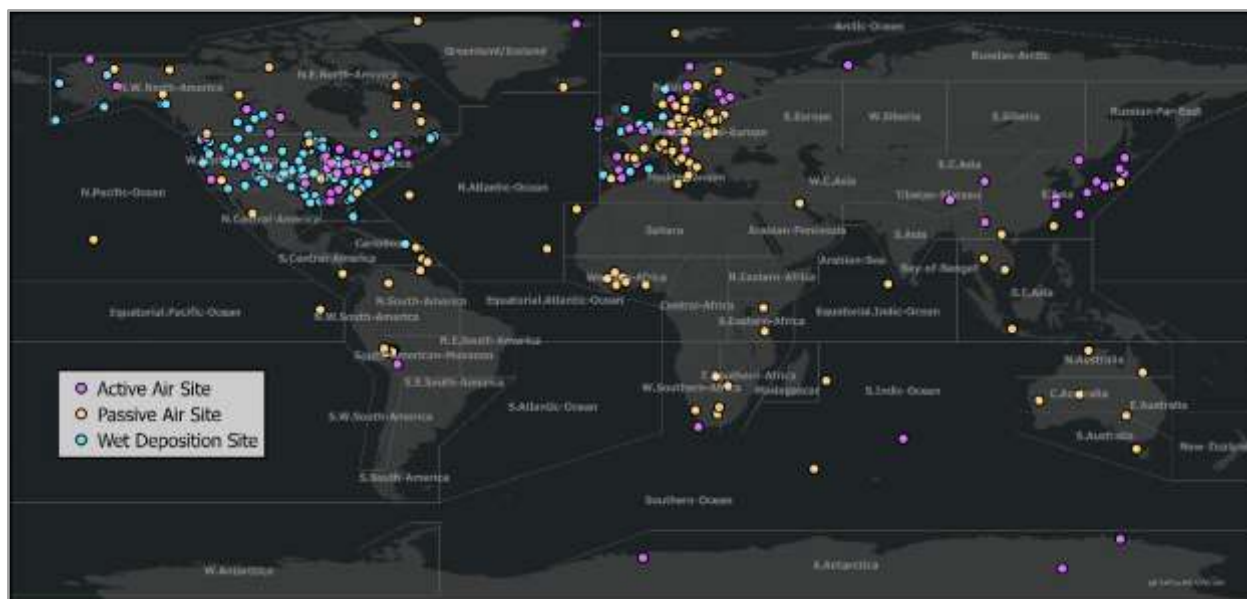


Figure Air 1: Map of the active, passive and wet deposition sites for which data was submitted from 2010 onward. Pink dots are the active sites, yellow dots are the passive sites and blue dots are the wet deposition sites.

Passive air samplers (PAS) for the measurement of GEM in the air is a newly developed method (McLagan et al., 2016). The sampler design was first published in 2016 and became commercially available in 2018. The number of sites per year has significantly changed and is shown in Figure Annex Air 4. Of the data submitted for use in the effectiveness evaluation, the number of sites with PAS measurements was highest in 2023, representing a consistent increase in usage. Note, 2024 only represents part of the year as that is what was received for this report at the time.

Wet deposition measurements are limited mainly to North America and Europe, with two sites in Japan, and only one available site in the southern hemisphere. Therefore, much of the world is not represented with any long-term monitoring data, and therefore, this summary only holds for a subset of regions. Precipitation plays an important role in wet deposition; thus, it is difficult to compare regions of the world as climates are so varied. Deposition in dry regions is very different than in wet regions based simply on the amount of precipitation carrying wet deposition to the surface. Wetter regions typically have lower Hg concentrations than dryer regions because the atmospheric mercury loading is distributed over larger volumes of precipitation.

2.3.2 What are current levels of mercury observed in ambient air in different regions of the world?

Active TGM: The levels of mercury observed in the ambient air for total gaseous mercury (TGM) vary around the world and over time. The global average concentration for active TGM for all the data collected is 1.51 (0.49–12.2) ng m^{-3} . The northern hemispheric average concentration for all the data collected is 1.54 (0.50 – 12.8) ng m^{-3} . The southern hemispheric average concentration for all the data collected is 0.99 (0.33 – 4.51) ng m^{-3} . These hemispherical differences have been reported elsewhere (Bencardino et al., 2024) and are consistent with the data presented here. Statistics for the mean TGM concentrations including mean, maximum, minimum, 25th and 75th percentiles are found in Annex Air Table 4.

Figures Air 2 a-b show the concentration levels of TGM active air samples and GEM passive air samples around the world. These data are reported as overall average concentrations.

For active air samples for TGM, most of the levels of mercury in the ambient air fall within the northern or southern hemispheric averages except for East Asia that is higher on average (Figure Air 2a). For the passive air sampling data areas in South America, Africa and East Asia also report above average concentration levels. The West/Central African data is very high (mean = 14.48 ng m^{-3}) and is investigated separately in a section below.



Figure Air A2: Averaged concentration of active mercury TGM (top) and passive GEM (bottom) for data from 2010 to 2024. Concentrations are expressed in ng m^{-3} .

Figure Air 3 a-d shows a frequency distribution of the TGM concentration in 4 regions. Other regions are not shown due to a lack of data. This figure shows regional differences in the distribution of concentrations in the active air data. For illustrative purposes a line was placed at the 1.5 ng m^{-3} mark to show the variation in concentration distribution in the different regions. The concentrations from Asia tend to skew above the 1.5 ng m^{-3} line, Europe and North America show some normal distribution around the 1.5 ng m^{-3} line but with outliers in the 3.5 ng m^{-3} or greater bin. The Oceania plot shows a significant skew of the data below the 1.5 ng m^{-3} line. This distribution of the concentration levels is within what has been previously reported (AMAP & Environment, 2019). However, additional data needs to be collected to show the “true” distribution of mercury levels globally. This data was limited to the collected active air data from which we know there are significant regional gaps. The data from the passive air network can fill in large gaps and will be investigated in the next draft of the report.

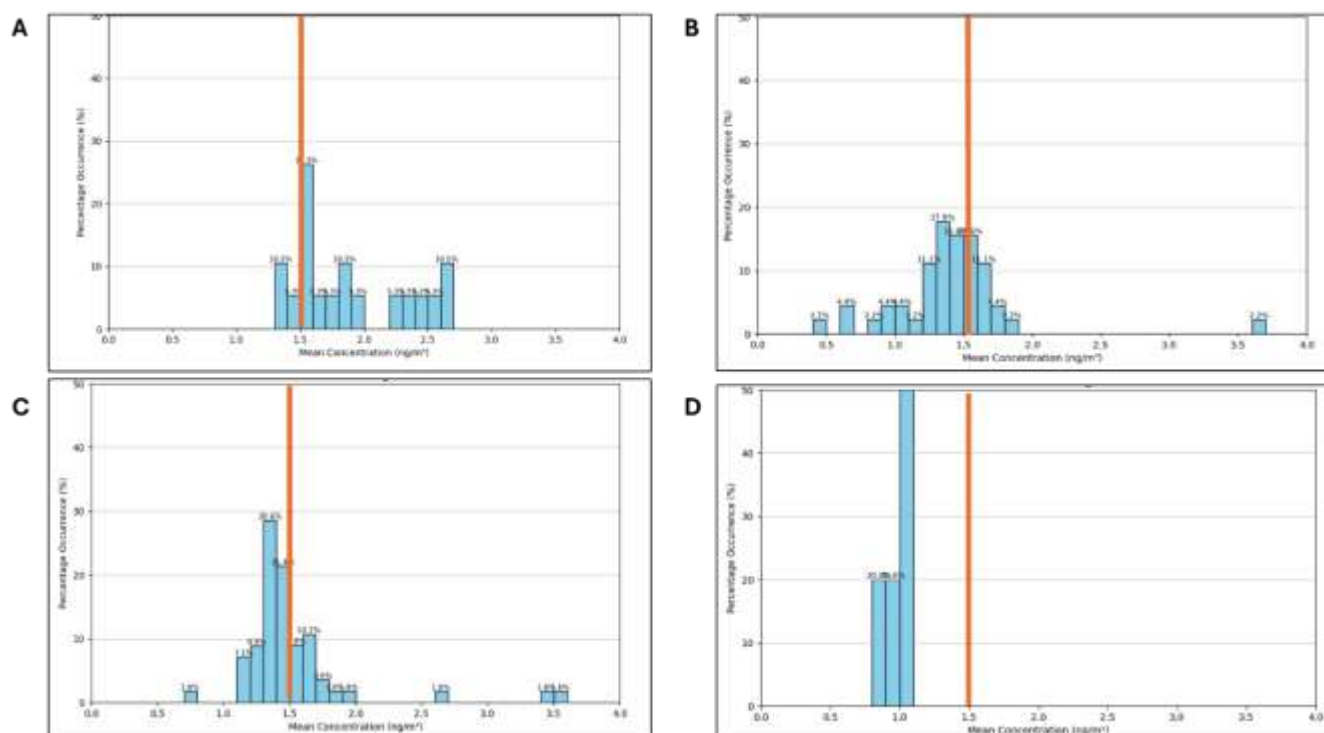


Figure Air 3: Frequency distribution of TGM concentration data (ng m⁻³) from A) Asia; B) Europe; C) North America and D) Oceania

Passive GEM: The levels of mercury observed in the ambient air for gaseous elemental mercury (GEM) vary as well around the world and over time. The levels reported through the passive work are different from those in the active sampling because they are results from largely different locations and over different time periods. The global average concentration for passive GEM for all the data collected is 1.45¹² (1.28-6.05) ng m⁻³. Figure Air 4 shows the difference between the mean concentration and the distribution of the GEM concentration data for these 3 regional areas¹³. The northern hemispheric average concentration for all the data collected is 1.54 (0.50 – 12.8) ng m⁻³. The tropical region has the highest GEM concentrations, with an average 1.53ng m⁻³ (SD 0.65 and range 0.94 - 2.62 ng m⁻³). The Southern Hemisphere has the lowest GEM with an average of 1.23 ng m⁻³ (SD 0.23 and range 0.98 - 1.39 ng m⁻³), consistent with previous and current understanding of the global distribution of atmospheric mercury. Northern Hemisphere GEM average concentration is 1.45 ng m⁻³ (SD 0.36 and range 1.01 - 2.14 ng m⁻³) which is consistent with current active estimates (Sprovieri et al., 2016).

¹² The data as part of this work are collected over a 3-month period. Concentrations were determined using the a sampling rate of 0.111m² day⁻¹ as recommend by the PAS manufacturer for use of MerPAS samplers at the time of data submission.

¹³ Sites that had an average ten times the standard deviation of the global passive air sampler average was removed from analysis to reduce skewing of results

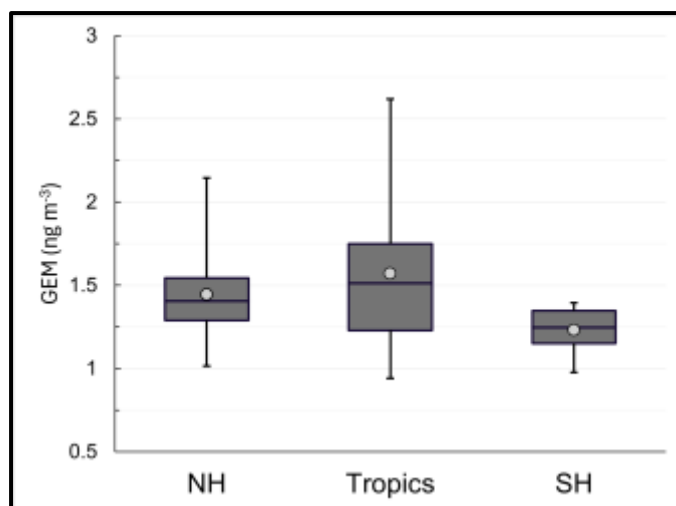


Figure Air 4: Box and whisker plots of hemispheric GEM concentration for data collected between 2019 and 2024. Boxes represent the 25th and 75th percent quartile range, grey dots are the mean, error bars show maximum and minimum concentration for each region.

Wet deposition: The overall “typical” concentration of mercury in precipitation is between 7.5 and 11.5 ng Hg L⁻¹ based upon the submitted long-term measurements in North America, Japan and Europe. Figure Air 5 shows the annual median of all precipitation weighted average concentrations (ng Hg L⁻¹) measured at all sites from 2010 to 2020. These medians can be thought of as typical concentrations of mercury in precipitation for the individual regions. Figure Air 5 clearly shows that dry, western North America has the highest typical concentration of 11-12 ng Hg L⁻¹. The other three regions have similarly wetter climates and have a typical concentration of 7.5 ng Hg L⁻¹. These values have been consistent since 2010, varying by only 2 to 3 ng Hg L⁻¹ over the years. Some of this variability is due to precipitation amount variability between the years; wetter (drier) conditions lead to lower (higher) concentrations when mercury content does not change. Measurements suggest there is a slight (not significant) tendency for decreasing concentrations over all sites. This tendency is more pronounced in western North America, but drought experienced during the early years is a complicating factor.

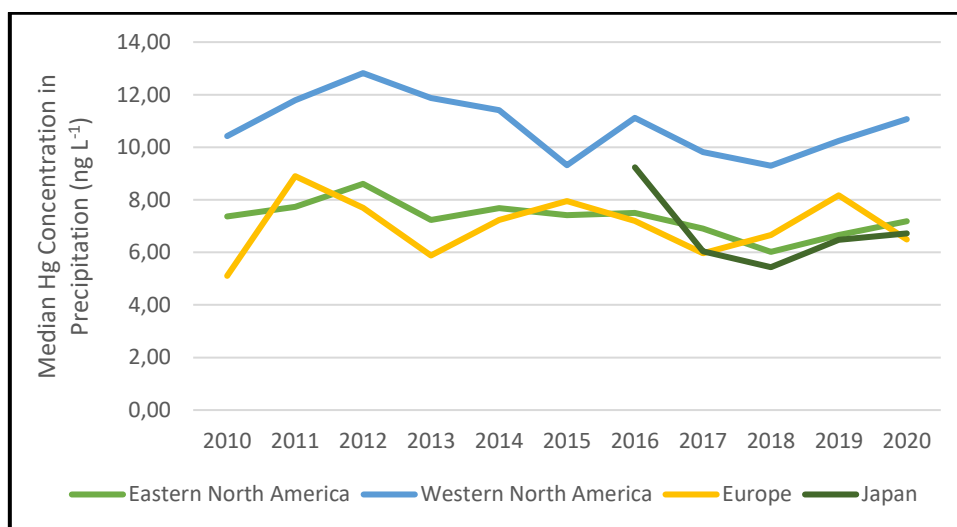


Figure Air 5: Median Precipitation Weighted Mean (PWM) Concentration, by Region and Year

The concentration values from the 4 identified regions were used to estimate regional deposition (Hg concentration x annual precipitation) with an approximation of their precipitation (40” or 1016mm for eastern

North America, Europe, and Japan; 15" or 381mm for western North America) (Figure Air 6). Some sites will have more (less) precipitation and; therefore, more (less) Hg deposition than others.

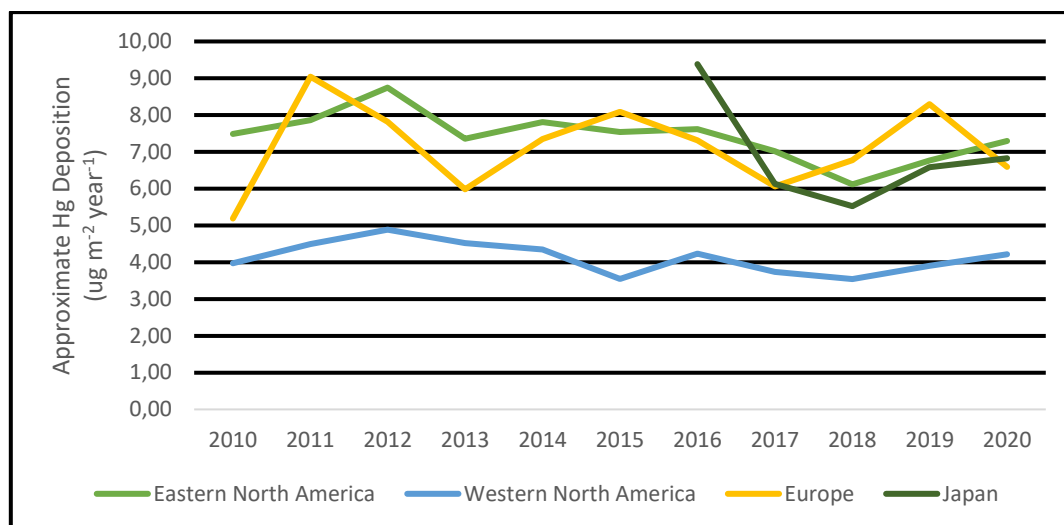


Figure Air 6: Typical mercury wet deposition rates for all Stations by region and year

Eastern North America, Japan and Europe all have similar levels of mercury deposition over all years (similar to the regional distribution of concentration discussed above). A typical deposition value is approximately $7 \text{ ug m}^{-2} \text{ year}^{-1}$ for these three wet regions. Precipitation variation contributes to the variability between the years.

Western North America, which is much drier, has significantly less wet deposition even if the concentrations appear higher. A typical value for wet deposition in western North America is approximately $4 \text{ ug m}^{-2} \text{ year}^{-1}$. Despite regional similarities, individual sites themselves can exhibit considerable variability. For example, deposition rates typically range from 3 (western U.S.) to $25 \text{ ug m}^{-2} \text{ year}^{-1}$ (Gulf Coast, U.S.). The same is true for Europe, although with not quite as large of a range (Spain vs Germany, for example).

Methods used:

TGM:

- The submitted data were averaged first to daily averages followed by averages for the whole period of data submitted.
- Analysis was performed using data supported in a Postgresql database with ancillary calculations using the Pandas module in Python.
- The mean concentration map shows the mean concentration of total gaseous mercury from 2010 onwards.
- The trend maps show sites that have at least 5 years of data and include data before 2010 so that sufficient time spans would exist to give the best available results for trend analysis.
- Histogram plots were produced for each continent showing the discrete distribution of daily mean concentrations of total gaseous mercury binned in 0.1 ng m^{-3} increments.
- Maps: All maps in this report were created using ArcGIS Pro (ESRI, 2025)

GEM:

- Averages and standard deviations were determined for data collected between 2019 and 2024. Sites whose averages exceeded 10x the standard deviation were excluded from analysis. These include Serpong Indonesia, Djekanou, Touba, Koutouba and Diawala in Cote d'Ivoire and Kenyase Ghana.
- Hemispheric statistics was determined by first assigning hemispheres based on latitude. Sites located above 23.5°N were classified as Northern Hemisphere, between 23.5°N and 23.5°S as Tropics and below 23.5°S as Southern Hemisphere.

- Data was also separated into regions based on IPCC climate regions. To avoid, where possible, regions with a single site adjacent regions were combined.

Wet Deposition:

- Data were averaged on an annual basis and plotted for all the years.
- Analysis was performed using data supported in a Postgresql database with ancillary calculations using the Pandas module in Python.
- The Global Mercury Assessment 2018 (AMAP & Environment, 2019) report includes assessment of Hg concentrations and wet deposition for the USA, Canada, Europe, Japan and China. Also included are results from the GMOS project with additional results sites mainly from stations in the southern hemisphere (Sprovieri et al., 2017)
- The results summarized here are consistent with the results presented in GMA 2018 (i.e. comparable median concentrations and wet deposition fluxes. Mercury concentrations and wet deposition from the southern hemisphere are generally somewhat lower than in the northern hemisphere in agreement with the location of the main emission areas. Some sites in the southern hemisphere have a high variability in precipitation amounts but the influence on Hg deposition was not as pronounced as in the northern hemisphere (Sprovieri et al., 2017).

Information used:

TGM:

- TGM information is based on data submitted to EE
- The majority of the data was collected through the Tekran 2537 series (A,B and X) and through the Tekran 1130/1135 system instrumentation (the GEM data collected through this method was used as TGM for the active data monitoring)
- For other methods used, please refer to the EBAS database ([EBAS home – ebas homepage](#)) for site specific methods employed for the European data sets

GEM:

- Based on data submitted to EE
- Passive Air samplers have been used to measure GEM in the air since 2019.
- Deployment lengths range between 1month to 1year. Typically, 3 months
- West/Central Africa data not presented – very high concentrations (Avg 14.48 ng m⁻³)

Wet deposition:

- Data submitted through the EE process and data was collected from known databases (e.g. EBAS)

Evaluation of confidence:

TGM:

- The methods used for data collection and analysis have been used in scientific literature over many evaluations (AMAP, 2005, 2011; AMAP & Environment, 2019; Katrina MacSween et al., 2022).

GEM:

- The methods used to collect and analyze the information for the passive GEM analysis are fairly recently as this method is new but it has been published and compared in the scientific literature (McLagan et al., 2016; McLagan et al., 2018; McLagan et al., 2019; Naccarato et al., 2021).

Wet deposition:

- We are confident in the quality and repeatability of these wet deposition measurements. All of the European, North American, and Japan measurements were made available from long-term and scientifically sound measurement networks. The networks are not perfectly aligned (*e.g.* the networks use different monitor manufactures, etc.), but they do use standard methods, standard quality assurance testing, and years of repeated measurements. Some observations in these networks extend back into the mid-1990s. These networks have been shown to be highly comparable. Therefore, the confidence in the interoperability between the measurements and the measurements themselves is quite high.
- Other network measurements exist (China network, Asia Pacific Mercury Monitoring Network), but their data was unavailable for this report.

Information and capacity needs:

TGM:

- Long term dedicated sampling programs with active air sampling remain necessary to understand the processes of atmospheric mercury in differing and changing environments.

GEM:

- Long term dedicated sampling program for this inexpensive and relatively easy method of data collection is needed in many underrepresented areas of the world. Capacity is needed for the laboratory and organizational operations.

Wet deposition:

- Additional measurements in locations/regions where no observations are present are needed. Capacity to undertake large networks in the southern hemisphere (Africa, South America), and more observations in all of Asia.
- Long-term observation of wet deposition across the oceans of the world; very little data exists here whatsoever and will not likely be available in the near future.

2.3.3 What are current levels of (GOM/ PBM, Litterfall, Dry Dep, Isotopes) in the air around the world?

at sites that are remote from anthropogenic sources

at sites expected to be affected by local anthropogenic sources

At this time, GOM/ PBM, Litterfall, Dry Dep, Isotopes are not being considered in the analysis. For the first cycle of the EE process, only TGM/GEM and wet deposition are being assessed due to the complexity and lack of information from these other atmospheric mercury species.

2.3.4 What is the spatial representativeness of current levels of Hg observed in air? Do we have geographic gaps in our observations?

As can be seen by the results shown in the previous section, there is spatial variation in the data. The TGM and GEM data show that there are higher levels of mercury in Figures Air 2, Air 3 and Air 4. For the TGM concentration levels shown in Figure Air 2, mercury levels are higher in the East Asian region than other regions ranging from 1 to $>2.5 \text{ ng m}^{-3}$ averaged concentration. Europe shows total average concentrations between <1 and 2 ng m^{-3} . The southern hemisphere shows the lowest TGM levels ranging from <1 to 1.5 ng m^{-3} and North America shows similar levels to Europe. It must be noted that this is the average of all the data submitted for each location and an analysis of the 2010 to present data is needed as well as an inspection of the temporal variety of the regional data. The passive regional GEM data is shown in Figure Air 5.

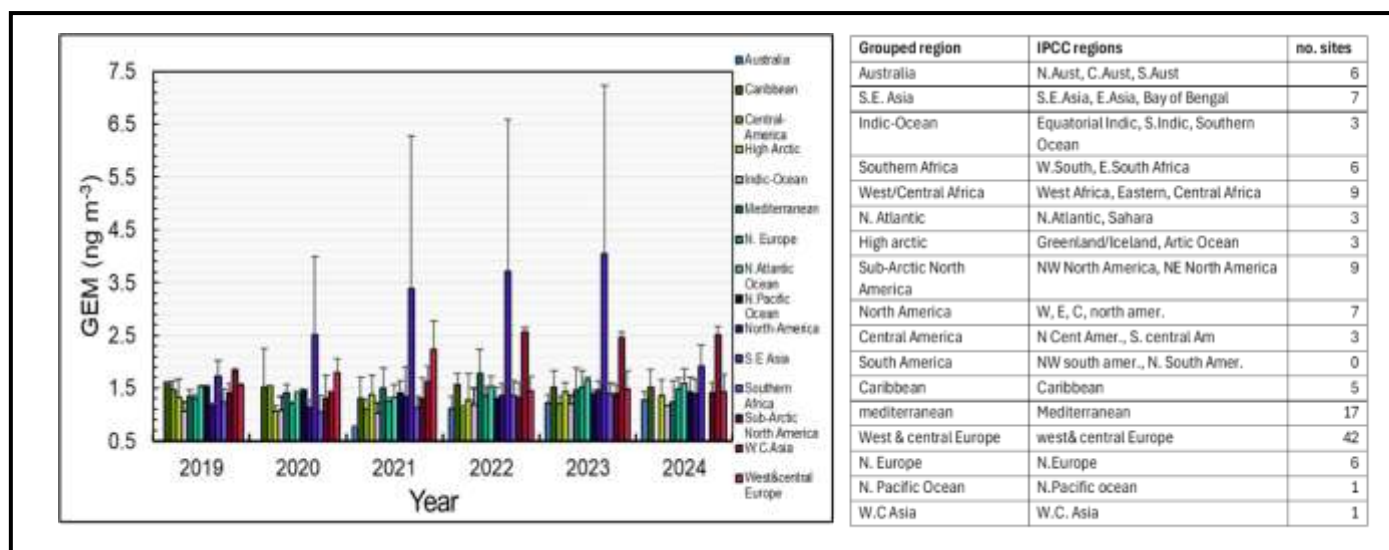


Figure Air 5: Annual average GEM concentration by grouped IPCC region for passive GEM measurements between 2019 and 2024. The error bars represent the standard deviation. Table (right) shows the grouped regions and the number of site data used for the regional analysis.

Figure Air 1 shows that much of the globe is not represented in the wet deposition measurements. Land measurements are missing from Asia, Africa and South America and there are no consistent measurements available over the oceans. Measurements from some of those regions (notably Asia) are available in the scientific literature and through networks but were not submitted as data for this report. The lack of data in some regions brings up the question of what can be said about mercury deposition in gap areas? Since we do not have direct measurements, estimates must be made from the data that we do have. It is reasonable to assume that concentration and deposition rates behave consistently between regions with and without measurements based on the general observation that mercury is typically present in the atmosphere for longer periods of time before being deposited. GOM and PHg can move in the atmosphere on a regional scale (at least) before being removed by wet deposition and they can also be re-emitted to the air further enhancing the potential transport distance. Therefore, we can generally assume that the concentration and deposition rates in the drier regions of Asia and Africa are similar to western North American rates. This is generally confirmed by measurements made in western China and South Africa and have been reported in the scientific literature (Gichuki & Mason, 2013). This is also likely true in South America. The moister regions of the landmasses are likely to have concentration and deposition rates similar to those measured in Europe, Japan, and Eastern North America. Local areas with large and local sources of mercury, with abundant rain will likely have the highest deposition rates (e.g., eastern China, Indonesia, etc.).

The wet deposition information provided herein has been collected primarily at regional background locations. For example, the monitoring data from Europe were generated within the European EMEP network where stations are positioned to reflect the regional background concentrations and are thus, not located near any local anthropogenic sources. However, a few sites may be classified as remote (e.g. Pallas in northern Finland). In Japan, the Cape Hedo site is in a remote area far from any anthropogenic emission sources and the Oga site (northern Japan) is in a rural area. While there are major anthropogenic emission sources around the sites, unknown/small emission sources can, at times, affect the air monitoring data rendering the sites to not be considered completely remote. For North America, the same is true as European observations. Most of the sites are in remote areas and are regional in nature. They are generally not close to any obvious sources of Hg emission. There are a few sites that are near urban areas (near Washington D.C., for example) but are generally considered regionally representative of their area.

For mercury monitoring, it is important to note that there are many sites between "completely remote" and "affected by major emission sources," and it is not easy to assess and classify the effects of anthropogenic sources on such sites.

2.3.5 How have levels of mercury species observed in the air in different regions of the world changed over the available record? Are there differences in the trends observed at sites that are a) remote from anthropogenic sources and b) expected to be affected by local anthropogenic sources?

This question, at this time, will be addressed through the time trends analysis in the following section.

Information Used (including applicable indigenous and traditional knowledge):

Methodology:

Answer/Response:

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Evaluation of Confidence:

Information and Capacity Needs:

2.3.6 Is there a change in the observed trends over the record of the data, where adequate data to establish a trend is available, and how do those changes in trends compare to the timeline of the Minamata Convention?

Information Used

TGM:

- Data submitted through the EE process and data was collected from known databases (e.g. EBAS).

Wet deposition:

- Data submitted through the EE process and data was collected from known databases (e.g. EBAS).

Methodology

TGM:

- Temporal trends were calculated annually using the Mann Kendall Test (MKT) with Sen's slope estimation (Gilbert, 1987).
- Data were averaged on a daily basis (where possible) and plotted for all years.

Wet deposition:

- Data were averaged on a monthly basis and plotted for all the years.
- Sites were included in the analysis based on the time span of the data, whereby sites that did not extend past 2016 were excluded since they did not support any trend analysis past 2013.
- Sites that did not span a minimum of 5 years were also excluded, since that was considered the minimum duration over which trends could be calculated¹⁴.
- Annual trends were calculated using the Mann Kendall Test (MKT) with Sen's slope estimation (Gilbert, 1987). The MKT was run for annual trends based on monthly precipitation weighted monthly concentrations and depositional sums.
- The MKT results were calculated for the candidate sites using a weighted monthly mean to remove possible measurement biases for months that had more measurements than others; a sum would be sensitive to that and affect trend analysis. In addition, fractional contributions for those samples that spanned monthly boundaries were applied to their respective months.

¹⁴ Sites have varying lengths of data; ranging from barely meeting the 5-yr threshold to over 20 years

- The MKT was performed through comparing month by month and averaged up over the year. A Chi-squared analysis was applied to the monthly P-values to determine an annual P-value to extrapolate an annual significance from monthly significance results.

Using the data submitted temporal trends for active TGM and wet deposition analysis has been done. The passive GEM data is not, at this stage, sufficiently long enough to reflect statistically significant trends and is thus not shown in this report.

Active TGM:

Temporal trends for active TGM measurements were calculated for the overall data sets. This analysis was undertaken for 77 of the data sets that met the required criteria¹⁵. The trends were calculated on the full data sets that were provided and will be updated to reflect the 2010 onward periods later. For each site an overall trend was calculated and assigned a status of no trend, increasing or decreasing. These results are shown in Annex Air Table 4 and in Figure Air 6. The number of years of data included in each trend range from 6 to 30 years. Thus, the results presented here are a snapshot over varying time frames.

In general, for this overall analysis, most of the concentration levels of TGM have no statistically significant trend. The calculated decreasing trends vary across the various regions. Of the total 77 sites, 49% show a decreasing trend and 49% show no trend and 1% show an increasing trend. It has been found that the trends of TGM concentration vary over time and can reflect changes in emission controls (Bencardino et al., 2024; Cole & Steffen, 2010; Cole et al., 2014; Ebinghaus et al., 2011; K. MacSween et al., 2022; Streets et al., 2019). Further analysis is required to assess more recent trends and how trends have changed over time.

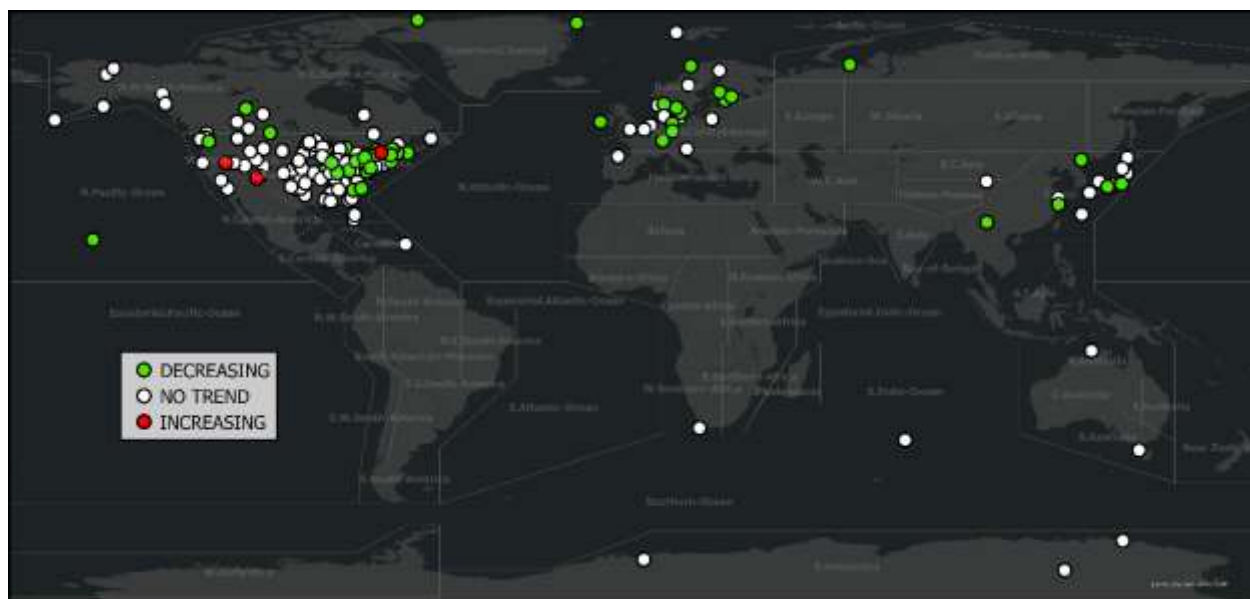


Figure Air 6: Trends of active TGM concentration for all reported data for all sites (that meet the required criteria). White dots indicate that no trend was found, red dots indicate an increasing trend and green dots indicate a decreasing trend.

Wet deposition:

132 sites were used in the analysis of the temporal trends for mercury concentration in precipitation and the temporal trends calculated for wet deposition.

¹⁵ The criteria includes a 75% completeness for daily data and minimum 5 years of data

Concentration: The frequency counts of trends in precipitation weighted mean (PWM) concentration show many stations with no statistically significant trend in concentration (Figure Air 7). Over the 132 sites in this report, only one out of 5 sites show a decrease in concentration over this time period and two sites show an increase in concentration over time. Most of the sites (80%) show no statistically significant change in concentration in either direction. Therefore, the general global conclusion is that a minority of sites have decreasing trends in precipitation concentration. Most sites show no significant change. The annual change in precipitation concentration rate (percentage per year) at monitoring sites (annual change rate calculated based on the slope of MKT) was estimated, and the distribution of change rates by site is shown (Figure Annex Air 5). Although many stations showed trends that were not statistically significant ($\alpha=0.05$), the change rate range at most stations fell between -10% and +10%. While these values likely contain uncertainty due to differing monitoring periods across sites and short monitoring periods at some sites, the general condition for those sites with significant trends are typically small and are decreasing between 0 and -2% per year in the PWM concentration of mercury.

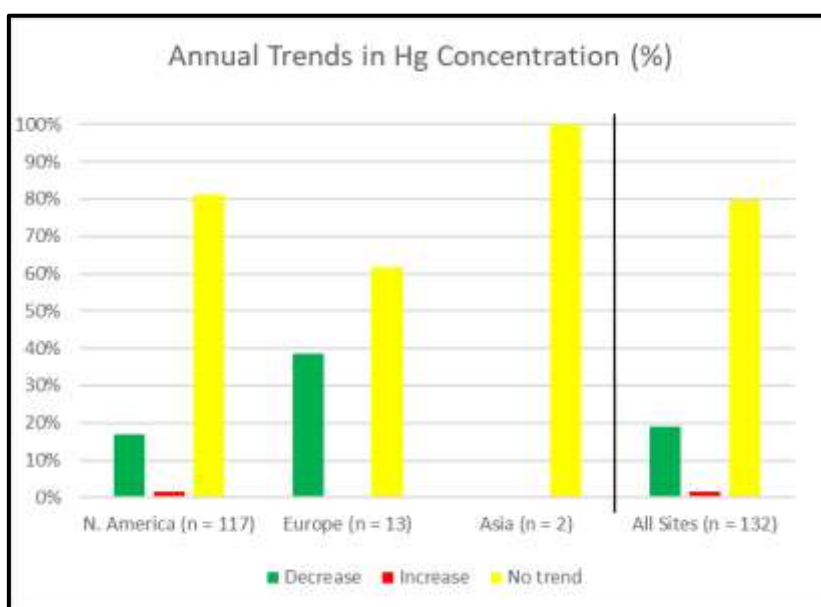


Figure Air 7: Annual trends in precipitation weighted mercury concentrations in precipitation in North America, Europe and Asia. “n” is the number of sites included in the analysis. Only stations with at least 5 years data are included.

Wet deposition and precipitation:

Regardless of concentration trends and potential trends in precipitation causing the concentration trends, it is instructive to look at deposition trends alone. From an ecological perspective, depositional trends are most important whether they are due to changes in mercury concentration, precipitation trends, or a combination of both. If more mercury is depositing, from an ecological perspective, the reason does not really matter. Figure Air 8 shows the annual trends in wet deposition of mercury.



Figure Air 8: Annual trends in wet deposition of mercury in precipitation.

Most sites show no significant trend in wet mercury deposition over their measurement time periods. Approximately 87% of the sites have no trend in either direction including Europe, North America, and Japan. A small fraction of the sites in central and northern Europe, as well as central and eastern North America show decreasing trends in deposition. For Europe, this observation could be driven by local source reductions, since concentration decreases are seen here as well. Most sites in Europe show no significant change in precipitation; thus, a decrease in precipitation is not likely to be the cause, particularly in central Europe. Two of the northern European sites have both significant decreases in precipitation and deposition, and; therefore, the decrease in precipitation is likely driving the decrease in wet deposition. One site showed both decreasing concentrations and decreasing precipitation leading to overall decreasing wet deposition. Two other sites in Europe had decreasing deposition rates at the 90% confidence level.

In Asia, no trends were found in the 2 sites in Japan for wet deposition. For these sites, no trend in precipitation were reported along with steady wet deposition and neither site showed trends even at the 90% confidence level.

No significant trends in wet deposition were found in western North America, except for one high elevation site that had increasing concentration. That site did not show a trend in precipitation, suggesting an increasing trend in actual mercury content. East of the 90th meridian, most of the sites have no changes in wet deposition at 95% confidence. So, the general conclusion is that there is no regional decrease of wet deposition of mercury in eastern North America. Five sites are situated along the US/Canada border that show significant increases in wet deposition. Several of these sites have increasing precipitation rates, suggesting more precipitation is driving some of these higher deposition rates. Figure Annex Air 6 shows the spatial distribution of precipitation trends. Overall, most sites have no significant trends in precipitation except those noted in northern North America in which some of these sites correspond to similar trends in wet deposition.

Overall, trends in wet deposition are driven by concentration and precipitation amounts. There is some regional variation (*i.e.* central Europe and eastern North America) but it is not consistent. With very little to no data in Asia, no regional conclusions can be drawn. Longer sampling over time would provide improved (statistically significant) information about the time trends for wet deposition of mercury. The data for wet deposition is currently limited to primarily North America and Europe although there is data from Asia that was not submitted that would provide improved trend information.

2.3.7 How do the changes in (TGM/GEM) observed in air between circa 2010 and the present compare to established national and international guidance or levels identified in recent scientific literature that are associated with adverse effects on human health, wildlife and environmental sustainability?

Information Used (including applicable indigenous and traditional knowledge)

- Air data - from circa 2010 to present day.
- Chronic level values compiled in Ciani et al. (2024).

Methodology

- Compare observed values to threshold level values
- Consider chronic = time-average of 1 month or longer
- Acute exposure (8-hr avg or shorter) can only be evaluated for data w/ hourly resolution

Answer/Response

Various threshold limit values for air have been established by international environmental health agencies for chronic and acute exposure. These range from 30-10,000 ng m⁻³ for chronic (long-term) exposure, 60-2,200,000 ng m⁻³ for 8-hour average exposure, and 600-16,000,000 ng m⁻³ for shorter acute exposures (Ciani et al., 2024).

Circa 2010, there was only one measurement site (Harwell, UK) with submitted data values that exceeded the acute exposure guidance levels. However, comparison of the submitted data to the official UK data portal showed these values were flagged during validation and should have been removed prior to submission. Four other sites in the data submitted occasionally had hourly values that exceeded 60 ng m⁻³. These sites are Flin Flon, Canada; Oga, Japan; Birmingham, UK and Indiana Dunes National Lakeshore, USA. However, these high levels did not experience a sustained period of 8 hours or more as would be required to exceed the California Environmental Protection Agency (EPA) 8-hour average Reference Exposure Level (REL) and no sites exceeded the 600 ng m⁻³ California EPA hourly REL¹⁶. Chronic thresholds were never exceeded at any site in the circa 2010 period.

In the present-day period (2022-2024), no sites with hourly data approached the acute exposure guideline values. However, this may be a limitation of the dataset as time averages of the data varied. In Peru, both chronic and acute exposure guidance levels were exceeded. This is a region where artisanal and small-scale gold mining (ASGM) using mercury is often practiced. Measurements in this region were only available as weekly or multi-week averages, yet the recorded average values were higher than thresholds designed for 1-hour, 8-hour or daily average exposure. Annex Air Table 5 shows the number of measurements (weekly or multi-week average) from Peruvian sites designated as “mining towns” that exceeded the various exposure guidance levels. As well, this table shows the percent of the total “mining towns” dataset that exceeded each guidance level. Note that the measurements were made in different locations over a region spanning ~150 km in distance and on different dates and with different sampling periods over a 2-year period. The numbers in Annex Air Table 5 should be considered indicative only.

In Diawala, Côte d’Ivoire and Kenyase, Ghana chronic exposure guidance levels for mercury were exceeded where ASGM using mercury is practiced. Diawala is a very large ASGM site where mercury is used in large quantities during the gold extraction process (Kpokro, 2025). At Diawala, GEM levels averaged 55.5 ng m⁻³ over a 12-month measuring period (Nov 2021 - Oct 2022). These levels exceed the California EPA 30 ng m⁻³ chronic REL (defined based on annual average) (Figure Air 9a). In one month (Dec 2022), GEM at Diawala exceeded the 300 ng m⁻³ limit considered by the US Environmental Protection Agency (US EPA) as the Reference Concentration¹⁷. The fact that the monthly average exceeds the daily exposure limits highlights the severity of the exposure. At Kenyase, measurements were available only as quarterly (3-month) averages. Measured GEM exceeded 30 ng m⁻³ chronic REL in 3 of the 6 samples collected over an 18-month measuring period. However, the rolling 12-month mean never exceeded the California EPA 30 ng m⁻³ chronic REL (defined on an annual basis) (Figure Air 9b).

¹⁶ REL defined as: “the concentration at or below which adverse health effects are not likely to occur in the general human population” EPA, California (2014) as cited in Ciani et al, 2024.

¹⁷ defined as ‘An estimate (with uncertainty spanning perhaps an order of magnitude) of a daily inhalation exposure of the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime,’ (US EPA 2003, as cited in Ciani et al., 2024) and by the US Department of Health and Human Services Agency for Toxic Substances and Disease Registry (ATSDR) as the Minimal Risk Level, defined as ‘An estimate of daily human exposure to a substance that is likely to be without an appreciable risk of adverse effects (noncarcinogenic) over a specified duration of exposure’ (ATSDR 2022, as cited in Ciani et al., 2024).

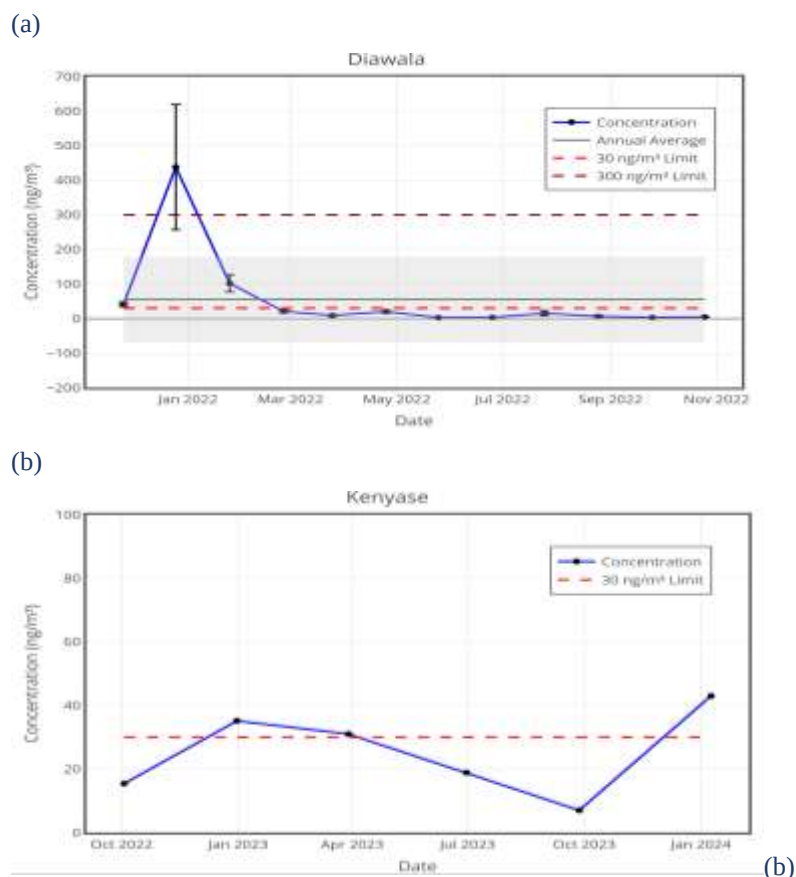


Figure Air 9: Monthly mean GEM at (a) Diawala, Côte d'Ivoire and (b) Kenyase, Ghana. Vertical lines indicate the standard deviations (Diawala only). The grey line and shading in (a) show the annual mean and standard deviation. The dashed lines indicate established guidance levels for chronic exposure: CalEPA Reference Exposure Level (red) and US EPA Reference Concentration / ATSDR Minimal Risk Level (dark red).

This data shows that in areas of ASGM activity exceedances to health guidelines are found. However, there is insufficient monitoring data available from locations with atmospheric mercury pollution to fully answer the question.

In the circa 2010 period, no sites exceeded acute 1-hour or 8-hour average exposure guidance levels or chronic levels. Of the 4 sites that had occasional hourly values near the 8-hour average limit, only two have measurements available in the present-day period. At these two sites, no values in the present day period approach the exceedance levels, potentially indicating that changes in (TGM/GEM) observed in air between circa 2010 and the present have decreased the likelihood of exceeding established national and international guidance or levels.

At locations where chronic and/or acute exceedances were recorded in the present-day dataset (Diawala and Kenyase), there were no measurements available from the circa-2010 period to answer the question. Monitoring at these sites is not ongoing and; thus, no conclusions can be drawn as to changes occurring.

Consistency of Answer with Existing Literature (especially the 2018 GMA)

- Not covered in GMA

Evaluation of Confidence

- Confidence is low - there are insufficient observations from GEM/TGM emission source locations to understand whether changes from circa 2010 to present day have increased or decreased the likelihood and frequency of exceedances of established guideline values.
- Chronic and acute exposure levels have been exceeded at ASGM sites during the present day period, but without historical or ongoing measurements, the direction of change cannot be ascertained.

Information and Capacity Needs

- Near-source, ongoing GEM/TGM measurements, especially in ASGM regions where values well above chronic exposure guideline levels have already been recorded.

References

ATSDR. Toxicological Profile for Mercury (Draft for Public Comment); U.S. Department of Health and Human Services, Public Health Service: Atlanta, GA, USA, 2022.

Bally, K. D. F. K. Contaminated Networks: Mercury Supply, Entrepreneurship, and Informal Gold Mining in Rural Sub-Saharan Africa. University of Surrey Surrey, UK, 2025.

CalEPA. Technical Support Document for the Derivation of Noncancer Reference Exposure Levels; Office of Environmental Health Hazard Assessment, California Environmental Protection Agency: San Francisco, CA, USA, 2014.

Ciani, F.; Costagliola, P.; Lattanzi, P.; Rimondi, V. Gaseous Mercury Limit Values: Definitions, Derivation, and the Issues Related to Their Application. *Sustainability* 2024, 16 (8), 3142. <https://doi.org/10.3390/su16083142>.

US EPA. A Review of the Reference Dose and Reference Concentration Processes; US EPA/630/P-02/002F, December 1, 2002; Risk Assessment Forum: Washington, DC, USA, 2003; 192p.

AMAP. (2005). *Arctic Monitoring and Assessment Programme (AMAP) Assessment 2002: Heavy Metals in the Arctic*.

AMAP. (2011). *AMAP Assessment 2011: Mercury in the Arctic* (ISBN - 978-82-7971-068-4). AMAP. <https://www.amap.no/documents/doc/amap-assessment-2011-mercury-in-the-arctic/90>

AMAP, & Environment, U. (2019). *Technical Background Report to the Global Mercury Assessment 2018* (ISBN- 798-82-7971-108-7). United Nations Environment Programme/Arctic Monitoring and Assessment Programme.

Ariya, P. A., Amyot, M., Dastoor, A., Deeds, D., Feinberg, A., Kos, G., Poulain, A., Ryjkov, A., Semeniuk, K., Subir, M., & Toyota, K. (2015). Mercury Physicochemical and Biogeochemical Transformation in the Atmosphere and at Atmospheric Interfaces: A Review and Future Directions. *Chemical Reviews*, 115(10), 3760-3802. <https://doi.org/10.1021/cr500667e>

Bencardino, M., D'Amore, F., Angot, H., Angiuli, L., Bertrand, Y., Cairns, W., Diéguez, M. C., Dommergue, A., Ebinghaus, R., Esposito, G., Komínková, K., Labuschagne, C., Mannarino, V., Martin, L., Martino, M., Neves, L. M., Mashyanov, N., Magand, O., Nelson, P.,...Pirrone, N. (2024). Patterns and trends of atmospheric mercury in the GMOS network: Insights based on a decade of measurements. *Environmental Pollution*, 363, 125104. <https://doi.org/https://doi.org/10.1016/j.envpol.2024.125104>

Ciani, F., Costagliola, P., Lattanzi, P., & Rimondi, V. (2024). Gaseous Mercury Limit Values: Definitions, Derivation, and the Issues Related to Their Application. *Sustainability*, 16, 3142. <https://doi.org/10.3390/su16083142>

Cole, A. S., & Steffen, A. (2010). Trends in long-term gaseous mercury observations in the Arctic and effects of temperature and other atmospheric conditions [Article]. *Atmospheric Chemistry and Physics*, 10(10), 4661-4672. <https://doi.org/10.5194/acp-10-4661-2010>

Cole, A. S., Steffen, A., Eckley, C. S., Narayan, J., Pilote, M., Tordon, R., Graydon, J. A., St. Louis, V. L., Xu, X., & Branfireun, B. A. (2014). A survey of mercury in air and precipitation across canada: Patterns and trends [Article]. *Atmosphere*, 5(3), 635-668. <https://doi.org/10.3390/atmos5030635>

Dastoor, A., Angot, H., Bieser, J., Brocza, F., Edwards, B., Feinberg, A., Feng, X., Geyman, B., Gournia, C., He, Y., Hedgecock, I. M., Ilyin, I., Kirk, J., Lin, C. J., Lehnher, I., Mason, R., McLagan, D., Muntean, M., Rafaj, P.,...Zolkos, S. (2025). The Multi-Compartment Hg Modeling and Analysis Project (MCHgMAP):

mercury modeling to support international environmental policy. *Geosci. Model Dev.*, 18(9), 2747-2860. <https://doi.org/10.5194/gmd-18-2747-2025>

Dastoor, A., Wilson, S. J., Travníkov, O., Ryjkov, A., Angot, H., Christensen, J. H., Steenhuisen, F., & Muntean, M. (2022). Arctic atmospheric mercury: Sources and changes [Article]. *Science of the Total Environment*, 839, Article 156213. <https://doi.org/10.1016/j.scitotenv.2022.156213>

Ebinghaus, R., Jennings, S. G., Kock, H. H., Derwent, R. G., Manning, A. J., & Spain, T. G. (2011). Decreasing trends in total gaseous mercury observations in baseline air at Mace Head, Ireland from 1996 to 2009. *Atmospheric Environment*, 45(20), 3475-3480. <https://doi.org/10.1016/j.atmosenv.2011.01.033>

ESRI. (2025). *ArcGIS Pro*. In

Gichuki, S. W., & Mason, R. P. (2013). Mercury and metals in South African precipitation. *Atmospheric Environment*, 79, 286-298. <https://doi.org/10.1016/j.atmosenv.2013.04.009>

Gilbert, R. O. (1987). *Statistical Methods for Environmental Pollution*. John Wiley & Sons.

Kpokro, B. K. D. F. (2025). *Contaminated Networks: Mercury Supply, Entrepreneurship, and Informal Gold Mining in Rural Sub-Saharan Africa* University of Surrey].

MacSween, K., Stupple, G., Aas, W., Kyllönen, K., Pfaffhuber, K. A., Skov, H., Steffen, A., Berg, T., & Mastromonaco, M. N. (2022). Updated trends for atmospheric mercury in the Arctic: 1995–2018 [Article]. *Science of the Total Environment*, 837, Article 155802. <https://doi.org/10.1016/j.scitotenv.2022.155802>

MacSween, K., Stupple, G., Aas, W., Kyllönen, K., Pfaffhuber, K. A., Skov, H., Steffen, A., Berg, T., & Mastromonaco, M. N. (2022). Updated trends for atmospheric mercury in the Arctic: 1995–2018. *Science of the Total Environment*, 837, 155802. <https://doi.org/10.1016/j.scitotenv.2022.155802>

McLagan, D. S., Mitchell, C. P. J., Huang, H., Lei, Y. D., Cole, A. S., Steffen, A., Hung, H., & Wania, F. (2016). A High-Precision Passive Air Sampler for Gaseous Mercury [Article]. *Environmental Science and Technology Letters*, 3(1), 24-29. <https://doi.org/10.1021/acs.estlett.5b00319>

McLagan, D. S., Mitchell, C. P. J., Steffen, A., Hung, H., Shin, C., Stupple, G., Olson, M. L., Luke, W. T., Kelley, P., Howard, D., Edwards, G. C., Nelson, P. F., Xiao, H., Sheu, G.-R., Dreyer, A., Huang, H., Hussain, B. A., Lei, Y. D., Tavshunsky, H., & Wania, F. (2018). Global evaluation and calibration of a passive air sampler for gaseous mercury *Atmospheric Chemistry and Physics*, 18, 5905-5919. <https://doi.org/10.5194/acp-18-5905-2018>

McLagan, D. S., Monaci, F., Huang, H., Lei, Y. D., Mitchell, C. P. J., & Wania, F. (2019). Characterization and Quantification of Atmospheric Mercury Sources Using Passive Air Samplers [Article]. *Journal of Geophysical Research: Atmospheres*, 124(4), 2351-2362. <https://doi.org/10.1029/2018JD029373>

Naccarato, A., Tassone, A., Martino, M., Moretti, S., MacAgnano, A., Zampetti, E., Papa, P., Avossa, J., Pirrone, N., Nerentorp, M., Munthe, J., Wängberg, I., Stupple, G. W., Mitchell, C. P. J., Martin, A. R., Steffen, A., Babi, D., Prestbo, E. M., Sprovieri, F., & Wania, F. (2021). A field intercomparison of three passive air samplers for gaseous mercury in ambient air [Article]. *Atmospheric Measurement Techniques*, 14(5), 3657-3672. <https://doi.org/10.5194/amt-14-3657-2021>

Prestbo, E. M., & Gay, D. A. (2009). Wet deposition of mercury in the U.S. and Canada, 1996–2005: Results and analysis of the NADP mercury deposition network (MDN). *Atmospheric Environment*, 43(27), 4223-4233. <https://doi.org/10.1016/j.atmosenv.2009.05.028>

Roberts, S. L., Kirk, J. L., Muir, D. C. G., Wiklund, J. A., Evans, M. S., Gleason, A., Tam, A., Drevnick, P. E., Dastoor, A., Ryjkov, A., Yang, F., Wang, X., Lawson, G., Pilote, M., Keating, J., Barst, B. D., Ahad, J. M. E., & Cooke, C. A. (2021). Quantification of Spatial and Temporal Trends in Atmospheric Mercury Deposition across Canada over the Past 30 Years [Article]. *Environmental Science and Technology*, 55(23), 15766-15775. <https://doi.org/10.1021/acs.est.1c04034>

Soerensen, A. L., Jacob, D. J., Streets, D., Witt, M., Ebinghaus, R., Mason, R. P., Andersson, M., & Sunderland, E. M. (2013). Hg0 trends in the North and South Atlantic. E3S Web of Conferences,

Sprovieri, F., Pirrone, N., Bencardino, M., D'Amore, F., Angot, H., Barbante, C., Brunke, E. G., Arcega-Cabrera, F., Cairns, W., Comero, S., Diéguez, M. D. C., Dommergue, A., Ebinghaus, R., Feng, X. B., Fu, X., Garcia, P. E., Gawlik, B. M., Hageström, U., Hansson, K.,...Zhang, H. (2017). Five-year records of mercury wet deposition flux at GMOS sites in the Northern and Southern hemispheres. *Atmos. Chem. Phys.*, 17(4), 2689-2708. <https://doi.org/10.5194/acp-17-2689-2017>

Sprovieri, F., Pirrone, N., Bencardino, M., D'Amore, F., Carbone, F., Cinnirella, S., Mannarino, V., Landis, M., Ebinghaus, R., Weigelt, A., Brunke, E. G., Labuschagne, C., Martin, L., Munthe, J., Wängberg, I., Artaxo, P., Morais, F., Barbosa, H. D. M. J., Brito, J.,...Norstrom, C. (2016). Atmospheric mercury concentrations observed at ground-based monitoring sites globally distributed in the framework of the GMOS network. *Atmos. Chem. Phys.*, 16(18), 11915-11935. <https://doi.org/10.5194/acp-16-11915-2016>

Sprovieri, F., Pirrone, N., Ebinghaus, R., Kock, H., & Dommergue, A. (2010). A review of worldwide atmospheric mercury measurements. *Atmospheric Chemistry and Physics*, 10(17), 8245-8265. <http://www.scopus.com/inward/record.url?eid=2-s2.0-77956330069&partnerID=40&md5=6ab0ac98a27f3f5300b1252aab68eefb>

Steffen, A., Bottenheim, J., Cole, A., Douglas, T. A., Ebinghaus, R., Friess, U., Netcheva, S., Nghiem, S., Sihler, H., & Staebler, R. (2013). Atmospheric mercury over sea ice during the OASIS-2009 campaign. *Atmospheric Chemistry and Physics*, 13(14), 7007-7021. <https://doi.org/10.5194/acp-13-7007-2013>

Streets, D. G., Horowitz, H. M., Lu, Z., Levin, L., Thackray, C. P., & Sunderland, E. M. (2019). Global and regional trends in mercury emissions and concentrations, 2010–2015 [Article]. *Atmospheric Environment*, 201, 417-427. <https://doi.org/10.1016/j.atmosenv.2018.12.031>

UNEP: *Global Mercury Assessment*. (2002). (United Nations Environment Programme Chemicals, Issue.

Weiss-Penzias, P. S., Gay, D. A., Brigham, M. E., Parsons, M. T., Gustin, M. S., & ter Schure, A. (2016). Trends in mercury wet deposition and mercury air concentrations across the U.S. and Canada [Article]. *Science of the Total Environment*, 568, 546-556. <https://doi.org/10.1016/j.scitotenv.2016.01.061>

2.4 **Biota**

2.4.1 **What data are available?**

2.4.1.1 Received and Acquired Data

The OESG received 57 submissions on total and methyl mercury measurements across a range of taxa, including invertebrates, amphibians, reptiles, fish, birds, mammals, and plants. Most of the long-term biota time-series are associated with monitoring of different tissues and organs of marine species (invertebrates, fish, seabirds and higher trophic level species, including marine mammals) or freshwater fish and terrestrial birds. Data on mercury concentrations in amphibians (3 observations), reptiles (18 observations) and plants (28 observations excluding mosses) remains limited; data on mercury in moss are available and were collected mainly to evaluate atmospheric mercury deposition. For the purpose of the first cycle of the effectiveness evaluation scientific report, the data analysis considered only total mercury measurements in invertebrates, fish, birds, and marine mammals.

Several multinational monitoring programs, including the Arctic Monitoring and Assessment Programme (AMAP), the OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR), the Helsinki Commission (HELCOM), and the Caribbean Region Mercury Monitoring Network (CRMMN), together with national and subnational organizations from Canada, the United States, Norway, France, Finland, Switzerland, the Netherlands, the United Kingdom, Germany, Peru, Australia, Sweden, Japan and other countries contributed data to the temporary OESG repository. The Biodiversity Research Institute (BRI) provided data spanning 37 years, derived from a database of about 1648 published articles and scientific research projects, covering a diverse range of taxa and tissue types. The collected data represent a compilation of repeated monitoring activities and scientific research projects (Annex 1).

While many biota datasets were submitted in accordance with the OESG biota template facilitating data integration and analysis, many submissions lacked key metadata, such as trophic level classifications, anthropogenic source impacts, size or age of organisms, or standardized coding (e.g., countries, ecosystems, tissues) either because the information was not collected or because of a lack of capacity to integrate it into the submission. In addition, data were not required to be submitted in the template leading to discrepancies in data included and requiring resources to enable integration with other datasets. These issues suggest that the template may require further refinement to improve data consistency, completeness, and usability and to better facilitate integration of new datasets with existing data archives.

Figure 1 shows sampling locations of submitted data, including both individual mercury measurements and aggregated data (i.e., mean values). Subsets of the submitted data visualized in Figure 1 were used in analyses. Only datasets, both from the published literature and direct submissions, meeting minimum data requirements (e.g., appropriate spatial or temporal resolution, ancillary data for standardization, etc.) for an analysis were used for answering a particular question. Many submitted data were unsuitable or found not to meet requirements for answering the types of questions being posed in the effectiveness evaluation. This highlights the need to align scientific mercury research, as well as the critical role of national and international monitoring programs in ensuring comprehensive and reliable data collection.

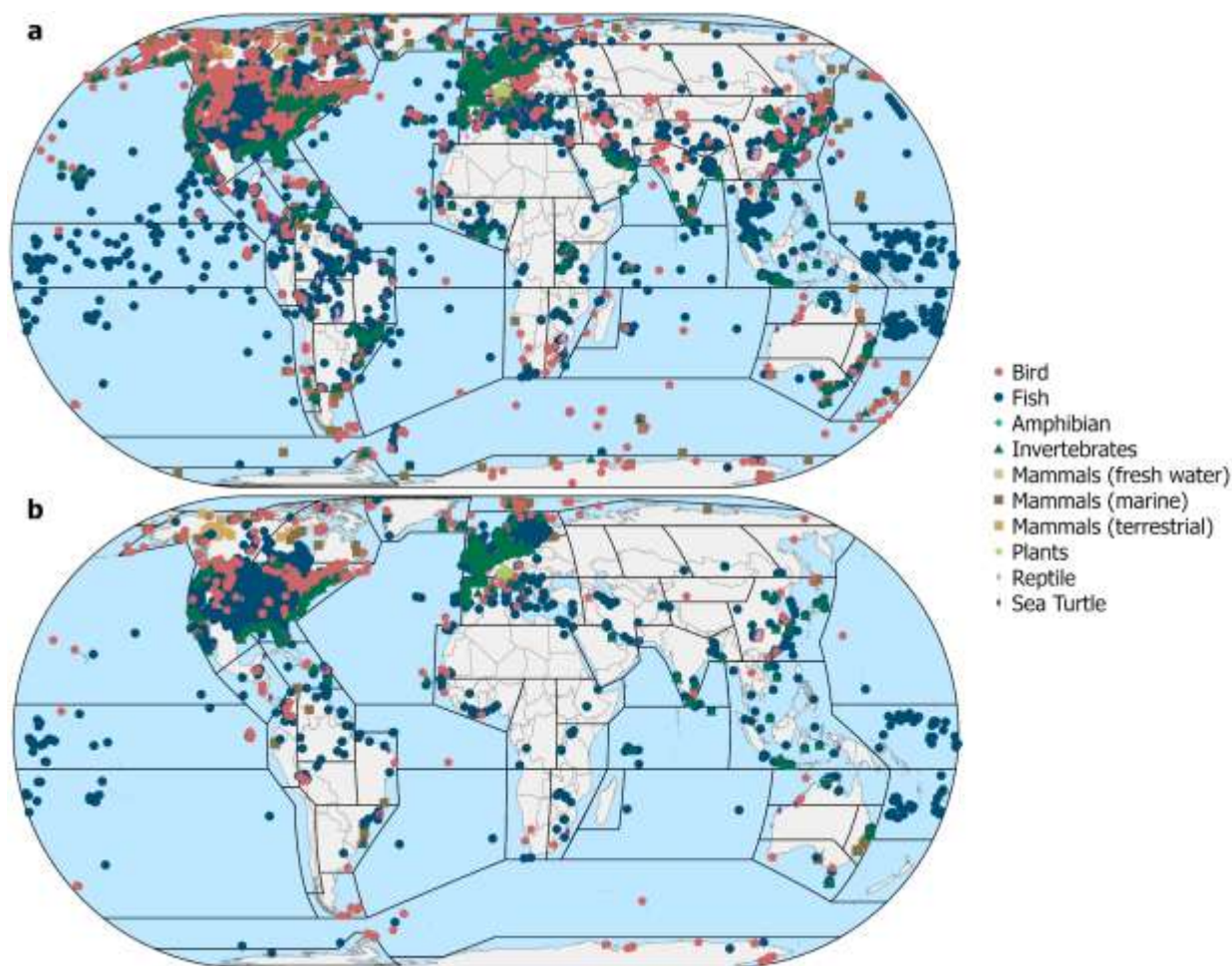


Figure 1. Geographic distribution of a) all sampling locations by taxa, overlaid on IPCC regional boundaries (black lines) and b) sampling locations in current period (2014-2024). A subset of suitable samples were identified for use in each analysis.

Following the collation of long-term data submitted to the OESG, significant geographic gaps have been identified in several regions, including the Pacific, Asia, western and central Africa, and Central and South America. Data available for assessing temporal trends remain especially limited.

Robust monitoring of temporal trends in biota generally requires long-term, consistent sampling programmes, ensuring standardized field, sampling, and analytical procedures to minimize sources of variability that could confound temporal trend interpretation. Several mercury biomonitoring programmes provide excellent examples of national or regional field monitoring focused on both human and ecological health (e.g., AMAP, Canada's Northern Contaminants Program, OSPAR, HELCOM, ARCTOX, and national networks in Canada, United States, Japan, European Union, and others). The experience from these monitoring programmes has shown that, for time series based on annual sampling, a period of more than 10 years is typically required to reliably detect the expected level of annual change at sites not directly affected by local sources, even when changes in mercury levels are in the range of 5-10% per year.

Although numerous peer-reviewed articles are publicly available, the extraction and standardization of these data were limited by time and personnel constraints. Nevertheless, key publications and reports have been identified and are supporting the interpretation of submitted data and the analysis of temporal trends.

2.4.1.2 Identified data that has not been acquired

The OESG has received intentions from various data holders, including international and national institutions, to submit datasets on mercury concentrations in invertebrates, fish, birds, mammals, and plants, that have not yet been formally submitted. Additionally, some known datasets remain inaccessible or are not in a standardized format. The missing data includes both local studies and broader national or regional programs across the Arctic, North America, Europe, Asia, Africa, and Australia. The datasets span from historical to ongoing monitoring, with timeframes ranging from the 1960s to the present.

2.4.2 Question 1: What are current levels of Hg in invertebrates, fish, birds, and mammals globally?

Understanding current mercury levels in biota is essential for evaluating progress toward meeting the objectives of the Minamata Convention. The fate and transport of mercury emitted or released into the environment is governed by complex chemical conversions and cycling. Consequently, the mercury that is bioavailable depends on both the mercury inputs and the methylation potential of a given system. Biotic mercury concentrations directly reflect that bioavailable fraction providing insight into spatial and temporal patterns and the magnitude of mercury body burdens in wildlife and humans.

- Current levels were defined as total mercury concentrations measured between **2014 and 2024** based on the following considerations:
 - Mercury levels in wildlife tend to change slowly, making decadal data more informative than short-term fluctuations.
 - Many countries began taking mercury-reduction actions prior and following the 2013 adoption of the Minamata Convention, even before the agreement's formal entry into force in 2017.
 - Short-term data are often limited due to infrequent sampling cycles (e.g., biota sampling typically occurs annually or biennially, depending on monitoring requirements).
- The Intergovernmental Panel on Climate Change (IPCC) regions were applied to create a consistent geographic framework across different biota matrices and to support spatial and integrated analyses where appropriate.

2.4.2.1 Invertebrates

Information Used:

Data were compiled from 17 submissions to the OESG repository (Annex 1, Figure 2).

The majority of invertebrate (n=15,305/16,414, 93%) data were provided by regional seas monitoring programmes covering the NE Atlantic (OSPAR Commission, with data co-reported to AMAP in overlapping Arctic waters), the Baltic Sea (Helsinki Commission, HELCOM), and national/sub-national monitoring programmes including the US-NOAA National Centers for Coastal Ocean Science Mussel Watch program, the Nyrstar Hobart/Derwent Estuary Program (Tasmania), and the Japanese national monitoring program.

If additional time and resources had been available, some further consideration of the data reported from other invertebrates, including cephalopods, crustacea and gastropods, would have been possible. However, for most of these groups the spatial coverage or low numbers of observations for individual species would have likely severely limited the value of any analysis in the context of the objectives of the effectiveness evaluation.

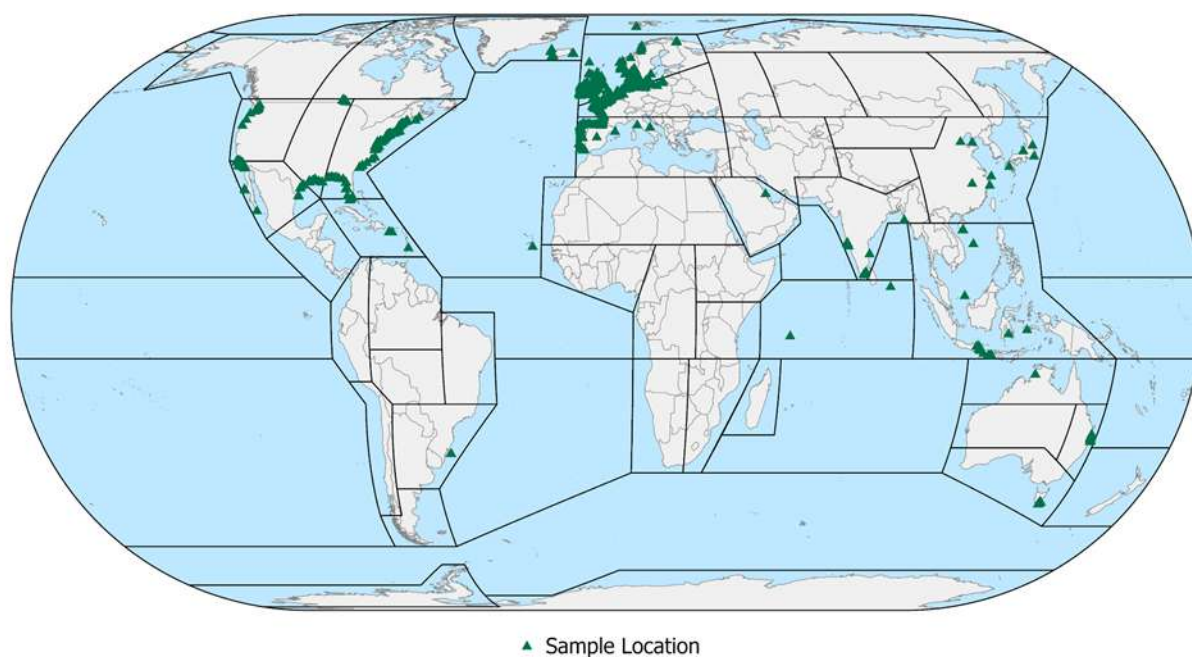


Figure 2: Geographic distribution of 2014-2024 sampling locations of invertebrate samples submitted, overlaid on IPCC regional boundaries.

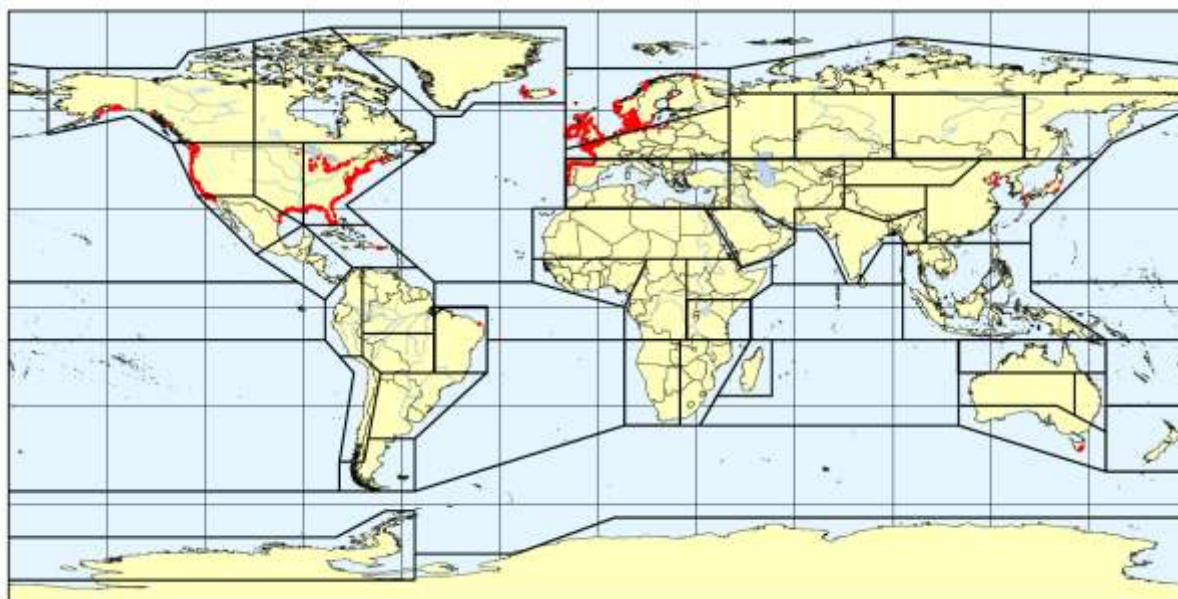


Figure 3: Map of locations of monitoring sites for the selected invertebrate species (red dots) for the period 2014-2023 in IPCC Regions.

Methodology:

Inclusion criteria: Observations were included if they: a) belonged to one of the four most common bivalve genera (*Mytilus*, *Crassostrea*, *Dreissena*, and *Ostrea*); b) reported individual or arithmetic mean total Hg (THg) concentrations (Mercury.species.measured* was coded as THg, Hg, or “NA” which was assumed to be THg); c) represented whole soft tissue concentrations (Matrix_consolidated coded as Whole organ(s), Whole, or Soft tissue); d) reported uncensored THg concentrations > 1 ng g⁻¹ dw.

Exclusion criteria: Observations were excluded if they: a) belonged to genera other than the four most common genera described above; b) reported individual or arithmetic mean MMHg concentrations; c) represented tissues other than whole soft tissue (i.e., Matrix_consolidated coded as Muscle, Other, or Unspecified); d) reported THg concentrations that were missing, censored by data suppliers (i.e., Qualifier* coded as “<”, “D”, or “Q”), or < 1 ng g⁻¹ dw.

Of the n=450 censored observations, n=414 were from OSPAR, n=22 were from Nyrstar Hobart/Derwent Estuary Program, and n=14 were from HELCOM. Limits of detection or quantification were listed for n=33 of these observations; all others were censored without an accompanying limit term. N=103 censored observations were for the period 2014-2024. Of these, n=17 censored observations were reported on a wet weight basis and n=86 were reported on a dry weight basis. Reported concentrations for dry weight basis observations ranged from 2-180 ng/g dw, with overlapping but relatively low concentrations compared to non-censored observations for this time period (see below). These values were excluded because censoring by data suppliers indicated lack of confidence in the reported values.

An additional n = 14 observations were excluded because they were assumed to be QA/QC samples (i.e., they were not assigned to an IPCC Region and had Site.Name = “Control”). Excluding observations on these bases returned a core dataset with n = 13,842 observations.

Data standardization and harmonization: The resulting core invertebrate dataset included n = 11,709 observations reported on a dry weight basis and n = 2,135 reported on a wet weight basis. Wet weight basis concentrations were converted to a dry weight basis assuming a percent moisture (% moisture) content of 86.5%. This assumption was based on the mean % moisture among the three bivalve genera for which observations were reported on a dry weight basis with an accompanying %moisture value (n = 3,269 observations). Wet weight concentrations were converted to dry weight concentrations as follows:

$$\text{dw concentration} = \text{wet weight concentration} / [(100-86.5)/100] = \text{wet weight concentration} / 0.135$$

Answer/Response:

Bivalves are generally considered useful bioindicators for coastal contamination monitoring because they are sessile (i.e., representative of the conditions from where they are harvested), abundant, widely distributed, and easy to sample. They have been employed for these reasons in a number of chemical monitoring programs since the 1970s around the world, including NOAA Mussel Watch (USA), the OSPAR Commission, and the Helsinki Commission.

Representativeness of the invertebrate datasets

There were 16,414 invertebrate observations representing 338 genera or species, of which the majority (92% of all “Invertebrate” observations) were bivalves belonging to four genera *Mytilus*, *Crassostrea*, *Dreissena*, and *Ostrea*. An additional 61 bivalve genera/species are relatively infrequently represented in the dataset (i.e., only representing n=493 additional observations). The analyses presented here focus on *Mytilus*, *Crassostrea*, *Dreissena*, and *Ostrea* as the most common Invertebrate genera among the submitted data. Of the 13,842 observations retained in the core Invertebrate dataset, n= 3,486 (25.2%) pertain to the period 2014-2023 (there were no observations for 2024).

Of the 13,842 observations retained in the core Invertebrate dataset, n= 3,486 (25.2%) pertain to the period 2014-2023 (there were no observations for 2024). The contemporary subset of the harmonized dataset represents 17 countries and 12 IPCC Regions (Table 1). The number of observations by IPCC Region ranges from 3 (Caribbean) to 2,028 (N.Europe). Observations from the Global North are dominant. The number of sites sampled by IPCC Region ranges from 1 (N.Atlantic-Ocean, a single site on the boundary with the N. Europe region so not representative of the N. Atlantic Ocean) to 348 (N.Europe) (Figure 3). It is important to note, therefore, that summaries by IPCC Region are only representative of the sampling locations submitted for the OESG and are not representative of the IPCC region as a whole. In some cases, the majority of data submitted for an entire IPCC Region are from monitoring in impacted sites (e.g., S. Australia data are primarily from the Derwent estuary in Tasmania which is impacted by historical activity of a large zinc smelter and paper mill). In general, it was not possible to evaluate the extent to which data in other IPCC regions are potentially impacted by local sources as in most cases this information was not provided; however, the regional seas programmes responsible for most of the data considered are typically designed to monitor background or representative sites rather than impacted sites.

Summary of current total mercury concentrations in bivalves

Across regions and species, THg concentrations ranged over two orders of magnitude from 10.2 to 2,070 ng g⁻¹ dw. IPCC Region-allocated mean $\pm$ SD THg concentrations (regardless of species) ranged from 50.5 $\pm$ 33 ng g⁻¹ dw (n=11) in E. Asia to 542 $\pm$ 270 ng g⁻¹ dw (n=299) in S. Australia. THg concentrations also varied within and among IPCC regions by species.

The evaluated bivalve data are not considered suitable for evaluation of spatial patterns on a global scale due to the limited geographical coverage (see Figure 3). Based on the number of sites monitored and the data available, spatial comparisons, even within a single region would only be warranted along near-coastal areas in a few cases (N. Europe, W. and Central Europe, East and West North America). For these regions spatial comparisons are made under the auspices of assessments conducted by OSPAR (see <https://oap.ospar.org/en/ospar-assessments/quality-status-reports/qsr-2023/thematic-assessments/hazardous-substances/>), HELCOM (see <https://indicators.helcom.fi/indicator/mercury/>) and the US-EPA.

The ranges of mercury concentrations reported in bivalves for the period 2014-2023 are relatively similar across the 12 IPCC regions for which data were available (Figure 3). Boxplots (Figure 4) indicate that concentrations in mussels and oysters are somewhat elevated in the S. Australia IPCC region, reflecting the predominance of data from monitoring sites affected by zinc smelting activities within the region. If the S. Australia region is excluded, mean THg values (based on reported measurements in pooled samples or averages of aggregated data) are in the range 50.5 $\pm$ 33 to 235 $\pm$ 530 ng g⁻¹ dw (Table 1).

It was not possible from the submitted biota data to reliably differentiate impacted and non-impacted sites as few of the reported data included this information. Mercury concentrations in the higher range of values in West and Central Europe and IPCC Mediterranean regions are associated with oysters. It is noted here that the IPCC Mediterranean region is characterized by samples at sites along the Atlantic coast of France, Portugal and Spain with no samples from the Mediterranean Sea itself, an example of the care that needs to be taken with respect to conclusions regarding ‘geographical context’. Mercury levels in the lower range of values are reported for mussels in the IPCC East Asia and Greenland/Iceland regions (samples from sites in Iceland only). Lower values at Arctic sites, generally considered remote from major anthropogenic source regions, may be expected; however, the lower values (for mussels in particular) at East Asian sites in Japan and northern China are notable, and possibly related to the different species monitored. Representing a region based on data from a very few sites is a general concern for almost all environmental monitoring data evaluated for the effectiveness evaluation, with implications also for work aiming to integrate results for different environmental matrices.

Table 1: THg concentrations (ng/g dw) by IPCC Region, bivalve species, and tissue type for the period 2014-2023. Data are summarized for sampling locations shown in Figure 3 (map) and should not be used to characterize the entire region as named.

IPCC_Name	Species	THg concentration (ng/g dw)			N obs	N sites
		mean	SD	median		
C.North-America	<i>Crassostrea virginica</i>	113	54	90.6	12	12
Caribbean	<i>Crassostrea rhizophorae</i>	95.9	9.8	100	3	3
E.Asia	<i>Mytilus galloprovincialis</i>	50.5	33	44.4	11	3
E.North-America	<i>Crassostrea virginica</i>	176	150	156	72	68
	<i>Mytilus edulis</i>	105	99	52	67	58
Greenland/Iceland	<i>Mytilus edulis</i>	62.6	26	60	37	10
Mediterranean	<i>Crassostrea gigas</i>	218	60	217	43	7
	<i>Mytilus edulis</i>	136	140	84	195	42
N. Atlantic Ocean	<i>Ostrea edulis</i>	155	79	108	8	1
N. Central-America	<i>Crassostrea virginica</i>	152	96	137	7	7
	<i>Mytilus californianus</i>	106	91	71	19	19
N. Europe	<i>Crassostrea gigas</i>	137	51	140	30	5
	<i>Mytilus edulis</i>	130	100	101	2028	342
	<i>Ostrea edulis</i>	130	100	100	10	2
S. Australia ¹	<i>Crassostrea gigas</i>	587	310	519	181	34
	<i>Mytilus edulis</i>	474	190	444	118	30
W.North-America	<i>Mytilus californianus</i>	235	530	90.6	14	14
	<i>Mytilus sp.</i>	64.2	39	48.2	17	17
West & Central-Europe	<i>Crassostrea gigas</i>	210	68	205	216	46
	<i>Dreissena polymorpha</i>	50.8	10	50	36	6
	<i>Mytilus edulis</i>	140	79	125	347	52
	<i>Mytilus galloprovincialis</i>	136	25	140	15	3

¹All S. Australia data are from the Nyrstar Hobart/Derwent Estuary Program monitoring Hg trends in an area impacted by historical activity of a large zinc smelter and paper mill. The data are therefore not representative of this IPCC region as a whole. These were the only data identified as being from sites with point sources, though others could be included in this dataset.

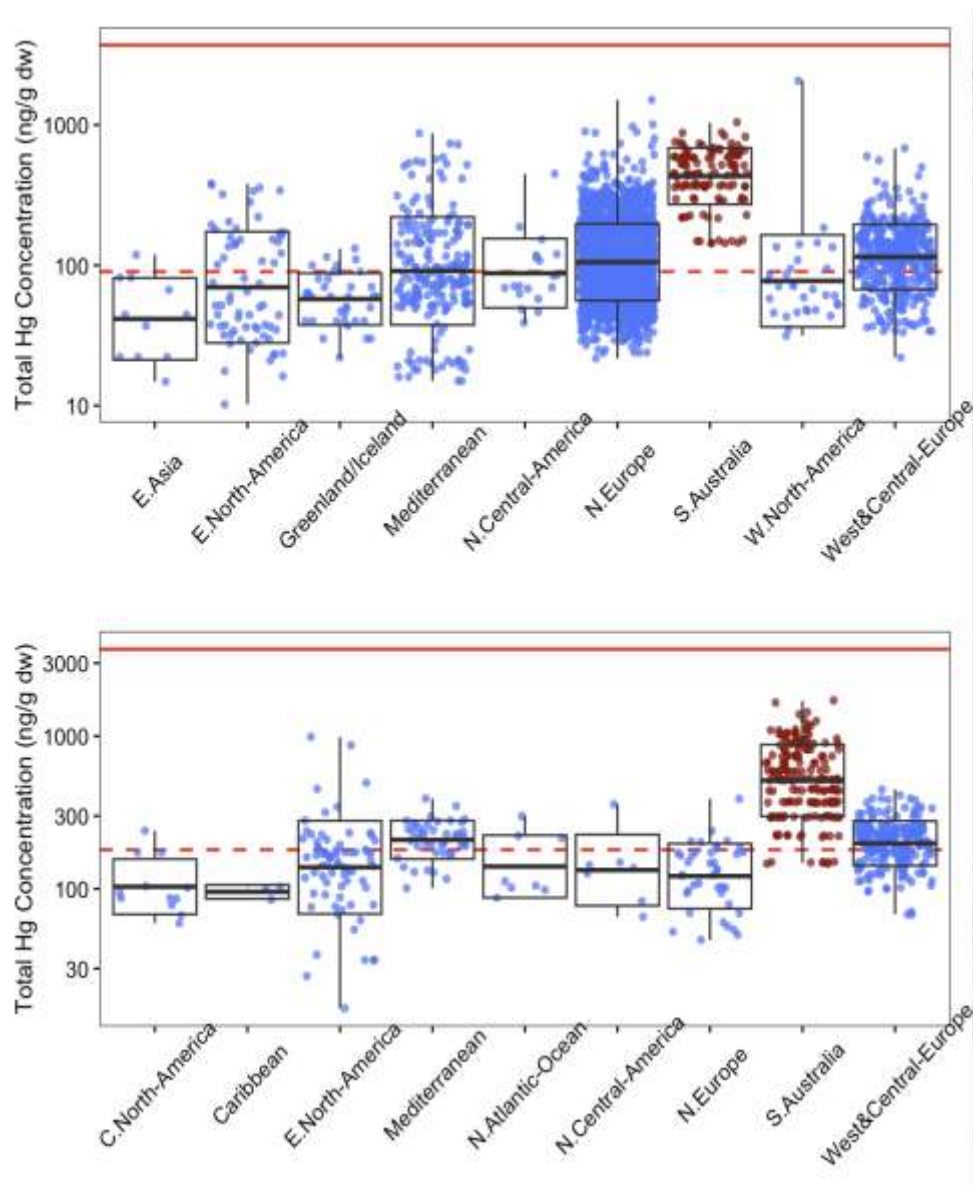


Figure 4: THg concentrations (ng/g dw) for the current period (2014-2023) by IPCC Region. Boxplots show min, mean – SD, mean, mean + SD, and max concentrations. Points represent individual observations. The top panel shows data for mussel species and the bottom panel shows data for oyster species. Horizontal dashed red lines represent Background Assessment Concentrations (BAC) below which concentrations are considered to be near background for the Northeast Atlantic region by the OSPAR Commission (OSPAR Commission, 2009); the BAC for mussels is 90 ng/g dw and the BAC for oysters is 180 ng/g dw. Horizontal solid red lines represent the Maximum Permissible Concentration (MPC) in food for the protection of public health for the Northeast Atlantic region by the OSPAR Commission (OSPAR Commission, 2009); the MPC value is 500 ng/g ww converted to 3704 ng/g dw assuming the average % moisture for bivalves in the presented dataset of 86.5%. Colors indicate whether data were associated with Hg point sources (red points). All S. Australia data are from the Nyrstar Hobart/Derwent Estuary Program monitoring Hg trends in the vicinity of a large zinc smelter and paper mill. The data are therefore not representative of this IPCC region as a whole. These were the only data identified as being from sites with point sources, though others could be included in this dataset. Data are summarized for sampling locations shown in Figure 1 (map) and should not be used to characterize the entire region as named.

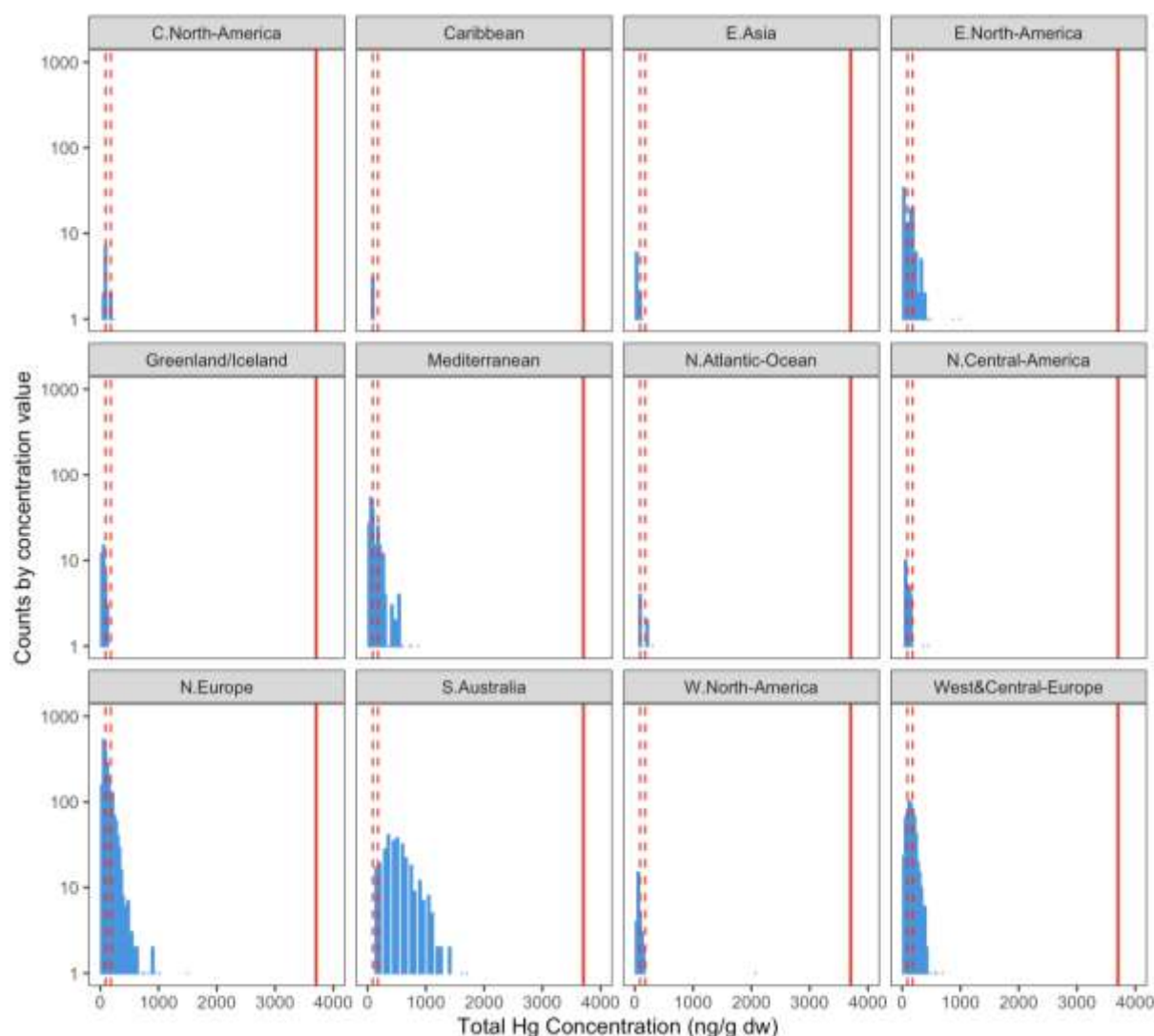


Figure 5: Histograms show relative counts of observations by THg concentration per IPCC Region for the period 2014-2023. Vertical dashed red lines represent Background Assessment Concentrations below which concentrations are considered to be near background by the OSPAR Commission (OSPAR Commission, 2009), 90 ng/g dw for mussels and 180 ng/g dw for oysters. The solid red lines represents the Maximum Permissible Concentration in food for the protection of public health for the Northeast Atlantic region by the OSPAR Commission (OSPAR Commission, 2009); the MPC value is 500 ng/g ww converted to 3704 ng/g dw assuming the average %moisture for bivalves in the presented dataset of 86.5%.

Consistency of Answer with Existing Literature:

Mercury concentrations in bivalves were not considered in the 2018 Global Mercury Assessment. The data evaluated herein include much of the data from ongoing monitoring programs, including HELCOM, OSPAR, US EPA, AMAP, and Australian datasets. Not all data are synthesized in publications, but many are included in monitoring program reports. Comparison with regional assessments by HELCOM, OSPAR, and US EPA are provided below in section 0.1.4.1

Evaluation of Confidence:

There is reasonable confidence in the quality of data included in invertebrate assessments presented here, as most data are from established monitoring programs. There is inherent uncertainty related to converting concentrations between dry and wet weight bases, especially given that none of the observations reported in wet weight units had

accompanying measures of percent moisture. Samples were instead converted to dry weight using the average percent moisture reported alongside the dry weight observations of the four most common genera (*Mytilus*, *Crassostrea*, *Dreissena*, and *Ostrea*) in the data set. As percent moisture can vary among species and as a function of storage time, this assumption introduces uncertainty to the converted values reported here. The evaluated bivalve data are not considered suitable for evaluation of spatial patterns on a global scale due to the limited geographical coverage (see Figure 3). Based on the number of sites monitored and the data available, spatial comparisons, even within a single region would only be warranted along near-coastal areas in a few cases (N. Europe, W. and Central Europe, East and West North America). However, even for IPCC regions represented here, there are concerns with representing a region based on data from a very few sites. It was also not possible from the submitted biota data to reliably differentiate impacted and non-impacted sites as few of the reported data included this information.

Information and Capacity Needs:

Enhanced monitoring is required in underrepresented regions, particularly Africa (AF), Latin America and the Caribbean (LAC) increasing both the frequency and resolution of sampling will improve the temporal and spatial resolution of mercury assessments.

Continued investment in data-sharing infrastructure and promotion of cross-sector collaboration will improve accessibility and integration of datasets across regions and monitoring networks.

2.4.2.2 Fish

Information Used:

Fish data were compiled from 30 submissions to the OESG repository (Annex 1). The fish dataset represents 87 countries (in several cases reported through national or regional monitoring programmes) and 48 IPCC Regions. The number of observations by IPCC Region ranges from one (W. Antarctica and W. Southern-Africa) to 30,181 (E. North-America). The number of individuals by IPCC Region ranges from two (Tibetan-Plateau) to 90,126 (N. Europe). Observations are not homogeneously distributed at a global scale, with the Global North much more heavily represented and significant data gaps in the Global South. The number of sites (defined by unique coordinates rounded to three decimal places to account for inconsistent site naming conventions across datasets and years) sampled by IPCC Region ranges from 1 (Bay-of-Bengal, W. Antarctica, W. Siberia, W. Southern-Africa) to 3,230 (E. North-America) (Figure 6).

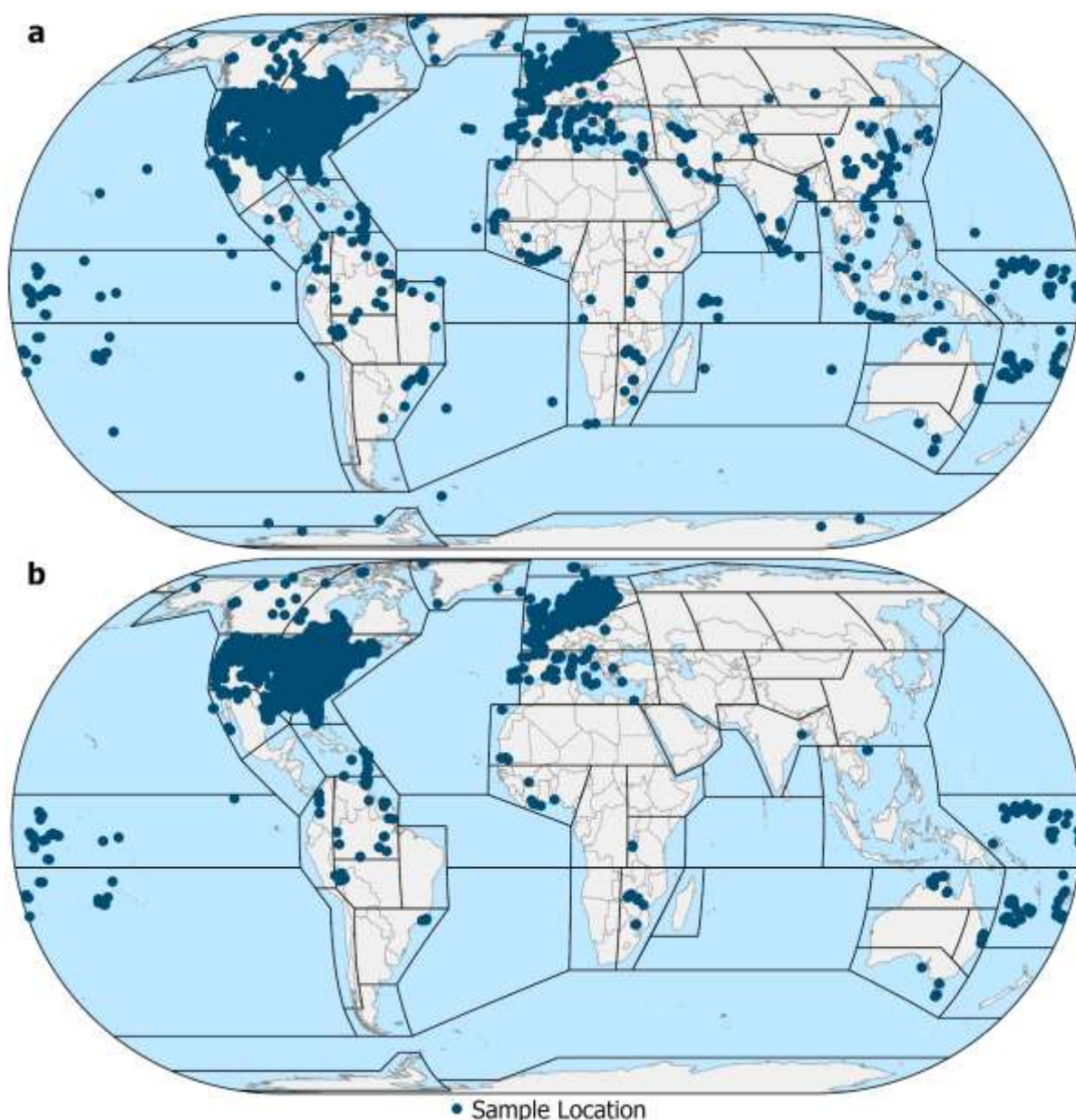


Figure 6: Geographic distribution of fish records from 2014-2024 **a)** included in the OESG repository, and **b)** filtered using the inclusion/exclusion criteria (see methodology), overlaid on IPCC regional boundaries.

Methodology:

Inclusion criteria: The full fish mercury dataset was filtered to include global submissions collected from 2014 to 2024. Submissions encompassed individual measurements, pooled arithmetic or geometric means, medians, or minimum and maximum values, reported on either a dry-weight or wet-weight basis. Only individual THg records or arithmetic means with valid mercury concentrations (THg > 0, not coded as NA) were considered for analysis.

Most of the qualified data came from muscle samples (91.7%) and whole specimens (4.8%), with the remaining data from eggs, liver, skin, organs, blood, or unknown matrices also included in smaller proportions. Only records reporting muscle THg concentrations were considered here, with other matrices not converted to a standard tissue type. Most submitted data were reported on a wet-weight basis (90.1%) followed by dry-weight measurements (7.9%) with the small remaining fraction of an unknown basis. Both wet weight and dry weight records were considered.

To ensure statistical robustness, only fish species with at least ten qualified records within each IPCC region were included in IPCC-level analyses. These species were further categorized by feeding guild (herbivorous, planktivorous, omnivorous, and carnivorous) and classified as marine or freshwater according to IPCC land-type categories.

Exclusion criteria: At the same time, strict exclusion criteria were applied to maintain data quality. Records reporting negative values, zeros, or missing values for THg were removed. Because more than 90% of records were fish muscle, we excluded all other matrices to avoid uncertainty from tissue conversions. All records noted as “Canned; Processed” were also excluded. Samples lacking critical spatial information, such as IPCC regional attribution, were also excluded from pertinent analyses. In addition, records from species represented by fewer than 10 qualified records in an IPCC region (records from a species with fewer records in an IPCC region were discarded even if records from other IPCC regions were kept) were not included in the main species comparisons.

Data standardization and harmonization: Prior to analysis, data were standardized to ensure comparability across diverse sources. All mercury values were converted into a consistent unit, nanograms per gram wet weight, with adjustments made where necessary. Results presented here are for muscle tissue on a wet-weight basis, which represented the vast majority of observations. Dry-weight observations were converted to wet-weight using the reported moisture content when available, or assuming 80% moisture content for fish muscle when moisture data were not reported.

Fish species were aligned using standardized taxonomy and assigned to families and feeding guilds using established references. Spatial information was harmonized through IPCC regional classifications.

Finally, harmonization procedures brought together the diverse records into a single coherent master dataset. Different sample matrices were consolidated, and taxonomic and geographic groupings ensured comparability across monitoring networks. The 2014–2024 time span was selected for “current levels” across taxa, with subsets of data suitable for annual monitoring identified for specific regions. Summary statistics such as means, medians, and percentiles were calculated, and visualizations such as boxplots and tables were prepared to enable meaningful comparisons across species, families, feeding guilds, and geographic regions.

Answer/Response:

Fish are widely used and well-established bioindicators of Hg bioaccumulation because they reflect not only their own exposure but also the exposure of the predators, including wildlife and humans, that consume them. In addition, fish can be classified into well-defined feeding guilds and trophic levels across ecosystems, allowing comparisons of Hg bioaccumulation among specific ecological groups (e.g., benthivores vs. pelagic feeding, different trophic levels) across geographic areas. Additionally, available fish Hg data are among the most abundant globally, making large-scale assessments more feasible and cost-effective. And lastly, fish total Hg concentrations are representative of methylmercury (MeHg) concentrations. Since MeHg is the toxic and most bioavailable and bioaccumulative form, combined with their global consumption, fish are particularly representative of MeHg exposures.

Representativeness of the fish datasets

The harmonized fish subset (species with 10 or more records in an IPCC region were retained) includes 70,000 records from 199 fish species across 67 families. Regional representation remained limited due to a strong dominance of data from North America and Europe while Africa, South Asia, and Latin America and the Caribbean were underrepresented.

The number of individuals by IPCC Region ranges from 38 (S.E.South-America) to 84,000 (N.Europe). Observations by IPCC Region range from 12 (S.E.Asia) to 25,753 (E.North-America), highlighting that observations from the Global North are dominant. The number of sites sampled by IPCC Region ranges from 2 (N.Pacific-Ocean, Equatorial.Atlantic-Ocean) to 2,690 (E.North-America).

Summary of current total mercury concentrations in fish

Across regions and species, THg concentrations in fish muscle samples included in the harmonized fish subset ranged over five orders of magnitude from 0.04 to 16,400 ng g⁻¹ ww. IPCC Region-specific mean ± SD muscle

THg concentrations (regardless of species) ranged from $7.18 \pm 2.54 \text{ ng g}^{-1} \text{ ww}$ (n=12) in S.E. Asia to $2,944.7 \pm 1,716.5 \text{ ng g}^{-1} \text{ ww}$ (n=108) in S.Eastern-Africa. The high muscle THg values in S. Eastern-Africa were mainly attributed to a specific study (Pascal et al. 2020) of ASGM. Other IPCC regions with elevated mean muscle THg were typically driven by intensive shark sampling (e.g., Le Croizier et al., 2020). THg concentrations also varied within and among IPCC regions by species (Table 2a,b).

Table 2: Summary of current THg concentrations ($\text{ng g}^{-1} \text{ ww}$) by feeding guild in the harmonized fish muscle subset across fish species with ≥ 10 records per IPCC Region for the period 2014-2024

Statistic	Individual THg Concentration (ng/g ww)				
	Carnivorous	Herbivorous	Omnivorous	Planktivorous	Total
n (records)	55,688	761	8,450	5,101	70,000
n (families)	45	6	25	7	67
n (species)	119	11	56	13	199
Minimum	0.0	1.1	0.4	0.6	0.0
5th percentile	41.0	5.1	19.4	14.0	28.5
Median	260.0	25.8	99.3	80.0	208.0
Mean	409.1	70.3	164.7	114.1	354.4
95th percentile	1,200.0	222.9	440.0	330.0	1,100.0
Maximum	16,400.0	1,436.9	7,788.0	1,000.0	16,400.0

Table 3a. Summary of current THg concentrations ($\text{ng g}^{-1} \text{ ww}$) by feeding guild across IPCC Regions in the harmonized fish muscle subset across fish species with ≥ 10 records per IPCC Region for the period 2014-2024

IPCC Name	Feeding Guild	n (records)	Individual THg Concentration (ng/g ww)		
			Mean	SD	Median

Arctic-Ocean	Carnivorous	131	71.0	92.0	35.0
C.North-America	Carnivorous	6,653	607.9	458.6	490.0
	Omnivorous	1,407	141.1	110.3	110.0
	Planktivorous	989	154.5	114.6	120.0
Caribbean	Carnivorous	228	230.5	530.7	103.4
E.Australia	Carnivorous	60	1,697.6	1,241.2	1,649.0
E.North-America	Carnivorous	19,518	455.6	446.9	330.0
	Omnivorous	4,678	136.8	131.8	92.5
	Planktivorous	1,557	140.9	130.0	100.0
E.Southern-Africa	Carnivorous	18	248.5	192.5	167.0
	Omnivorous	120	60.4	94.8	26.5
Equatorial.Atlantic-Ocean	Carnivorous	375	279.7	186.3	202.1
Equatorial.Pacific-Ocean	Carnivorous	126	205.5	402.1	52.4
Greenland/Iceland	Carnivorous	268	242.0	366.5	109.5
Mediterranean	Carnivorous	1,114	696.4	833.8	447.5
	Omnivorous	157	397.4	417.8	264.2
	Planktivorous	12	139.9	250.0	48.3
N.Atlantic-Ocean	Carnivorous	43	2,942.3	2,288.0	2,350.0
N.Australia	Carnivorous	988	207.8	141.0	172.0

	Planktivorous	11	24.2	12.1	26.1
N.Central-America	Carnivorous	70	667.5	652.9	453.2
	Omnivorous	114	89.7	83.3	66.5
N.E.North-America	Carnivorous	5,618	607.5	532.5	450.0
	Omnivorous	1,151	152.4	134.3	120.0
	Planktivorous	1,194	142.2	106.9	110.0
N.Europe	Carnivorous	16,141	211.3	220.5	150.0
	Planktivorous	1,027	28.4	19.7	22.1
N.Pacific-Ocean	Carnivorous	60	2,115.9	524.5	2,105.7
N.South-America	Carnivorous	327	362.7	436.4	157.9
	Omnivorous	10	186.4	127.5	180.0
	Planktivorous	11	22.5	10.1	22.0
	Herbivorous	27	124.7	80.0	100.0
N.W.North-America	Carnivorous	1,145	265.9	167.5	237.0
	Omnivorous	14	69.4	46.2	51.0
	Planktivorous	11	37.2	10.3	40.7
N.W.South-America	Carnivorous	285	107.4	155.0	75.3
	Omnivorous	346	48.7	79.0	27.0
	Herbivorous	235	26.1	46.4	16.8
S.Asia	Herbivorous	13	108.2	71.8	74.0

S.Australia	Carnivorous	441	738.0	1,077.1	390.0
	Omnivorous	45	914.0	481.5	730.0
S.E.Asia	Carnivorous	12	7.2	2.5	6.5
S.E.South-America	Carnivorous	38	966.7	2,807.4	196.5
S.Eastern-Africa	Carnivorous	36	3,370.7	1,914.6	3,034.0
	Omnivorous	72	2,731.7	1,579.7	2,367.5
S.Pacific-Ocean	Carnivorous	275	243.5	266.7	142.2
Sahara	Carnivorous	40	1,219.8	664.2	1,165.0
	Omnivorous	23	51.5	42.8	44.2
South-American-Monsoon	Carnivorous	500	412.6	1,064.8	144.5
	Omnivorous	224	134.6	354.8	60.2
	Planktivorous	10	119.8	95.5	93.6
	Herbivorous	486	87.6	197.8	34.9
W.North-America	Carnivorous	363	117.8	167.3	70.5
	Omnivorous	62	116.0	92.2	83.6
West&Central-Europe	Carnivorous	815	89.4	108.8	63.8
	Planktivorous	279	25.3	11.8	23.0
Western-Africa	Omnivorous	27	74.4	50.0	67.0

Table 3b. Summary of mercury (Hg) concentrations ($\text{ng g}^{-1} \text{ ww}$) in fish species across IPCC regions within the harmonized fish muscle subset. Only species with ≥ 10 records per IPCC region for the period 2014–2024 are included. For each region, the table presents summary statistics for the two species with the highest number of records

IPCC_Name	Species	n (records)	Individual THg Concentration (ng/g ww)		
			mean	sd	median
Arctic-Ocean	<i>Gadus morhua</i>	91	31.0	21.3	26.0
	<i>Salvelinus alpinus</i>	40	162.1	122.2	117.1
C. North-America	<i>Sander vitreus</i>	1,939	728.6	526.7	580.0
	<i>Esox lucius</i>	1,842	662.4	466.0	550.0
Caribbean	<i>Coryphaena hippurus</i>	65	150.2	94.9	129.9
	<i>Acanthocybium solandri</i>	42	102.0	163.5	52.7
E. Australia	<i>Carcharodon carcharias</i>	47	2,099.1	1,101.3	1,904.0
	<i>Acanthopagrus australis</i>	13	246.2	107.6	240.0
E. North-America	<i>Sander vitreus</i>	4,998	581.1	455.6	470.0
	<i>Esox lucius</i>	3,852	550.6	419.0	430.0
E. Southern-Africa	<i>Clarias gariepinus</i>	57	68.1	109.4	31.1
	<i>Oreochromis niloticus</i>	49	22.6	9.9	22.0
Equatorial.Atlantic-Ocean	<i>Thunnus obesus</i>	201	330.7	219.1	237.6
	<i>Thunnus albacares</i>	174	220.9	114.4	182.9
	<i>Katsuwonus pelamis</i>	109	62.6	50.5	47.7

Equatorial.Pacific-Ocean	<i>Carcharhinus falciformis</i>	17	1,122.0	463.2	948.0
Greenland/Iceland	<i>Salvelinus alpinus</i>	254	253.6	373.0	123.0
	<i>Gadus morhua</i>	14	30.7	6.5	31.0
Mediterranean	<i>Merluccius merluccius</i>	442	119.9	341.7	52.6
	<i>Thunnus thynnus</i>	175	902.5	502.2	774.0
N. Atlantic-Ocean	<i>Prionace glauca</i>	43	2,942.3	2,288.0	2,350.0
N. Australia	<i>Lates calcarifer</i>	988	207.8	141.0	172.0
	<i>Nematalosa erebi</i>	11	24.2	12.1	26.1
N. Central-America	<i>Ariopsis felis</i>	44	124.6	76.2	104.0
	<i>Bagre marinus</i>	19	250.4	170.7	218.0
N.E. North-America	<i>Esox lucius</i>	1,980	631.1	605.8	410.0
	<i>Sander vitreus</i>	1,946	667.5	488.9	550.0
N. Europe	<i>Perca fluviatilis</i>	10,602	211.6	174.1	165.0
	<i>Gadus morhua</i>	1,990	125.5	115.5	92.0
N. Pacific-Ocean	<i>Carcharodon carcharias</i>	60	2,115.9	524.5	2,105.7
N. South-America	<i>Micropogonias furnieri</i>	42	185.0	131.4	154.1
	<i>Scomberomorus cavalla</i>	27	99.5	38.6	96.6
N.W. North-America	<i>Salvelinus namaycush</i>	524	297.1	148.6	269.5

	<i>Lota lota</i>	388	282.1	166.3	238.0
N.W. South-America	<i>Prochilodus magdalenae</i>	138	29.3	49.3	18.7
	<i>Hypostomus hondae</i>	97	21.6	41.8	11.3
S. Asia	<i>Labeo rohita</i>	13	108.2	71.8	74.0
S. Australia	<i>Platycephalus bassensis</i>	401	446.3	284.5	360.0
	<i>Acanthopagrus butcheri</i>	45	914.0	481.5	730.0
S.E. Asia	<i>Trachinotus ovatus</i>	12	7.2	2.5	6.5
S.E. South-America	<i>Rhizoprionodon lalandii</i>	22	1,548.9	3,610.7	222.5
	<i>Rhizoprionodon porosus</i>	16	166.3	88.5	139.5
S. Eastern-Africa	<i>Haplochromis spp.</i>	36	3,035.6	1,726.3	2,734.0
	<i>Oreochromis spp.</i>	36	2,427.8	1,375.5	2,185.5
S. Pacific-Ocean	<i>Thunnus albacares</i>	115	135.6	83.3	113.2
	<i>Katsuwonus pelamis</i>	101	132.9	61.8	122.5
Sahara	<i>Prionace glauca</i>	40	1,219.8	664.2	1,165.0
	<i>Clarias gariepinus</i>	12	68.1	42.9	53.2
South-American-Monsoon	<i>Hoplias malabaricus</i>	159	478.1	1,327.4	184.8
	<i>Potamorhina altamazonica</i>	139	149.5	300.8	55.5
W. North-America	<i>Leptocottus armatus</i>	69	26.1	12.2	25.4

	<i>Paralichthys californicus</i>	56	95.3	41.1	90.5
West&Central-Europe	<i>Perca fluviatilis</i>	276	63.2	24.2	61.2
	<i>Clupea harengus</i>	257	25.8	11.8	23.4
Western-Africa	<i>Tilapia spp.</i>	16	88.7	51.4	74.0
	<i>Sarotherodon melanotheron</i>	11	53.5	41.5	55.0

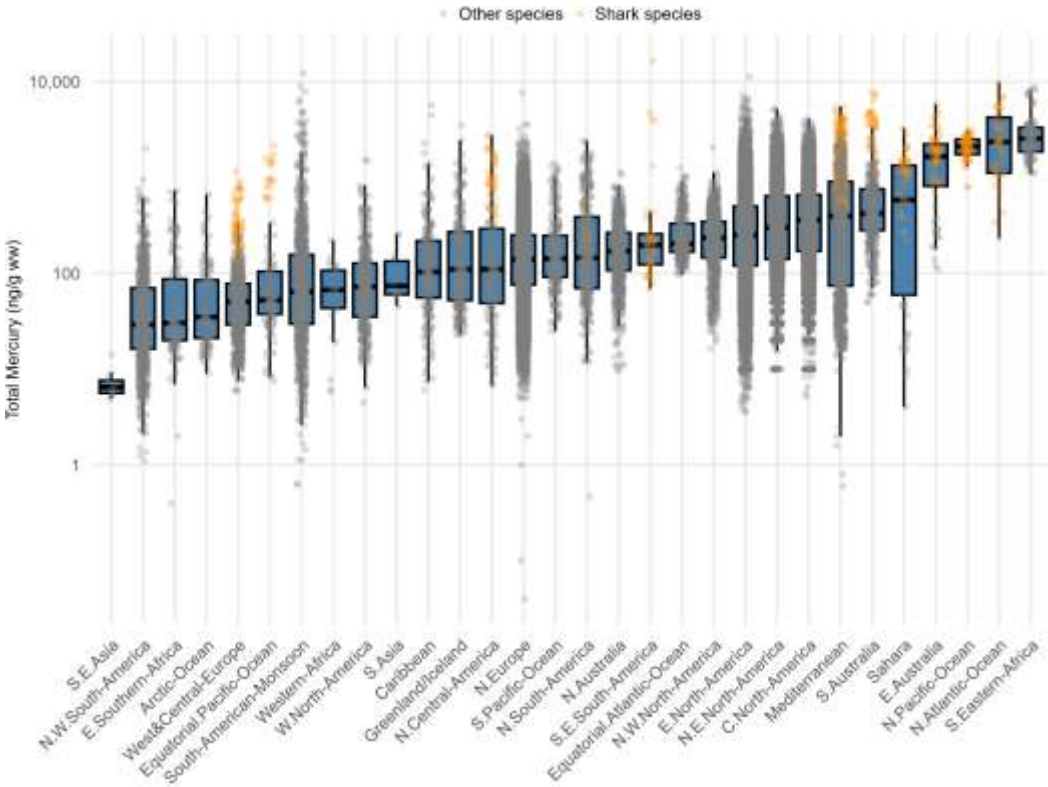


Figure 7. Total mercury (THg) concentrations (ng g^{-1} wet weight) in fish muscle collected during the current period (2014–2024) across IPCC regions within the harmonized fish muscle subset. Only species with ≥ 10 records per IPCC region for the period 2014–2024 are included. Boxplots display the median (central line) and interquartile range (boxes), with whiskers extending to $1.5 \times$ the interquartile range. Individual points represent all measured observations, with color indicating fish group (shark in orange vs. other species in gray).

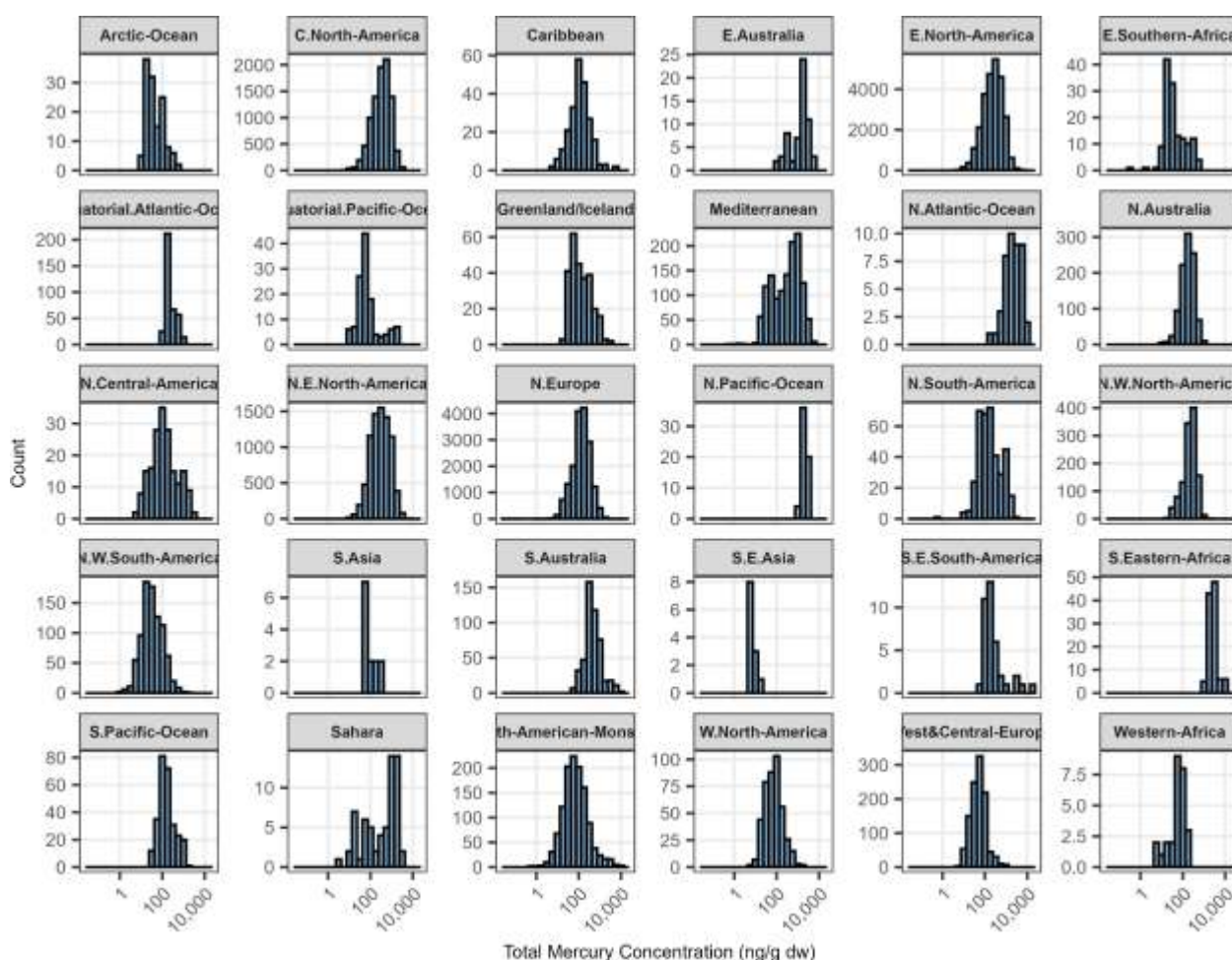


Figure 8: Histograms showing the distribution of total mercury (THg) concentrations in fish muscle within the harmonized fish muscle subset by IPCC region during 2014–2024. Only species with ≥ 10 records per IPCC region for the period 2014–2024 are included. The x-axis is displayed on a log10 scale, while the y-axis indicates the count of observations.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

The biota data included in the Global Mercury Assessment (GMA) were compiled from peer-reviewed studies and therefore reflect the highly specific spatial and temporal contexts each study was designed to address as opposed to an integrated global picture. The GMA fish data incorporate both freshwater and marine fish of a wide range of trophic levels and biomes – from the Arctic to tropical regions – based on published research conducted, often, independently over many decades. The GMA presented and discussed data spatially and temporally for specific species groups and case studies. However, the raw data from the underlying studies was inconsistently available, and when it was not the GMA relies on aggregated or average values resulting in reduced understanding of within-study variability.

This OESG report encountered many of the same issues as the GMA, and incorporated the underlying data. However, it compiles monitoring data collected over a specific, defined, and contemporary period (2014–2024) to evaluate spatial variability in Hg concentrations in fish. The spatial coverage of the incorporated datasets is highly variable, a pattern also highlighted in the GMA. Additionally, there were only limited areas and programs with available and consistent temporal data that could be compared resulting in temporal analyses being done as case studies much like the GMA. Because the GMA aggregated data from multiple decades, it does not set a baseline for Hg in fish in the specific decade (2014–2024) of interest to the OESG. Like the GMA, the data coverage in this report is dominated by the northern hemisphere (North America and Europe) and poorly represents the Southern Hemisphere and Asia.

Evaluation of Confidence:

There is high confidence in the robustness of fish mercury concentration data from North America and Europe, where large numbers of qualified records spanning multiple species and years provide reliable temporal and spatial baselines. Confidence is also relatively strong for specific long-term monitoring sites in Latin America and Australia, where sustained and intensive sampling allow for meaningful analysis.

High confidence in the internal consistency of the dataset is further supported by the systematic standardization of mercury values to a consistent basis (wet-weight), harmonization of taxonomy into species, families, and feeding guilds, and consistent use of IPCC regional classifications. Most records derive from muscle tissue, a standard matrix for fish Hg monitoring, which further strengthens comparability.

However, there is lower confidence in regional representativeness arising from significant spatial gaps. Observational coverage is lacking in Africa, South Asia, and parts of the Pacific, where available data are sparse or absent. Even within better-covered continents, certain IPCC regions have limited records, reducing the strength of regional comparisons.

Regional comparisons are also challenging due to the heterogeneous sampling objectives of the individual studies or monitoring program integrated for this OESG report. In IPCC regions with sparse data, results may be disproportionately influenced by sampling focus or protocol of a single study. For example, fish THg data from some IPCC regions consist entirely of shark samples, while other regions are composed mostly of low-trophic level herbivores. Similarly, IPCC regions with a heavy proportion of studies targeting understanding point source pollution (e.g., ASGM sites, industrial contamination) may not have data reflective of the whole IPCC region, especially when reference areas were not also included. Additionally, inconsistent metadata collection and reporting (e.g., length, age) further complicates standardization and reduces confidence in regional comparisons.

QA/QC procedures included removal of invalid or biologically implausible records, correction of unit errors, and filtering out of zero or negative values. High confidence can therefore be placed in the quality of the cleaned dataset, although some uncertainty remains due to the limited metadata accompanying certain submissions (e.g., missing geographic coordinates for some records).

Methodological robustness is reinforced by the harmonization of data across multiple monitoring networks, ensuring comparability of Hg measurements despite diverse data sources. Confidence is particularly high in the statistical summaries (means, medians, percentiles) and groupings by species, feeding guilds, and regions, which enable consistent cross-comparison. However, lower confidence applies to the detection of global temporal trends, since the strong dominance of North American and European data may mask changes in underrepresented regions.

Information and Capacity Needs:

The collection, cleaning, and analysis of the data for this report have revealed critical information and capacity needs required for evaluating the spatial and temporal patterns of Hg in fish, with many challenges still remaining. These needs, observed for fish and biota in general, also likely pertain to other media. Overall, there is a continued need to strengthen capacity for data collection, cleaning, harmonization, and analysis.

While fish are a good, well-established bioindicator group, the many differences among datasets, taxa, regions, and methodologies make analyses challenging. Not only are methodologies for sample collection and Hg analysis not standardized but there is a need for improved metadata reporting – including site, species, and individual information – essential for enhancing the interpretability and comparability of datasets across regions. Establishing and propagating standardized reporting guidelines and protocols for entities conducting Hg research and monitoring would further reduce the risk of missing or inconsistent data. Future efforts to collect data require continued investment in QA/QC procedures needed to maintain data integrity, including training and technical support for laboratories in developing regions to ensure consistent analytical standards. Capacity-building efforts should focus on strengthening local expertise in both mercury analysis and data management. For true global data, there is a need for further development of data-sharing infrastructure and promotion of cross-sector collaboration among governments, research institutions, and monitoring networks which will improve accessibility, harmonization, and integration of biomonitoring data globally. Such efforts will enable more comprehensive assessments of mercury exposure in aquatic ecosystems and enhance the scientific basis for policy decisions under

the Minamata Convention. Similarly, sustained investment in the time for knowledgeable professionals in the Hg field is needed to do a proper global analysis of such heterogeneous biotic data.

In terms of global monitoring for comparable and usable Hg data, there is a need for enhanced monitoring in underrepresented regions, particularly Africa, South Asia, and parts of the Pacific, where current data are sparse or absent. Expanding observational networks in these regions will be critical to improving global representativeness and reducing uncertainty in regional comparisons. Increasing both the frequency and resolution of sampling is needed, especially in regions where records exist but time series are too short or intermittent to support robust temporal trend analysis. Greater emphasis on consistent annual monitoring of focal species at sentinel sites would strengthen the ability to detect and attribute changes in mercury exposure over time.

2.4.2.3 Birds

Information Used:

Data were compiled from 20 submissions to the OESG repository, with 18 datasets containing data collected from 2014-2024 (Annex 1).

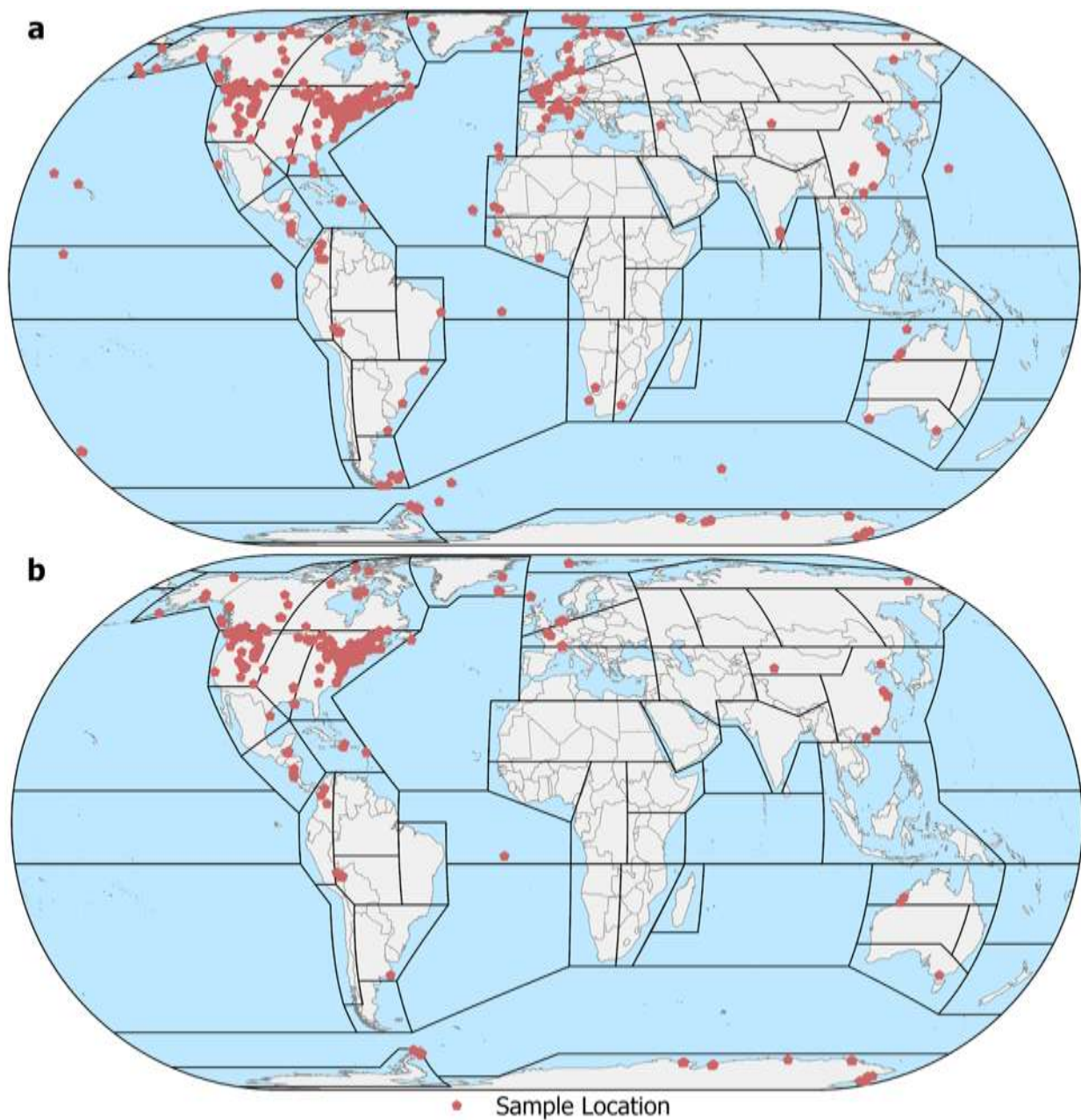


Figure 9: Geographic distribution of bird sampling location records from 2014-2024 **a)** included in the OESG repository, and **b)** filtered using the inclusion/exclusion criteria (see methodology), overlaid on IPCC regional boundaries.

Methodology:

The analysis was conducted on data collected for eggs, feathers, and blood within the bird subgroup. For each matrix, different approaches and considerations were applied, which are described in the corresponding subsections.

Eggs

A quality assurance process was applied to the egg THg dataset before integration. This included the removal of one record with a highly questionable value (19.4 µg/g dw), one record without geographic coordinates, eleven records containing mean THg values from publications that duplicated individual egg data already submitted to the OESG, and 12 records with uncertain accuracy.

Procedures applied before data plotting: prior to plotting, the dataset was restricted to the 2014–2024 period. Matrix descriptions referring to eggs, egg contents, and egg homogenate of yolk and albumen were treated as equivalent. Concentrations were standardized to µg/g, converting from ng/g where necessary.

Converted concentrations between dry weight and wet weight was done on the data. When sample-specific moisture content was available, it was applied; otherwise, a 75% moisture content assumption was used, following Ackerman et al. (2024). No classification by Hg source type (point vs. non-point) was made due to lack of information. Similarly, no separation between individual egg concentrations and mean values was performed; the dataset is composed primarily of individual egg measurements (approximately 98%), with less than 2% of entries representing means of multiple eggs. The fact that species lay a single egg or several eggs was not taken into account, nor was the number of the analysed egg in the laying sequence. No classification by species trophic level was possible due to insufficient metadata.

Blood

A quality assurance process was applied on the blood THg dataset before integration. The dataset was restricted to the 2014–2024 period. Matrix descriptions referring to blood type (whole blood, blood, and red blood cells) was considered. Given that rapid changes in physiology and diet of juvenile birds can affect blood MeHg concentrations and complicate the interpretation of exposure (Ackerman et al. 2011, Chételat et al. 2020), only known adult bird blood THg concentrations were considered for spatial analyses. Only individual blood THg concentrations were selected for analyses, excluding aggregated values. Concentrations were standardized to ww where necessary. When sample specific moisture content was available, it was used to convert dw to ww THg concentrations, otherwise a 77.2% moisture context was assumed (Ackerman et al. 2019). Dry-weight red blood cell THg concentrations were converted to blood THg concentrations following Ackerman et al (2024). Bird records without sufficient species information (e.g., bird sp., hybrid sp.) were excluded. Trophic niches were assigned following Tobias et al. (2022).

Feathers

A quality assurance process was applied on the feather THg dataset before integration. The dataset was restricted to the 2014–2024 period. Matrix descriptions referring to feather type (body, down, flight, tail, tail cover, wing covert, unspecified) was considered. The concentrations were not converted between dry weight, fresh weight and wet weight, as a moisture content of ~10% was used for feathers (based on lab validation). Concentrations were standardized to µg/g (dw), converting from ng/g where necessary. No classification by Hg source type (point vs. non-point) was made due to a lack of information. Only individual data were used, since less than 10 data points correspond to mean/median values. No classification by species trophic level was possible due to insufficient metadata.

Answer/Response:

Birds are abundant, widespread and occupy diverse habitats, making them good bioindicators for monitoring biotic mercury exposure in the environment (Provencher et al. 2014; Evers 2018). Non-invasive sampling of MeHg-rich bird tissues (blood, egg, and feather) can be advantageous for mercury monitoring, though each tissue represents a different exposure period that requires careful interpretation (Albert et al. 2019, Chételat et al. 2020). Birds have been widely studied for toxicological effects of mercury, making toxicity endpoints readily available for risk assessment (Ackerman et al. 2024).

Representativeness of the bird datasets:

Eggs

Following standardization, the dataset comprises 2213 records covering 11 IPCC regions, 9 countries, and 39 bird species. Most data originate from North America (n = 1728; 78%), followed by Northern Europe (n = 415; 19%). All other regions are represented by 45 or fewer records, indicating a significant underrepresentation of other continents in the egg THg dataset.

Feathers

A total of 12117 records of bird feather THg are available following standardization, data is available for 23 IPCC regions, 16 countries, 412 bird species. Most data come from E. North-America (n = 5192, 42.85%) followed by W. North-America (n = 1787, 14.75%), unspecified region (n=962, 7.94%), and West & Central-Europe (n = 739, 6.10%), C. North-America (n=713, 5.88%) and E. Asia (n=622, 5.13%). Noted lack of representation (n < 45) for other IPCC regions (e.g., Africa and Australia-related regions, E.C. Asia, E. Europe, Equatorial Atlantic-Ocean, S.E. South-America, Southern-Ocean and Caribbean) in the feather THg dataset.

Blood

A total of 5,015 bird blood THg records were available after standardization. Records are available from 12 IPCC regions, 10 countries, and 195 species from 39 families. The majority of the data is from E.North-America (n=3951, 78.8%). W.North-America (n=301, 6.0%), N.Central-America (n=232, 4.6%), N.W.North-America (n=113, 2.3%), and Caribbean (n=111, 2.2%) were other IPCC regions with over 100 records. There is a notable lack of representation outside of North and Central America and the Caribbean.

*Summary of current total mercury concentrations in birds**Eggs*

Egg THg concentrations ranged four orders of magnitude from 0.003 to 6 µg/g dw. The median and mean concentrations were 0.65 and 0.84 µg/g dw, respectively. The highest THg concentrations in bird eggs were found in regions of North America, Northern Europe and the North Atlantic region, although data were not available for many areas of the world.

Feathers

Feather THg concentrations ranged over five orders of magnitude from <0.01 to 72.80 µg/g dw. The median and mean concentrations were 1.05 and 2.22 µg/g dw, respectively. Based on the maximum value of feather THg, the highest THg concentrations in bird feather were found in regions of South-American-Monsoon (max: 72.80 µg/g dw) from sites with ASGM activity, followed by E. North-America (61.65 µg/g dw) and E. Asia (38.63 µg/g dw), although high THg (26.25 µg/g dw) also found in an unspecified region. According to the mean value of feather THg, the highest concentrations (sample size greater than 45) in feathers were found across regions of N. Central-America and in N.W. South-America and E. North-America (See details in Figure 11).

Blood

Blood THg concentrations ranged from 0.001 to 8.69 µg/g ww in adult birds (Table 4). The median and mean (±sd) THg concentrations were 0.22 and 0.51±0.72 µg/g ww, respectively. The highest mean adult THg concentrations was 1.4 µg/g ww found in N.W. South-America (n=1, Wattled Jacana [*Jacana jacana*]: a species

commonly found in wetlands and marshes) and $0.78 \mu\text{g/g ww}$ in N. Central-America ($n=232$, all Peregrine Falcons [*Falco peregrinus*], a high trophic level predator) (Figure 12).

Interpretation of spatial patterns in bird mercury body burdens is complicated by the composition of sampled species as foraging behavior and habitat usage impact the risk of mercury exposure (Figure 13). Vertivores ($n=344$, mean THg $\pm$ sd = $0.76 \pm 0.66 \mu\text{g/g dw}$) and aquatic predators ($n=1531$, $0.84 \pm 0.94 \mu\text{g/g dw}$) had the highest mean adult THg concentrations, while nectarivores ($n=4$, $0.01 \pm 0.01 \mu\text{g/g dw}$) and frugivores ($n=7$, $0.02 \pm 0.01 \mu\text{g/g dw}$) had the lowest .

Table 4. Summary statistics for standardized feather, egg, and blood THg records.

Statistic	Feather THg ($\mu\text{g/g dw}$)	Egg THg ($\mu\text{g/g dw}$)	Blood THg ($\mu\text{g/g ww}$)
Minimum	< 0.01	0.003	< 0.01
25th Percentile	0.52	0.33	0.11
Median	1.05	0.65	0.22
Mean	2.22	0.84	0.51
SD	3.78	0.75	0.72
75th Percentile	2.18	1.15	0.61
Maximum	72.80	6.00	8.69

The Figure 10 shows Box plot for egg THg concentration by IPCC region

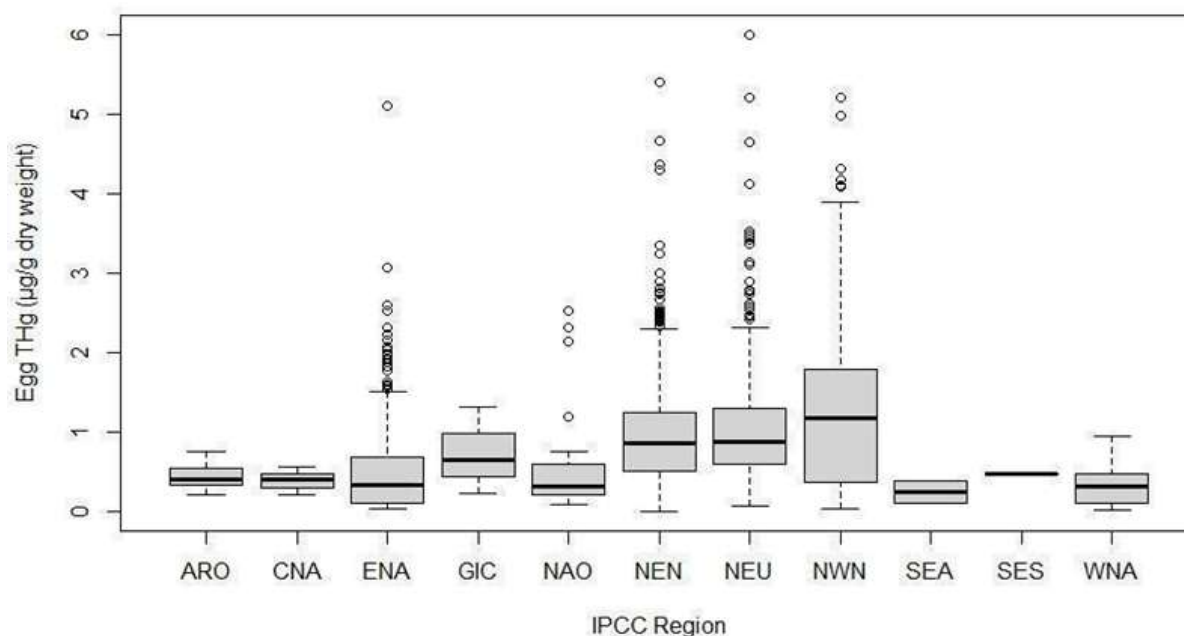


Figure 10: Box plot of THg concentrations in eggs from bird samples, according to IPCC regions

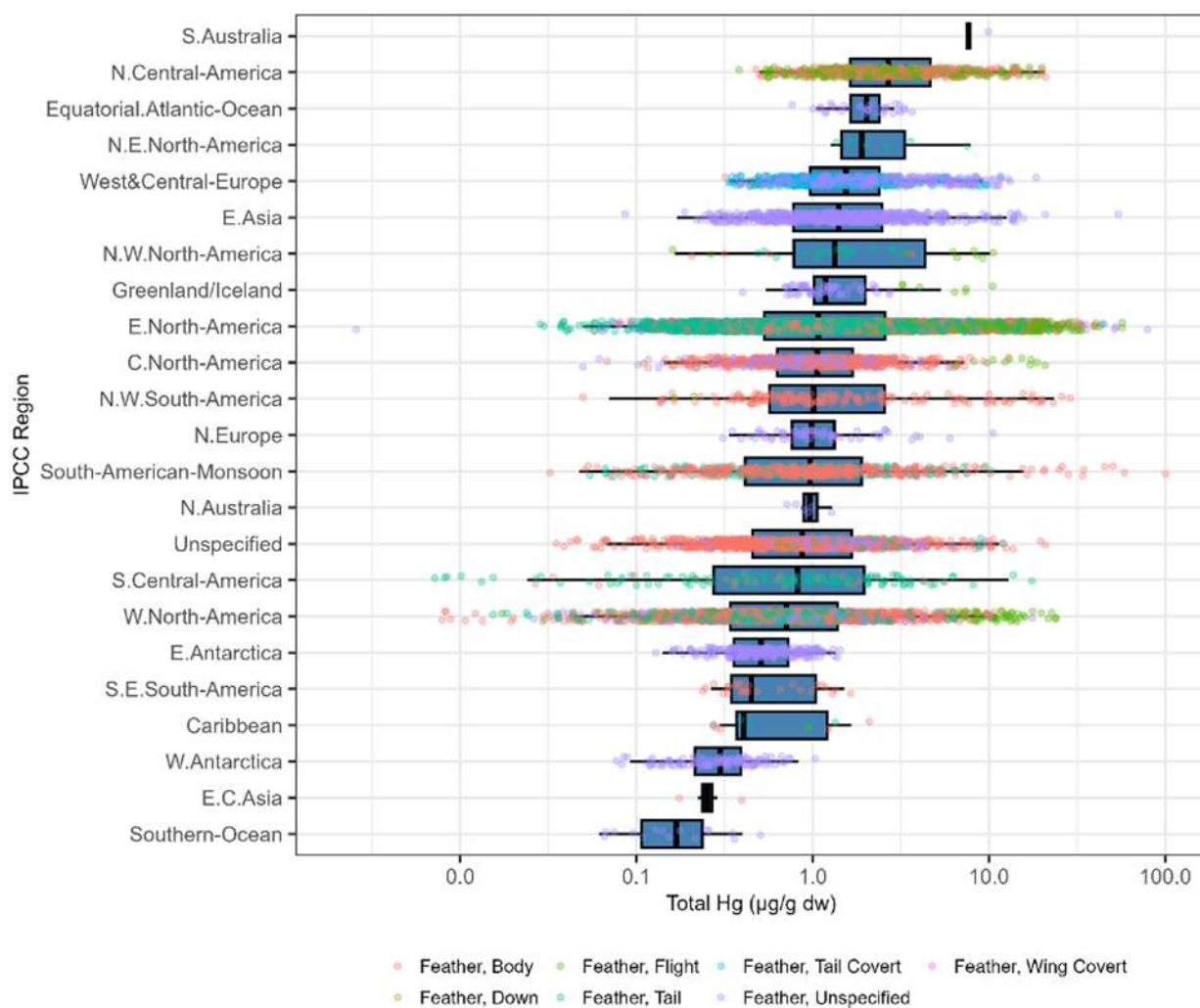


Figure 11. Total Hg (THg, g/g, dw) concentration in feathers across feather types and different IPCC regions. The X-axis is log₁₀ scale.

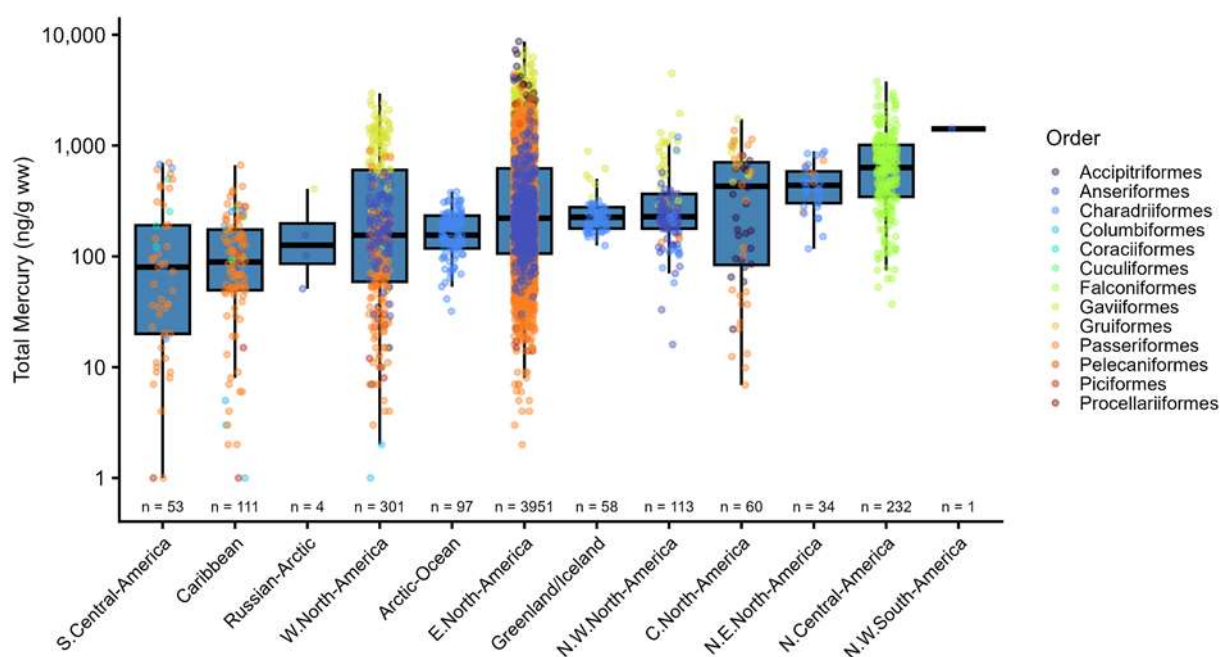
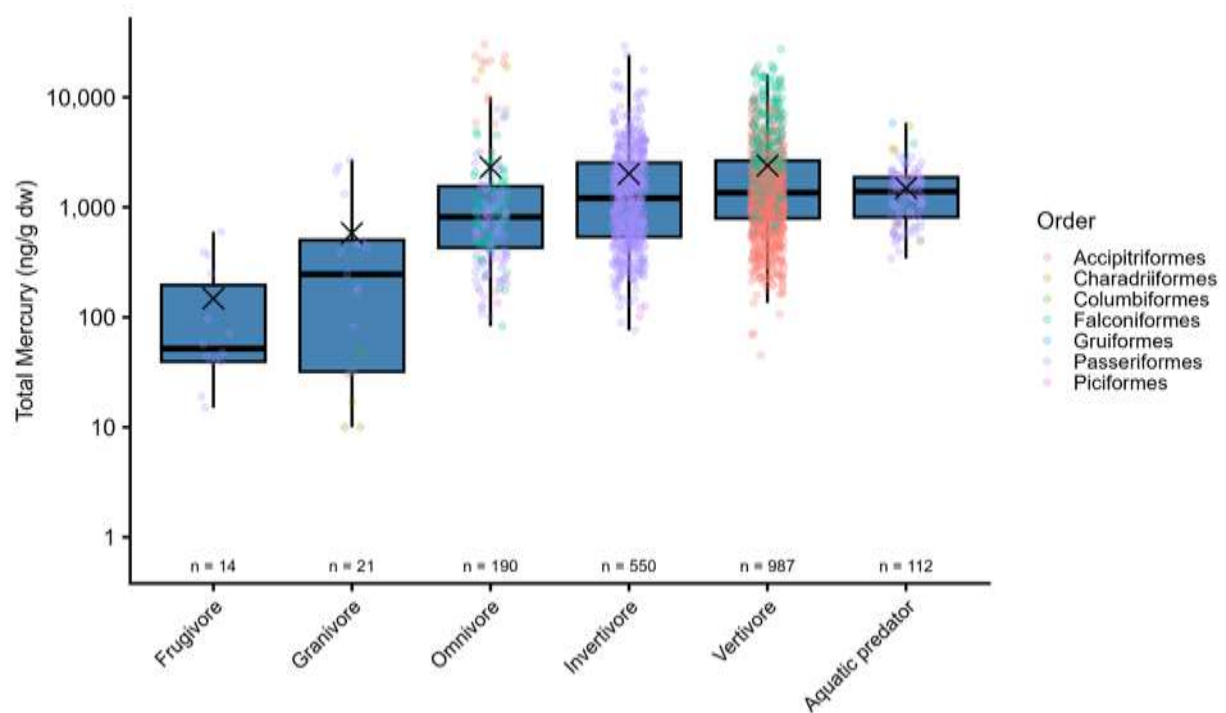


Figure 12. Total mercury (THg) concentrations (ng/g wet weight) in bird blood collected during the current period (2014–2024) across IPCC regions within the harmonized adult bird blood subset.

A



B

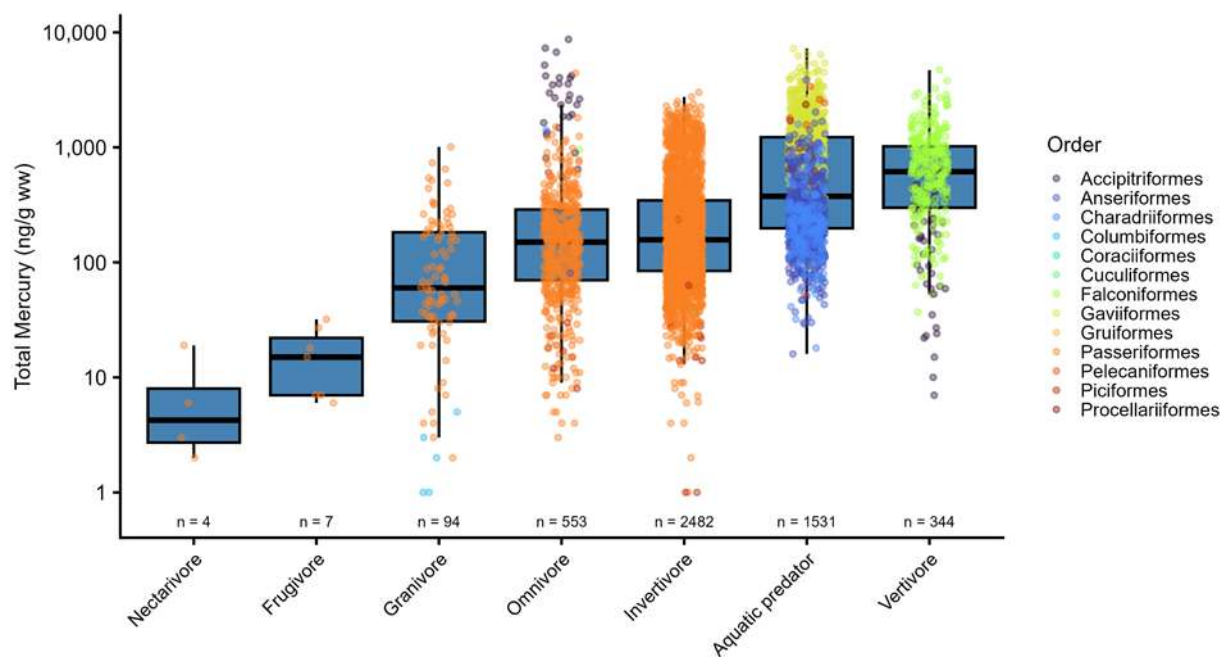


Figure 13. A) Total mercury (THg) concentrations (g/g, dw) in adult body feathers collected in the current period (2014–2024) by trophic niche. B) Total mercury (THg) concentrations (ng/g wet weight) in bird blood, both collected during the current period (2014–2024) by trophic niche (Tobias et al. 2022) within the harmonized adult bird blood subset.

Table 5. Summary of feather Hg ($\mu\text{g/g dw}$) across IPCC Name by feather type

IPCC Name	n (%)	Feather Type							Combined
		Flight	Unspecified	Down	Body	Wing Covert	Tail	Tail Covert	
C.North-America	713 (5.9%)	12.38 $\pm$ 4.30 [5.42, 17.91]	0.99 $\pm$ 0.76 [0.07, 5.00]	0.25	1.38 $\pm$ 1.21 [0.07, 10.65]				1.53 $\pm$ 1.96 [0.07, 17.91]
Caribbean	11 (0.1%)	0.90			0.80 $\pm$ 0.61 [0.36, 1.65]		0.72 $\pm$ 0.50 [0.29, 1.34]		0.77 $\pm$ 0.5 [0.3, 1.65]
E.Antarctica	217 (1.8%)		0.56 $\pm$ 0.26 [0.14, 1.35]						0.56 $\pm$ 0.26 [0.14, 1.35]
E.Asia	622 (5.1%)		2.10 $\pm$ 2.63 [0.11, 38.63]						2.1 $\pm$ 2.63 [0.11, 38.63]
E.C.Asia	2 (0.0%)				0.25 $\pm$ 0.04 [0.22, 0.29]				0.25 $\pm$ 0.04 [0.22, 0.29]
E.North-America	5192 (42.8%)	10.06 $\pm$ 7.26 [0.24, 61.65]	4.32 $\pm$ 7.29 [<0.01, 61.65]	5.10 $\pm$ 1.84 [2.05, 8.51]	2.05 $\pm$ 3.15 [<0.01, 35.46]	0.36 $\pm$ 0.14 [0.26, 0.46]	1.32 $\pm$ 1.93 [<0.01, 34.40]		2.69 $\pm$ 4.56 [<0.01, 61.65]
Equatorial.Atlantic-Ocean	34 (0.3%)		1.97 $\pm$ 0.49 [0.91, 2.90]						1.97 $\pm$ 0.49 [0.91, 2.9]
Greenland/Iceland	66 (0.5%)	5.50 $\pm$ 1.73 [3.23, 7.75]	1.34 $\pm$ 0.54 [0.54, 3.09]						1.78 $\pm$ 1.48 [0.54, 7.75]
N.Australia	6 (0.0%)		1.01 $\pm$ 0.16 [0.86, 1.29]						1.01 $\pm$ 0.16 [0.86, 1.29]
N.Central-America	609 (5.0%)	3.81 $\pm$ 3.85 [0.50, 20.66]	1.90		3.86 $\pm$ 3.48 [0.53, 27.35]				3.84 $\pm$ 3.63 [0.5, 27.35]
N.E.North-America	6 (0.0%)						3.03 $\pm$ 2.54		3.03 $\pm$ 2.54 [1.26, 7.88]

							[1.26, 7.88]		
N.Europe	63 (0.5%)		1.45 ± 1.39 [0.34, 7.67]						1.45 ± 1.39 [0.34, 7.67]
N.W.North- America	31 (0.3%)	5.86 ± 3.69 [0.16, 10.13]			1.33 ± 2.00 [0.22, 4.33]		1.62 ± 1.06 [0.50, 4.36]		2.82 ± 2.93 [0.17, 10.13]
N.W.South- America	165 (1.4%)	0.18 ± 0.05 [0.14, 0.22]			3.34 ± 5.35 [0.07, 23.41]				3.3 ± 5.33 [0.07, 23.41]
S.Australia	1 (0.0%)		7.67						7.67
S.Central-America	147 (1.2%)				1.00 ± 1.36 [0.00, 6.39]		1.56 ± 2.13 [0.01, 12.88]		1.45 ± 2.01 [<0.01, 12.88]
S.E.South-America	26 (0.2%)				0.68 ± 0.40 [0.27, 1.52]				0.68 ± 0.4 [0.27, 1.52]
South-American- Monsoon	587 (4.8%)				2.73 ± 7.33 [0.03, 72.79]		1.47 ± 1.67 [0.06, 8.61]		2.27 ± 5.97 [0.03, 72.79]
Southern-Ocean	15 (0.1%)		0.18 ± 0.11 [0.06, 0.40]						0.18 ± 0.11 [0.06, 0.4]
Unspecified	962 (7.9%)	0.85 ± 0.23 [0.69, 1.19]	2.07 ± 1.50 [0.14, 7.93]		1.34 ± 1.76 [0.04, 17.77]		3.97 ± 3.59 [1.00, 12.40]		1.45 ± 1.88 [0.04, 17.77]
W.Antarctica	116 (1.0%)		0.31 ± 0.15 [0.09, 1.22]						0.31 ± 0.15 [0.09, 1.22]
W.North-America	1787 (14.7%)	6.62 ± 5.78 [0.02, 24.26]	1.05 ± 2.34 [<0.01, 22.80]		1.04 ± 1.13 [0.01, 10.56]		0.81 ± 0.84 [0.01, 5.95]		1.38 ± 2.36 [<0.01, 24.26]
West&Central- Europe	739 (6.1%)		3.73 ± 2.91 [0.41, 14.49]		1.37 ± 0.62 [0.34, 2.92]			1.76 ± 1.48 [0.34, 11.42]	2.2 ± 2.08 [0.34, 14.49]
TOTAL	12117 (100.0%)	7.75 ± 6.79 [0.02, 61.65]	2.20 ± 3.89 [<0.01, 61.65]	4.84 ± 2.10 [0.25, 8.51]	1.82 ± 3.14 [<0.01, 72.79]	0.36 ± 0.14 [0.26, 0.46]	1.33 ± 1.90 [<0.01, 34.40]	1.76 ± 1.48 [0.34, 11.42]	0.51 ± 0.72 [<0.01, 8.69]

Table 6. Summary of total mercury (THg) concentrations (ng g⁻¹ ww) in bird families across IPCC regions within the harmonized adult bird blood subset

IPCC Name	Family	n	Individual THg Concentration (ng/g ww)		
			Mean	SD	Median
Arctic-Ocean	<i>Alcidae (Auks, Murres, and Puffins)</i>	10	64.9	19.5	66.6
	<i>Laridae (Gulls, Terns, and Skimmers)</i>	87	188.2	72.9	176.9
C.North-America	<i>Accipitridae (Hawks, Eagles, and Kites)</i>	23	307.3	251.8	174.0
	<i>Corvidae (Crows, Jays, and Magpies)</i>	1	12.5		12.5
	<i>Gaviidae (Loons)</i>	13	841.2	349.3	698.0
	<i>Hirundinidae (Swallows)</i>	2	42.3	7.7	42.3
	<i>Parulidae (New World Warblers)</i>	5	460.4	560.4	320.0
	<i>Passerellidae (New World Sparrows)</i>	6	272.9	395.9	82.1
	<i>Troglodytidae (Wrens)</i>	6	908.3	196.7	900.0
	<i>Turdidae (Thrushes and Allies)</i>	1	49.8		49.8
	<i>Tyrannidae (Tyrant Flycatchers)</i>	1	102.0		102.0
	<i>Vireonidae (Vireos, Shrike-Babblers, and Erpornis)</i>	2	29.6	10.6	29.6
Caribbean	<i>Anatidae (Ducks, Geese, and Waterfowl)</i>	1	81.0		81.0
	<i>Ardeidae (Herons, Egrets, and Bitterns)</i>	5	163.0	92.7	142.0
	<i>Columbidae (Pigeons and Doves)</i>	3	3.0	2.0	3.0
	<i>Cuculidae (Cuckoos)</i>	4	191.8	75.8	202.5

	<i>Laridae (Gulls, Terns, and Skimmers)</i>	1	260.0		260.0
	<i>Mimidae (Mockingbirds and Thrashers)</i>	6	43.3	18.1	51.0
	<i>Parulidae (New World Warblers)</i>	69	136.0	109.2	109.0
	<i>Phaenicophilidae (Greater Antillean Tanagers)</i>	4	11.0	6.5	10.5
	<i>Picidae (Woodpeckers)</i>	2	8.0	9.9	8.0
	<i>Scolopacidae (Sandpipers and Allies)</i>	3	174.0	88.5	189.0
	<i>Thraupidae (Tanagers and Allies)</i>	4	7.5	7.9	4.5
	<i>Todidae (Todies)</i>	1	97.0		97.0
	<i>Turdidae (Thrushes and Allies)</i>	2	52.0	5.7	52.0
	<i>Tyrannidae (Tyrant Flycatchers)</i>	4	235.5	167.9	277.5
	<i>Vireonidae (Vireos, Shrike-Babblers, and Erpornis)</i>	2	52.0	0.0	52.0
E.North-America	<i>Accipitridae (Hawks, Eagles, and Kites)</i>	44	2102.7	2065.3	1560.5
	<i>Alcedinidae (Kingfishers)</i>	1	241.0		241.0
	<i>Alcidae (Auks, Murres, and Puffins)</i>	21	876.2	251.6	831.0
	<i>Anatidae (Ducks, Geese, and Waterfowl)</i>	498	301.1	293.6	227.0
	<i>Bombycillidae (Waxwings)</i>	4	13.0	12.7	7.0
	<i>Cardinalidae (Cardinals and Allies)</i>	42	90.0	82.7	65.0
	<i>Certhiidae (Treecreepers)</i>	5	171.6	30.5	175.0
	<i>Charadriidae (Plovers and Lapwings)</i>	2	146.0	9.9	146.0

<i>Corvidae (Crows, Jays, and Magpies)</i>	9	156.8	112.7	154.0
<i>Cuculidae (Cuckoos)</i>	1	38.0		38.0
<i>Falconidae (Falcons and Caracaras)</i>	62	1129.1	756.7	987.0
<i>Fringillidae (Finches, Euphonias, and Allies)</i>	10	10.2	11.4	5.0
<i>Gaviidae (Loons)</i>	479	1789.2	1042.1	1569.0
<i>Hirundinidae (Swallows)</i>	18	216.3	78.4	209.5
<i>Hydrobatidae (Northern Storm-Petrels)</i>	25	787.8	468.5	676.0
<i>Icteridae (Troupials and Allies)</i>	75	155.3	130.4	126.0
<i>Mimidae (Mockingbirds and Thrashers)</i>	122	182.9	175.9	127.0
<i>Paridae (Tits, Chickadees, and Titmice)</i>	121	231.6	236.7	127.0
<i>Parulidae (New World Warblers)</i>	638	166.4	162.4	113.0
<i>Passerellidae (New World Sparrows)</i>	878	567.0	548.4	315.5
<i>Pelecanidae (Pelicans)</i>	8	1581.8	1214.8	1683.0
<i>Picidae (Woodpeckers)</i>	24	87.2	129.6	35.0
<i>Polioptilidae (Gnatcatchers)</i>	7	1139.7	613.9	1226.0
<i>Rallidae (Rails, Gallinules, and Coots)</i>	4	449.8	294.3	347.0
<i>Scolopacidae (Sandpipers and Allies)</i>	13	501.7	237.7	448.0
<i>Sittidae (Nuthatches)</i>	26	249.6	136.4	196.5
<i>Troglodytidae (Wrens)</i>	59	506.1	436.1	345.0

	<i>Turdidae (Thrushes and Allies)</i>	427	134.6	130.3	92.9
	<i>Tyrannidae (Tyrant Flycatchers)</i>	70	430.6	465.5	307.0
	<i>Vireonidae (Vireos, Shrike-Babblers, and Erpornis)</i>	258	233.3	176.6	174.0
Greenland/Iceland	<i>Alcidae (Auks, Murres, and Puffins)</i>	52	217.1	48.4	215.5
	<i>Gaviidae (Loons)</i>	6	556.2	188.2	520.5
N.Central-America	<i>Falconidae (Falcons and Caracaras)</i>	232	778.3	624.0	632.5
N.E.North-America	<i>Icteridae (Troupials and Allies)</i>	6	539.8	168.6	568.5
	<i>Scolopacidae (Sandpipers and Allies)</i>	28	448.2	213.6	399.6
N.W.North-America	<i>Alcidae (Auks, Murres, and Puffins)</i>	12	288.4	367.1	116.9
	<i>Anatidae (Ducks, Geese, and Waterfowl)</i>	72	224.9	94.4	218.5
	<i>Gaviidae (Loons)</i>	23	1000.1	850.3	883.0
	<i>Icteridae (Troupials and Allies)</i>	6	203.7	72.9	184.5
N.W.South-America	<i>Jacanidae (Jacanas)</i>	1	1414.2		1414.2
Russian-Arctic	<i>Anatidae (Ducks, Geese, and Waterfowl)</i>	3	103.0	52.5	102.0
	<i>Gaviidae (Loons)</i>	1	407.0		407.0
S.Central-America	<i>Alcedinidae (Kingfishers)</i>	4	266.0	164.4	221.5
	<i>Corvidae (Crows, Jays, and Magpies)</i>	1	12.0		12.0
	<i>Furnariidae (Ovenbirds and Woodcreepers)</i>	4	384.5	220.5	425.5
	<i>Hirundinidae (Swallows)</i>	3	6.7	2.5	7.0
	<i>Icteridae (Troupials and Allies)</i>	3	207.7	205.5	191.0

	<i>Parulidae (New World Warblers)</i>	10	180.8	199.5	85.5
	<i>Passerellidae (New World Sparrows)</i>	1	37.0		37.0
	<i>Picidae (Woodpeckers)</i>	1	1.0		1.0
	<i>Scolopacidae (Sandpipers and Allies)</i>	5	303.8	322.0	112.0
	<i>Thraupidae (Tanagers and Allies)</i>	2	163.5	178.9	163.5
	<i>Troglodytidae (Wrens)</i>	2	133.5	14.8	133.5
	<i>Turdidae (Thrushes and Allies)</i>	2	5.0	5.7	5.0
	<i>Tyrannidae (Tyrant Flycatchers)</i>	12	108.6	193.0	38.5
	<i>Vireonidae (Vireos, Shrike-Babblers, and Erpornis)</i>	3	24.3	10.1	23.0
W.North-America	<i>Accipitridae (Hawks, Eagles, and Kites)</i>	12	57.1	66.9	31.0
	<i>Anatidae (Ducks, Geese, and Waterfowl)</i>	77	214.8	127.7	178.0
	<i>Bombycillidae (Waxwings)</i>	3	20.0	6.2	18.0
	<i>Cardinalidae (Cardinals and Allies)</i>	2	20.0	18.4	20.0
	<i>Columbidae (Pigeons and Doves)</i>	2	1.5	0.7	1.5
	<i>Falconidae (Falcons and Caracaras)</i>	2	178.0	73.5	178.0
	<i>Fringillidae (Finches, Euphonias, and Allies)</i>	3	5.0	1.7	4.0
	<i>Gaviidae (Loons)</i>	80	1157.6	528.5	1113.5
	<i>Hirundinidae (Swallows)</i>	2	244.0	21.2	244.0
	<i>Icteridae (Troupials and Allies)</i>	9	257.9	230.6	180.0

<i>Mimidae (Mockingbirds and Thrashers)</i>	5	44.2	29.6	35.0
<i>Paridae (Tits, Chickadees, and Titmice)</i>	2	25.0	2.8	25.0
<i>Parulidae (New World Warblers)</i>	20	125.4	113.3	107.0
<i>Passerellidae (New World Sparrows)</i>	43	113.9	205.3	44.0
<i>Picidae (Woodpeckers)</i>	4	28.3	25.1	21.0
<i>Regulidae (Kinglets)</i>	2	70.0	22.6	70.0
<i>Troglodytidae (Wrens)</i>	3	98.0	59.9	121.0
<i>Turdidae (Thrushes and Allies)</i>	9	86.4	30.9	74.0
<i>Tyrannidae (Tyrant Flycatchers)</i>	12	109.4	55.7	113.5
<i>Vireonidae (Vireos, Shrike-Babblers, and Erpornis)</i>	9	66.9	28.0	57.0

Consistency of Answer with Existing Literature (especially the 2018 GMA):

There was not an extensive analysis of THg in bird eggs, blood, and feathers in the 2018 UN/AMAP Global Mercury Assessment (GMA) for comparison. The bird data included in the 2018 Global Mercury Assessment were compiled from peer-reviewed, published manuscripts. Consequently, these studies were primarily designed to address highly specific research questions within defined spatial and temporal contexts, rather than for integration into a global synthesis. Similar to the present effort, the GMA analyses were limited by inconsistent access to metadata. Additionally, the GMA incorporated all available data without restricting analyses to a specific, contemporary period (2014–2024). These factors make direct comparisons with the GMA, and with other existing literature, challenging.

The bird data included in the 2018 GMA were largely presented as case studies (e.g., marine birds, loons) limiting the ability to compare broadly across taxa and tissues. For comparison, we report weighted THg concentrations derived from both individual and arithmetic mean in this section to align with the GMA methodology.

THg concentrations in adult Procellariidae body feathers were very similar between this study (10.4 ± 10.6 µg/g fw, n = 9 records from 130 individuals) and the GMA (10.5 ± 10.6 µg/g fw, n = 103 records from 2364 individuals). By contrast, THg in adult Diomedidae and Oceanitidae (reported as Hydrobatidae in the GMA) was less consistent, with this study showing 3.4 ± 0.3 µg/g fw (n = 2 [39]) and 1.1 ± 0.8 µg/g fw (n = 2 [20]), compared with 17.2 ± 10.0 µg/g fw (n = 42 [977]) and 12.5 ± 5.2 µg/g fw (n = 27 [726]) in the GMA. For blood, THg concentrations in adult Hydrobatidae were comparable (this study: 0.79 ± 0.47 µg/g fw, n = 25 [25]; GMA: 0.75 ± 0.14 µg/g fw, n = 5 [131]), as were adult Gaviiformes (this study: 1.6 ± 1.0 µg/g fw, n = 602 [602]; GMA: 1.8 ± 1.5 µg/g fw, n = 101 [4927]).

These comparisons (Table 7) highlight that while some results show strong agreement with the GMA, others differ, likely reflecting differences in sampling design, spatial coverage, and temporal scope. Sample sizes in the 2018

GMA were generally much larger, in part because it incorporated studies published prior to 2014, whereas the present synthesis focuses on the contemporary period (2014–2024). The largest discrepancies were observed for Diomedidae and Oceanitidae body feathers, both of which were represented by only two records in this study, underscoring the limited data available for these groups. Overall, these comparisons emphasize the inherent variability in THg concentrations among marine birds and loons. The species highlighted here and in the GMA span a range of generally high trophic level foraging ecologies shaped by differences in diet, behavior, habitat use (e.g., benthic vs pelagic), and life history traits, all of which contribute to spatially and temporally heterogeneous mercury body burdens. These patterns highlight the importance of long-term, consistent monitoring programs to capture the range of variability, account for confounding factors, and facilitate a reliable assessment of mercury body burdens in avian species.

Table 7. Comparison of total mercury (THg) concentrations in selected bird taxa between this study and the 2018 UN/AMAP Global Mercury Assessment (GMA). Values are reported as weighted mean $\pm$ standard deviation ($\mu\text{g/g}$ fresh weight) with the number of records and individual birds in brackets.

Family	Tissue	This study		2018 GMA	
		n (records [individuals])	THg ($\mu\text{g/g}$ fw)	n (records [individuals])	THg ($\mu\text{g/g}$ fw)
Procellariidae	Body feathers	9 [130]	10.4 ± 10.6	103 [2364]	10.5 ± 10.6
Diomedidae	Body feathers	2 [39]	3.4 ± 0.3	42 [977]	17.2 ± 10.0
Oceanitidae	Body feathers	2 [20]	1.1 ± 0.8	27 [726]	12.5 ± 5.2
Hydrobatidae	Adult blood	25 [25]	0.8 ± 0.5	5 [131]	0.75 ± 0.14
Gaviidae	Adult blood	602 [602]	1.6 ± 1.0	101 [4927]	1.8 ± 1.5

Evaluation of Confidence:

Overall, there is low confidence in the spatial analysis of mercury in bird eggs, feathers, and blood because of the comparison of many species across geographic regions and lack of accounting for biological and ecological influences on bioaccumulation including age, sex, trophic position, habitat and migration. In the case of feathers, all available data were used, including all feather types. Observational coverage is lacking for many parts of the world, including South America, Africa, South Asia and Oceania.

Information and Capacity Needs:

The compilation and analysis of bird mercury data highlighted critical gaps and the need for enhanced capacity to reliably assess spatial patterns of mercury in avian species. Many of these challenges are similar to those observed in invertebrates, fish, mammals, and other biota reflecting broader difficulties in assessing mercury in biota. In particular, enhanced monitoring is required in underrepresented regions, particularly Africa (AF), Latin America and the Caribbean (LAC), and other data-sparse areas. Increasing both the frequency and resolution of sampling will improve the temporal and spatial resolution of mercury assessments. Birds are widely used bioindicators, particularly of ecological health, and have been extensively studied over multiple decades due to their diverse foraging ecologies, trophic levels, life history traits, and broad geographic distributions. However, confounding factors (habitat or landscape context, demographic characteristics, physiology, tissue type) are often unreported,

requiring careful consideration when integrating datasets. As a result, the standardization of sampling, reporting, and analysis is crucial to ensure comparability across regions and through time.

Finally, continued investment in data-sharing infrastructure and promotion of cross-sector collaboration between governments, research institutions, and established monitoring networks will improve accessibility and integration of datasets. These efforts will improve efforts to assess mercury levels in birds and support evaluation of the Minamata Convention.

2.4.2.4 Marine Mammals

Information Used:

Total mercury concentration data for marine mammals were submitted for 88 species including more than 8,700 individual concentrations and 1,400 average concentrations (see Annex 1). Analyses were conducted predominantly in liver and muscle tissues, results were also available for blood, blubber/fat/oil, brain, epidermis/skin, hair, heart, and kidney.

The majority of data were collected from North America and Europe & Central Asia, where more than 8,000 samples were analyzed in marine mammals (Figure 12). A few hundred samples were available from East Asia & Pacific and Latin America & Caribbean, the smallest number of data received came from the Middle East & Africa and from Antarctica. Analytical methods included direct mercury analyzer, cold vapor atomic absorption spectroscopy, and ICP-MS and were reported for a very limited number of datasets. A minority of the data represented samples collected during the last decade (2014-2024): 28% of the liver data and 23% of the muscle data, with a skewed representation from the northern hemisphere (see below).

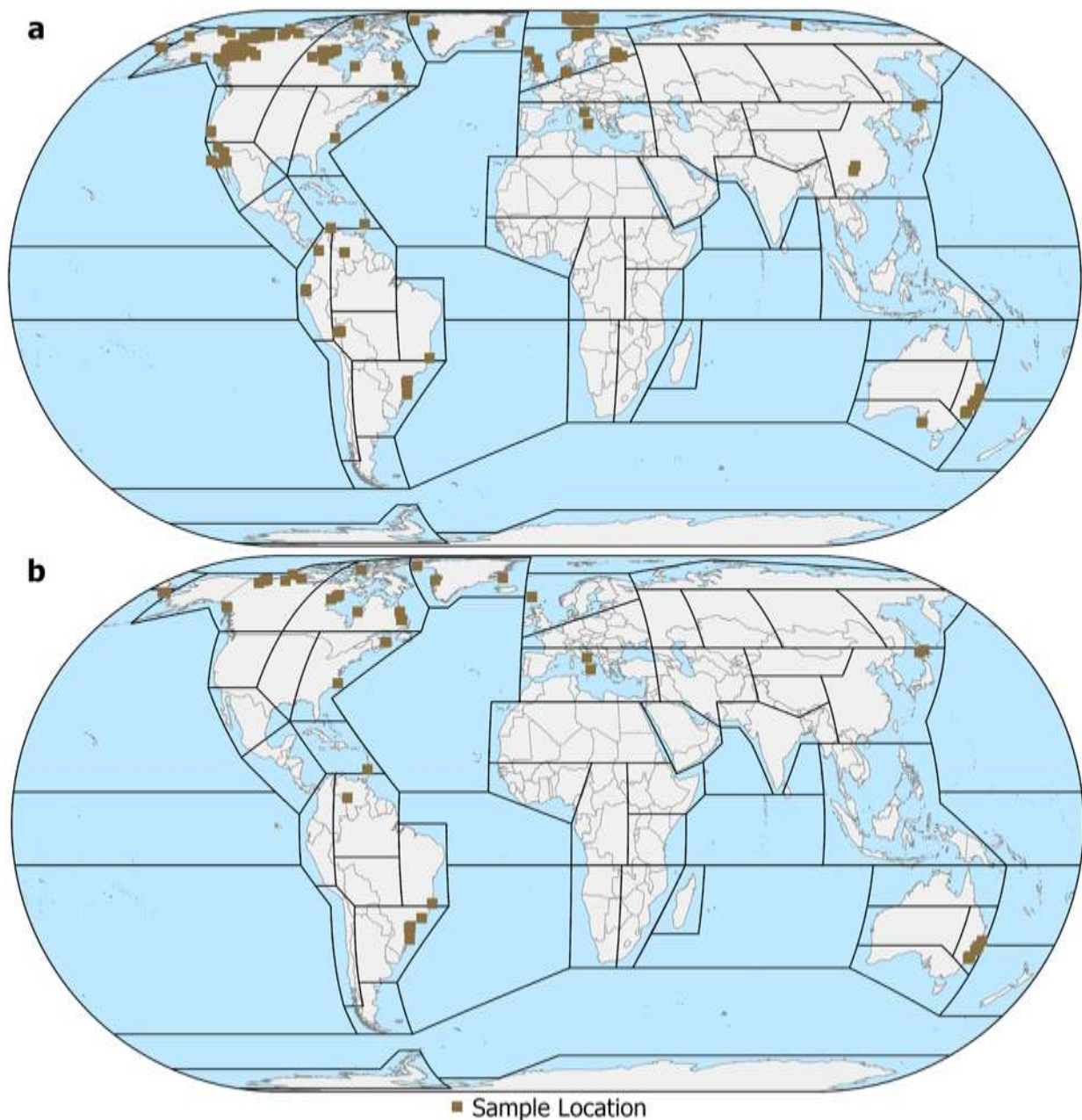


Figure 12. Geographic distribution of a) all 2014-2024 sampling locations and b) sampling locations used in current level analyses of marine mammal samples, overlaid on IPCC regional boundaries.

Methodology:

Inclusion criteria:

For geographical comparisons, data from all marine mammal taxa collected between 2014 and 2024 were included. The inclusion of data from the last decade reduced the number of data points by 40% compared to the inclusion of data collected in 2010-2014. As a minor part of the data was denoted as zeros (< 1% of the data), all data were included in geographical comparisons. The zeros were included without further processing as the information on detection limits was missing in most cases.

Exclusion criteria:

As most measurements were done in the liver followed by muscle tissue, the other matrices were excluded for geographical comparisons. Data lacking latitude and longitude information were also excluded.

Data standardization and harmonization:

Individual measurements for liver, muscle, and kidney and mean values for liver and muscle were all converted on a dry weight basis for comparisons. Moisture contents of 70% for liver, 75% for muscle and kidney, and 80% for brain were used to convert concentrations from wet weight to dry weight (Hawk et al. 2024, Yang et al. 2003).

Concentrations measured in liver and muscle were plotted separately to allow comparability. Because the submitted data included both individual measurements and mean values for populations across multiple species and tissues, the data were aggregated into $5^{\circ} \times 5^{\circ}$ grid cells, which roughly corresponded to the sampled populations. Within each cell, arithmetic means weighted by sample size were then calculated for each species and tissue.

Answer/Response:

Marine mammals are an important indicator for the purpose of the Minamata Convention because of their long lifespan and the diversity of their trophic levels and habitats. Marine mammal position in food webs span from primary predators (baleen whales) to tertiary predators (polar bears, killer whales). They therefore feed on diverse organisms, from pelagic to deepwater prey, providing information for different marine ecosystems. Particularly in the Arctic, several marine mammals are a crucial food source for Indigenous Peoples and can therefore be an important link to human exposure resulting from high mercury levels in parts of the traditional diet.

Only a small set of the submitted data represented samples collected during the last decade (2014-2024): 28% of the liver data and 23% of the muscle data overall. Mapping the most recent muscle concentrations of THg data (2014-2024) indicates that marine mammal populations with the highest concentrations are found in equatorial and mid latitudes (Figure 13). Further inspection of the data shows that the highest contaminated species were all toothed whales. The mapping of liver concentrations shows high THg levels in populations sampled in the Northern Europe, Greenland/Iceland and Latin America & Caribbean regions, in addition to a few samples from East Australia (Figure 13). These species included long-finned pilot whales (*Globicephala melas*) from the Faroe Islands, polar bears (*Ursus maritimus*) from East Greenland, false killer whales (*Pseudorca crassidens*) from Caribbean and beaked whales (*Mesoplodon* sp.) from E. Australia. If data from 2010 onwards are included, highest THg concentrations are also found in toothed whales and polar bears, but some other regions and species with high THg levels emerge, such as Risso's dolphins (*Grampus griseus*, Israel), Indo-Pacific bottlenose dolphins (*Tursiops aduncus*, Japan), and Blainville's beaked whales (*Mesoplodon densirostris*, China).

The lowest overall mean THg concentrations were found in walruses, seals, and baleen whales.

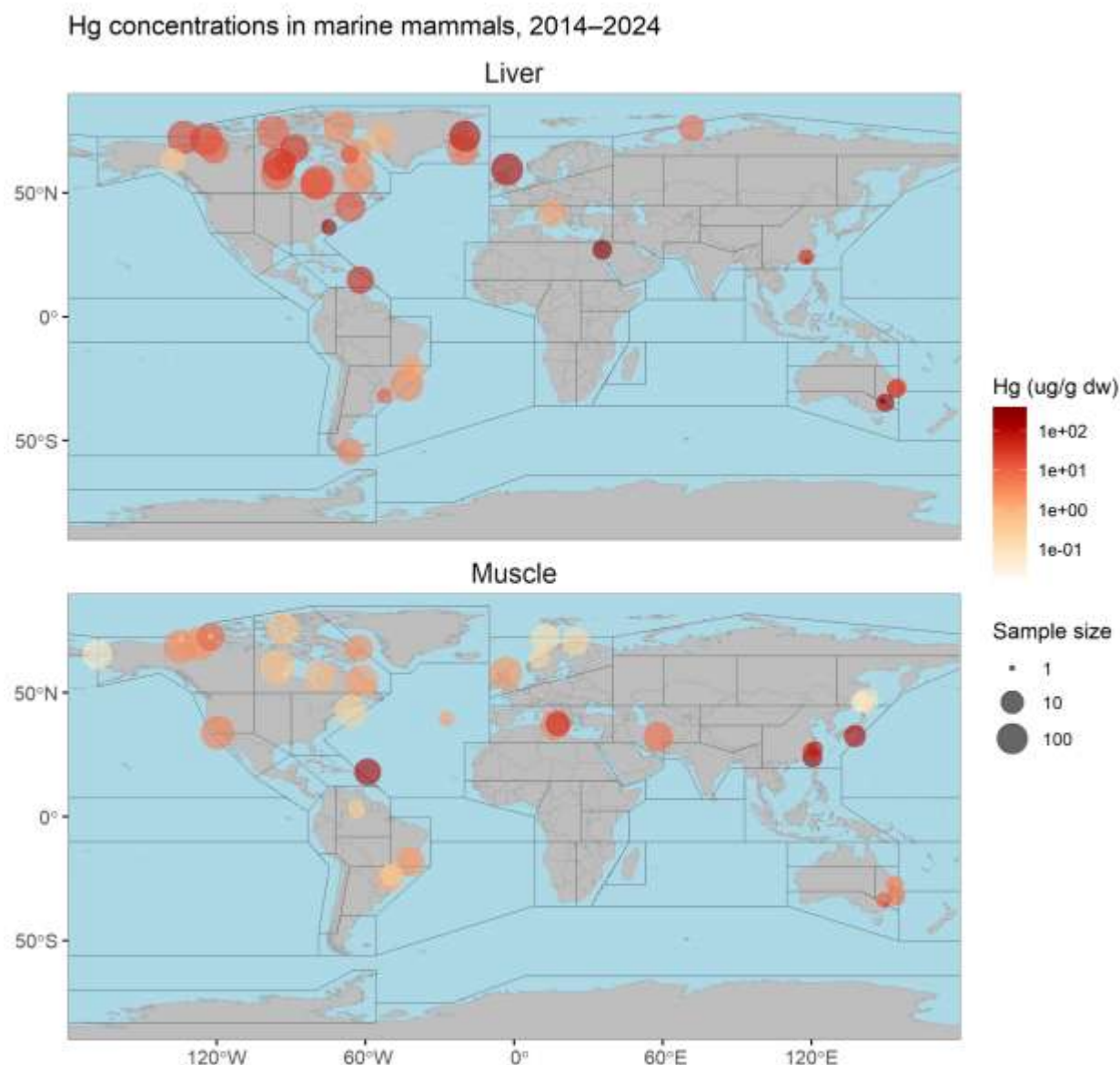


Figure 13. Concentrations of total mercury in liver and muscle samples from marine mammals collected in 2014–2024. Each dot represents an arithmetic mean level in one species collected within a $5^\circ \times 5^\circ$ grid cell. Points were jittered slightly to improve visibility of overlapping observations.

Geographical representativeness of the marine mammal datasets

In comparison to other regions, the Arctic is over-represented with regards to spatial and temporal for THg concentration data availability. There is a complete lack of information for some regions (e.g., Africa, S.E. Asia), in other areas only one sample was collected. Furthermore, in regions other than the Arctic and Northern Europe (Færoe Islands), liver and muscle samples often originate from stranded animals which is not optimal as these animals may have been old or very young and potentially sick. In contrast, samples in the Arctic and Northern Europe are crucial food sources collected from usually healthy individuals as part of subsistence activities. In several cases, more information about the health and appearance of the animals is available through Indigenous Knowledge (see the beluga case study below).

Consistency of Answer with Existing Literature (especially the 2018 GMA):

There was insufficient time and resources to undertake a comprehensive literature review. The 2018 GMA did not include much information and very limited data on marine mammals. Some sections refer to data from the

AMAP 1998 assessment, where mean Hg for Arctic marine mammals ranged from 0.08 - 2 µg g⁻¹ w.w. in seal muscle, and up to 300 µg g⁻¹ ww in whale liver. A study from 1998 on Hong Kong's Indo-Pacific hump-backed dolphins is referred to and reports a maximum of 906 µg kg⁻¹ dw in animals close to point sources. The very large ranges of mercury levels reported globally for marine mammals in the various tissues are consistent with the current findings.

Evaluation of Confidence:

Overall, there is high confidence in data generated through national and regional monitoring programs, which are subject to standard QA/QC procedures, and (in the case of AMAP) additional round-robin laboratory testing to ensure comparability of the results between different facilities.

Information and Capacity Needs:

Enhanced monitoring is required in underrepresented regions, particularly Africa (AF), Latin America and the Caribbean (LAC), and other data-sparse areas. Increasing both the frequency and resolution of sampling will improve the temporal and spatial resolution of mercury assessments. For example, species in regions where high concentrations were found before 2010 should be sampled again to investigate more recent accumulation.

Continued investment in data-sharing infrastructure and promotion of cross-sector collaboration will improve accessibility and integration of datasets across regions and monitoring networks.

Key findings/messages

- Current levels were defined as total mercury (THg) measured between 2014 and 2024. Only a small set of the submitted data (31%) represents samples collected during the last decade.
- The most frequently sampled matrices were fish (muscle), followed by birds (feathers, blood, and eggs), invertebrates (soft tissues), and marine mammals (liver and muscle). Mercury concentrations varied widely, often spanning several orders of magnitude depending on tissue and species: invertebrates (10.2 to 2,070 ng g⁻¹ dw), fish (<0.1-16,400 ng g⁻¹ ww (muscle), birds (<0.1-6,000 ng g⁻¹ dw (eggs) and 2-72,790 ng g⁻¹ dw (feathers), 1- 8,691 ng g⁻¹ ww (blood)), marine mammals (16.5 to 5,962,000 ng g⁻¹ dw (liver) and 3-472,000 ng g⁻¹ dw (muscle)).
- Data coverage is uneven: intensive monitoring exists in North America, Europe, and East Asia, but large gaps remain in Africa, Latin America, South Asia, and parts of the Pacific. Monitoring in artisanal and small-scale gold mining (ASGM) regions is critical, as these are major mercury emission hotspots.
- The submitted biota data are suitable for analyzing spatial patterns at regional scales, but they cannot support global-scale assessments. This limitation reflects incomplete global geographic coverage and broad variability on the species and trophic positions among different regions. Spatial comparisons may be more feasible at regional level where the same species are well sampled such as North America, Europe or the Arctic.
- Greater, equitable and ethical utilization of knowledge, sciences and practices of Indigenous Peoples will strengthen the understanding of ecosystems, sources and pathways of mercury contamination, and greatly increase the relevance, inclusivity, and cultural appropriateness of monitoring programs.

2.4.3 Question 2: How do current levels of THg observed in (invertebrates, fish, birds, mammals) compare to established national / international benchmark and scientific levels associated with adverse effects on human health, wildlife and environmental sustainability?

2.4.3.1 Invertebrates

Information Used

The dataset described in section 0.1.2.1 was filtered to include only observations since 2014. Observed concentrations were compared to the Background Assessment Concentrations for the Northeast Atlantic region by the OSPAR Commission (OSPAR Commission, 2009) and the Maximum Permissible Concentration in food for the protection of public health by the European Commission (1881/2006/EC) were used to calculate the proportion of individuals exceeding thresholds for the period 2014-2023. In addition, within the European Water Framework Directive, Environmental Quality Standard (EQS) for mercury have been developed as a means to protect humans and organisms from secondary poisoning (20 µg/kg ww). This EQS applies to top predators and its use for bivalves requires its conversion to trophic level 2, which would therefore decrease its value. In absence of global marine trophic magnification factor (TMF), this conversion is not possible and it is taken as a high estimate of Hg EQS for bivalve. However, this EQS converted to 150 µg/kg dw, assuming the average %moisture for bivalves in the presented dataset of 86.5%, is a top value for EQS for bivalve and it falls in the range of the OSPAR BAC. It might therefore be considered for the present exercise that values higher than OSPAR BAC would also be higher than WFD EQS.

Methodology

Contemporary concentrations in oysters and mussels (time period 2014-2024) were compared to 1) the Background Assessment Concentrations (BACs) established by the OSPAR Commission to evaluate mercury concentrations in biota, and 2) the Maximum Permissible Concentration (MPC) in food for the protection of public health for the Northeast Atlantic region by the OSPAR Commission (OSPAR Commission, 2009). Observed concentrations well below BACs are considered to be near background levels (see here). Of note, the BAC and MPC thresholds are originally derived for the NE Atlantic area covered by OSPAR assessments.

Answer/response

Sixty five percent of all oyster observations and 59% of all mussel observations were above BACs (Figure 15). No observations exceeded the MPC (Figure 14).

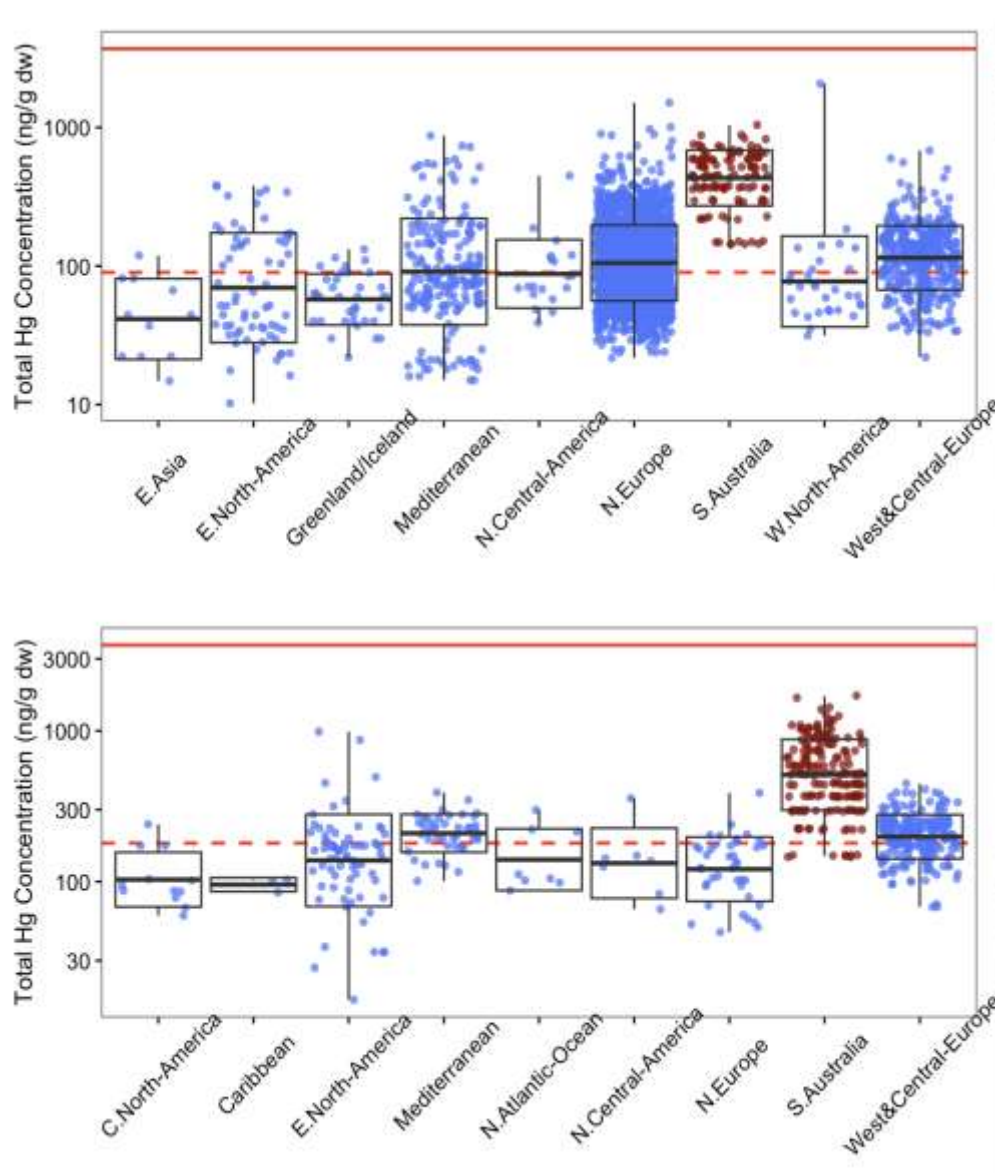


Figure 14. THg concentrations ($\text{ng g}^{-1} \text{ dw}$) for the current period (2014-2024) by IPCC Region. Boxplots show min, mean – SD, mean, mean + SD, and max concentrations. Points represent individual observations. The Top panel shows data for mussel species and the bottom panel shows data for oyster species. Horizontal dashed red lines represent Background Assessment Concentrations (BAC) below which concentrations are considered to be near background for the Northeast Atlantic region by the OSPAR Commission (OSPAR Commission, 2009); the BAC for mussels is 90 ng/g dw and the background assessment concentration for oysters is 180 ng/g dw . Horizontal solid red lines represent the Maximum Permissible Concentration (MPC) in food for the protection of public health by the European Commission (1881/2006/EC); the MPC value is 500 ng/g ww converted to 3704 ng/g dw assuming the average %moisture for bivalves in the presented dataset of 86.5%. Colors indicate whether data were associated with Hg point sources (red points). All S. Australia data are from the Nyrstar Hobart/Derwent Estuary Program monitoring Hg trends in the vicinity of a zinc smelter. The data are not representative of this IPCC region as a whole. These were the only data identified as being from sites with point sources, though others could be included in this dataset. Data are summarised for sampling locations shown in Figure 1 (map) and should not be used to characterize the entire region as named.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Mercury concentrations in bivalves were not considered in the 2018 Global Mercury Assessment.

Evaluation of Confidence: There are few mercury thresholds specific to bivalves. The BAC and MPC and thresholds from the OSPAR Commission were selected because they are established thresholds for bivalves. The BAC threshold was not altered, as it is expressed on a dry weight basis and therefore aligned with the dry weight basis of the majority of bivalve observations discussed herein. The OSPAR Commission MPC threshold was

converted from wet weight to dry weight units using the average % moisture (86.5%) of the bivalves included in invertebrate assessments. The BAC and MPC thresholds are originally derived for the NE Atlantic area covered by OSPAR assessments and may not be representative of background concentrations in other regions.

Information and Capacity Needs: Harmonized benchmarks representative of broader spatial areas are needed to improve comparability and interpretation across spatial areas. Continued development of analytical tools, such as species-specific risk calculators, alongside sustained investment in biomonitoring capacity, will further support effective assessment and management of mercury exposure biota.

2.4.3.2 Fish

Information Used

Toxicity benchmarks were selected by choosing Toxicity Reference Values (TRVs) that were mathematically derived using a benchmark dose analysis framework when available, as recommended by the U.S. Environmental Protection Agency and European Food Safety Authority. When mathematically derived Toxicity Reference Values were not available, we selected the best available benchmarks from subjectively derived Toxicity Reference Values from the specific recent review papers (Annex 1).

The toxicity benchmarks used in this assessment are based on a summary of derived TRVs for the effects of methylmercury on fish. Where possible, TRVs were selected from values mathematically derived using a benchmark dose analysis framework, following the recommendations of the U.S. Environmental Protection Agency (2012) and the European Food Safety Authority (Hardy et al. 2017). This benchmark dose methodology incorporates dose–response modeling and identifies the Benchmark Dose Lower Bound (BMDL) as the preferred toxicity benchmark, reflecting a conservative and protective estimate of adverse effect thresholds (U.S. EPA 2012; Davis et al. 2011; Hardy et al. 2017; Mayfield and Skall 2018; Gift et al. 2020).

When mathematically derived TRVs were not available, the best available benchmarks from recent peer-reviewed review papers were selected. These subjectively derived TRVs, while less rigorous than BMDL-based estimates, still provide valuable points of reference and represent consensus values within the scientific literature.

By comparing observed fish mercury concentrations against these TRVs, we can assess the likelihood of adverse effects on endpoints such as reproductive success, behavior, growth, and survival, thereby providing a consistent framework for evaluating ecological and human health risks across regions and taxa.

Methodology

Toxicity benchmarks values were used to calculate the proportion of individuals exceeding thresholds over 2014–2024.

Answer/response

Across nearly all regions, the vast majority (90.8%) of fish muscle THg concentrations (wet weight) exceeded the lowest TRV of 0.04 mg/kg ww, associated with impaired reproductive success. In half of the IPCC regions represented in the harmonized fish muscle subset exceedances were above 90%, with full exceedance (100%) in regions in Australia, Asia, Africa, and South America as well as regions in both the Atlantic and Pacific Oceans.

At the 0.50 mg/kg ww TRV (impaired behavior), exceedance rates were more variable but still widespread. In North America, as high as 36.2% (C. North-America) of fish muscle observations exceeded this threshold, with higher values in the S. Eastern-Africa (100%), Australia (E. Australia: 78.3%, S. Australia: 42.8%), Sahara (54%), and Mediterranean (42.6%), along with the North Atlantic and Pacific Oceans (93% and 100%, respectively).

Exceedance at the 1.44 mg/kg ww TRV (impaired growth) was less common overall, but substantial in certain regions. More than 60% of fish THg concentrations in the North Atlantic and Pacific Oceans and S. Eastern Africa and E. Australia exceeded this level.

At the highest TRV of 2.80 mg/kg ww (impaired survival), exceedances were rare overall but again regionally concentrated. High proportions were observed in S. Eastern Africa (38.9%) and N. Atlantic Ocean (39.5%).

Shark species, where present, displayed particularly high exceedance rates. All observations in the N.Atlantic- and N.Pacific-Ocean regions were sharks with at least 90% of observations exceeding 0.50 mg/kg ww TRV (impaired behavior). In the North Atlantic Ocean, where almost all records were sharks, exceedances remained very high, with 65.1% above the growth TRV and 39.5% above the survival TRV. A notable exception was S.E. South-America where 13.2% of the 38 observations exceeded this threshold despite all being sharks. However, closer examination reveals that only 10 observations were adults and four of those had muscle THg exceeding the highest TRV of 2.80 mg/kg ww (impaired survival).

Regions with limited sample sizes (e.g. S.E. and E. Asia, S.E. South America, Western Africa) showed mixed results, with low confidence due to small sample sizes.

Table 6. Proportion of fish muscle total mercury (THg) concentrations (wet weight, ug/g) exceeding toxicity reference values (TRVs) concentrations in fish muscle within the harmonized fish muscle subset by IPCC region during 2014–2024. TRVs correspond to impaired reproductive success (0.04), impaired behavior (0.50), impaired growth (1.44), and impaired survival (2.80). Values are expressed as percentages of all records per region; N = number of samples. The proportion of shark species are noted.

IPCC_Name	n (records)	Reproductive threshold	Behavioral threshold	Growth impairment threshold	Survival impairment threshold	Shark
		0.04 µg/g ww	0.50 µg/g ww	1.44 µg/g ww	2.80 µg/g ww	
Arctic-Ocean	131	43.5%	0.8%	0.0%	0.0%	0.0%
C.North-America	8644	96.9%	37.5%	4.2%	0.2%	0.0%
Caribbean	228	83.3%	7.9%	1.8%	0.9%	0.0%
E.Australia	60	100.0%	78.3%	60.0%	13.3%	78.3%
E.North-America	25197	93.7%	25.3%	2.4%	0.2%	0.0%
E.Southern-Africa	138	39.1%	2.9%	0.0%	0.0%	0.0%
Equatorial.Atlantic-Ocean	375	100.0%	12.5%	0.0%	0.0%	0.0%
Equatorial.Pacific-Ocean	126	72.2%	13.5%	4.0%	0.0%	13.5%
Greenland/Iceland	268	86.6%	11.6%	1.1%	0.4%	0.0%
Mediterranean	1,283	86.3%	42.6%	12.5%	3.0%	17.6%
N.Atlantic-Ocean	43	100.0%	93.0%	65.1%	39.5%	100.0%
N.Australia	999	96.6%	4.2%	0.0%	0.0%	0.0%

N.Central-America	184	78.8%	17.4%	5.4%	0.0%	20.1%
N.E.North-America	7,963	96.6%	32.7%	5.0%	0.4%	0.0%
N.Europe	17,168	87.7%	7.1%	0.3%	0.0%	0.0%
N.Pacific-Ocean	60	100.0%	100.0%	93.3%	10.0%	100.0%
N.South-America	375	91.7%	21.6%	3.7%	0.0%	5.9%
N.W.North-America	1,170	96.8%	7.9%	0.1%	0.0%	0.0%
N.W.South-America	866	39.7%	1.2%	0.1%	0.0%	0.0%
S.Asia	13	100.0%	0.0%	0.0%	0.0%	0.0%
S.Australia	486	100.0%	42.8%	8.8%	5.6%	8.2%
S.E.Asia	12	0.0%	0.0%	0.0%	0.0%	0.0%
S.E.South-America	38	100.0%	13.2%	10.5%	10.5%	100.0%
S.Eastern-Africa	108	100.0%	100.0%	89.8%	38.9%	0.0%
S.Pacific-Ocean	275	95.6%	13.5%	0.0%	0.0%	0.0%
Sahara	63	82.5%	54.0%	19.0%	3.2%	63.5%
South-American-Monsoon	1,220	65.2%	7.1%	3.2%	1.6%	0.0%
W.North-America	425	70.8%	2.6%	0.5%	0.0%	0.0%
West&Central-Europe	1,094	60.8%	1.4%	0.0%	0.0%	5.6%
Western-Africa	27	77.8%	0.0%	0.0%	0.0%	0.0%

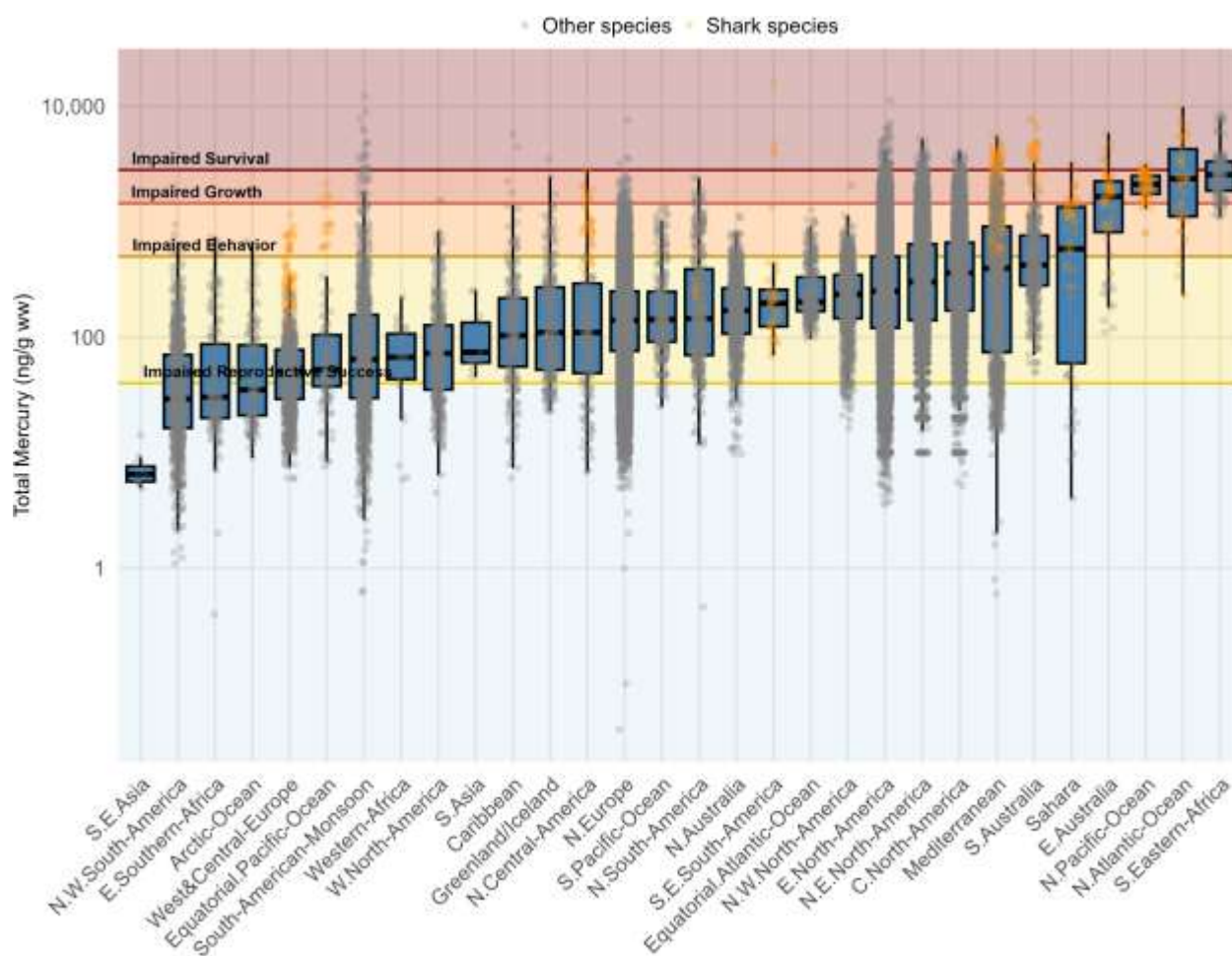


Figure 15. Boxplots of fish muscle total mercury (THg) concentrations (wet weight, ng/g within the harmonized fish muscle subset by IPCC region during 2014–2024. Data points are shown for individual records (orange = shark species, grey = other species). Horizontal colored lines indicate toxicity reference values (TRVs): 0.04 mg/kg (reproductive success, yellow), 0.50 mg/kg (behavior, orange), 1.44 mg/kg (growth, orange-red), and 2.80 mg/kg (survival, dark red). The y-axis is a log₁₀ scale.

(standardized to 80% moisture), this figure shows substantially greater variability and extreme values, which likely reflect data submission and QA/QC issues (e.g., inconsistent unit conversions, moisture content assumptions, or incomplete metadata) rather than biological differences.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

Our analysis of Toxicity Reference Value (TRV) exceedance patterns in fish aligns broadly with conclusions of the Global Mercury Assessment 2018 (UNEP, 2019a). The GMA emphasized that methylmercury bioaccumulation in aquatic food webs poses the most significant risk to wildlife and humans, with regional hotspots of concern and significant data gaps.

In our exceedance analysis, these same regions emerge with elevated frequencies of benchmarks being surpassed, suggesting consistency across independent datasets. However, some discrepancies are also apparent. For instance, our analysis shows relatively higher exceedance rates in North American freshwater systems compared to GMA's more generalized regional synthesis, which may reflect differences in dataset composition, QA/QC harmonization, and the inclusion of more recent records in our compilation. Importantly, both GMA and our study highlight the uneven global coverage of biotic mercury monitoring, with persistent data gaps in Africa, South Asia, and parts of the Pacific, underscoring the continued need for improved spatial representation in global mercury biomonitoring. Overall, while the GMA provides a broad global framing, our exceedance-based approach refines the interpretation by directly linking observed concentrations to health-relevant thresholds for wildlife.

Our exceedance-based assessment reinforces the Global Mercury Assessment 2018 findings by confirming that mercury concentrations in fish and wildlife frequently surpass toxicity benchmarks in key regions while also underscoring persistent data gaps in Africa, South Asia, and the Pacific that limit global representativeness. Our analysis also highlights the importance of standardized and targeted sampling protocols as variability in individual, species, site factors significantly impact the interpretability of patterns in Hg concentrations across space and time.

Evaluation of Confidence:

The validation of benchmark values was conducted to ensure their scientific robustness, and the compatibility of reported units and tissue matrix types was verified, including conversion between wet weight and dry weight values and the standardization of equivalent matrices.

Comparison of fish mercury datasets derived from dry weight versus directly reported wet weight values highlighted important QA/QC considerations. While converted wet weight data provides a harmonized and consistent basis for dataset integration and cross-regional comparisons, the assumption of 80% moisture content when sample specific moisture was not available inherently introduces uncertainty due to natural variability in tissue moisture between individuals and samples. This uncertainty reflects inconsistencies in data submission practices (e.g., differing assumptions on tissue moisture, unit errors, or incomplete metadata).

These discrepancies emphasize the importance of standardized reporting templates requiring both wet and dry weight concentrations, explicit documentation of moisture content, and unit verification prior to data integration. Overall, confidence is high in the TRV exceedance framework where data have been standardized and quality-controlled, while confidence is lower in regions or datasets where reporting practices are heterogeneous or sample sizes are small.

Information and Capacity Needs:

Broader adoption of harmonized benchmark frameworks, including systematic use of mathematically derived TRVs with transparent documentation, will strengthen the scientific basis of exceedance assessments. Enhanced biomonitoring in underrepresented regions (Africa, South Asia, the Pacific, Latin America and the Caribbean), coupled with more consistent sampling of focal species at sentinel sites, is needed to improve temporal and spatial resolution. Strengthened QA/QC procedures—such as standardized templates requiring both wet and dry weight reporting, moisture content, and verified metadata—will reduce uncertainties and improve comparability. Continued investment in analytical tools (e.g., risk calculators, data-sharing platforms) and laboratory capacity, particularly in developing regions, will close monitoring gaps and support more robust global mercury assessments under the Minamata Convention.

2.4.3.3 Birds

Information Used

Toxicity benchmarks were selected by choosing Toxicity Reference Values (TRV) that were mathematically derived using a benchmark dose analysis framework when available, as recommended by the U.S. Environmental Protection Agency and European Food Safety Authority. When mathematically derived TRVs were not available, we selected the best available benchmarks from subjectively derived TRVs from the specific recent review papers (Annex 1).

Methodology

TRVs were used to calculate the proportion of individuals exceeding thresholds using data from the time period, 2014-2023.

Answer/response

Eggs

Egg THg concentrations (on a wet weight basis) were largely below a toxicity reference value (TRV) of 0.7 µg/g. Only 2% of observations exceeded the threshold for a risk of 10% decline in toxicological response (EC10) due

to mercury exposure. Exceedances of the TRV occurred in most regions of North American and also in Northern Europe.

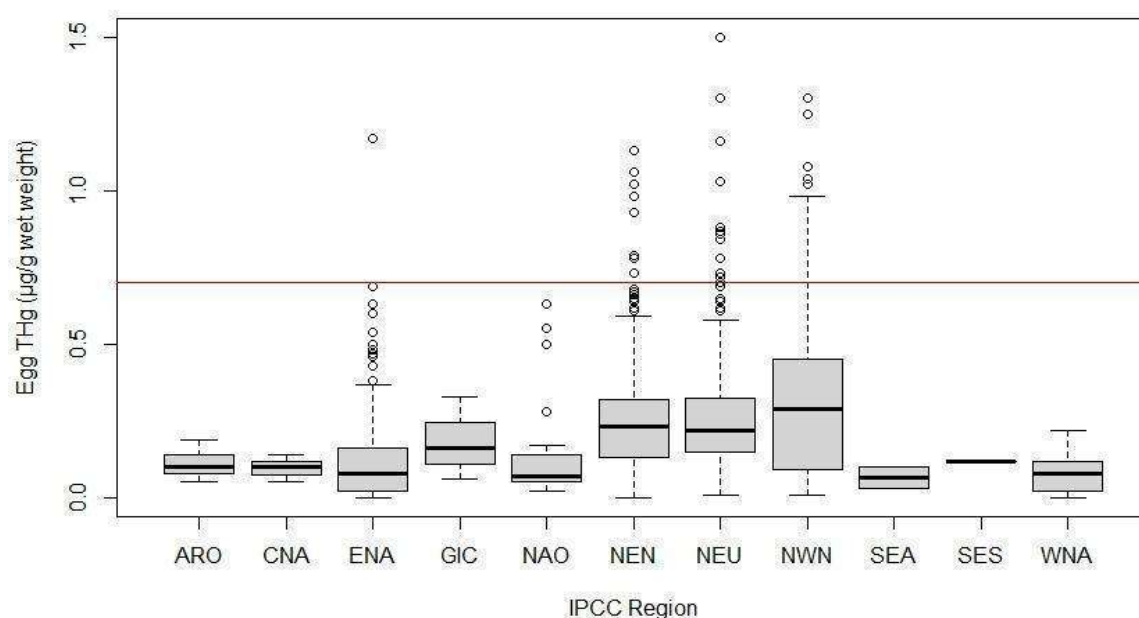


Figure 16: THg concentrations in eggs from bird samples, according to IPCC regions

Feathers

Nearly 35% of feather THg concentrations (on a dw basis) were above a toxicity reference value at no apparent effect category (Chastel et al., 2022). Nearly 23% (2872 of 12123) were at low risk, while 6% (731), 1 % (128) and 4 % (447) were at median, high and severe risk categories, respectively (Figure 17, Table 7).

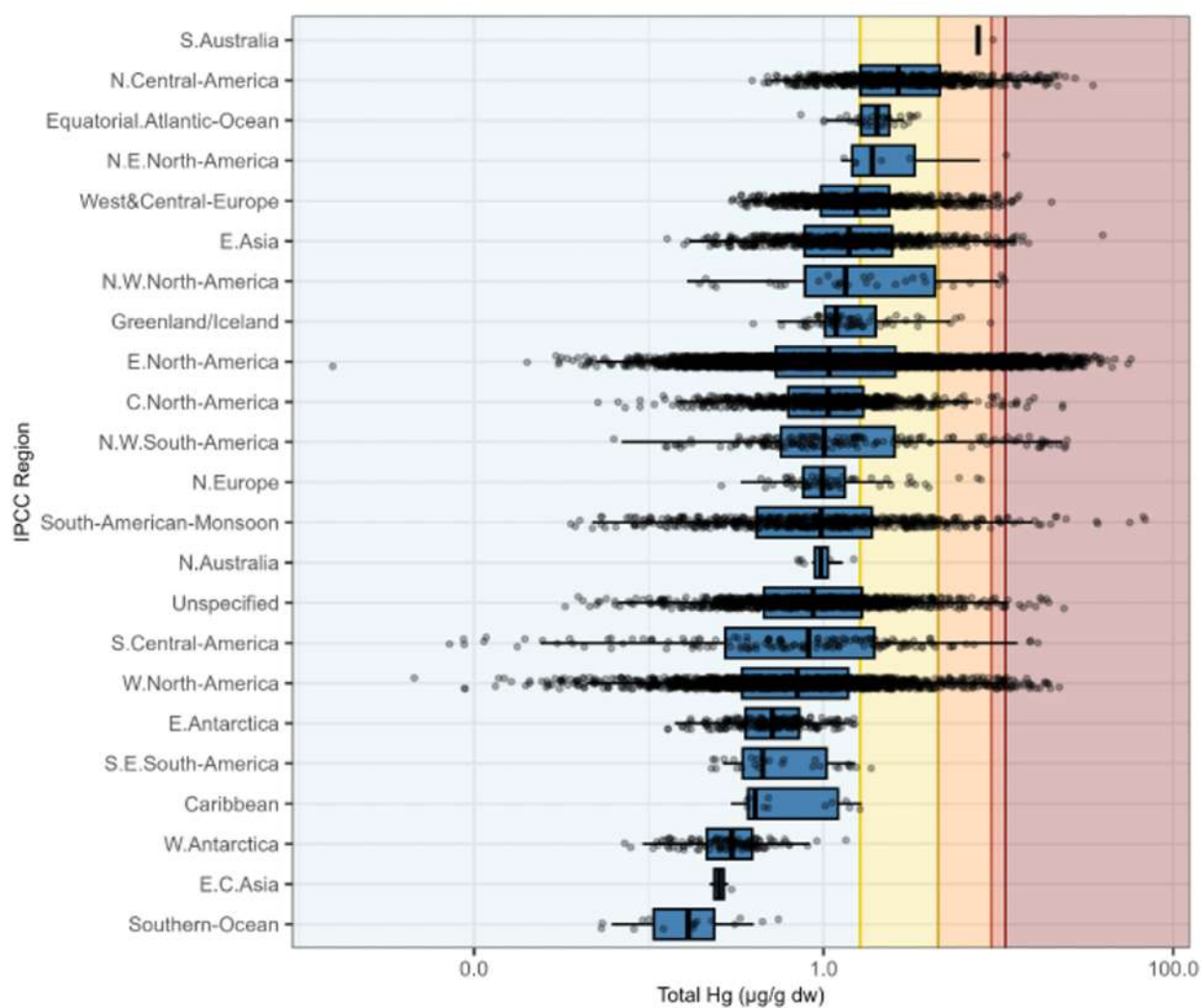


Figure 17. Total Hg (THg, $\mu\text{g/g}$, dw) concentration in feathers across different continents. The x-axis is truncated at $80 \mu\text{g/g}$ for clarity, data points of $100.90 \mu\text{g/g}$ and $123.28 \mu\text{g/g}$ are not shown. Vertical red dashed lines show Hg toxicity thresholds Chastel et al.(2022). From left to right, representing no apparent risk ($<1.62 \mu\text{g/g}$), low risk ($1.62\text{--}4.53 \mu\text{g/g}$; health, physiology, behaviour and reproduction affected), moderate risk ($4.53\text{--}9.14 \mu\text{g/g}$; severe impairment to health and reproduction), high risk ($9.14\text{--}10.99 \mu\text{g/g}$; physiological and reproductive effects, often complete reproductive failure), and severe risk ($\geq 10.99 \mu\text{g/g}$; severe physiological and reproductive effects, including adult mortality). The X-axis is a log10 scale.

Table 7: The risk category for THg in feathers

Risk Category	Percentage(%)	Number
No apparent effect	65.5	7942
Low risk	23.7	2869
Median risk	6.0	731
High risk	1.1	128
Severe risk	3.7	447

Blood

For bird blood, Toxicity Reference Values (TRVs) from Ackerman et al. (2024) were used to assess the likelihood of adverse impacts on ecological health across regions and species. The TRVs were derived using analyses across taxa, so are applied for all Orders in the blood subset. The TRVs combined survival and reproduction endpoints recommended for assessing injury to wildlife.

Across all IPCC regions, 20.3% of adult blood THg concentrations were below the EC01 effect level (low injury; 0.09 $\mu\text{g/g ww}$). An additional 54.4% of samples exceeded the EC01 threshold, 14.6% exceeded EC05 (moderate injury; 0.6 $\mu\text{g/g ww}$), 9.6% exceeded EC10 (high injury; 1.3 $\mu\text{g/g ww}$), and 1.2% of samples exceeded the EC20 threshold (severe injury; 3.2 $\mu\text{g/g ww}$).

Exceedances were distributed across all IPCC regions. The only sample (Wattled Jacana [*Jacana jacana*]) from N.W. South-America exceeded the EC10 threshold. A total of 12.9% of the only species sampled in N. Central-America (Peregrine Falcons [*Falco peregrinus*]) exceeded either EC10 or EC20. E. North-America (12.1%) was the remaining IPCC region with >10% of samples exceeding EC10 or EC20, with 63.7% of samples (n=479) of Common Loon (*Gavia immer*) in the region exceeding these thresholds.

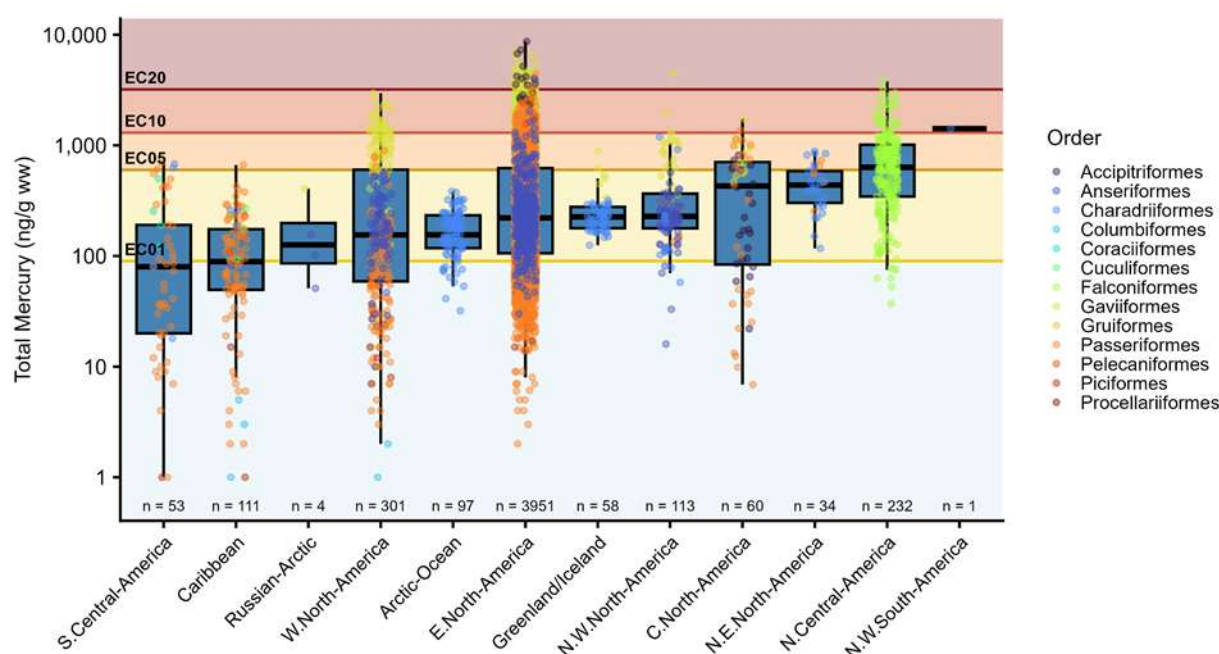


Figure 18. Boxplots of adult bird blood total mercury (THg) concentrations (ng/g ww) within the harmonized adult bird blood subset by IPCC region during 2014–2024. Data points are shown for individual records (colors by Order). Horizontal colored lines indicate toxicity reference values (TRVs): 90 ng/g (EC01, yellow), 600 ng/g (EC05, orange), 1300 ng/g (EC10, orangered), 3200 ng/g (EC20, darkred). The y-axis is a log10 scale.

Consistency of Answer with Existing Literature:

A limited comparison indicated that the Toxicity Reference Value (TRV) exceedance assessment broadly aligns with the conclusions of the 2018 Global Mercury Assessment (GMA; UNEP, 2019a). While the 2018 GMA did not quantitatively assess mercury body burdens in relation to TRVs, it emphasizes the variability in mercury body burdens in birds, driven by a suite of confounding factors, resulting in a heterogeneous distribution. Consequently, some species and regions exhibit elevated risk, as reflected in the individuals with body burdens exceeding the established toxicity benchmarks. The GMA notes that elevated body burdens can have adverse impacts (e.g., reduced reproductive capacity, altered behavior) underscoring the ecological significance of mercury contamination. This analysis demonstrated a similar heterogeneity in body burdens across individuals, species, and regions highlighting the importance of monitoring to identify patterns and populations at risk.

Evaluation of Confidence:

The validation of benchmark values was conducted, and the compatibility of reported units and matrix types was verified, including the conversion between wet weight and dry weight values and the standardization of equivalent matrices. While the framework of reporting mercury body burdens in birds in the context of TRVs is established and has a high degree of confidence, there is lower confidence in assessing regions or taxa with small sample sizes and incomplete metadata. This highlights the importance of standardized, targeted sampling, comprehensive metadata collection, and sufficient replication.

Information and Capacity Needs:

The broader adoption of harmonized benchmark frameworks is needed to improve comparability and interpretation across studies. Continued development of analytical tools, such as species-specific risk calculators, alongside sustained investment in biomonitoring capacity, will further support effective assessment and management of mercury exposure to biota.

2.4.3.4 Marine Mammals

Information Used

Risk categories for adverse effects of mercury on the reproduction, physiology, condition, and behaviour of mammals have been developed by Dietz et al. (2022). To evaluate the risk of mercury toxicity in marine mammals, we compared total mercury concentrations in liver to the four risk categories: $\leq 16 \mu\text{g g}^{-1} \text{ ww}$ (No risk), $16\text{--}83 \mu\text{g g}^{-1} \text{ ww}$ (Low risk), $83\text{--}126 \mu\text{g g}^{-1} \text{ ww}$ (High risk), and $\geq 126 \mu\text{g g}^{-1} \text{ ww}$ (Severe risk).

Methodology

Inclusion criteria: Comparisons with toxicity thresholds were conducted twice: first including all marine mammal taxa collected from 1969 to 2023 (55 species, Figure 18); then including only marine mammal taxa collected from 2014 onwards (15 species, Figure 19). As these benchmarks were estimated only in liver tissue and on a wet weight basis, we included only hepatic mercury concentrations ($n = 4295$ for all years; $n = 1999$ for the 2014–2024 period), expressed as $\mu\text{g g}^{-1} \text{ ww}$.

Exclusion criteria: Hg measurements in all other tissues except liver were excluded.

Data standardization and harmonization: Prior to comparison with benchmarks, data expressed on a dry weight basis were converted to wet weight using a moisture content of 70%, as estimated for liver tissue in several marine mammal species (Hawk et al., 2024; Yang et al., 2003).

Answer/response

When including taxa from 1969 to 2023, mercury levels reported in the liver of marine mammals fell within the risk range ($\geq 16 \mu\text{g g}^{-1} \text{ ww}$) for 34 out of 55 species (62%), while 21 out of 55 species (38%) presented individuals in the high to severe risk range. Short-finned and long-finned pilot whales (*Globicephala macrorhynchus* and *G. melas*), striped dolphins (*Stenella coeruleoalba*), pygmy killer whales (*Feresa attenuata*), polar bears, ringed seals (*Pusa hispida*), grey seals (*Halichoerus grypus*), belugas (*Delphinapterus leucas*), Cuvier's beaked whales (*Ziphius cavirostris*), and strap-toothed beaked whales (*Mesoplodon layardii*) consistently presented individuals exceeding severe-risk mercury thresholds across regions.

When considering only data from the 2014–2024 period, only 15 species remained in the dataset. Mercury levels reported in the liver of marine mammals were in the risk range ($\geq 16 \mu\text{g g}^{-1} \text{ ww}$) for 8 out of 15 species (53%), while 6 out of 15 species (40%) presented individuals in the high to severe risk range. Consistent with the 1969–2023 analysis, beluga, Blainville's beaked whale, false killer whale, grey seal, long-finned pilot whale, polar bear, ringed seal, and strap-toothed beaked whale presented individuals at risk across regions.

Most of the listed odontocetes occupy meso- and bathypelagic foraging niches, feeding largely on high-trophic-level prey such as cephalopods and deep-water fishes (Cioffi et al., 2021; McInnes et al., 2024; Plint et al., 2023; Santos et al., 2014; Shearer et al., 2022). Polar bears and grey seals are apex predators of their respective ecosystems, feeding on other marine mammals (e.g., ringed, harp, and hooded seals or harbour porpoises for polar bears; harbour porpoises for grey seals) (Stringell et al., 2014; Galicia et al., 2021). Polar bears are also exposed

to terrestrial inputs, including seabird eggs (Petherick et al., 2021). The extensive use of offshore and deep-ocean habitats by toothed whales, and of sea ice or terrestrial habitats by polar bears and other Arctic species, places these animals high in marine food webs where mercury efficiently bioaccumulates.

Polar bears, ringed seals, and long-finned pilot whales are also harvested and consumed by Arctic Indigenous Peoples, and northern communities. Ringed seals and polar bears have long provided essential sources of food and cultural identity across Arctic regions, while long-finned pilot whales are hunted and consumed in the North Atlantic, particularly in the Faroe Islands (AMAP 2021b). This human consumption pathway means that elevated mercury levels observed in these species may pose potential health risks for northern populations relying on them as part of traditional diets (AMAP 2021b).

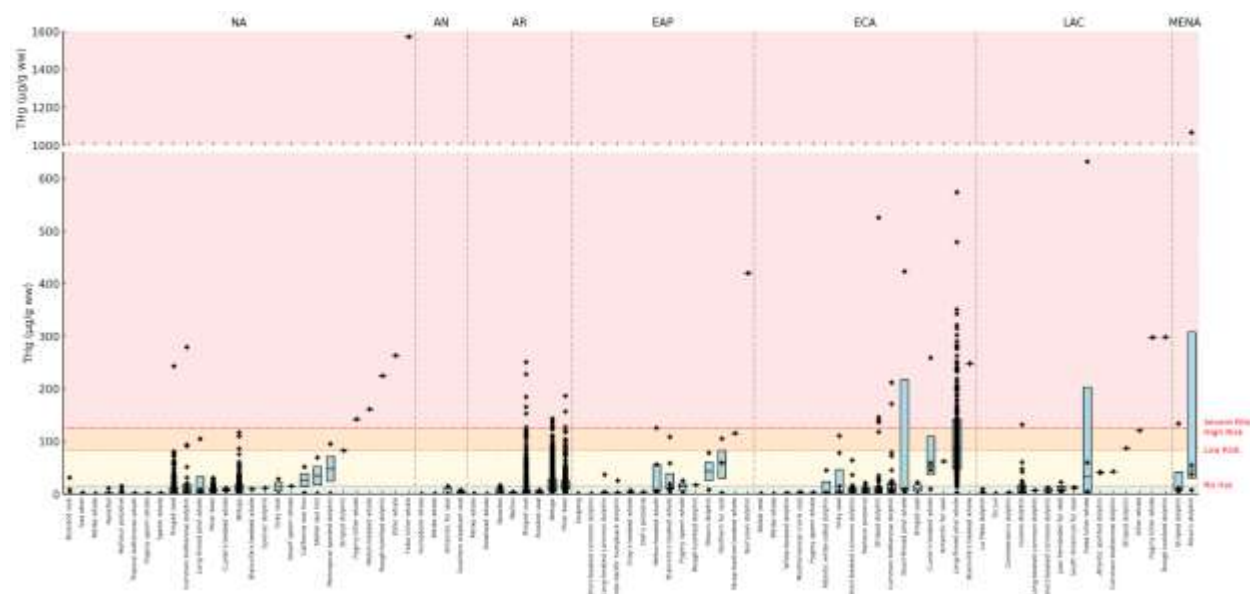


Figure 18. Individual THg concentrations in marine mammal liver ($\mu\text{g/g ww}$) for the period 1969 to 2024. The red-dotted lines indicate the thresholds for low ($< 16 \mu\text{g/g ww}$), moderate ($16 - 83 \mu\text{g/g ww}$), high risk ($83-126 \mu\text{g/g ww}$) and severe risk ($> 126 \mu\text{g/g ww}$) of adverse effects of mercury on the reproduction, physiology, condition and behaviour of mammals according to the categorization made by Dietz et al. 2022. Concentrations are reported as median and by region. NA: North America, AN: Antarctica, AR: Arctic, EAP: East Asia & Pacific, ECA: Europe & Central Asia, LAC: Latin American & Caribbean, MENA: Middle East & North Africa

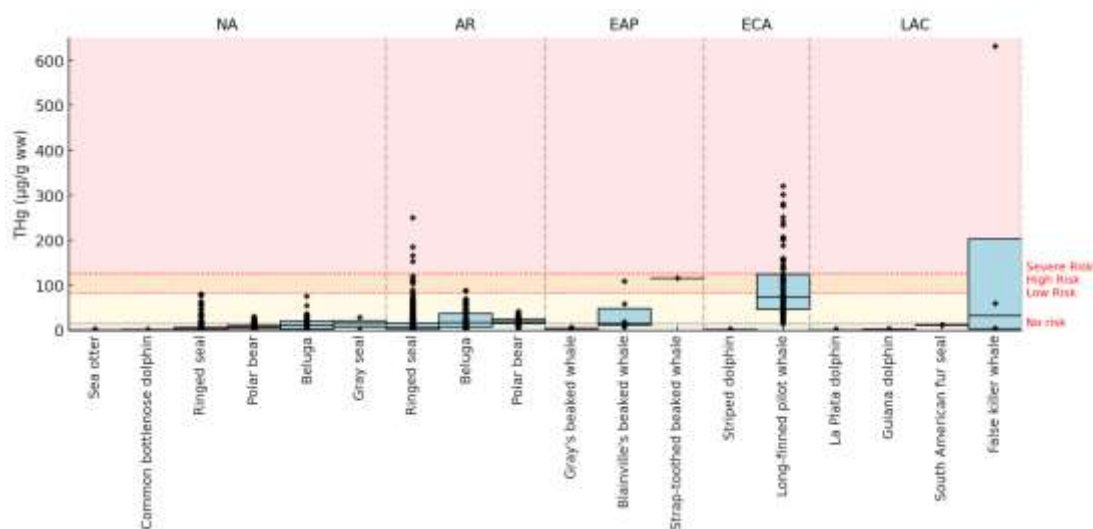


Figure 19. Individual THg concentrations in marine mammal liver ($\mu\text{g/g ww}$) for the period 2014-2024. The red-dotted lines indicate the thresholds for low ($< 16 \mu\text{g/g ww}$), moderate ($16 - 83 \mu\text{g/g ww}$), high risk ($83-126$

µg/g ww) and severe risk (> 126 µg/g ww) of adverse effects of mercury on the reproduction, physiology, condition, and behaviour of mammals according to the categorization made by Dietz et al. 2022. Concentrations are reported as median and by region. NA: North America, AN: Antarctica, AR: Arctic, EAP: East Asia & Pacific, ECA: Europe & Central Asia, LAC: Latin American & Caribbean, MENA: Middle East & North Africa.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

There was insufficient time and resources to undertake a literature review and comparisons with the 2018 GMA to determine consistency with earlier research.

Evaluation of Confidence:

The validation of benchmark values was conducted, and the compatibility of reported units and matrix types was verified, including the conversion between wet weight and dry weight values and the standardization of equivalent matrices.

Information and Capacity Needs:

The broader adoption of harmonized benchmark frameworks is needed to improve comparability and interpretation across studies. Continued development of analytical tools, such as species-specific risk calculators, alongside sustained investment in biomonitoring capacity, will further support effective assessment and management of mercury exposure to biota. Only a limited number of mercury toxicity thresholds have been established to date, and many are based on very old experimental studies (e.g., the four risk categories defined by Dietz et al. 2022 were derived from Donald et al.'s live exposure experiment on harp seals in 1977). Others have been extrapolated from terrestrial mammalian species, which may not be appropriate for marine mammals given their unique physiology and ecology.

Key findings/messages

- **Invertebrates:** Over half of oyster and mussel samples exceeded the background assessment concentrations defined by OSPAR Commission for the Northeast Atlantic region, but none surpassed food safety limits, indicating low human health risk.
- **Fish:** Mercury contamination is widespread, with significant reproductive, behavioral, growth and survival risks especially in Europe, the Mediterranean, and the Arctic. Sharks show consistently high levels, often exceeding all toxicity thresholds. Data gaps remain in East Asia, Northern South America, Latin America and the Caribbean, and Western Africa.
- **Birds:** Around one-third of feather samples exceeded toxicity reference values; ~10% between median and severe risk categories; most egg samples were below risk thresholds, but blood concentrations indicate varying risk of adverse effects across regions
- **Marine mammals:** Many species, particularly toothed whales, show liver mercury levels above low to severe risk thresholds.
-

2.4.4 Question 3: How do THg in species vary geographically over time (2000-current)? (for non-migratory species?)

Due to the limited availability of global-scale data, temporal analyses of THg concentrations in certain taxa were primarily based on a small number of case studies, conducted at local or regional levels.

Recommended criteria for analysing temporal trends:

- Data series spanning at least five years
- Data series ending in 2015 or later
- Ideally, one species should be sampled consistently over time; however, comparisons at a higher taxonomic level are also possible
- Same tissue type sampled
- Same location or region sampled
- Use of standardized sampling and analytical procedures

2.4.4.1 Invertebrates

Information Used:

Temporal analyses of THg concentrations in bivalves can be pursued for subsequent reports should sufficient time allow for a robust approach. In the meantime, we summarized results of prior analyses by AMAP/OSPAR, HELCOM, and NOAA Mussel Watch as the monitoring programs contributing the majority of the bivalve data discussed herein.

OSPAR: In the OSPAR regions, in sediment, fish and bivalves (a different picture is observed for long lived organisms), the OSPAR convention (<https://oap.ospar.org/en/ospar-assessments/quality-status-reports/qsr-2023/thematic-assessments/hazardous-substances>) reported in 2023 that most metals follow a general downward trend over the last 20 years, though smaller trends are observed in the most recent year compared with former decades (Figure 20).

However, in the more populated OSPAR Regions, mercury concentrations in fish and bivalves did not decrease in the last 20 years. There is even an overall increasing trend for pollution in the Southern North Sea in Region II (within the IPCC regions NEU: North Europe) with three times more upward than downward trends, whereas no significant improvements have been seen in the Northern North Sea, the English Channel, or the Northern Bay of Biscay (IPCC regions NEU and CEU: Central Europe), which have poor environmental status and consequently, do not move close to background values for naturally occurring mercury.

Analysis of mercury inputs, however (see: Inputs of Mercury, Cadmium and Lead via Water and Air to the OSPAR Maritime Area), shows that those from air and rivers are decreasing, which is also reflected in the decreasing mercury concentrations found in sediment. The generally upward time trends for fish and shellfish could be due to remobilisation of historically contaminated materials from sediment to water and marine organisms or to a change in the uptake of metals by fish and shellfish owing to climate change. Indeed, observed oxygen depletion (hypoxia and anoxia) and temperature increases in the marine environment provide a favorable environment for mercury methylation, and therefore favors its subsequent uptake in food webs and in several cases has also been found to increase the toxicity of contaminants. Continuation of the current dependence on coal in Russia, China, and India for power generation, together with increasing risks of forest fires, could imply a risk of increasing atmospheric deposition of mercury in the future.

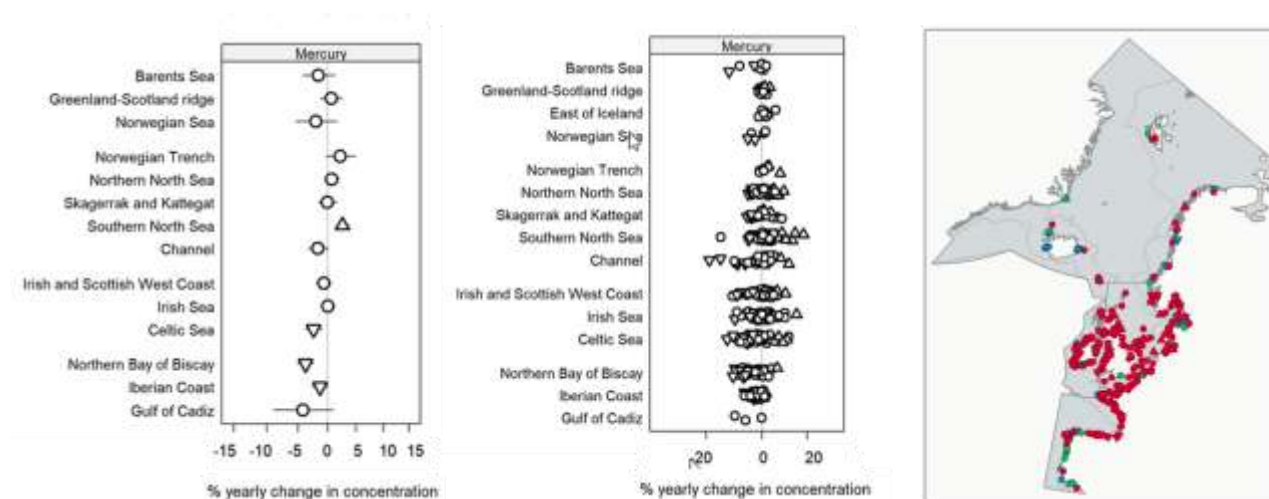


Figure 20. Temporal change in the last twenty years in mercury concentrations in biota (fish and bivalves) from the North East Atlantic covering AMAP and OSPAR regions (North East Atlantic). On the left panel: regional trend estimates with pointwise 95% confidence intervals are plotted. On the middle panel: trend estimates from the individual time series on which the regional assessment is based. On the right panel: geographical spread of the time trend analysis status assessment. The symbols in all these plots have the following interpretation: downward triangle: the mean concentration is significantly decreasing ($p < 0.05$), circle: there is no significant change in mean concentration ($p > 0.05$), upward triangle: the mean concentration is significantly increasing ($p < 0.05$). The colours have the following meaning: blue: the mean concentration is significantly ($p < 0.05$) below the Background Assessment Concentration (BAC), green: the mean concentration is significantly ($p < 0.05$) below the Environmental Quality Standard based on secondary poisoning (EQSsp = 0.020 mg/kg ww), red: the mean concentration is not significantly below the EQSsp. From the 2025 annual assessment on the OSPAR Hazardous Substances Assessment Tool (OHAT: <https://dome.ices.dk/ohat/?assessmentperiod=2025>) using HARSAT approach.

HELCOM: The HELCOM data cover the Baltic Sea area, adjacent to the Northern European Seas covered by OSPAR and also contributing to the data for the N. Europe IPCC region (Figure 21).



Figure 21. HELCOM area, covering the Baltic Sea zone

HELCOM mercury monitoring data (a regional focus has been agreed on biota) are regularly assessed in relation to established criteria, with good status achieved when the concentrations of mercury are below the regionally agreed threshold values. For invertebrate molluscs (*M. edulis*, + *M. baltica* + *Dreissena polymorpha*) a threshold (EQS biota secondary poisoning) of 20 µg/kg ww is applied. The soft body tissue material is used in analysis.

HELCOM assessments integrate results from different media (i.e., selected fish species also monitored); conclusions reflected below are therefore not based on invertebrate data alone and the majority of the blue mussel monitoring sites are in the southern Baltic Sea (also reflecting species distribution patterns). In its most recent assessment the available mercury data are considered sufficient to allow a wide-ranging evaluation to be made for the Baltic Sea. Despite apparent reductions in mercury inputs and also decreasing trends (i.e. lower concentrations) at some stations where distinct trends could be assigned, the mercury status objectives were not reached in almost all evaluated sub-areas, with all but two coastal areas and all open sea assessment units evaluated as failing to achieve Good Environmental Status for mercury (<https://indicators.helcom.fi/indicator/mercury/>). The HELCOM evaluation of mercury includes a large number of high-quality biota datasets with long time-series and the possibility to assign statistical trends. Distinct downward trends were recorded at 13 stations and 8 stations showed trends of increasing concentrations ([Thematic Assessment of hazardous substances 2023](#)).

NOAA Mussel Watch: For the purposes of this report, we draw on two recently published trend analyses from the NOAA National Centers for Coastal Ocean Science for the Gulf of Maine (IPCC region = E. North America) and Pacific Northwest (IPCC region = W. North America) regions of North America. Bivalve (*Mytilus* sp.) Hg concentrations at Mussel Watch and Gulf Watch sites in the Gulf of Maine (including coastal sites in Massachusetts, New Hampshire, Maine, and Nova Scotia) were generally low (<LD to 100 ng/g dw) with a few sites identified as having medium concentrations (100 to 200 ng/g dw) and one site as having high concentrations (300 to 1300 ug/g dw). Trend analysis (Figure 23b and c) indicates that 2015/2016 concentrations were mostly low relative to historic patterns. Only one site showed a significant increasing trend for the period 1986-2013, whereas four sites showed a significant decreasing trend over the same time period. Overall, concentrations for the period 1986-2016 presented in this report are spatially and temporally variable. *Mytilus* sp. sampling in the Pacific Northwest of the USA represents coastal sites in Washington and Oregon (Figure 22). Concentrations are generally low in this region. In 2019, concentrations among sampling sites ranged from 30 to 190 ng/g dw, with 94% of sites having low concentrations (LD to 160 ng/g dw). Trend analysis indicates that in 2019, the most recent year of sampling for this report, concentrations at the majority of surveyed sites are similar or lower than historic median concentrations, with two sites showing significantly decreasing concentrations and one site showing significantly increasing concentrations (Figure 23b). There have not been significant changes in tissue concentrations regionally over time.

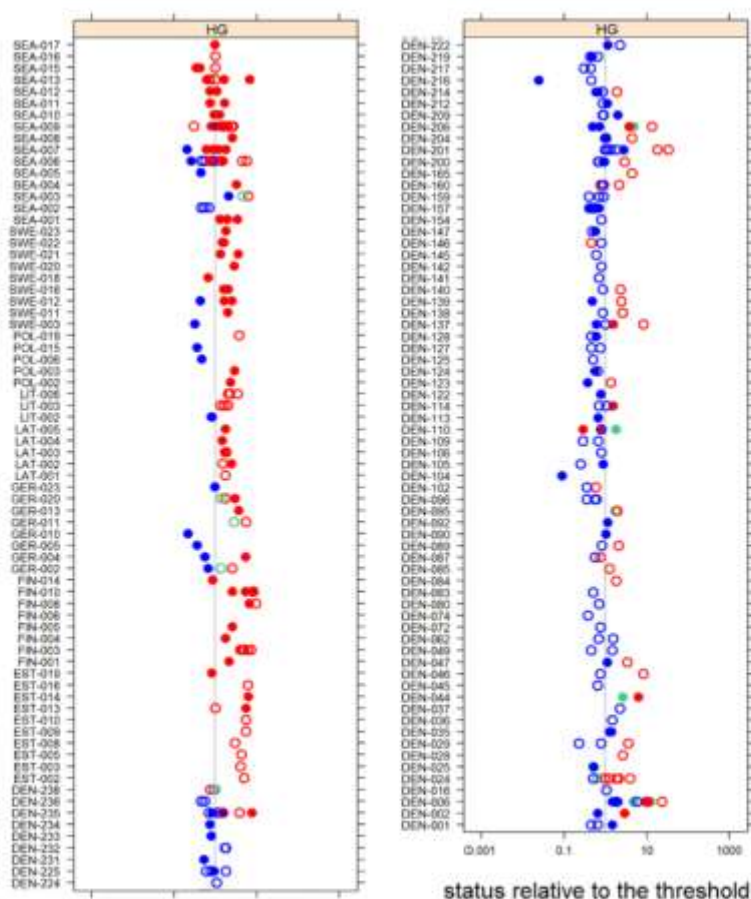


Figure 22. The same assessment units as shown in Figure 23 but each assessment unit visualises the individual stations included in making the assessment unit level status evaluation. Potential differences in evaluation outcome due to sampling matrices are highlighted: Red=fish muscle, blue=mussel soft body, and green=fish liver.

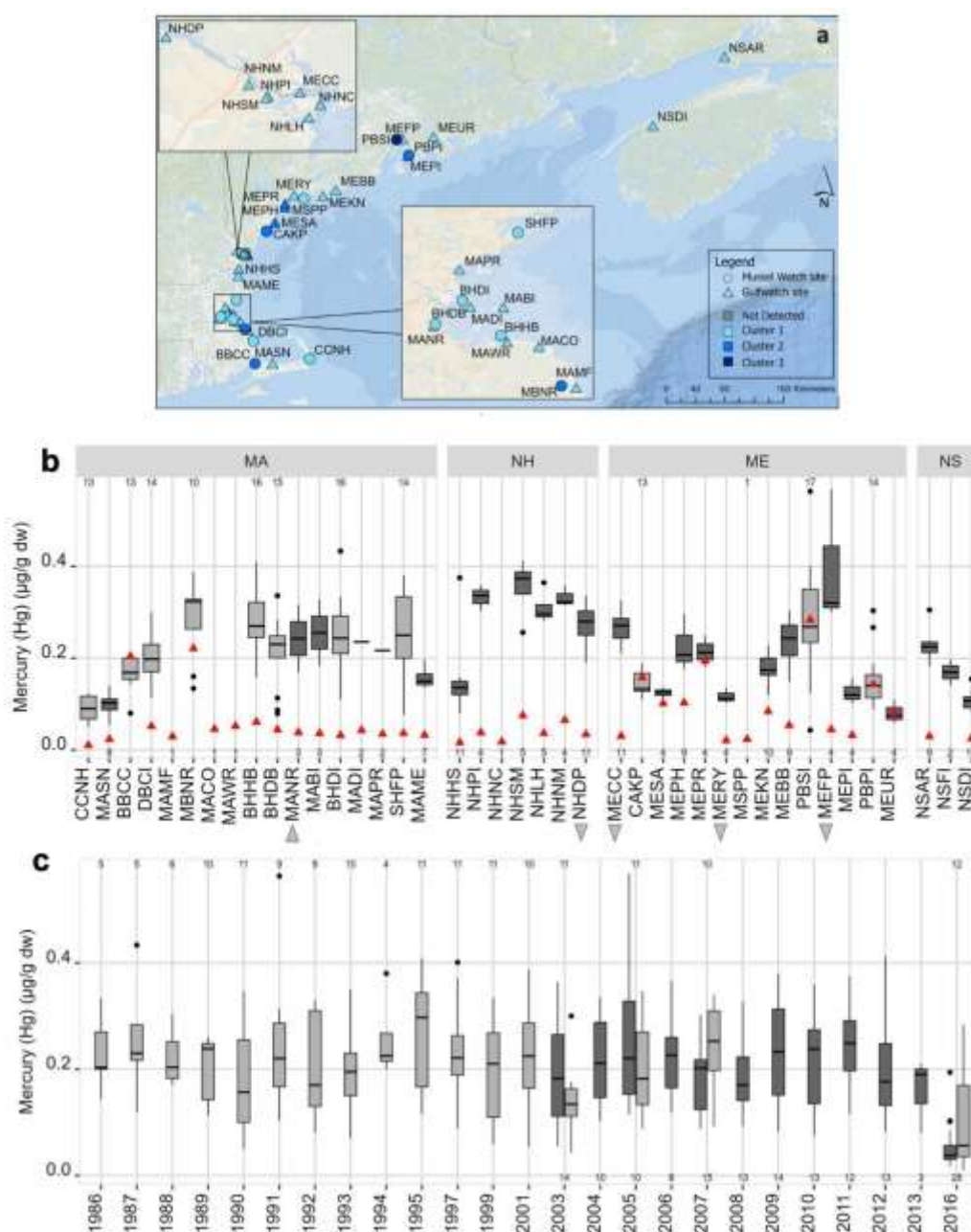


Figure 23. a) Map of Mussel Watch and Gulf Watch sampling locations in the Gulf of Maine. b) Trend analysis showing 2015/2016 sampling results (red triangles) relative to historic concentrations shown in light grey boxplots (Mussel Watch program) and dark grey boxplots (Gulf Watch program) for the period 1986-2013. Significant trends are noted by arrows beneath specific sites, with upwards arrows indicating increasing trend and downward facing arrows indicating decreasing trends. Sites are grouped by state or region (MA: Massachusetts, USA; NH: New Hampshire, USA; ME: Maine, USA; NS: Nova Scotia, Canada). The number of years sampled since 1986 is noted above the plot for the Mussel Watch program and below the plot for the Gulf Watch program. c) Boxplots (light grey, Mussel Watch program; dark grey, Gulf Watch program) of temporal patterns across sites from 1986-2016. The number of sites sampled since 1986 is noted above the site for the Mussel Watch program and below the plot for the Gulf Watch program. Figure modified from Swam et al. (2023).

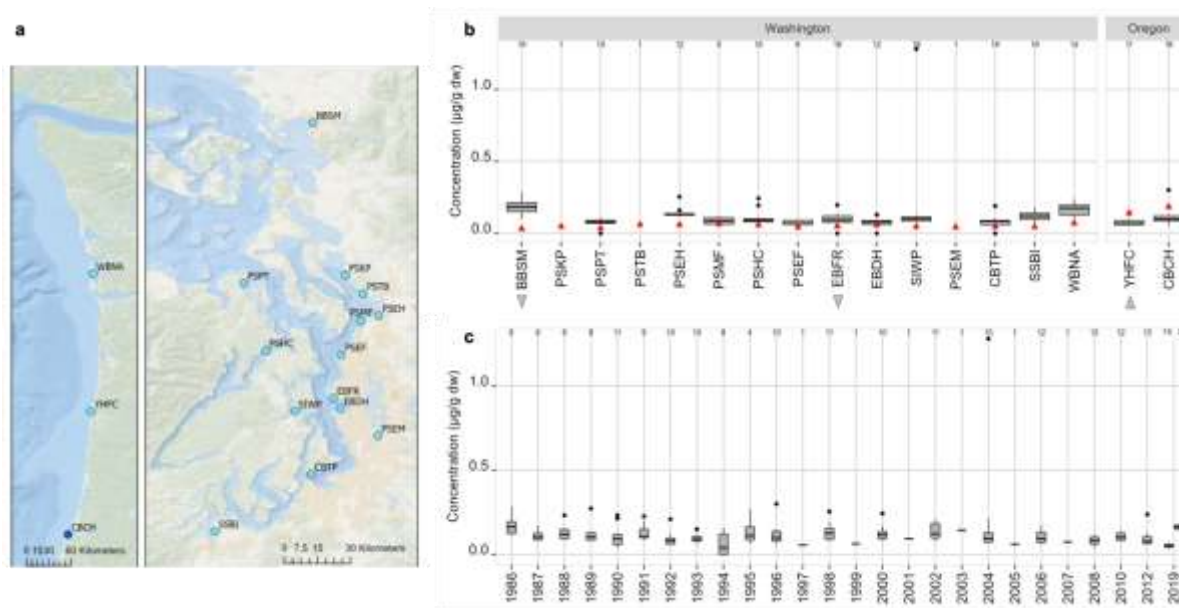


Figure 24. a) Map of Mussel Watch sampling locations in the Pacific Northwest, USA. b) Trend analysis showing 2019 sampling results (red triangles) relative to historic concentrations shown as grey boxplots for the period 1986-2012. Significant trends are noted by arrows beneath specific sites, with upwards arrows indicating increasing trend and downward facing arrows indicating decreasing trends. Sites are grouped by state and number of years sampled since 1986 is noted above the plot. c) Boxplots of temporal patterns across sites from 1986-2019. The number of sites sampled since 1986 is noted above the plot. Figure modified from Swam et al. (2025).

2.4.4.2 Fish

Information Used:

Data were compiled from different data sources: submissions to the OESG repository and public domain (Annex 1). Several long-term datasets exist for certain sites, which could support temporal trend analyses. In the meantime, we summarized results of analyses for Great Lakes and Nordic Fish.

Answer/response

Great Lakes

Both the US and Canada have monitoring programs for fish in the Great Lakes that include Hg and other contaminants. In the US, data are collected by the US Environmental Protection Agency (USEPA) in several programs including the National Aquatic Resources Surveys (NARS), National Coastal Condition Assessment (NCCA), and the Great Lakes Fish Monitoring and Surveillance Program (GLFMSP). The USEPA data are a combination of muscle and whole fish samples collected for evaluation of both human and ecosystem health. In Canada, data for the Great Lakes are collected by the Ontario Ministry of the Environment, Conservation and

Parks (MECP). The fish samples are filet samples collected for the establishment of fish consumption advisories for the Canadian waters of the Great Lakes.

Total mercury concentrations in fish muscle (ww) tissue were selected because they made up more than 90% of the available samples. We also selected the five most sampled species (walleye, lake trout, lake whitefish, smallmouth bass, and yellow perch) for further analysis. All mercury concentrations were first length-standardized to the mean length for each species by quantifying the species-specific effect of length on muscle mercury concentrations, before being natural log transformed to improve normality.

We then used a linear mixed effects model to better understand patterns in mercury concentrations in fish in the Great Lakes. The length-standardized and log-transformed total mercury concentrations were modelled as a function of the interaction between Year, Lake, and Species, along with a random intercept for sampling site to account for site-level variation. Mercury concentrations in fish muscle varied significantly across lakes and through time, with multiple significant interactions between year, lake, and species, highlighting the spatial and temporal complexity of mercury trends in the Great Lakes (Figure 25) with several species-lake combinations showing significant positive or negative trends. When comparing the fish tissue concentrations to human health criterion for human exposure, all lakes have fish exceeding the USDA/EPA criterion (Figure 26).

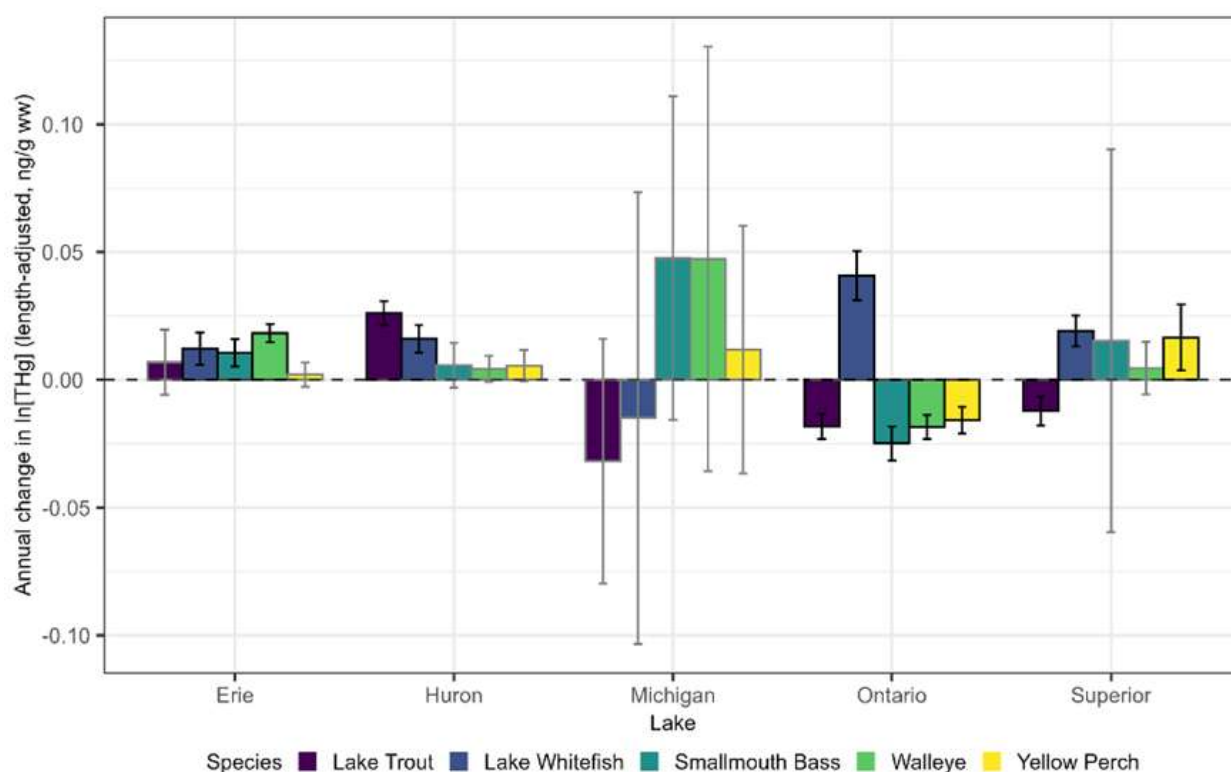


Figure 25. Estimated annual trends in length-adjusted total mercury concentrations ($\ln[\text{THg}]$, ng/g muscle wet weight) in five fish species from the five Great Lakes. Temporal trends were estimated using a linear mixed-effects model with fixed effects for Year, Lake, Species, and their interaction, and including sampling site as a random effect. Bars show model-estimated slopes for each combination of species and lakes with lines for 95% confidence intervals. Black outlines denote statistically significant trends ($p < 0.05$), while gray outlines denote non-significant trends. Only the five most-sampled species were modelled.

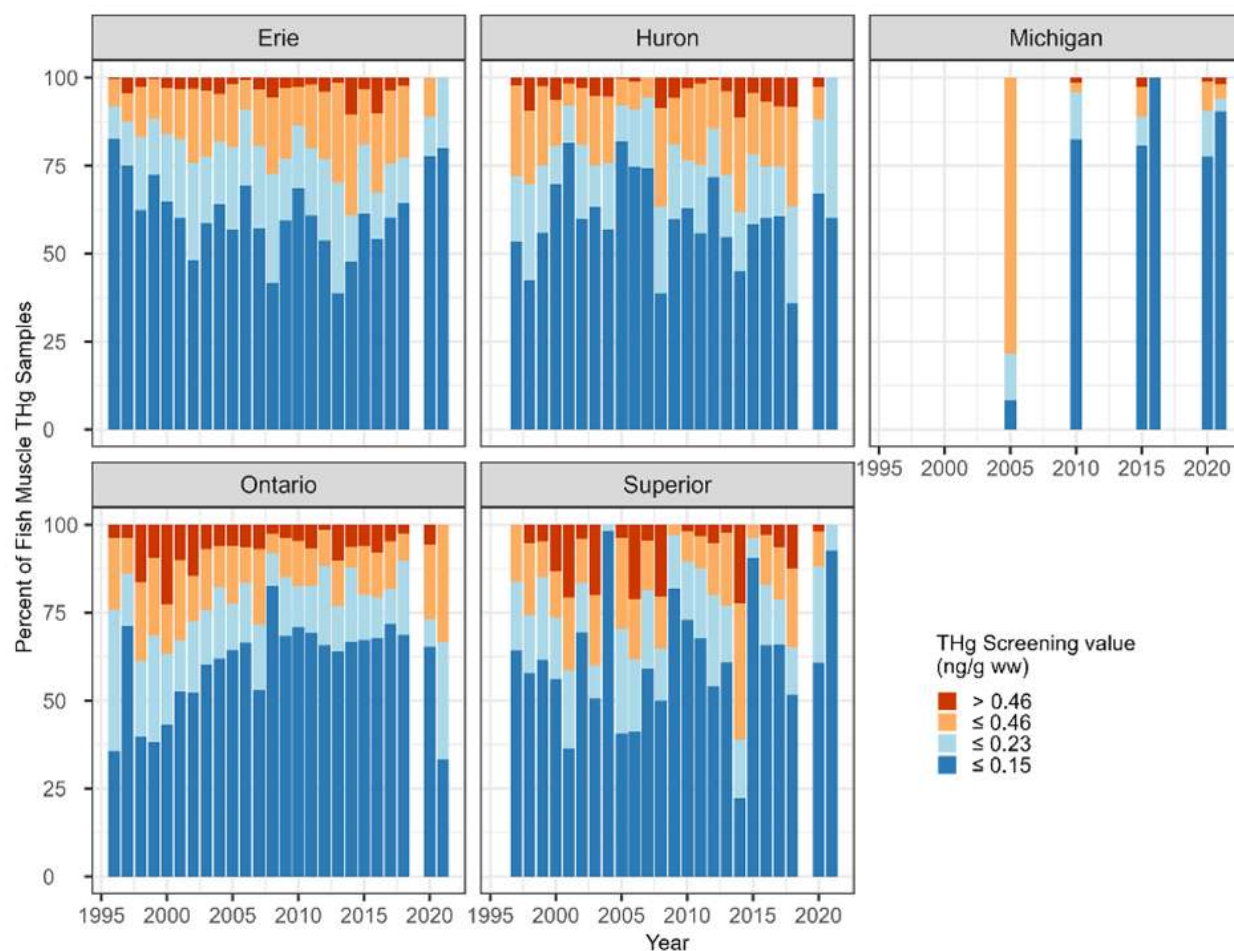


Figure 26: Percent of all fish samples in each of the five Great Lakes with mercury fish tissue concentrations categorized by the USFDA/EPA fish consumption guidelines for sensitive human populations indicating best choices (≤ 0.15 = 3 weekly fish servings), good choices (≤ 0.23 = 2 weekly fish servings or ≤ 0.46 = 1 weekly fish serving), or choices to avoid (> 0.46 = 0 weekly fish servings) (US FDA/EPA 2024).

- Nordic fish mercury

Widespread mercury contamination in the biota of Fennoscandia was recognized already in the 1960s, during the same decade as the cause of Minamata disease was becoming apparent in Japan. The Nordic concerns about mercury led to initiation of mercury monitoring in the environment to determine both the risks and the causes of that contamination. The record of mercury in over 50,000 Fennoscandian fish collected since the mid-1960s is a valuable case study for addressing central questions for first effectiveness assessment of the UNEP Minamata Convention: can reductions in the release of Hg to the environment reduce Hg in the biota. The most recently published analysis of these data Braaten et al. (2019) found that mercury levels in aquatic invertebrates and fish have decreased since the 1970s, consistent with reduced atmospheric concentrations. The rate of response, however, is slow, and mercury levels in many fish higher in the food web remain over levels safe for human consumption and wildlife. Both the challenges of harmonizing the data from different fish monitoring programs, and confounding influences (e.g. changes in land use and climate) contribute to variation in the fish mercury levels over time (Figure 27). The availability of Hg in environmental archives and surface waters, provides important evidence that supports the interpretation of linkages between emissions, different environmental media, and the biota. In summary, reducing emissions is of value, but it requires diligent follow-up, often over decades, to observe the benefits when it comes to reduced atmospheric Hg levels and fish.

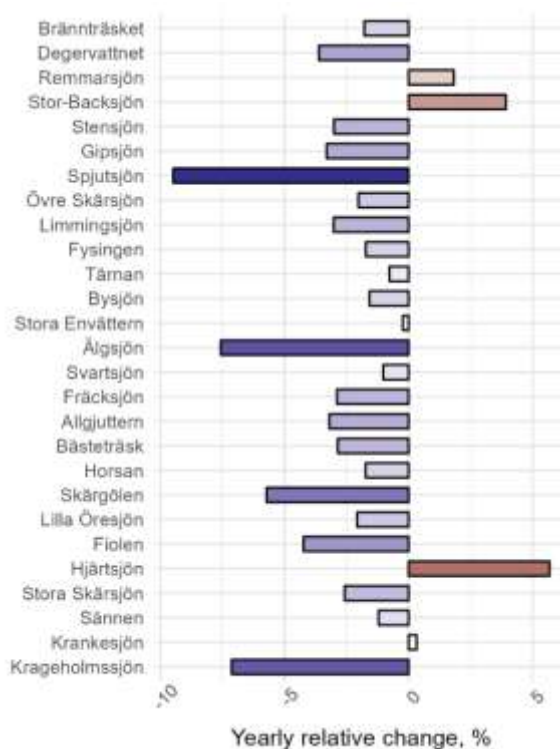
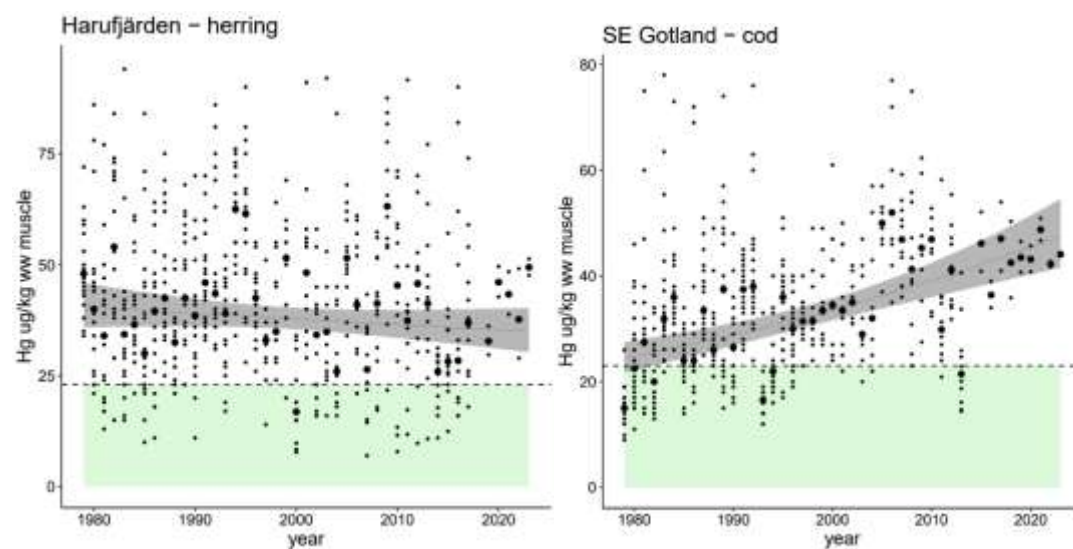


Figure 27: Fitted trend for Swedish perch (*Perca fluviatilis*) Hg concentrations 2013-2022. Source: Faxneld et al 2024



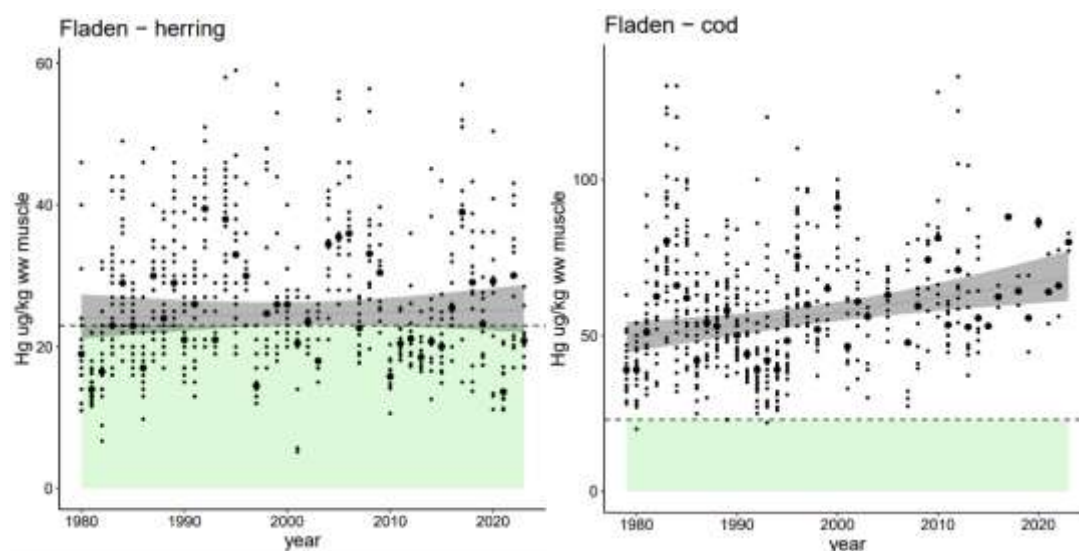


Figure 28. Time-series examples of fish Hg changes from 1980 to 2023 across different Swedish monitoring sites. The dotted horizontal line represents an Environmental Quality Standard (EQS) level of 20 ng/g wet weight. Source: Soerensen et al 2025

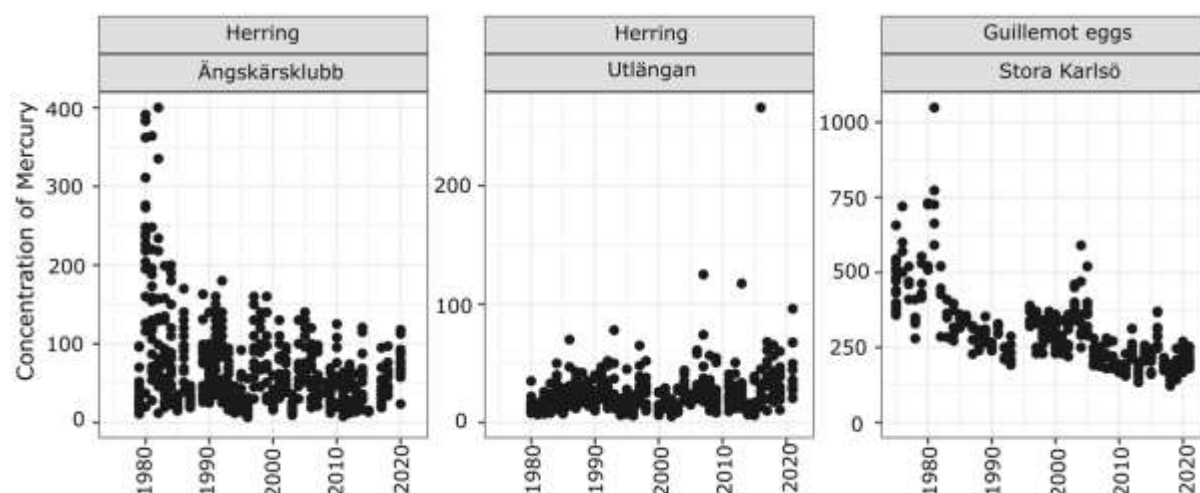


Figure 29: Time-series examples from the MCoM after technical validation for mercury ($\text{ng.g}^{-1} \text{ ww}$) concentration in herring (*Clupea harengus*) from the autumn sampling at the stations: Ängskärsklubb and Utlången and in the guillemot (*Uria aalge*) eggs from Stora Karlsö in Baltic Sea. Source: Ammar et al. 2024.

Mercury trends in key fish species

Analysis of mercury concentrations in freshwater perch from multiple lakes reveals predominantly declining trends over time. Despite this general decrease, very few lakes show Hg levels below the Environmental Quality Standard (EQS) threshold of 20 ng/g wet weight (Figure 28). This persistence of elevated mercury suggests legacy contamination in these ecosystems, coupled with ongoing bioaccumulation within certain catchments.

In contrast, mercury concentrations in Baltic Sea herring have remained mostly stable or have exhibited slight increases (Figure 29). Most herring Hg levels consistently exceed the EQS, indicating the presence of persistent mercury sources or ecological factors driving bioaccumulation, such as dietary shifts or slower mercury elimination rates in marine environments.

Cod populations in the Baltic Sea demonstrate a marked increase in mercury concentrations since the early 1990s, particularly in deeper waters. Mercury levels in cod are well above the EQS and continue to rise, underscoring ongoing contamination pressures and bioaccumulation in this species.

Evaluation of Confidence:

Confidence in the results varies depending on data consistency and coverage. High confidence is attributed to long-term datasets with consistent species, tissue types, and geographic regions. Moderate confidence applies to assessments where metadata or sampling methods showed some variation. However, confidence is limited in areas with sparse data or where inconsistencies exist in tissue types or temporal coverage.

Information and Capacity Needs:

To strengthen mercury monitoring and assessment, expanded sampling efforts are needed in underrepresented regions and ecosystems. Greater inclusion of Indigenous-led monitoring programs and integration of traditional knowledge frameworks are essential for more comprehensive and culturally relevant assessments. Continued refinement of models that incorporate biogeochemical, ecological, and climate variables will improve understanding of mercury dynamics.

2.4.4.3 Birds

Information Used

Available bird datasets in the OESG repository were compiled but limited temporal trends could be conducted. Further temporal analyses of THg concentrations in birds can be pursued for subsequent reports should sufficient time and additional monitoring data become available to support a robust trend assessment. For the present report, we analyzed a single dataset of Common Loon (*Gavia immer*) mercury concentrations. Some additional published studies are described below for temporal trends of THg in birds to give broader geographical and ecological context, though a comprehensive literature review was not conducted.

Methodology

Only studies with data series containing a minimum of five years of data and terminating in 2015 or later were included in discussion of analyzed published temporal trends for bird THg concentrations.

Answer/Response

Common Loons are obligate piscivores that have been identified as a good bioindicator of MeHg bioavailability in temperate lakes and so have been intensively studied for several decades. The Common Loon dataset contained more than 7,000 THg samples beginning in 1988. We selected wet-weight whole blood (>58% of the available Common Loon samples) because it represented the largest sample type. Blood THg concentrations also reflect shorter-term Hg exposure and therefore capture local, lake-specific ecological drivers of exposure, compared to the spatially and temporally integrative profile provided by feathers – an important distinction in a migratory bird species.

Adult Common Loon blood THg concentrations ranged from 10 to 9,750 ng/g ww across 2,990 samples from three countries (Canada, Iceland, United States). We converted male blood THg concentrations to female blood THg concentrations using standard published relationships (Evers et al. 2011), in order to increase the standardized sample size usable in each Lake x Year. Samples from lakes with five or more years of data extending past 2015 were included in the lake-specific temporal trend analyses.

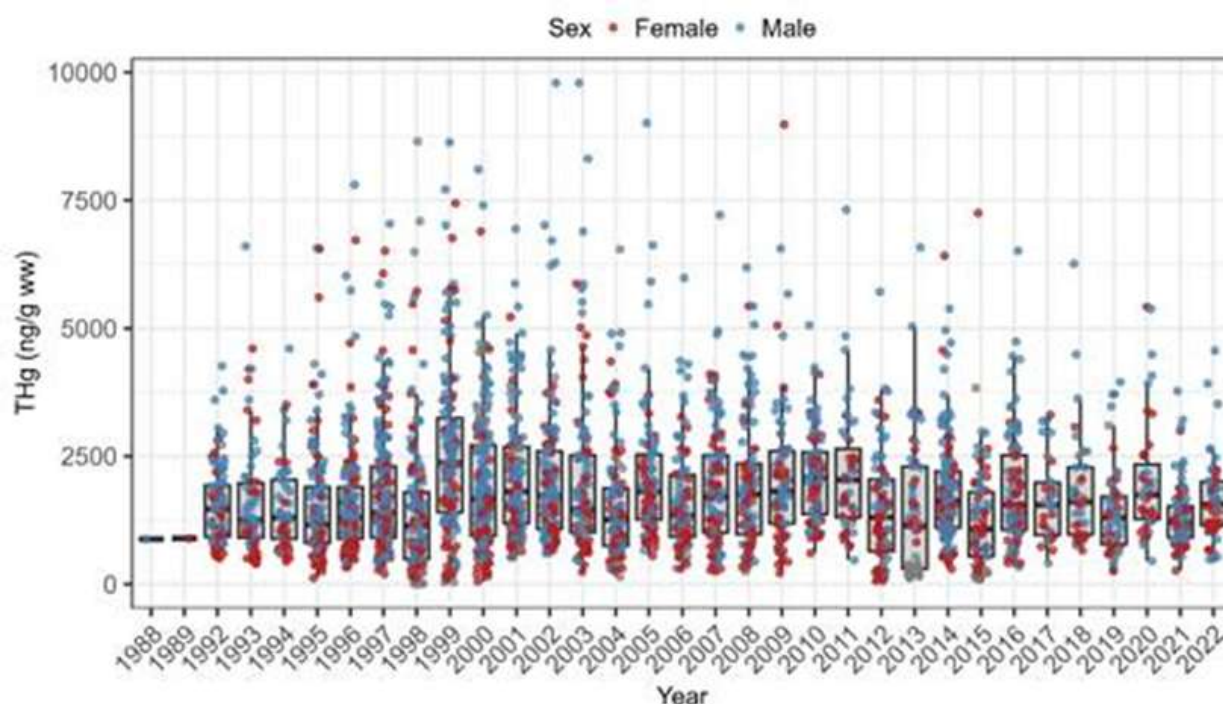


Figure 30. Total mercury concentrations in adult Common Loon (*Gavia immer*) blood by year. Regional analysis ($\ln[\text{THg}] \sim \text{Year} + \text{Sex} + (1 \mid \text{Site})$) showed no significant temporal trend ($p < 0.05$), although male THg concentrations were significantly higher than females ($p < 0.05$).

Variation in temporal and spatial sampling, coupled with annual-, individual-, and site- specific drivers of Hg methylation, bioaccumulation, and biomagnification (e.g., depressed trophic position in some smaller pond systems) confounds interpretation of long-term trends. To better account for this variability, we independently analyzed temporal trends in 32 lakes in North America that had five or more years of dating ending in 2015 or later. In total there were 991 adult blood THg concentrations from the 32 lakes in two IPCC regions (E.North America and W.North America) that met the sampling criteria.

The temporal trends observed in adult Common Loon blood were heterogeneous across both space and time. Most lakes in the dataset exhibited relatively stable concentrations since 1988, with significant directional change observed at only a small number of lakes. In total, four lakes demonstrated significantly decreasing THg trends, while three lakes exhibited significantly increasing THg change ($p < 0.05$). These trends were modelled over the course of multiple decades of declining Hg emissions in the United States (U.S. Environmental Protection Agency's National Emissions Inventory). This heterogeneity may reflect the importance of localized drivers of THg dynamics in adult Common Loon blood rather than uniform regional exposure trends. These findings emphasize the continued need for long-term, consistent monitoring, particularly in lakes with intermittent sampling, to understand mechanisms driving trends and to inform risk assessment and management of mercury in freshwater ecosystems.

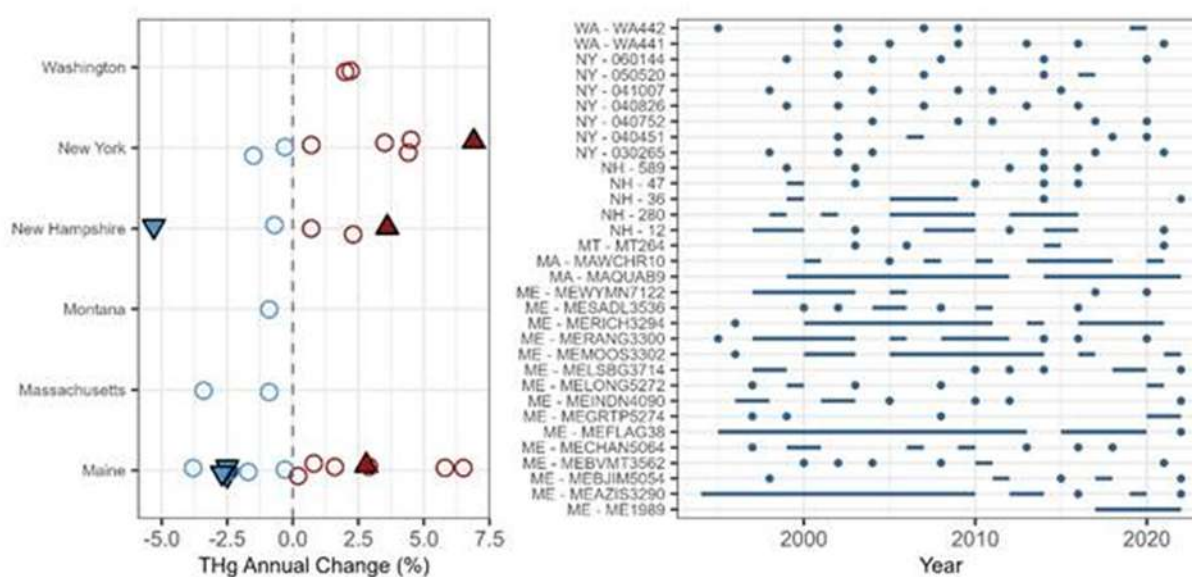


Figure 31. Temporal change since 1988 in total mercury concentrations in Common Loons (*Gavia immer*) from North America are shown on the left panel. Trends were calculated individually for each lake with five or more years of data, including data in 2015 or later. Circle outlines indicate no significant change in mean blood THg ($p \geq 0.05$). Blue downward triangles indicate a significantly decreasing THg ($P < 0.05$) and red upward triangles indicate a significantly increasing THg ($P < 0.05$). Trends are calculated using the HARSAT approach consistent with other trend analyses (HARSAT: <https://harsat.amap.no/>). The right panel visualizes the sampling year effort for each of the lakes meeting the minimum criteria for inclusion in temporal analyses. Lines indicate years of consecutive data collection, while points indicate an isolated sampling year.

Literature review

The most comprehensive regional analysis of temporal trends of THg in birds, currently available, was for the circumpolar Arctic (Morris et al. 2022) (see also section 0.1.4.5). Seabirds were monitored for mercury exposure using eggs, feathers or blood. Increasing, decreasing, and stable mercury concentrations were observed across 9 monitoring sites, which ranged in temporal coverage from 10–40 years. In other geographic areas, non-linear trends were reported for THg in feathers of egrets in the Florida Everglades (Lange et al. 2020) and for THg in eggs of seabirds breeding on the Pacific Coast of Canada (Kestic et al. 2024). Feathers of an owl species in Norway showed no temporal trend of THg over a 30-year period (Devalloir et al. 2023), and similarly, multi-decadal variation of THg in gull eggs from a large freshwater lake in northern Canada showed no trend after controlling for diet variation (Hebert et al. 2021). These published studies highlight the complexity and variable temporal trends of mercury exposure to birds in Europe and North America.

Consistency of Answer with Existing Literature (especially the 2018 GMA)

The heterogeneity in trends in individual lakes and lack of a significant trend at a regional scale are broadly consistent with the literature (e.g., Schoch et al (2020) found Common Loon tissue Hg concentrations increased significantly from 1998–2010 and then stabilized from 2011–2016). The 2018 GMA was not able to analyze temporal trends in Common Loons, but noted a spatial gradient with increasing THg concentrations moving west to east across North America then dropping again in Iceland. This qualitative pattern is largely reflected in the geometric means of this dataset when aggregating by IPCC region after extracting known marine wintering samples in California, Louisiana, and Virginia (E.North-America>W.North-America>C.North-America>N.W.North-America>Greenland/Iceland>Marine). A key distinction is the 2018 GMA aggregated sites in western Canada with the western continental United States while the IPCC region approach aggregates them with Alaska.

Evaluation of Confidence

Confidence in the results presented is low overall due to limitations in data available. We analyzed one species and, despite its intensive sampling over multiple decades, only 32 lakes in the United States met the criteria (≥ 5 years of data ending in 2015 or later) for temporal analyses. In order to increase confidence in temporal trends, robust analyses need to be conducted for multiple species spanning taxonomic groups and with adequate metadata and statistical approach to account for the multitude of confounding factors driving MeHg exposure in birds.

Information and Capacity Needs

A more comprehensive analysis is needed for temporal trends of THg in birds, which accounts for better geographic coverage and the influence of ecological and biological factors. In particular, the spatial and temporal linkages between emissions and releases, atmospheric concentrations, deposition, and the MeHg concentrations in birds is still poorly understood (see case studies in this report highlighting decoupling of patterns in different compartments in other taxa). The identification of these knowledge gaps, further harmonized data collection with associated standardized metadata, and capacity to implement robust statistical frameworks are critical for understanding spatial and temporal trends in biota across ecoregions and taxa while accounting for the many other drivers impacting Hg concentrations trends (e.g., global climate change, invasive species, land use conversion, resource exploitation, biophysiology).

2.4.4.4 Marine Mammals

Information Used (including applicable indigenous and traditional knowledge):

More than 60 time series with over five years of data collection for different tissues were submitted for the report (Figure 30). A majority of those trends ended before 2015 (24 included to study the changes from 2000 until 2015 and after). Half of the datasets submitted were from polar regions. Monitoring programs in certain northern countries such as the Kingdom of Denmark, Norway, and Canada have allowed the continuous measurements of mercury concentrations over decades in marine mammals. In the Arctic Monitoring and Assessment Programme 2021 Mercury Assessment (AMAP 2021a), most of the Arctic marine mammal species investigated, except for beluga, had at least some time series with significantly increasing trends of Hg in liver over the last 20 years (50 time series analyzed with a minimum of 6 years of data post-2015; AMAP 2021a). Mercury time series for Arctic marine mammals were found to be statistically powerful for muscle (AMAP 2021a). The Arctic data and the beluga time trend are described in more detail in the case study section below. In particular the beluga case study shows how Indigenous co-led monitoring can lead to more comprehensive and culturally relevant assessments to explain the observed mercury trends.

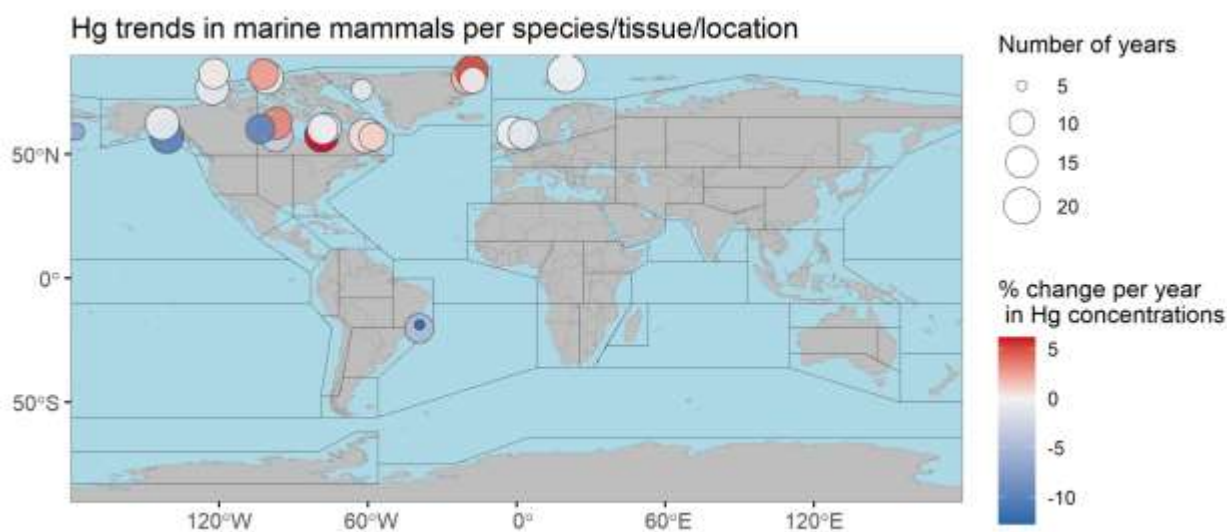


Figure 32. Temporal changes in total mercury concentrations in marine mammals since 2000. Each dot represents the rate of change for a given location, species and tissue with 30 observations or more. Only data series with a minimum of five years of data and terminating in 2015 or later are included. Data is aggregated by a 20° × 20° grid cell. Red indicates increasing trends through time, blue decreasing trends.

Methodology

Only data series with a minimum of five years of data and terminating in 2015 or later were included in analysis.

Answer/response

Results indicated that temporal trends varied by species and locations. Long-time data series from southern regions (East Asia/Pacific; Europe/Central Asia; Latin America) were limited in numbers and time scale (Figure 29). Analyses of most recent time trends since 2000 and reaching 2015 or later (minimum five years of data) indicated slightly increasing concentrations in several populations from the Arctic, while concentrations in some populations from lower/sub Arctic and South America were decreasing.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

There was insufficient time and resources to undertake a literature review and comparisons with the 2018 GMA to determine consistency with earlier research.

Evaluation of Confidence:

Confidence in the results varies depending on data consistency and coverage. High confidence is attributed to long-term datasets with consistent species, tissue types, and geographic regions. Moderate confidence applies to assessments where metadata or sampling methods showed some variation. However, confidence is limited in areas with sparse data or where inconsistencies exist in tissue types or temporal coverage.

Information and Capacity Needs:

To strengthen mercury monitoring and assessment, expanded sampling efforts are needed in underrepresented regions and ecosystems over extended periods of time. As illustrated with the beluga case study, greater inclusion of Indigenous-led monitoring programs and utilization of Indigenous Knowledge frameworks can be essential for

more comprehensive and culturally relevant assessments of mercury trends. Continued refinement of models that incorporate biogeochemical, ecological, and climate variables will improve understanding of mercury dynamics.

2.4.4.5 Integrated biota case studies (several taxa)

❖ Arctic Case Study: Time trends of mercury bioaccumulation in the circumpolar Arctic

The Arctic Monitoring and Assessment Programme (AMAP), an international working group of the Arctic Council, performs assessments of contaminants in the Arctic, including for mercury. A core component of AMAP assessments is the synthesis of national monitoring datasets to evaluate temporal trends of mercury bioaccumulation in animals across the circumpolar Arctic (Figure 31). This effort is one of the most comprehensive regional evaluations of change to levels of mercury in animals, given the geographic scope, diversity of bioindicator species, and lengths of the time series. The most recent analysis was completed in 2021, focusing on 77 time series of mercury concentrations of Arctic biota including mussels, marine and freshwater fish, seabirds, and marine and terrestrial mammals (Morris et al. 2022). Together, the compilation of time series covered the period of 1968-2018, and individual datasets contained on average 17 years of measurements with at least one of the years between 2013-2018. Most time series (60%) showed no change in biotic mercury concentration over the past 20 years (1999-2018), while 22% showed a statistically significant increase and 18% showed a statistically significant decrease. There was no pattern in the types of Arctic biota or geographic regions that showed these temporal trends. These divergent results make it challenging to draw an overall conclusion about changes in mercury exposure to Arctic animals and food webs over the last two decades. Further, the findings highlight that combined effects of long-range transport of mercury, local environmental conditions, and a broad range of processes have complex influences on mercury burdens. The AMAP assessment underscores the importance of carefully designing temporal trend monitoring to account for the influence of transport, biogeochemical, ecological and biological processes on mercury accumulation in animals.

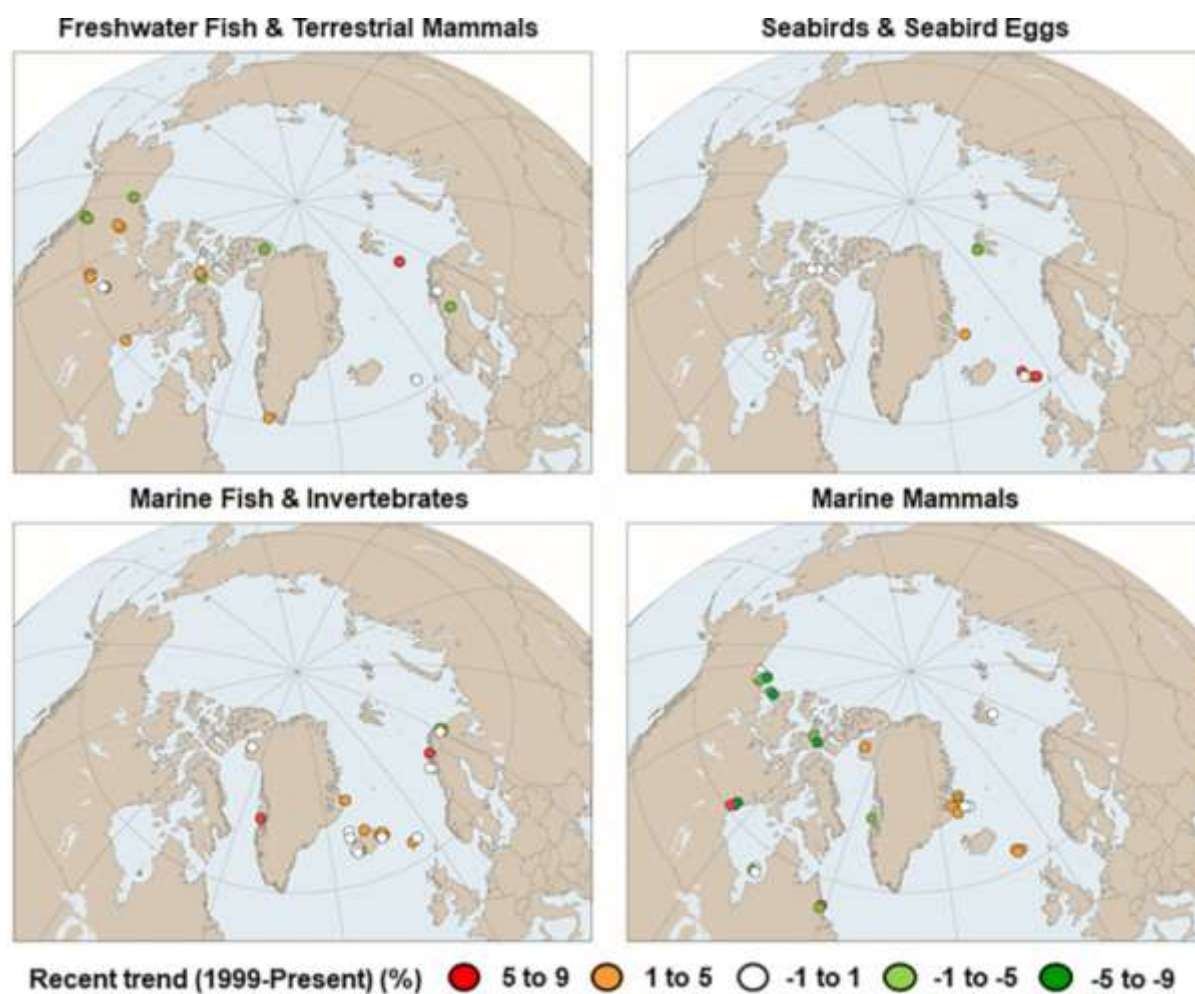


Figure 33. Geographical patterns in recent (last 20 years) trends of total mercury concentrations in tissues of biota, where trends were significant and/or time series were of a statistical power $\geq 80\%$. The figures show freshwater fish and terrestrial mammals (top left); seabirds (top right); marine invertebrates (mussels) and marine fish (bottom left) and marine mammals (bottom right). Overlapping symbols have been dispersed to make them visible, resulting in clusters of points around some monitoring locations where trends are available for multiple time series (source: Morris et al. 2022).

❖ Beluga Monitoring Case Study: Community participatory approaches

As part of Inuvialuit and their ancestors' stewardship of beluga whales, a resource they depend on, they have maintained the Beluga Harvest Monitoring Program since the initiation in 1973. The program formalized in 1980 and later expanded in 2001 to include annual sampling for contaminant monitoring (Harwood and Smith 2002; Loseto et al., 2015) and has since grown to include a range of health-related analyses (e.g. Noel et al., 2018; Nielsen et al., 2018; Ostertag and Chan 2018; Choy et al., 2020; Figure 34). The program is directed by Inuvialuit partners to ensure that it responds to community and regional concerns. Through the practice of meaningful engagement and collaborative research that respects cultural and societal expectations, the monitoring program has provided more than 40 years of mercury data for the eastern Beaufort Sea beluga whale population.

Beluga whale harvesting is a central feature of Inuvialuit culture, dictating summer gathering and activity, providing food for the whole year (Nuligak, 1996; Alunik et al., 2003; Worden, 2018). Most of the whale hunting occurs near the mouth of the Mackenzie River (Day 2002; Usher, 2002). Inuvialuit have in-depth knowledge about beluga, qilalugaq, and the surrounding ecosystem, that they have acquired through first-hand experience and intergenerational knowledge transfer that extends many generations, referred to as Inuvialuit Knowledge (IK; ICC et al., 2006). The rich IK provides direction on beluga research and monitoring and the opportunity to co-design

research informed by two knowledge systems - Inuvialuit and western science. The successful co-design of beluga research and monitoring is enabled by a solid foundation of strong relationships between researchers and communities. These relationships have been developed over decades, with consistent communication throughout the research process. The relationship respects and acknowledges the capacity and realities of communities and partner organizations. This process of respectful engagement and collaboration has resulted in an extensive dataset of beluga samples and research programs that are inclusive of IK, that also drives research and monitoring directions (e.g. MacMillan et al., 2023; Mayette et al., 2022; Noel et al., 2022; Ostertag et al., 2018; Ovitz et al., 2023; Figure 34).

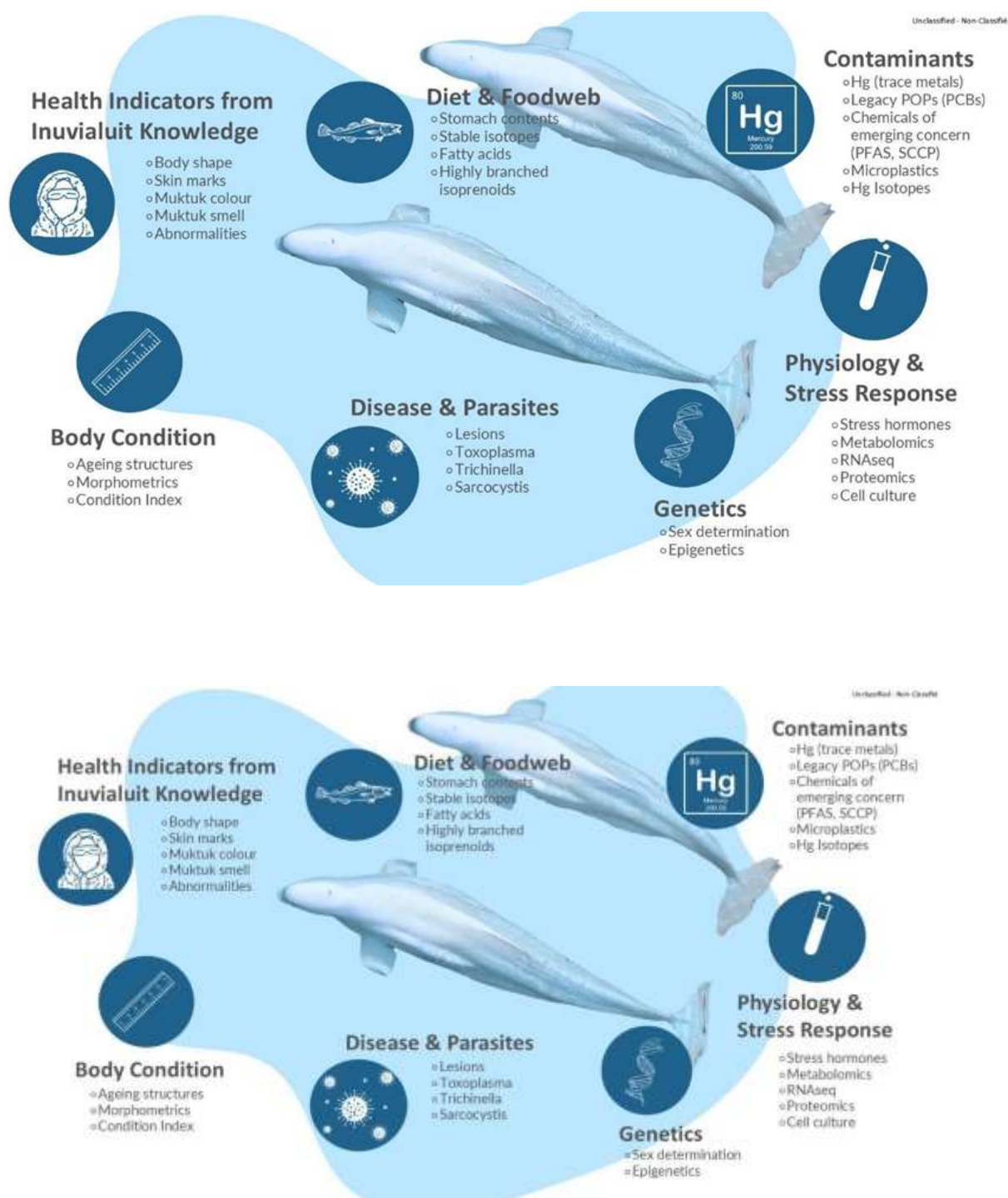


Figure 34. Schematic of the measurements, data and samples collected on harvested beluga whales through a partnership between Inuvialuit and researchers.

Mercury levels in the beluga whales harvested by Inuvialuit were found to be increasing from 1981 to 2002 and were shown to be higher than beluga whales harvested in other Canadian Inuit communities (Lockhart et al., 2005, Figure 33). Since this time, mercury levels have declined and a closer evaluation of size, age and foraging habits were identified as key interpretive factors (Loseto et al., 2015; Morris et al., 2022; Noel et al., in press. Figure 35). Questions remain about the driver of high mercury levels measured between 1996 to 2002, and were recently evaluated using environmental forcing models with parameters informed by IK (ICC et al., 2006). Findings pointed to ecosystem productivity as a key driver of beluga mercury trends (Gilles et al., 2024).

Regional differences in beluga mercury levels across the Canadian Arctic were recently explored through considering local contexts, particularly around harvester preferences in two communities, Arviat (Nunavut) and Tuktoyaktuk (Northwest Territories) (Sudlovenick et al., 2024). Community interviews together with historical data highlighted context and the marked preferences, with Tuktoyaktuk hunters harvesting the largest belugas and hunters from Arviat harvesting across an even spread of belugas. The re-interpretation across common size and age reduced the regional disparity and highlighted the nuanced complexity of the beluga harvest and context of the individual animal harvest preferences. Findings remind us to consider where the samples come from and how these contexts can be applied to data interpretation (Sudlovenick et al., 2025).

The partnerships and emphasis on dialogue between scientists and Inuvialuit knowledge-holders to guide research has broadened the beluga monitoring program to respond to critical questions about the significance of mercury in the Arctic food web for Inuvialuit harvesters. Harvesting, preparing and eating country foods, including beluga dry meat and maktaaq (the outer skin and blubber layers), continue to support physical and mental health in the ISR.

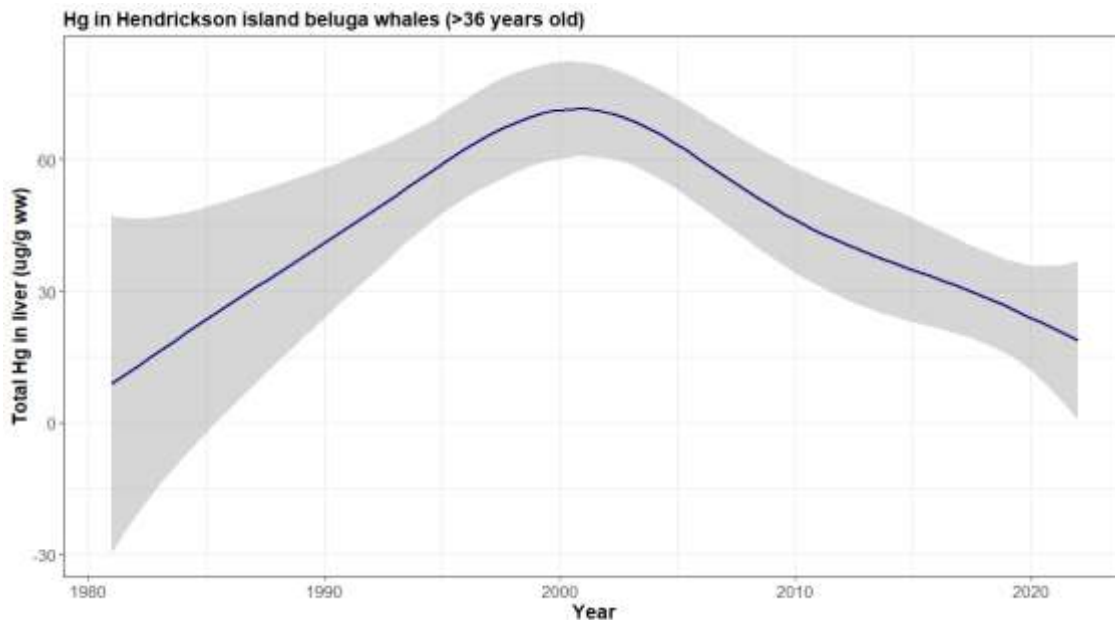


Figure 35. Temporal trend of mercury in Eastern Beaufort Sea beluga harvested in the Inuvialuit Settlement Region from 1981 to 2023

Monitoring contaminants in beluga whales comes with the responsibility to respond to questions about the safety of eating beluga and other country foods, and providing background on legacy contaminants and heavy metals in the Arctic ecosystem. In 2016, Inuvialuit participants at the ISR's Beluga Summit emphasized the need for research to respond to questions about beluga health from a human perspective, including how the preparation of beluga affects contaminant and nutrient concentrations (Murray et al., 2024). The Country Foods for Good Health

project was initiated to respond to these questions and expanded to country foods in general, and used a multi-disciplinary approach to answer questions through western science, youth perspectives and Elders' IK, and improve communication of dietary health messages (Gyapay et al., 2022; Gyapay et al., 2025; Ramirez Prieto et al., 2025)

The leadership of Inuvialuit has led to the success of the beluga monitoring program. Beluga whales are a critical source of food and connection to culture in the ISR and other Inuit regions. Inuvialuit are at the table with scientists developing the research questions and guiding the direction of research, including defining key and meaningful research questions, ensuring the study designs are fail proof, enhancing the programs with inclusion of knowledge holders and including future leaders within the research programs. These strong partnerships and respectful working relationships have generated what is likely the most reliable long-term trend on mercury in toothed whales that there is globally, providing crucial information that can be used regionally, nationally, and globally to inform policies, explain impacts of climate change, ecosystem health and provide critical information to Inuvialuit harvesters on the health risks and benefits of consuming beluga whale products.

❖ Integrated Analysis of Mercury in the Derwent Estuary, Australia

Description of the case study

The Derwent Estuary covers an area of about 200 km² and is an integral part of Tasmania's natural, cultural and economic heritage in Australia (Macleod & Coughanowr, 2019). It supports diverse urban, industrial, and recreational uses (Macleod & Coughanowr, 2019). Historically, the estuary has been impacted by the operations of a large zinc smelter (in operation since 1917) and a paper mill (established in 1941). Although modern environmental management practices have significantly improved, the legacy of past industrial activity continues to affect the estuary and is likely to persist for decades (Figure 34).

Several environmental issues affect the Derwent Estuary, including contamination by metals, loss of habitats and species, introduced pests, weeds, altered river flow regimes, blocked migratory pathways for fish and eels, elevated nutrients, algal growth and hypoxia in localised areas (Macleod & Coughanowr, 2019).

The Derwent Estuary Program, initiated in 1999, plays a central role in advancing the understanding and management of these issues. Cooperative monitoring arrangements with the State Government, industries, local governments and the scientific community have generated a wealth of information on water and sediment quality, seafood safety and estuarine habitats and species (DEP, 2020, 2021). Metal levels in the Derwent sediments, water column, fish and some invertebrates have been monitored for several decades (Bloom & Ayling, 1977; Macleod & Coughanowr, 2019; Ratkowsky et al., 1975; Whitehead, 2013) (Figures 34 a and b). Additional data on mercury emissions and releases is available through both the Australia's National Pollutant Inventory and industry reports (NPI, 2025; Nyrstar, 2025). These monitoring datasets represent one of the rare long-term environmental data series from the Southern Hemisphere, a region often underrepresented in global data collection efforts.

The Derwent Estuary contains some of the highest recorded concentrations of mercury, zinc, lead, and cadmium in sediments compared to other estuaries worldwide (Macleod et al 2019, Jones 2013). While the primary source of these metals is historical industrial activity, ongoing contributions from fugitive dust, diffuse groundwater inflows, and point-source discharges from current zinc smelting operations may continue to influence both bound and soluble metal levels in the estuary (Nyrstar 2025).

Mercury observations in invertebrates

To monitor bioavailability to organisms, oysters have been deployed annually to surface waters throughout the estuary and then retrieved to measure the concentration of accumulated metals including mercury (Nyrstar, 2025). Wild oysters and mussels were also collected on a triennial basis to determine possible human health risks of exposure to metals through collection of wild shellfish (Nyrstar, 2025). Oysters and mussels are exposed to both particulate-bound and dissolved mercury through filter feeding and direct absorption from the water column. As effective bioindicators, they exhibited changes in mercury concentrations influenced by anthropogenic inputs and environmental factors such as sediment resuspension and water quality.

Deployed oysters at nine impacted sites in the middle estuary accumulated higher concentrations of mercury than those deployed to the control site (Mickey's Bay) and showed the highest loads from sites nearest to the zinc smelter (Nyrstar Wharf) (DEP, 2020, 2021) between 2014 and 2024 (Figure 35a).

Wild oysters and mussels collected in the middle (near the zinc smelter) and lower zones of the Derwent Estuary showed higher concentrations of mercury than those from sites furthest from zinc smelting activities (the D'Entrecasteaux Channel and Mickey's Bay) (Figures 35b and c). Within the estuary, the highest concentrations were typically recorded in the middle estuary and Ralphs Bay, whilst the western shore, exhibited the lowest levels. These spatial patterns likely reflect the influence of estuarine circulation, ambient water metal concentrations, proximity to contaminant sources, estuarine processes and longer water and sediment retention times at Ralphs Bay. Sites in the middle estuary are closest to the zinc smelter, whereas marine-influenced regions, such as the western shore, are generally exposed to lower concentrations of metals in the water column.

In 2025, the Derwent Estuary Program scientists assessed mercury temporal trends in deployed oysters using the Harmonised Regional Seas Assessment Tool (HARSAT) being developed as a collaboration between AMAP (Arctic Monitoring and Assessment Programme), HELCOM (Baltic Sea Regional Seas Convention) and OSPAR (NE Atlantic Regional Seas Convention). Between 2004 and 2024, four sampling sites, including the control site, showed no statistically significant trends in mercury concentrations in deployed oysters. In contrast, six impacted sites within the estuary exhibited significant linear or nonlinear trends. At three of these sites, mercury concentrations increased annually by 1.6% to 3.4% over the 2004-2024 period (Table 8a).

Mercury observations in fish

Environmental organisms at higher trophic levels are primarily exposed to organic mercury through dietary sources. Sand flathead (*Platycephalus bassensis*) being selected as a bioindicator of metal contamination because of their high site fidelity, abundance and regular consumption by humans (Jones et al., 2013; Ratkowsky et al., 1975). Sand flathead is a benthic carnivore, most likely exposed to mercury through dietary intake of epibenthic fauna such as crabs, which in turn feed at the sediment surface where methyl mercury can be produced (Jones, 2013). Sand flathead also shows high site fidelity and a relatively small home range (~5-10 km) with limited migration downstream during the spring/summer, presumably for spawning purposes (Tracey et al., 2020).

The control region (Mickey's Bay) mercury concentrations in fish ($0.20 \pm 0.02 \text{ mg kg}^{-1}$) were significantly lower than mercury concentrations in fish from within the Derwent Estuary ($0.49 \pm 0.01 \text{ mg kg}^{-1}$) between 1975 and 2011 (Jones et al., 2013).

In 2025, the Derwent Estuary Program scientists plotted current mercury concentrations in fish (Figure 35d). Mercury concentrations in fish from the control site (Mickey's Bay) were lower than those measured in fish collected from within the Derwent Estuary during 2014–2024. The observed spatial variability in mercury concentrations across impacted sites is likely influenced by differences in fish growth rates (Jones, 2013) and potentially greater mercury bioavailability and longer water and sediment retention times in Ralphs Bay (Ralphs Bay Spit and Richardson Beach) (Jones et al., 2014).

Assessing the temporal change of Hg bioaccumulation in fish is complicated by shifts in habitat, fish length, growth rates, prey preferences and the potential for seasonal movements or migration (Jones, 2013). Jones et al 2013 used linear and non-linear models to examine spatial and temporal variations between regions. The authors concluded that calculating length-standardized mercury concentrations are important to accurately evaluate spatial and temporal trends. Both linear (data from 1974-2011) and non-linear models (data from 1991-2011) that take into consideration the fish biometrics, showed no significant change in mercury concentrations over sampling time despite reductions of mercury inputs (Jones et al., 2013).

In 2025, the Derwent Estuary Program scientists used the Harmonised Regional Seas Assessment Tool (HARSAT) to assess long-term trends in mercury concentrations in fish across sampling sites. No statistically significant trends were detected across all sites between 2002 and 2024, consistent with the findings of Jones et al. (2013) (Table 8b). In the current study, length standardisation was not applied due to lack of log-linear relationship between mercury concentration and fork length. However, applying the length-standardisation method described by Jones et al. (2013) may be appropriate for adjusting mercury concentrations for fish size in future trend analyses.

Mercury observation in birds

Methylmercury is absorbed largely through the diet, and birds which feed on fish and other aquatic organisms are most at risk of exposure (Einoder et al., 2018). Biomagnification of mercury has been observed in the piscivorous birds residing in the Derwent Estuary, with concentrations reaching levels considered toxic (Einoder et al., 2018). For example, feathers collected near the refinery showed mercury concentrations of 24.8 ± 4.8 mg Hg/kg dry weight in White-bellied Sea-Eagles and 19.3 ± 6.9 mg Hg/kg dry weight in Black-faced Cormorants (Einoder et al., 2018).

Human consumption advice

Understanding mercury concentrations in fish and invertebrates remain crucial as a mechanism for monitoring and in protecting humans who consume seafood. Observed mercury concentrations in deployed oysters and wild oysters and mussels are significantly lower than Food Standards Australia and New Zealand national guidelines for mercury (Maximum Permitted Limit; 0.5 mg kg^{-1}) over last two decades (DEP, 2020, 2021; Nyrstar, 2025).

Of all the legal sized flathead sampled from the Derwent in the 2014-2022 period ($n=325$), 68.5% had a mercury concentration exceeding Food Standards Australia and New Zealand national guidelines for mercury (Maximum Permitted Limit; 0.5 mg kg^{-1})(DEP, 2020, 2021).

Key Findings

The Derwent Estuary case highlights complex and persistent mercury contamination issues. Modern control measures have substantially reduced mercury emissions from industrial activities over recent decades. However, elevated mercury concentrations in fish and invertebrates at impacted sites, compared to control sites, indicate that historical contamination remains a major source of exposure.

Metal concentrations in all biota were consistently lower at the control site (Mickey's Bay) compared to impacted sites within the Derwent Estuary. Mercury spatial patterns in biota likely reflect the influence of proximity to contaminant sources, estuarine circulation, and estuarine processes. For example, elevated mercury concentrations in biota from Ralphs Bay, despite its greater distance from the zinc smelter, suggest additional environmental factors influencing mercury bioavailability and bioaccumulation in this region. Ralphs Bay has been previously characterised by longer water and sediment retention times and greater mercury bioavailability.

There were some differences in temporal trends between deployed oysters and resident benthic fish, likely reflecting distinct exposure pathways, bioaccumulation mechanisms and fish mobility. Despite reductions in total anthropogenic mercury inputs over the past two decades, no significant temporal decline in mercury concentrations in fish has been observed between 2002 and 2024. In comparison, mercury concentrations in deployed oysters showed either no significant changes or increased annually by 1.6% to 3.4% over 2004-2024. Although these concentrations in deployed oysters remain well below thresholds of concern for human health, further investigation and predictive modelling are required to better understand the processes driving these observed changes. An assessment of the mercury discharge from industrial activities, environmental and biogeochemical drivers, such as climatic conditions (wind and rainfall), sediment geochemistry, co-contaminants, mercury methylation, and food-web dynamics is suggested as all these processes and inputs may influence mercury bioavailability in biota.

Overall, the results suggest that significant reductions in anthropogenic mercury inputs achieved to date have not translated into corresponding decreases in mercury concentrations in fish and oysters. This highlights the complexity of mercury cycling processes and interactions within the Derwent Estuary, as well as the need for long-term monitoring and targeted management strategies.

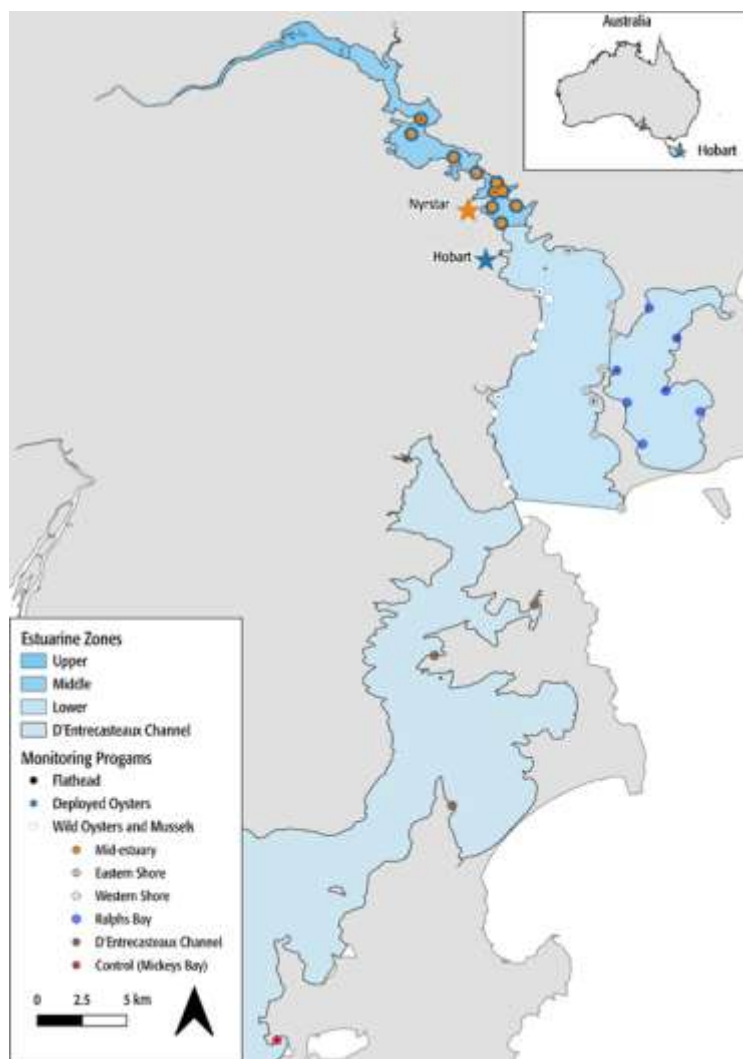


Figure 36a. Derwent Estuary Program fish (flathead), deployed oysters, wild oysters and mussels sampling locations.



Figure 36b. Derwent Estuary Program fish (flathead) and deployed oysters sampling locations.

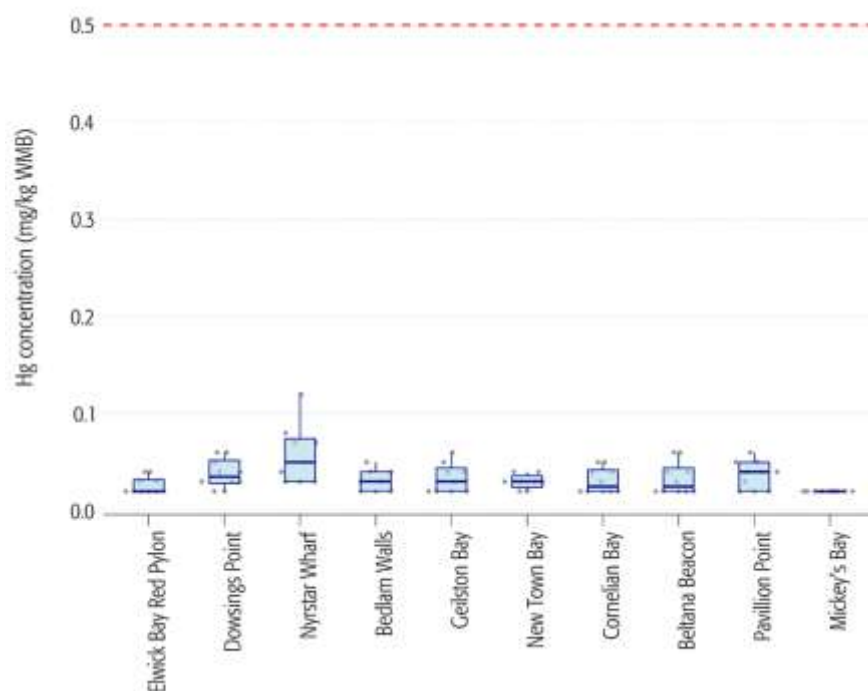


Figure 37a. Total mercury concentrations in surface-deployed Pacific Oysters' soft tissues (2014-2024). Box plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at nine impacted sampling sites in the middle estuary and a control site in Mickey's Bay. Grey dots show the raw data. Horizontal red line represents Food Standards Australia and New Zealand national guideline for mercury (Maximum Permitted Limit; 0.5 mg kg⁻¹). Mercury concentrations are expressed in mg/kg wet matter basis (WMB). Outliers were removed using the interquartile range (IQR) method. Values that were more than 1.5 times the IQR below the first quartile or above the third quartile were excluded.

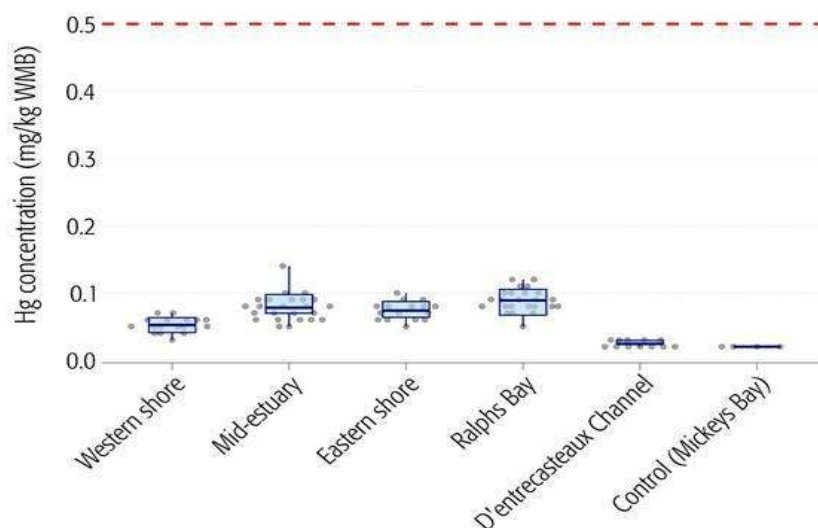


Figure 37b. Total mercury concentrations in wild mussels' soft tissues (2014-2024). Box plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at five regions and a control site in Mickey's Bay. Grey dots show the raw data. Horizontal red line represents Food Standards Australia and New Zealand national guideline for mercury (Maximum Permitted Limit; 0.5 mg kg⁻¹). Mercury concentrations are expressed

in mg/kg wet matter basis (WMB). Outliers were removed using the interquartile range (IQR) method. Values that were more than 1.5 times the IQR below the first quartile or above the third quartile were excluded

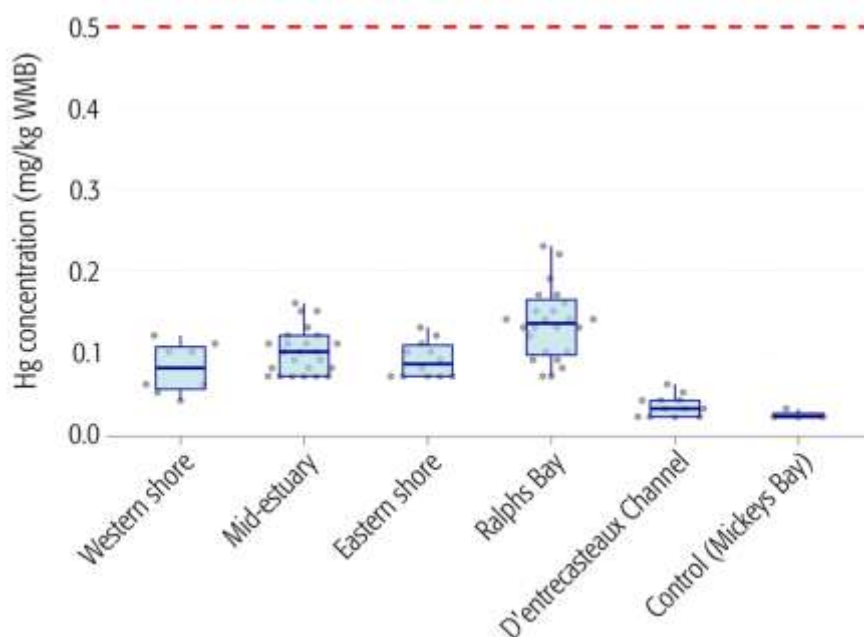


Figure 37c. Total mercury concentrations in wild oysters' soft tissues (2014-2024). Box plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at five regions and a control site in Mickey's Bay. Grey dots show the raw data. Horizontal red line represents Food Standards Australia and New Zealand national guideline for mercury (Maximum Permitted Limit; 0.5 mg kg⁻¹). Mercury concentrations are expressed in mg/kg wet matter basis (WMB). Outliers were removed using the interquartile range (IQR) method. Values that were more than 1.5 times the IQR below the first quartile or above the third quartile were excluded.

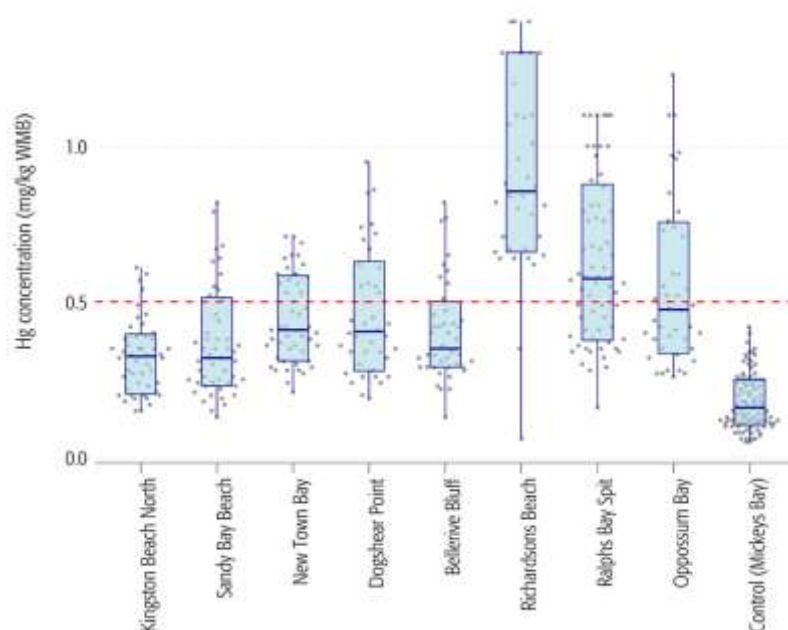


Figure 37d. Total mercury concentrations in fish (flathead) muscle tissues (2014-2024). Box plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at eight impacted sampling sites and a control site in Mickey's Bay. Grey dots show the raw data. Horizontal red line represents Food Standards

Australia and New Zealand national guideline for mercury (Maximum Permitted Limit; 0.5 mg kg⁻¹). Mercury concentrations are expressed in mg/kg wet matter basis (WMB). Outliers were removed using the interquartile range (IQR) method. Values that were more than 1.5 times the IQR below the first quartile or above the third quartile were excluded. 2018 data points were removed because the skin was left on and data was erroneous. Mercury concentrations were not length-standardised.

The Derwent Estuary Program scientists analysed temporal trends of mercury in biota by using Harmonised Regional Trend Analysis Tool - HARSAT tool. Statistical time-trend analysis was performed in R version 4.5.1 using the HARSAT package version 1.0.3 (<https://harsat.amap.no/>). The concentrations were log transformed and changes in the log concentrations over time were modelled using linear mixed models. When there are more than 15 years of data, both a linear model and a smoother on 2,3 and 4 degrees of freedom (df) were fitted to the data. Model selection was based on the lowest Akaike Information Criterion (AICc). The primary metric used to summarise changes over time was the change in total Hg concentrations from 2002(4) to 2024, with significance assessed at the 5% level. The significance of each fitted trend—whether linear or non-linear—was assessed using a likelihood ratio test. Further methodological details are available in Morris et al. 2022.

Table 8a. Trend analysis of total mercury in surface-deployed oysters across nine impacted sites in the Derwent Estuary and one control site (Mickey's Bay) from 2004 to 2024, using the HARSAT tool. To account for variability in control oyster sources, baseline correction was applied by subtracting control oyster values from retrieved oyster measurements. This approach ensures that observed changes reflect environmental influences rather than differences in source oyster conditions. Where this correction resulted in values below the limit of reporting (LOR), data were adjusted to equal the LOR. The figures show the fitted trend along with pointwise 90% confidence bands (grey shaded areas).

Sampling Station	Trend	p-value	Concentrations change over 2004-2024
Bedlam Walls	No significant temporal trend	0.298	Concentrations at the end of the time series were not significantly different from those at the start of the time series.
Beltana Beacon	No significant temporal trend	0.783	Concentrations at the end of the time series were not significantly different from those at the start of the time series.
Cornelian Bay	Significant nonlinear trend	0.005	Concentrations at the end of the time series were not significantly different from those at the start of the time series.
Elwick Bay	Significant log-linear trend in the time series	0.045	Concentrations have increased by an estimated 1.6% per year over the course of the time series.
Mickey's Bay (control)	No significant temporal trend	0.691	Concentrations at the end of the time series were not significantly different from those at the start of the time series.
New Town Bay	Significant nonlinear trend	0.011	Concentrations at the end of the time series were significantly higher than those at the start of the time series.
Nyrstar Wharf	Significant nonlinear trend	0.018	Concentrations at the end of the time series were not significantly different from those at the start of the time series.
Pavillion Point	Significant log-linear trend	0.007	Concentrations have increased by an estimated 3.4% per year over the course of the time series.
Dowsing Point	Significant log-linear trend	0.032	Concentrations have increased by an estimated 3% per year over the course of the time series.

Geilston Bay	No significant temporal trend	0.950	Concentrations at the end of the time series were not significantly different from those at the start of the time series.

Table 8b. Trend analysis of total mercury in wild caught fish (flathead) across six impacted sites in the Derwent Estuary and one control site (Mickey’s Bay) from 2002 to 2024, using the HARSAT tool. 2018 data points were removed because the skin was left on and data was erroneous.

Sampling Station	Trend	p-value	Concentrations change over 2002-2024
New Town Bay	No significant trend	0.930	Concentrations at the end of the time series were not significantly different from those at the start of the time series.
Bellerive Bay	No significant trend	0.822	
Sandy Bay Beach	No significant trend	0.673	
Ralphs Bay Spit	No significant trend	0.214	
Oppossum Bay	No significant trend	0.518	
Kingston Beach North	No significant trend	0.076	
Mickey’s Bay (control)	No significant trend	0.326	

❖ Flin Flon Case Study

The Cu-Zn smelter at Flin Flon was a major mercury and heavy metal emitter. Hg emissions began declining in the late 1980s and ceased when the Cu smelter closed in 2010 because of its inability to meet target SO₂ (and Hg) emission reductions. Within 10 km of the smelter, the landscape and lake sediments remain highly contaminated with mercury (and other metals) deposited over the more than 90 years of smelter operation and continue to enter lake ecosystems through re-emission and runoff. Mercury concentrations in northern pike in lakes within 10 km of the smelter were extremely low in 1982 due either to selenium interference with uptake or metal toxicity interfering with mercury methylation. Mercury concentrations have increased in recent years as mercury, other metals and SO₂ emissions have declined. Highest mercury concentrations occur in northern pike 70 km to the north on the Boreal Shield due to a combination of fish age, growth rates and unknown factors. Trend monitoring would better elucidate the response of Flin Flon lakes to the major reductions in Hg, SO₂, and toxic metal emissions.

❖ Japanese Monitoring Case Study

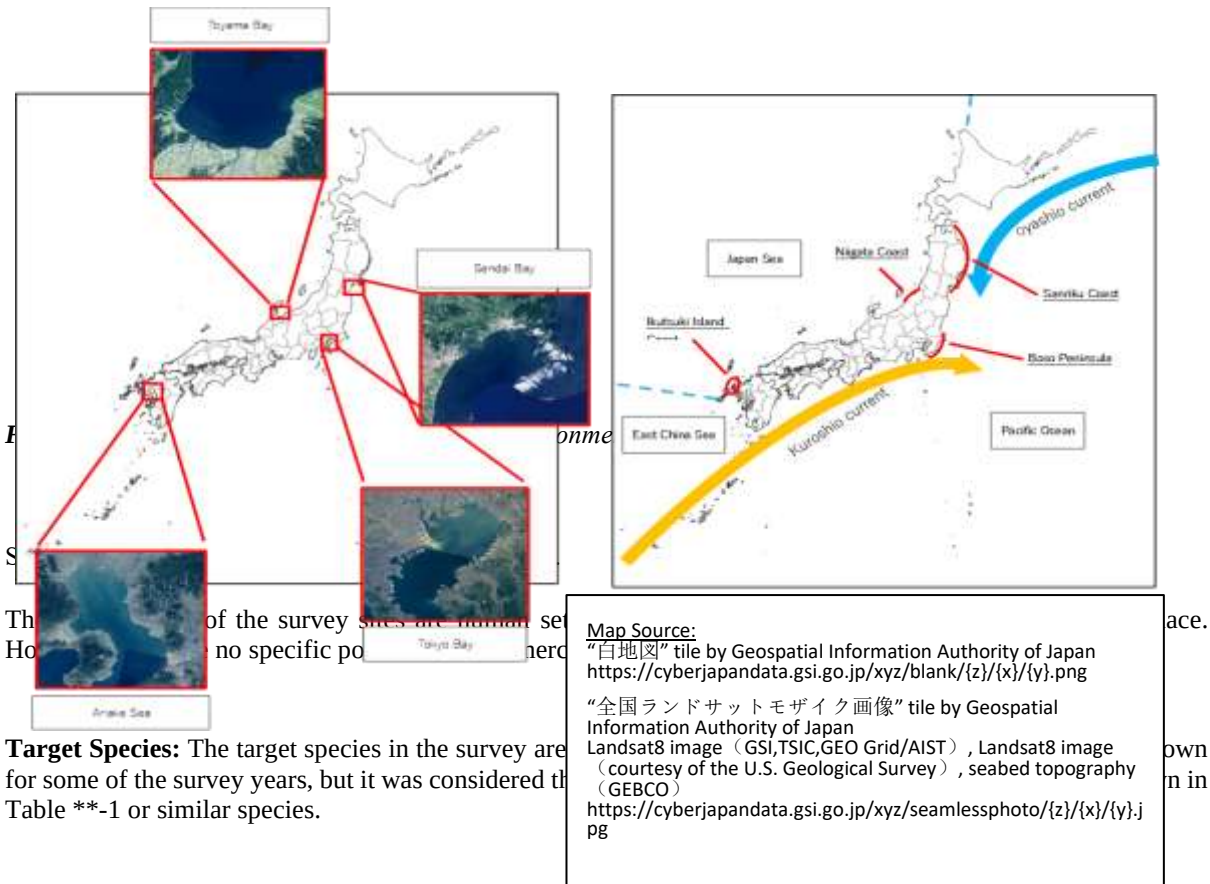
Information Used:

This case study utilizes data submitted from the long-term monitoring program “Survey of Ocean Environment” conducted in Japan. Although the survey commenced prior to 2000 and remains ongoing, the present analysis focuses on data collected during Japan’s fiscal years 2000 to 2022 (April 2000 to March 2023).

Answer/response

Sampling was conducted across both coastal and offshore areas. Coastal sites include Sendai Bay, Tokyo Bay, Ariake Sea, and Toyama Bay. Offshore sampling covered broader marine regions such as the Sea of Japan, the East China Sea, and the North Pacific Ocean, specifically within the Kuroshio and Oyashio current areas.

The map of survey area as shown in Figure 38.



Target Species: The target species in the survey are for some of the survey years, but it was considered the Table **-1 or similar species.

Table 9. Target species of “Survey of Ocean Environment”

	Example Species (Latin name)	Survey site	Target organ
Invertebrates			
Bivalves	Mediterranean mussel (<i>Mytilus galloprovincialis</i>), Purplish bifurcate mussel (<i>Septifer virgatus</i>)	Sendai Bay, Tokyo Bay, Ariake Sea, Toyama Bay	Soft tissue
Squids	Japanese flying squid (<i>Todarodes pacificus</i>),	Oyashio current area, Kuroshio current area, East China Sea, Sea of Japan	Liver
Crustaceans	Red snow crab (<i>Chionoecetes japonicus</i>)	Oyashio current area, Sea of Japan	Whole specimen excluding shell
Fish			
Benthic Sharks	Starspotted smooth-hound shark (<i>Mustelus manazo</i>) Spotless smooth-hound shark (<i>Mustelus griseus</i>), North pacific Spiny dogfish (<i>Squalus suckleyi</i>)	Sendai Bay, Tokyo Bay, Ariake Sea, Toyama Bay	Liver

Cods	Pacific cod (<i>Gadus macrocephalus</i>)	Oyashio current area, Sea of Japan	Liver

Most samples were a mixture of a few individuals, so it was difficult to evaluate and standardize the concentration data based on sample size.

Temporal trend

The data of mercury plotted over date are shown in Figure 39.

Figure 39: Temporal trends of mercury

Although the number of data points for each survey site and year may have been limited, no significant trends were found during the survey. Mercury levels in marine species (bivalves, squids, crustaceans, benthic sharks, and cod) did not show major changes from 2000 to 2023.

Evaluation of Confidence:

The survey, sample collection, and chemical analysis were conducted in the same manner for each survey in the monitoring program. Fixed quality assurance/quality control (QA/QC) procedures were applied, including limit of detection (LOD) estimation and confirmation of repeatability and reproducibility. Thus, the survey and analysis were considered with high confidence. However, due to the limited number of samples and ancillary information, such as sample size, there may be uncertainty regarding the representativeness of the samples and their characteristics (size).

Information and Capacity Needs:

Long-term monitoring is valuable for illustrating regional situations and temporal trends. Providing information on other species, particularly those at different trophic levels, offers insight into the ecological situation. Additionally, including ancillary information with the mercury data, such as sample size, will make the analysis more precise.

❖ **Mercury Pollution from Artisanal Gold Mining in Madre de Dios, Peru: A cross-media case and source attribution study**

- Description of the Case Study

Over the last 30 years, ASGM has deforested and degraded nearly 100,000 ha of high-biodiversity rainforest landscapes in the department of Madre de Dios, located in the southern Peruvian Amazon (Caballero Espejo et al., 2018), and created highly degraded landscapes that are pockmarked by thousands of hectares of mining ponds (Gerson et al., 2020). A 2018 study estimated that 181 tons of mercury are released to the region's waterways, soils, and air every year. As of this writing, Madre de Dios is considered the largest hotspot of ASGM activity and ASGM-related mercury pollution in Latin America (Cardo & Vargas, 2017).

This case study illustrates the causal link between mercury emissions from artisanal and small-scale gold mining (ASGM) activities—including amalgam burning in gold shops and mercury releases to soils and waters at mining sites—and elevated mercury in biota from aquatic and terrestrial ecosystems. The objective is to demonstrate how ASGM contamination manifests across environmental media, with implications for both biodiversity and human exposure.

Between 2017 and 2019, The *Centro de Innovación Científica Amazonica* (CINCIA) is a Peruvian non-profit scientific research centre, developed a framework to characterize the presence, magnitude, and spatial distribution of mercury pollution in and around ASGM sites. CINCIA focused on measuring mercury concentrations in biota, sediment, and air across an area that spanned more than 2,000 km² to evaluate mercury levels in different environments and media across the region to identify Hg hotspots.

ASGM and control sites were identified based on the presence and distance of mining activities. Control sites were selected to have very similar features to the impacted sites, but without ASGM. The sites defined as ASGM sites were initially located in rural environments, such as mining ponds formed during mining activity and subsequently abandoned after the cessation of the activity.

Mercury in abandoned mining ponds: sediment and fish

The methodology to assess mercury levels in mining ponds included fish muscle, collected in mining ponds located in different ASGM areas of the Madre de Dios region. Natura oxbow lakes, upriver mining activity were used as control sites.

Mercury around Mining ponds: Bird and bats

CINCIA's biomonitoring surveys were initially focused on fish because they were the predominant source of methylmercury dietary exposure for humans. However, it later expanded to include wildlife species, such as birds and bats, to develop a better understanding of the transfer of mercury along the food web (biomagnification) and the potential impact of mercury exposure on terrestrial biodiversity. In this case, bat fur and bird feathers from around Mining ponds and natural oxbow lakes were sampled for mercury analysis.

- Cross-media integration of observed levels/trends

- Mining Ponds Assessment.** The results showed that all the biota matrix (fish muscle, bird feather and bat fur) evaluated in and around mining ponds (impacted sites) and oxbows lakes (control sites), showed similar trend: *Mercury levels were consistently and significantly higher in biota from ASGM sites in comparison with control sites (fig1, 2 and 3).* In birds, we reported the highest Hg level reported for bird feather in South America. We recorded in bats from ASGM sites the highest mercury levels in bat fur reported in the Neotropics. (Sayer et al 2023; Pisconte et al 2024; Portillo et al 2023).

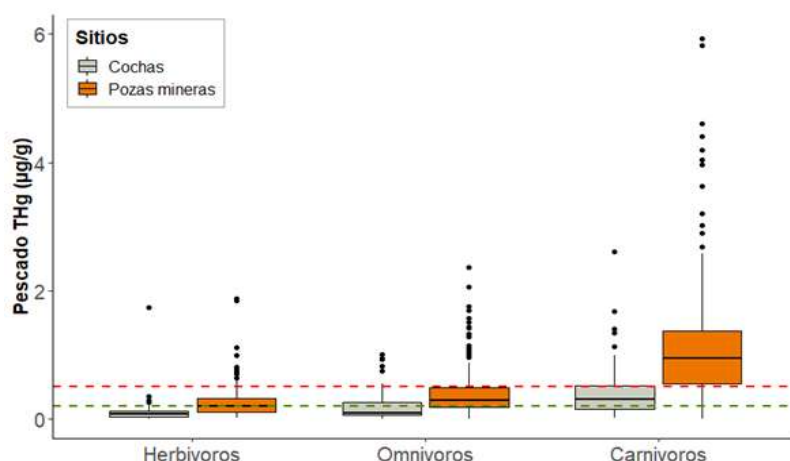


Figure 40. Mercury concentrations in fish: ASGM ponds vs. oxbow lakes (controls). The red dotted line is for WHO reference of 0.5 mg/Kg and the green one is EPA reference Level in fish of 0.3 mg/kg

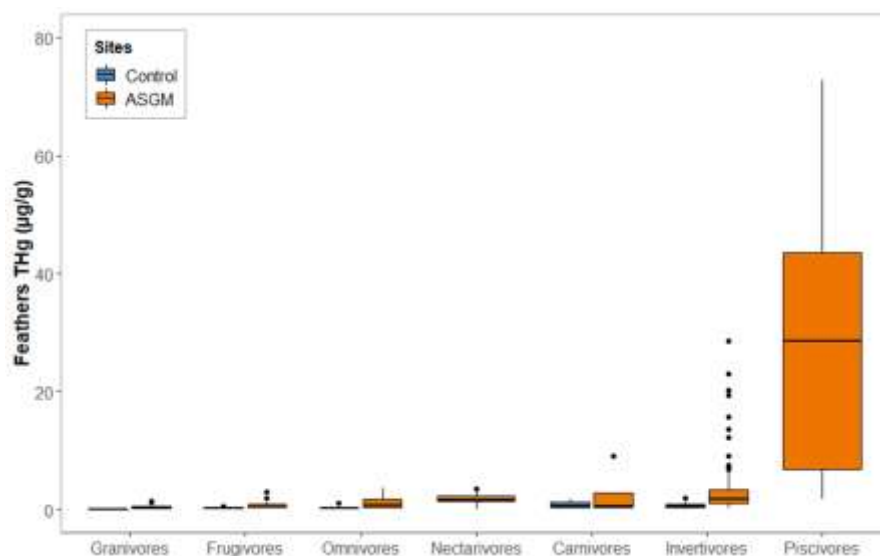


Figure 41. Mercury concentrations in bird feathers: ASGM ponds vs. oxbow lakes (controls).

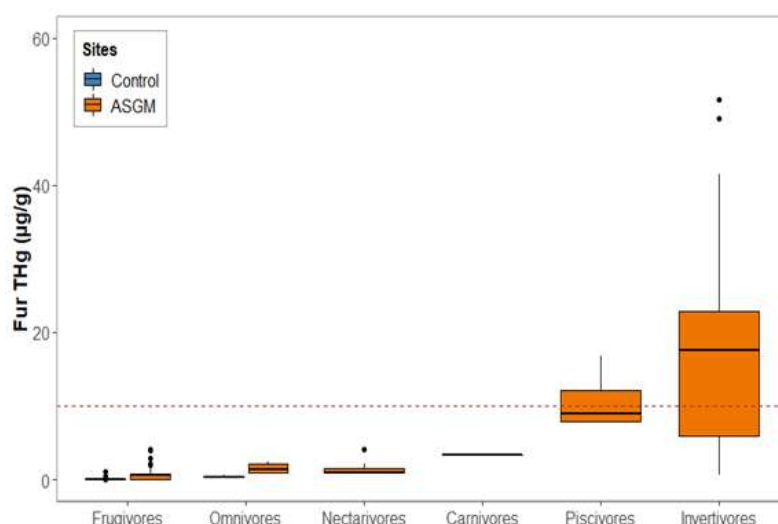


Figure 42. Mercury concentrations in bat fur: ASGM ponds vs. oxbow lakes (controls). The red dotted line shows the 10 mg/kg toxicity threshold for small mammals (Burton et al. 1977, Nam et al. 2012)

Taken together, these findings provide strong evidence of cross-media contamination, where mercury emissions from ASGM are simultaneously detectable in aquatic and terrestrial wildlife.

- Relevance to effectiveness evaluation

This case study is one of the few that directly examines mercury contamination from ASGM across multiple environmental matrices during the same time range in similar sites. The data provide clear evidence of how mercury from ASGM is distributed throughout mining regions and demonstrate the value of using cross-media indicators to characterize contamination. While bottom sediment samples did not consistently differentiate between ASGM ponds and natural oxbow lakes, biota indicators told a different story: fish, birds, and bats in and around ASGM sites consistently showed significantly higher mercury concentrations than those from control sites, further confirming the breadth of ASGM's impact across both aquatic and terrestrial systems.

These findings are directly relevant to the Minamata Convention's effectiveness evaluation, particularly under Article 7 (ASGM). They show that, despite ongoing national and international commitments, ASGM in Madre de Dios continues to release large amounts of mercury that are traceable across environmental media. This case underscores the urgent need to strengthen Peru's National Action Plan on ASGM, tighten controls on mercury trade and use, and develop targeted hotspot monitoring that can provide measurable evidence of the Convention's effectiveness.

- Gaps, Limitations and Recommendations

While this case study provides clear evidence of cross-media mercury contamination from ASGM, several important gaps and limitations should be acknowledged. The monitoring period, conducted between 2017 and 2019, was relatively short and may not fully capture longer-term temporal trends. For example, sediment samples did not consistently differentiate between ASGM and control sites, likely reflecting time lags in mercury deposition. In addition, although mercury concentrations were measured in fish, birds, and bats, the study did not include direct human exposure assessments such as dietary surveys or biomonitoring of hair or blood. As a result, the link between observed environmental contamination and community health impacts remains indirect.

Geographically, the study focused exclusively on Madre de Dios, which is the largest ASGM hotspot in Latin America but does not represent the full diversity of mining contexts across the Amazon Basin.

These limitations point to several recommendations. Future work should expand biomonitoring to include human exposure pathways and extend geographic coverage to other ASGM regions in Peru and neighboring countries. Standardizing protocols across media would improve comparability across studies and strengthen regional assessments. Finally, the integration of isotope tracing into national mercury monitoring frameworks would provide a powerful tool to attribute sources more precisely and support enforcement of Minamata Convention commitments.

Key findings/messages

Across regional and local case studies (AMAP, Beluga Case Study, OSPAR/HELCOM, NOAA Mussel Watch, Great Lakes, Fennoscandia, Australian Derwent Estuary Program, Common Loon blood mercury study, Japan's case study), mercury in biota exhibits different trends (stable, increasing, and decreasing trends), driven by a complex interplay of local ecology, food-web dynamics, species traits, climate influences, legacy contamination, biogeochemistry, and hydrodynamics. While mercury emission reductions provide benefits, biotic responses are delayed, often slow, uneven, and masked by legacy stores and ecological factors. Exceedances of health and environmental thresholds remain common in some systems.

In the Derwent Estuary, spatial gradients near the zinc smelter and mixed invertebrate trends suggest that local conditions strongly shape mercury bioavailability. The absence of a clear temporal decline in fish, despite emission reductions, is consistent with outcomes in the Great Lakes and Fennoscandia, highlighting the role of legacy stores and ecosystem processes.

Interestingly, at the Flin Flon Cu-Zn smelter, mercury concentrations in fish increased in nearby lakes even after the smelter's closure in 2010 and major reductions in emissions, highlighting the complex and sometimes counterintuitive ecological responses to emission decline.

Early studies using biota monitoring typically interpreted temporal trends in terms of changes in (global, regional or local) emissions and releases of mercury, often making relatively simplistic assumptions in this regard. Complex trend patterns, e.g. differing trends in different tissues of the same animals, and trends that did not match anticipated relationships to changing emissions, however, lead to a need for greater attention to potential confounding factors and processes when attributing causes of changes in levels in biota.

The impacts of climate change over the past 20 years have added a new dimension, in particular in regions such as the Arctic where environmental change is occurring at an unprecedented rate. This is altering not only pathways impacting long-range transport of contaminants such as mercury, but also potentially (re)mobilizing mercury that has accumulated in water and soils from historical emissions, as well as altering ecosystem structures and food-webs that are important factors in determining mercury levels in certain (especially higher trophic level) species.

Thus, considerable care needs to be taken in attribution of trends to changes in emissions or releases, especially in areas away from local sources. An evaluation of the degree of change in levels of mercury in environmental media, including biota, that may be expected as a result of the actions under the Minamata Convention is an appropriate initial consideration in this respect.

While regional and local monitoring programs provide valuable insights, significant gaps remain. These include limited trend data for artisanal and small-scale gold mining (ASGM)-impacted areas, Southern Hemisphere ecosystems, and certain taxa and regions. Addressing these gaps is critical for supporting the Minamata Convention effectiveness evaluation and its objectives.

Moving forward, integrated models linking emissions, deposition, methylmercury production and biotic mercury are essential. These models should incorporate local ecological and climate drivers, utilise Indigenous Knowledge where possible, and apply holistic (such as One Health), as well as robust statistical approaches to improve attribution and trend interpretation. See integrated analysis (section X) for further case studies.

References

- Ackerman, J.T., Eagles-Smith, C.A., Herzog, M.P., 2011. Bird mercury concentrations change rapidly as chicks age: Toxicological risk is highest at hatching and ledging. *Environ. Sci. Technol.* 45, 5418–5425.
- Ackerman, JT, CA Eagles-Smith, MP Herzog, CA Hartman, SH Peterson, DC Evers, AK Jackson, JE Elliott, SS Vander Pol, and CE Bryan. 2016. Avian mercury exposure and toxicological risk across western North America: a synthesis. *Science of the Total Environment* 568:749-769.]
- Ackerman, JT, MP Herzog, SH Peterson, and JL Yee. 2024b. The Bird Mercury Tool: Estimating methylmercury injury to birds: U.S. Geological Survey Software Release, <https://doi.org/10.5066/P9MSD8X0>.
- Ackerman, JT, SH Peterson, MP Herzog, and JL Yee. 2024a. Methylmercury effects on birds: A review, meta-analysis, and development of toxicity reference values for injury assessment based on tissue residues and diet. *Environmental Toxicology and Chemistry* 43:1195-1241. <https://doi.org/10.1002/etc.5858>.
- AMAP Adam D. Morris, Simon J. Wilson, Rob J. Fryer, Philippe J. Thomas, Karista Hudelson, Birgitta Andreassen, Pierre Blévin, Paco Bustamante, Olivier Chastel, Guttorm Christensen, Rune Dietz, Marlene Evans, Anita Evenset, Steven H. Ferguson, Jérôme Fort, Mary Gamberg, David Grémillet, Magali Houde, Robert J. Letcher, Lisa Loseto, Derek Muir, Marianna Pinzone, Amanda Poste, Heli Routti, Christian Sonne, Gary Stern, Frank F. Rigét, Temporal trends of mercury in Arctic biota: 10 more years of progress in Arctic monitoring, *Science of the Total Environment*, Volume 839, 2022. <https://doi.org/10.1016/j.scitotenv.2022.155803>.
- AMAP, 2021. AMAP Assessment 2021: Mercury in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Tromsø, Norway. 324 pp.
- Ammar, Y., Faxneld, S., Sköld, M. et al. Long-term dataset for contaminants in fish, mussels, and bird eggs from the Baltic Sea. *Sci Data* 11, 400 (2024). <https://doi.org/10.1038/s41597-024-03216-0>
- Basu, N, AM Scheuhammer, C Sonne, RJ Letcher, EW Born, and R Dietz. 2009. Is dietary mercury of neurotoxicological concern to wild polar bears (*Ursus maritimus*)? *Environmental Toxicology and Chemistry* 28:133–140.
- Bloom, H.; Ayling, G. M. Heavy metals in the Derwent Estuary. *Environmental Geology* 1977, 2 (1), 3-22. DOI: 10.1007/BF02430661.
- Braaten et al. 2019 *Environmental Science and Technology* <https://doi.org/10.1021/acs.est.8b06399>
- Caballero Espejo, J., Messinger, M., Román-Dañobeytia, F., Ascorra, C., Fernandez, L. E., & Silman, M. (2018). Deforestation and forest degradation due to gold mining in the Peruvian Amazon: A 34-year perspective. *Remote Sensing*, 10(12), 1903.
- Cardo, M. A., & Vargas, P. M. (2017). Proyecto: Plan nacional de acción sobre mercurio en el sector de la minería de oro artesanal y de pequeña escala en el Perú. (Artisanal Gold Council 2017)
- Céline Albert, Marina Renedo, Paco Bustamante, Jérôme Fort, Using blood and feathers to investigate large-scale Hg contamination in Arctic seabirds: A review. *Environmental Research*, Volume 177, 2019, 108588, <https://doi.org/10.1016/j.envres.2019.108588>.
- Chastel, O. et al. Mercury contamination and potential health risks to Arctic seabirds and shorebirds. *Sci. Total Environ.* 844, 156944(2022).
- Chételat, J., Ackerman, J.T., Eagles-Smith, C.A., Hebert, C.E., 2020. Methylmercury exposure in wildlife: A review of the ecological and physiological processes affecting contaminant concentrations and their interpretation. *Science of the Total Environment*.
- Choy, E.S., Giraldo, C., Rosenberg, B. Roth, J.D., Stasko, A., Majewski, A., Swanson, H., Power, M., Reist, J.D., Loseto, L.L. 2020. Variation in the diet of beluga whales in response to changes in prey availability: insights on changes in the Beaufort Sea ecosystem. *MEPS*, 647: 195-210.
- Cioffi, W. R., Quick, N. J., Foley, H. J., Waples, D. M., Swaim, Z. T., Shearer, J. M., Webster, D. L., Friedlaender, A. S., Southall, B. L., Baird, R. W., Nowacek, D. P., & Read, A. J. (2021). Adult male Cuvier's beaked whales (*Ziphius cavirostris*) engage in prolonged bouts of synchronous diving. *Marine Mammal Science*, 37(3), 1085–1100. <https://doi.org/10.1111/mms.12799>
- D. A.; Dix, T.; Wilson, K. Mercury in fish in the Derwent Estuary, Tasmania, and its relation to the position of the fish in the food chain. *Marine and Freshwater Research* 1975, 26, 223-231.

Davis, J. A., J. S. Gift, and Q. J. Zhao. 2011. Introduction to benchmark dose methods and U.S. EPA's benchmark dose software (BMDS) version 2.1.1. *Toxicology and Applied Pharmacology* 254:181–191.

DEP. Metal contamination in fish and shellfish of the Derwent estuary - A summary of results to December 2020; Derwent Estuary Program, 2021.

DEP. State of the Derwent Estuary 2020 - An update and review of environmental data and activities; Derwent Estuary Program, 2020.

Depew, DC, N Basu, NM Burgess, LM Campbell, EW Devlin, PE Drevnick, CR Hammerschmidt, CA Murphy, MB Sandheinrich, and JG Wiener. 2012. Toxicity of dietary methylmercury to fish: Derivation of ecologically meaningful threshold concentrations. *Environmental Toxicology and Chemistry* 31:1536–47.

Dietz, R., Letcher, R., Aars, J., Andersen, M., Boltunov, A., et al. 2022. A risk assessment review of mercury exposure in Arctic marine and terrestrial mammals. *Science of the Total Environment*, 829, 154445. <https://doi.org/10.1016/j.scitotenv.2022.154445>

Dillon, T, N Beckvar, and J Kern. 2010. Residue based mercury dose-response in fish: An analysis using lethality equivalent test endpoints. *Environmental Toxicology and Chemistry* 29:2559–2565.

Einoder, L. D.; MacLeod, C. K.; Coughanowr, C. Metal and Isotope Analysis of Bird Feathers in a Contaminated Estuary Reveals Bioaccumulation, Biomagnification, and Potential Toxic Effects. *Arch Environ Contam Toxicol* 2018, 75 (1), 96-110. DOI: 10.1007/s00244-018-0532-z From NLM.

EPA Tasmania. Environment Protection Notice (EPN) No. 7043/6. Tasmania, Australia, 2024.

Faxneld, S., Eliassen, J., Jonsson, C., Soerensen, A.L. 2024. Graphic and statistical overview of temporal trends and spatial variations within the Swedish National Monitoring Programme for Contaminants in Freshwater Biota (until 2022 year's data). NRM Report NV-00839-24

Gerson, J. R., Szponar, N., Zambrano, A. A., Bergquist, B., Broadbent, E., Driscoll, C. T., Vega, C & Bernhardt, E. S. (2022). Amazon forests capture high levels of atmospheric mercury pollution from artisanal gold mining. *Nature communications*, 13(1), 1-10.

Galicía, M. P., Thiemann, G. W., Dyck, M. G., & Ferguson, S. H. (2021). Polar bear diet composition reveals spatiotemporal distribution of Arctic marine mammals across Nunavut, Canada. *Ecological Indicators*, 132, 108245. <https://doi.org/10.1016/j.ecolind.2021.108245>

Gift, J., J. A. Davis, T. Blessinger, M. Wheeler, L. Olszyk, C. Simmons, Y. Xu, B. Allen, and M. E. Brown. 2020. Benchmark Dose Software (BMDS). Version 3.2 User Guide. EPA/600/R-20/216. U.S. Environmental Protection Agency.

Gilles, E.J., Li, Mi-Ling, Christensen, V., Hoover, C., Sora, K.J., Loseto, L.L., Cheung, W.W.L., Angot, H., Giang, A. 2024. Exploring drivers of historic mercury trends in beluga whales using an ecosystem modeling approach. *ACS Environmental Au* 2024, 4, 5 219-235. <https://doi.org/10.1021/acsenvironau.3c00072>

Gyapay J, Noksana K, Ostertag S, Wesche S, Laird BD, Skinner K. Informing the Co-Development of Culture-Centered Dietary Messaging in the Inuvialuit Settlement Region, Northwest Territories. *Nutrients*. 2022 May 3;14(9):1915. doi: 10.3390/nu14091915. PMID: 35565882; PMCID: PMC9099519.

Gyapay, J., Ostertag, S., Wesche, S., Laird, B., Skinner, K., (2025), "[Characterizing the development and dissemination of dietary messaging in the Inuvialuit Settlement Region, Northwest Territories](#)", HSSCommons: (DOI: 10.15353/cfs-rcea.v10i1.569)

Hardy, A., D. Benford, T. Halldorsson, M. J. Jeger, K. H. Knutsen, S. More, A. Mortensen, H. Naegeli, H. Noteborn, C. Ockleford, A. Ricci, G. Rychen, V. Silano, R. Solecki, D. Turck, M. Aerts, L. Bodin, A. Davis, L. Edler, U. Gundert - Remy, S. Sand, W. Slob, B. Bottex, J. C. Abrahantes, D. C. Marques, G. Kass, and J. R. Schlatter. 2017. Update: use of the benchmark dose approach in risk assessment. *EFSA Journal* 15:1–41.

HARSAT: Harmonized Regional Seas Assessment Tool. R package version 1.0.3.1008; Arctic Monitoring and Assessment Programme (AMAP); Helsinki Commission (HELCOM); OSPAR Commission (OSPAR); International Council for the Exploration of the Sea (ICES); AmbieSense Ltd (2025): <https://github.com/osparcomm/HARSAT>.

Hawk, D.W., Dunkin, R.C., Batac, F., Miller, M., Weiss-Penzias, P. 2024. Assessment of mercury concentrations in fur, liver, and brain tissue from necropsied southern sea otters (*Enhydra lutris nereis*). *Marine Mammal Science* 40: e13112. <https://doi.org/10.1111/mms.13112>

Houde, M.; Wang, X.; Colson, T.L.L.; Gagnon, P.; Ferguson, S.H.; Ikonou, M.G.; Dubetz, C.; Addison, R.F.; Muir, D.C.G. Trends of persistent organic pollutants in ringed seals (*Phoca hispida*) from the Canadian Arctic. *Sci Total Environ* 2019;665:1135-1146 <https://doi.org/10.1016/j.scitotenv.2019.02.138>

Jones, H. J. Accumulation of mercury in estuarine food webs: biogeochemical and ecological considerations. University of Tasmania, 2013.

Jones, H. J.; Swadling, K. M.; Tracey, S. R.; Macleod, C. K. Long term trends of Hg uptake in resident fish from a polluted estuary. *Marine Pollution Bulletin* 2013, 73 (1), 263-272. DOI: <https://doi.org/10.1016/j.marpolbul.2013.04.032>.

Joshua T. Ackerman, Sarah H. Peterson, Mark P. Herzog, Julie L. Yee, Methylmercury Effects on Birds: A Review, Meta - Analysis, and Development of Toxicity Reference Values for Injury Assessment Based on Tissue Residues and Diet, *Environmental Toxicology and Chemistry*, Volume 43, Issue 6, 1 June 2024, Pages 1195–1241, <https://doi.org/10.1002/etc.5858>

Loseto, L.L., Stern, G.A., Macdonald, R.M. 2015. Distant drivers or local signals: where do mercury trends in western Arctic belugas originate? *Science of the Total Environment*, 509-510 (226-236).

Macleod, C.; Coughanowr, C. Heavy metal pollution in the Derwent estuary: History, science and management. *Regional Studies in Marine Science* 2019, 32, 100866. DOI: <https://doi.org/10.1016/j.rsma.2019.100866>.

MacMillan, K., Hoover, C., Iacozza, J., Peyton, J., Loseto, L.L. 2023. Understanding Trends and Variability of Beluga Whale Body Condition and Influence of Large-Scale Climatic Drivers. *Arctic Science*, 9 (3) 678-688. <https://doi.org/10.1139/as-2021-0026>

Mayette, A., Loseto, L., Pearce, T., Hornby, C., Marcoux, M. 2022. Group characteristics and spatial organization of the eastern Beaufort Sea beluga whales (*Delphinapterus leucas*) population using aerial photographs. *CJZ*, 100(6): 363-375. <https://doi.org/10.1139/cjz-2021-0232>

Mayfield, D. B., and D. G. Skall. 2018. Benchmark dose analysis framework for developing wildlife toxicity reference values. *Environmental Toxicology and Chemistry* 37:1496–1508.

McInnes, J. D., Lester, K. M., Dill, L. M., Mathieson, C. R., West-Stap, P. J., Marcos, S. L., & Trites, A. W. (2024). Foraging behaviour and ecology of transient killer whales within a deep submarine canyon system. *PLOS ONE*, 19(3), e0299291. <https://doi.org/10.1371/journal.pone.0299291>

Morris, A. et al. 2022. Temporal trends of mercury in Arctic biota: 10 more years of progress in Arctic monitoring. *STOTEN*, 839: 155803. <https://doi.org/10.1016/j.scitotenv.2022.155803>

Murray, L., Campbell, T.K.F., MacPhee, S., Elliott, A., Hansen-Craik, K., Aklavik Hunters & Trappers Committee, Inuvik Hunters & Trappers Committee, Olokhtomiut Hunters & Trappers Committee, Paulatuk Hunters & Trappers Committee, Sachs Harbour Hunters & Trappers Committee, Tuktoyaktuk Hunters & Trappers Committee, and Loseto, L. 2024. Harvest data and sample collection metadata from the 2020 beluga (*Delphinapterus leucas*) subsistence harvest monitoring program in the Inuvialuit Settlement Region. Canada/Inuvialuit Fisheries Joint Management Committee Report. 2024-01: vi + 41 p

Noel, AL., Devred, E., Iacozza, J., Marcoux, M., Hornby, C., Loseto, L.L. 2022. Exploring potential shifts in late summer beluga distribution in the context of climate change: A case study of inshore regions in the Mackenzie Estuary and TN MPA, Western Canadian Arctic. *Arctic Science*, 8: 1305-1319.

Noel, M., Stern, G.A., Loseto, L.L. 2018. Legacy contaminants in the Eastern Beaufort Sea beluga whales (*Delphinapterus leucas*): Are temporal trends reflecting regulations? *Arctic Science*, 4: 373-387.

NPI. National Pollutant Inventory data within Australia - Mercury and mercury compounds from Hobart Zinc Smelting Source. 2025.}

Nyrstar. Nyrstar Hobart Triennial Public Environment Report 2022 – 2024; 2025.

Ovitz, K., Matari, K., O'Hara, S., Esogak, D., Loseto, L.L. 2023. Observations of social and environmental change on Kendall Island (Ukiivik), a traditional whaling camp in the Inuvialuit Settlement Region. *Arctic Science*, 00: 1-29. <https://cdnsiencepub.com/doi/abs/10.1139/AS-2022-0016>.

Petherick, A. S., Reuther, J. D., Shirar, S. J., Anderson, S. L., & DeSantis, L. R. G. (2021). Dietary ecology of Alaskan polar bears (*Ursus maritimus*) through time and in response to Arctic climate change. *Global Change Biology*, 27(13), 3109–3119. <https://doi.org/10.1111/gcb.15573>

Pisconte, J. N., Vega, C. M., Sayers, C. J., Sevillano-Ríos, C. S., Pillaca, M., Quispe, E., ... & Fernandez, L. E. (2024). Elevated mercury exposure in bird communities inhabiting Artisanal and Small-Scale Gold Mining landscapes of the southeastern Peruvian Amazon. *Ecotoxicology*, 33(4), 472-483.

Plint, T., ten Doeschate, M. T. I., Brownlow, A. C., Davison, N. J., Hantke, G., Kitchener, A. C., Longstaffe, F. J., McGill, R. A. R., Simon-Nutbrown, C., & Magill, C. R. (2023). Stable isotope ecology and interspecific dietary overlap among dolphins in the Northeast Atlantic. *Frontiers in Marine Science*, 10. <https://doi.org/10.3389/fmars.2023.1111295>

Portillo, A., Vega, C. M., Mena, J. L., Bonifaz, E., Ascorra, C., Silman, M. R., & Fernandez, L. E. (2023). Mercury bioaccumulation in bats in Madre de Dios, Peru: implications for Hg bioindicators for tropical ecosystems impacted by artisanal and small-scale gold mining. *Ecotoxicology*, 1-13.

Raes, E. J.; Holmes, B. H.; Karsh, K.; Hillyer, K. E.; Green, M.; van de Kamp, J.; Bodrossy, L.; Whitehead, S.; Proemse, B.; Taylor, U.; et al. Organic matter and metal loadings influence the spatial gradient of the benthic bacterial community in a temperate estuary. *Marine and Freshwater Research* 2022, 73 (4), 428-440. DOI: <https://doi.org/10.1071/MF21225>.

Ramirez Prieto M, Ostertag S, Noksana K, Arey S, Wolki D, Memogana S, Wolki C, Yurris A, Skinner K. 'Keep learning, keep trying': exploring food and cultural experiences and supports of Inuvialuit youth in the Inuvialuit Settlement Region. *Glob Public Health*. 2025 Dec;20(1):2516704. doi: 10.1080/17441692.2025.2516704. Epub 2025 Jun 18. PMID: 40531079.

Rigét, F.; Bignert, A.; Braune, B.; Dam, M.; Dietz, R.; Evans, M.; Green, N.; Gunnlaugsdóttir, H.; Hoydal, K.S.; Kucklick, J.; Letcher, R.; Muir, D.; Schuur, S.; Sonne, C.; Stern, G.; Tomy, G.; Vorkamp, K.; Wilson, S. Temporal trends of persistent organic pollutants in Arctic marine and freshwater biota. *Sci Total Environ* 2019;649:99-110 <https://doi.org/10.1016/j.scitotenv.2018.08.268>

Santos, M. B., Monteiro, S. S., Vingada, J. v., Ferreira, M., López, A., Martínez Cedeira, J. A., Reid, R. J., Brownlow, A., & Pierce, G. J. (2014). Patterns and trends in the diet of long - finned pilot whales (*Globicephala melas*) in the northeast Atlantic. *Marine Mammal Science*, 30(1), 1–19. <https://doi.org/10.1111/mms.12015>

Sayers, C. J., Evers, D. C., Ruiz-Gutierrez, V., Adams, E., Vega, C. M., Pisconte, J. N., ... & Fernandez, L. E. (2023). Mercury in Neotropical birds: a synthesis and prospectus on 13 years of exposure data. *Ecotoxicology*, 32(8), 1096-1123.

Scheuhammer, AM, MW Meyer, MB Sandheinrich, and MW Murray. 2007. Effects of environmental methylmercury on the health of wild birds, mammals, and fish. *Ambio* 36:12–18.

Shearer, J., Jensen, F., Quick, N., Friedlaender, A., Southall, B., Nowacek, D., Bowers, M., Foley, H., Swaim, Z., Waples, D., & Read, A. (2022). Short-finned pilot whales exhibit behavioral plasticity in foraging strategies mediated by their environment. *Marine Ecology Progress Series*, 695, 1–14. <https://doi.org/10.3354/meps14132>

Shore, RF, E Pereira, LA Walker, and DR Thompson. 2011. Mercury in nonmarine birds and mammals. Pages 609–626 in WN Beyer and JP Meador, editors. *Environmental Contaminants in Biota: Interpreting Tissue Concentrations*, Second Edition. CRC Press, Boca Raton, Florida.

Soerensen, A. L., C. Jonsson, A. Forsberg, J. Eliassen, N. Stjärnkvist, S. Faxneld. 2025. The Swedish National Monitoring Programme for Contaminants in Marine Biota (until 2023 year's data) - temporal trends and spatial variations. Swedish Museum of Natural History, Stockholm, Sweden.

Stringell, T., Hill, D., Rees, D., Rees, F., Rees, P., & Morgan, G. (2015). Short Note: Predation of Harbour Porpoises (*Phocoena phocoena*) by Grey Seals (*Halichoerus grypus*) in Wales. *Aquatic Mammals*, 41(2), 188–191. <https://doi.org/10.1578/AM.41.2.2015.188>

Sudlovenick, E., Pokiak, V., Loseto, L.L. 2025. When research relies on wildlife samples obtained from communities: a case study on local cultural contexts of beluga whale harvesting in Tuktoyaktuk, NT and Arviat, NU. *Arctic Science*, In press. <https://doi.org/10.1139/as-2024-0036>

Sudlovenick, E., Pokiak, V., Swanson, H., Kirk, J., Loseto, L.L. 2024. Inuit harvester selection and observed mercury levels in Eastern Beaufort Sea and Western Hudson Bay beluga whales (*Delphinapterus leucas*). *STOTEN*. Dec 2024, 176317. <https://doi.org/10.1016/j.scitotenv.2024.176317>.

Swam et al 2025. [to be completed]

Szponar, N., Vega, C. M., Gerson, J., McLagan, D. S., Pillaca, M., Delgado, S., ... & Bergquist, B. A. (2025). Tracing atmospheric mercury from artisanal and small-scale gold mining. *Environmental Science & Technology*, 59(10), 5021-5033.

Tobias, J.A., Sheard, C., Pigot, A.L., Devenish, A.J., Yang, J., Sayol, F., Neate - Clegg, M.H., Alioravainen, N., Weeks, T.L., Barber, R.A. and Walkden, P.A. (2022). AVONET: morphological, ecological and geographical data for all birds. *Ecology letters*, 25(3), 581-597.

Tracey, S. R.; Hartmann, K.; McAllister, J.; Lyle, J. M. Home range, site fidelity and synchronous migrations of three co-occurring, morphologically distinct estuarine fish species. *Science of The Total Environment* 2020, 713, 136629. DOI: <https://doi.org/10.1016/j.scitotenv.2020.136629>.

U.S. Environmental Protection Agency 2012 Benchmark dose technical guidance. EPA/100/R-12/001. Risk Assessment Forum, Washington, D.C.

UNEP. UNEP Global Mercury Partnership Study report on mercury from non-ferrous metals mining and smelting; Geneva, 2021.

USEPA. 1997. Mercury Study Report to Congress Vol. VII: Characterization of Human Health and Wildlife Risks from Mercury Exposure in the United States. Office of Air Quality Planning and Standards and Office of Research and Development, U.S. Environmental Protection Agency, EPA-452/R-97-009. EPA, Washington, DC.]

USFDA/EPA (2024) Technical Information on Development of FDA/EPA Advice about Eating Fish for Those Who Might Become or Are Pregnant or Breastfeeding and Children Ages 1-11 Years. <https://www.fda.gov/food/metals-and-your-food/technical-information-development-fdaepa-advice-about-eating-fish-those-who-might-become-or-are>

Waugh, D., Pearce, T., Ostertag, S., Pokiak, V., Collings, P., Loseto, L. 2018. Inuvialuit Traditional Ecological Knowledge of Beluga Whale (*Delphinapterus leucas*) Under Changing Climatic Conditions in Tuktoyaktuk, NT. *Arctic Science*, 4: 242-258.

Whitehead, J. C., Christine; Rushton, Michael; Derwent Estuary surface sediments comparison between 2000 & 2011 surveys: - metals, clay/silt ratio & % organic carbon; Derwent Estuary Program and Tasmanian Department of Primary Industries, Parks, Water and Environment, 2013.

Wiener, JG, DP Krabbenhoft, GH Heinz, and AM Scheuhammer. 2003. Ecotoxicology of mercury. In: *Handbook of Ecotoxicology* (2nd ed). Hoffman, D.J., Rattner, B.A., Burton, G.A. and Cairns, J. (eds). CRC Press, Boca Raton, Florida, pp. 409-463.

Worden, E., Pearce, T., Gruben, M., Ross, D., Kowana, C., Loseto, L. 2020. Social-ecological changes and implications for understanding the declining beluga whale (*Delphinapterus leucas*) harvest in Aklavik, NT. *Arctic Science*, 6: 229-246.

Yang, J., Miyazaki, N. 2003. Moisture content in Dall's porpoise (*Phocoenoides dalli*) tissues: a reference base for conversion factors between dry and wet weight trace element concentrations in cetaceans. *Environmental Pollution*, 121: 345-347. [https://doi.org/10.1016/S0269-7491\(02\)00239-7](https://doi.org/10.1016/S0269-7491(02)00239-7) .

2.4.5 Annex 1

Table S1. List of observational data used to evaluate current levels. [to be completed]

Data Provider or Source	Brief Description of Dataset	Type of Data	Time Span	Geographic scope	Citation

HELCOM	Hg concentrations in zebra mussels (<i>Dreissena polymorpha</i>) and blue mussels (<i>Mytilus edulis</i>) from the HELCOM HOLAS III hazardous substances assessment.	Invertebrate (bivalve)	2014-2021	IPCC Regions: N. Europe and West & Central Europe; Countries: Lithuania, Germany, Denmark, Poland, Sweden	ICES (2023). Input data files for the HELCOM HOLAS III hazardous substances assessment 2016-2021. Data Outputs. Dataset. https://doi.org/10.17895/ices.data.24042021.v1
A Agrawal and JE Brandt, University of Connecticut, USA	Hg concentrations in Eastern oysters (<i>Crassostrea virginica</i>) deployed at coastal sites along the northern boundary of Long Island Sound	Invertebrate (bivalve)	2022-2023	IPCC Regions: E. North America; Connecticut, USA	Agrawal, Anika, et al. "Macroparasite infection and mercury contamination as coupled stressors in Eastern oysters." <i>Ecotoxicology and Environmental Safety</i> 302 (2025): 118672.
USA NOAA Mussel Watch Program	Hg concentrations in blue mussels (<i>Mytilus edulis</i>), California mussels (<i>Mytilus californianus</i>), <i>Mytilus</i> sp., Eastern oysters (<i>Crassostrea virginica</i>), and Mangrove cupped oysters (<i>Crassostrea rhizophorae</i>) sampled at various coastal sites in the USA.	Invertebrate (bivalve)	2015-2021	IPCC Regions: E. North America, W. North America, C. North America, N. Central America, Caribbean	Data provided by NOAA Mussel Watch Program Personnel. https://coastalscience.noaa.gov/science-areas/pollution/mussel-watch/

Nyrstar Hobart Derwent Estuary Program	Hg concentrations in blue mussels (<i>Mytilus edulis</i>) and Pacific oysters (<i>Crassostrea gigas</i>) from various sites in the Derwent Estuary monitoring network.	Invertebrate (bivalve)	2014-2023	IPCC Region: S. Australia	Derwent Estuary Program (DEP), 2021. Metal contamination in fish and shellfish of the Derwent estuary - A summary of results to December 2020. https://www.derwentestuary.org.au/assets/Metals_in_fish_FINAL_24Sep2021.pdf Derwent Estuary Program (DEP), 2020. State of the Derwent Estuary 2020 - An update and review of environmental data and activities. https://www.derwentestuary.org.au/news/state-of-the-derwent-2020-update/
OSPAR Commission	Hg concentrations in blue mussels (<i>Mytilus edulis</i>), Mediterranean mussels (<i>Mytilus galloprovincialis</i>), <i>Mytilus</i> sp., Eastern oysters (<i>Crassostrea virginica</i>), and European flat oysters (<i>Ostrea edulis</i>) from various sites in the OSPAR	Invertebrate (bivalve)	2014-2022	IPCC Regions: N. Europe, West & Central Europe, N. Atlantic Ocean, Mediterranean, Greenland/Iceland; Countries: Belgium, Denmark, France, Germany, Iceland, Ireland, Netherlands, Norway, Portugal, Spain, Sweden, United Kingdom	
Japanese Survey of Ocean Environment	Hg concentrations in Mediterranean mussels (<i>Mytilus galloprovincialis</i>) from the Ariake Sea, Tokyo Bay, and Toyama Bay, Japan.	Invertebrate (bivalve)	2014, 2016, 2020-2022	IPCC Region: E. Asia, Country: Japan	https://tenbou.nies.go.jp/gisplus/download.html

Table S2. List of observational data used to evaluate temporal levels. [to be completed]

Data Provider or Source	Brief Description of Dataset	Type of Data	Time Span	Geographic scope	Citation
HELCOM	Hg concentrations in zebra mussels (<i>Dreissena polymorpha</i>) and blue mussels (<i>Mytilus edulis</i>) from the HELCOM HOLAS III hazardous substances assessment.	Invertebrate (bivalve)	2000-2021	IPCC Regions: N. Europe and West & Central Europe; Countries: Lithuania, Germany, Denmark, Poland, Sweden	ICES (2023). Input data files for the HELCOM HOLAS III hazardous substances assessment 2016-2021. Data Outputs. Dataset. https://doi.org/10.17895/ices.data.24042021.v1
A Agrawal and JE Brandt, University of Connecticut, USA	Hg concentrations in Eastern oysters (<i>Crassostrea virginica</i>) deployed at coastal sites along the northern boundary of Long Island Sound	Invertebrate (bivalve)	2022-2023	IPCC Regions: E. North America; Connecticut, USA	Agrawal, Anika, et al. "Macroparasite infection and mercury contamination as coupled stressors in Eastern oysters." <i>Ecotoxicology and Environmental Safety</i> 302 (2025), 118672.
USA NOAA Mussel Watch Program	Hg concentrations in blue mussels (<i>Mytilus edulis</i>), California mussels (<i>Mytilus californianus</i>), Mediterranean mussels (<i>Mytilus galloprovincialis</i>), <i>Mytilus</i> sp., Eastern oysters (<i>Crassostrea virginica</i>), Pacific oysters (<i>Crassostrea gigas</i>), <i>Dreissena</i> sp., and Mangrove cupped oysters (<i>Crassostrea rhizophorae</i>) sampled at various coastal sites in the USA.	Invertebrate (bivalve)	2000-2012, 2015-2021	IPCC Regions: E. North America, W. North America, N.W. North America, C. North America, N. Central America, Caribbean	Data provided by NOAA Mussel Watch Program Personnel. https://coastalscience.noaa.gov/science-areas/pollution/mussel-watch/

Nyrstar Hobart / Derwent Estuary Program	Hg concentrations in blue mussels (<i>Mytilus edulis</i>) and Pacific oyster (<i>Crassostrea gigas</i>) from various sites in the Derwent Estuary monitoring network.	Invertebrate (bivalve)	2002, 2004-2023	IPCC Region: S. Australia	Derwent Estuary Program (DEP) 2021. Metal contamination in fish and shellfish of the Derwent estuary - A summary of results to December 2020. https://www.derwentestuary.org.au/assets/Metals_in_fish_FINAL_24Sept2021.pdf Derwent Estuary Program (DEP) 2020. State of the Derwent Estuary 2020 - An update and review of environmental data and activities. https://www.derwentestuary.org.au/news/state-of-the-derwent-2020-update/
OSPAR Commission	Hg concentrations in blue mussels (<i>Mytilus edulis</i>), Mediterranean mussels (<i>Mytilus galloprovincialis</i>), <i>Mytilus</i> sp., Eastern oysters (<i>Crassostrea virginica</i>), and European flat oysters (<i>Ostrea edulis</i>) from various sites in the OSPAR	Invertebrate (bivalve)	2000-2022	IPCC Regions: N. Europe, West & Central Europe, N. Atlantic Ocean, Mediterranean, Greenland/Iceland; Countries: Belgium, Denmark, France, Germany, Iceland, Ireland, Netherlands, Norway, Portugal, Spain, Sweden, United Kingdom	
Japanese Survey of Ocean Environment	Hg concentrations in Mediterranean mussels (<i>Mytilus galloprovincialis</i>) from the Ariake Sea, Sendai Bay, Tokyo Bay, and Toyama Bay, Japan.	Invertebrate (bivalve)	2000, 2002-2008, 2010, 2014, 2016, 2020-2022	IPCC Region: E. Asia, Country: Japan	https://tenbou.nies.go.jp/gisplus/download.html
Global Biotic Mercury Synthesis (GBMS) database	Hg concentrations in Mytilidae spp. and Mangrove cupped oysters (<i>Crassostrea rhizophorae</i>) from area of oil	Invertebrate (bivalve)	2004	IPCC Region: N.E. South America, Country: Brazil	Lacerda, L. D., Campos, R. C., & Santelli, R. E. (2013). Metals in water, sediments, and biota of an offshore oil exploration area in the Potiguar Basin, Northeastern Brazil. Environmental monitoring and assessment, 185(5), 4427-4447.

	exploitation in Brazil				
Global Biotic Mercury Synthesis (GBMS) database	Hg concentrations in blue mussels from the Bay of Fundy, Gulf of Maine.	Invertebrate (bivalve)	2001, 2003	IPCC Region: E. North America, Country: Canada	Harding, G., Dalziel, J., & Vass, P. (2018). Bioaccumulation of methylmercury within the marine food web of the outer Bay of Fundy, Gulf of Maine. PloS one, 13(7), e0197220.
Global Biotic Mercury Synthesis (GBMS) database	Hg concentrations in blue mussels (<i>Mytilus edulis</i>) and Pacific oyster (<i>Crassostrea gigas</i>) from the Bohai Sea, China	Invertebrate (bivalve)	2013	IPCC Region: E. Asia, Country: China	Meng, M., Sun, R. Y., Liu, H. W., Yu, B., Yin, Y. G., Hu, L. G., ... & Jiang, G. B. (2020). Mercury isotope variations within the marine food web of Chinese Bohai Sea: Implications for mercury sources and biogeochemical cycling. Journal of hazardous materials, 384, 121379.

Table S3 . List of observational data submitted for the assessment, including regions and countries, species, tissues, years of collection, range of individual THg concentrations ($\mu\text{g/g ww}$), sample size, and means submitted for **marine mammals**. (Excel file)

2.5 Humans

2.5.1 What data are available?

The data submitted through the online data repository managed by BRI include 16 data sets of biomonitoring data from nine countries: Canada, Japan, Brazil (Amazon region), Lithuania, Arctic region (Canada, Faroe Islands, Iceland, Greenland, United States, Sweden, Russia). In total, 21 intend to submit forms have been received, and the countries that intended to but have not yet submitted their data are: Peruvian Amazon, Palau, Switzerland, Norway, Zimbabwe, Lithuania, and Japan. The data from the received datasets represent the period from 2000 to 2023. The data collected is described in **Annex II**. In summary, most of the data represent Indigenous and other Arctic populations, and Indigenous populations from the Amazon region. Two datasets received from Japan and Lithuania represent national human biomonitoring programs.

In addition to data submission through the OESG template, a systematic literature search was conducted to collect all publicly available data on mercury exposure in various human populations across the globe published from 2018 onwards, to complement data collected for the Global Mercury Assessment 2018 by Basu et al.^[1] Adopting the methodological approach described by Basu et al., the Web of Science database was screened for publications using the following keywords: (ALL=("mercury") OR ALL=("methylmercury") OR ALL=("methyl" AND "mercury") OR ALL=("MeHg")) AND (ALL=("blood") OR ALL=("hair") OR ALL=("urine")). The search resulted in 3.288 records, after selecting only for articles and proceedings papers, we got 2.980 papers. First, the titles were screened and after excluding non-human, in-vitro or cell lines research, analytical or modelling papers, papers dealing with poisoning events and blood pressure related studies (Hg-containing devices) and national HBMs (collected separately), 1.153 papers remained, which were further screened by reading through the abstracts. After excluding the papers following the same criteria as described above for title screening, we got 652 papers for the full review. From these, 273 were found eligible as cross-sectional studies. One additional study was later identified by one of the OESG group members and six more through the full papers review, so in total **280 papers** were eligible and were included in the database. We excluded the papers that contained data collected before the year 2000, had a sample size <50, used inappropriate biomarker for Hg exposure, did not report measures of central tendency, or could not be retrieved, as well as papers whose data was already reported in the OESG template or included in GMA2018. Papers reporting data for **50 birth cohorts** were identified and were included in a separate database. The flow diagram of the search is presented in Figure 1. To summarize the literature review, the data from the papers represent **62 countries**, and cover the entire target time period – from 2000 to 2024.

[1] Basu N., Horvat M., Evers D.C., Zastenskaya I., Weihe P., Tempowski J. A State-of-the Science Review of Biomarkers in Human Populations Worldwide between 2000 and 2018. *Environmental Health Perspectives* 126 (10), 2018.

Within the 280 papers on cross-sectional studies we've identified 589 population groups (with at least one biomarker), which were either not stratified or stratified by geographical region, age (life stage), type of environmental (contaminated vs. non-contaminated; industrial vs. rural, ...) or year. There were 324 population groups classified as general (background) population, 104 as Indigenous (40 as Arctic and 64 as other), 55 as fish consumers, 42 living on contaminated sites (other than ASGM) and 41 in ASGM sites with direct exposure. There was only one study/population on dental workers, while 18 study populations on occupation exposure other than dental. Only three eligible studies were found on the Hg exposure in relation to cosmetics use. Most of the study groups from cross-sectional studies had Hg measured in whole blood (n=286), followed by hair (n=249), urine (n=144), and cord blood Hg measurement (n=16). Methyl Hg was measured in addition to total Hg in only 5% of studies.

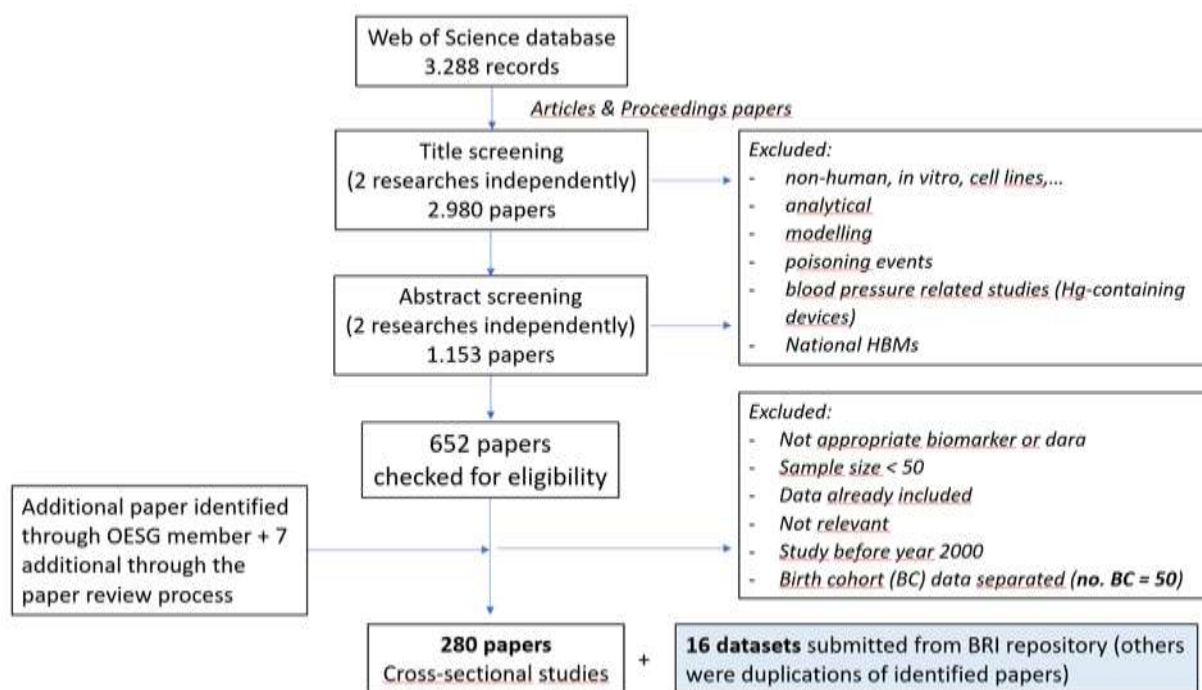


Figure 1. Flow diagram of literature search for human biomonitoring data indicating the number of papers that were identified, screened, and included in the metadata file.

Data collections for the national HBM programs were also obtained from different sources: 1) reports (United States), 2) publications (France, Republic of Korea, Spain), 3) Canadian Biomonitoring Dashboard (Canada), and 4) European HBM dashboard (Belgium, Czech Republic, Germany). For Slovenia, the data on Hg have not been submitted through the OESG repository, as the laboratory analysis was completed just recently, and the national reporting is in progress. The data of the national programs span years from 2007 to 2024.

The data from publications listed in the Annex II will be combined with the data identified and used in the last assessment by Basu et al. (2018) to form a common database for the current assessment within the Minamata Convention, which will be particularly useful to evaluate time trends and patterns for the general population (background exposure) from nationally representative data and also in the most vulnerable population groups, e.g. Indigenous Arctic populations.

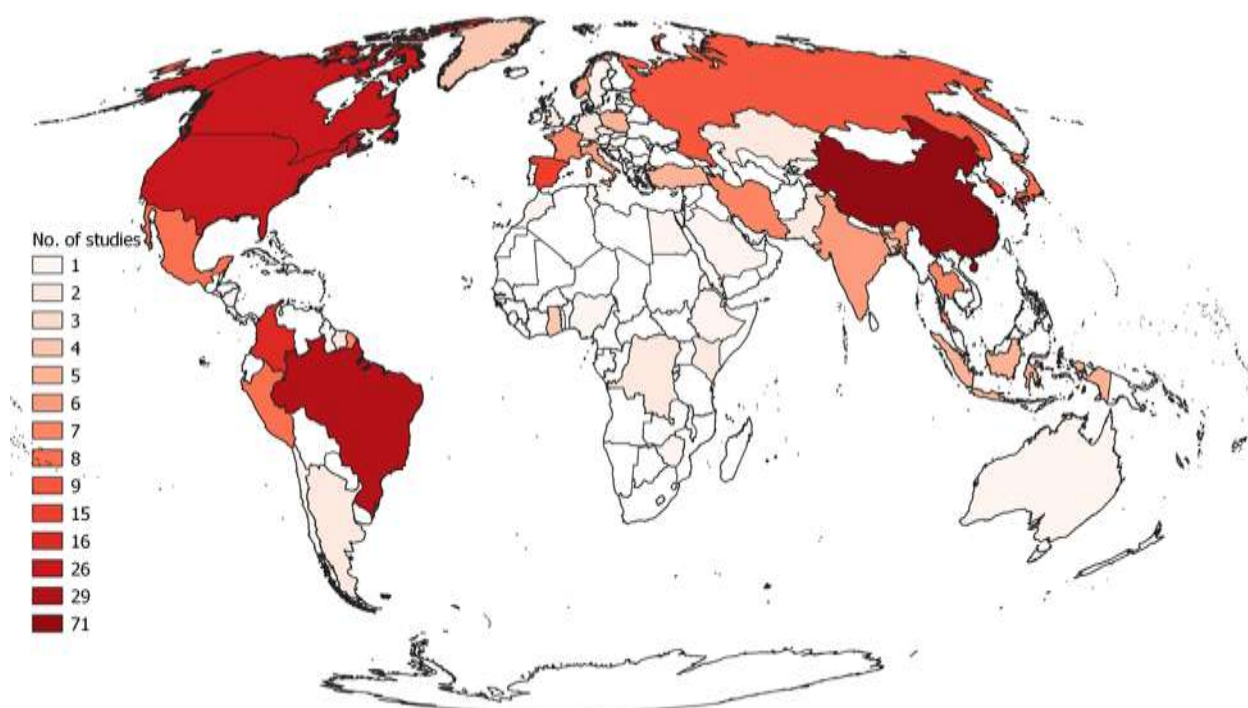


Figure 2. Number of studies or birth cohorts by countries. The map includes human biomonitoring data obtained through the OESG repository and literature search.

2.5.2 What are current levels of THg observed in (blood, urine, hair) of (vulnerable sub-populations including Indigenous Peoples, highly exposed population (occupationally exposed at ASGM, non occupationally exposed at ASGM, use of skin lightening creams) and the general population -) that are a) remote from anthropogenic sources (i.e. indigenous vs others) and b) at anthropogenic sources (highly exposed population; iHg, orgHg exposure)?

Information Used (including applicable Indigenous and traditional knowledge):

We used all available data described in the previous chapter: data from national HBM programs, data submitted through the OESG repository and data obtained through literature search.

Methodology:

For the purpose of the research question, the data of cross-sectional studies were classified according to the *type of population* as follows:

- 1) The general population (background exposure)
- 2) Fish consumers (further classified into consumers of: Inland fish, Marine fish, Marine mammals, Multiple)
- 3) Indigenous Peoples from Arctic regions
- 4) Indigenous communities from non-Arctic regions (further classified into riverside South American, First Nations, and Other)
- 5) ASGM communities with direct exposure to Hg vapour
- 6) Contaminated sites
- 7) Occupational exposure – dental workers
- 8) Occupational exposure – other
- 9) Skin-lightening cream users

Each population type was assigned a *group* based on the type of exposure:

- 1) BKGD: The general population (background exposure)
- 2) DIET: dietary exposure sources (Indigenous Peoples from Arctic regions, Indigenous non-Arctic communities away from direct exposure to Hg vapour (absence of ASGM practice), Fish consumers – includes non-indigenous or mixed populations)
- 3) POINT: direct exposure through Hg vapour in the ASGM areas, contaminated areas, and occupational exposure
- 4) FETUS: fetal exposure (regardless of the population type)
- 5) COSMETICS: skin-lightening cream users

For each dataset the following information was collected either through the harmonized data template or from the scientific publications: country of the study, WHO and UN region, type of population (see above), year(s) of sampling, sex, age range, life stage (infant, toddler, child, teenager, adult), main source of Hg exposure (fish-marine, fish-inland, marine mammals, seafood other than fish, rice, mining-ASGM, mining-other, industry-non-mining, amalgam, cosmetics, multiple), primary source of Hg exposure (e.g. ASGM mining in fish-consuming riverside areas contaminated from ASGM), biomarker (whole blood, urine, hair, cord blood), Hg measured (Total Hg, Methyl Hg, Hg₂, Hg₀), unit of reported Hg concentrations, sample preparation method, detection method, QA/QC, central tendency metrics (median, geometric mean or mean), and upper bound (P75, P90, P95, or Max).

For the purpose of harmonizing the reported levels on Hg exposure, in cases where whole blood Hg was not measured, but hair Hg was available, the latter were converted into whole blood levels (*blood-equivalent levels*) by dividing the hair Hg concentration in ng/g by 250 to obtain whole blood concentration in µg/L, but only in cases where no external contamination was expected. Although the ratio is known to be highly variable among individuals, it allows us to compare the levels among all population groups. The transformation was not applied in cases where exposure was known to occur via inhalation of mercury vapor. For transformation from cord blood to maternal whole blood levels, the concentration in cord blood was divided by 1.7.

Answer/Response:

The following plot (Fig. 3) shows median whole blood Hg levels by the population types described in the Methodology section above. Fish-consuming populations and Arctic Indigenous Peoples were further stratified by their main dietary source, while non-Arctic Indigenous populations were stratified by geographic region.

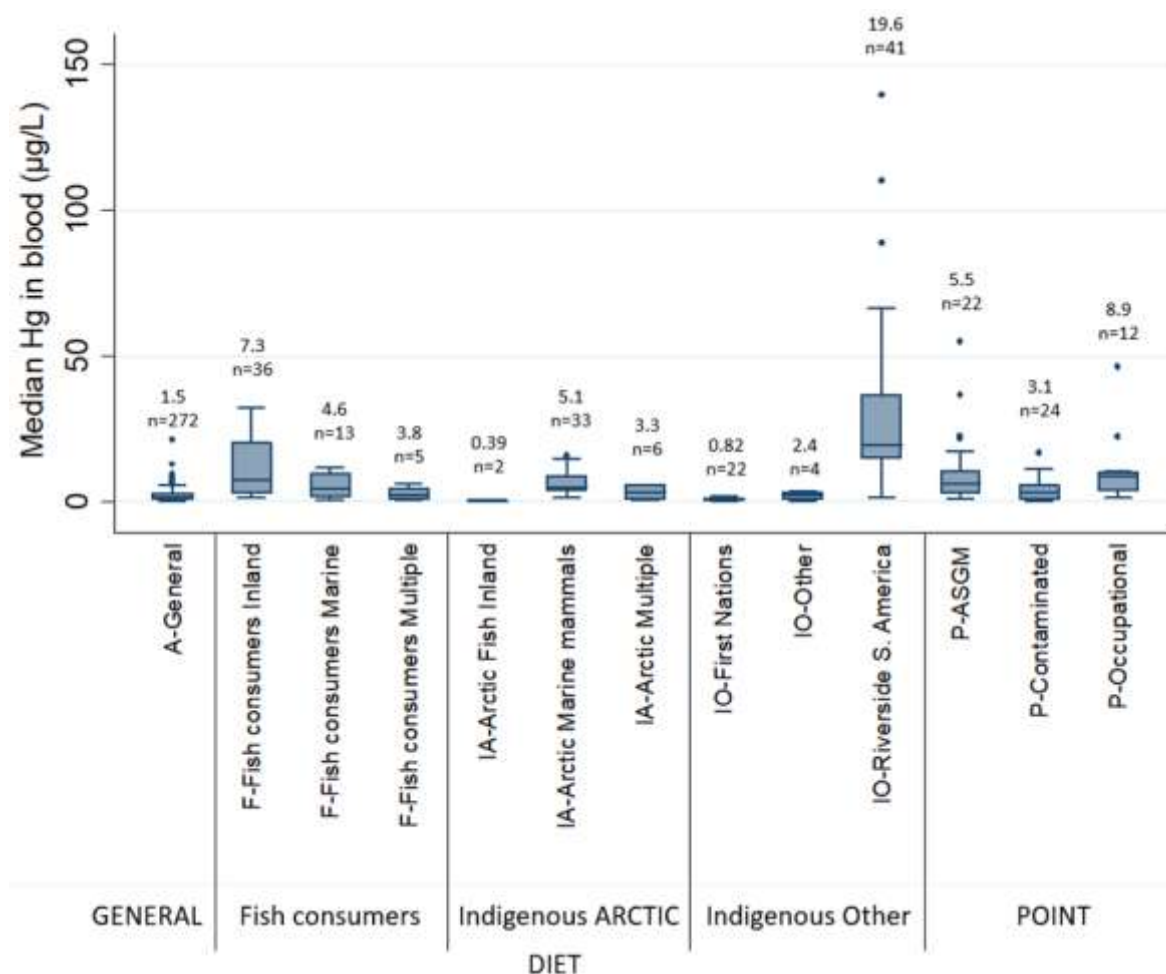


Figure 3. Distribution of median Hg blood or blood-equivalent levels across different population groups (n) identified from Cross-sectional studies stratified by main groups of exposure: GENERAL (Background exposure), DIET (dietary sources of exposure, further classified into Fish consumers, Indigenous Arctic and other populations), POINT (point sources of exposure: ASGM, contaminated sites and occupational exposure). DIET group: in cases hair or cord blood Hg levels were the only biomarkers available, whole blood Hg was derived using a conversion factor of 250 and 1.7, respectively.

Notes:

IO-Riverside S. America includes data from Brazil, Peru, Colombia, Suriname, Guyana

IO-Other includes data from China, Kenya, New Zealand

IA-Arctic Marine mammals Include also one population from Faroese Islands

Among non-occupationally exposed populations, the highest Hg levels were observed for riverside Indigenous communities from South America (Median = 19.6 µg/L; data from Brazil, Peru, Colombia, Suriname, Guyana). They were followed by fish-consuming populations whose main source of exposure is freshwater (inland) fish (Median = 7.3 µg/L), ASGM communities with direct exposure to Hg vapour (Median = 5.5 µg/L), and Arctic Indigenous Peoples consuming marine mammals (Median = 5.1 µg/L). A strong link to gold mining contamination was evident in the South American populations. Among riverside South American Indigenous communities, **86 % of study populations were reported to live on ASGM-contaminated sites**. Furthermore, fish consumers whose main source of Hg exposure were freshwater (inland) fish were in majority from South American countries, and **61 %** of these population groups were reported to reside in ASGM-contaminated areas (Fig. 4).

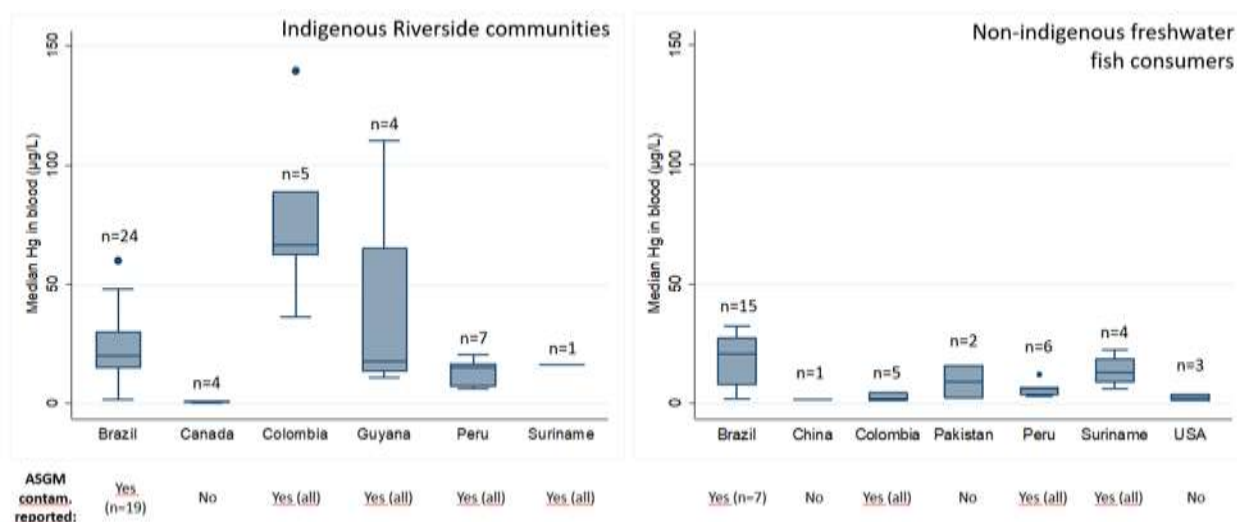


Figure 4. Distribution of median blood or blood-equivalent levels Hg levels in **Indigenous** and **non-indigenous consumers of freshwater (inland) fish**, by country. Note: 'n' refers to the number of distinct population groups. For each country, we indicated whether contamination from ASGM activities was identified in the paper/data source.

Overall, the highest exposure levels were observed among Indigenous communities from Colombia, all of which were affected by ASGM mining. However, the levels shown in the figure should be interpreted with caution, as for most countries (except Brazil), the estimates are based on only a small number of studies, and some report arithmetic means, which are not appropriate for log-normally distributed biomarker data. Moreover, for many population groups, where the means/medians are reported for different villages, the sample size is often below 50.

Figure 5 shows median hair Hg levels in freshwater fish consuming communities, either Indigenous or non-Indigenous from non-Arctic regions, stratified by *reported* ASGM contamination in the area. In cases of ASGM contamination, no direct exposure to Hg vapour was expected in these populations, but this can not be completely excluded.

Median Hg levels in Indigenous riverside communities being affected by ASGM contamination were 12 times higher than those of the other riverside Indigenous communities.

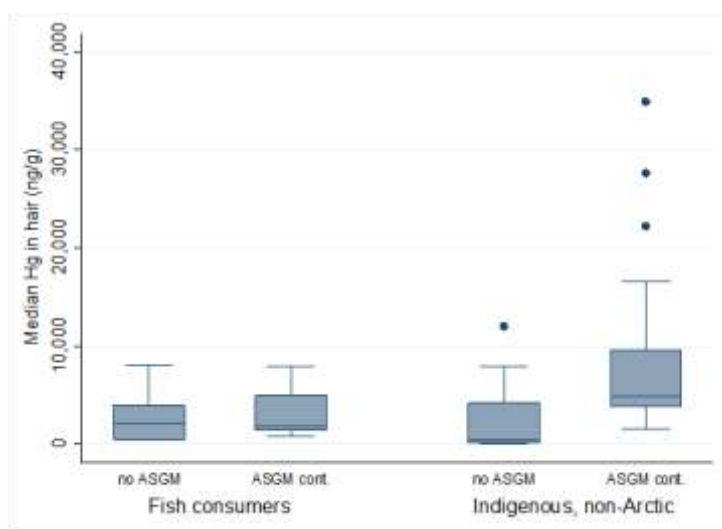


Figure 5. Distribution of median hair Hg levels in **non-indigenous (fish consumers)** and **Indigenous riverside populations**, by absence or presence of ASGM contamination, as identified in the reviewed publications.

Among the ASGM communities with direct exposure to Hg vapours (important: dietary exposure via contaminated fish consumption is not excluded and may also contribute), most studies (12) were conducted in Colombia with 18 distinct population groups and median level of 6.4 µg/L in whole blood and 7.5 µg/L in urine (Fig. 6). Maximum levels were also reported for Colombia, namely 22.7 µg/L in whole blood and 26.9 µg/L in urine.

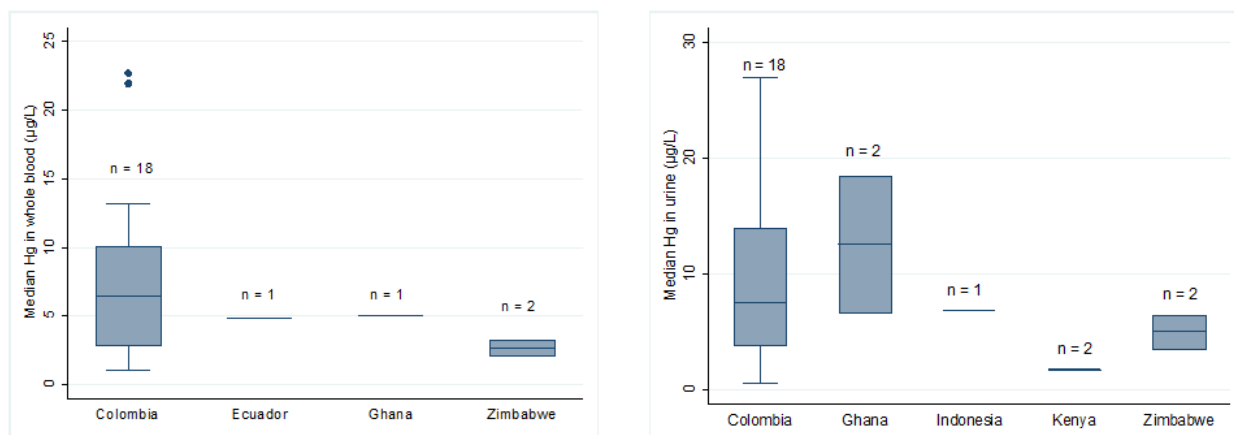


Figure 6. Distribution of median whole blood and urine Hg levels in **ASGM communities**, by country. Note: ‘n’ refers to the number of distinct population groups.

Based on the above observations, in Table 1 we present all cross-sectional studies of Hg data that are relevant to ASGM activities: direct exposure through inhalation of Hg vapour, dietary exposure in Indigenous communities and dietary exposure in non-indigenous fish-consuming communities. According to Table 1, the most exposed on account to ASGM are Indigenous fish-consuming communities.

Table 1. Total Hg concentrations in urine, whole blood and hair in different ASGM-related communities.

Biomarker	ASGM – direct exposure*	n	ASGM – dietary exposure <u>Indigenous</u>	n	ASGM – dietary exposure <u>non-Indigenous</u>	n
Urine THg (µg/L)	6.6	25	n.a.	/	n.a.	/
Blood THg (µg/L)	5.5	22	7.4	1	2.5	6
Hair THg (ng/g)	1700	37	4900	36	1875	18

*Dietary exposure also possible

These findings show that the highest Hg non-occupational exposures from the current literature review were associated with freshwater (inland) fish consumption and among the consumers the levels were the highest in Indigenous communities living in ASGM-contaminated areas.

In the following section (Fig. 7) we have looked into the Hg exposure levels for fish (and other seafood) consumers, who reside in the areas (presumably) free from ASGM contamination. Despite the latter, the highest median and maximum levels were still observed for freshwater (inland) fish consumers, which mostly represent the South American riverside communities (see Fig. 4). Second largest median levels were observed for consumers of Marine mammals, which represent populations from the Arctic region (Indigenous Peoples from Canada and Greenland, and Faroese population). *Exposure from marine fish consumption will be analysed further in the next version of the report.*

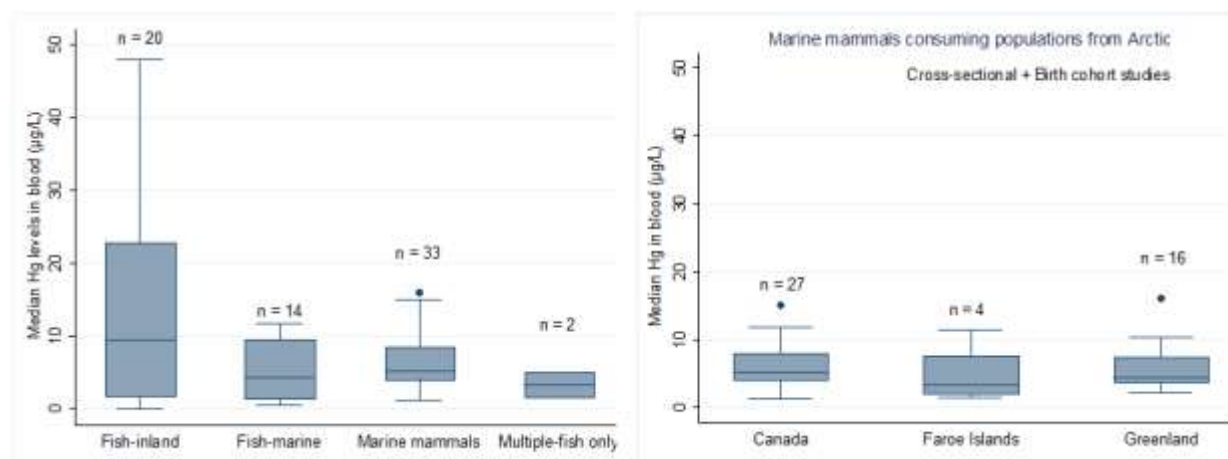


Figure 7. Distribution of median blood or blood-equivalent levels Hg levels in Fish (or other seafood) consuming communities by main source of Hg (population groups with known ASGM contamination excluded). Note: 'n' refers to the number of distinct population groups. Plot on the right also includes data from Birth cohort studies to ensure representativeness from all Arctic populations.

In addition to data from cross-sectional studies, we also collected data from nationally representative HBM surveys. The Japanese national program was conducted for the first time in 2021-23 and shows markedly higher Hg exposure levels than other national data (Fig. 8). The nationally representative Japanese median is consistent with levels observed in fish-consuming populations identified in the literature, when restricted to adults (Median Hg in blood = 5.6 µg/L), and likely reflects exposure primarily through seafood consumption.

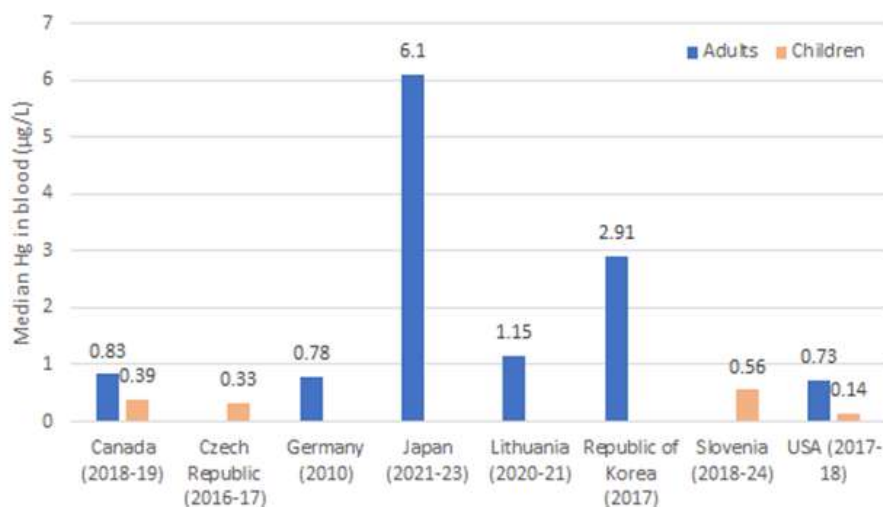


Figure 8. Median Hg levels in whole blood from the most recent available national HBM surveys.

Table 1 lists all study populations identified in the literature search that were classified as occupationally exposed. The highest exposure was reported for Hg miners from Mexico, with median urinary Hg levels of 550 µg/g creatinine. Very high exposures were also observed among workers in thermometer and lamp factories in China, with median urinary levels of 150 µg/g crt. These were followed by chlorine production workers (median urine Hg = 40.7 µg/g crt) and fluorescent lamp production workers (median urine Hg = 20 µg/g crt). Other exposures were related to lead and tin smelting in India and China, for whom whole-blood mercury concentrations around 10 µg/L were observed.

Table 1. Hg exposure data for occupational exposure, presented by different activities as obtained from the literature review.

Country	Specific population	SEX	YEAR	Biomarker (Total Hg)	UNIT	Mean, median or GM	Type	n
India	Pb smelting workers	M	2003-04	Whole blood	µg/L	9.60	Mean	127
India	Pb smelting executives	M	2003-04	Whole blood	µg/L	8.92	Mean	121
China	Pb-Zn industry	Both	2022-23	Whole blood	µg/L	4.18	GM	401
China	fluorescent lamp production workers	Both	2017-20	Urine	µg/g crt	19.7	Median	530
China	thermometer and lamp factory: chronic Hg exposure	Both	2016	Urine	µg/g crt	2.13	Median	40
China	thermometer and lamp factory: absorption group	Both	2016	Urine	µg/g crt	150	Median	40
China	Tin Smelting Workers: comprehensive work	Both	2019	Whole blood	µg/L	10.4	GM	62
China	Tin Smelting Workers: maintenance	Both	2019	Whole blood	µg/L	10.3	GM	34
China	Tin Smelting Workers: primary smelting	Both	2019	Whole blood	µg/L	9.04	GM	73
China	Tin Smelting Workers: refining	Both	2019	Whole blood	µg/L	7.99	GM	49
Mexico	Mercury miners (mine "La Estrella")	M	2019	Urine	µg/g crt	551	Median	34
Poland	Chlorine production factory workers	M	2016-17	Urine	µg/g crt	40.7	Median	38
Poland	chloralkali workers	M	2015	Whole blood	µg/L	6.06	Median	131
Poland	chloralkali workers	M	2015	Urine	µg/g crt	11.8	Median	131

Only 3 studies were identified that assessed Hg exposure associated with the use of skin-lightening creams. Of these, only one (conducted more than 20 years ago) measured Hg in whole blood and urine (Table 3). For the other two studies (one conducted in 2017 and one for which the study period was not reported), only hair Hg levels were available, which do not reliably reflect exposure from skin-lightening product use.

Table 2. Hg exposure data for users of skin-lightening creams.

Country	Specific population	Year	Biomarker (Total Hg)	Unit	Mean	n
China	Users of a beauty cream cosmetic who contacted the Department of Health following a public announcement	2002	Whole blood	µg/L	17.1	183
China	Users of a beauty cream cosmetic who contacted the Department of Health following a public announcement	2002	Urine	µg/L	45.2	282

Consistency of Answer with Existing Literature (especially the 2018 GMA):

The median blood Hg levels stratified by basic population groups and exposure sources can be compared to the levels presented in the latest Hg exposure assessment by Basu et al (2018) (Fig 9). In general, some reduction in exposure can be observed, particularly for POINT and DIETARY exposure sources. Particularly, the exposure in Indigenous Peoples and ASGM communities seems to be somewhat lower, however, the [non-Arctic Indigenous](#)

communities still show high dietary exposure, which seems to be to a large extent a consequence of ASGM activities in South American countries, as described above (Figures 4 and 5). Consequently also the exposure stratified by WHO regions changed from the last assessment and is now the highest in ‘Americas’ region, on account of high levels in South American countries where ASGM is taking place.

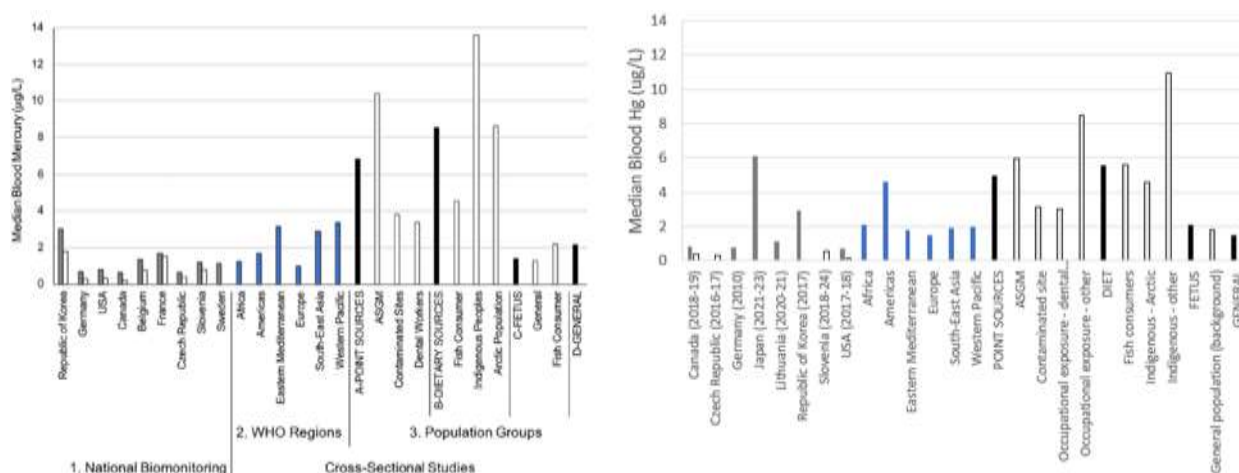


Figure 9. Comparison of pooled median blood or blood-equivalent levels by basic population type between assessment in 2018 (Basu et al, 2018, left) and current assessment (right).

The levels obtained through national HBM biomonitoring programs have not changed significantly from the last assessment (Fig. 9), except for the new nationally-representative data available for Japan, which shows markedly higher levels than other national data available (Fig. 8).

Evaluation of Confidence:

Several limitations can be noted, particularly for the highly exposed populations in remote areas:

- The data often relies on small sample size;
- In many cases arithmetic mean levels are reported instead of geometric mean or medians, which may overestimate the exposure estimates. Arithmetic mean is not appropriate in log-normally distributed data, as is the case for exposures to toxic elements;
- In ASGM settings, total mercury in hair may not be an optimal biomarker, because hair can be externally contaminated by airborne mercury in addition to reflecting dietary methylmercury uptake. In the absence of mercury speciation, it is therefore difficult to distinguish externally deposited from internally incorporated mercury.

In order to increase the confidence of the data, we are currently:

- contacting the publication authors to provide us median or geometric mean levels for their study populations;
- checking the ASGM-related data with local collaborators in the OESG group, to distinguish with higher confidence between populations with direct exposure to Hg from ASGM activities and those that are exposed indirectly through fish consumption.

Information and Capacity Needs:

To be added in the next version.

2.5.3 How have levels of THg observed in (vulnerable subpopulations including Indigenous Peoples, general population) changed over the available record? Is there a change in the observed trends over the available record and how do those changes in trends compare to the timeline of the Minamata Convention? Locally, Regionally, Globally

Information Used (including applicable Indigenous and traditional knowledge):

For the purpose of this draft, we only used:

- 1) Nationally-representative HBM data (national HBM programs), where monitoring was repeated over time.
- 2) Regional data, repeated over time for the same population group (Nunavik region and central Sweden).

Methodology:

At this stage, the analysis is limited to visualization of mean mercury levels by year of study conduct, and therefore only includes datasets with repeated measurements over time.

Answer/Response:

The trends in total mercury concentrations in the adult population appear broadly stable after the year 2010. For example, Canada's biomonitoring fact sheet on mercury reports that total mercury levels remained essentially unchanged between 2007–2009 (the first CHMS data collection period) and 2018–2019 (the most recent period for which data have been reported). Similar patterns are observed in other HBM surveys.

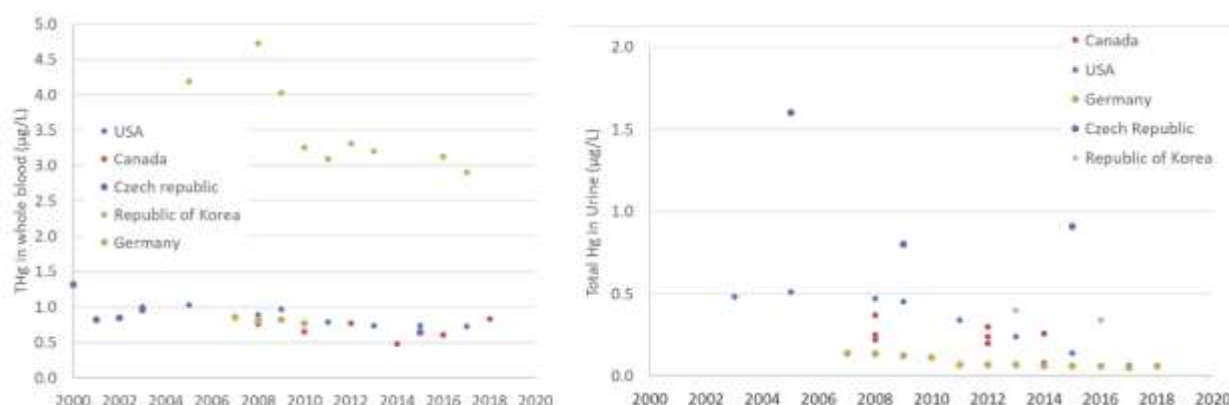


Figure 10. Temporal data from available national HBM programs (ADULTS) by years of sampling.

Regional temporal data is available for pregnant Indigenous women from the Arctic region of Nunavik and showed marked decrease in Hg exposure from year 2000 (Fig. 11, left). Gradual decrease can be observed also at lower levels of exposure, such as in the general population of Swedish primiparous women (Fig. 11, right).

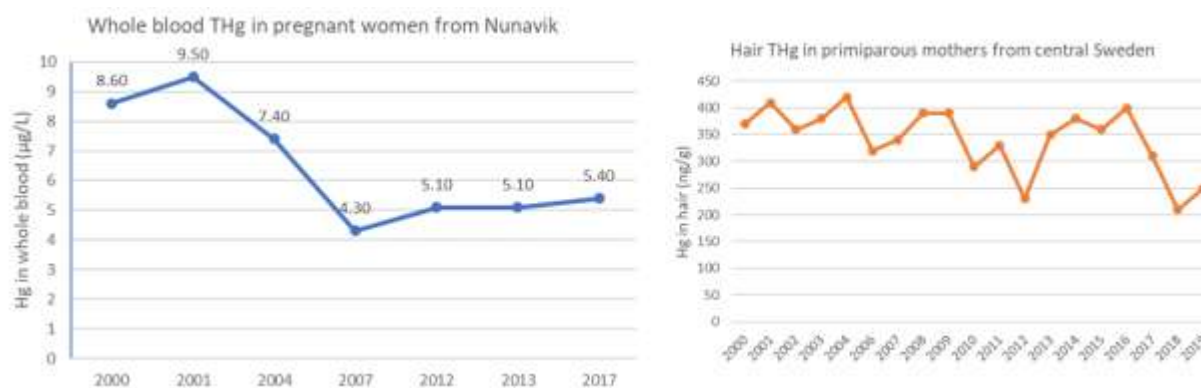


Figure 11. Regional temporal data.

Although national and regional human biomonitoring data generally indicate a decline or stabilization in mercury exposure, a more comprehensive analysis of time trends will be conducted using all available cross-sectional data to provide a more complete assessment of temporal patterns across different population groups and regions.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

The time trends using nationally representative data have been re-produced from GMA 2018. The next version of the report will include a more comprehensive analysis of time trends using all available cross-sectional data across the globe.

Evaluation of Confidence:

To be added in the next version.

Information and Capacity Needs:

To be added in the next version.

2.5.4 How do current levels of THg observed in (blood, urine, hair) of (vulnerable sub-populations and the general population) vary geographically at the global scale?

Information Used (including applicable Indigenous and traditional knowledge):

We used all available data described in the previous chapter: data from national HBM programs, data submitted through the OESG repository and data obtained through literature search.

Methodology:

For each dataset, the country of the study, the WHO and UN region were collected along with other information. Each study was located as accurately as possible based on the information provided by data providers (OESG template) or authors of the scientific publications. These locations were either wider regions within countries, towns, or villages. The location will be used to visualise the collected Hg levels on a global map, stratified by population types/exposure sources. Each study population was classified also according to the place of living (coastal - Atlantic, Pacific, Mediterranean, Arctic, other; inland - general; inland - river or lake).

Answer/Response:

In addition to Fig. 8, which shows median blood Hg levels by WHO regions, we stratified the levels also according to the UN regions. Among the UN Regions Latin America and Caribbean show almost 6-time higher exposure than other regions, which is on account of high levels observed in riverside communities of South America, largely affected by ASGM activities as described in the previous chapters.

Across the general (background) population, pooled median Hg concentrations in blood, urine, and hair were typically low, however, some variability can be observed according to the (geographical) place of living (Fig. 12). The median blood and blood-equivalent levels were higher in the populations residing in coastal regions than inland, which is most plausibly driven by dietary habits, particularly seafood and fish consumption.

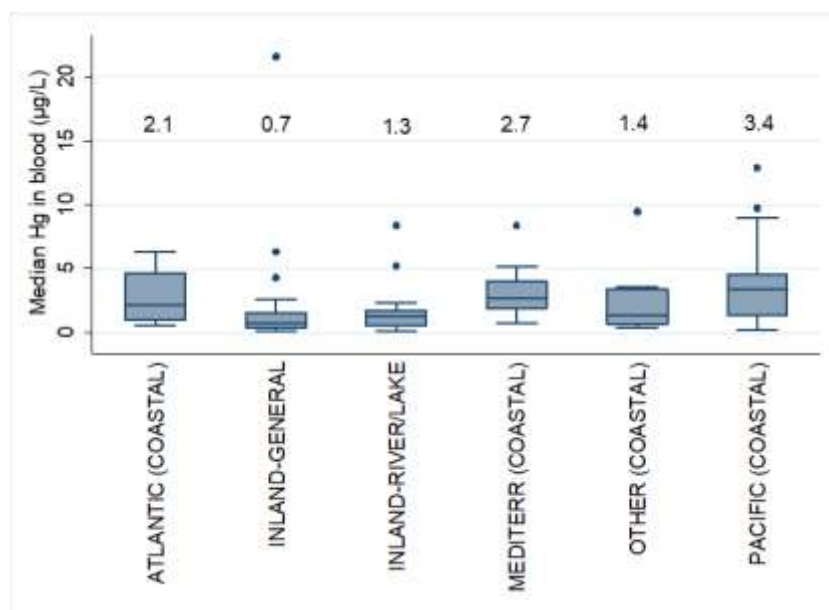


Figure 12. Distribution of median Hg blood or blood-equivalent levels in the general (background) population by place of living. Medians of pooled data are indicated on the graph.

Consistency of Answer with Existing Literature (especially the 2018 GMA):

To be added in the next version.

Evaluation of Confidence:

To be added in the next version.

Information and Capacity Needs:

To be added in the next version.

2.5.5 How do current levels of THg observed in (blood, urine, hair) of (vulnerable sub-populations and the general population) compare to established national and international guidance levels associated with adverse effects on human health? Locally, Regionally, Globally

Information Used (including applicable Indigenous and traditional knowledge): n.a.

Methodology: n.a.

Answer/Response: n.a.

Consistency of Answer with Existing Literature (especially the 2018 GMA): n.a.

Evaluation of Confidence: n.a.

Information and Capacity Needs:

In order to be able to answer this question, we need internationally-established guidance values. In addition, reporting of percentages of population above the available reference values/ health-based guidance level is needed.

2.5.6 How do the changes in the levels of THg observed in (blood, urine, hair) of (vulnerable sub-populations and the general population) between circa 2010 and the present compared to established national and international benchmark levels associated with adverse effects on human health? Locally, Regionally, Globally

Information Used (including applicable Indigenous and traditional knowledge): n.a.

Methodology: n.a.

Answer/Response: n.a.

Consistency of Answer with Existing Literature (especially the 2018 GMA): n.a.

Evaluation of Confidence: n.a.

Information and Capacity Needs:

In order to be able to answer this question, we need internationally-established guidance values. In addition, reporting of percentages of population above the available reference values/ health-based guidance level is needed.

2.5.7 How are the changes in the levels of THg observed in (blood, urine, hair) of (vulnerable sub-populations and the general population) attributable to the mitigation measures influenced by the Convention? Locally, Regionally, Globally

Information Used (including applicable Indigenous and traditional knowledge):

Although in many regions and in nationally representative studies we are observing declining trends in Hg exposure, we have no evidence that the decline can be directly attributed to the mitigation measures influenced by the Convention. However, there have been case studies conducted regionally or locally that demonstrate a decline in exposure following certain measures or interventions. The three cases that can be used to answer the question are the following:

1) Dental amalgam phase-out for children < 15 years (Slovenia)

Urinary mercury concentrations from four human biomonitoring (HBM) studies were analysed to assess the impact of dental amalgam phase-out measures on mercury exposure in Slovenian children aged 6–11 years. These studies included the EU 6th Framework project *Public Health Impact of Mixed Element Exposure in Susceptible Populations* (PHIME; 2007–2008), the EU Life+ project *Demonstration of a Study to Coordinate and Perform Human Biomonitoring on a European Scale* (DEMOCOPHES; 2011–2012), the EU 7th Framework project *European Coordination Action on Human Biomonitoring* (CROME; 2016), and the *Second National Human Biomonitoring Program in Slovenia* (SLO-HBM-II; 2018–2024). Children originated from 13 geographically distinct regions across the country, encompassing urban, rural, and potentially contaminated areas. The latter included the former Hg-mining town of Idrija and other regions affected by industrial activities. Parents completed standardized questionnaires in all studies. For this analysis, data on the presence (yes/no) and number of amalgam fillings, frequency of fish consumption (<1 per month vs. ≥1 per month), and basic child characteristics (age, sex, body mass index, BMI) were used. U-Hg levels were analysed both as unadjusted concentrations (ng mL⁻¹) and creatinine-adjusted values (µg g⁻¹ creatinine). Differences between cohorts were assessed using non-parametric tests and regression models adjusting for age, sex, BMI, and residence area type. Associations between U-Hg

levels and the presence/number of dental amalgams were examined using correlation analyses in both pooled and cohort-specific data. Temporal trends were evaluated with mixed-effects models that accounted for clustering by town and included key predictors such as amalgam status, fish consumption, urinary creatinine, residence area type and child characteristics. Interactions of year with amalgam and fish consumption were added to test hypothesized changes in contribution of exposure sources over time.

Source: Usenik, V., Alilović Osolin, A., Snoj Tratnik, J. et al. Human biomonitoring in support of the Minamata Convention: a case of phasing out dental amalgam. *Environ Health* (2026). <https://doi.org/10.1186/s12940-025-01255-7>

2) Dietary recommendations/interventions in the Arctic

There are various dietary recommendations and interventions applied across Arctic regions, especially among Indigenous Peoples, which are all aimed at protecting health with encouraging the consumption of healthy traditional foods. Authorities prioritise trusted, locally-tailored communication that pairs the benefits of traditional diets (nutrition, part of cultural and spiritual identity, food security) with clear guidance on limiting high-contaminant foods and choosing safer alternatives (Krümmel et al., 2024). Traditional diets differ across the Arctic: marine mammals are central for many Indigenous Peoples in Alaska, Canada, Greenland, and far-eastern Russia, whereas terrestrial game is more prominent in Norway, Finland, Sweden, and western central Arctic Russia, while fish is a major component for both Indigenous and non-Indigenous populations in many Arctic areas (AMAP, 2021). Although most Arctic dietary advisories are largely mercury-centred, they are in a few cases multi-contaminant in scope, incorporating PCBs, dioxins, and other pollutants when local evidence indicates risk. Dietary advice is commonly life-stage specific (especially for pregnant/breastfeeding women, women planning pregnancy, and children), location and species-specific (emphasising mercury in predatory fish and, where relevant, PCBs/dioxins and other pollutants in fish/game/sea mammals, etc.), and increasingly evidence-based via biomonitoring (e.g., Alaska's hair Hg program) (Krümmel et al., 2024; Arnold et al., 2005). It is often communicated through practical tools, such as serving limits, "Best/Good/Avoid" lists, point systems, location-specific notices, etc., delivered by trusted sources (State of Alaska, 2020; Government of Canada, 2019; Government of Greenland, 2020; Bjerregaard & Mulvad, 2012; AMAP, 2015; Krümmel & Gilman, 2016; Icelandic Directorate of Health, 2018; Faroese Food and Veterinary Authority, 2011; Finnish Food Authority, 2019). Nevertheless, important gaps remain in the formal evaluation of the success of risk communication strategies, as well as balancing risk reduction with cultural identity. Long-term success depends on targeted communication from trusted sources, developed in an equitable partnership approach with communities, and the availability of healthy food alternatives. However, it was found that communication of contaminant risk is a complex undertaking and not a sustainable solution to ensure low levels of contaminants in human populations. Regulations to lower contaminant levels in the environment are thus crucial. (Krümmel et al., 2024).

Sources:

AMAP, 2021. AMAP Assessment 2021: Human Health in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Tromsø, Norway. x+240pp

AMAP, 2015. AMAP Assessment 2015: Human Health in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. vii + 165 pp.

Arnold SM, Lynn TV, Verbrugge LA, Middaugh JP. Human biomonitoring to optimize fish consumption advice: reducing uncertainty when evaluating benefits and risks. *Am J Public Health*. 2005; 95(3): 393-397.

Bjerregaard P, Mulvad G. The best of two worlds: how the Greenland board of nutrition has handled conflicting evidence about diet and health. *Int J Circumpolar Health*. 2012; 71(1): 18588.

Faroese Food and Veterinary Authority. Dietary recommendation on the consumption of pilot whale meat and blubber. J No: 461-200800444-70. 2011 [cited 2025 Oct 14]. Available from: <https://hfs.fo/webcenter/ShowProperty?nodeId=%2Fhfs2cs%2FHFS053891%2F%2FidcPrimaryFile&revision=latestreleased>

Finnish Food Authority. Safe use of fish. Date modified: 7 February 2019. 2019 [cited 2025 Oct 13]. Available from: <https://www.ruokavirasto.fi/en/foodstuffs/instructions-for-consumers/safe-use-of-foodstuffs/safe-use-of-fish/>

Government of Canada. Mercury in fish. Consumption advice: making informed choices about fish. Date Modified: 27 July 2024. 2024 [cited 2025 Oct 13]. Available from: <https://www.canada.ca/en/health-canada/services/food-nutrition/food-safety/chemical-contaminants/environmental-contaminants/mercury/mercury-fish.html>

Government of Greenland. De 10 kostråd [The 10 dietary tips]. 2020 [cited 2025 Oct 13]. Available from: https://peqgik.gl/Emner/Livsstil/Kost/GodeRaadOmKost?sc_%20lang=da-DK

Icelandic Directorate of Health. Nutrition - recommendations from the Directorate of Health. 2018 [cited 2025 Oct 13]. Available from: <https://island.is/en/nutrition-recommendations/naering-a-medgongu>

Krümmel EM, Boyd AD, Brandow D, et al. Updated review on contaminant communication experiences in the circumpolar Arctic. *International Journal of Circumpolar Health*. 2024; 83(1): 2371623.

Krümmel EM, Gilman A. An update on risk communication in the Arctic. *Int J Circumpolar Health*. 2016; 75(1): 33822.

State of Alaska. Fish facts & consumption. 2020 [cited 2025 Oct 13]. Available from: <https://health.alaska.gov/en/education/fish-facts/>

Weihe P, Bjerregaard P, Bonefeld-Jørgensen E, et al. Overview of ongoing cohort and dietary studies in the Arctic. *International Journal of Circumpolar Health*. 2016; 75:33803.

Weihe P, Grandjean P. Cohort studies of Faroese children concerning potential adverse health effects after the mothers' exposure to marine contaminants during pregnancy. *Acta Veterinaria Scandinavica*. 2012; 54:S7.

3) Dietary intervention study: HBM4EU-Mom

The HBM4EU-Mom study was a multicentre randomized controlled trial within the European Human Biomonitoring Initiative (HBM4EU), conducted in five coastal, high fish-consuming countries (Cyprus, Greece, Spain, Portugal, Iceland). It integrated human biomonitoring (HBM) with a dietary intervention, providing seafood-consumption advice to pregnant women to reduce methylmercury (MeHg) exposure, while also documenting other potential mercury sources. Pregnant women (≥ 120 per country) were enrolled during the first trimester (gestational age ≤ 20 weeks). They provided a baseline hair sample for total mercury (THg) determination and completed a questionnaire covering lifestyle, pregnancy status, pre- and during-pregnancy diet (including seafood), and potential non-dietary mercury exposures. Next, the women were randomly assigned to either a control group (that would continue their habitual practices) or an intervention group (that received harmonized HBM4EU-MOM dietary advice for fish consumption during pregnancy and were encouraged to follow it). Around the child's delivery, the women provided a second hair sample and completed another tailored questionnaire.

Source: Katsonouri et al., 2023 (<https://doi.org/10.1016/j.ijheh.2023.114213>); HBM4EU-Mom Final Report

Methodology:

No particular methodology applied, as results from published papers/reports were used. Evidence was compiled from existing publications and official reports and summarized descriptively.

Answer/Response:

1) Dental amalgam phase-out for children < 15 years (Slovenia)

Across Slovenian children studied between 2007 and 2024, dental amalgam use was consistently associated with higher urinary mercury (U-Hg), with the number of fillings showing a significant positive correlation with U-Hg (Spearman's $\rho = 0.315$ and 0.340 for U-Hg in $\mu\text{g/L}$ and in $\mu\text{g/g}$ creatinine; both $p < 0.001$). Over this period,

median U-Hg concentrations declined markedly, from 0.76 ng mL⁻¹ in the 2007 PHIME cohort to 0.22 µg/L in the 2018–2024 SLO-HBM-II cohort, in parallel with a pronounced decrease in amalgam prevalence in children (from 65% to 3%) and in the mean number of fillings among carriers (from 3.02 to 1.85; Figure 13). In mixed-effects models, inclusion of amalgam variables (either as a binary indicator or as the number of fillings) substantially attenuated the calendar-year effect, suggesting that changes in amalgam use accounted for a large share of the observed temporal pattern in U-Hg. As amalgam exposure declined, fish consumption emerged as a relatively stronger predictor of U-Hg in more recent cohorts, reflecting a shift in the relative importance of exposure sources. Study-specific analyses confirmed this pattern, with amalgam effects strongest in the oldest study, PHIME, and fish consumption influence strongest in the most recent study, SLO-HBM-II. Children living in historically contaminated areas continued to show higher U-Hg, indicating the persistence of environmental background contributions alongside changes in dental-related exposure.

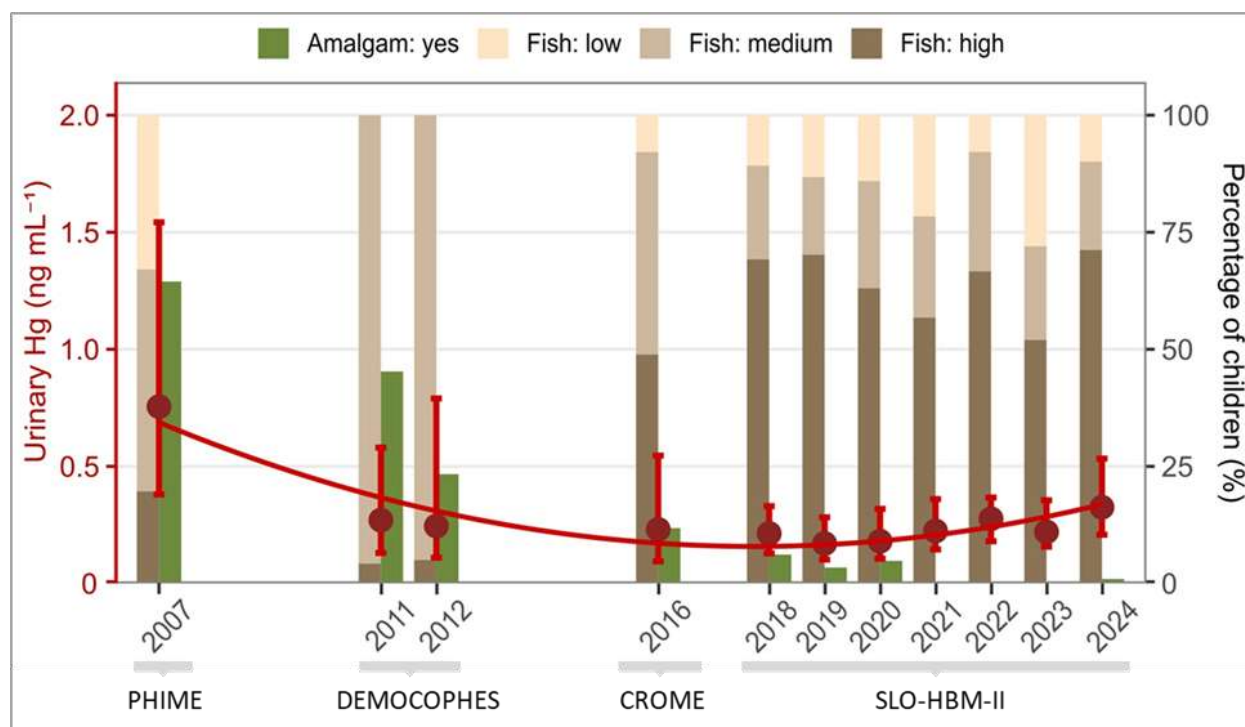


Figure 13. Urinary mercury concentrations and exposure-related characteristics in the recruited Slovenian children: urinary Hg (µg/L; median and interquartile range, left y-axis) and on the right y-axis, the percentage of participants with at least one dental amalgam and the percentages in low-, medium- and high-fish consumption groups by year (2007–2024).

Taken together, these findings indicate that the phase-down of dental amalgam use has been accompanied by measurable reductions in children's mercury body burden in Slovenia. Although residual amalgam exposure remains detectable, dietary sources now represent a relatively larger share of ongoing exposure. The results demonstrate the value of human biomonitoring for evaluating policy effectiveness, tracking shifts in exposure pathways, and informing further efforts to minimize non-essential mercury exposure in children.

2) Dietary recommendations/interventions in the Arctic

While there are still many regions of the Arctic with limited human biomonitoring data, levels of several POPs and metals are declining in the Arctic regions where time trend data exist, although not uniformly or consistently across regions (AMAP, 2021; Weihe et al., 2016). Multiple cohort studies in the Faroe Islands show that the levels of POPs and Hg are decreasing. This is most likely due to decreasing consumption of pilot whale meat, as the levels in pregnant women and children in the latest cohort are much lower than those observed in the first one (Weihe & Grandjean, 2012), while pilot whale levels have been continuously increasing (AMAP 2015, Krümmel and Gilman 2016). This serves as an example of how overall exposure trends can decline despite high environmental contaminant levels when messaging is consistent, culturally sensitive and supported by accessible, healthy food alternatives. However, the success in the Faroe Islands came at the cost of loss of cultural identity

(AMAP 2021), and healthy food alternatives are often not available in highly food insecure Indigenous communities.

Due to the effects of colonization and a dominating impact of Western culture, most Arctic populations have experienced a transition towards imported and highly processed foods. Health effects of such transitions depend on the characteristics of the newly adopted food items and the extent to which the traditional diet is maintained, but a switch to highly processed store-bought foods has been mostly negative for Arctic Indigenous Peoples. It has led to a decline in nutrient levels associated with reduced traditional food intake, such as lower vitamin D (Alaska, Canada), omega-3 fatty acids (Greenland), and iodine (young women in Iceland and Norway). Health effects include hypovitaminosis (across the European North of Russia), rising obesity, metabolic disorders, and dental problems, which are linked to higher consumption of sugar-rich and processed products (AMAP, 2021). Reported benefits of the dietary change include a reduction in contaminant burdens (e.g., mercury in the Faroe Islands and Greenland), increased intake of dietary fibre (reported for Greenland and Norway), and fruit and vegetables (reported for Norway and Sweden – while still too low), as well as reduced trans-fat intake (as seen in Iceland). Policy and program responses have included communication and access initiatives to promote low-contaminant, nutrient-dense traditional foods (Canada, Greenland), food-handling education, vitamin D fortification/supplementation, and iodine fortification (Finland).

Future efforts should prioritise reducing or eliminating environmental contaminants and promoting culturally grounded, contaminant-aware diets that highlight the benefits of traditional country foods, coupled with an adequate evaluation of the effectiveness of dietary advice. Cohort studies and biomonitoring programs remain essential for establishing time trends of contaminants, following populations with elevated levels, studying health effects and identifying contaminants of emerging Arctic concern.

3) Dietary intervention study: HBM4EU-Mom

Mercury levels in the hair of women participating in Phases 1 and 2 of the study are presented in Figure x. In both phases, the highest Hg levels in hair were observed in Portugal (GMs 1555 and 1368 ng/g, for Phase 1 and 2, respectively), followed by Spain (902 and 721 ng/g), Greece (440 and 464 ng/g), Iceland (360 and 362 ng/g) and Cyprus (307 and 309 ng/g).

The levels of Hg in hair were compared with the available health-based levels that were established to protect an unborn child against neurodevelopmental toxicity. The lowest cut-off value, established by Bellanger et al. (2013) (580 ng/g), was exceeded by approximately half of the participants in both Phases 1 and 2. The US EPA reference cut-off value of 1000 ng/g (NRC, 2000) was exceeded by approximately one third of participants in both phases. The value of 1800 ng/g established by EFSA (2012) was exceeded by 13% of the women in Phase 1 and 8% of the women in Phase 2. The value of 2300 ng/g, established by FAO/WHO (2004), was exceeded by 7% of the women in Phase 1 and 4% of the women in Phase 2.

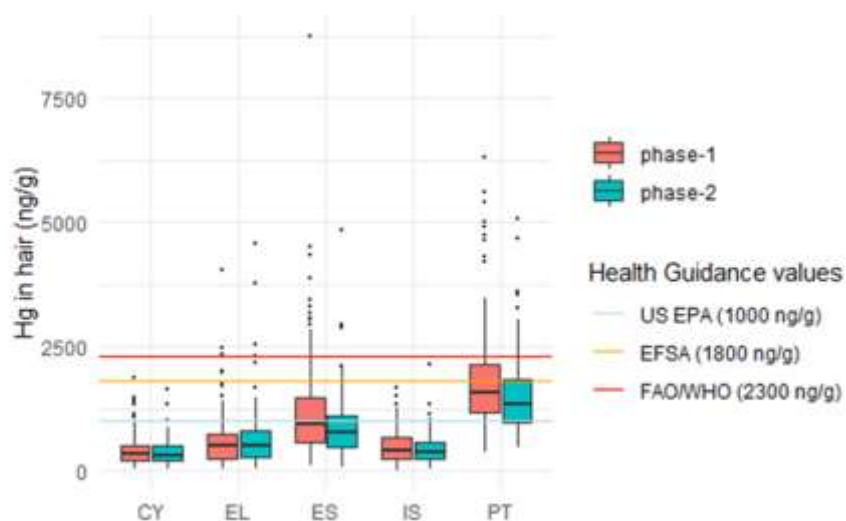


Figure 14. Distribution of Hg levels in hair (ng/g) of the HBM4EU-Mom study participants in Phases 1 (1st trimester of pregnancy) and 2 (after receiving dietary advice on fish consumption). CY=Cyprus; EL=Greece; ES=Spain; IS=Iceland; PT=Portugal

Marked differences in exceedances of these cut-off values were observed between the countries. Portugal had the highest percentage of women exceeding the available cut-off values, followed by Spain, Greece, Iceland and Cyprus. Overall, a drop in the percentage of women exceeding the cut-off values was observed after following dietary advice (Fig. 15). The sharpest drop from Phase 1 to Phase 2 was observed in Spain.

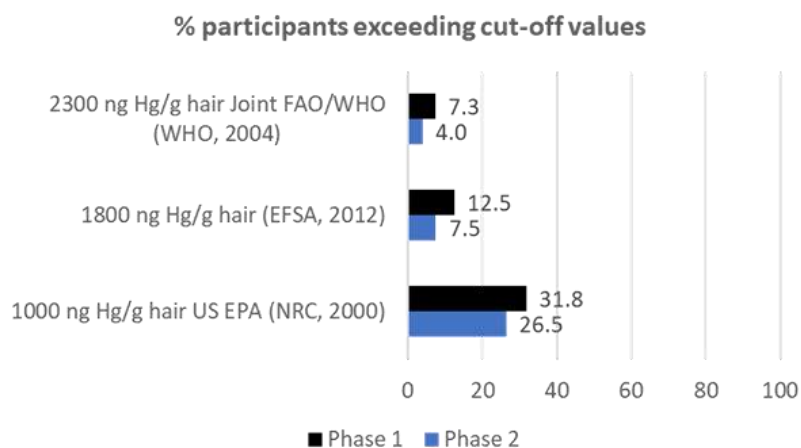


Figure 15. Percentage of participants exceeding available health-based guidance values for Hg in hair of HBM4EU-Mom study participants in Phases 1 (1st trimester of pregnancy) and 2 (after receiving dietary advice on fish consumption).

Consistency of Answer with Existing Literature (especially the 2018 GMA):

To be added in the next version.

Evaluation of Confidence:

To be added in the next version.

Information and Capacity Needs:

To be added in the next version.

2.6 Soil

2.6.1 What data are available?

Soil data are available from Australia, Africa, Europe, New Zealand, North America, and South America (Figure 1 and Appendix II). Due to the low coverage of the available data, we used countries or continents as specific regions and we do not use the IPCC regions. Submissions include measurements for total Hg in soil. Ancillary data for pH, carbon content, and percentage of clay, silt and sand are available for a very limited number of sites. Table 1 reveals a highly uneven distribution of available soil Hg data across continents and countries. North America, led by the United States, dominates with an exceptional 14,571 Hg sampling sites, indicating extensive monitoring efforts. Oceania follows, primarily due to Australia, which contributes 2,401 sites. In South America, Peru and Ecuador stand out with 733 and 561 sites, respectively, while other countries like Brazil and Argentina show modest data availability. Europe presents a large coverage but moderate dataset, with France (236), Germany (146), and Finland (130) among the most sampled. Africa remains underrepresented, with only Tanzania (2) and Uganda (3) reporting data. Asia appears largely absent from the dataset, suggesting significant gaps in Hg soil monitoring across the continent. This global snapshot highlights both the strengths and disparities in environmental Hg surveillance, emphasizing the need for expanded efforts in underrepresented regions.

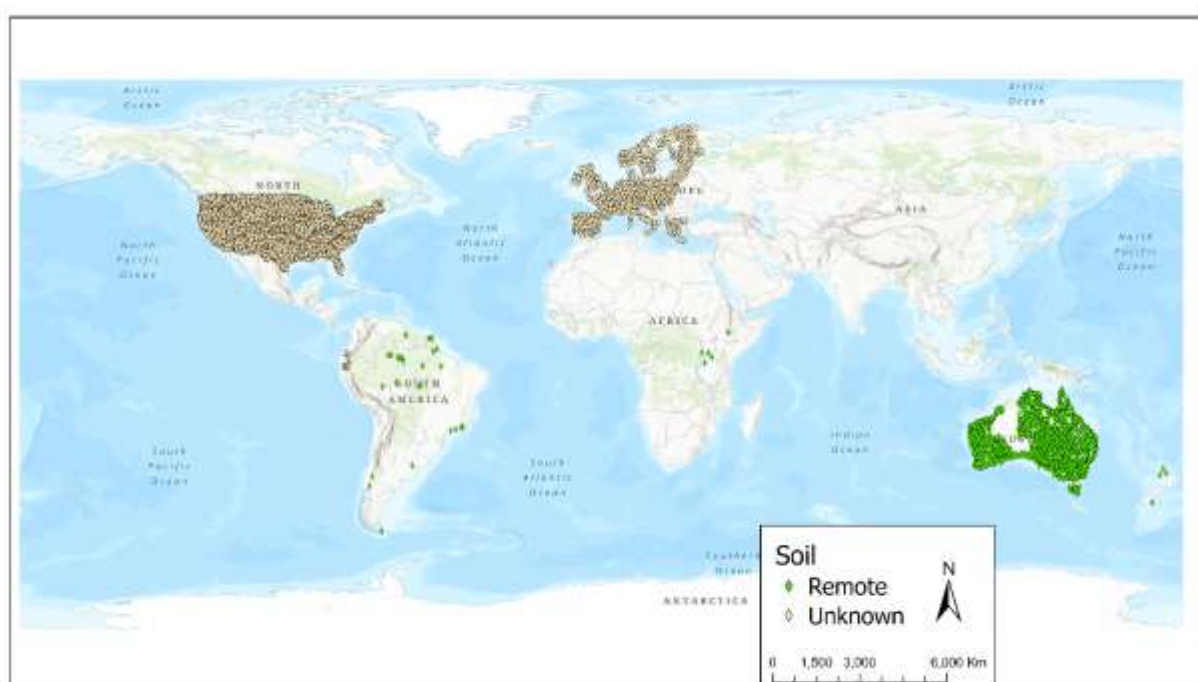


Figure 1 : Map showing the distribution of the available soil data.

Additional datasets with more recent soil Hg data from Europe (LUCAS database) as well as individual European monitoring program (e.g. Switzerland, Germany, Poland) were identified but could not be accessed. Existing data from monitoring programs in China, Canada and Peru either were not submitted or are pending data use authorization.

Table 1: World survey of Hg soil data

Continent	Country	Hg data 2025
Europe	Albania	4
	Austria	34
	Belgium	10

	Croatia	26
	Czech Republic	20
	Denmark	10
	Estonia	22
	Finland	130
	France	236
	Germany	146
	Greece	82
	Hungary	27
	Ireland	22
	Italy	98
	Latvia	16
	Lithuania	30
	Netherlands	15
	New Zealand	4
	Norway	116
	Poland	112
	Portugal	38
	Slovakia	29
	Slovenia	10
	Spain	135
	Sweden	102
	Switzerland	19
	United Kingdom	120
Africa	Tanzania	2
	Uganda	3
	Argentina	4
America	Brazil	21
	Ecuador	561
	Peru	733
	United States of America	14571
	Venezuela	1

	French Guiana	3
Oceania	Australia	2401

The datasets span different time periods depending on the region (Figure 2). For Latin America, data were submitted for Ecuador for the years 2012 to 2023 and a dataset from Peru was identified that spans from 2015 to 2023. For Australia, available data spans from 2007 to 2009 (Figure 2). Publicly available data for Europe was accessed from the FOREGS Geochemical database and contains data for soil samples taken between 1997 and 2001 from the following countries: Albania, Austria, Belgium, Croatia, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, The Netherlands, Norway, Poland, Portugal, Slovak Republic, Slovenia, Spain, Sweden, Switzerland and the UK. The publicly available data from the USA covers the period between 2007 and 2010. Limited data for Argentina, French Guyana, Brazil, New Zealand, Tanzania, Uganda and Venezuela was obtained from tables and supporting information presented in published scientific publications and spans from 1978 to 2020 .

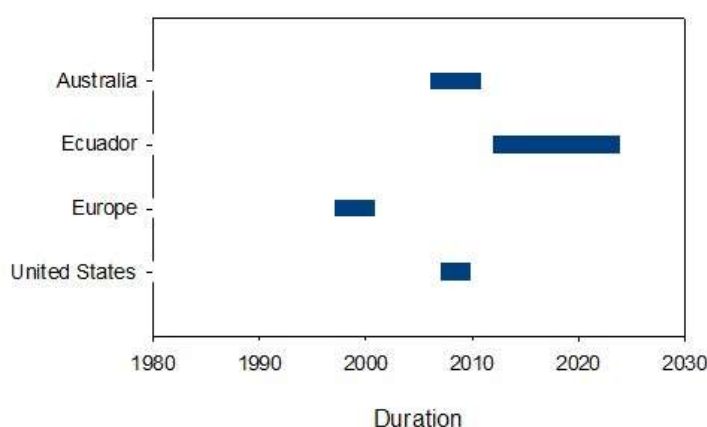


Figure 2 : Years with available observations for soil.

2.6.2 What are current levels of THg and MMHg (and other species/fractions if applicable) observed in soil that are remote from or near anthropogenic sources?

Information Used (including applicable indigenous and traditional knowledge): Received (submitted) or acquired data from publicly available databases. Only THg was available.

Methodology: Both datasets for remote from or near anthropogenic sources were restricted to soil samples within the 0–10 cm to 0–30 cm depth intervals and excluded entries with missing or zero total Hg concentrations. In both cases, the reported limit of detection (LOD) was considered, with values of 0.005 ng Hg (absolute values – DMA) and 100 ng / g (ICP-MS)- Laboratory quality documentation was reported as “ISO/IEC 17025”, “EPA Method 200.7”, or “Not Available”. Common sample preparation methods included sieving prior to analysis and chemical digestion using hydrochloric acid, nitric acid, aqua regia, as well as direct analysis without digestion. THg detection methods overlapped substantially, with both datasets reporting CV-AAS, ICP-MS, and DMA.

However, key differences exist. The anthropogenically influenced dataset comprises samples collected between 2007 and 2022, reflecting areas impacted by human activities, and standardized sieving to <2 mm. In contrast, the non-influenced dataset spans a broader temporal range from 1978 to 2023, includes sieving sizes from 0.075 mm to 2 mm, and represents soils with no recent anthropogenic impact. Additionally, the non-influenced dataset recorded an extra detection technique (CVAFS) not present in the influenced dataset. Finally, while both datasets were filtered to retain the most recent year with at least five observations, the broader historical coverage of the non-influenced dataset reflects its inclusion of legacy and modern records, contrasting with the more contemporary focus of the anthropogenically influenced dataset.

Answer/Response: This question can only be partially answered since the data collected does not consistently label the sampling points as either remote or anthropogenically influenced and the distance to potential point sources are missing. Also, the data used is not homogenous (see “Methodology” above). Nonetheless, Ecuador and the United States show elevated Hg levels that are primarily associated with industrial and large-scale mining activities, with Ecuador showing the highest concentrations among the countries analyzed (Figure 3 and 4). Peru, influenced by artisanal and small-scale gold mining (ASGM), presents intermediate Hg levels, reflecting localized contamination.

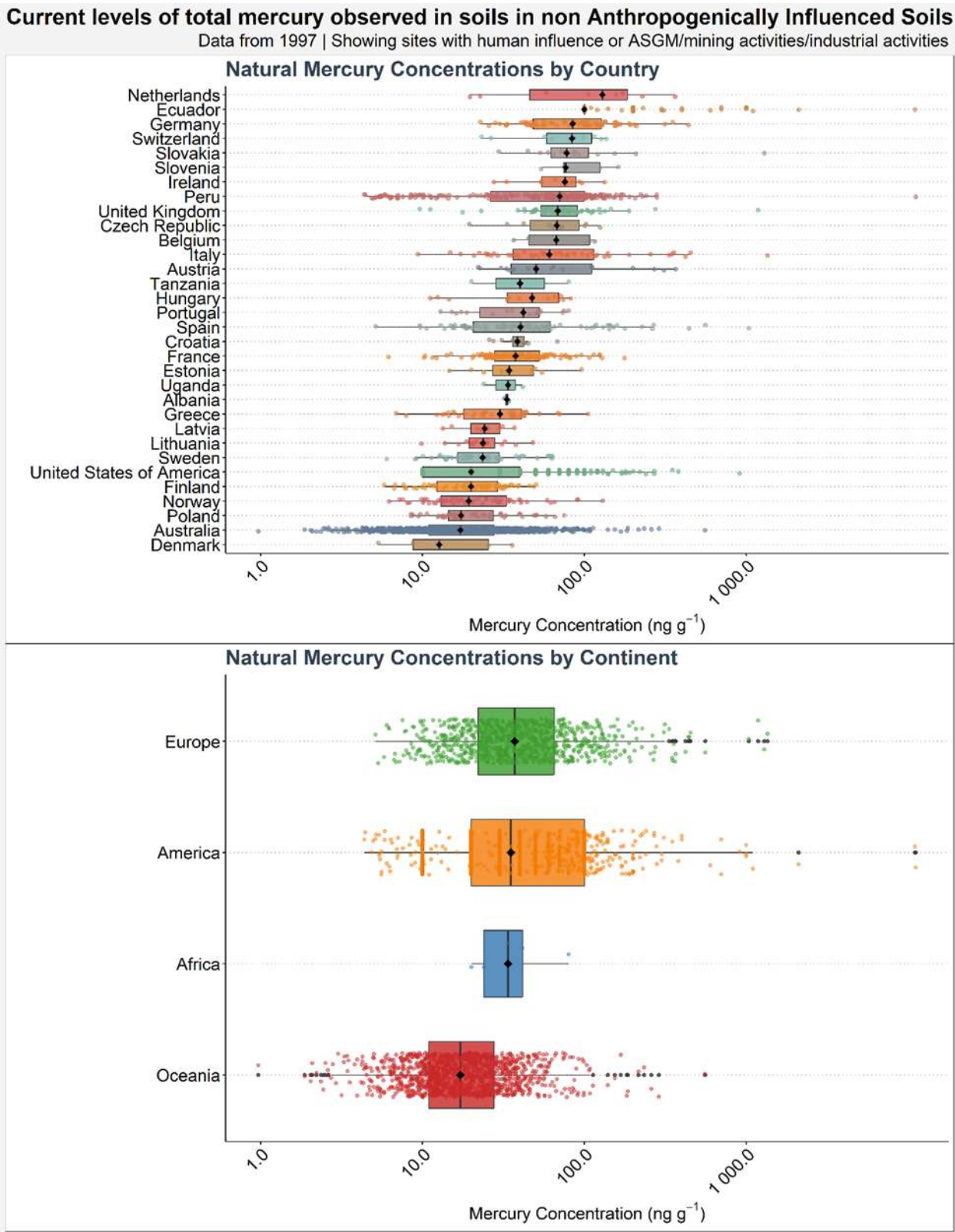


Figure 3: Current levels of THg in soils in non-anthropogenically influenced soils

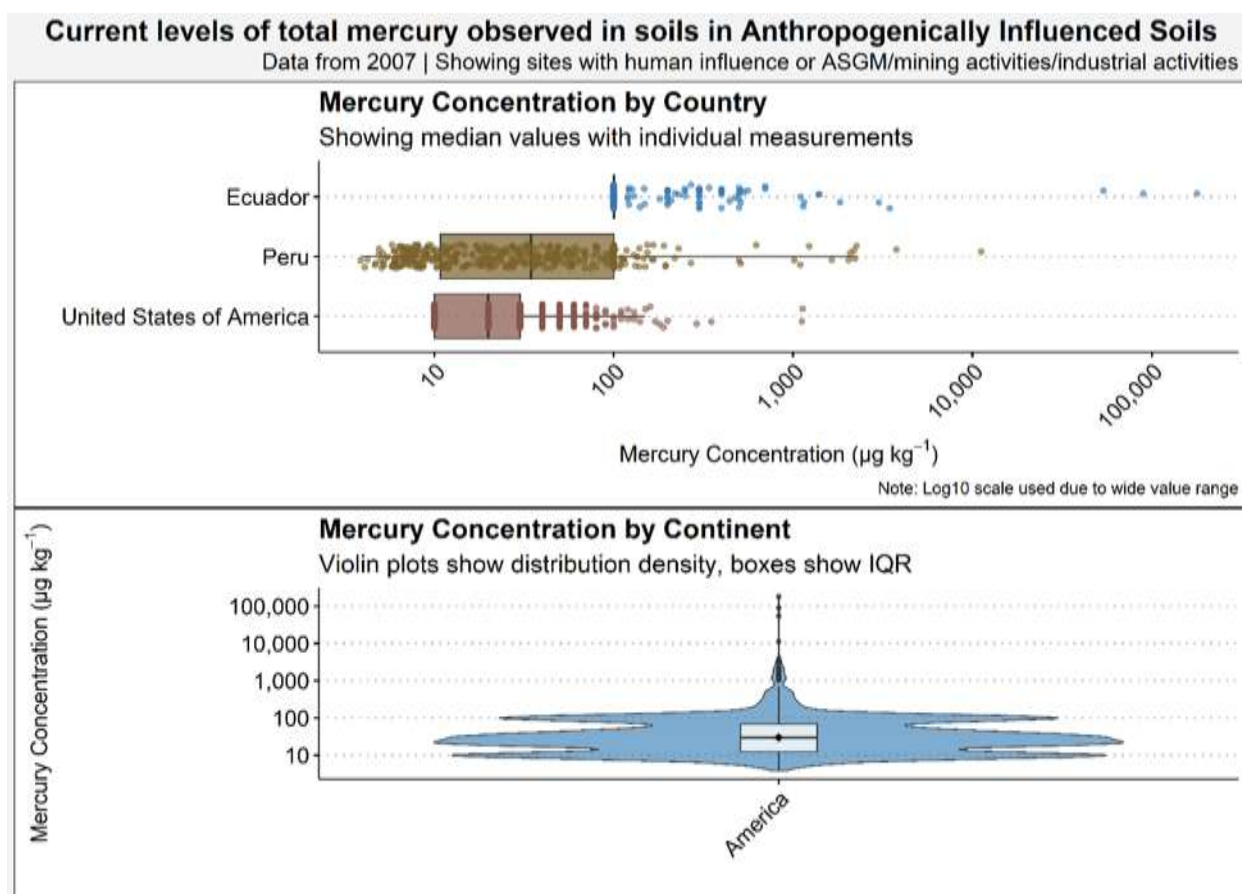


Figure 4: Current levels of THg in soils in anthropogenically influenced soils

Consistency of Answer with Existing Literature (especially the 2018 GMA): There is only one study, in Europe, that has addressed this question at a large scale, to the best of our knowledge (Ballabio et al., 2021). There, the top 1% of the measured Hg concentrations ($> 492 \mu\text{g kg}^{-1}$) were classified as likely outliers corresponding to sites subject to geogenically enriched bedrock or topsoil Hg contamination (42% of these could be associated with known mining sites). There are likely many small individual studies on soils impacted by large and small-scale mining or other anthropogenic activities that might be useful to answer this question but collecting and analysing such data would represent a huge logistical challenge that is currently not possible with the available human resources.

Evaluation of Confidence: While the current dataset provides limited but valuable insights into THg concentrations in soils near anthropogenic sources, it is important to recognize that differences in soil types, land use, and the nature of human activities across countries limit the ability to draw statistically significant comparisons. Furthermore, as highlighted in “Methodology”, above, the datasets are not rigorously homogenous (depth, sieving etc.). Finally, only very few datasets were available.

Information and Capacity Needs: Collection of data that systematically records the anthropogenic source, type and distance are necessary as well as homogenous data using similar sampling and sample preparation schemes.

2.6.3 How have levels of THg and MMHg (and other species/fractions if applicable) observed in soil changed over the available record?

Information Used (including applicable indigenous and traditional knowledge): Received (submitted) or acquired data from publicly available databases. Only THg was available.

Methodology: The evaluation of THg concentrations in soil was based on country, continent, year, and sample counts. The anthropogenically influenced dataset spans 2007–2022, includes Ecuador, Peru, United States of America with a total of 7,936 samples, all from the American continents. The non-influenced dataset spans 1997–2023 and includes Australia, Austria, Belgium, Brazil, Croatia, Czech Republic, Denmark, Ecuador, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Netherlands, New

Zealand, Norway, Peru, Poland, Portugal, Slovakia, Slovenia, Spain, Sweden, Switzerland, United Kingdom, United States of America, located across Oceania, Europe, and America, with a total sample counts per continent of 6,921 for America, 1,600 for Europe and 2,400 for Oceania.

Sampling depths included 0–10 cm, 0–20 cm, 0–30 cm, and below 30 cm. Sieving sizes ranged from 0.075 mm to 2.0 mm. Reported detection limits (LOD) ranged from 0.005 to 100 ng Hg. Sample digestion or extraction methods were hydrochloric acid, nitric acid, and aqua regia. Hg detection methods included Direct Mercury Analysis (no digestion), CV-AAS, CV-AFS, and ICP-MS. Laboratory quality management systems were reported as “ISO/IEC 17025”, “EPA Method 200.7, Rev.4.4”, or “Not Available”.

Answer/Response: This question is difficult to answer for soils since soils are not typically sampled periodically. Also, any change in soil concentrations are happening over long periods of time while the data we collected represent short time periods for each dataset (Fig. 2).

Across non-anthropogenically influenced soils compiled in the available dataset, THg levels show no consistent temporal decline. This might be due to a shift in the geographical origin of the available data through the years although we cannot exclude that the reduction of Hg emissions is not (yet) reflected in soils which are large Hg reservoirs. Before 2013, the record is dominated by Australia and the United States, with low background levels ($\sim 15\text{--}30 \mu\text{g kg}^{-1}$), while post 2013 data come mainly from Andean–Amazon countries (Ecuador, Peru), where natural and legacy Hg levels are higher ($\sim 70\text{--}100 \mu\text{g kg}^{-1}$). This geographic change inflates the apparent global median after 2013, and makes any statistical data treatment very difficult. Europe and Africa remain vastly underrepresented whereas Oceania shows stable low values with sparse recent data.

Consistency of Answer with Existing Literature (especially the 2018 GMA): To be added

Evaluation of Confidence: Low due to short time periods for each dataset. Differences in soil types, land use, and the nature of human activities across countries/datasets limit the ability to draw statistically significant comparisons. Furthermore, as highlighted in “Methodology”, above, the datasets are not rigorously homogenous (depth, sieving etc.). Finally, only very few datasets were available.

Information and Capacity Needs: A monitoring program with consistent sampling and analyses over time (decadal time scale) for a few selected background sites would help to answer this question.

2.6.4 How do temporal trends of Hg levels observed in soils compare to or co-vary with variations and trends of related pollutants or environmental variables over the available record? (TOC, SOM, DOC, TSS)?

This question cannot be answered with the available data due to its scarcity.

Consistency of Answer with Existing Literature (especially the 2018 GMA): There does not seem to be any existing literature on the topic, to the best of our knowledge.

Information and Capacity Needs: same as 0.8.3, a monitoring program with consistent sampling over time (decadal time scale) for a few selected sites would allow to answer this question.

2.6.5 How do THg and MMHg levels (and other species/fractions if applicable) co-vary temporally in soils?

This question cannot be addressed since no data was submitted or available for MMHg.

Consistency of Answer with Existing Literature (especially the 2018 GMA): Compared to sediments and waters, there are relatively few studies on the levels and production pathways of MMHg in natural soils. These studies are usually carried out on permanently or periodically flooded soils, as suboxic and anoxic conditions are more conducive to Hg methylation, with a focus lately on the northern regions due to increasingly warmer and thawing conditions (St. Pierre et al., 2015; Yang et al., 2016; Olson et al., 2018; Fahnestock et al., 2019; Tarbier et al., 2021; Varty et al., 2021). Despite their limited number, they indicate that MMHg concentrations and contribution to THg (%MMHg) can be highly fluctuating across seasons, i.e. concentrations can vary by several orders of magnitude and contribution to THg from 0 up to 10 %, depending on soil depth/horizons, temperature, redox conditions, composition of bacterial communities, electron donors and acceptors, and Hg availability. On the other hand, studies on Hg methylation in contaminated soils are more numerous and largely focused on floodplain and paddy soils (O’Connor et al., 2019; Zhao et al., 2020). MMHg levels at contaminated sites are usually higher due to a larger pool of Hg being available for transformations but they are also largely fluctuating. Therefore it is unlikely that a general temporal relationship would be uncovered between HgT and MMHg in soils globally even if such data would be available.

Information and Capacity Needs: Monitoring programs including MMHg measurements are missing as they are cost and time intensive. This would be potentially possible for specific/representative selected sites.

2.6.6 How do THg and MMHg levels (and other species/fractions if applicable) found in soils differ by world regions?

Information Used (including applicable indigenous and traditional knowledge): Received (submitted) or acquired data from publicly available databases. Only THg was available.

Methodology: Available data ranged from 1978 to 2023. Sampling depths included 0–10 cm, 0–20 cm, 0–30 cm, and below 30 cm. Sieving sizes ranged from 0.075 mm to 2.0 mm. Reported detection limits (LOD) were 0.005 ng Hg (Absolute value value-DMA), 0.01 ng Hg (Absolute value - DMA), and 0.1 (CVAAS), 1 (CVAAS), 10 (CVAAS), and 100 (ICPMS) ng g⁻¹. Sample digestion or extraction methods were hydrochloric acid, nitric acid, and aqua regia. Hg detection methods included Direct Mercury Analysis (no digestion), CV-AAS, CV-AFS, and ICP-MS. Laboratory quality management systems were reported as “ISO/IEC 17025”, “EPA Method 200.7, Rev.4.4”, or “Not Available”.

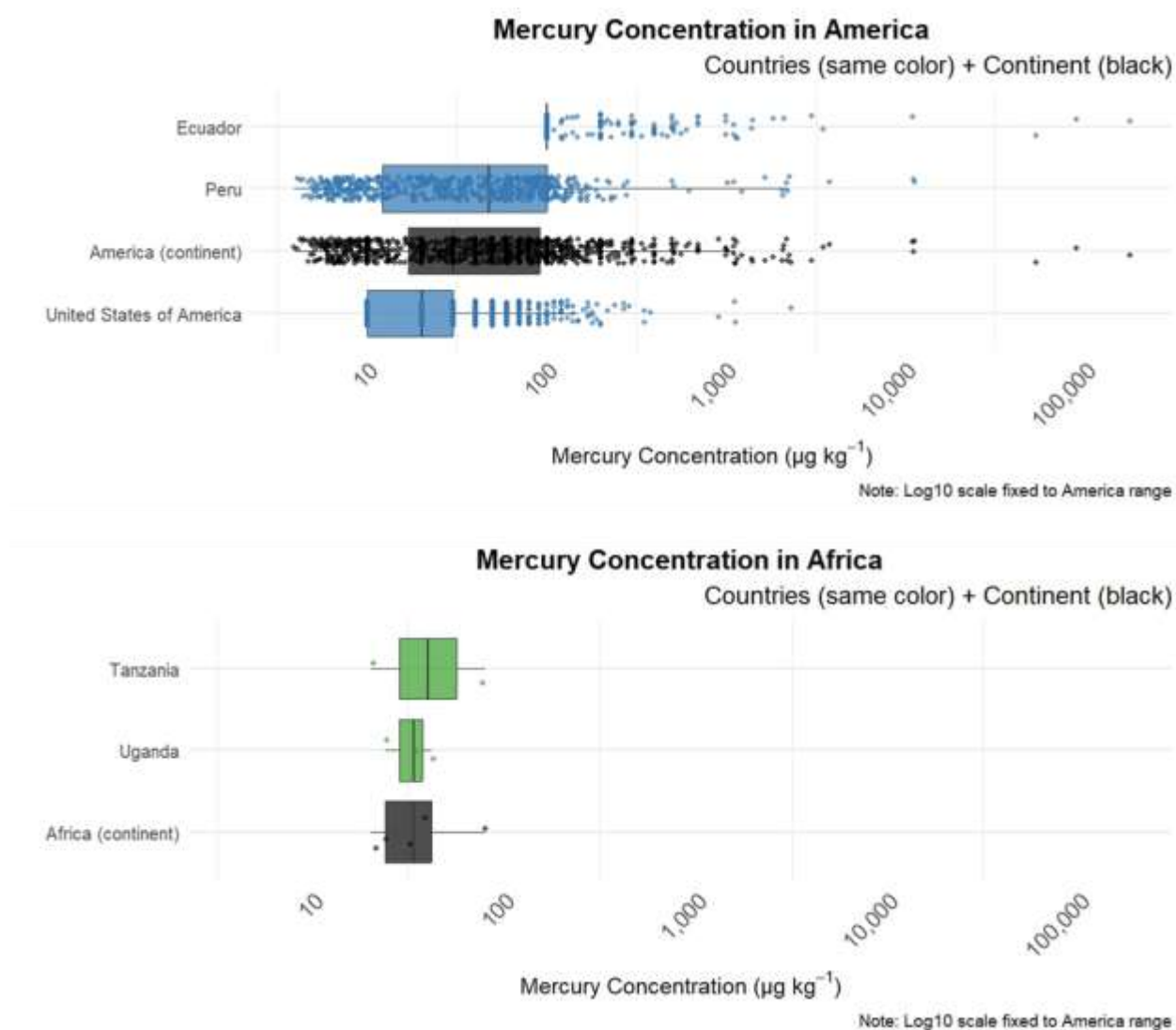


Figure 5: Distribution of THg concentrations in America and Africa. Please note that due to the number of data points available, the ranges for each continent are not representative but rather illustrative.

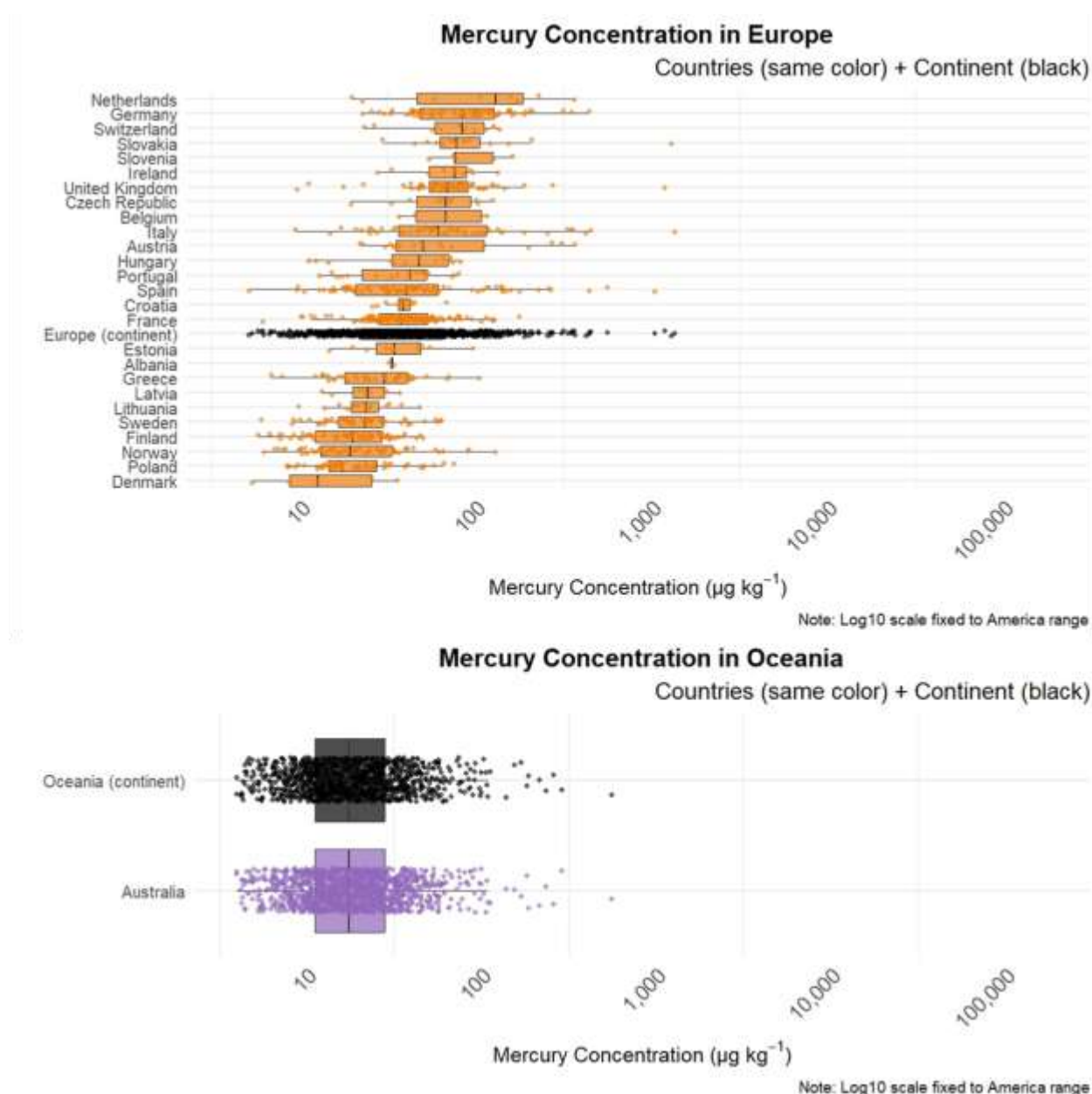


Figure 6: Distribution of THg concentrations in Europe and Oceania. Please note that due to the number of data points available, the ranges for each continent are not representative but rather illustrative.

Answer/Response: The analysis of THg concentrations in soil across continents, based on available data, provides a valuable snapshot of the global state of Hg levels (Figure 5 and 6). It should be noted that due to the low number of data points available, the ranges for each continent are not representative but rather illustrative. America and Europe show the highest concentrations and variability, with notable outliers in countries such as the USA and several European countries. Africa presents the lowest and most consistent levels, while Oceania, particularly New Zealand, shows moderate to low concentrations with fewer anomalies. Although these patterns suggest regional differences, no concrete answer can be given; rather, the available data indicates trends that merit further investigation. Given the long time span and global coverage of the dataset, this evaluation serves as a baseline for future monitoring, emphasizing the importance of identifying key indicators and regions for continued observation and scientific assessment.

Consistency of Answer with Existing Literature (especially the 2018 GMA): In addition to the extensive datasets for the US and Australia, there are large scale studies covering Europe and China (data not available for this report) that can help answer the question. The average $\pm$ SD topsoil Hg concentration in Europe (0 – 20 cm) was $55.4 \pm 4.9 \mu\text{g kg}^{-1}$ while the median was $23.3 \mu\text{g kg}^{-1}$ with an inter-quartile range of $35.7 \mu\text{g kg}^{-1}$ (Ballabio et al., 2021). Higher concentrations were found in forested soils compared to other land uses (at the exception of

contaminated soils). A north-south gradient was also noted due to different type of vegetations, normalized difference vegetation index (NDVI), and soil organic carbon content. This is consistent with data from the US where the mean ranges from 17 to 87 $\mu\text{g kg}^{-1}$ (median from 11 - 73 $\mu\text{g kg}^{-1}$) in the top 5 cm, depending on the land cover or ecoregions (Olson et al., 2022). No such detailed analysis about land cover or ecoregions has been conducted for the Australian dataset where the average Hg concentration is $30 \pm 18 \mu\text{g kg}^{-1}$ (Reimann and de Caritat, 2017; Schneider et al., 2023).

Evaluation of Confidence: low due to the scarcity of data available.

Information and Capacity Needs: Data is missing for many parts of the world. Also, there is a critical need to establish consistent sampling, analysis and reporting protocols so data can be compared across regions.

2.6.7 How do temporal trends in THg and MMHg (and other species/fractions if applicable) found in soils differ by world regions?

The available data does not allow us to address this question.

Consistency of Answer with Existing Literature (especially the 2018 GMA): There does not seem to be any existing literature on the topic, to the best of our knowledge.

Information and Capacity Needs: Data is missing for many parts of the world. Also, there is a critical need to establish consistent sampling, analysis and reporting protocols so data can be compared across regions.

2.6.8 How do THg and MMHg levels (and other species/fractions if applicable) found in soils differ by watershed land cover or land use?

The available data does not allow us to address this question as land cover/use are not consistently or not at all reported.

Consistency of Answer with Existing Literature (especially the 2018 GMA): Beyond the distance to anthropogenic activities and the presence of lithogenic Hg, the presence of vegetation, especially forest, is a critical factor to explain the concentrations of Hg in soils (Ballabio et al., 2021; Obrist et al., 2016; Wang et al., 2019; Zhang et al., 2020). This is because plants act as a biological pump for atmospheric Hg (Jiskra et al., 2018; Obrist et al., 2017), absorbing elemental Hg through leaves' stomata and sequestering it under its oxidized form. This Hg is then transferred to soils during litterfall, which represents one of the largest fluxes in the Hg cycle and the largest sink for atmospheric Hg (Wang et al., 2016; Zhou et al., 2021; Zhou and Obrist, 2021). Therefore, Hg concentrations in soils under forests is usually twice as high as in other land use/cover (Almeida et al., 2005; Lacerda et al., 2004; Gamby et al., 2015; Ballabio et al., 2021; Olson et al., 2022) but it can largely vary depending on the ecosystem, type of vegetation and land-use considered.

Information and Capacity Needs: A clear reporting and categorization of land-use/cover is needed for future monitoring programs for terrestrial ecosystems.

2.6.9 How do THg and MMHg levels (and other species/fractions if applicable) co-vary spatially in soils?

This question cannot be addressed since no data was submitted or available for MMHg.

Consistency of Answer with Existing Literature (especially the 2018 GMA): As mentioned in 0.8.5, very few studies on MMHg levels in soils are available, especially at large scale so it is difficult to answer this question. A positive relationship between THg and MMHg has been reported in one case (St. Pierre et al., 2015). However, since it is the "bioavailable fraction" of THg (and not THg per se) that is the required substrate for methylation process, it is unlikely that such a relationship would be found over a wide range of conditions, even if data would be available. Such relationship was nonetheless previously evidenced in sediments and waters of various ecosystems, see e.g. Cossa et al. (Cossa et al., 2014), which demonstrates the importance of THg as the substrate for the reaction even if other parameters are also important.

Information and Capacity Needs: same as 0.8.5 above, monitoring programs including MMHg measurements are missing ut perhaps possible for specific/representative sites.

2.6.10 How do spatial trends of Hg levels observed in soils compare to or co-vary with variations and trends of related pollutants or environmental variables over the available record? (TOC, SOM, DOC, TSS)?

Information Used (including applicable indigenous and traditional knowledge): Received (submitted) or acquired data from publicly available databases. The dataset included variables such as sampling depth (cm), sieving size (mm), individual THg concentration (ng/g), pH, organic carbon (%), clay (%), silt (%), sand (%), electrical conductivity (dS/m), and cation exchange capacity (cmol+/kg).

Methodology: To assess the co-variation of THg with other environmental variables, only the data from America and Europe were compiled due to data availability for correlations. Sampling depths considered were 0–10 cm and 60–80 cm, representing surface and subsurface layers. Sieving was applied prior to analysis using the following classes: 0.075 mm, 0.25 mm, 0.5 mm, 1.0 mm, and 2.0 mm. These data were used to evaluate relationships between THg concentration and soil attributes using common statistical approaches.

Answer/Response: In both America and Europe, correlations between Hg and variables such as pH, Clay (%), Silt (%), Sand (%), Electrical Conductivity (EC), and Cation Exchange Capacity (CEC) were generally weak or negligible (Figure 6). Only Europe showed a modest positive correlation between Hg and Organic Carbon (%) ($r = 0.23$), suggesting a limited role of organic matter in Hg retention. In contrast, the American dataset, which includes data from Ecuador and Peru, reflects diverse soil conditions and anthropogenic influences that likely contribute to the absence of clear correlations. These findings highlight that, while temporal analysis is not feasible, the available data allows for some regional comparison of Hg covariation with key soil properties.

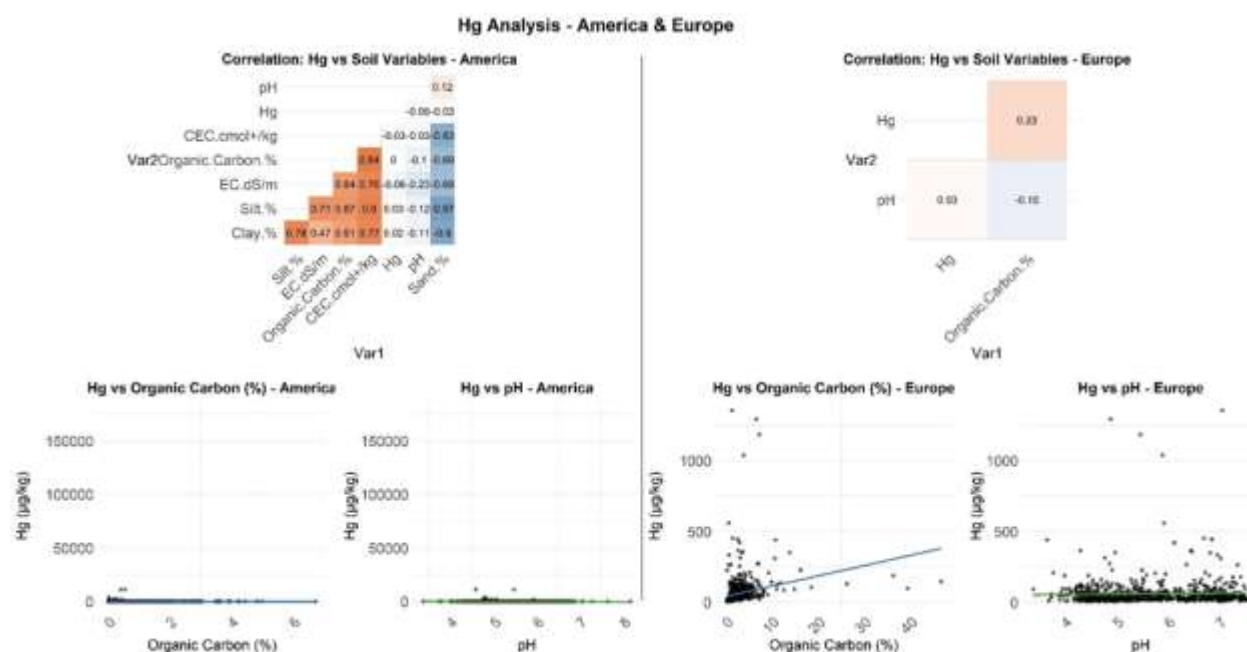


Figure 6: Covariance analysis of THg and relevant soil properties.

Consistency of Answer with Existing Literature (especially the 2018 GMA): Many previous studies have found various relationship, at the local and up to continental scale, between THg and soil physico-chemical parameters, such as texture, CEC, elemental content but especially with soil organic matter (Schuster et al., 2018; Zhang et al., 2020; Tarbier et al., 2021) as Hg presents strong affinity to reduced sulfur associated to organic matter (Sjallberg et al., 2006). However, there are marked differences between temperate and (sub)arctic soils on one hand and tropical soils on the other one. Positive (but often weak) relationships between THg and SOM are usually found in surface soils from temperate and (sub)arctic ecosystems. However, these relationships only hold-up to a certain SOM level, above which the supply of Hg is limiting. On the other hand, in deeper mineral horizons of tropical forest soils, positive relationships are usually found with pedogenic oxides, to which Hg is associated since the organic matter content is very low in these soil layers (Grimaldi et al., 2008; Guedron et al., 2009; Roulet et al., 1998).

Evaluation of Confidence: Low due to the fact that very few datasets were available. The lack of available data on related pollutants or environmental variables limits the ability to draw statistically significant comparisons.

Furthermore, as highlighted in “Methodology”, above, the datasets are not rigorously homogenous (depth, sieving etc.).

Information and Capacity Needs: Hg concentrations in soils depend on soil physico-chemical parameters on one hand but also largely on Hg inputs (wet and dry) on the other one. As the relative importance of sources vs. retention has not been clearly established (yet), it is not obvious if focusing on soil parameters is key to understand and predict Hg concentrations at the moment.

2.6.11 Key findings: Soil Hg observations

The available data was very sparse except for the USA and Australia and the available datasets covered different periods, making comparisons and interpretations very difficult. No sites were sampled over time preventing any evaluation of temporal trends and no MMHg data was available.

A clear difference between background and anthropogenic levels was difficult to establish with the available data due to the intrinsically high spatial variability of soil THg.

THg levels in soils seem to be regulated by the presence of soil organic matter on the surface and/or the presence of metallic oxides (e.g. iron or manganese oxides) at depth but the relationships observed can strongly vary.

References

- Almeida, M.D., Lacerda, L.D., Bastos, W.R., Herrmann, J.C., 2005. Mercury loss from soils following conversion from forest to pasture in Rondônia, Western Amazon, Brazil. *Environ. Pollut.* 137, 179–186. <https://doi.org/10.1016/j.envpol.2005.02.026>
- Ballabio, C., Jiskra, M., Osterwalder, S., Borrelli, P., Montanarella, L., Panagos, P., 2021. A spatial assessment of mercury content in the European Union topsoil. *Sci. Total Environ.* 769, 144755. <https://doi.org/10.1016/j.scitotenv.2020.144755>
- Cossa, D., Garnier, C., Buscail, R., Elbaz-Poulichet, F., Mikac, N., Patel-Sorrentino, N., Tessier, E., Rigaud, S., Lenoble, V., Gobeil, C., 2014. A Michaelis–Menten type equation for describing methylmercury dependence on inorganic mercury in aquatic sediments. *Biogeochemistry* 119, 35–43. <https://doi.org/10.1007/s10533-013-9924-3>
- Fahnestock, M.F., Bryce, J.G., McCalley, C.K., Montesdeoca, M., Bai, S., Li, Y., Driscoll, C.T., Crill, P.M., Rich, V.I., Varner, R.K., 2019. Mercury reallocation in thawing subarctic peatlands. *Geochem. Perspect. Lett.* 33–38. <https://doi.org/10.7185/geochemlet.1922>
- Gamby, R.L., Hammerschmidt, C.R., Costello, D.M., Lamborg, C.H., Runkle, J.R., 2015. Deforestation and cultivation mobilize mercury from topsoil. *Sci. Total Environ.* 532, 467–473. <https://doi.org/10.1016/j.scitotenv.2015.06.025>
- Grimaldi, C., Grimaldi, M., Guedron, S., 2008. Mercury distribution in tropical soil profiles related to origin of mercury and soil processes. *Sci. Total Environ.* 401, 121–129. <https://doi.org/10.1016/j.scitotenv.2008.04.001>
- Guedron, S., Grangeon, S., Lanson, B., Grimaldi, M., 2009. Mercury speciation in a tropical soil association; Consequence of gold mining on Hg distribution in French Guiana. *Geoderma* 153, 331–346. <https://doi.org/10.1016/j.geoderma.2009.08.017>
- Jiskra, M., Sonke, J.E., Obrist, D., Bieser, J., Ebinghaus, R., Myhre, C.L., Pfaffhuber, K.A., Wängberg, I., Kyllönen, K., Worthy, D., Martin, L.G., Labuschagne, C., Mkololo, T., Ramonet, M., Magand, O., Dommergue, A., 2018. A vegetation control on seasonal variations in global atmospheric mercury concentrations. *Nat. Geosci.* 11, 244–250. <https://doi.org/10.1038/s41561-018-0078-8>
- Lacerda, L.D., De Souza, M., Ribeiro, M.G., 2004. The effects of land use change on mercury distribution in soils of Alta Floresta, Southern Amazon. *Environ. Pollut.* 129, 247–255. <https://doi.org/10.1016/j.envpol.2003.10.013>
- Obrist, D., Agnan, Y., Jiskra, M., Olson, C.L., Colegrove, D.P., Hueber, J., Moore, C.W., Sonke, J.E., Helmig, D., 2017. Tundra uptake of atmospheric elemental mercury drives Arctic mercury pollution. *Nature* 547, 201–204. <https://doi.org/10.1038/nature22997>

- Obrist, D., Pearson, C., Webster, J., Kane, T., Lin, C.-J., Aiken, G.R., Alpers, C.N., 2016. A synthesis of terrestrial mercury in the western United States: Spatial distribution defined by land cover and plant productivity. *Sci. Total Environ.* 568, 522–535. <https://doi.org/10.1016/j.scitotenv.2015.11.104>
- O'Connor, D., Hou, D., Ok, Y.S., Mulder, J., Duan, L., Wu, Q., Wang, S., Tack, F.M.G., Rinklebe, J., 2019. Mercury speciation, transformation, and transportation in soils, atmospheric flux, and implications for risk management: A critical review. *Environ. Int.* 126, 747–761. <https://doi.org/10.1016/j.envint.2019.03.019>
- Olson, C., Jiskra, M., Biester, H., Chow, J., Obrist, D., 2018. Mercury in Active-Layer Tundra Soils of Alaska: Concentrations, Pools, Origins, and Spatial Distribution. *Glob. Biogeochem. Cycles* 32, 1058–1073. <https://doi.org/10.1029/2017GB005840>
- Olson, C.I., Geyman, B.M., Thackray, C.P., Krabbenhoft, D.P., Tate, M.T., Sunderland, E.M., Driscoll, C.T., 2022. Mercury in soils of the conterminous United States: patterns and pools. *Environ. Res. Lett.* 17, 074030. <https://doi.org/10.1088/1748-9326/ac79c2>
- Reimann, C., de Caritat, P., 2017. Establishing geochemical background variation and threshold values for 59 elements in Australian surface soil. *Sci. Total Environ.* 578, 633–648. <https://doi.org/10.1016/j.scitotenv.2016.11.010>
- Roulet, M., Lucotte, M., Saint-Aubin, A., Tran, S., Rhéault, I., Farella, N., De Jesus Da Silva, E., Dezencourt, J., Sousa Passos, C.-J., Santos Soares, G., Guimarães, J.-R.D., Mergler, D., Amorim, M., 1998. The geochemistry of mercury in central Amazonian soils developed on the Alter-do-Chão formation of the lower Tapajós River Valley, Pará state, Brazil. The present investigation is part of an ongoing study, the CARUSO project (IDRC-UFPa-UQAM), initiated to determine the sources, fate, and health effects of MeHg in the Lower Tapajós area. *Sci. Total Environ.* 223, 1–24. [https://doi.org/10.1016/S0048-9697\(98\)00265-4](https://doi.org/10.1016/S0048-9697(98)00265-4)
- Schneider, L., Fisher, J.A., Diéguez, M.C., Fostier, A.-H., Guimaraes, J.R.D., Leaner, J.J., Mason, R., 2023. A synthesis of mercury research in the Southern Hemisphere, part 1: Natural processes. *Ambio* 52, 897–917. <https://doi.org/10.1007/s13280-023-01832-5>
- Schuster, P.F., Schaefer, K.M., Aiken, G.R., Antweiler, R.C., Dewild, J.F., Gryziec, J.D., Gusmeroli, A., Hugelius, G., Jafarov, E., Krabbenhoft, D.P., Liu, L., Herman-Mercer, N., Mu, C., Roth, D.A., Schaefer, T., Striegl, R.G., Wickland, K.P., Zhang, T., 2018. Permafrost Stores a Globally Significant Amount of Mercury: Permafrost Stores a Globally Significant Amount of Mercury. *Geophys. Res. Lett.* 45, 1463–1471. <https://doi.org/10.1002/2017GL075571>
- Skylberg, U., Bloom, P.R., Qian, J., Lin, C.M., Bleam, W.F., 2006. Complexation of mercury(II) in soil organic matter: EXAFS evidence for linear two-coordination with reduced sulfur groups. *Environ. Sci. Technol.* 40, 4174–4180.
- St. Pierre, K.A., St. Louis, V.L., Kirk, J.L., Lehnher, I., Wang, S., La Farge, C., 2015. Importance of Open Marine Waters to the Enrichment of Total Mercury and Methylmercury in Lichens in the Canadian High Arctic. *Environ. Sci. Technol.* 49, 5930–5938. <https://doi.org/10.1021/acs.est.5b00347>
- Tarbier, B., Hugelius, G., Kristina Sannel, A.B., Baptista-Salazar, C., Jonsson, S., 2021. Permafrost Thaw Increases Methylmercury Formation in Subarctic Fennoscandia. *Environ. Sci. Technol.* <https://doi.org/10.1021/acs.est.0c04108>
- Varty, S., Lehnher, I., St. Pierre, K., Kirk, J., Wisniewski, V., 2021. Methylmercury Transport and Fate Shows Strong Seasonal and Spatial Variability along a High Arctic Freshwater Hydrologic Continuum. *Environ. Sci. Technol.* 55, 331–340. <https://doi.org/10.1021/acs.est.0c05051>
- Wang, X., Bao, Z., Lin, C.-J., Yuan, W., Feng, X., 2016. Assessment of Global Mercury Deposition through Litterfall. *Environ. Sci. Technol.* 50, 8548–8557. <https://doi.org/10.1021/acs.est.5b06351>
- Wang, X., Yuan, W., Lin, C.-J., Zhang, L., Zhang, H., Feng, X., 2019. Climate and Vegetation As Primary Drivers for Global Mercury Storage in Surface Soil. *Environ. Sci. Technol.* 53, 10665–10675. <https://doi.org/10.1021/acs.est.9b02386>
- Yang, Z., Fang, W., Lu, X., Sheng, G.-P., Graham, D.E., Liang, L., Wulfschleger, S.D., Gu, B., 2016. Warming increases methylmercury production in an Arctic soil. *Environ. Pollut.* 214, 504–509. <https://doi.org/10.1016/j.envpol.2016.04.069>
- Zhang, Z.-Y., Li, G., Yang, L., Wang, X.-J., Sun, G.-X., 2020. Mercury distribution in the surface soil of China is potentially driven by precipitation, vegetation cover and organic matter. *Environ. Sci. Eur.* 32, 89. <https://doi.org/10.1186/s12302-020-00370-1>

Zhao, L., Meng, B., Feng, X., 2020. Mercury methylation in rice paddy and accumulation in rice plant: A review. *Ecotoxicol. Environ. Saf.* 195, 110462. <https://doi.org/10.1016/j.ecoenv.2020.110462>

Zhou, J., Obrist, D., 2021. Global Mercury Assimilation by Vegetation. *Environ. Sci. Technol.* 55, 14245–14257. <https://doi.org/10.1021/acs.est.1c03530>

Zhou, J., Obrist, D., Dastoor, A., Jiskra, M., Ryjkov, A., 2021. Vegetation uptake of mercury and impacts on global cycling. *Nat. Rev. Earth Environ.* 2, 269–284. <https://doi.org/10.1038/s43017-021-00146-y>

2.7 Sediment Cores

2.7.1 What data are available?

The lake sediment core archive has global geographic coverage, with core data from every continent (Figure 20). Of the 446 cores compiled, the majority (390 cores, 87%) were collected from the Northern Hemisphere, primarily from Canada (182 cores, 40.8% of the total), the USA (161 cores, 36%), and China (20 cores, 4.5%) (Figure 21). Cores from the Tropics (defined as between -30° and $+30^{\circ}$ latitude) accounted for 10.5% of the dataset (47 cores), with the majority originating from Brazil (13 cores) and Peru (5 cores). The Southern Hemisphere contributed the fewest cores (9 cores, 2%), with samples from Argentina (3 cores), New Zealand (3 cores), Australia (2 cores), and the South Shetland Islands (1 core) (Figure 22). Ancillary measurements for other environmental variables, such as Total Organic Carbon (TOC), aluminum (Al), and percent loss on ignition (%LOI) are available for most of the cores.



Figure 20. A map showing the distribution of the available data for sediment cores.

From the total 446 cores, 339 were collected from lakes, 28 from coastal regions and mangrove estuaries, and 19 from reservoirs. A total of 347 cores (77.8%) were collected from locations without a known nearby anthropogenic source, while 99 cores (22.2%) were influenced by local industry. Cores collected from extremely remote regions—where no human settlements or infrastructure, including roads or rudimentary camps, were present—represented 168 cores (37.7%) of the dataset. In contrast, cores collected from areas within dense human settlements, characterized by buildings, roads, and agricultural activities, accounted for 52 cores (11.7%).

The dataset was compiled from publicly available data sourced from 110 peer-reviewed journal articles, government reports, and thesis publications. An additional 86 publications that reported mercury concentrations in dated lake sediment cores were identified but not included because the raw data was not publicly available.

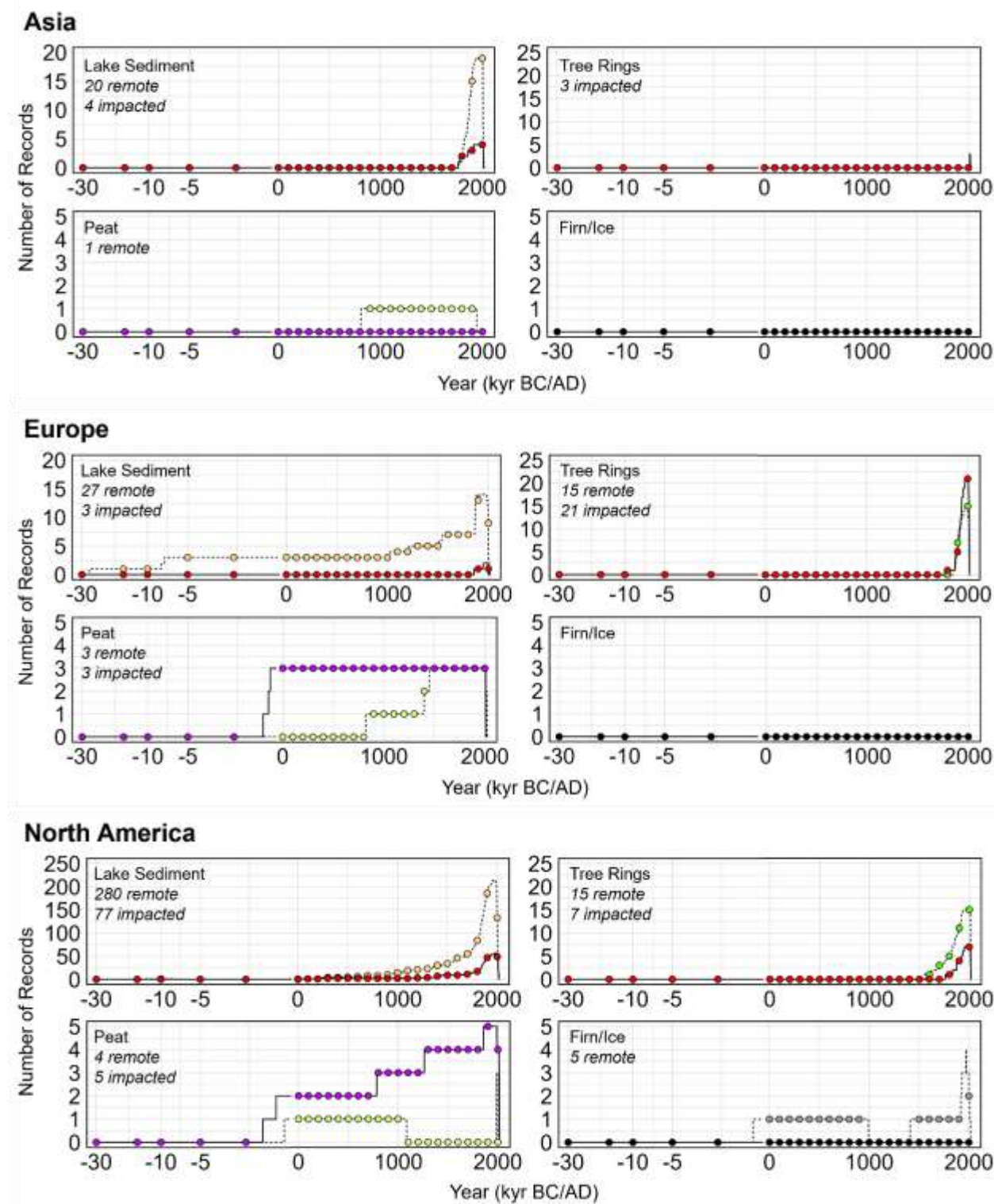


Figure 21. Number and length of peat cores from impacted (purple) and unimpacted (light green) areas; sediment cores from impacted (red squares) and unimpacted (yellow) areas; and tree rings from impacted (red circles) and unimpacted (green circles) areas from Asia, Europe, and North America.

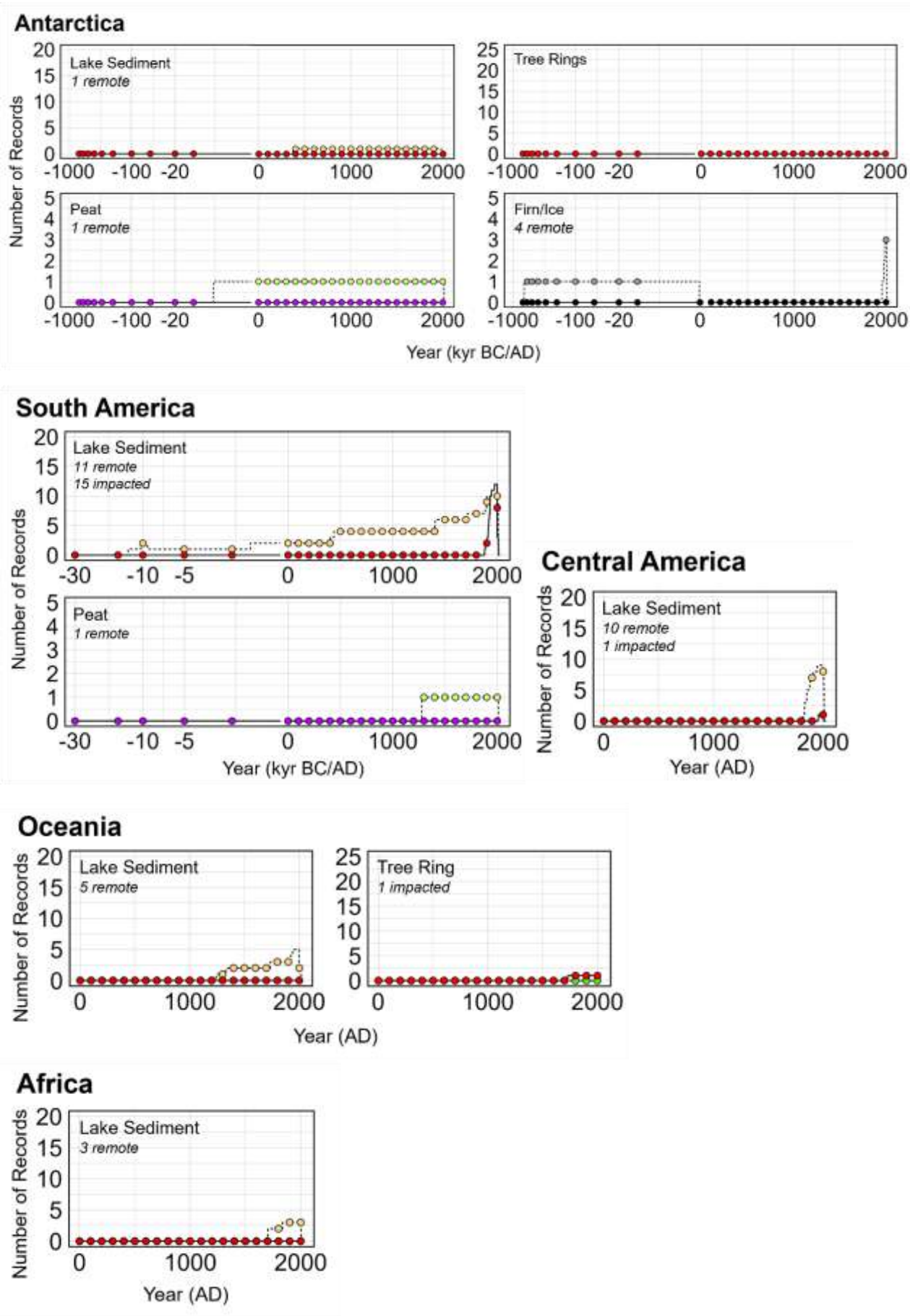


Figure 22. Number and length of peat cores from impacted (purple) and unimpacted (light green) areas; sediment cores from impacted (red squares) and unimpacted (yellow) areas; tree rings from impacted (red circles) and unimpacted (green circles) areas from South and Central America, Africa and Oceania. Number and length of ice cores from Antarctica.

2.7.2 What are the current levels of THg and MMHg (and other species/fractions if applicable) observed in sediment cores that are remote from or near anthropogenic sources?

Information Used. To assess current THg levels (measured as concentrations and fluxes), lake sediment core data representing the time period 1990–2020 was used. The Hg concentration dataset ($\mu\text{g g}^{-1}$) includes 396 lake sediment cores from 96 published references globally, with 71% (281 cores) from North America and 29% (114 cores) from watersheds with known anthropogenic Hg sources (Figure 1). The Hg flux dataset ($\mu\text{g m}^{-2} \text{yr}^{-1}$) includes 467 cores from 109 published references, with 70% (329 cores) from North America and 24% (113 cores) from impacted watersheds (Supplemental Figure 1).

Methodology. To assess current Hg levels, THg concentrations ($\mu\text{g g}^{-1}$) and/or THg fluxes ($\mu\text{g m}^{-2} \text{yr}^{-1}$) measured in the top sediment horizons and corresponding to the period 1990–2020 were examined. THg fluxes represent the rate of Hg deposition to the lake bottom originating from atmospheric deposition and catchment inputs. They are calculated as the product of Hg concentrations and sediment mass accumulation rates ($\text{g cm}^{-2} \text{yr}^{-1}$) and have been proven useful for reconstructing changes in Hg deposition over time (Dastoor et al. 2025, Muir et al. 2009, Engstrom et al. 2014, Wiklund et al. 2017, Roberts et al. 2019, Roberts et al. 2021). Hg concentration and flux data were log-transformed prior to analysis. Sample size (n), median (M), and mean (μ) log values were calculated for each modern time period examined (i.e., 1990–2020 as well as 1990–2000, 2000–2010, and 2010–2020; Figures 3 and 4). Wilcoxon rank-sum tests were used to test if Hg concentrations and fluxes were statistically different among impact status groups and time periods.

The sites where sediment cores were collected from were classified as unimpacted if no known local Hg source was identified in the publication the data originated from while those that identified an anthropogenic Hg source in the watershed were classified as impacted. Anthropogenic Hg sources within the dataset include artisanal small scale gold mining (ASGM) sites, large scale gold-mining, other mining and non-ferrous metal processing facilities, oil and natural gas processing facilities, cement clinker production facilities, chlor alkali plants, coal-fired power plants, military zones, and waste incineration facilities.

Answer/ response. Current (1990–2020) global median Hg concentrations measured in lakes sediment cores collected from unimpacted sites are $\sim 0.113 \mu\text{g g}^{-1}$ (range $0.004 - 9.47 \mu\text{g g}^{-1}$, $n = 282$) and are significantly lower than those from impacted sites for the same time period (median $\sim 0.132 \mu\text{g g}^{-1}$, range $0.007 - 551 \mu\text{g g}^{-1}$, $n = 114$, $p < 0.001$). Similarly, current sediment derived depositional Hg fluxes in unimpacted sites are $\sim 28.6 \mu\text{g m}^{-2} \text{yr}^{-1}$ (range $0.002 - 2257 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 354$) and are significantly lower than those measured in impacted sites (median $\sim 86.8 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $0.1 - 86,507 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 111$, $p < 0.001$).

To assess Hg levels in the most recent time period available, Hg concentrations and fluxes were also examined for the 2010–2020 time period. However, the number of impacted sediment cores available for this period is substantially lower (~ 40 cores) than for the previous two decades (> 80 cores). In unimpacted sites, median Hg concentrations over the most recent time period are $\sim 0.10 \mu\text{g g}^{-1}$ (range $0.02 - 9.77 \mu\text{g g}^{-1}$, $n = 60$) and surprisingly, are significantly greater than those measured in sediment cores from impacted sites (median $0.065 \mu\text{g g}^{-1}$, range $0.017 - 76.7 \mu\text{g g}^{-1}$, $n = 45$, $p < 0.01$). Comparison of sediment Hg concentrations over the three most current decades (1990–2000 versus 2000–2010 and 2010–2020) demonstrates that in unimpacted sites, Hg concentrations are comparable across these periods ($p > 0.05$). In contrast, in impacted sites, Hg concentrations are significantly lower in the most recent time period (2010–2020) compared to the previous two decades ($p < 0.001$). Sediment derived depositional Hg fluxes showed similar trends, with recent (2010–2020) fluxes in unimpacted sites (median $\sim 33.9 \mu\text{g m}^{-2} \text{yr}^{-1}$; range $0.51 - 9498$, $n = 68$) significantly higher than those in impacted cores (median $19.9 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $3.3 - 6644 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 39$, $p < 0.001$). In unimpacted sites, Hg fluxes are significantly higher over the 2010–2020 period compared to the previous two decades ($p < 0.005$) while in impacted sites, the opposite is true with measured Hg fluxes significantly lower in the most recent decade compared to the previous 20 years ($p < 0.0001$). Together, these results suggest that while Hg levels in unimpacted sites have increased over the most recent time period (2010–2020) compared to the previous two decades, Hg levels near point sources have

decreased over this same period. While these trends could result from the lower sample sizes available for the 2010–2020 period compared to earlier decades, the decreasing trend in impacted sites likely reflects decreased Hg inputs due to emission controls, especially in parts of the world with the largest datasets (i.e., North America). Sediment cores from impacted sites also exhibit substantial variability over the full 1990–2020 period, with Hg concentrations and fluxes spanning ~5 and 8 orders of magnitude, respectively. In contrast, variability in the 2010–2020 subset is markedly lower, with concentrations and fluxes spanning 1 and 4 orders of magnitude, respectively. The markedly lower variability in the 2010–2020 dataset, resulting in part from smaller dataset may have made it more likely to detect a trend, underscoring the need for continued measurements to encompass a greater range of sites and resolve recent trends.

Consistency with Existing Literature. For unimpacted sites, the observed values and recent increases in both Hg concentrations and fluxes (i.e., between 2010–2020 compared to the previous two decades) align with findings from two lake sediment core compilations for North America, both of which are included in this global dataset. Roberts et al. (2021) analyzed 82 cores from across Canada and reported regional differences, with Hg levels decreasing in western Canada but increasing in eastern Canada over the period 1990–2018. The magnitude of Hg flux varied between region and impact status, with lowest fluxes measured in remote central Alberta ($5.1 \pm 1.9 \mu\text{g m}^{-2} \text{yr}^{-1}$) and highest fluxes in close proximity to a large metals smelter (Flin Flon, Manitoba) that was recent decommissioned ($436 \pm 614 \mu\text{g m}^{-2} \text{yr}^{-1}$), while highly industrial regions such as the Greater Toronto Area displayed intermediate values ($71.2 \pm 30.8 \mu\text{g m}^{-2} \text{yr}^{-1}$). Similarly, Drevnick et al. (2016) analyzed 165 sediment cores from across western North America and identified median Hg fluxes of $27 \mu\text{g m}^{-2} \text{yr}^{-1}$ for the period 2000–2005 in unimpacted cores, similar to the median flux of $23 \mu\text{g m}^{-2} \text{yr}^{-1}$ for this dataset. Additionally, while the authors calculated a median Hg flux of $19 \mu\text{g m}^{-2} \text{yr}^{-1}$ for the most recent time period measured (2005–2010), the authors deemed this value untrustworthy compared to earlier intervals due to low sample numbers ($n = 35$), which aligns with our interpretation that the decline in Hg flux in impacted cores may reflect lower sample numbers for this time period.

Evaluation of Confidence. The findings are consistent with previous studies, lending confidence in the results. However, impacted core classification was based on literature-reported proximity to Hg sources, which may have overlooked unidentified sources affecting some “unimpacted” sites. Moreover, the relatively low number of sediment cores spanning the most recent time period ($n=41$ for 2010–2020 for impacted sites) complicates distinguishing between true environmental changes and artifacts of sampling discrepancy.

Information and Capacity Needs. The dataset remains heavily weighted toward North America (67–81% of cores), with major gaps in Africa (6 cores), Oceania (4 cores), and regions of Eastern Europe/ Mediterranean/ Middle East (1 core) (Figures 1–2, Supplemental Figure 1). There is also much less data for the most recent time period ($n=61$ for the 2010–2020 time period) compared to the earlier two decades ($n=183$ –269 for the 1990–2000 and 2000–2010 time periods, respectively) highlighting the need for new lake sediment core collections, especially at previously cored lakes, to enable more robust trend assessments. There are no dated lake sediment cores with MMHg data available as MMHg is not well preserved down core. However, MMHg could be analyzed in surface lake sediment core slices to assess spatial patterns in sediment MMHg levels. Finally, a more systematic classification method for Hg source presence, including specific distance to source, is recommended.

2.7.3 How have levels of THg and MMHg (and other species/fractions if applicable) observed in sediment cores changed over the available record?

Information used. To assess changes in THg levels over time, temporal trends in concentration, flux, and enrichment factors were examined. Hg concentrations ($\mu\text{g g}^{-1}$) data were available for 435 lake sediment cores from 104 published references globally, with 67% (290 cores) from North America and 27% (116 cores) known to have an anthropogenic Hg source in the watershed. Hg flux ($\mu\text{g m}^{-2} \text{yr}^{-1}$) data was available for 488 cores from 116 published references, with 69% (338 cores) from North America and 118 from impacted watersheds. Flux-calculated enrichment factors (EFs) were available for 248 cores from 74 published references, with 82% (203 cores) from North America and 45 from impacted watersheds.

Methodology. Hg concentrations and fluxes were log-transformed for analysis. Sample size (n), median (M), and mean (μ) log values were calculated for different time periods spanning the period 1500–2020 (Figures 3 and 4). Based on data availability and time periods of significance for Hg emission and deposition, as previously defined by Streets et al. 2019, Hg concentration and flux data were examined over the following different time periods: pre-anthropogenic period (pre-1500); pre-industrial (1760–1850), post-industrial (1850–1950), and decadal

between 1950–2020. Enrichment factors (EFs) were calculated for each sediment core as the ratio of untransformed mean decadal Hg flux versus mean pre-industrial (1760–1850) flux as follows:

$$\text{Enrichment factor}_{\text{period } x \text{ vs. baseline}} = \frac{\text{Hg flux}_{\text{period } x}}{\text{Hg flux}_{\text{baseline}}}$$

The pre-industrial period (1760–1850) was used as the baseline rather than the pre-anthropogenic period (pre-1500) to increase the sample size. In unimpacted sites, Hg concentrations and fluxes were statistically similar between the pre-anthropogenic and pre-industrial periods ($p > 0.05$), supporting the use of either as a baseline. In impacted sites, Hg concentrations also did not differ significantly between the pre-anthropogenic and pre-industrial periods (median EF = 1.13, $p > 0.05$), but Hg fluxes were significantly higher during the pre-industrial period (median EF = 1.89, $p < 0.001$), suggesting potential early influence of anthropogenic activity.

The sites that sediment cores were collected from were classified as unimpacted or impacted as described above under the methodology for Question 1.

Answer/ response. In unimpacted sites, global Hg concentrations and fluxes over the pre-industrial (1760–1850) period were low ($\sim 0.061 \mu\text{g g}^{-1}$ and $6.83 \mu\text{g m}^{-2} \text{yr}^{-1}$, respectively, $n = 170\text{--}206$) and generally comparable to those in impacted sites (median $\sim 0.053 \mu\text{g g}^{-1}$ and $5.21 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 44$) (Figures 3–4). After the pre-industrial period, Hg concentrations and fluxes in unimpacted cores rose steadily peaking in the most modern period measured (2010–2020; $\sim 0.10 \mu\text{g g}^{-1}$ and $33.9 \mu\text{g m}^{-2} \text{yr}^{-1}$, respectively $n = 60\text{--}68$). In impacted sites, global Hg concentrations and fluxes rose more steeply than in unimpacted sites, peaking in 1990–2000 ($0.16 \mu\text{g m}^{-2} \text{yr}$ and $282.3 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 101\text{--}104$) before declining sharply to significantly lower values in 2010–2020 ($0.07 \mu\text{g g}^{-1}$ and $\sim 19.9 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 39\text{--}45$). As discussed in detail under Question 1, the sharp decline in both Hg concentrations and fluxes in the most modern sediment horizons in impacted sites may reflect decreased Hg emissions and deposition due to emission controls or sampling bias due to the lower number of cores available for this time period. Hg concentrations and fluxes were significantly lower in unimpacted sites compared to impacted sites for all post-1850 time periods except the period 2010 – 2020 as discussed above in detail.

In unimpacted sites, EFs, which are particularly useful for summarizing temporal trends, steadily increased from ~ 1.6 (range 0.07 – 18.7, $n = 201$) in 1900–1950, peaking in the most recent time periods at ~ 3.4 (range 0.4 – 36.1, $n = 116$) in 2000–2010 and 3.6 (range 0.7 – 23.8, $n = 27$) in 2010–2020. In impacted sites, EFs increased from ~ 3.0 (range 0.9 – 167.4, $n = 44$) in 1900–1950 and, in contrast to Hg concentrations and fluxes, reached maximum values in most recent time periods measured (5.1; range 0.5 – 146.1, $n = 33$ in 2000–2010 and 5.7; range 0.3 – 145.9, $n = 16$ in 2010–2020). However, concentration and flux data were available for a much larger number of cores ($n = 41$) than EFs ($n = 16$). Globally, EFs are significantly higher in impacted sites compared to unimpacted sites for all post-industrial time periods examined ($p < 0.05$, Figure 5) except for the most recent 2010 – 2020 period ($p = 0.10$) as discussed in Question 1.

Consistency of Answer with Existing Literature. Hg fluxes reported here are generally higher than values in other large lake sediment core compilations (e.g., Engstrom et al. 2014 range 4 – $12 \mu\text{g m}^{-2} \text{yr}^{-1}$ for the North America, South America and East Africa regions; Roberts et al. 2021 range 2 – $17 \mu\text{g m}^{-2} \text{yr}^{-1}$ for 82 sites across Canada; Drevnick et al. 2016 range 7 – $27 \mu\text{g m}^{-2} \text{yr}^{-1}$); many of which have been included in this dataset (Muir et al. 2009, Drevnick et al. 2016, Chen et al. 2024; Roberts et al. 2021). The higher flux values reported here likely reflect that we were not able to correct Hg fluxes for sediment focusing, the process by which sediments are redistributed from shallow to deeper areas of a lake basin. Flux-based EFs are robust even without sediment focusing corrections and were much more comparable to existing literature. For example, for unimpacted sites, EFs reported here for the global dataset ($\sim 2\text{--}4$) align well with estimates of modern vs. pre-industrial EFs reported in Engstrom et al. 2014 (range 2 – 5), Muir et al. 2009 (range 0.5 – 7), Drevnick et al. 2016 (~ 4), Biester et al. 2007 (~ 3), and Chen et al. 2024 (~ 5). Emission inventories suggest that globally, Hg emissions to air increase greatly around 1970 and remain high thereafter showing only modest declines (Streets et al. 2019). This is consistent with EF values being highest in the most modern time periods as a result of continued accumulation of Hg in the environment and the lag time in Hg being transported from catchments into lakes, even if in some regions (e.g., N. America) emissions to air are thought to have declined more significantly.

Evaluation of Confidence. Flux-based EFs align well with literature values (Engstrom et al. 2014; Muir et al. 2009, Drevnick et al. 2016; Biester et al. 2007; Chen et al. 2024), supporting confidence in the results. As discussed above, unlike sediment-derived fluxes, EFs are robust to the mathematical effects of sediment focusing corrections, allowing for more reliable comparison to literature values. The use of the pre-anthropogenic baseline

for the EF calculation increased the samples size but may have lowered modern EFs in regions with early anthropogenic influence (i.e. some regions of Central American, South American, and European; see Question 6 for a detailed discussion).

Information and Capacity Needs. The dataset remains heavily weighted toward North America (67–81% of cores depending on metric) with data for other regions, especially in Africa (6 cores), and Oceania (4 cores) needed. Additional data is also needed for the most recent time periods (2010–2020 and onward) as the low sample numbers for 2010–2020 compared to the earlier two decades make it difficult to assess current temporal trends with confidence.

Focusing factors to estimate sediment focusing values for all world regions would allow quantitative spatial comparison of depositional fluxes. Currently focus factor estimates for Asia ($n = 7$), Africa ($n = 4$), Europe ($n = 6$), Oceania ($n = 3$), South America ($n = 0$), and Central America ($n = 0$) are especially lacking. The focusing factors used to make the sediment focusing correction are based on the known significant linear relationship between atmospheric lead-210 fall out and latitude that, to our knowledge, has only been reported for North America (e.g., Muir et al. 2009).

Finally, a more systematic method for classifying unimpacted versus impacted sites is needed to compare sediment derived Hg concentrations, fluxes and EFs among unimpacted and impacted sites. Future work could incorporate Hg emission data to categorize impacted versus unimpacted sites, though these may underrepresent sources like artisanal small-scale gold mining, where atmospheric emissions are more uncertain.

2.7.4 How do temporal trends of Hg levels observed in sediment cores compare to or co-vary with variations and trends of related pollutants or environmental variables over the available record? (TOC, SOM, DOC, TSS)?

Information used. To assess how THg levels covary with other environmental variables, temporal trends of THg concentration, THg flux, percent loss-on-ignition (% LOI) and aluminum (Al, $\mu\text{g g}^{-1}$) were compared. Measurements of THg concentrations ($\mu\text{g g}^{-1}$) and %LOI were available for 86 lake sediment cores from 20 published references, with 48% (42 cores) known to have an anthropogenic Hg source in the watershed (i.e. impacted). Measurements of THg concentration and Al were available for 162 lake sediment cores from 34 published references with 37% (60 cores) from impacted watersheds. Measurements of Hg flux ($\mu\text{g m}^{-2} \text{ yr}^{-1}$) and %LOI were available for 80 cores from 16 published references, with 50% (40 cores) from impacted watersheds, while measurements of Hg flux and Al were available for 143 lake sediment cores from 25 published references, with 39% (56 cores) from impacted watersheds.

Methodology. The regions that sediment cores were collected from were classified as unimpacted or impacted as described under the methodology for Question 1. Spearman correlations were used to assess relationships between THg concentration or flux and %LOI or aluminum (Al) across different time periods spanning from 1500–2010. %LOI was used as a proxy for organic matter concentration, which influences Hg binding and methylation potential, while Al served as an indicator of lithogenic mineral inputs of catchment-derived material. Correlations were calculated for the same time bins used for temporal trends analysis in Question 2. Correlations were calculated only where ≥ 5 paired observations were available per time bin. For each time bin, Spearman's ρ , p -values, and 95% confidence intervals (via Fisher z -transformation) were computed. Correlation strength was classified as very weak ($|\rho| < 0.2$), weak ($0.2–0.4$), moderate ($0.4–0.6$), or strong ($|\rho| \geq 0.6$), and further grouped by direction (positive or negative) and significance ($p < 0.05$).

Answer/ Response. In unimpacted sites, THg concentrations show positive relationships with %LOI over the entire temporal record, with significant positive relationships observed during the earliest time periods examined (1500–1850; $\rho = 0.49–0.55$ or moderate, $p < 0.05$, $n = 8–15$) and between 1950–2010 ($\rho = 0.26–0.57$ or weak to moderate, $p < 0.05$) (Figure 6, Supplemental Figure 2). In impacted sites, the relationship between THg and %LOI is weak, negative, and not significant throughout most of the record (for 1500–2000, $\rho = -0.26$ to 0.07 , $p > 0.05$, $n = 7–31$), likely reflecting the role of nearby anthropogenic sources of varying size and nature (i.e., atmospheric emissions versus releases to water) in controlling Hg levels in lake sediments (Figure 6, Supplemental Figure 2). Of the seven impacted cores with data prior to 1800, two are from Peru nearby the Santa Bárbara cinnabar mine that operated from ~800 BCE to ~1975 CE (Cooke et al. 2009). Excluding these two cores and leaving only the five impacted cores from the Yellowknife region in Canada's Northwest Territories, the correlation between THg and %LOI over the 1500–1850 period shifts to a positive but statistically variable trend ($\rho = 0.27–0.44$, $p = 0.008–0.2$, $n = 5$). The positive correlations between THg concentration and %LOI over all time periods in unimpacted cores, and in some early periods in impacted cores demonstrates the well-established co-transport of Hg and organic matter in aquatic ecosystems, and ultimately, lake sediment (Sanei and Goodarzi, 2006). Unlike Hg concentrations, Hg fluxes in unimpacted cores show significant and weak to moderate strength negative

correlations to %LOI during all post-industrial (after ~1850) periods examined ($\rho = -0.3$ to -0.5 , $p < 0.05$, $n = 26-36$), with similar patterns observed in impacted sites (Supplemental Figure 3). These results demonstrate the dominant role of sedimentation rate in controlling Hg deposition rates to lake sediments.

For Al, the relationship between THg and Al **concentrations** is weak and insignificant over most of the time periods examined in both unimpacted and impacted sites, which isn't surprising given that Hg and Al are not co-transported within aquatic ecosystems (Supplemental Figure 2). In contrast, in unimpacted sites, THg **fluxes** show strong positive correlations with Al concentrations in all periods examined ($\rho = 0.43-0.76$, $p < 0.05$, $n = 3-77$) reflecting the dominant role of sedimentation rates, particularly through erosion and the delivery of mineral rich catchment material, in controlling sediment derived Hg depositional fluxes (Figure 6, Supplemental Figure 2). In impacted sites, strong, significant positive correlations between Hg flux and Al concentration were also observed in pre-anthropogenic (1500–1760) and most recent (2000–2020) time periods ($\rho = 0.45-0.58$, $p < 0.05$, $n = 14-35$), suggesting that recent increases in Hg delivery to sediments is tied to land disturbance, runoff, or enhanced erosion (Figure 6). Overall, the results suggest that mineral sediment inputs strongly influence depositional Hg flux rates over long time scales across diverse environmental conditions, including both disturbed and undisturbed systems.

Consistency of Answer with Existing Literature. The strong and positive correlations between THg concentration and organic carbon concentration are consistent with existing literature, including findings in China (Wu et al. 2013), Africa (Campbell et al. 2003), Canada (Stern et al. 2009; Sanei et al. 2014; Ethier et al. 2010), Oceania (Schneider et al. 2024), Europe (Jian et al. 2011; Rutkowski et al. 2021) and South America (Biester et al. 2017). As noted by Sanei and Goodarzi (2006), strong Hg-organic matter correlations typically arise from specific organic fractions, such as thermally labile, algal-derived material, which affect chemical reactivity and binding. The observed positive relationship between Hg flux and Al in both unimpacted and impacted lake sediments is consistent with previous research identifying mineral lithogenic inputs as key drivers of Hg delivery to lakebeds. Studies such as Rydberg et al. (2012) and Muir et al. (2009) demonstrated that Al and mineral-associated matter explain a substantial portion of Hg accumulation, highlighting the role of lithogenic inputs in delivering Hg to lake sediments.

Evaluation of Confidence. The patterns identified are consistent with key observations from existing literature. However, confidence is limited by the relatively small number of references (9–12 references, ~60 cores) supporting the dataset, especially in comparison to other questions in this report that draw on larger compilations (>200 cores).

Information and Capacity Needs. There are limited sediment cores from a limited number of world regions with concurrent measurements of Hg and %LOI (i.e., $n=13-15$ from North America, 29–32 from Europe, 22 from Central America, 15 from South America, and 1–2 from Africa and no data for Oceania or Asia for THg and %LOI). While there is a slightly greater availability of THg and Al sediment data, most is still clustered in North America, Europe, and Central and South America. Additionally, standard %LOI and %OM metrics do not distinguish between organic fractions that influence Hg binding potential, such as particle surface area or chemical reactivity (Sanei and Goodarzi, 2006). Improved organic matter characterization would provide information on how organic matter quality—rather than quantity—affects Hg accumulation in lake sediments.

2.7.5 How do THg and MMHg levels (and other species/fractions if applicable) co-vary temporally in sediment cores?

This question cannot be addressed. MMHg is not faithfully preserved and archived in lake sediments as it is continually produced and decomposed by microbial organisms present in sediments. Therefore, MMHg concentrations in lake sediment cores do not provide an indication of temporal trends and we cannot evaluate whether THg and MMHg co-vary temporally in lake sediments.

2.7.6 How do local THg and MMHg levels (and other species/fractions if applicable) found in sediment cores differ by world regions?

Information used. Sediment core derived THg concentration, flux and EF data were compiled for 31 Intergovernmental Panel on Climate Change (IPCC) regions (Iturbide et al. 2020) across all continents with the EF dataset covering 26 of these regions (Figures 1 and 2, Supplemental Figure 1). Regional core counts are summarized in Figure 7 and Supplemental Figures 4-5 with greatest data coverage in North and South America and Europe. Regions with only one core were excluded from regional trend figures (East-Central Asia, Russian-Arctic, Mediterranean, North Central America, South Australia, West Antarctica, and West/Central Europe). See Question 2 for further details on data sources.

Methodology. Hg concentrations and fluxes were log-transformed for analysis. Cores were grouped by impact status, continent, and/or IPCC region. Enrichment factors (EFs) were calculated from Hg fluxes following the methodology described under Question 2. Sample size (n), median (M), and mean (μ) log values were calculated for different time periods spanning the modern 1990–2020 period (Supplemental Figure 4 for Hg concentration and Figure 5 for Hg flux). Statistical significance of differences between regions was assessed using Wilcoxon rank-sum tests. To evaluate regional differences within a given time bin or period, Kruskal-Wallis tests were used to detect overall variation among IPCC regions, followed by Dunn's test with Benjamini-Hochberg correction to identify specific pairwise differences in Hg flux and their statistical significance (padj).

Answer/ Response

Unimpacted Sites: In unimpacted sites, current (1990–2020) THg concentrations are generally highest in Northern Hemisphere regions, ranging from $0.09 \mu\text{g g}^{-1}$ in N.W. North America to $0.27 \mu\text{g g}^{-1}$ in N. Europe. The Tropics have lower values ($0.04 \mu\text{g g}^{-1}$ in the Caribbean to $0.09 \mu\text{g g}^{-1}$ in S. Central America), as does the Southern Hemisphere ($\sim 0.07 \mu\text{g g}^{-1}$ in S. South America and S. Australia). Median THg concentrations ranked as follows: N. Europe > E. North America > N.E. North America > C. North America > W. North America > S. Central America > N.W. North America > S.E. Asia > Arctic Ocean > S. Australia > S. South America > S.E. South America > Mediterranean > Caribbean > Tibetan Plateau. Concentrations in E. North America, N.E. North America, C. North America, N. Europe, and the 2010–2020 period in W. North America were statistically similar and significantly higher than other regions (padj < 0.05).

In contrast to THg concentrations, current median THg fluxes are highest in tropical regions, particularly in S.E. South America ($502 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $80 - 1830 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 2), S. Central America ($208 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $0.02 - 1734 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 11), and S.E. Asia ($113 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $38 - 658 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 3) while the lowest fluxes are in E. Asia ($0.011 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $0.008 - 16.5 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 3), E.C. Asia ($0.021 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $0.005 - 0.03 \mu\text{g m}^{-2} \text{yr}^{-1}$, n=2), and S. Eastern Africa ($0.026 \mu\text{g m}^{-2} \text{yr}^{-1}$, range $0.01 - 0.09 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 4). Differences in global trends among concentrations and fluxes reflects the important role of sedimentation rates in governing Hg deposition rates to lake sediments. Median THg fluxes across all IPCC regions ranked as follows: S.E. South America > S. Central America > N. Europe > S.E. Asia > Caribbean > Arctic Ocean > Mediterranean > W. North America > E. North America > New Zealand > S. Eastern Africa > E.C. Asia > N. E. North America > Tibetan-Plateau > C. North America > S. W. South America > N.W. North America > E. Asia > Greenland/Iceland > S. South America.

Impacted Sites: In impacted sites, current median Hg concentrations are highest in N.E. North America ($\sim 1.8 \mu\text{g g}^{-1}$; range $0.007 - 19 \mu\text{g g}^{-1}$, n= 13) and N.W. South America ($0.56 \mu\text{g g}^{-1}$; $0.09 - 6.8 \mu\text{g g}^{-1}$, n= 4), and lowest in E. Siberia ($0.03 \mu\text{g g}^{-1}$; $0.03 - 0.05 \mu\text{g g}^{-1}$, n= 3) and S. Central America ($0.05 \mu\text{g g}^{-1}$; $0.02 - 0.9 \mu\text{g g}^{-1}$, n= 14). Median Hg fluxes exceed $500 \mu\text{g m}^{-2} \text{yr}^{-1}$ in several regions, with extreme values in N.W. South America ($2907 \mu\text{g m}^{-2} \text{yr}^{-1}$; $45 - 15,109 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 4), the Caribbean ($1620 \mu\text{g m}^{-2} \text{yr}^{-1}$; $211 - 9600 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 5), S.E. South America ($763 \mu\text{g m}^{-2} \text{yr}^{-1}$; $130 - 51,744 \mu\text{g m}^{-2} \text{yr}^{-1}$, n= 2), and N.E. North America ($512 \mu\text{g m}^{-2} \text{yr}^{-1}$; 0.7

– 4883 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=13$). The lowest fluxes occurred in C. North America (11.4 $\mu\text{g m}^{-2} \text{yr}^{-1}$; 0.1 – 27 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=6$), and N.W. North America (13.3 $\mu\text{g m}^{-2} \text{yr}^{-1}$; 2 – 92 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=21$).

Across impacted and unimpacted cores, some regions exhibit consistent Hg rankings that may reflect natural patterns, with high fluxes in S.E. South America and N. Europe, high concentrations in N.E. and E. North America and low fluxes in E. Asia, S. Eastern Africa and N.W. North America. By contrast, certain regions rank consistently across concentration and flux (e.g., N.E. North America, N. Europe), whereas others display discrepancies (e.g., N.W. South America, C. North America, Caribbean), which may reflect regional differences in sedimentation rates, sampling density, or Hg sources.

Consistency with Existing Literature. Hg flux values reported in this study align with Chen et al. (2024) primarily in polar and high-altitude regions, where both datasets show low fluxes: Chen et al. report mean fluxes of $\sim 7 \mu\text{g m}^{-2} \text{yr}^{-1}$ in the Arctic and $\sim 20 \mu\text{g m}^{-2} \text{yr}^{-1}$ in high-altitude Central Asia, while this study finds median fluxes of 9 $\mu\text{g m}^{-2} \text{yr}^{-1}$ in Greenland/Iceland, 9 $\mu\text{g m}^{-2} \text{yr}^{-1}$ on the Tibetan Plateau, and 14 $\mu\text{g m}^{-2} \text{yr}^{-1}$ in N.W. North America. Both studies also identify elevated fluxes in tropical regions such as Latin America, though this study reports higher 1990–2020 median fluxes ($\sim 500 \mu\text{g m}^{-2} \text{yr}^{-1}$ in S.E. South America and $\sim 170 \mu\text{g m}^{-2} \text{yr}^{-1}$ in S. Central America) compared to Chen et al.’s predicted mean of $\sim 34 \mu\text{g m}^{-2} \text{yr}^{-1}$.

Discrepancies arise because Chen et al. values are predicted using a generalized additive model (GAM) with fewer cores (159 vs. 354), report mean fluxes instead of medians, interpolate missing years, and apply different regional definitions and temporal averaging. In East Asia, Chen et al. report a high mean flux ($\sim 414 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n=5$), while this study finds low median fluxes in E. Asia (10.7 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=3$) and E.C. Asia (20.6 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=2$) but elevated fluxes in S.E. Asia (113 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=3$); note that sample sizes are small in both studies. Arctic Ocean (61 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=2$) and N. Europe ($\sim 127 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n=34$) median fluxes exceed Chen et al.’s mean values ($\sim 7 \mu\text{g m}^{-2} \text{yr}^{-1}$ and $\sim 39 \mu\text{g m}^{-2} \text{yr}^{-1}$, respectively). In North America, this study finds lower median fluxes in C. North America (19 $\mu\text{g m}^{-2} \text{yr}^{-1}$, $n=23$) than E. North America (36 $\mu\text{g m}^{-2} \text{yr}^{-1}$) and W. North America (39 $\mu\text{g m}^{-2} \text{yr}^{-1}$), which contrasts with Chen et al.’s sub-regional pattern, although enrichment factors in C. North America were significantly greater than elsewhere.

Global trends in Hg concentrations and fluxes, for impacted sites could not be evaluated due to the lack of published literature reporting these metrics across different world regions.

Evaluation of Confidence. Confidence in the reported Hg metrics is highest in regions where findings align closely with published literature and where sample numbers are substantial. For example, elevated Hg fluxes in tropical regions such as Latin America and lower fluxes in polar and high-altitude areas show strong agreement with existing global datasets, supporting the reliability of these results. Similarly, regions with larger sample sizes, including N. Europe and parts of North America, provide more robust median values.

Information and Capacity Needs. As discussed under the Information and Capacity Needs section of Questions 1 and 2, greater spatial representation from regions outside of North America is needed to assess spatial trends in Hg concentrations and fluxes across world regions. Also as discussed in Question 2, to enable cross-region comparisons of Hg fluxes, there is a need for spatially resolved and region-specific estimates of Pb-210 atmospheric fallout for the calculation of lake-specific sediment focusing factors. Developing and making publicly available global map layers of estimated Pb-210 fluxes would enable more consistent application of focusing factor corrections across study sites, which in turn would facilitate the comparison of flux values across lakes/sites.

2.7.7 How do local temporal trends in THg and MMHg (and other species/fractions if applicable) found in {media} differ by world regions?

Information used. Flux-calculated enrichment factors (EFs) were compiled for 26 Intergovernmental Panel on Climate Change (IPCC) regions at 8 time periods relative to pre-industrial (1760 – 1850) Hg fluxes, as described in the methodology section of Question 2. Regional core counts of EF data are summarized in Figure 6 with the greatest data coverage in North America and South America. Regions with only one core are excluded from the regional trend figure and discussion including unimpacted cores from: Caribbean, E. Asia, E.C. Asia, Equatorial-Pacific-Ocean, Mediterranean, N.W. South America, S. Australia, South-American-Monsoon, W. Antarctica, West/ Central Europe; and impacted cores from: E. Siberia, N. Central America, N.E. North America, N. Europe, West/Central Europe.

EF data was available for 248 cores (203 unimpacted, 45 impacted). N.W. North America had the greatest number of cores (63 unimpacted, 15 impacted) while many regions had fewer than 5 cores (i.e., N. Europe, S. Eastern Africa

S.W. South America, Greenland/Iceland, Tibetan Plateau, S. Central America, S. South America, Arctic Ocean, New Zealand, Caribbean, E. Asia, N.W. South America, South-American-Monsoon.

Comparisons of intra-region impacted vs. unimpacted EFs was possible for 5 regions: N.W. North America, W. North America, E. North America, C. North America, S. Eastern Africa. See Question 2 for further details on data sources.

Methodology. To assess spatio-temporal differences in Hg levels between IPCC regions, enrichment factors (EFs) were calculated following the methodology described in Question 2. The focus on EFs stems from the lack of reliable sediment focusing corrections for Hg fluxes in regions outside North America, which restricts direct comparison across waterbodies. EFs normalize Hg levels relative to pre-industrial baselines, accounting for spatial variability not addressed by raw concentration or flux data. While some Central American, South American, and European regions may exhibit elevated Hg fluxes before the industrial period, this was assessed by calculating the flux ratio of pre-anthropogenic (pre-1500) to pre-industrial (1760–1850) fluxes, with the pre-industrial flux chosen as the baseline to maximize data retention. Sample size (n), median (M), and mean (μ) EF values were calculated for different time periods spanning the periods 0–1500 (pre-anthropogenic) and 1900–2020 (Figure 6). Statistical significance of differences between time periods and IPCC regions was assessed using Wilcoxon rank-sum tests. To evaluate regional differences within a given time bin or period, Kruskal-Wallis tests were used to detect overall variation among IPCC regions, followed by Dunn's test with Benjamini-Hochberg correction to identify specific pairwise differences in Hg flux.

Answer/ Response

Unimpacted Sites: In unimpacted sites, early EFs (calculated by comparing pre-anthropogenic (0–1500):pre-industrial (1760–1850) Hg fluxes) were below 1 in most regions, indicating that Hg fluxes were lower in earlier time periods, as expected. In the 1850–1900 period, median EFs remain below 1.2 in 8 of the 15 IPCC regions, with lowest values in S.W. South America (~0.9, range 0.6 – 1.5, n= 3), S. Eastern Africa (~1.00, 0.8 – 1.2, n= 2), and S. South America (~1.05, range 1.0 – 1.1, n= 3). The regions with highest EFs in the 1850–1900 period were N. Europe (~3.3, range 1.8 – 3.5, n= 3), S. Central America (~1.8, range 1.3 – 5.3, n= 3), and the Tibetan-Plateau (median 1.6, range 0.8 – 3.5, n=4). However, there are no significant differences in EFs across regions over these early (0–1500 and 1850–1900) periods (padj > 0.05).

Over the 20th century, EFs gradually increased over time in most world regions with distinct peaks observed only in the Arctic Ocean and N. Europe IPCC in the 1970s (~8.5, range 4.9 – 23.7, n= 3), and 1950s (~20.5, range 8.6 – 32.4, n= 2), respectively (Figure 6); however, the values in the peak period were statistically similar to the preceding and following decades in both regions ($p < 0.05$). Over the 20th century, EFs continue to be lowest in S.W. South America (period 1900–1990 median EF range: 0.5 – 0.9, n= 2-4), S. South America (range 1.1 – 1.5, n= 1-3), followed by Greenland/Iceland (range 1.1 – 2.2, n= 1-4). EFs in New-Zealand increased at a greater rate than the South America regions, with median Hg fluxes ranging from ~1.6 in 1900–1950 (range 1.3–2.1, n= 3) to ~2.9 in 1990–2000 (range 2.5 – 6.9, n= 3). Over the 20th century, median EFs continued to be highest in N. Europe (median range 4.2 – 20.5, n= 2-4), the Tibetan Plateau (range 4.0 – 12.0, n= 4), and S. Central America (range 6.4 – 17.6, n= 3). The Arctic Ocean IPCC region also experienced fast increase in EFs during the 20th century, growing at a rate of ~1.16-fold per decade from a median of ~2.8 in 1900–1950 (range 2.7 – 5.5, n= 3), to a peak of ~8.5 in 1970–1980, and then increasing once again to ~9.7 in 1990–2000 (range 5.8 – 13.5, n= 2). Intermediate EFs were observed in all North American IPCC regions as well as in S. Eastern Africa as follows: W. North-America (range 2.1–15.1, n= 2-37) > E. North America (range 2.1–5.2, n= 14-32) $\approx$ C. North America (range 2.3–5.7, n= 10-12) > N.E. North America (range 1.4–3.2, n= 6-17) $\approx$ N.W. North America (range 1.4–2.6, n= 15-63) $\approx$ S. Eastern Africa (range 1.2–2.4, n= 2). In the 21st century, two regions experienced notable but statistically insignificant increases in EFs between 2010 and 2020: the Tibetan Plateau where EFs were ~10.5, range 1.5 – 21.0, n= 4 in 2000-2010 and ~20.4, range 17.1 – 23.8, n= 2, $p = 0.7$ in 2010-2020, and W. North America where EFs were ~4.5, range 0.5 – 35.4, n= 32 in 2000-2010 and ~15.1, range 12.7 – 17.6, n= 2, $p = 0.09$ in 2010-2020.

Impacted Sites: In impacted sites, EFs in the pre-anthropogenic (0–1500) period were below 1, demonstrating that fluxes were lower in this period relative to the pre-industrial period (1760-1850), and statistically similar across all regions. In increasing order of median EF: ~0.28 in N.W. South America > ~0.34 in N.W. North America and > ~0.81 in E. North America. By the 1850–1900 period, median EFs were highest in C. North America (~15.4, range 5.5 – 25.0, n= 2), the Caribbean (~2.6, range 1.9 – 3.4, n= 2), and W. North America (~1.5 range 1.4 – 3.0, n= 5), and lowest in S. Eastern Africa (~0.9, range 0.6 – 1.2, n= 2), and the South-American-Monsoon (~1.2, range 0.9 – 1.4, n= 2). Post-1900 EFs with values > 10 were observed at all time periods in the Caribbean (1900–2010 median range 12.6 – 63.0, n= 2) and C. North America (1900–2020 median range 40.0 – 94.0, n= 2), and

in W. North America over the 1970–2000 (median range 9.6 – 10.9, $n=7-8$). EFs remained low over the 20th century in S. Eastern Africa (median range 1.8 – 4.4, $n=2$), the South-American Monsoon (range 1.8 – 4.4, $n=2$), N.W. North America (range 2.8 – 4.3, $n=9-15$), N.W. South America (median range 1.0 – 2.0, $n=2$) and E. North America (range 1.9 – 4.8, $n=2-4$).

Consistency with Existing Literature. The trends in unimpacted cores observed in this study generally align with other global compilations of lake sediment cores, including continental Hg flux trends reported by Chen et al. (2024) and the hemispheric differences reported in Li et al. (2020). In Europe, we observe mid-20th century peaks in Hg enrichment, which align with findings of Chen et al. (2024) —who note that Europe is the only continental region where Hg fluxes have declined since the mid-20th century— and Streets et al. (2019), who reported peak European emissions during 1960–1980. For New Zealand/Oceania, our results agree with Chen et al. (2024), who reported slow, steady Hg flux increases since the 1750s. However, in New Zealand/Oceania cores, neither us nor Chen et al. (2024) observe a pronounced peak during the Gold Rush (1860–1920) as is observed for Hg emissions by Streets et al. (2019). For South America, our unimpacted cores in South America and Southwest South America show relatively low and stable EFs over the 20th century, which is in agreement with Chen et al. (2024), who reported mostly stable fluxes since 1850 with a small peak in ~1980. In Africa, our data show increased Hg enrichment over time which is consistent with Chen et al. (2024), who found steady Hg flux increases from 1850–2010, and the Streets et al. (2019) emissions inventory. Our results differ from Chen et al. (2024) for North America, who reported peak Hg fluxes in ~1970 ($n=91$ cores) while we observed a constant rise in Hg enrichment over the 20th century in the different IPCC regions of North America ($n=166$ cores).

Compared to the flux-based EFs reported by Li et al. (2020), our analysis finds similar or slightly higher EFs in unimpacted sites of the Northern Hemisphere (range: 2.7–23.2; $n=187$) compared to their reported ~5-fold enrichment ($n=4$). In the Southern Hemisphere, unimpacted cores in Oceania, South America, and Africa show lower modern EFs (ranging from ~1.6 to 2.6; $n=17$) relative to Li et al.’s estimate of ~3.5-fold enrichment ($n=8$) compared to pre-industrial levels; however, both this report and Li et al. are consistent in finding that EFs are higher in the Northern Hemisphere relative to the Southern Hemisphere.

Evaluation of Confidence. Confidence in regional and continental Hg flux trends from unimpacted sediment cores varies with data density, temporal coverage, and geographic representation. Confidence is highest for Europe, where observed trends align with other sediment core compilations and independent emissions inventories (e.g., Chen et al. 2024; Streets et al. 2019) and in North America, where datasets include well-dated cores spanning pre-industrial and modern time periods. Confidence is moderate for Asia, where core coverage is more limited but trends are consistent across IPCC regions and match reported emissions (Streets et al. 2019). Confidence is lower for Oceania, Africa, South America, and Central America due to fewer available cores, incomplete temporal records, and reliance on individual sites. EF estimates are more robust where multiple cores support consistent trends, but uncertainty increases in regions with high inter-core variability or where only one or two cores are available (e.g., Central North America, parts of Africa).

Information and Capacity Needs. For details on the Information and Capacity Needs in evaluating regional spatiotemporal Hg trends, see the Information and Capacity Needs section of Questions 2 and 5.

2.7.8 How do local THg and MMHg levels (and other species/fractions if applicable) found in sediment cores differ by watershed land cover or land use?

Information used. To assess differences in Hg levels across lakes with varying watershed land cover, sediment derived enrichment factors (EF) were compared across lakes with varying catchment-to-lake area ratios (CA:LA) including: small, closed-basin systems (CA:LA <1.5), small lakes with predominantly atmospheric Hg inputs (CA:LA = 1.5–5), lakes with a mix of atmospheric and catchment sources (CA:LA = 5–20), primarily catchment-dominated systems (CA:LA = 20–50), and large-lake systems with extensive catchments where Hg delivery is largely influenced by terrestrial processes (CA:LA > 50). Catchment-to-lake area ratios have been found to be strong predictors for Hg fluxes in the Chen et al. 2024 global dataset (159 cores) and in the Swain et al. 1992 dataset of 7 lake sediment cores from northeastern Minnesota, but not in the Muir et al. 2009 dataset of 50 North American lakes or the Engstrom et al. 2007 dataset of 55 Minnesota lakes. Available sediment core data was distributed across CA:LA categories as follows: 46 in the <1.5 group, 227 in the 1.5–5 group, 506 in the 5–20 group, 215 in the 20–50 group, and 207 in the >50 group. The 2000–2010 period was the most recent time interval evaluated, as no cores in the <1.5 CA:LA category included data for 2010–2020.

To evaluate Hg levels in lake sediment cores with varying watershed land use, coring sites were classified following the “degree of development” framework adapted from Drevnick et al. (2016). This classification scheme excludes cores from impacted sites and includes four categories: 0 (pristine; no evident development), 1 (minimally disturbed; e.g., presence of roads or campgrounds), 2 (moderately developed; e.g., limited residential or agricultural activity), and 3 (heavily developed; e.g., extensive agriculture or dense residential use). Cores influenced by known point sources of Hg (category 4) were excluded due to variability in source type, timing, and intensity.

Sediment Hg concentration ($\mu\text{g g}^{-1}$) and flux ($\mu\text{g m}^{-2} \text{yr}^{-1}$) data, respectively, was available from 319 and 370 lake sediment cores from 82 and 92 studies including 222 and 271 cores from North America, 41 and 42 from Europe, 25 and 4 from Oceania, 12 and 16 from South America, 8 and 11 from Central America, 8 and 21 from Asia, 2 and 4 from Africa, and 1 from Antarctica. For Hg concentration and flux, respectively, 140 and 164 cores were classified as category 0, 58 and 94 as category 1, 54 and 53 as category 2, and 67 and 59 as category 3. EF data was available from 203 sediment cores from 62 studies, with a similar geographic and development category distribution to the concentration and flux dataset.

Methodology. THg concentrations and fluxes were log-transformed prior to statistical analysis. EFs were calculated from THg flux values as described above for Question 2. Cores were grouped according to development category and/or CA:LA classification and sample size (n), median (M), and mean (μ) log-transformed values were calculated for different time periods spanning the period 1500–2020 (Figures 8–9 and Supplemental Figure 6). Lake categories were delineated based on visual interpretation of land use and shoreline characteristics using Google Earth imagery from May to July 2024. The Wilcoxon rank-sum test was applied to assess the statistical significance of differences in THg concentrations, fluxes, and EFs between temporal groups. Kruskal-Wallis tests were conducted to evaluate differences across multiple CA:LA categories.

Answer/ Response. Examination of Hg enrichment in watersheds with no known Hg source and with varying catchment:lake area ratios showed that median EFs did not differ significantly among CA:LA categories over any time period examined (Kruskal-Wallis tests, $p > 0.05$). Although the lowest CA:LA group (<1.5) often showed higher median EFs (i.e., 2.1 in 1900–1950 (range 1.3 – 3.9, $n = 7$) and 3.3 in 2000–2010 (range 1.5 – 8.3, $n = 5$), values overlapped with those from larger catchments, such as the >50 group (e.g., 1.2 in 1900–1950 (range 0.5 – 8.5, $n = 38$) and 3.0 in 2000–2010 (range 0.4 – 26.5, $n = 25$)). Overall, CA:LA ratio does not appear to be a strong predictor of EF magnitude at the global scale; however, the analysis was limited by the size of the dataset.

Examining Hg levels across different watershed development categories showed that during the pre-industrial (1760 – 1850) period, Hg concentrations and fluxes are statistically similar ($p > 0.05$) across all development categories. In pristine watersheds, THg concentrations and fluxes rose modestly from pre-industrial values of $\sim 0.06 \mu\text{g g}^{-1}$ and $6.1 \mu\text{g m}^{-2} \text{yr}^{-1}$, respectively ($n = 85$ and 102)), with peak Hg concentrations over the 1980–2000 period ($\sim 0.103 \mu\text{g g}^{-1}$, $0.02 - 0.58 \mu\text{g g}^{-1}$, $n = 113$) and peak fluxes in the most recent 2010–2020 period ($\sim 32.6 \mu\text{g m}^{-2} \text{yr}^{-1}$, $2.1 - 250 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 22$). Similarly, in minimally developed watersheds, concentrations and fluxes peaked in 1990–2000 at $\sim 0.15 \mu\text{g g}^{-1}$ and $28.7 \mu\text{g m}^{-2} \text{yr}^{-1}$, respectively ($n = 50 - 86$) but then significantly decreased to $\sim 0.09 \mu\text{g g}^{-1}$ and $15.2 \mu\text{g m}^{-2} \text{yr}^{-1}$ ($n = 12$ and 20) over the most recent decade measured. In moderately and heavily developed watersheds, concentrations and fluxes exhibited a steady rise after industrialization, peaking in the most recent decade measured (2010–2020: ~ 0.26 and $0.13 \mu\text{g g}^{-1}$, $n = 8$ and 19 , and ~ 135.9 and $91.2 \mu\text{g m}^{-2} \text{yr}^{-1}$, $n = 8$ and 18). The differing trends over the most recent decade may reflect recent

changes in Hg inputs across the different landscape development categories examined; for example, the most disturbed watersheds are showing later peaks in Hg levels due to on-going development. However, the differences may also reflect decreased sample sizes for this time period as has extensively been discussed in earlier sections.

Enrichment factors (EFs), calculated relative to pre-industrial Hg fluxes, more clearly illustrate the influence of watershed development on the magnitude of Hg enrichment over time. In pristine watersheds, median EFs remained low throughout the record, increasing gradually from ~1.5 (range 0.07 – 18.7, n= 99) in 1900–1950 to ~2.5 (range 0.7 – 19.6, n= 13) in 2010–2020, with median values <3 across all periods. Minimally developed sites showed a similar pattern over time but with higher enrichment observed after the 1950s (i.e., EFs rose from ~1.9 in 1900–1950 (range: 0.8 – 15.6, n= 52) to ~5.6 in 2010–2020 (range 1.1 – 23.8, n= 10)). Moderately and heavily developed sites exhibited similar enrichment to each other across all time periods ($p > 0.05$) but showed higher enrichment than the lakes in the minimally developed category (i.e., median EFs increased from ~2.0–2.4 in 1900–1950 (range 1.0 – 17.3, n= 19–32) to ~7.2–8.9 in 2000–2010 (range 0.7–36.1, n= 10–19). EFs were also highly elevated in both of these disturbance categories in the most recent decade measured; however the sample size was very small (n= 2). Taken together, the results suggest that even minimal catchment development can produce a small but measurable enhancement of sediment derived Hg depositional fluxes and that minimal and major catchment disturbance result in a more pronounced and sustained Hg enrichment.

The observed trends between higher watershed development and Hg enrichment are largely driven by North American lake sediment core data where EF magnitude also varies across IPCC region (Table 1). Heavily developed watersheds in Western and Eastern North America show particularly high median EFs (>10), while lower EFs (~2–4) are observed in Northwest and Northeast North America with less variation by development level. In contrast, data from other continents display more variable patterns, often lacking a clear relationship between development intensity and enrichment (Table 1). For example, in Asia and Central America, sites within minimally and moderately developed watersheds have high Hg enrichment, while those within watersheds with heavy development are not consistently elevated. These discrepancies may reflect smaller sample sizes or regional differences in land use, industrial history, or Hg sources.

Table 1. Summary of Hg flux Enrichment Factors (EFs) across watersheds with varying levels of development in different IPCC regions. Enrichment Factors were calculated as the ratio of Hg fluxes from 1990–2020 to pre-industrial Hg fluxes in unimpacted sites with no known Hg source. Degree of development was classified as no development; minimal development (e.g., road, campground); minor development (e.g., small city or agricultural field); and major development (e.g., large city or agricultural hub).

Region	Degree of Development	# Unique Cores	Median EF	Mean EF	Interquartile range	Standard error
North America	None	78	2.54	3.27	1.60	0.04
	Minimal	40	3.93	4.73	3.15	0.12
	Moderate	28	5.04	6.47	5.79	0.18
	Heavy	13	9.61	12.31	10.57	1.03
Western North America	None	20	3.27	3.99	2.62	0.16
	Minimal	4	4.23	7.07	1.84	2.63
	Moderate	9	8.82	9.21	8.26	0.62
	Heavy	4	12.74	17.03	4.50	3.65
	None	38	2.25	2.96	1.02	0.09

<i>Northwestern North America</i>	Minimal	8	3.35	3.18	1.77	0.17
	Moderate	5	1.09	1.07	0.23	0.04
	Heavy	8	2.31	3.85	1.92	0.40
<i>Eastern North America</i>	None	4	2.95	2.94	0.68	0.13
	Minimal	20	4.64	5.27	3.64	0.21
	Moderate	7	6.32	6.89	2.96	0.64
	Heavy	1	33.40	33.40	0	NA
<i>Northeastern North America</i>	None	13	2.72	3.44	1.63	0.17
	Minimal	2	4.43	4.02	1.56	0.80
	Moderate	1	2.49	2.49	0	NA
Europe	None	5	3.20	5.52	6.31	0.94
	Minimal	2	7.36	8.25	5.45	2.61
	Moderate	1	0.66	0.66	0	NA
	Heavy	1	4.31	4.31	0	NA
Asia	Minimal	4	16.77	13.79	17.11	2.43
	Moderate	1	4.06	4.065	0	NA
	Heavy	1	2.17	2.17	0	NA
Central America	Moderate	2	19.41	20.27	1.62	1.33
	Heavy	1	7.40	7.40	0	NA
South America	None	5	1.69	1.59	0.76	0.10
	Minimal	5	1.55	1.48	0.64	0.11
Africa	None	1	2.62	2.62	0	NA
	Minimal	1	2.25	2.25	0	NA

Consistency of Answer with Existing Literature. Our findings showing a lack of relation between CA:LA and Hg enrichment (measured as Hg flux based EFs) are generally consistent with Drevnick et al. 2012, who found variable regional relationships between CA:LA and Hg flux ratios, with significant relations in some subregions but not universally across their study area (the Laurentian Great Lakes Region). Chen et al. 2024, however, recently reported a strong concave relationship between CA:LA and Hg fluxes for 159 globally distributed lake sediment cores with fluxes peaking at intermediate CA:LA values and declining at larger ratios (> 75). We did not observe any relationship between CA:LA with Hg concentration, flux, or EF. Differences among study findings may reflect variations in dataset composition and geographic coverage.

We did observe higher Hg enrichment in lakes with higher watershed development which is in accordance with results for Western North America, where Drevnick et al. 2016 found that lakes in heavily disturbed watersheds

(category 3) had Hg accumulation rates that were quite high, resulting in EFs of ~5 by the 1960s, and significantly greater than those with no or minimal disturbance.

Evaluation of Confidence. CA:LA values were not available for all lake sediment cores resulting in small sample sizes for each CA:LA category and making assessment at a global-scale challenging. To assess the impact of watershed land-use of Hg levels, watershed development classifications were assigned using recent satellite imagery, providing a globally consistent but qualitative assessment. This method captures broad spatial patterns but lacks the precision and reproducibility of metrics based on quantified land use data (e.g., percent watershed development as was used in Drevnick et al. 2016). Results generally align with published literature and show consistent trends across time periods, continents, and core types, particularly in North America where data coverage is highest. Confidence is lower in regions with limited representation, including Africa, South America, and parts of Asia.

Information and Capacity Needs. To improve future global assessments, expanded lake sediment Hg measurements that include ancillary data, such as CA:LA, in underrepresented regions, particularly Africa, South America, and Asia is needed. Harmonization and updating of land use, land cover, and watershed disturbance data over time would also support more precise evaluations of development impacts on Hg enrichment.

2.7.9 How do THg and MMHg levels (and other species/fractions if applicable) co-vary spatially in sediment cores?

This question cannot be addressed as there was no lake sediment core datasets which reported both THg and MMHg levels. But this question may be able to be addressed using monitoring datasets which include THg and MMHg concentrations in surficial lake sediment, however these types of data are not the focus of this particular report.

2.7.10 How do spatial trends of Hg levels observed in sediment cores compare to or co-vary with variations and trends of related pollutants or environmental variables over the available record? (TOC, SOM, DOC, TSS)?

Information used. To assess spatial trends in how lake sediment THg levels co-vary with other environmental variables, relationships between THg concentrations and fluxes, %LOI and Al concentration were assessed (as described for Question 3) across different IPCC regions as delineated in Iturbide et al. (2020) and for Questions 5-6. For both unimpacted and impacted sites, sediment cores with %LOI or Al data are available for 7-8 of the IPCC regions with most data from parts of North America, Northern Europe, Central and South America and the Caribbean (Figures 10-11). To supplement lacking %LOI data, percent total organic carbon (%TOC) was used as an additional metric for organic matter content and filled in data gaps in for the IPCC regions of North, Central and South America (Figure 10).

Methodology. The regions that sediment cores were collected from were classified as unimpacted or impacted as described under the methodology for Question 1. Spearman correlations were used to assess the relationship between THg concentration or flux and organic matter concentration (as %LOI or %TOC) or aluminum concentration (Al, $\mu\text{g/g}$), for time periods spanning from 1500–2010 as described under the methodology for Question 3. For each region and time period, Spearman's ρ , 95% confidence intervals, and associated p-values were calculated. Correlation strength was classified as very weak ($|\rho| < 0.2$), weak ($0.2-0.4$), moderate ($0.4-0.6$), or strong ($|\rho| \geq 0.6$), and further grouped by direction (positive or negative) and significance ($p < 0.05$).

Answer/ Response. Globally, at unimpacted sites, sediment Hg concentrations generally exhibit positive correlations with %LOI, as described in Question 3. Regionally, these trends show greater variability but still generally support the well-established co-transport of Hg and organic matter to lake sediments (Figure 10). In unimpacted sites of N. Europe, N.W. North America and S. Central America, where %LOI data is available, relationships between THg and %LOI are generally positive, with significant correlations in approximately a third of the correlations carried out ($p < 0.05$ for 9 of 24 correlations across 5-11 time periods and 3 IPCC regions) (Figure 10). Similar to %LOI, relationships between sediment THg concentration and %TOC, available for 5 IPCC regions in North America, are generally positive and frequently significant ($p < 0.05$ for 25 of 49 correlations across 11 time periods and 5 IPCC regions).

As with the global dataset, at impacted sites, the strength and direction of the relationships between Hg concentration and organic carbon are more heterogeneous over time and across regions than in unimpacted sites, likely reflecting the role of nearby anthropogenic sources of varying size and nature (i.e., atmospheric emissions versus releases to water) in controlling Hg levels in lake sediments. In the Caribbean, N.W. North America, N.W. South America, South-American-Monsoon and W. North America, relationships between THg concentration and %LOI are generally positive with just less than half of the correlations significant ($p < 0.05$ for 23 of 54 correlations across 5-11 time periods and 7 IPCC regions).

In contrast, at impacted sites of N. and S. Central America, relationships between sediment THg and %LOI are generally significantly negative ($p < 0.05$ for 9 of 13 correlations across 4-11 time periods and 2 IPCC regions), likely reflecting the importance of oil refining and waste incineration or the shared coastal Mexico location among coring sites driving this pattern (14 of 17 cores); however, since Caribbean cores (also coastal but influenced by mining and metallurgical activities) exhibit positive correlations in 5 of 8 periods, this suggests that source type, rather than coastal setting, drives the observed differences. Relationships between sediment THg concentration and %TOC, available for 2 IPCC regions of North America and 2 for South America, are generally positive in E. North America, S.E. South America and South-American-Monsoon, although most of these correlations are not significant ($p < 0.05$ for 5 of 18 positive correlations across 4-11 time periods and 3 IPCC regions). In contrast, in impacted sites of N.W. North America, relationships between sediment THg and %TOC are predominantly significantly negative ($p < 0.05$ for 5 of 11 time periods). This is in contrast to %LOI for cores from this same IPCC region (N.W. North America), which show strong, significantly positive relationships with THg over most

of the record. These contrasting results for the same IPCC region highlight the role of local factors, such as Hg sources and local landscape characteristics, which can over-ride the influence of organic matter in driving Hg concentration in sediments.

Similar to the global dataset (see Question 3), in the 3-6 IPCC regions where data is available, Hg fluxes in both unimpacted and impacted sites often show negative correlations with %LOI, although the correlations are not as frequently significant as for the global dataset ($p < 0.05$ for 24 of 68 correlations across 4-11 time periods and 3-6 IPCC regions) (Figure 10). Relationships between THg flux and %TOC are also generally negative but vary in both strength and direction across IPCC regions and time periods (Figure 10). For example, positive correlations are observed in earlier parts of the record (before 1960) for unimpacted sites of E. North America and N.W. North America suggesting strong links between Hg and OC transport in these areas.

Similar to the global dataset, in both unimpacted and impacted sites, THg and Al concentrations were generally weakly and sometimes negatively correlated across the 6-8 IPCC regions where data is available (Figure 11). Exceptions include positive, significant THg:Al relationships over much of the temporal record in unimpacted sites of E. North America and N. Europe ($p < 0.05$ for 11 of 20 correlations) and impacted sites of South American Monsoon ($p < 0.05$ for 6 of 7 correlations). In the global dataset, we observed significant, positive relationships between THg flux and Al concentrations in unimpacted and, to a lesser degree, impacted, regions (see Question 3 and Figure 6). Regionally, correlations between Hg flux and Al concentration are weaker and more variable with strongest positive relationships observed in unimpacted sites of N.W. North America, S. Central America, and W. North America ($p < 0.05$ for 13 of 25 correlations) and impacted sites of N.W. North America ($p < 0.05$ for 6 of 11 time periods) suggesting that mineral lithogenic inputs, and by extension sedimentation rate, is the dominant driver of sediment Hg accumulation rates in these regions (Figure 11).

Consistency of Answer with Existing Literature. To our knowledge, this is the first-time relationships between sediment THg concentrations and fluxes, %LOI, %TOC and Al concentration have been assessed across global IPCC regions. See Question 3 for a discussion of how relationships between THg, organic carbon and Al and observed in this dataset compare to findings from individual studies.

Evaluation of Confidence. Confidence is limited by uneven data availability across geographic regions. Core distribution is heavily skewed toward North America, Central America, and Northern Europe with many regions, such as Africa, Asia, and Oceania, absent. Additionally, variability in the temporal resolution of datasets, small sample sizes in some regions (e.g., $n = 2-4$), and limited availability of co-located THg, %LOI, %TOC, and Al concentration data constrain the robustness of correlation analyses at the regional level. Confidence is higher in regions with more extensive records (e.g., N.W. North America, N. Europe, E. North America), where patterns are supported by larger sample sizes and greater temporal coverage.

Information and Capacity Needs. Expanding sediment core collections in regions with limited geographic coverage, especially Africa, Southeast Asia, Oceania, and parts of South America would improve the understanding how spatial trends in sediment Hg co-varies with other environmental variables globally. Second, although many studies measure co-located variables such as %LOI or Al concentration alongside Hg concentrations, these ancillary data are often excluded from published supplementary materials or archived datasets. Addressing this gap requires improvements in data reporting practices and repository standards to ensure that key contextual variables are made publicly accessible alongside Hg measurements.

2.7.11 Key Findings

Global lake sediment records show that Hg concentrations and fluxes were low and comparable between unimpacted and impacted sites during the pre-industrial period, but diverged thereafter, with unimpacted/remote sites showing steady increases to 2010–2020 while impacted sites collectively peaked around 1990–2000 and declined thereafter, particularly in the IPCC regions of Northern Europe, Eastern North America, Northeast North America, South-Central America, and Northwest-South America.

Flux based enrichment factors (EFs) were used to assess the magnitude of Hg enrichment over various time periods and showed that, as expected and consistent with prior literature, modern (2010-2020) EFs are significantly higher in impacted regions (~ 5.7) than in unimpacted regions (~ 3.6). In unimpacted regions, highest modern EFs are observed in Northern Europe, the Tibetan Plateau, and the Arctic while the lowest EFs are observed in South and Southwest-South America.

In unimpacted regions and during some early periods in impacted regions, Hg concentrations were positively correlated with organic matter indicative of the well-established co-transport of Hg and organic matter to lake sediments.

In both impacted and unimpacted regions, Hg fluxes were strongly correlated with aluminum, reflecting the dominant role of sedimentation rates, particularly the delivery of mineral rich catchment material, in controlling sediment derived Hg depositional fluxes.

While catchment:lake area ratios were not a strong predictor of enrichment across the global dataset, watershed development showed a clear influence, with pristine catchments maintaining low EFs (<3) over the entire temporal record and minimally to heavily developed catchments exhibiting progressively higher and more sustained Hg enrichment (median EFs ~ 7 – 9 by 2000–2010).

References:

- Biester *et al.* Solar irradiance and primary productivity controlled mercury accumulation in sediments of a remote lake in the Southern Hemisphere during the past 4000 years. *Limnol. Oceanogr.* 63, 540 – 549. (2017) <https://doi.org/10.1002/lno.10647>
- Biester, H., *et al.* Modeling the past atmospheric deposition of mercury using natural archives. *Environ. Sci. Technol.* 41, 14, 4851 – 4860 (2007). <https://doi.org/10.1021/es0704232>
- Campbell, L., Hecky, R., Muggide, R., *et al.* Variation and distribution of total mercury in water, sediment and soil from northern Lake Victoria, East Africa. *Biogeochemistry* 65, 195–211 (2003). <https://doi.org/10.1023/A:1026058417584>
- Chen, Q. *et al.* Mercury records from natural archives reveal ecosystem responses to changing atmospheric deposition. *J. Nat. Sci. Rev.* 11, 12, nwae417 (2024). <https://doi.org/10.1093/nsr/nwae417>
- Cooke, C. A. *et al.* Caribbean Lead and Mercury Pollution Archived in a Crater Lake. *Environ. Sci. Technol.* 56, 3, 1736 – 1742 (2022). <https://doi.org/10.1021/acs.est.1c06791>
- Cooke, C. A. *et al.* Spatial and temporal patterns in trace element deposition to lakes in the Athabasca oil sands region (Alberta, Canada). *Environ. Res. Lett.* 12, 12, 124001 (2017). <https://doi.org/10.1088/1748-9326/aa9505>
- Dastoor, A., *et al.* The Multi-Compartment Hg Modeling and Analysis Project (MCHgMAP): mercury modeling to support international environmental policy. *Geosci. Model Dev.* 18, 2747–2860. (2025) <https://doi.org/10.5194/gmd-18-2747-2025>
- Drevnick, P. E. *et al.* Spatiotemporal patterns of mercury accumulation in lake sediments of western North America. *Sci. Tot. Environ.* 15, 568, 1157 – 1170 (2016). <https://doi.org/10.1016/j.scitotenv.2016.03.167>
- Engstrom, D.R. Atmospheric Hg emissions from preindustrial gold and silver extraction in the Americas: A reevaluation from lake-sediment archives. *Environ. Sci. Technol.* 48, 12, 6533–6543 (2014) <https://doi.org/10.1021/es405558e>
- Ethier, A.L.M., *et al.* Mercury empirical relationships in sediments from three Ontario lakes. *Sci. Tot. Environ.* 408, 2087 – 2095 (2010). <https://doi.org/10.1016/j.scitotenv.2009.12.037>
- Iturbide, M. *et al.* An update of IPCC climate reference regions for subcontinental analysis of 604 climate model data: definition and aggregated datasets. *Earth Syst. Sci. Data* 12, 2959–2970 (2020).
- Jiang, S., Liu, X., Chen, Q. Distribution of total mercury and methylmercury in lake sediments in Arctic Ny-Ålesund. *Chemosphere*. 83, 1108 – 1116 (2011). <https://doi.org/10.1016/j.chemosphere.2011.01.031>
- Li, C. *et al.* Unequal Anthropogenic Enrichment of Mercury in Earth's Northern and Southern Hemispheres. *ACS Earth Space Chem.* 4, 11, 2073–2081 (2020). <https://doi.org/10.1021/acsearthspacechem.0c00220>
- Muir, D.C.G. *et al.* Spatial Trends and Historical Deposition of Mercury in Eastern and Northern Canada Inferred from Lake Sediment Cores. *Environ. Sci. Technol.* 43, 13, 4802–4809 (2009). <https://doi.org/10.1021/es8035412>
- Roberts, S. L. *et al.* Quantification of spatial and temporal trends in atmospheric mercury deposition across Canada over the past 30 years. *Environ. Sci. Technol.* 55, 23, 15766 – 15775 (2021). <https://doi.org/10.1021/acs.est.1c04034>
- Roberts, S., Kirk, J.L., Wiklund, J.A., *et al.* Mercury and metal(loid) deposition to remote Nova Scotia lakes from both local and distant sources. *Sci. Tot. Environ.* 675, 192–202 (2019). <https://doi.org/10.1016/j.scitotenv.2019.04.167>
- Rutkowski, C. *et al.* Mercury in Sediment Core Samples From Deep Siberian Ice-Rich Permafrost. *Front. Earth Sci.* 9: 718153 (2021) <https://doi.org/10.3389/feart.2021.718153>
- Rydberg, J. *et al.* Assessment of the spatial distributions of total- and methyl-mercury and their relationship to sediment geochemistry from a whole-lake perspective. *J. Geophys. Res.* 117, G04005 (2021) <https://doi.org/10.1029/2012JG001992>
- Sanei, H. and Goodarzi, F. Relationship between organic matter and mercury in recent lake sediment: The physical–geochemical aspects. *Appl. Geochem.* 21, 11, 1900 – 1912 (2006). <https://doi.org/10.1016/j.apgeochem.2006.08.015>
- Sanei, H. *et al.* Classification of mercury–labile organic matter relationships in lake sediments. *Chem. Geol.* 373, 87–92 (2014). <https://doi.org/10.1016/j.chemgeo.2014.02.029>

Schneider, M. A., *et al.* Long-term mercury accumulation and climate reconstruction of an Australian alpine lake during the late Quaternary. *Glob. Planet. Change.* 240: 104539 (2024). <https://doi.org/10.1016/j.gloplacha.2024.104539>

Stern *et al.* Historical Interrelated Variations of Mercury and Aquatic Organic Matter in Lake Sediment Cores from a Subarctic Lake in Yukon, Canada: Further Evidence toward the Algal-Mercury Scavenging Hypothesis. *Environ. Sci. Technol.* 43, 20, 7684–7690 (2009). <https://doi.org/10.1021/es902186s>

Streets, D. G., *et al.* Five hundred years of anthropogenic mercury: spatial and temporal release profiles. *Environ. Res. Lett.* 14, 8, 084004 (2019). <https://doi.org/10.1088/1748-9326/ab281f>

Swain, E. B. *et al.* Increasing Rates of Atmospheric Mercury Deposition in Midcontinental North America. *Science.* 257 (5071) 748 – 787 (1992). <https://doi.org/10.1126/science.257.5071.784>

Wiklund, J.A., Kirk, J.L., Muir, D.C.G., *et al.* Anthropogenic mercury deposition in Flin Flon Manitoba and the Experimental Lakes Area Ontario (Canada): A multi-lake sediment core reconstruction. *Sci. Tot. Environ.* 586, 685-695 (2017). <https://doi.org/10.1016/j.scitotenv.2017.02.046>

Wu, F., Xu, L., Liao, H. *et al.* Relationship between mercury and organic carbon in sediment cores from Lakes Qinghai and Chenghai, China. *J. Soils Sediments.* 13, 1084 – 1092 (2013). <https://doi.org/10.1007/s11368-013-0694-2>

2.8 Fresh Waters (rivers, lakes, and coastal estuaries)

2.8.1 What data are available?

Seven (7) submissions for surface water measurements were received from four (4) continents – Australia, Europe, North America, and South America (Figure 1). They represent a mixture of inland and estuary sites. All submissions have data for total Hg but not for methyl Hg. Ancillary data (pH, dissolved oxygen, conductivity) is only available for the datasets from Belgium and Bulgaria.

Most of the submissions were from the Northern Hemisphere, primarily Canada (24%), Europe (34%) and USA (59%). Southern Hemisphere is represented by Ecuador (4%) and Australia (4%). Submissions do not provide good spatial coverage because majority of the data is from a single site (Australia, Belgium) or local monitoring (Canada, USA).

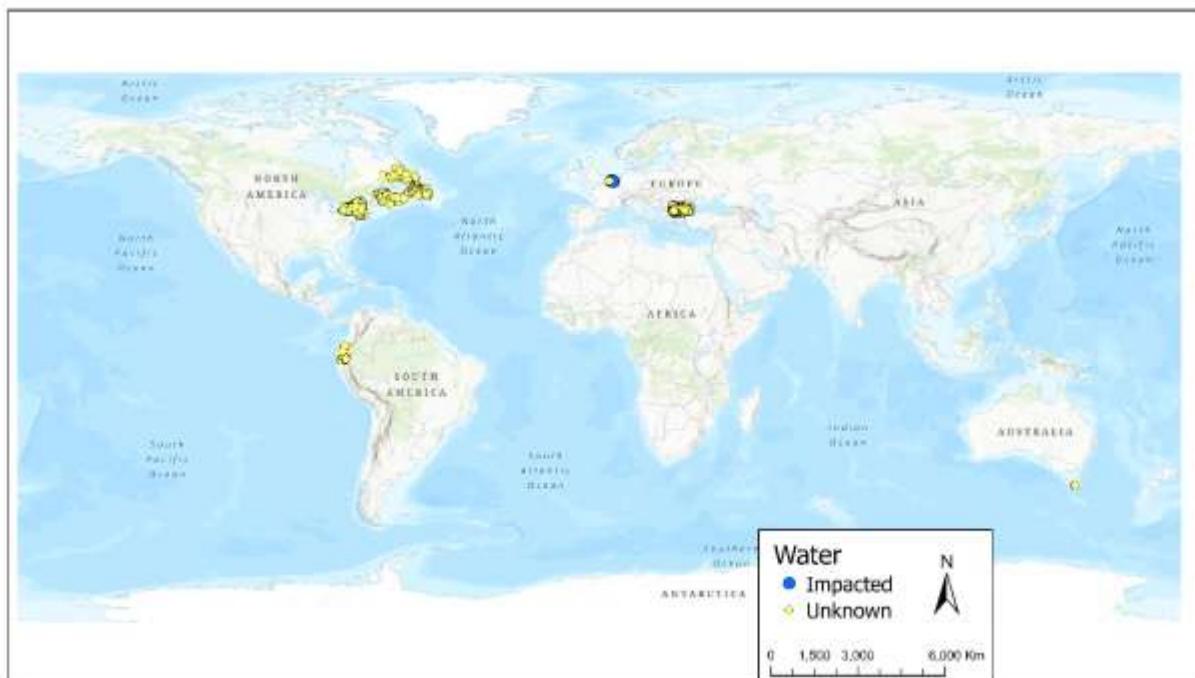


Figure 1. A map showing the distribution of the available freshwater data.

The only data received from South America (Ecuador) is from a national monitoring program of effluents of treatment plants. This data was included in the total count and is presented on the map (Figure 1) but cannot be used for assessment of freshwater because the data collection did not occur in the receiving waterbody.

Two additional inland monitoring programs for individual waterbodies were identified in the United States which are awaiting data authorization and submission.

Data usability. The quality and usability of the data vary considerably across countries. Australia's, Bulgaria's, and Ecuador's data have very high detection limit for natural water (0.05 µg/L; 0.01-0.02 µg/L, and 0.2 µg/L, respectively), resulting in reported values at or below detection limit. Belgium's data appears acceptable but limited in scope and duration, with unclear proximity to pollution sources. Canada's data shows inconsistency in units and detection limits; Hg species are unspecified, possibly using different analytical methods over the years. The USA provides usable aggregated data but lacks detection limits and has missing location details. While not explicitly stated, the long historical records from Canada and USA - dating back from 1970s – suggest that different analytical methods were likely used over the years.

2.8.2 What are current levels of THg and MMHg (and other species/fractions if applicable) observed in freshwater that are remote from or near anthropogenic sources?

Information used. There is currently insufficient data in the OESG dataset to address this question. Limitations of the available data due to analytical uncertainty and methodology are discussed above. With the exception of the Australian data, which was from a single contaminated site, the proximity of contaminated sites were not

provided by the data owners. There is also inadequate spatial coverage of THg and MMHg concentrations in freshwater to assess regional patterns.

Methodology. N/A

Answer/Response. N/A

Consistency of Answer with Existing Literature. There is limited information in the scientific literature to broadly assess the effect of anthropogenic point sources on Hg levels in natural waters. Research studies on Hg drivers in water are largely limited to North America and Europe. A review of the influence of contaminated sites on streams found catchments containing a point source of contamination can discharge >10 times the flux of Hg associated with undisturbed watersheds (Domagalski et al., 2016; Eckley et al., 2020). Because Hg concentrations in stream are usually correlated with discharge and TSS, due to erosion and mobilization of Hg-laden particles at high flow, river export of Hg can be highly variable (Eckley et al., 2020). Several studies of river water downstream of anthropogenic Hg releases have reported an increase in Hg concentrations relative to upstream sources – including the Sudbury River, U.S.A. downstream of the Nyanza Superfund site (Waldron et al., 2000), the Hackensack River, U.S.A. downstream of the Berry’s Creek Superfund site (Cardona-Marek et al., 2007), whereas particulate Hg but not dissolved Hg concentrations were elevated in the Penobscot River, U.S.A., a tidal river downstream of historic chemical plant (Turner et al., 2018). In the English-Wabigoon River in Canada, a site of a former-chlor alkali factory which has heavily contaminated downstream fish population with Hg, the water THg concentrations downstream of the point source are elevated, exceeding upstream levels by more than 10-fold (Rudd et al., 2021). The impacts of point sources of contamination on MeHg concentrations in the rivers are less clear. The percentage of Hg present as MeHg in natural waters are low (5-20%), and are often lower in contaminated sites where Hg concentrations are elevated, due to the MeHg concentrations being controlled by methylation potential, rather than Hg concentration (Eckley et al., 2020). It is also noted that concentrations of MeHg can be elevated where Hg concentrations are high, despite a low percentage of Hg present as MeHg. Elevated MeHg concentrations can be a concern at distances further downstream from point sources (Eckley et al., 2020), particularly if streams pass through wetlands in lower reaches (Gilmour et al., 2018; Waldron et al., 2000).

There is less research studying the impact of point sources of contamination on Hg concentrations in lakes. The current understanding remains limited to a few systems, each with its own contamination timeline and monitoring intensity; therefore, the effects are difficult to discern on a broad, global scale. Concentrations of Hg in the water column of pristine lakes are regulated by water quality parameters, particularly DOC concentration and phytoplankton density (Driscoll et al., 1994; Lescord et al., 2018; Ogorek et al., 2021), making general conclusion about the influence of point sources across different freshwater environments difficult to discern. MeHg concentrations in lakes are driven by internal cycling, development of hypolimnetic hypoxia linked to expression of sulfate-reducing conditions and methylation of Hg (Todorova et al. 2009; McCord et al. 2016; Millard et al. 2023), and watershed inputs. Some landscapes (eg. Florida Everglades, U.S., Kejimikujik National Park, Canada) are susceptible to efficient conversion of Hg to MeHg, resulting in high water column concentrations of MeHg. Point sources of Hg contamination to these “hotspot” landscapes can be particularly susceptible to methylation leading to elevated concentrations of MeHg in the water column (Gilmour et al., 2018).

Evaluation of Confidence. N/A

Information and Capacity Needs. To assess the impact of point sources of contamination on THg and MeHg in the water column requires more Hg data in freshwater ecosystems, along with information on the proximity of point sources to the water bodies being studied, flow regime for rivers, and stratification conditions for lakes. There is particularly a need for data in South America and Africa, where artisanal gold mining is thought to be a major source of Hg release in rivers.

2.8.3 How have levels of THg and MMHg (and other species/fractions if applicable) observed in freshwater changed over the available record?

Information Used (including applicable indigenous and traditional knowledge). The data record spans different time periods (Figure 2), beginning as early as 1970 for a dataset from Northeastern USA and 1978 for Northeastern Canada.

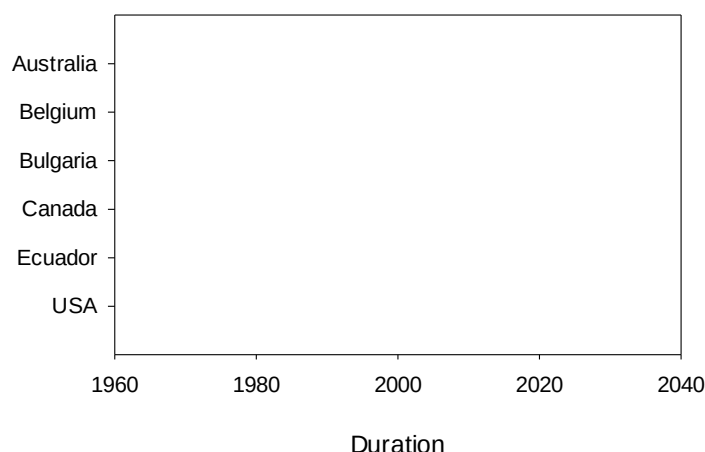


Figure 2. Years with available observations for freshwater.

Methodology. Methodology. USA dataset was examined for potential use in trend analysis. The MeHg data was determined unusable due to low temporal coverage with fewer than four years per site.

The THg data was clustered by location with ~10 m precision. In total, 786 sites/clusters were identified for 9,295 samples. Decreasing the precision to regroup more sites did not change the clustering significantly. Out of 786 sites, only 66 of them have at least 6 different years of data. Even after narrowing further to 22 sites with at least 10 years of data, which is considered as a minimum to detect long-term environmental trends, the time spread across the sampling period was not sufficient. Moreover, the available data period for many of the sites started in the 1970s and stopped before or around 1992. It limits any analysis to historical trends: it will not inform us about current or ongoing changes in freshwater data.

Due to the analytical limitations in 1970-1990 period, when instrumentation and detection limits were less sensitive, many of the recorded values for these sites fall below detection limits, with some years consisting entirely of non-detects (Figure 3).

Although 22 candidate sites were identified to have longer time series, their short temporal spread (ending in 1992) and the high proportion of values below detection limit make them problematic for performing trend analysis: any detected trend on this data would be very uncertain and not suitable for drawing broader conclusions.

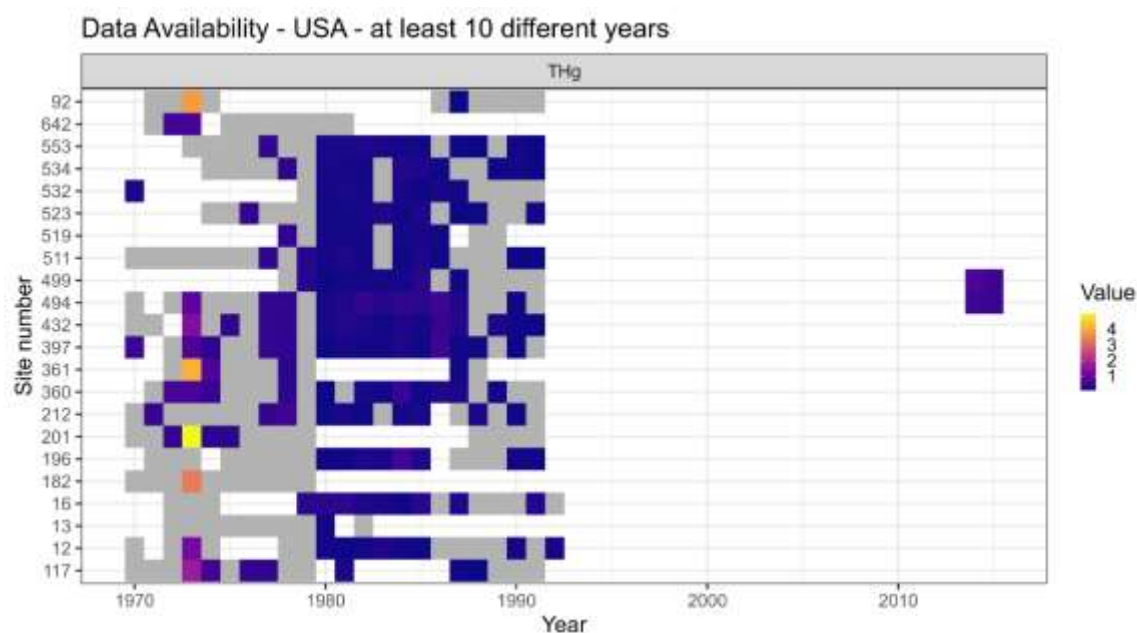


Figure 31: Time map of yearly average values of the 22 US sites that have at least 10 years of records. The color code refers to concentrations in ng/L. Years that have only values under the detection limits are coloured in grey.

Answer/Response. N/A

Consistency of Answer with Existing Literature. There is a limited number of multidecadal studies examining the long-term trends in THg and MMHg concentrations in lotic and lentic aquatic systems away from point sources. This lack of data limits our ability to quantitatively assess the timing and magnitude of changes in the Hg concentrations over time. A recently published study using mercury monitoring data collected between 2000 and 2020 under the national Swedish program on water chemistry found variable trends in THg concentrations (Eklöf et al., 2024; All data can be accessed from the database Miljödata MVM¹⁸ at SLU the Swedish University of Agricultural Sciences)¹⁹. Data from monthly sampling in >20 rivers and streams showed decreasing trends in about 15% of the waterbodies. The decrease was more pronounced between 2000 and 2005. However, in the last five years of the studied period (2015–2020), more waterbodies showed significantly increasing Hg trends.

¹⁸ <https://miljodata.slu.se/MVM/>

¹⁹ <https://www.slu.se/en/environment/statistics-and-environmental-data/environmental-data-catalogue/river-mouths/>

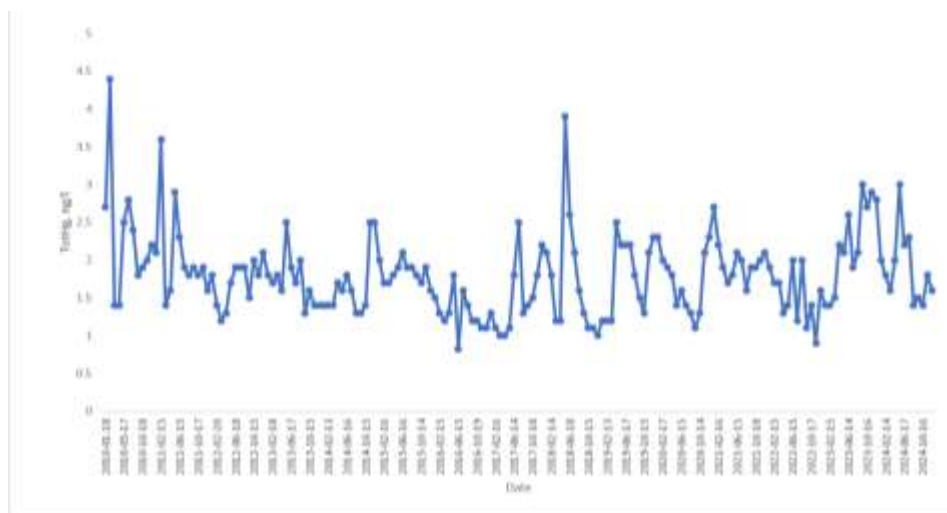


Figure 4. Long-term trends in total Hg concentrations in Ljusnan River.

Evaluation of Confidence. N/A

Information and Capacity Needs. There is a substantial data gap in long-term monitoring in lotic and lentic aquatic systems. Most existing studies connecting Hg depositions to biota and humans rely on indirect inferences about concentrations of different Hg species in water. Therefore, there is a critical need for sustained and harmonized long-term monitoring efforts across diverse geographic regions. Instead of establishing a global monitoring network, strategically selected river basins and lakes from different regions could serve as representative sites.

2.8.4 How do temporal trends of Hg levels observed in freshwater compare to or co-vary with variations and trends of related pollutants or environmental variables over the available record? (TOC, SOM, DOC, TSS)?

Information Used. To address this question we are limited to the data that was submitted by the Parties to the OESG. As mentioned above – there were a small number of datasets submitted, and for several of these, the Hg data has analytical and methodological limitations. Only the Everglades dataset has Hg data with adequate detection limits as well as DOC data; however, the ancillary parameters (including DOC concentration) have not been included in the official submission. In a publication summarizing the Everglades dataset, atmospheric Hg delivery had not changed over the study time period (2008-2018), and DOC delivery varied spatially but not temporally. There were no temporal trends in Hg or MeHg in the water column, so no temporal relationship between Hg and other parameters were observed (Janssen et al., 2022).

Methodology. N/A

Answer/Response. N/A

Consistency of Answer with Existing Literature. The mobility and transport of Hg is widely intertwined with that of DOC. Hg is readily mobilized to subsurface and surface waters along with DOM by hydrologic forcing (Demers et al., 2010; Dittman et al., 2010, 2009; Jiskra et al., 2017; Riscassi and Scanlon, 2011). In streams, both Hg and DOC concentrations increase with discharge. In headwater streams, DOC, as well as the absorbance of ultraviolet (UV) light at a wavelength of 254 nanometers (UV254), which is a common parameter used to assess the concentration of terrestrial DOC, has been strongly correlated with Hg within streams and can be used as a proxy for Hg concentrations over time (Dittman et al., 2010, 2009; Shanley et al., 2022). The temporal relationship of MeHg to DOC in rivers is less clear – MeHg concentrations have been found to increase or decrease with discharge, and MeHg production follows temperature which drives microbial activity. In streams with a large area (>10%) of wetlands and saturated soils in their catchment, MeHg concentrations often increase along with DOC during periods of high discharge, when hydrologic connectivity with the watershed increases (Riscassi and Scanlon, 2011; Schelker et al., 2011; Vidon et al., 2010). In streams where MeHg is predominantly formed within the stream channel, MeHg concentrations can be higher at baseflow, and decrease with dilution during high discharge events, which is the opposing pattern of DOC (Riscassi and Scanlon, 2011).

In lakes, DOC also plays a role in Hg stabilization in the water column, but temporal relationships between Hg and DOC are difficult to assess as variability of both parameters tends to be low over time. The temporal relationship between Hg species and DOC has been found to vary with lake watersheds. In lakes with a large area of riparian wetland, MeHg and Hg may enter the lake water column along with DOC during periods of high runoff. In lakes with minimal watershed, where precipitation is the major source of Hg, temporal relationships with DOC are less clear. In a long-term study of northern lakes in Wisconsin, a record of Hg and MeHg in lake water spanning three decades, found a strong decadal relationship between Hg and oscillation in water levels. A lake within this study having a large riparian wetland had a strong temporal relationship between Hg, MeHg and DOC, explained by periodic exposure and reflooding of littoral sediments, which promotes the release of Hg and DOC, and the conversion of Hg to MeHg by anaerobic bacteria under reducing conditions (Watras et al., 2020). The relationship with DOC was less clear in a lake where Hg delivery was predominantly through deposition (Watras et al., 2020).

Evaluation of Confidence. N/A

Information and Capacity Needs. Hg and MeHg in lakes and rivers are strongly tied to DOM; longterm monitoring of water column Hg and MeHg, along with DOM, particularly in lakes, is needed to better understand the complex role of DOM in the internal cycling driving Hg and MeHg concentrations.

2.8.5 How do THg and MMHg levels (and other species/fractions if applicable) co-vary temporally in freshwater?

Information Used. The datasets submitted by the Parties lack information about MMHg concentrations. Therefore, assessing temporal variability among Hg species is not possible.

Methodology: N/A

Answer/Response: N/A

Consistency of Answer with Existing Literature: Temporal variability of THg and MMHg is influenced by the type of freshwater systems. In lotic systems, which are characterized by continuous movement, like rivers, Hg species are typically associated with suspended particles larger than 1 µm (Guimarães et al. 2000; Åkerblom et al.

2022). As a result, their temporal variability is driven by the flowrate and the transport of particulate matter. Conversely, in lentic systems—such as lakes, reservoirs, and wetlands—mercury tends to accumulate in fine sediments and organic matter at the bottom, with MMHg formation often being more pronounced within the system due to anoxic conditions and internal cycling. Temporal variability in lentic systems is therefore driven more by seasonal stratification, redox changes, and biogeochemical cycling.

Evaluation of Confidence: N/A

Information and Capacity Needs: To enhance analysis and support more comprehensive effectiveness evaluation, the inclusion of MMHg measurements in future monitoring programs is essential. MMHg provides critical insight into Hg transformation processes and would facilitate more accurate assessment of health risks.

2.8.6 How do local THg and MMHg levels (and other species/fractions if applicable) found in freshwater differ by world regions?

Information Used. The dataset on surface freshwater concentrations does not have sufficient spatial coverage to identify differences in THg and MMHg levels between world regions. Currently, the database contains samples from two European nations (Belgium and Bulgaria), two North American nations (the United States and Canada), a South American nation (Ecuador) and Australia. Within the available national datasets, only the data from Belgium and the United States have Hg concentrations consistently above relevant reporting limits for natural waters (e.g., reporting limits of <0.5 ng/L for THg and <0.05 ng/L for MMHg). It is important to note that there is no reliable information to determine whether reported concentrations are influenced by proximate point sources. The data from Belgium and the United States are also limited to a small number of sites within each country.

The information used to assess differences in THg and MMHg levels between world regions is based on the current data submitted by the Parties to the OESG group. The inability to determine global-scale spatial differences in surface freshwater THg and MMHg levels is consistent with the existing scientific literature, which lacks studies that have attempted to synthesize freshwater datasets on a global scale.

Methodology. N/A

Answer/Response. While there have been many studies where surface freshwater THg and MMHg concentrations have been measured throughout the globe, the vast majority of this data has not been incorporated in the database and therefore can be utilized to determine global-scale spatial differences. Furthermore, much of this data is based on hypothesis driven research questions and was collected by different methods, over short and varying time scales, and does not lend itself to synthesis needed to identify broad-scale trends.

Consistency of Answer with Existing Literature. Low-level Hg water sampling can be easily impacted by the sample collection, preservation, storage, and analytical methods utilized (Creswell et al., 2015). This is particularly important for water samples because the concentrations in lakes and rivers is usually in a similar range as blanks and analytical reporting limits. Due to the impact that stream/river discharge has on THg and MMHg concentrations in flowing water (Domagalski et al., 2016), using stream/river data is not conducive to detecting spatial or temporal changes unless sufficient data is collected to establish concentrations/discharge relationships across sites. Lakes often have less variable surface water concentrations and are better candidates for long-term monitoring programs than rivers. However, in stratified lakes there can be large seasonal differences in Hg (especially MMHg) with lake depth. These variables (analytical/hydrological/depth) all contribute to the inability to determine differences in THg and MMHg between global regions.

Evaluation of Confidence. The information needed to identify differences in THg and MMHg between global regions would require surface freshwater data that has been collected using appropriate trace-metal sampling methodologies and analyzed using instrumentation that has a reporting limit that is able to capture the range of ambient concentrations. Furthermore, ancillary information on proximity to point-source/contaminated sites, hydrological conditions during sampling, and sample depth are necessary to determine spatial trends with confidence.

Information and Capacity Needs.

2.8.7 How do local temporal trends in THg and MMHg (and other species/fractions if applicable) found in freshwater differ by world regions?

Information Used. The dataset on surface freshwater concentrations does not have sufficient temporal coverage to identify differences in THg and MMHg levels between world regions. Currently, the database contains samples from two European nations (Belgium and Bulgaria), two North American nations (the United States and Canada), a South American nation (Ecuador) and Australia. However, within these national datasets only the data from Belgium and the United States have data that are consistently above relevant reporting limits for ambient concentrations in natural waters (e.g., reporting limits of <0.5 ng/L for THg and <0.05 ng/L for MMHg). The dataset from Belgium spans from 2020 to 2023 and does not provide a long enough record of measurements from which to identify temporal trends. The period of measurements in the United States is much longer, spanning from 1970 to 2019, with the Adirondacks region of New York state having the longer period of measurements (46 years—though samples not collected each year and many of years early in the record are all below reporting limits that are well above ambient concentrations in natural waters). The other region in the United States is the Everglades National Park in the state of Florida which has a long temporal record of measurements (11 years—between 2008 and 2019) and were analyzed with appropriately low analytical reporting limits (e.g. <0.5 ng/L for THg and <0.05 ng/L for MeHg). While the data from this region in the United States can be used to assess temporal trends within this one region, there is insufficient temporal data from other to draw conclusion about temporal trends between world regions.

Methodology.

Answer/Response. N/A

Consistency of Answer with Existing Literature. While there have been some studies where surface freshwater THg and MMHg concentrations have been monitored consistently throughout the globe, the vast majority of this data has not been incorporated in the current database (data submitted by the Parties) and therefore can be utilized to determine global-scale temporal differences.

Low-level Hg water sampling can be easily impacted by the sample collection, preservation, storage, and analytical methods utilized (Creswell et al., 2015). This is particularly important for water samples because the concentrations in lakes and rivers are usually in a similar range as the blanks and analytical reporting limits. To assess temporal trends within a dataset, there needs to be consistent sample collection, preservation, storage, and analytical methods utilized over time—especially because the magnitude of change over time is usually relatively small in ambient natural waters. Furthermore, due to the impact that stream/river discharge has on THg and MMHg concentrations in flowing water (Domagalski et al., 2016), using stream/river data is not conducive to detecting spatial or temporal changes unless sufficient data is collected to establish concentrations/discharge relationships across sites.

Evaluation of Confidence. N/A

Information and Capacity Needs. The information needed to identify differences in THg and MMHg between global regions would require surface freshwater data that has been collected and analyzed using appropriate trace-metal sampling methodologies and analyzed using instrumentation that has a reporting limit that is able to capture the range of ambient concentrations. Furthermore, ancillary information on proximity to point-source/contaminated sites, hydrological conditions during sampling, and sample depth are necessary to determine spatial trends with confidence.

2.8.8 How do local THg and MMHg levels (and other species/fractions if applicable) found in freshwater differ by watershed land cover or land use?

Information Used. The dataset on surface freshwater concentrations does not have sufficient spatial coverage to identify differences in THg and MMHg levels because of land cover or land use. Currently, the database contains samples from two European nations (Belgium and Bulgaria), two North American nations (the United States and Canada), a South American nation (Ecuador) and Australia. However, within these national datasets only the data from Belgium and the United States have data that are consistently above relevant reporting limits for ambient concentrations in natural waters (e.g., reporting limits of <0.5 ng/L for THg and <0.05 ng/L for MMHg). The datasets from Belgium and the United States include samples from watersheds with different land covers and land uses. For example, the data from Belgium mostly originates from the Scheldt Estuary which receives water from a large watershed containing multiple anthropogenic land uses (urban, industry, agriculture). Data from the United States consists of two distinct watershed land cover/use types. The data from the Adirondacks region of New York state is predominantly a mountainous deciduous forest with minimal anthropogenic activities and the data from the Everglades National Park in the state of Florida consists of mostly upstream agricultural land use. While the available data were collected from watersheds with distinct land covers/use; there is insufficient replication and variability in influential ancillary parameters to draw broad conclusions based on differences in land cover/use.

Three case studies presented below from watersheds in the United States highlight the long-term trends in three different systems.

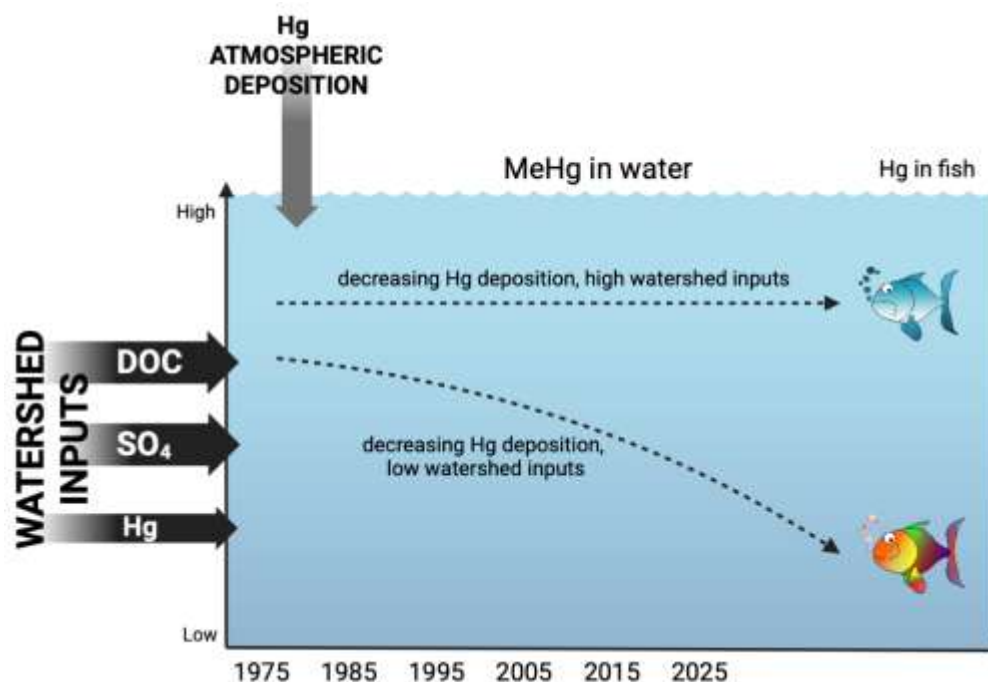
Introduction: Freshwater plays a critical role in the global Hg cycle, linking Hg transfer from the atmosphere and terrestrial environment to aquatic biota. Water is not widely included in long-term Hg monitoring and poses analytical and sampling challenges but presents an important gap in global Hg modeling and interpretation.

Overview of data: Long-term monitoring datasets for water are scarce. Here we review three case studies from U.S. lakes: Florida Everglades (2008 - 2018 (Janssen et al. 2022)), Wisconsin lakes (1988 to 2017 (Watras et al. 2019)), Voyageurs National Park, Minnesota (1998 - 2018 (Brigham et al. 2021)). All three studies include ancillary measures of water chemistry, and paired monitoring of Hg deposition and Hg concentrations in aquatic biota.

Observed trends: Florida Everglades (Janssen et al. 2022): An 11-year, 76-site monitoring record in this large, coastal wetland ecosystem found no long-term trends in Hg concentrations in water, precipitation or fish. However, spatial and annual fluctuations in aqueous Hg and MeHg concentrations inform the drivers of Hg bioaccumulation in this hot spot ecosystem (Rumbold et al. 2008). The Everglades receive precipitation with Hg concentrations 2-3 times higher than the national U.S. average (5) and are heavily impacted by upland water drainage, agricultural runoff and urban land use. A lack of change in Hg deposition from 1997-2018 (Janssen et al. 2022, Weiss-Penzias et al. 2016) is in contrast with the nationwide trend of declining Hg deposition, and attributed to frequent thunderstorms in the region, which have been found to efficiently scavenge Hg in the upper atmosphere (Guentzel et al. 2001, Holmes et al. 2010), accessing Hg from global as well as regional sources. Annual fluctuation of Hg concentrations in water was weakly correlated with Hg deposition, supporting the notion that atmospheric delivery of Hg is the primary source to the Everglades water column, though the influence of water management practices was also evident. The ecosystem supports efficient microbial methylation of Hg, with filtered water Hg concentrations being the strongest driver of MeHg concentrations across sites. The high susceptibility of Hg to methylation in this ecosystem is attributed to inputs of SO₄ and DOC from agricultural sources, coupled with redox and climate conditions (Benoit et al. 1999, Gilmour et al. 1998). Across the Park, there were strong regional patterns of Hg in mosquitofish and jewelfish, which were elevated in the coastal region and inland slough, following patterns of filtered MeHg, Hg, DOC and SO₄.

Northern Wisconsin (Watras et al. 2019): This study tracked Hg and MeHg concentrations in two contrasting lakes in Northern Wisconsin over three decades. Delivery of Hg to clear, mesotrophic, Little Rock Lake is predominantly atmospheric whereas surrounding wetlands are likely the major source of Hg to the dark water Trout Bog. Over the course of the record, there was a decline in Hg concentrations in water and atmospheric deposition, but trends in MeHg and Hg concentrations in lake water as well as Hg in fish from 176 lakes throughout the region were most strongly associated with regional oscillations in lake water level. In Little Rock Lake, aqueous Hg concentrations declined over the study period and were weakly correlated with wet deposition of Hg. However, aqueous MeHg concentration did not correlate with wet deposition, but was strongly associated with a regional, decadal oscillation in lake water level. In Trout Bog, concentrations of Hg and MeHg were 4-10 times higher than in Little Rock Lake; here, Hg, as well as SO₄ and DOC, tracked with lake water level, though MeHg did not. The influence of lake level on Hg and MeHg concentrations is linked to the wetting and drying of the terrestrial littoral zone, which drives mobilization of Hg, SO₄ and DOC, as well as causing shifts in pH and redox. This “reservoir effect” is attributed to an increase in MeHg production (St. Louis et al. 2004), and is suggested to drive temporal patterns in Hg bioaccumulation in the region.

Voyageurs, MN (Brigham et al. 2021): This study examined Hg concentrations in water and biota from four neutral-pH lakes in northern Minnesota, with minimally disturbed, flat, watersheds underlain by shallow soils and bedrock, and dominated by boreal forests. Three lakes receive inflow from a large upstream lake, and the fourth solely from headwater streams. Across the four lakes, epilimnetic MeHg and Hg declined on average by 44% and 27%, respectively from 1998 to 2018. This partly reflected wet deposition of Hg monitored at two nearby NADP sites, which declined 22% between 1998 and 2009, then stabilized through 2018. The decline in surface-water Hg extended beyond 2010, suggesting a watershed lag in Hg delivery to lake water. The large decrease in MeHg decrease reflects reductions in both Hg and SO_4 , a driver of Hg methylation. Fish Hg concentrations tracked aqueous MeHg, though significant declines were observed in only one lake. A similar pattern to the Wisconsin study, where aqueous MeHg concentrations peaked during high water levels, was evident in one lake. However, the trend was absent in the other lakes, suggesting that elevated MeHg was not driven by rewetting of watershed soils after low lake levels, but rather from increased inflow from an upstream lake. The contrasting landscapes between regions likely explains this difference: Voyageurs lakes are surrounded by shallow soils over bedrock, whereas northern Wisconsin lakes are underlain by deep, permeable outwash sands, which better favor conditions that enhance microbial methylation during wetting.



Relevance to effectiveness evaluation: Across these three case studies, monitoring aqueous Hg concentrations and water chemistry parameters informed the drivers of Hg methylation and biotic Hg concentrations. Spatial and temporal patterns of Hg and MeHg in water can identify landscape and hydrologic drivers, which can cause a lag or opposing trend to atmospheric deposition.

Data gaps and limitations: These studies are all within the U.S. and cover a limited scope of landscape and hydrologic characteristics, and proximity to point sources of pollution.

Opportunities and recommendations: Together, these studies highlight the role of water monitoring in deciphering regional and site-specific controls on temporal Hg and MeHg sources and bioaccumulation. Inclusion of aqueous Hg and MeHg, as well as water chemistry (DOC , SO_4) in long-term study sites is recommended to improve interpretation and modeling capabilities.

Consistency of Answer with Existing Literature. However, the influence of watershed land cover/use have been assessed for specific types of regions in multiple published studies and it is clear that watershed characteristics can have a large influence of Hg and MMHg concentrations in freshwater ecosystems (Balogh et al., 2005; Brigham et al., 2009; Domagalski et al., 2016; Hsu-Kim et al., 2018; Hurley et al., 1995; Shanley et al., 2008; Stoken et al., 2016). In addition, forestry operations can have a large influence on Hg and MeHg releases from soils to surface waters with differences in response related to the specific watershed characteristics and the harvest practices (Eklöf et al. 2025).

Evaluation of Confidence. The inability to determine global-scale trends between watershed with different land use/covers for THg and MMHg levels is consistent with the existing scientific literature.

Information and Capacity Needs. The information needed to identify the influence of watershed variables such as land use and cover on THg and MMHg dynamics on a global scale exists within the scientific literature for specific regions of the world, but consistent monitoring techniques across multiple global regions is needed to expand the understanding of these dynamics on a global scale.

2.8.9 How do THg and MMHg levels (and other species/fractions if applicable) co-vary spatially in freshwater?

This question cannot be addressed as there not enough datasets which reported both THg and MMHg concentrations.

2.8.10 How do spatial trends of Hg levels observed in freshwater compare to or co-vary with variations and trends of related pollutants or environmental variables over the available record? (TOC, SOM, DOC, TSS)?

Information Used. As discussed above, there was insufficient data as well as limitations in the data submitted by the Parties that prevented analysis of this question. In the Florida Everglades dataset, DOC was strongly correlated with Hg concentrations, and DOC and sulfate were strongly correlated with dissolved MeHg in the water column, and they were drivers of Hg methylation (Janssen et al., 2022; Benoit et al., 1999; Gilmour et al., 1998).

Methodology. N/A

Answer/Response. N/A

Consistency of Answer with Existing Literature. Mercury cycling is closely tied to DOC; Hg and DOC are co-mobilized from watershed soils by erosion and hydrologic forcing (Demers et al., 2013; Shanley et al., 2022), and DOC from both autochthonous and terrestrial sources stabilize Hg in the water column (Janssen et al., 2022; Lescord et al., 2018). A global meta-analysis of Hg and DOC found both THg and MeHg levels in lakes and streams increase with DOC concentration, but the slope of the relationship changes across systems (Lavoie et al., 2019). Across all studies, the relationship between DOC and Hg was stronger than with MeHg, and aromatic fraction of DOC was more strongly coupled with Hg than bulk DOC (Lavoie et al., 2019).

Evaluation of Confidence: N/A

Information and Capacity Needs: The relationship between DOC and Hg is well established, and on local scales DOC is a major driver of Hg variability. Future monitoring of Hg in water is strongly recommended to include DOC as an ancillary parameter, as this variable is a major driver of Hg variability across sites.

2.8.11 Key Findings

Freshwater data has limited spatial coverage, mostly from the Northern Hemisphere. Most data are site-specific or local. Inadequate spatial coverage makes it difficult to assess regional and global patterns.

All submissions include total Hg, none report methyl Hg.

Data usability for trend analysis is limited due to analytical uncertainties and inconsistent methodologies. Many datasets with higher analytical detection limits and data censored at the reporting limit.

Few multidecadal scientific monitoring studies exist on total Hg and methyl Hg in freshwater systems away from point sources, restricting quantitative assessments of change over time.

A national study in rivers in the Northern Hemisphere highlighted shifting trends in Hg concentrations with most recent increases.

In rivers (lotic systems), Hg variability is driven by flow and particle transport, and its mobility is closely linked to dissolved organic carbon (DOC), especially during high discharge.

In lakes (lentic systems), Hg and methyl Hg are influenced by sediment interactions, seasonal redox conditions, lake water level oscillations, or upstream inflow, with DOC playing a stabilizing role.

References:

- Guimarães, J.R.D., Ikingura, J. & Akagi, H. Methyl Mercury Production and Distribution in River Water-Sediment Systems Investigated Through Radiochemical Techniques. *Water, Air, & Soil Pollution* **124**, 113–124 (2000). <https://doi.org/10.1023/A:1005206109083>
- Balogh SJ, Nollet YH, Offerman HJ. A comparison of total mercury and methylmercury export from various Minnesota watersheds. *Science of the Total Environment* 2005; 340: 261-270.
- Brigham ME, Wentz DA, Aiken GR, Krabbenhoft DP. Mercury Cycling in Stream Ecosystems. 1. Water Column Chemistry and Transport. *Environmental Science & Technology* 2009; 43: 2720-2725.
- Creswell JE, Carter A, Engel VL, Metz JA, Davies CA. An Interlaboratory Comparison for Total Mercury and Methylmercury in Water at Environmentally Relevant Concentrations. *Water Air and Soil Pollution* 2015; 226.
- Domagalski J, Majewski MS, Alpers CN, Eckley CS, Eagles-Smith CA, Schenk L, et al. Comparison of mercury mass loading in streams to atmospheric deposition in watersheds of Western North America: Evidence for non-atmospheric mercury sources. *Science of the Total Environment* 2016; 568: 638-650.
- Eklöf, K., de Wit, H., Eckley, C.S., Eagles-Smith, C.A., Eggert, S.L., Mackereth, R.W., Skjellberg, U., Ukonmaanaho, L., Verta, M., Allan, C., Emilson, E.J.S., Kidd, K.A., Mitchell, C.P.J., Munthe, J., Sallantausta, T., Segersten, J., Bravo, A.G., Kolka, R.K., McCarter, C.P.R., Porvari, P., Ring, E., Sebestyen, S.D., Sikström, U. and Sahlén Zetterberg, T. *Environmental Science & Technology* 2025 59 (30), 15944-15955. DOI: 10.1021/acs.est.5c02787
- Eklöf, K., von Brömssen, C., Huser, B., Åkerblom, S., Augustaitis, A., Veiteberg Braaten, H.F., de Wit, H.A., Dirnböck, T., Elustondo, D., Grandin, U., Holubová, A., Kleemola, S., Krám, P., Lundin, L., Löfgren, S., Markensten, H., Moldan, F., Pihl Karlsson, G., Rönnback, P., Valinia, S., Vuorenmaa, J Trends in mercury, lead and cadmium concentrations in 27 European streams and rivers: 2000–2020, *Environmental Pollution*, 360, 2024, 124761, <https://doi.org/10.1016/j.envpol.2024.124761>.
- Hsu-Kim H, Eckley CS, Acha D, Feng XB, Gilmour CC, Jonsson S, et al. Challenges and opportunities for managing aquatic mercury pollution in altered landscapes. *Ambio* 2018; 47: 141-169.
- Hurley JP, Benoit JM, Babiarz CL, Shafer MM, Andren AW, Sullivan JR, et al. Influences of watershed characteristics on mercury levels in Wisconsin Rivers. *Environmental Science & Technology* 1995; 29: 1867-1875.
- Shanley JB, Mast MA, Campbell DH, Aiken GR, Krabbenhoft DP, Hunt RJ, et al. Comparison of total mercury and methylmercury cycling at five sites using the small watershed approach. *Environmental Pollution* 2008; 154: 143-154.
- Stoken OM, Riscassi AL, Scanlon TM. Association of dissolved mercury with dissolved organic carbon in US rivers and streams: The role of watershed soil organic carbon. *Water Resources Research* 2016; 52: 3040-3051.
- Janssen SE, Tate MT, Poulin BA, Krabbenhoft DP, DeWild JF, Ogorek JM, et al. Decadal trends of mercury cycling and bioaccumulation within Everglades National Park. *Sci Total Environ.* 2022 Sep 10;838:156031.
- Watras CJ, Grande D, Latzka AW, Tate LS. Mercury trends and cycling in northern Wisconsin related to atmospheric and hydrologic processes. 2019;846(August 2018):831–46.
- Brigham ME, Vandermeulen DD, Eagles-Smith CA, Krabbenhoft DP, Maki RP, Dewild JF. Long-term trends in regional wet mercury deposition and lacustrine mercury concentrations in four lakes in voyageurs national park. *Appl Sci.* 2021;11(4):1–20.
- Rumbold DG, Lange TR, Axelrad DM, Atkeson TD. Ecological risk of methylmercury in Everglades National Park, Florida, USA. *Ecotoxicology* [Internet]. 2008 Oct [cited 2025 Aug 16];17(7):632–41. Available from: <https://pubmed.ncbi.nlm.nih.gov/18679795/>
- NADP. <http://nadp.slh.wisc.edu/data/mdn/> last accessed 15 Jan 2024. National Atmospheric Deposition Program. 2024.
- Weiss-Penzias PS, Gay DA, Brigham ME, Parsons MT, Gustin MS, ter Schure A. Trends in mercury wet deposition and mercury air concentrations across the U.S. and Canada. *Sci Total Environ.* 2016;568:546–56.

Guentzel JL, Landing WM, Gill GA, Pollman CD. Processes influencing rainfall deposition of mercury in Florida. *Environ Sci Technol* [Internet]. 2001 [cited 2025 Aug 17];35(5):863–73. Available from: <https://pubs.acs.org/journals/esthag/index.html>

Holmes CD, Jacob DJ, Corbitt ES, Mao J, Yang X, Talbot R, et al. Global atmospheric model for mercury including oxidation by bromine atoms. *Atmos Chem Phys*. 2010;10(24):12037–57.

Benoit JM, Gilmour CC, Mason RP, Heyes A. Sulfide controls on mercury speciation and bioavailability to methylating bacteria in sediment pore waters. *Environ Sci Technol* [Internet]. 1999 Mar 15 [cited 2025 Aug 18];33(6):951–7. Available from: <https://pubs.acs.org/doi/full/10.1021/es9808200>

Gilmour CC, Riedel GS, Ederington MC, Bell JT, Benoit JM, Gill GA, et al. Methylmercury concentrations and production rates across a trophic gradient in the northern Everglades. *Biogeochemistry* [Internet]. 1998 [cited 2025 Aug 18];40(2–3):327–45. Available from: <https://link.springer.com/article/10.1023/A:1005972708616>

St. Louis VL, Rudd JWM, Kelly CA, Bodaly RA, Paterson MJ, Beaty KG, et al. The Rise and Fall of Mercury Methylation in an Experimental Reservoir. *Environ Sci Technol*. 2004;38(5):1348–58.

2.9 Ocean Water

2.9.1 What data are available?

The ocean links atmospheric concentrations with those of marine food webs. Currently, there is no systematic monitoring of marine Hg water concentrations, but an increasing number of research cruise data is being published. For this work, a total of 60 datasets containing information on total mercury (THg) and methylmercury (MeHg = MMHg+DMHg) measurements in the marine water column were compiled (Figure 1). The observations cover 20 years spanning the period 2005–2024. Most of the datasets were downloaded from public repositories (22; with the GEOTRACES data hosted at the British Oceanographic Data Centre contributing data from 11 campaigns). Another 23 datasets have been obtained from tables and supporting information presented in published scientific publications, and 15 datasets have been provided directly by authors. All datasets were from scientific work and none from monitoring programs. The datasets cover campaigns that a) move in space and time along a cruise track, b) cover a time period at a specific location or c) represent one or a few measurements/profiles in time and space.

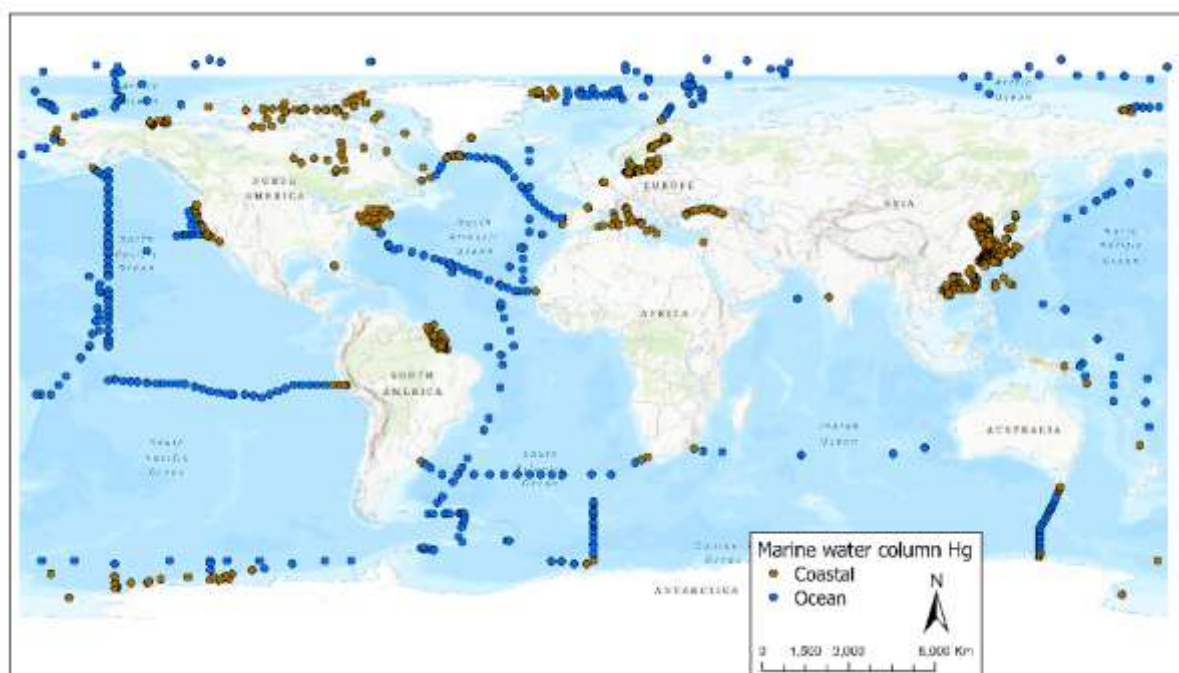


Figure 1. Location of surface observations or profiles (multiple observations with depth). Observations are divided into open ocean and coastal ocean observations based on the IPCC regions (Iturbide et al., 2020). Datasets included: Adams et al. (2024; 2025), Agather et al. (2019), Bergamaschi et al. (2012), Bowman et al. (2015, 2016), Bratkic et al. (2016), Capo and Cayian et al. (2022), Carrasco et al. 2024, Chakraborty et al. (2019), Chen et al. (2024), Ci et al. (2016), Coale et al. (2018), Cossa et al. (2011; 2018), do Nascimento et al. (2025), Eom et al. (2025), Fu et al. (2010), GEOTRACES Intermediate Data Product Group (2023), Gosnell et al. (2017; 2023), Hammerschmidt and Bowman (2012), Hammerschmidt et al. (2013), Jonsson et al. (2022), Kim et al. (2017; 2020), Kirk et al. (2008), Kohler et al. (2022), Lehnher et al. (2011), Lindeman et al. (2023), Malcolm et al. (2010), Mastromonaco et al. (2017a; 2017b), Munson et al. (2015), Perrot et al. (2023), Schlitzer and Mieruch-Schnülle (2024), Sharif et al. (2014), Soerensen et al. (2013; 2016; 2018), Starr et al. (2025), Sunderland et al. (2009), Tate et al. (2025), Torres-Rodriguez et al. (2023), Umhau et al. (2024), Wang et al. (2009; 2016; 2018; 2020), Yang et al. (2017, 2023), Yue et al. (2023).

It is not possible to go through the various methodologies used for the 60 studies. Generally acknowledged approaches for low level detection of Hg in marine environments can be found in for example Bowman et al. (2020).

Information on level of detection or level of quantification (here both are referred to as LOD for simplicity) was included in the data compilation when available as part of the original dataset, otherwise an attempt was made to extract the information from the scientific publication presenting the dataset. For each data analysis it is indicated if the analysis excluded observations below LOD or included them by setting them to $\text{LOD}/\sqrt{2}$. When available, additional information on salinity, temperature and oxygen were included in the data compilation.

Recent work has shown loss of DMHg during filtering of water samples using some filters (no loss for glass, 40% loss for Nuclepore™ and >95% loss for PES syringe filters (West et al., 2023)). This not only biases DMHg observations, but also MeHg (MMHg+DMHg) and total Hg (inorganic Hg+MeHg). The fractional importance of this uncertainty is smaller for the surface ocean (where DMHg in general contributes a lower fraction due to air-sea exchange and DMHg demethylation (Adams et al., 2024; Soerensen et al., 2016a) and smaller for THg than MeHg (possible fractional loss: DMHg > MeHg > THg). Twenty-five percent of the cruises in the data compilation reports only filtered Hg observations. In the analysis, we excluded filtered observations of DMHg and MeHg but kept filtered observations of THg following the methodology of a recent meta-analysis for Hg in the Arctic ocean (Eom et al., 2025). Filtered THg was only included when no unfiltered samples existed and both filtered and unfiltered THg observations are jointly referred to as THg in the analysis. Unfiltered MMHg and DMHg observations were combined to provide data on MeHg in cases where MeHg was not reported.

For the data analysis, the observations are divided into IPCC regions (Iturbide et al., 2020). There are 55 regions that contain ocean surfaces. These are categorized as land, ocean or land-ocean by *Iturbide et al.* (2020). For our purpose, land and land-ocean were joined to the category “coastal ocean”, which should reflect the coastal and shelf waters likely to be most heavily affected by human activity. This resulted in 12 open ocean regions and 33 coastal ocean regions. An initial analysis indicated that the regions are generally giving a useful representation of the coastal-oceanic split in observations even though the regions were not constructed with a focus on representing the coastal ocean. However, corrections were done for the coastal observations from *do Nascimento et al.* (2025), which were moved from the Equatorial Atlantic Ocean region (open ocean) to the North-East South-America region (coastal ocean), and the observations from *Sharif et al.* (2014) from the European west coast which were moved from the Mediterranean region to the West and Central European region. Further, samples collected west of the Strait of Gibraltar categorized were moved from the Mediterranean region to the North Atlantic Ocean. Of the 55 IPCC regions, the Hg data compilation covered 36 regions, 34 regions for THg (62% of all ocean regions) and 23 regions for MeHg (42% of all ocean regions; Figure 2). The open ocean regions are represented with observations in 83% of the 12 regions and the coastal ocean regions are represented with observations in 60% of the 33 regions.

Focus in the analysis was on THg and MeHg observations as most information was available for these species but similar analysis could be conducted for other Hg species (also included in the data compilation) like Hg₀, dissolved gaseous Hg (DGM), MMHg, DMHg. In the future, also Hg isotopes could be included in the data compilation and used for analysis.

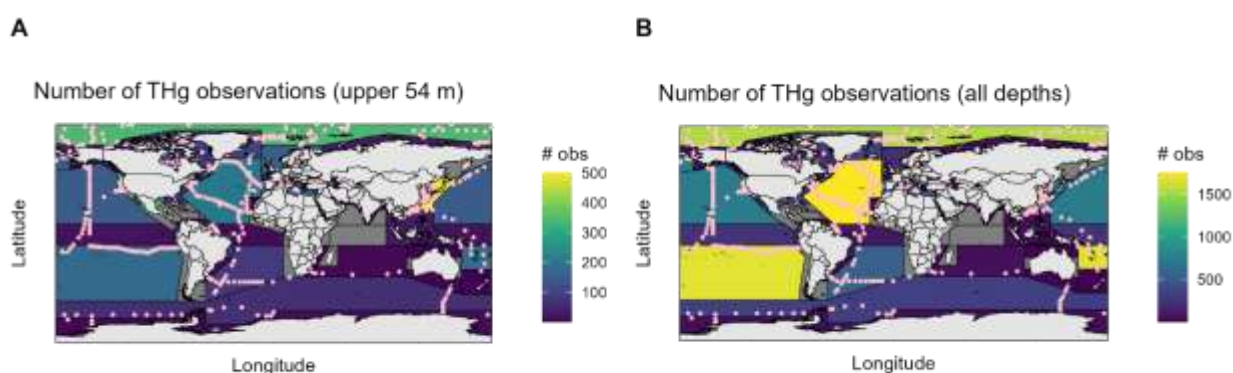


Figure 2. Number of THg observations A) in the upper 54 m of the water column (global average mixed layer depth (*de Boyer Montégut et al.*, 2004)) and B) across all depths. Pink circles indicate the sampling locations.

2.9.2 How do THg and MeHg levels found in the ocean differ by IPCC region?

The distribution of Hg in the surface ocean is primarily influenced by deposition and rivers, settling and sediment resuspension, upwelling and ocean transport (Soerensen et al., 2010). In the intermediate and deep ocean, settling and ocean circulation are most important for the spatial distribution but hydrothermal vent inputs can also play a role near the ocean bottom (Bowman et al., 2015; Torres-Rodriguez et al., 2024). The influence

of changes in the source load is highest in the surface ocean (turnover time less than a year for surface waters (Soerensen et al., 2010) but centuries in the deeper parts of the global ocean (Amos et al., 2013; Geyman et al., 2025)).

Methodology. Average THg and MeHg concentrations per IPCC region were calculated for the surface ocean (upper 54 m). Observations below LOD were included as $\text{LOD}/\sqrt{2}$ for the calculation of averages. Depth profiles were calculated by applying a gam smoother to observations from the open ocean regions.

Answer. Figure 3 shows the distribution of regional average Hg concentrations in the surface ocean. IPCC region THg averages range from 0.2 ± 0.2 pM ($n=176$) to 106 pM ($n=1$) in the surface ocean. Region MeHg averages range from <LOD to 880 ± 550 fM ($n=361$) in the surface ocean. Highest THg concentrations in the open ocean were found at high latitudes (Southern Ocean: 1.8 ± 1.1 pM; Arctic Ocean: 1.2 ± 0.7 pM), then followed the South Atlantic and North Pacific oceans (0.8 – 0.9 pM). The Equatorial, North Atlantic and South Indian oceans are all in the 0.5 – 0.6 pM range and the South Pacific Ocean has the lowest concentrations (0.2 ± 0.2 pM, $n=176$). For MeHg, the latitudinal distribution is similar with highest concentrations at high latitudes (Arctic 125 ± 94 fM; Southern Ocean 84 ± 97 fM), similar concentration range for the North and South Atlantic and North Pacific oceans (41 – 49 fM) and lowest concentrations in the South Pacific Ocean (10 ± 13 , $n=5$).

Figure 4 presents the average depth profiles for the open ocean IPCC regions. Except for a few regions, the expected nutrient-like profile is seen for THg in the open ocean regions around 500 m depth (Bowman et al., 2020). For the Arctic Ocean, concentrations at the surface are elevated as expected due to high input loads to the surface and stratification of the water column (Dastoor et al., 2022; Eom et al., 2025). For the South Atlantic Ocean the lack of a ~500m peak is related to large differences in concentrations between the two cruises providing observations for the region (~0.5 pM versus ~1.5 pM averages (Bratkič et al., 2016; Schlitzer and Mieruch-Schnülle, 2024)), and we therefore ascribe the profile to analytical inconsistencies. Below 1000 m's depth (where the thermocline waters are depleted of oxygen during microbial respiration) the Arctic and Atlantic open oceans show lower concentrations than the three Pacific oceans. The Southern Ocean has the highest concentrations across all depths. For MeHg, a nutrient-like profile with a peak at depths similar to THg is seen (Figure 4B). Kim et al. (2017) suggested that elevated MeHg at 500 m depth in the Equatorial Pacific Ocean (>750 fM) was caused by upwelling and active organic carbon mineralization in the convergence-divergence zone. For the Arctic, the peak is found at a shallower depth (few hundred meters) than seen for other oceans as previously reported (Heimbürger, 2015; Wang et al., 2018). A smaller number of observations are available at depths below 2000 m but like THg, the Atlantic and Arctic oceans MeHg concentrations are similar and even the Antarctic Ocean falls in this interval, with the North Pacific Ocean having a minimum not seen for the other regions around 2000 m's depth. Low concentrations of MeHg in the South Atlantic Ocean compared to other regions are suggested by the authors collecting the samples to be a bias related to long storage and analytical methods (Bratkič et al., 2016).

Existing Literature. The regional distribution of Hg in the ocean has not been systematically examined based on a data compilation of available observations in previous Hg assessment reports (AMAP, 2021; AMAP/UN Environment, 2019) but Bowman et al. (2020) conducted a comprehensive meta-analysis on published open ocean Hg profiles (see discussion above).

Confidence. There are a few IPCC regions where enough observations have been collected over the last two decades, to build high confidence in the representativeness of average Hg concentrations, spatial and seasonal patterns and their drivers. The Arctic Ocean is the open ocean region with the highest confidence (~1500 and ~700 open and coastal ocean THg observations) and process oriented meta-studies (e.g. Eom et al. (2025)). Additional regions with high confidence are the two North American west coast regions (~900 THg observations), where campaigns include unfiltered THg, MeHg and DMHg conducted by several research groups (Adams et al., 2024; Coale et al., 2018) and monthly variability of THg has been observed across multiple years (Adams et al., 2025), and the coastal East Asia where data from 10 campaigns with a good spatial coverage including unfiltered THg, MeHg and Hg₀ have been conducted (~570 THg observations; for example Chen et al. (2024a); Wang et al. (2020); Yang et al. (2017)). For other regions zero or very limited data is available (Figure 2). For MeHg, the choice of removing filtered MeHg water samples in the analysis has resulted in a limited number of samples despite a lot of samples having been collected.

Capacity Needs. Observations are lacking for many coastal ocean regions and the spatial coverage is sporadic for most open ocean regions. Generating new observational data is important to close this gap, both through water sampling and analysis but other methods, like the use of passive samplers (Blondet et al., 2025), should also be explored. Further, addition of existing data from coastal research or monitoring work, not currently included in the data compilation, can also improve the coverage. Uncertainty in the confidence of filtered

samples should be clarified by incorporating information on specific filtering methodologies into the data compilation. MMHg could also be analyzed to include some information lost by the exclusion of filtered MeHg.

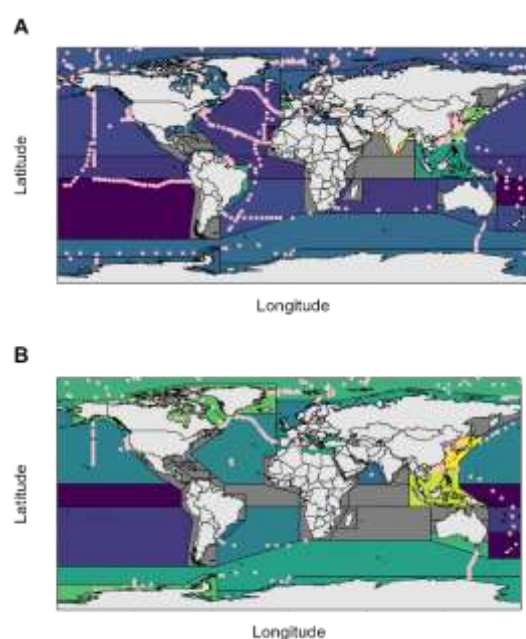


Figure 3. Spatial averages of THg and MeHg in the upper 54 m by IPCC region. Grey area indicates that no observations are available for the region. The number of THg observations per region is indicated in Figure 2A. Pink circles indicate sampling locations.

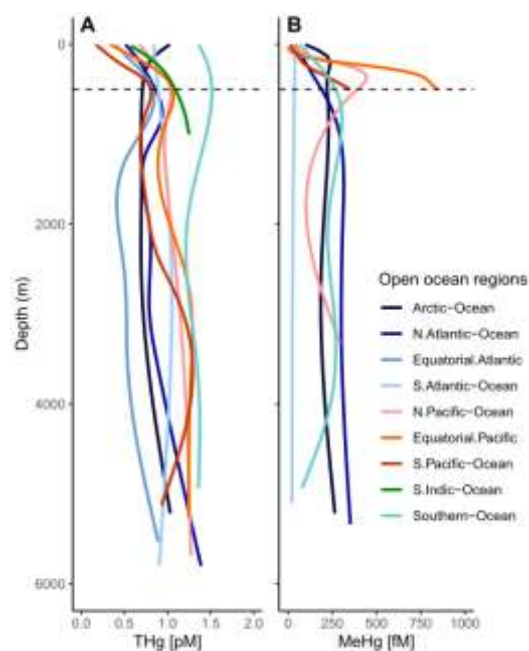


Figure 4. THg (n=7000) and MeHg (n=2200) profiles for nine open ocean IPCC regions. A gam smoother was applied to observations. Number of observations per region is shown in Figure 2B. Punctuated line indicates 500 m depth.

2.9.3 What are current levels of THg and MeHg observed in the ocean that are remote from or near anthropogenic sources?

Anthropogenic point sources have a large impact on marine Hg levels but the effect declines with distance to sources. Hg from anthropogenic point sources can be released directly to coastal water from industry or waste water discharge or through atmospheric deposition due to high industrial emissions close to the coast (Bernardello et al., 2006; Fitzgerald et al., 2018; Gascon Diez et al., 2014). The anthropogenic impact becomes more diffuse when the sources are located further inland and are transported to the ocean by rivers and/or regional atmospheric transport (Jaffe and Strode, 2008; Liu et al., 2021a).

Method. All surface ocean regions are to some extent impacted by historic Hg emissions and long-range transport. However, due to the nature of anthropogenic sources being primarily associated with land, coastal regions are likely to be more closely associated with anthropogenic primary releases than the open ocean. While this does not mean that all coastal areas are highly affected by anthropogenic primary releases, a coastal ocean:open ocean comparison between IPCC regions was used to explore the impact of anthropogenic sources on nearby and remote areas. We divided the global ocean into five latitudinal bands based on the primary location of IPCC regions (see Figure 5). The difference between coastal and open ocean observations were calculated based on mean and median concentrations for the upper 54 m within the latitudes. Focus is on the surface ocean, which has a fast response time to changes in source loads (Soerensen et al., 2010).

Answer. IPCC region average THg range from 0.2 ± 0.2 pM ($n=176$) to 1.8 ± 1.1 pM ($n=65$) in open surface ocean regions and 0.4 ± 0.1 pM ($n=4$) to 106 pM ($n=1$) in coastal surface ocean regions (Figure 3). Region average MeHg ranged from <LOD to 125 ± 94 fM ($n=197$) in open surface ocean regions and <LOD to 880 ± 550 fM ($n=361$) in coastal surface ocean regions. There is thus a factor 100 difference between the highest concentrations found in coastal regions compared to open regions for THg but less than a factor 10 for MeHg. Figure 5 presents the difference between THg and MeHg for coastal and open ocean as a function of latitude. For the Polar oceans, there is a relatively small difference between average and median THg from coastal and open ocean (93-144%) while MeHg is slightly higher in coastal regions (40-90%). For the Southern midlatitudes, Equatorial ocean and Northern midlatitudes, the coastal ocean has 400-1500% and 250-600% higher mean and median concentration, respectively, for THg and 400-900% and 350-450%, respectively, for MeHg. Overall, these results highlight a large variability in exposure of coastal ocean waters. Results from the Polar regions reflect that these areas are not directly impacted by anthropogenic sources but mainly affected by long-range transport (by direct deposition or by transport through drainage basins (Dastoor et al., 2022)). For the midlatitudes and the equatorial ocean, direct anthropogenic inputs affect many coastal observations. For the Southern midlatitudes, the enrichment of THg and MeHg is similar (400-570%), while THg shows a higher enrichment (1500%) for the Northern midlatitudes than MeHg (900%) but similar lower medians (250-350%). This reflects that MeHg from areas with high anthropogenic impact (examples: (Wang et al., 2009) (measured 15 pM MMHg), (Ci et al., 2016; do Nascimento et al., 2025; Perrot et al., 2023)) were not included in measurements from most of the Northern midlatitude (and Equatorial ocean) studies and that few very high observations of both THg and MeHg (not found in the Southern midlatitude dataset) dominate the average (for example from sampling close to harbour and industrial areas (Wang et al., 2009)), while there is a fast dilution of concentrations away from point sources.

Existing Literature. The differences in concentrations between sites near point sources and remote areas has not been systematically examined at a larger scale based on data compilations of available observations in previous Hg assessment reports (AMAP, 2021; AMAP/UN Environment, 2019) or meta-studies. Some relevant literature is discussed in the section above.

Confidence. The choice of using coastal regions as a proxy for anthropogenically polluted areas rather than gathering information on proximity of Hg sources lowers the confidence of the results. However, the fact that the presence of anthropogenic point sources in the open ocean are low, the definition of remote and “possibly impacted” regions has a high confidence. Therefore, as long as the variability in proximity of point sources in coastal areas are kept in mind during the interpretation, there is high confidence in the results. A small number of surface ocean MeHg observations in the Equatorial ocean, Southern midlatitudes and Southern ocean (<200) results in a lower confidence for these regions.

Capacity Needs. Observations are missing for many coastal regions. Further, regions are often defined by one or a few studies focused either on an anthropogenic impacted hotspot or river, the South Asia coast is an example of this (Chakraborty et al., 2019). This can result in concentrations being biased high for the region compared to studies with observations from coast to open ocean transects or studies in coastal areas away from point sources and rivers. An indication for each observation on its status as being near anthropogenic sources would improve our understanding of the direct anthropogenic influence and the impact of point sources versus diffuse sources. Alternatively, as release inventories to the coastal ocean continues to improve (Liu et al., 2020,

2021a) detailed geospatially distributed global inventories for atmospheric and land based sources (Kocman et al., 2017) could be joined with information on coastal Hg concentrations, to identify the type and level of regional/local anthropogenic influence of coastal areas across the global coastal ocean.

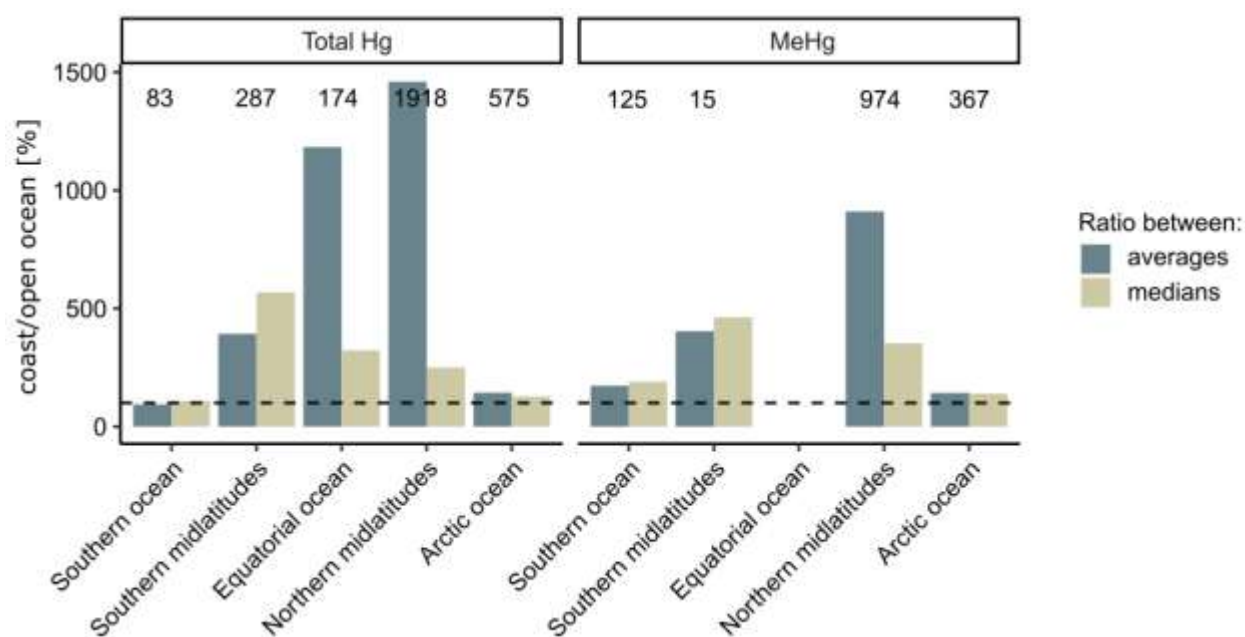


Figure 5. Mean and median differences (%) between coastal and open ocean (0-54 m depth). The five latitudinal categories were created based on the primary location of the IPCC regions (approximate latitudes: Southern ocean > 35-55° S, Southern midlatitudes 35-55° S-10° S, Equatorial ocean 10° S-8° N, Northern midlatitudes 8° N-65° N, Arctic ocean >65° N). The punctuated line indicates no difference between open and coastal ocean concentrations. Observations <LOD were included as $\text{LOD}/\sqrt{2}$. Numbers above bars indicate how many observations were used to calculate the distribution.

2.9.4 How do THg and MeHg levels co-vary spatially in the ocean?

It is well established that Hg methylation takes place both in marine sediments and the water column (Gilmour and Riedel, 1995; Lehnher et al., 2011; Schartup et al., 2025). In coastal regions Hg, in its various forms including MeHg, is also introduced by rivers or other landbased sources (Liu et al., 2021a). Hg methylation and demethylation, and thereby the percentage of THg found as MeHg, are controlled by environmental and ecological factors (e.g., temperature, light intensity, pH, redox condition, organic carbon and nutrient concentrations; (Bouchet et al., 2023; Heimbürger et al., 2010; Soerensen et al., 2017)). A thorough description of these mechanisms and the supporting literature are presented in previous assessments (AMAP, 2021; AMAP/UN Environment, 2019) and will be discussed in the next section on environmental co-variables.

Method. The percent MeHg (%MeHg: MeHg/THg) was used to examine how THg and MeHg co-vary spatially in the surface ocean and intermediate water (150-1000 m), which include the subsurface chlorophyll maximum. %MeHg was calculated for individual water samples where both MeHg and THg were above LOD (to avoid a positive bias of the results as MeHg more often than THg is below LOD). The %MeHg was then averaged per IPCC region (Table 1).

Answer. Regional average %MeHg varied between 2 and 15% for the surface ocean with a similar range found in the open ocean and coastal ocean (Table 1). The latitudinal distribution indicate a lower %MeHg in the southern hemisphere (including the Southern Ocean, ~2-5% for the open ocean and 4-10% for the coastal ocean) than the northern hemisphere (including the Arctic, ~6-12% for open ocean and 4-15% for the coastal ocean). For the intermediate ocean (150-1000 m) the range of %MeHg is also similar between open (15-36%) and coastal (7-42%) ocean. For the open ocean, the intermediate waters also seem to have lower %MeHg in the Southern Hemisphere (15%) than the Northern hemisphere (28-36%) while the coastal oceans have a more uniform distribution range (20-39% and 7-42%, respectively). A few coastal regions in the northern hemisphere (Northern Europe, West and Central Europe and East Asia) show lower %MeHg (3-5%) than the other regions in the surface ocean (9-15%) but not in intermediate water.

Table 1. %MeHg in the upper 54 m and 150-1000 m by IPCC region. Colors indicate latitudinal distribution: Arctic (blue), Northern hemisphere (green), Southern hemisphere (orange) and Antarctic (purple).

Type	IPCC region	Months (n)	Cruises (n)	MeHg _{<54} (%)	Samples (n)	MeHg ₁₅₀₋₁₀₀₀ (%)	Samples (n)
Ocean			5			33.8±23.1	253
	Arctic-Ocean	4		12.0±9.5	135		
	N.Atlantic-Ocean	3	2	10.6±5.4	62	35.7±19.2	121
	N.Pacific-Ocean	4	4	6.5±6.2	69	27.6±14.2	118
	S.Pacific-Ocean	1	1	4.5	1	14.9±10.1	11
	Southern-Ocean	4	2	2.2±1.3	42	15.1±19.2	59
Coast	Greenland/Iceland	4	6	15.1±7.0	58	36.3±21.2	70
	Russian-Arctic	1	1	9.0±4.2	25		
	N.W.North-America	3	3	12.6±10.0	12	33.5±10.3	11
	N.E.North-America	5	5	12.3±10.7	75	31.1±28.9	75
	N.Europe	3	3	3.5±3.5	44	31.3±37.9	21

West&Central-Europe	2	2	5.0±1.5	11	7.3±6.3	2
Mediterranean	2	2	9.9±13.9	48	41.9±19.3	64
W.North-America	3	2	10.1±8.2	271	25.6±12.5	85
E.Asia	3	3	4.4±4.0	293	34.9±22.6	10
S.E.Asia	1	1	10.0±3.7	20	20.3±17.2	3
S.Australia	1	1	8.4±7.7	4	39.1±8.9	3
E.Antarctica	2	2	3.9±2.7	9	21.5±17.2	14

Existing Literature. *Bowman et al.* (2020) present %MeHg for a range of open ocean cruises divided by latitude (<150 m and 150-1000 m), which range from 4-22% and 7-26%, respectively. Except for a few cruises, these fractions are higher for the surface ocean than found for the IPCC regions (<54 m). As MeHg is often depleted at the surface the inclusion of observations down to 150 m in their analysis could possibly explain this. Furthermore, it is also unclear how <LOD values were treated in the analysis. If these were included, it could result in a positive bias of the %MeHg. For the intermediate water (150-1000 m) *Bowman et al.* (2020) found %MeHg lower than our results. This could be due to the inclusion of filtered MeHg samples in the analysis, which is likely to impact the subsurface waters more than the surface ocean as DMHg often constitute >30 % of MeHg in subsurface waters but <20 % in surface water (Adams et al., 2024; Jonsson et al., 2022; Schartup et al., 2025) due to gaseous evasion and photodegradation (Soerensen et al., 2016a). *Bowman et al.* (2020) did not see a similar latitudinal gradient in the %MeHg for the open ocean found in our analysis. We have no suggestion for the drivers of such a latitudinal pattern.

The Northern European region (including the Baltic Sea) and the East Asia region (including the Yellow Sea and East Chinese Sea) are both relatively closed seas highly impacted by anthropogenic sources (Liu et al., 2021b; Soerensen et al., 2016b) where the system efficiency of MeHg production has previously been shown to be low (Chen et al., 2024b; Soerensen et al., 2018). Important drivers of low %MeHg in these systems were found to be input of terrestrial DOM (high inputs decreasing inorganic Hg availability and increasing MeHg photodemethylation rates), and the methylation potential in the system (Bouchet et al., 2023; Chen et al., 2024b; Soerensen et al., 2017).

Confidence. The number of samples for which it was possible to calculate %MeHg is low compared to the information we have for the individual observations of THg and MeHg. Especially the low number of %MeHg observations and cruise campaigns (which will incorporate an averaging effect from laboratory uncertainty of individual research groups) for the southern hemisphere is low (<80 observations for the surface ocean and <100 for the deeper ocean). This lowers the confidence in the calculated %MeHg in the southern hemisphere and the interhemispheric differences observed.

Capacity Needs. While MeHg (as well as MMHg and DMHg) can be analyzed with a low uncertainty and LOD for research groups dedicated to aquatic Hg work, often doing extensive work in the open ocean (Adams et al., 2024; Kohler et al., 2022; Starr et al., 2025), many laboratories working on a more regional/national scale, often with a focus on the coastal ocean, does not have the capabilities to analyse water samples for MeHg or have high LOD on their analysis. This hampers our understanding of the co-variability and the factors controlling the THg:MeHg relationship in marine systems. *Chen et al.* (2024b) collected information on estuarine and coastal systems averages from around the world, showing that a simple analysis is possible. However, they identified no southern hemisphere studies, showing the current bias in Hg studies towards the northern hemisphere. Expanding the collaboration between research groups working on a national scale with some of the well established Hg laboratories could help bridge this gap. The use of passive samplers could facilitate such collaboration, as sampling can be administered on site without Hg experts present and the membranes later analyzed in laboratories with well established protocols (Blondet et al., 2025).

2.9.5 How do spatial trends of THg and MeHg levels observed in the ocean co-vary with environmental variables?

Hg can co-vary with environmental variables through common sources like rivers (DOM, POM, nutrients), through co transportation and adsorption in the ocean and through the impact on speciation of environmental variables (Bravo et al., 2018; Lamborg et al., 2016). MeHg can further co-vary with variables impacting the methylation and demethylation processes (Bravo and Cosio, 2020).

Method. When available, salinity, temperature and oxygen was included in the data compilation (for 55%, 45% and 20% datasets, respectively). We focus on the spatial distribution of correlations between Hg and salinity in the surface water (<54 m) and oxygen for intermediate water (150-1000 m (Bowman et al., 2016; Munson et al., 2015)). Salinity is used as a proxy for distance to freshwater sources (and therefore a measure of the effect of anthropogenic influence that can supplement the open-coastal ocean comparison presented in Figure 5). Oxygen level is suggested to be an indicator for organic matter remineralization fueling MeHg production (Cossa et al., 2011; Heimbürger et al., 2010; Kim et al., 2017). In order to capture coast to open ocean surface salinity gradients, open ocean regions were combined with nearby coastal regions to “basins”. This also meant that observations from coastal regions containing deeper profiles were included in the Hg-oxygen analysis. Basins with less than 30 observations were not included.

Answer. For the THg-salinity correlation, we find significant correlations for non-Polar ocean basins (R^2 : 0.1-0.4) except the South Atlantic and Mediterranean while salinity did not explain the variability in Polar waters (R^2 : ~0; Figure 6). While salinity only explained a smaller part of the variability in THg, the result supports the findings from the regional comparison of differences in THg levels between coast and open ocean (little difference in Polar areas and large differences elsewhere). For MeHg, there is no latitudinal pattern, and R^2 is low (<0.2). This suggests that while MeHg concentrations are elevated in coastal oceans (Figure 5), they are not necessarily from river inputs but could be the effect of elevated in situ production in coastal regions (Soerensen et al., 2017). For several basins (North Atlantic and North Pacific) the THg concentration is variable and does not follow the trendline for observations below ~10 psu (Figure 6B). This highlights how concentrations measured close to freshwater sources are very affected by presence/absence of point sources while dilution and mixing results in a more predictable salinity-Hg relationship further from the coast. For the THg-oxygen correlation, significant correlations with $R^2 > 0.2$ were only found for the North Atlantic and South and Equatorial Pacific. For MeHg, there is data from a limited number of basins but these suggest that oxygen utilization could be driving the MeHg variability in some northern midlatitude systems (R^2 : 0.05-0.3).

Existing Literature. The data compilation used for the analysis only contained three environmental variables that have been found to co-vary with THg and MeHg of which we used only two. Salinity is often used as a proxy to source regions for coastal studies (for example *do Nascimento et al.* (2025), *Carrasco et al.* (2024)) or for determination of water masses (for example *Starr et al.* (2025), *Kim et al.* (2020)). However, we are not aware it has been used at the basin scale before. While most work has been done on the MeHg-oxygen relationship, a relationship between THg and AOU (which can approximate our THg-oxygen correlation) has also been reported (Kim et al., 2017; Yang et al., 2017). *Bowman et al.* (2020) have summarized studies exploring links between MeHg and oxygen (in the form of organic carbon remineralization (OCRR) and apparent oxygen utilization rate (AOUR)). They concluded that not all open ocean regions showed a correlation and that other factors are needed to predict the MeHg concentration. The lack of correlation for the Arctic Ocean has also previously been noted (Agather et al., 2019). This supports our findings that the oxygen level is not a good predictor for MeHg concentrations at the basin scale but this should be reevaluated with a larger dataset.

The *AMAP/UN Environment* (2019) and *AMAP* (2021) reports have more extensive descriptions of additional variables and mechanisms not available in the data compilation (although no meta-analysis were conducted). Environmental variables that can be expected to co-vary with THg and MeHg, either through co-transport or direct influence on rates of MeHg production and degradation, air temperature, light intensity, pH, redox condition, organic carbon/organic matter (and associated thiol groups) and nutrient concentrations (Bouchet et al., 2023; Bravo and Cosio, 2020; Lamborg et al., 2016; Seelen et al., 2023).

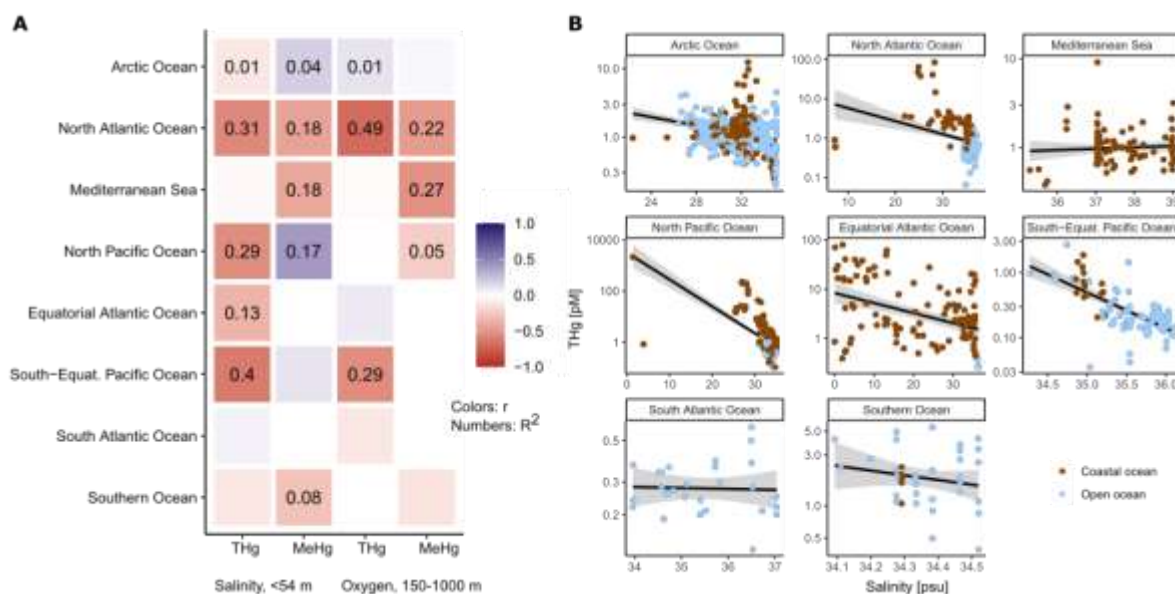


Figure 6. A) Pearson's correlation coefficient (R^2) between Hg and salinity [psu] (excluding salinities below 8) in the surface ocean (<54 m) and oxygen [$\mu\text{mol kg}^{-1}$] in the intermediate water (150-1000 m; open ocean region combined with nearby coastal regions). Only R^2 for significant correlations ($p < 0.05$) are given, B) Correlations between THg and salinity in the surface ocean (<54 m). The Mediterranean Sea is included even if it is not an open ocean and the South and Equatorial Pacific has been combined.

Confidence. Salinity observations are available for more than 50% of the datasets resulting in a relative good representation of regions. We further see an agreement between the main result from this analysis and the analysis of coast to open ocean Hg concentration differences presented in a previous section. For oxygen, the number is smaller with data missing for several of the large regions and the uncertainty of the results are therefore higher.

Capacity Needs. More data on environmental variables connected to the datasets are likely available but did not get included in the data compilation. Additional work should therefore be put into identifying important environmental variables and complementing the existing data compilation where these observations are available. Variables most likely to be available is TOC, TSS, DOC, chlorophyll fluorescence and nutrients (N and P) as well as complement with information on OCRR and AOU which requires information on oxygen, salinity and temperature.

2.9.6 How do THg and MeHg levels found in the ocean differ by watershed land cover or land use?

The impact of land cover and land use is most likely to affect THg and MeHg concentrations significantly in coastal areas, small coastal seas (like the Baltic Sea) and the small Arctic Ocean basin (Schartup et al., 2022; Soerensen et al., 2017). Except for cases where there are direct anthropogenic inputs to land or the natural background in soil is high, the differences in releases of THg and MeHg are controlled by the ability of a given land cover at a given time to withhold the Hg received from atmospheric deposition and the conditions for Hg methylation in the system.

Answer. With the information contained in the current ocean water column Hg data compilation it is not possible to investigate this question quantitatively.

Existing Literature. Diffuse sources from managed land (not including anthropogenic point sources like mining activities, industry, waste water treatment plants etc) or remobilization of Hg during humanly induced changes in land cover can result in an increase in Hg and/or MeHg releases to freshwater and coastal systems (Kocman et al., 2017). An example of this is forestry logging activities or conversion of forest to agricultural land (Eklöf et al., 2016; Gamby et al., 2015). Hg can also be remobilized during land cover changes as an indirect anthropogenic effect of climate change. An example of this is thawing permafrost, which can remobilize Hg that has been accumulating for centuries (Campeau et al., 2022; Lim et al., 2019). AMAP (2021) concluded that it is currently not possible to quantify how much permafrost thaw contributes to Arctic watershed export. Wetland areas with their oxygen-depleted conditions and often high dissolved organic matter (DOM) content have been linked to high MeHg production (Zhang et al., 2023). Bravo *et al.* (2018) found that for rivers across Europe, soil derived DOM inputs controlled the THg concentration, while the autochthonous DOM was important for the MeHg percentage. Watershed land cover and land use affects in different ways THg and MeHg concentrations and availability in freshwater systems. However, it is currently unclear to what degree this might be important on the regional and global scale for Hg in marine systems.

Capacity Needs. In the data compilation on ocean water column Hg concentrations, no information of watershed land cover or land use of nearby areas is included and combining detailed information on land cover and land use from other sources is outside the scope of what is possible in this analysis. For future Effectiveness Evaluations, a specific analysis linking coastal ocean Hg concentration with information on watershed land cover and land use, as for example presented in Potapov *et al.* (2022) (or one of many other options), and resulting releases (Kocman et al., 2017) could be attempted. Such an analysis would need to identify areas without point sources, which would likely dominate over the variability associated with land cover and land use. The analysis could focus on variability within regions, closely connected areas or areas with similar land cover types. An analysis focused on Hg in river discharge rather than the coastal ocean might be preferable in order to better link the watershed land cover and land use to the Hg concentrations.

2.9.7 How have levels of THg and MeHg observed in the ocean changed over the available record and how do their temporal trends co-vary with each other, other pollutants or environmental variables and between world regions?

The major limitation of ocean Hg observations is linked to the estimation of temporal trends, which is currently not feasible, neither long term (pre-2005) nor for the past decades. The lack of long-term stations measuring Hg in the ocean means that we must rely on sporadic cruises (different time of year, different ship tracks). Furthermore, for deeper offshore water the ocean response time can be decades to centuries (Amos et al., 2013), while the variability in Hg observations for coastal and surface water are likely to be higher than anthropogenically driven decadal trends. There is also uncertainty related with observations performed prior to the introduction of clean measurements technique, which were established in the 1980s but not uniformly used until later (Gill and Fitzgerald, 1987). This makes it difficult to know if/how to include older observations (that could otherwise help explore multi-decadal temporal trends). This conclusion is in line with previous global and Arctic assessments that have not included time trend analysis on oceanic Hg observations (AMAP, 2021; AMAP/UN Environment, 2019).

Answer. With the information contained in the current ocean water column Hg data compilation it is not possible to investigate this question quantitatively.

Existing Literature. A few research publications have discussed temporal trends based on the limited material available. These studies suggested an increase in the Pacific Ocean (total Hg, ~95% increase, 1987-2006 (Sunderland et al., 2009)) and a decrease in the subsurface Atlantic Ocean (total Hg, ~80% decline, 1979-2008 (Soerensen et al., 2012)) and Labrador Sea Water (Arctic, total Hg, ~25% decline, 1993-2015 (Bowman et al., 2020)). However, these studies are few and the uncertainty high due to the sporadic nature of a limited number of observations (time and season) and the late introduction of clean measurement techniques (Gill and Fitzgerald, 1987). Recently, the GEOTRACES GP15 cruise in the Pacific (Starr et al., 2025) compared their THg profiles (collected 2018) with the nearest profiles collected in 2006 (CLIVAR P16N (Sunderland et al., 2009)), 2009 (SaFe station (Hammerschmidt and Bowman, 2012)), 2011 (Metzyme cruise (Munson et al., 2015)), and 2013 (GEOTRACES GP16 (Bowman et al., 2016)) and concluded that THg concentrations broadly agreed across the cruises. Lower THg concentrations in subsurface water (200-800 m) in 2018 filtered samples compared to 2006 unfiltered samples (Starr et al., 2025; Sunderland et al., 2009) could indicate a decline in concentrations but could also be related to the difference in methodology (filtered versus unfiltered samples). For MeHg, Starr et al. (2025) found large discrepancies between the five datasets and concluded that, while they did not understand what caused the problem, the quantitative differences in concentrations were unlikely to be real. These case studies, indicate the current problems to establish temporal trends of oceanic THg and MeHg.

Capacity Needs. The increase in number of cruises in recent years, resulting in cross-over of stations or even cruise tracks (Starr et al., 2025) and conduction of intercalibration studies to lower measurements uncertainty (Lamborg et al., 2012; Živković et al., 2025) are important for securing the observational record that will allow us to evaluate trends for some ocean basins in the coming decades. Longer term, systematic monitoring at coastal locations, like it has been done at the North American West coast (Adams et al., 2025), can also over time inform on decadal temporal trends. In addition, developing and improving existing ocean biogeochemical models and simpler box models for the whole atmosphere-ocean-soil system has been, and will continue to be, important to fill the gaps in our knowledge on Hg spatiotemporal trends in the ocean (Dastoor et al., 2025). See [chapter XXX](#) for results on ocean spatiotemporal trends from model simulation.

2.9.8 Key Findings: Ocean Water Columns

Ocean THg and MeHg water column data was available in 62% and 42% of the 55 IPCC open ocean and coastal regions, respectively.

No water column Hg monitoring data was available and data from the scientific literature is not sufficient to conduct an evaluation of temporal trends.

For both THg and MeHg, highest concentrations in the surface ocean were found in Polar regions and lowest concentrations in the South Pacific Ocean.

For both THg and MeHg, the largest difference between coastal and open ocean concentrations were found in midlatitudes and equatorial regions, while little difference was seen for the Polar regions.

%MeHg varied between 2 and 15% for the surface ocean with a similar range found in the open ocean and coastal ocean but with an indication of lower %MeHg in the southern hemisphere open ocean.

THg correlated with salinity outside of the Polar regions while no correlation was found between MeHg and salinity. Low salinity (coastal) Hg concentrations varied by several orders of magnitude.

Better data coverage including more coastal regions, sampling method development and more laboratory/method intercomparison studies, better collaboration and use of passive samplers, and an official database extending the current data compilation and including more ancillary data are needed.

References

- Adams, H.M., Cui, X., Lamborg, C.H., Schartup, A.T., 2024. Dimethylmercury as a Source of Monomethylmercury in a Highly Productive Upwelling System. *Environ. Sci. Technol.* 58, 10591–10600.
- Adams, H.M., Wu, P., Kübler-Dudgeon, I., Lamborg, C.H., Bowman, J.S., Schartup, A.T., 2025. Coastal inorganic mercury time series reveals interannual and seasonal variability driven by regional climate factors. *Commun. Earth Environ.* 6, 293.
- Agather, A.M., Bowman, K.L., Lamborg, C.H., Hammerschmidt, C.R., 2019. Distribution of mercury species in the Western Arctic Ocean (US GEOTRACES GN01). *Mar. Chem.* 216, 103686.
- AMAP, 2021. AMAP Assessment 2021: Mercury in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Tromsø, Norway. viii + 324pp.
- AMAP/UN Environment, 2019. Technical Background Report for the Global Mercury Assessment 2018. Arctic Monitoring and Assessment Programme, Oslo, Oslo, Norway/UN Environment Programme, Chemicals and Health Branch, Geneva, Switzerland.
- Amos, H.M., Jacob, D.J., Streets, D.G., Sunderland, E.M., 2013. Legacy impacts of all - time anthropogenic emissions on the global mercury cycle. *Glob. Biogeochem. Cycles* 27, 410–421.
- Bergamaschi, B. A., D. P. Krabbenhoft, G. R. Aiken, E. Patino, D. G. Rumbold, and W. H. Orem (2012), Tidally driven export of dissolved organic carbon, total mercury, and methylmercury from a mangrove-dominated estuary, *Environ. Sci. Technol.*, 46(3), 1371-1378.
- Bernardello, M., Secco, T., Pellizzato, F., Chinellato, M., Sfriso, A., Pavoni, B., 2006. The changing state of contamination in the Lagoon of Venice. Part 2: Heavy metals. *Chemosphere* 64, 1334–1345.
- Blondet, I., Rioual, F., Graco, M.I., Diaz Ruiz, K., Aguirre-Velarde, A., Flye-Sainte-Marie, J., Laffont, L., Heimbürger-Boavida, L.-E., Torres-Rodriguez, N., Bonnet, S., Guieu, C., Raimbault, V., Point, D., 2025. Diffusive gradient in thin film for ultra-trace methylmercury measurements in the open and coastal ocean.
- Bouchet, S., Soerensen, A.L., Björn, E., Tessier, E., Amouroux, D., 2023. Mercury Sources and Fate in a Large Brackish Ecosystem (the Baltic Sea) Depicted by Stable Isotopes. *Environ. Sci. Technol.* 57, 14340–14350.
- Bowman, K.L., Hammerschmidt, C.R., Lamborg, C.H., Swarr, G., 2015. Mercury in the North Atlantic Ocean: The US GEOTRACES zonal and meridional sections. *Deep Sea Res. Part II Top. Stud. Oceanogr.* 116, 251–261.
- Bowman, K.L., Hammerschmidt, C.R., Lamborg, C.H., Swarr, G.J., Agather, A.M., 2016. Distribution of mercury species across a zonal section of the eastern tropical South Pacific Ocean (US GEOTRACES GP16). *Mar. Chem.* 186, 156–166.
- Bowman, K.L., Lamborg, C.H., Agather, A.M., 2020. A global perspective on mercury cycling in the ocean. *Sci. Total Environ.* 710, 136166.
- Bratkich, A., Vahčić, M., Kotnik, J., Obu Vazner, K., Begu, E., Woodward, E.M.S., Horvat, M., 2016. Mercury presence and speciation in the South Atlantic Ocean along the 40° S transect. *Glob. Biogeochem. Cycles* 30, 105–119.
- Bravo, A.G., Cosio, C., 2020. Biotic formation of methylmercury: A bio–physico–chemical conundrum. *Limnol. Oceanogr.* 65, 1010–1027.
- Bravo, A.G., Kothawala, D.N., Attermeyer, K., Tessier, E., Bodmer, P., Ledesma, J.L., Audet, J., Casas-Ruiz, J.P., Catalán, N., Cauvy-Fraunié, S., 2018. The interplay between total mercury, methylmercury and dissolved organic matter in fluvial systems: A latitudinal study across Europe. *Water Res.* 144, 172–182.

- Campeau, A., Eklöf, K., Soerensen, A.L., Åkerblom, S., Yuan, S., Hintelmann, H., Bieroza, M., Köhler, S., Zdanowicz, C., 2022. Sources of riverine mercury across the Mackenzie River Basin; inferences from a combined HgC isotopes and optical properties approach. *Sci. Total Environ.* 806, 150808.
- Capo, E., C. Feng, A. G. Bravo, S. Bertilsson, A. L. Soerensen, J. Pinhassi, M. Buck, C. Karlsson, J. Hawkes, and E. Björn (2022), Expression Levels of hgcAB Genes and Mercury Availability Jointly Explain Methylmercury Formation in Stratified Brackish Waters, *Environ. Sci. Technol.*, 56(18), 13119-13130.
- Carrasco, N., McGovern, M., Evenset, A., Søreide, J.E., Arts, M.T., Jonsson, S., Poste, A.E., 2024. Seasonal riverine inputs may affect diet and mercury bioaccumulation in Arctic coastal zooplankton. *Sci. Total Environ.* 906, 167643.
- Chakraborty, P., Jayachandran, S., Lekshmy, J., Padalkar, P., Sitlhou, L., Chennuri, K., Shetye, S., Sardar, A., Khandeparker, R., 2019. Seawater intrusion and resuspension of surface sediment control mercury (Hg) distribution and its bioavailability in water column of a monsoonal estuarine system. *Sci. Total Environ.* 660, 1441–1448.
- Chen, L., Cheng, G., Zhou, Z., Liang, Y., Ci, Z., Yin, Y., Liu, G., Cai, Y., Li, Y., 2024a. Methylmercury cycling in the Bohai Sea and Yellow Sea: Reasons for the low system efficiency of methylmercury production. *Water Res.* 258, 121792.
- Chen, L., Li, K., Li, Y., 2024b. System efficiency of methylmercury production, an important indicator for evaluating the long-term risk of mercury: From a case study in the Jiaozhou Bay to global coastal seas. *J. Hazard. Mater.* 480, 136286.
- Ci, Z., Zhang, X., Yin, Y., Chen, J., Wang, S., 2016. Mercury redox chemistry in waters of the eastern Asian seas: from polluted coast to clean open ocean. *Environ. Sci. Technol.* 50, 2371–2380.
- Coale, K., Heim, W., Negrey, J., Weiss-Penzias, P., Fernandez, D., Olson, A., Chiswell, H., Byington, A., Bonnema, A., Martenuk, S., 2018. The distribution and speciation of mercury in the California current: Implications for mercury transport via fog to land. *Deep Sea Res. Part II Top. Stud. Oceanogr.* 151, 77–88.
- Cossa, D., Heimbürger, L.-E., Lannuzel, D., Rintoul, S.R., Butler, E.C., Bowie, A.R., Averty, B., Watson, R.J., Remenyi, T., 2011. Mercury in the southern ocean. *Geochim. Cosmochim. Acta* 75, 4037–4052.
- Cossa, D., L.-E. Heimbürger, F. F. Pérez, M. I. García-Ibáñez, J. E. Sonke, H. Planquette, P. Lherminier, J. Boutorh, M. Cheize, and J. L. Menzel Barraqueta (2018), Mercury distribution and transport in the North Atlantic Ocean along the GEOTRACES-GA01 transect, *Biogeosciences*, 15(8), 2309-2323.
- Dastoor, A., Angot, H., Bieser, J., Brocza, F., Edwards, B., Feinberg, A., Feng, X., Geyman, B., Gournia, C., He, Y., 2025. The Multi-Compartment Hg Modeling and Analysis Project (MCHgMAP): mercury modeling to support international environmental policy. *Geosci. Model Dev.* 18, 2747–2860.
- Dastoor, A., Angot, H., Bieser, J., Christensen, J.H., Douglas, T.A., Heimbürger-Boavida, L.-E., Jiskra, M., Mason, R.P., McLagan, D.S., Obrist, D., 2022. Arctic mercury cycling. *Nat. Rev. Earth Environ.* 3, 270–286.
- de Boyer Montégut, C., Madec, G., Fischer, A.S., Lazar, A., Iudicone, D., 2004. Mixed layer depth over the global ocean: An examination of profile data and a profile - based climatology. *J. Geophys. Res. Oceans* 109.
- do Nascimento, L.S., Pestana, I.A., Seidel, M., da Silva Junior, J.M., de Oliveira, B.C.V., Gomes, P.R., Koschinsky, A., Dittmar, T., de Rezende, C.E., 2025. Mercury concentrations along the Amazon estuary and plume: Spatial trends and geochemical processes. *Environ. Res.* 121129.
- Eklöf, K., Lidskog, R., Bishop, K., 2016. Managing Swedish forestry’s impact on mercury in fish: Defining the impact and mitigation measures. *Ambio* 45, 163–174.
- Eom, S., Soerensen, A.L., Rhee, T.S., Hong, J.K., Son, P., Rho, T.K., Han, S., 2025. Properties of inflowing Pacific and Atlantic water govern total and methylated mercury profiles in the Arctic Ocean. *Environ. Pollut.* 374, 126254.
- Fitzgerald, W.F., Engstrom, D.R., Hammerschmidt, C.R., Lamborg, C.H., Balcom, P.H., Lima-Braun, A.L., Bothner, M.H., Reddy, C.M., 2018. Global and local sources of mercury deposition in coastal New England reconstructed from a multiproxy, high-resolution, estuarine sediment record. *Environ. Sci. Technol.* 52, 7614–7620.
- Fu, X., X. Feng, G. Zhang, W. Xu, X. Li, H. Yao, P. Liang, J. Li, J. Sommar, and R. Yin (2010), Mercury in the marine boundary layer and seawater of the South China Sea: Concentrations, sea/air flux, and implication for land outflow, *J. Geophys. Res.-Atmos.*, 115(D6).

- Gamby, R.L., Hammerschmidt, C.R., Costello, D.M., Lamborg, C.H., Runkle, J.R., 2015. Deforestation and cultivation mobilize mercury from topsoil. *Sci. Total Environ.* 532, 467–473.
- Gascon Diez, E., Bravo, A.G., à Porta, N., Masson, M., Graham, N.D., Stoll, S., Akhtman, Y., Amouroux, D., Loizeau, J.-L., 2014. Influence of a wastewater treatment plant on mercury contamination and sediment characteristics in Vidy Bay (Lake Geneva, Switzerland). *Aquat. Sci.* 76, 21–32.
- GEOTRACES Intermediate Data Product Group (2023), The GEOTRACES Intermediate Data Product 2021v2 (IDP2021v2). NERC EDS British Oceanographic Data Centre NOC. doi:10.5285/ff46f034-f47c-05f9-e053-6c86abc0dc7e
- Geyman, B.M., Streets, D.G., Olson, C.I., Thackray, C.P., Olson, C.L., Schaefer, K., Krabbenhoft, D.P., Sunderland, E.M., 2025. Cumulative Anthropogenic Impacts of Past and Future Emissions and Releases on the Global Mercury Cycle. *Environ. Sci. Technol.*
- Gill, G.A., Fitzgerald, W.F., 1987. Picomolar mercury measurements in seawater and other materials using stannous chloride reduction and two-stage gold amalgamation with gas phase detection. *Mar. Chem.* 20, 227–243.
- Gilmour, C., Riedel, G., 1995. Measurement of Hg methylation in sediments using high specific-activity ²⁰³Hg and ambient incubation. *Water. Air. Soil Pollut.* 80, 747–756.
- Hammerschmidt, C.R., Bowman, K.L., 2012. Vertical methylmercury distribution in the subtropical North Pacific Ocean. *Mar. Chem.* 132, 77–82.
- Hammerschmidt, C. R., M. B. Finiguerra, R. L. Weller, and W. F. Fitzgerald (2013), Methylmercury accumulation in plankton on the continental margin of the Northwest Atlantic Ocean, *Environ. Sci. Technol.*, 47(8), 3671-3677.
- Heimbürger, L.-E., 2015. Mercury measurements in the ocean-GEOTRACES intercalibration exercises. Presented at the AGU Fall Meeting Abstracts, pp. GC51F-1157.
- Heimbürger, L.-E., Cossa, D., Marty, J.-C., Migon, C., Averty, B., Dufour, A., Ras, J., 2010. Methyl mercury distributions in relation to the presence of nano-and picophytoplankton in an oceanic water column (Ligurian Sea, North-western Mediterranean). *Geochim. Cosmochim. Acta* 74, 5549–5559.
- Iturbide, M., Gutiérrez, J.M., Alves, L.M., Bedia, J., Cerezo-Mota, R., Cimadevilla, E., Cofiño, A.S., Di Luca, A., Faria, S.H., Gorodetskaya, I.V., 2020. An update of IPCC climate reference regions for subcontinental analysis of climate model data: Definition and aggregated datasets. *Earth Syst. Sci. Data* 12, 2959–2970.
- Jaffe, D., Strode, S., 2008. Sources, fate and transport of atmospheric mercury from Asia. *Environ. Chem.* 5, 121–126.
- Jonsson, S., Mastromonaco, M.G.N., Gårdfeldt, K., Mason, R.P., 2022. Distribution of total mercury and methylated mercury species in Central Arctic Ocean water and ice. *Mar. Chem.* 242, 104105.
- Kirk, J. L., V. L. St. Louis, H. Hintelmann, I. Lehnher, B. Else, and L. Poissant (2008), Methylated mercury species in marine waters of the Canadian high and sub Arctic, *Environ. Sci. Technol.*, 42(22), 8367-8373.
- Kim, H., Soerensen, A.L., Hur, J., Heimbürger, L.-E., Hahm, D., Rhee, T.S., Noh, S., Han, S., 2017. Methylmercury mass budgets and distribution characteristics in the Western Pacific Ocean. *Environ. Sci. Technol.* 51, 1186–1194.
- Kim, J., Soerensen, A.L., Kim, M.S., Eom, S., Rhee, T.S., Jin, Y.K., Han, S., 2020. Mass budget of methylmercury in the East Siberian Sea: the importance of sediment sources. *Environ. Sci. Technol.* 54, 9949–9957.
- Kocman, D., Wilson, S.J., Amos, H.M., Telmer, K.H., Steenhuisen, F., Sunderland, E.M., Mason, R.P., Outridge, P., Horvat, M., 2017. Toward an assessment of the global inventory of present-day mercury releases to freshwater environments. *Int. J. Environ. Res. Public. Health* 14, 138.
- Kohler, S.G., Heimbürger-Boavida, L.-E., Petrova, M.V., Digernes, M.G., Sanchez, N., Dufour, A., Simić, A., Ndungu, K., Ardelan, M.V., 2022. Arctic Ocean’s wintertime mercury concentrations limited by seasonal loss on the shelf. *Nat. Geosci.* 15, 621–626.
- Lamborg, C.H., Hammerschmidt, C.R., Bowman, K.L., 2016. An examination of the role of particles in oceanic mercury cycling. *Philos. Trans. R. Soc. Math. Phys. Eng. Sci.* 374, 20150297.

- Lamborg, C.H., Hammerschmidt, C.R., Gill, G.A., Mason, R.P., Gichuki, S., 2012. An intercomparison of procedures for the determination of total mercury in seawater and recommendations regarding mercury speciation during GEOTRACES cruises. *Limnol. Oceanogr. Methods* 10, 90–100.
- Lehnher, I., St. Louis, V.L., Hintelmann, H., Kirk, J.L., 2011. Methylation of inorganic mercury in polar marine waters. *Nat. Geosci.* 4, 298–302.
- Lim, A.G., Sonke, J.E., Krickov, I.V., Manasypov, R.M., Loiko, S.V., Pokrovsky, O.S., 2019. Enhanced particulate Hg export at the permafrost boundary, western Siberia. *Environ. Pollut.* 254, 113083.
- Lindeman, M. R., F. Straneo, H. M. Adams, M. J. S. Nelson, and A. T. Schartup (2024), Low mercury concentrations in a Greenland glacial fjord attributed to oceanic sources, *Commun. Earth. Environ.*, 5(1), 320.
- Liu, C., Chen, L., Liang, S., Li, Y., 2020. Distribution of total mercury and methylmercury and their controlling factors in the East China Sea. *Environ. Pollut.* 258, 113667.
- Liu, M., Zhang, Q., Maavara, T., Liu, S., Wang, X., Raymond, P.A., 2021a. Rivers as the largest source of mercury to coastal oceans worldwide. *Nat. Geosci.* 14, 672–677.
- Liu, M., Zhang, Q., Yu, C., Yuan, L., He, Y., Xiao, W., Zhang, H., Guo, J., Zhang, W., Li, Y., 2021b. Observation-based mercury export from rivers to coastal oceans in East Asia. *Environ. Sci. Technol.* 55, 14269–14280.
- Malcolm, E. G., J. K. Schaefer, E. B. Ekstrom, C. B. Tuit, A. Jayakumar, H. Park, B. B. Ward, and F. M. Morel (2010), Mercury methylation in oxygen deficient zones of the oceans: no evidence for the predominance of anaerobes, *Marine Chemistry*, 122(1-4), 11-19.
- Marumoto, K., A. Takeuchi, S. Imai, H. Kodamatani, and N. Suzuki (2018), Mercury evasion fluxes from sea surfaces of the Tsushima Strait and Kuroshio Current in the East China Sea, *Geochemical Journal*, 52(1), 1-12.
- Mastromonaco, M. G. N., K. Gårdfeldt, K. M. Assmann, S. Langer, T. Delali, Y. M. Shlyapnikov, I. Zivkovic, and M. Horvat (2017a), Speciation of mercury in the waters of the Weddell, Amundsen and Ross Seas (Southern Ocean), *Marine Chemistry*, 193, 20-33.
- Mastromonaco, M. G. N., K. Gårdfeldt, and I. Wängberg (2017b), Seasonal and spatial evasion of mercury from the western Mediterranean Sea, *Marine Chemistry*, 193, 34-43.
- Munson, K.M., Lamborg, C.H., Swarr, G.J., Saito, M.A., 2015. Mercury species concentrations and fluxes in the Central Tropical Pacific Ocean. *Glob. Biogeochem. Cycles* 29, 656–676.
- Perrot, V., Ma, T., Vandeputte, D., Smolikova, V., Bratkic, A., Leermakers, M., Baeyens, W., Gao, Y., 2023. Origin and partitioning of mercury in the polluted Scheldt Estuary and adjacent coastal zone. *Sci. Total Environ.* 878, 163019.
- Potapov, P., Hansen, M.C., Pickens, A., Hernandez-Serna, A., Tyukavina, A., Turubanova, S., Zalles, V., Li, X., Khan, A., Stolle, F., 2022. The global 2000-2020 land cover and land use change dataset derived from the Landsat archive: first results. *Front. Remote Sens.* 3, 856903.
- Schartup, A.T., Soerensen, A.L., Angot, H., Bowman, K., Selin, N.E., 2022. What are the likely changes in mercury concentration in the Arctic atmosphere and ocean under future emissions scenarios? *Sci. Total Environ.* 836, 155477.
- Schartup, A.T., Soerensen, A.L., Calder, R.S., Heimbürger-Boavida, L.-E., 2025. Vertical Distribution of Methylmercury in the Central Arctic Ocean Explained by In Situ Methylation and Demethylation.
- Schlitzer, R., Mieruch-Schnülle, S., 2024. THE GEOTRACES INTERMEDIATE DATA PRODUCTS. *Oceanography* 37, 25–33.
- Seelen, E., Liem-Nguyen, V., Wünsch, U., Baumann, Z., Mason, R., Skjellberg, U., Björn, E., 2023. Dissolved organic matter thiol concentrations determine methylmercury bioavailability across the terrestrial-marine aquatic continuum. *Nat. Commun.* 14, 6728.
- Sharif, A., Monperrus, M., Tessier, E., Bouchet, S., Pinaly, H., Rodriguez-Gonzalez, P., Maron, P., Amouroux, D., 2014. Fate of mercury species in the coastal plume of the Adour River estuary (Bay of Biscay, SW France). *Sci. Total Environ.* 496, 701–713.
- Soerensen, A.L., Jacob, D.J., Schartup, A.T., Fisher, J.A., Lehnher, I., St. Louis, V.L., Heimbürger, L., Sonke, J.E., Krabbenhoft, D.P., Sunderland, E.M., 2016a. A mass budget for mercury and methylmercury in the Arctic Ocean. *Glob. Biogeochem. Cycles* 30, 560–575.

- Soerensen, A.L., Jacob, D.J., Streets, D.G., Witt, M.L., Ebinghaus, R., Mason, R.P., Andersson, M., Sunderland, E.M., 2012. Multi - decadal decline of mercury in the North Atlantic atmosphere explained by changing subsurface seawater concentrations. *Geophys. Res. Lett.* 39.
- Soerensen, A.L., Schartup, A., Skrobonja, A., Bouchet, S., Amouroux, D., Liem - Nguyen, V., Björn, E., 2018. Deciphering the role of water column redoxclines on methylmercury cycling using speciation modeling and observations from the Baltic Sea. *Glob. Biogeochem. Cycles* 32, 1498–1513.
- Soerensen, A.L., Schartup, A.T., Gustafsson, E., Gustafsson, B.G., Undeman, E., Björn, E., 2016b. Eutrophication Increases Phytoplankton Methylmercury Concentrations in a Coastal Sea: A Baltic Sea Case Study. *Environ. Sci. Technol.* 50, 11787–11796.
- Soerensen, A.L., Schartup, A.T., Skrobonja, A., Björn, E., 2017. Organic matter drives high interannual variability in methylmercury concentrations in a subarctic coastal sea. *Environ. Pollut.* 229, 531–538.
- Soerensen, A.L., Sunderland, E.M., Holmes, C.D., Jacob, D.J., Yantosca, R.M., Skov, H., Christensen, J.H., Strode, S.A., Mason, R.P., 2010. An improved global model for air-sea exchange of mercury: High concentrations over the North Atlantic. *Environ. Sci. Technol.* 44, 8574–8580.
- Soerensen, A.L., R. P. Mason, P. H. Balcom, and E. M. Sunderland (2013), Drivers of surface ocean mercury concentrations and air–sea exchange in the West Atlantic Ocean, *Environ. Sci. Technol.*, 47(14), 7757-7765.
- Starr, L.D., He, Y., Mason, R.P., Hammerschmidt, C.R., Newell, S.E., Lamborg, C.H., 2025. Mercury distribution and speciation along the US GEOTRACES GP15 Pacific meridional transect. *J. Geophys. Res. Oceans* 130, e2024JC021672.
- Sunderland, E.M., Krabbenhoft, D.P., Moreau, J.W., Strode, S.A., Landing, W.M., 2009. Mercury sources, distribution, and bioavailability in the North Pacific Ocean: Insights from data and models. *Glob. Biogeochem. Cycles* 23.
- Tate, M. T., D. P. Krabbenhoft, and S. E. Janssen (2025), Total Mercury Concentrations in waters from the Pacific (2006), Indian (2009), Southern (2011) and Atlantic (2013) Ocean Basins, US Geological Survey (USGS) Data Release, 581.
- Torres-Rodriguez, N., Yuan, J., Petersen, S., Dufour, A., González-Santana, D., Chavagnac, V., Planquette, H., Horvat, M., Amouroux, D., Cathalot, C., 2024. Mercury fluxes from hydrothermal venting at mid-ocean ridges constrained by measurements. *Nat. Geosci.* 17, 51–57.
- Wang, C., Wang, Z., Zhang, X., 2020. Characteristics of mercury speciation in seawater and emission flux of gaseous mercury in the Bohai Sea and Yellow Sea. *Environ. Res.* 182, 109092.
- Wang, K., Munson, K.M., Beaupré-Laperrière, A., Mucci, A., Macdonald, R.W., Wang, F., 2018. Subsurface seawater methylmercury maximum explains biotic mercury concentrations in the Canadian Arctic. *Sci. Rep.* 8, 14465.
- Wang, Shaofeng, Jia, Y., Wang, Shuying, Wang, X., Wang, H., Zhao, Z., Liu, B., 2009. Total mercury and monomethylmercury in water, sediments, and hydrophytes from the rivers, estuary, and bay along the Bohai Sea coast, northeastern China. *Appl. Geochem.* 24, 1702–1711.
- Wang, C., Z. Ci, Z. Wang, and X. Zhang (2016), Air-sea exchange of gaseous mercury in the East China Sea, *Environ. Pollut.*, 212, 535-543.
- West, J., Babi, D., Azaroff, A., Jonsson, S., 2023. Dimethylmercury in natural waters—analytical and experimental considerations. *Limnol. Oceanogr. Methods* 21, 837–846.
- Umhau, B. P., L. C. Motta, J. D. Blum, H. G. Close, J. C. Drazen, B. N. Popp, and C. R. Benitez-Nelson (2024), Particulate mercury export in the Central Pacific Ocean using $^{234}\text{Th}/^{238}\text{U}$ disequilibria, *Marine chemistry*, 265, 104433.
- Yang, J., Kim, H., Kang, C.-K., Kim, K.-R., Han, S., 2017. Distributions and fluxes of methylmercury in the East/Japan Sea. *Deep Sea Res. Part Oceanogr. Res. Pap.* 130, 47–54.
- Yang, S., P. Li, K. Sun, N. Wei, J. Liu, and X. Feng (2023), Mercury isotope compositions in seawater and marine fish revealed the sources and processes of mercury in the food web within differing marine compartments, *Water Research*, 241, 120150.
- Yue, F., Y. Li, Y. Zhang, L. Wang, D. Li, P. Wu, H. Liu, L. Lin, D. Li, and J. Hu (2023), Elevated methylmercury in Antarctic surface seawater: The role of phytoplankton mass and sea ice, *Sci. Total. Environ.*, 882, 163646.

Zhang, J., Li, C., Tang, W., Wu, M., Chen, M., He, H., Lei, P., Zhong, H., 2023. Mercury in wetlands over 60 years: research progress and emerging trends. *Sci. Total Environ.* 869, 161862.

Živković, I., Heimbürger-Boavida, L.-E., Petrova, M.V., Dufour, A., Begu, E., Horvat, M., 2025. Achieving measurement comparability in mercury speciation analysis in seawater: Key requirements and best practices. *Mar. Chem.* 104498.

2.10 Other Natural Archives

2.10.1 What data are available?

2.10.1.1 Peat Cores

A total of 18 cores across six (6) datasets are available for analysis (Figure 23 and Appendix II). Measurements include total Hg concentrations ($n = 6$ datasets), total mercury fluxes ($n = 4$), inorganic mercury flux ($n = 1$) and mercury stable isotopes ($n = 2$) representing ten (10) remote sites and eight (8) impacted sites. Peat records cover the period 4650 BC to 2018 AD, and cores have been retrieved from the following countries: France (2), Falkland Islands (1; the only core in the Southern Hemisphere), China (1), Canada (2), and Sweden (1). All compiled data are publicly available.

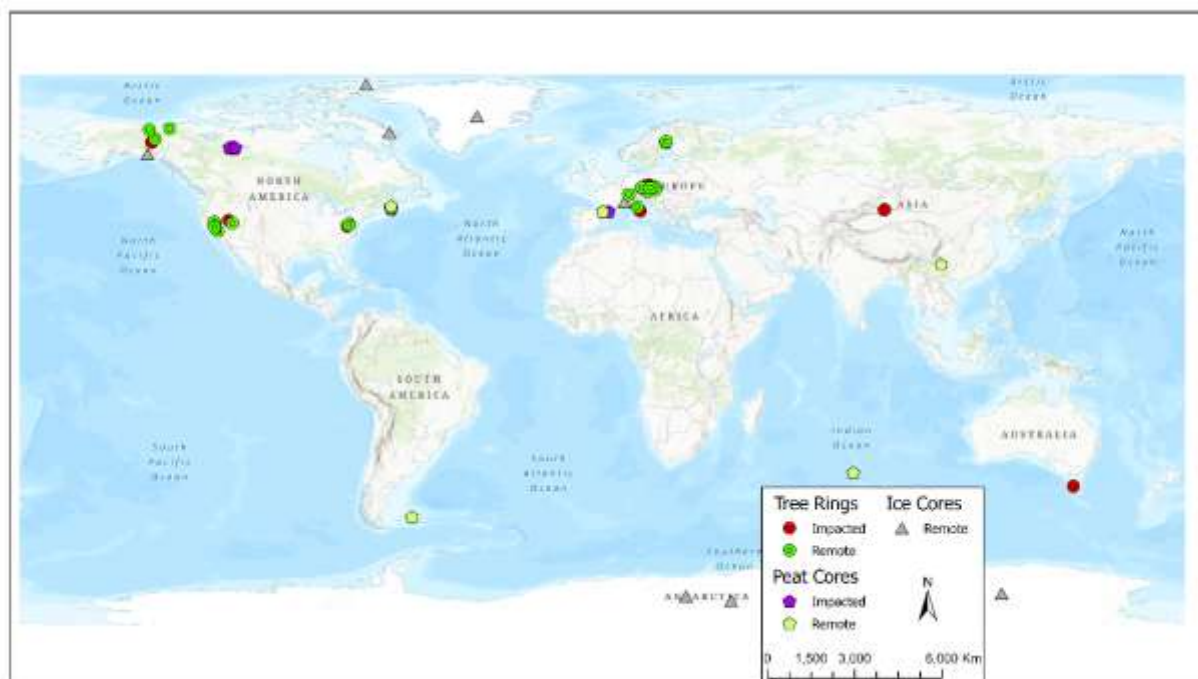


Figure 23. A map showing the distribution of the available data for peat cores, ice cores and tree ring. Tree rings cores depicted with circles. Ice cores depicted with triangles, and peat cores depicted with stars.

Forty-five (45) additional datasets have been identified but are in the possession of private data holders and have not been obtained and/or are awaiting Data Use Authorization. These datasets represent a total of 98 cores. 85 cores are in the Northern Hemisphere in 14 different countries, covering the period 16240 BC to 2019 AD (Figure 8). Nearly 70% ($n = 58$) of these cores are in five countries: Spain (16), Canada (13), China (10), Sweden (10), Czech Republic (9). Thirteen (13) cores are located in the Southern Hemisphere: Chile (6), South Africa (3), Bolivia (2), Brazil (1), and Lesotho (1), spanning the period 69050 BC to 2018 AD (Figure 21 and Figure 22).

2.10.1.2 Ice Cores

A total of 10 records (unique cores or snow pits) across 8 datasets are available for analysis (Figure 23). Measurements include total Hg concentrations ($n = 9$ datasets), inorganic mercury concentrations ($n = 3$), methyl Hg concentrations ($n = 4$), and total mercury flux ($n = 4$) in ice/firn cores (6) and snowpits (4), in 10 remote sites (Appendix II). Ice records cover the period 670050 BC to 2017 AD (Figure 10). Ice records are found primarily in the Northern Hemisphere, including Canada (3), Greenland (1), and France (1). Three records in the Southern Hemisphere are found solely in Antarctica. Compiled ice data are publicly available ($n = 7$) or have been shared by private data holders ($n = 1$).

Fifteen (15) additional datasets have been identified but are in the possession of private data holders and have not been obtained and/or are awaiting Data Use Authorization. These datasets include total mercury measurements in 18 ice/firn cores and 18 snowpits, as well as a single dataset of Hg stable isotopes, together

spanning the period 13636 BC to 2012 AD. These privately held datasets originate from the following locations: Antarctica (4 datasets), Greenland (5), China (3), Canada (2), and one each in Russia and United States. In addition, two of these datasets include measurements of gaseous elemental mercury measured in the pores of two ice cores retrieved from Greenland and represent unique archives of historic trends in atmospheric mercury.

2.10.1.3 Tree Rings

A total of 62 records (records identified as Hg time series of a unique tree species and unique site) across 16 datasets are available for analysis (Figure 23). Measurements include total Hg concentrations ($n = 16$ datasets) and stable isotopes ($n = 1$) in tree rings, representing 30 remote sites and 32 impacted sites with point source(s) of mercury identified by authors in published literature (Appendix II). Tree ring records cover the period 1550 to 2020. All records are in the Northern Hemisphere, excluding one, and include the following countries: Italy (1), Germany (1), Canada (4), United States (2), China (1), Czech Republic (5), Switzerland (1), and Tasmania (1; only South Hemisphere record) (Figure 21 and Figure 22). Compiled tree ring data are publicly available ($n = 11$) or have been shared by private data holders ($n = 5$).

Seventeen (17) additional datasets have been identified but are in the possession of private data holders and have not been obtained and/or are awaiting Data Use Authorization. These private datasets include at least 21 records representing remote locations and 37 records reflecting one or more point sources, spanning the period 1790 to 2022 AD. Private datasets originate primarily from the Northern Hemisphere, and include data in the following countries: China (3), France (2), Italy (2), Republic of Korea (2), United States (2), and one each in Austria, Canada, Russia, Serbia, and Turkey. A single dataset in Peru represents the only privately held dataset in the Southern Hemisphere and is awaiting publication.

3 Integrated Analysis

The chapter “Integrated analysis” focuses on four set of key question groups, namely,

- Cross Media Comparisons of Observed Trends and Patterns
- Process Understanding and Model Evaluation
- Process and Source Attribution for Air and Oceans
- Process and Source Attribution for Biota, Humans, and Other Media

Every set of key questions would require information on the cross-media levels across air, ocean, biota, humans and others and the integrated understanding of environmental processes by modeling or other scientific approaches.

Although we have outcomes of the analysis in each media as shown in earlier sections in this report, those outcomes are not necessarily well-organized to enable scientific cross media comparisons among media considering geographical or temporal consistencies, biological nature in the mercury processes, human exposure and environmental and social conditions, and other factors to be considered. Modeling study is substantially in progress, which would be the methodology to integrate cross media levels and processes in global scale. However, current efforts are around process understanding and model evaluation, which suggest the difficulties to employ the current outcomes of modeling study as the basis of process integration at this stage. Integration of biota and human media and global, regional, to local environmental levels in media may be difficult in the current framework of global modeling study or in other scientific information.

Based on the above understanding of the science and available data basis, this chapter of integrated analysis will work on two categories of information to give feasible and sound cross-media integration in the first cycle of effectiveness evaluation.

- ✓ Modeling assessment: mainly focused to answer questions around “process understanding and model evaluation”
- ✓ Cross-media case studies experienced in global, regional or national scales submitted by experts of OESG, to show the existing examples of cross-media integration among media in various scientific studies. Those focused on cross-media temporal trends and geographical patterns and understandings of process and/or attributions from sources to the media, based on observations, modeling and other information.

Table 3.1 showed summaries of indexed information from modeling and case studies. Outcomes of current progress of modeling study and submitted case studies were evaluated whether each would give whole, part, or none of answer to the key questions set out as four set of question groups as shown in above. The section listed each of key questions one by one, and identified and described how information summarized in Table 3.1. could give, or difficult to give, the answer. The purpose of the analysis would summarize the starting basis of integrated analysis of scientific evaluation in next or future effectiveness evaluation cycles.

Table 3.1. Model assessment and case studies referred in the integrated analysis

Index	Topic	Leading experts	Reference
MA-1 to 15	Model Assessment of Hg Levels, Spatial Patterns and Temporal Trends (Progress Report)	The Multi-Compartment Mercury Modelling and Analysis Project (MCHgMAP)	See reference* and section 3.2 and 3.3
FFL-1	Impacts of reductions in mercury emissions on the Flin Flon ecosystem	Marlene Evans	Appendix-FFL
IATT-1 to 4	Integrated Analysis Study of Mercury in Tropical Tuna Species	Robert Mason, Anaïs Médieu, Raphael Lavoie, Mi-Ling Li	Appendix-IATT

MEDT-1 to 2	Mercury in the Mediterranean Region and the Minamata Convention	Milena Horvat, Jože otnik, Igor Živkovič, Jan Gačnik, Oleg Travnikov, Adna Alilović, Janja Snoj Tratnik, Vanja Usenik, David Kocman, Ermira Begu	Appendix-MEDT
ASGMP-1	Mercury Pollution from Artisanal Gold Mining in Madre de Dios, Peru: A Cross-Media Case Study	Claudia Vega	Appendix-ASGMP
GL-1	Great Lakes Regional Case Study	Mark Burton, Celia Chen	Appendix-GL
SWED-1	Integrated Analysis of Mercury in Sweden	Pianpian Wu, Kevin Bishop, Karin Eklöf, John Munthe, Anne Sørensen, Erik Björn, Claudia von Brömssen	Appendix-SWED
AUST-1	Integrated Analysis of Mercury in the Derwent Estuary, Australia	Olha FURMAN, Akira Weller-Wong	Appendix-AUST
ECS-1	Oceanic mercury monitoring in the East China Sea	Koji Marumoto, Akinori Takeuchi	Appendix-ECS

*<https://gmd.copernicus.org/articles/18/2747/2025/gmd-18-2747-2025.html>

10.1 Cross Media Comparisons of Observed Trends and Patterns

10.1.1 Are the spatial patterns and temporal trends of Hg observed in air consistent with trends in estimated emissions?

The Northern Hemispheric average concentration is higher than the Southern Hemispheric average concentration, which is consistent with historical anthropogenic emission patterns. In recent decades, anthropogenic emissions in much of the Northern Hemisphere have declined or stayed constant. This trend in emissions is broadly consistent with the trends observed by active TGM measurements, which are mostly in the Northern Hemisphere. Emissions in the Southern Hemisphere, mostly associated with ASGM, have been increasing. Passive air samplers have been used recently to observe high air concentrations in ASGM areas. However, an increase in the Southern Hemispheric average air concentrations has not been observed at the limited number of sites where long term observations are available. Additional sustained observations in the Southern Hemisphere are needed.

Relating information appears in model evaluation section (10.2) which will be useful to answer this question in future.

10.1.2 Are the spatial patterns and temporal trends in Hg observed in soils, sediments, surface and ocean waters consistent with trends in estimated releases to land and water?

Information Used (including applicable indigenous and traditional knowledge)

There is no straightforward information to answer this question. Two regional case studies may give part of answer on the question, and we refer the two in this section.

Methodology

FFL-1 “Impacts of reductions in mercury emissions on the Flin Flon ecosystem” provided by Marlene Evans, Canada, introduced long term efforts on the copper-zinc smelter at Flin Flon, Manitoba, Canada, including emission reduction, spatial and temporal patterns in mercury in soils, peat bogs, and lake sediments, and research on smelter emissions and the biological environment. The study is an example of cross-media integration of patterns and trends among emission/releases, environmental media and biota. Although the study includes biota information, the study is located in this section as major observations seem to be in emission, soil and sediments.

ECS-1 “Oceanic mercury monitoring in the East China Sea” provided by Koji Marumoto and Akinori Takeuchi, Japan, introduced concurrent monitoring of atmosphere and ocean in East China Sea. This small case study is an example of cross-media integration of trends in same region showing ocean-sea exchange.

Answer/Response

Case study FFL-1 showed decreasing concentrations in lake sediments with increasing distance from the smelter due to the increasing dilution of emissions with distance and deposition. The emission of mercury declined to 4% of 1975-1997 average in 2008, however levels in sediment seems not clearly changing in the period.

Case stud ECS-1 showed decreasing concentrations in both in surface water in East China Sea and atmospheric concentration in Cape Hedo, which is assumed to reflect emissions in the regions, although authors did not give clear implication on how the concurrent decreasing trend could be interpreted.

Both case studies showed general observation that emission/releases reduction may not be directly reflected in the environmental media, however, there is not clear interpretations in terms of the mechanisms.

Consistency of Answer with Existing Literature (especially the 2018 GMA)

It is difficult to argue consistency between the global assessment and regional case studies.

Evaluation of Confidence

Information shown in the case studies are confident, at least in the context of each case studies. General comments in Answer/Response may need future evaluation of confidence.

Information and Capacity Needs

Long term monitoring and appropriate interpretation are necessary to assess the effectiveness of the convention further.

Linkages among regional case studies and global analysis may also be necessary for further study.

Summary of referred case studies

FFL-1

Impacts of reductions in mercury emissions on the Flin Flon ecosystem

Description for the case study: The copper-zinc smelter at Flin Flon, Manitoba, began its operation in 1930, using two short (53 and 69 m tall) stacks to release its emissions. It was until improvements were made in the operation of its Zn processing facility in 1982, the single largest Hg emitter in North America (Pai et al. 2000, Wiklund et al. 2017) with emissions of 20,000 kg in 1990, representing 57% of total Canadian emissions (Outridge et al. 2011, Environment and Climate Change Canada 2016). It also was a major emitter of Pb, Zn and other heavy metals with an average particulate emission rate of 7,150 t/yr over 1931-1974 and 6,834 t/yr over 1985-1995 (McMartin et al. 1999) and the fourth 4th largest single SO₂ emitter in Canada emitting 192,320 tonnes/yr (Franklin et al. 1985).

In an effort to reduce toxic emissions from smelters, The Government of Canada (2006) set 2008 and 2015 target releases. For Flin Flon, the 2008 SO₂ target release was 187,000 tonnes/yr with further reductions to 33,500 tonnes/yr by 2015, reductions to 88% and 16% respectively of the 212,000 tons/yr releases in the late 1970s and early 1980s (Franklin et al. 1985). Target reductions for particulates was 930 tonnes/yr by 2008 and 616 tonnes/yr by 2015 down to 14% and 9% respectively of 1975-1997 average dust emissions of 6,834 t/yr (McMartin et al. 1999). Individual targets were not set for metals except for Hg to 373 kg/y by 2008 down to 4% of 1975-1997 average emissions of 9.3 t/yr (McMartin et al. 1999) and 2% of peak estimates of 20 tonnes/y estimates in the late 1970s early 1980s (Wiklund et al. 2017). Challenges were encountered in meeting target SO₂ emissions and the Cu smelter Description closed in 2010 although the zinc refinery continued to operate through the early 2020s.

Cross-media integration of observed levels/trends

Reductions in Hg and other emissions: Flin Flon met with some success in reducing its emissions to meet The Government of Canada (2006) 2008 and 2015 target releases (Fig 1). Hg emissions began to decline after 1980 from ca. 20,000 kg to ca. 4,200 kg in 1995 with further declines through to 2010 (Wiklund et al. 2017); the 2008 target was not met with Hg emissions decreasing below this target with the 2010 closure of the copper smelter.

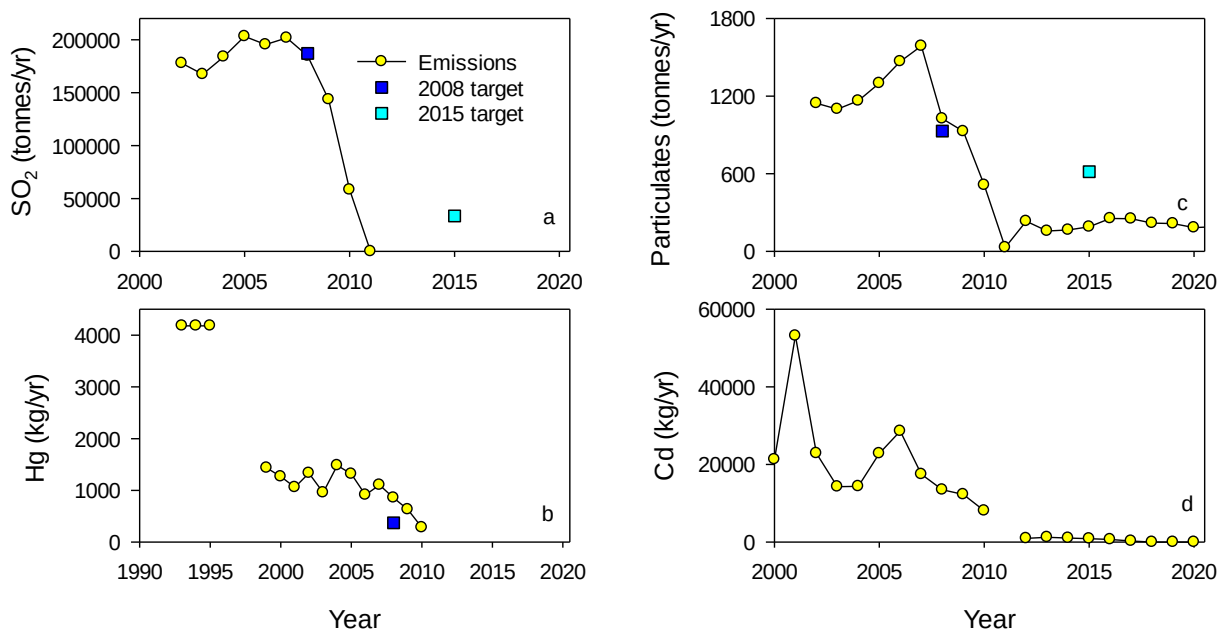


Figure 1. Declines in SO₂ (a), Hg (b), particulates (c) and Cd (d) emissions from the copper zinc smelter at Flin Flon, Manitoba. Also shown are the target reductions in SO₂, Hg, and particulate emission targets by the Government of Canada (2006). No targets were set for metal other than Hg.

Spatial and temporal patterns in mercury in soils, peat bogs, and lake sediments

Harrison and Klaverkamp (1990) measured Hg in addition to Cu, Cd, Se and Se concentrations in the upper 2 cm of lake sediments at distances ranging from 4.5 to 84 km of the smelter (Fig 2). They observed a classic pattern of decreasing concentrations with increasing distance from the smelter due to the increasing dilution of emissions with distance and deposition.

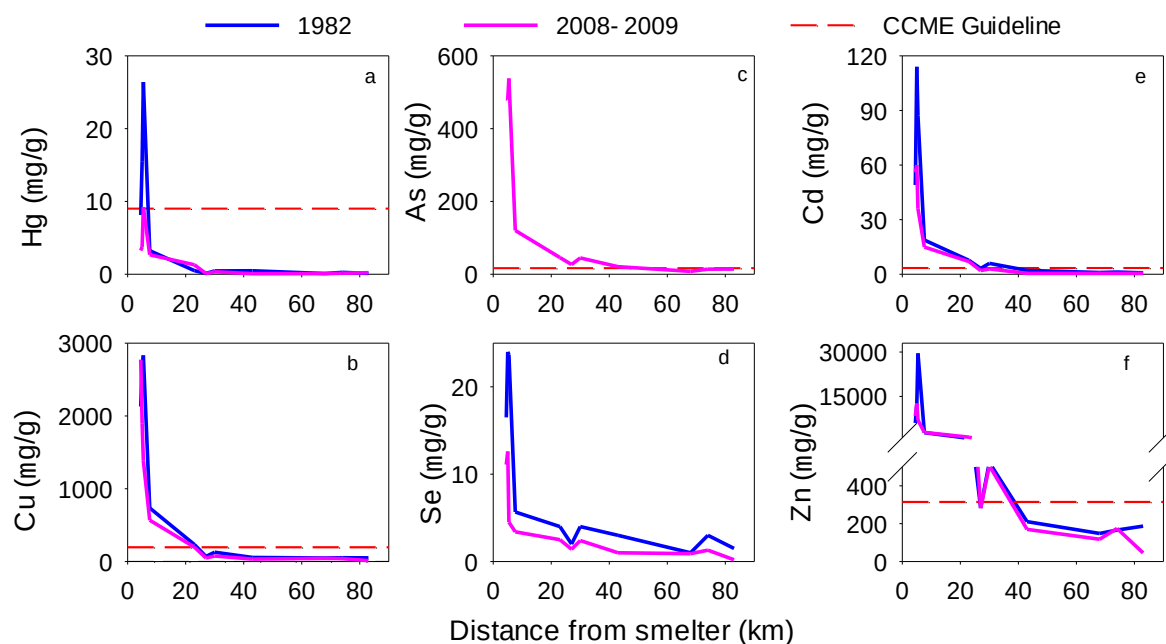


Figure 2. Metal concentrations in surface sediments in lakes at varying distances from the Flin Flon smelter as measured in 1982 by Harrison and Klaverkamp (1990) and in 2008-2009 by Evans et al. (In review). Arsenic (2c) was not measured in 1982. Also shown are the Canadian Council of Ministers of the Environment (2014) Probable Effects Guidelines for the protection of aquatic life in sediments. No guidelines have been established for selenium (Fig. 2 d).

Smelter emissions and the biological environment.

Harrison and Klaverkamp (1990) conducted the primary study investigating the impact of smelter emissions on the metal content on Northern pike (*Esox lucius*) and white sucker (*Catostomus commersoni*) from the same 13 lakes located 4.5 to 84 km from smelter where they conducted sediment core studies. They determined, as already noted, that metal concentrations were extremely low in northern pike (and white sucker) in lakes whose sediments were highly contaminated with mercury and other metals. While they did not measure metals in water, we can assume that metals would have shown the same spatial patterns and possibly similar concentration as observed in Evans et al. (In review). Other limnological variables seemed to have little influence on metal concentrations in fish. Over 2009-2017, Evans conducted various studies of mercury concentrations in northern pike (and white suckers) presenting findings at scientific meetings the most recent being Evans et al. (2017) with a manuscript now in preparation building on Evans et al. (In review), and including a closer examination of the mercury fish data reported in Harrison et al. (1989). The higher mercury concentrations in lakes 74 km to the north is within the normal variability of mercury concentrations in pike and appears to be due to a combination of fish age and undetermined features of the watersheds and lakes.

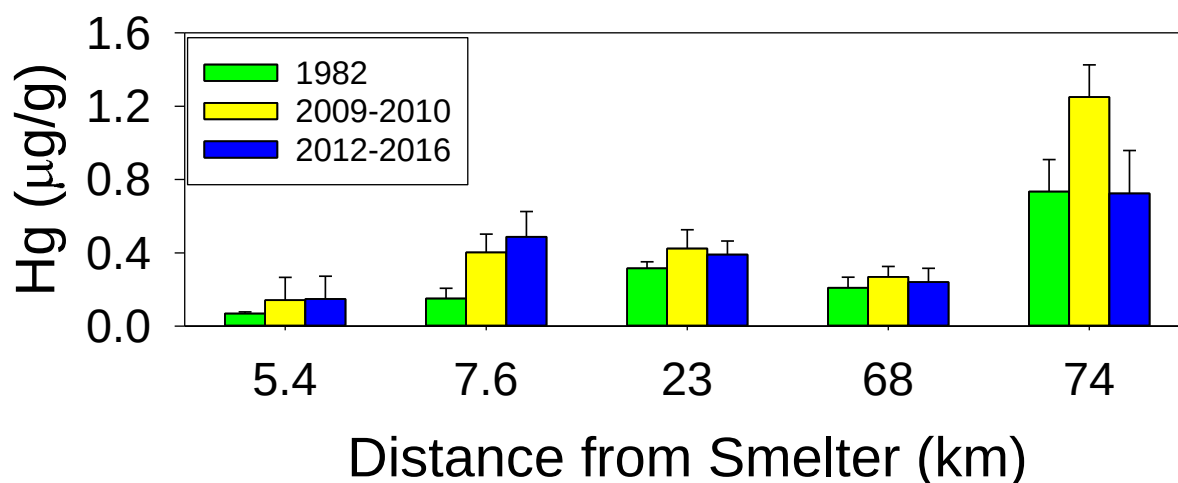


Figure 4. Spatial and temporal trends in mercury concentrations in length-adjusted (525 mm) northern pike fillet in lakes varying distance from the smelter during full smelter operation (1982), shorter before and during the closure of the copper smelter (2009-2010) and subsequent years (2012-2016). From (Evans et al. 2017).

Relevance to effectiveness evaluation:

Studies conducted at Flin Flon have shown that while the Cu-Zn smelter was a major mercury emitter, these mercury emissions appear to have had little negative impact on the aquatic environment. Unlike studies conducted around pulp and paper mills where large quantities of mercury were released into the aquatic environment where it rapidly entered the food chain reaching dangerously high concentrations in fish (Armstrong and Scott 1979, Gewurtz et al. 2010, Neff et al. 2012), high emission rates of other heavy metals may have occurred in sufficiently high concentrations in the aquatic environment as to impair mercury methylation rates; high concentrations of selenium may have interfered with mercury uptake.

Information and capacity needs:

There are several research needs for the Flin Flon area including a better understanding of the factors affecting the low mercury concentrations in northern pike in lakes near the smelter. The potential role of heavy metal toxicity in affecting mercury methylation rates requires a fuller understanding. Furthermore, if high concentrations of Se are interfering with Hg uptake, this needs to be better understood as it potentially could be used as a remedy elsewhere in limiting Hg uptake including through the consumption of organisms containing high mercury concentrations.

A long-term research and monitoring plan is desirable for Flin Flon to investigate how lake ecosystems respond to declining emissions and the re-entry of metals into lakes from the watershed. Fish mercury monitoring and studies are particularly desirable in lakes located with 7 km of the smelter. Lakes located 73-84 km to the northwest also are of interest because north pike can have high mercury concentrations in these lakes for reasons not fully understood. Waters are occasionally acidic which may facilitate mercury methylation rates.

ECS-1

Oceanic mercury monitoring in the East China Sea

Description for the case study

On the northwestern edge of the Pacific Ocean, there are three marginal seas connected by warm currents branching off from the Kuroshio Current; the East China Sea, the East/Japan Sea, and the Sea of Okhotsk. These seas are important components of the marine system in the northwestern Pacific Ocean, influencing regional weather patterns and supporting diverse marine ecosystems. Especially, the East China Sea is one of the world's richest fishing grounds because its primary production is supported by nutrient supply from various processes, including atmospheric deposition, ocean currents, upwelling, eddies, river and groundwater inflows, and human activities [1,2].

We mainly focused on the Kuroshio Current region of the East China Sea, where data are scarce, and examined the vertical and horizontal distributions of dissolved Hg, particulate Hg, and dissolved MeHg in seawater and

total Hg and MeHg in the planktons. In addition, Hg evasion fluxes were investigated using a two-layer gas exchange model [3].

Cross-media integration of observed levels/trends

Time series trend of Hg in surface seawaters

In the East China Sea, Hydrographic and Oceanographic Department, Japan Coast Guard (JCG) have observed total Hg in surface seawaters every year [5]. As shown in Figure 1, the decreasing trend of Hg concentrations in surface seawater in the East China Sea as well as North Atlantic Ocean {6}. The collateral decreasing trend of atmospheric gaseous elemental Hg (GEM) at Cape Hedo, Okinawa was observed (Figure 2). In addition, the stronger decreasing trend of GEM was observed in warm season compared with that in the cold season, which air masses including some air pollutants from the Asian continent are transported to the monitoring site by westerlies and northwestern monsoons [7]. Therefore, one possible explanation on these decreasing trends is the influence of the interactions of air-sea exchange.

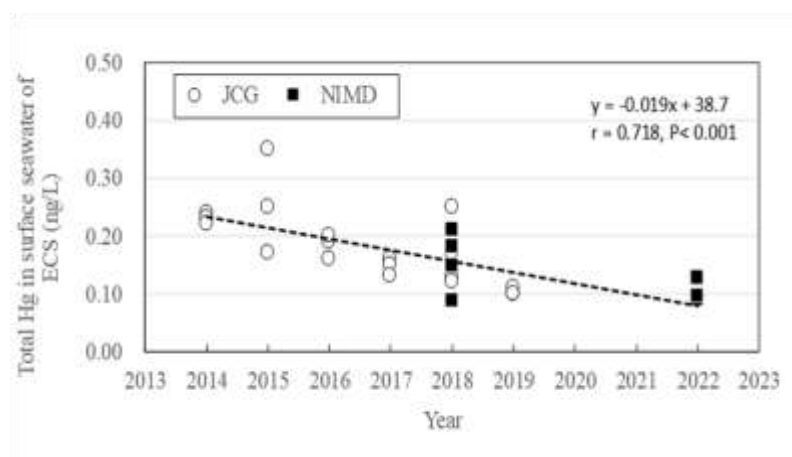


Figure 1 Long-term trends of total Hg in surface seawater of the East China Sea

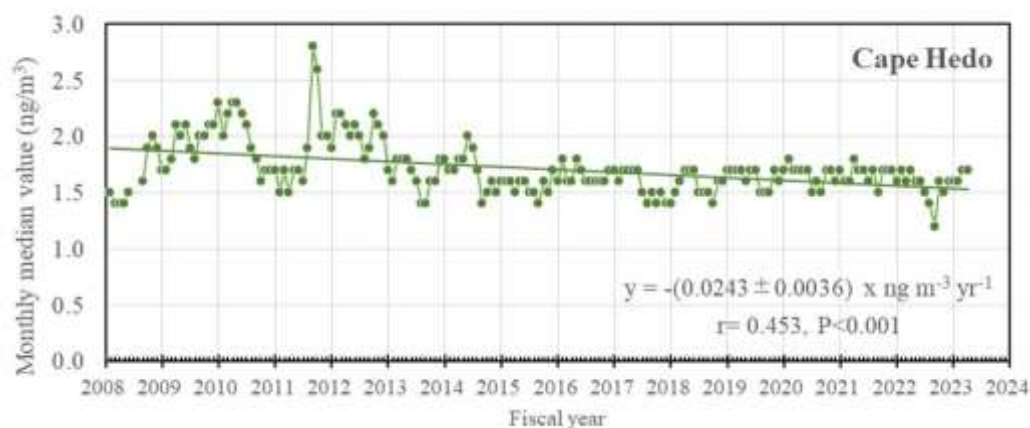


Figure 2 Long-term trends of atmospheric GEM concentration at Cape Hedo, Okinawa [7]

10.1.3 Are the spatial patterns and temporal trends in Hg observed in air, soils, sediments, surface and ocean waters consistent with trends in biota or humans?

Information Used (including applicable indigenous and traditional knowledge)

There is no straightforward information to answer this question. Four regional case studies may give part of answer on the question, and we refer the two in this section.

Methodology

MEDT-1 “Mercury in the Mediterranean Region and the Minamata Convention” provided by Milena Horvat, et al., Slovenia, introduced comprehensive summary of cross-media monitoring data on air, water and sediments, biota, and human biomonitoring together with the discussion of cross-media integration. As the study showed spatial trend of various media in the region and temporal trend in some monitoring examples, we could discuss relations among abiotic and biotic media in terms of consistency.

ASGMP-1 “Mercury Pollution from Artisanal Gold Mining in Madre de Dios, Peru: A Cross-Media Case Study” provided by Claudia Vega, Peru, introduced the study trying to illustrate the causal link between mercury emissions from artisanal and small-scale gold mining (ASGM) activities and elevated mercury levels in air, aquatic ecosystems, and terrestrial biodiversity.

SWED-1 “Integrated Analysis of Mercury in Sweden” provided by Pianpian Wu, et al., Sweden, introduced long-term efforts and observations in Sweden on widespread mercury contamination in the biota of Fennoscandia already in the 1960s, during the same decade as the cause of Minamata disease. The study covered mercury emissions and releases, atmospheric mercury deposition, and mercury in surface waters, fish, and birds.

AUST-1 “Integrated Analysis of Mercury in the Derwent Estuary” provided by Olha FURMAN, Australia, introduced long-term efforts and observation in Australia on The Derwent Estuary located within Hobart, the capital of Tasmania. The study covered modern control measures, emission/releases, and mercury observations in ambient water and sediments, invertebrates, fish and birds.

Answer/Response

Case study MEDT-1 showed both spatial distribution of mercury/methylmercury levels in the surface waters and mercury levels in fish species in the region. Also it showed mercury levels in biota having the temporal trends from the period from 1974-1990 to 2011-2025. Those observations are the basis of further discussions on cross-media integration of both spatial and temporal trends in the region.

Case study ASGMP-1 showed comparative observations in sediment and fish, birds and bats, and levels in air between ASGM sites and control sites. The study showed mercury levels were consistently and significantly higher in biota from ASGM sites in comparison with control sites.

Case study SWED-1 showed several observations: Hg deposition has dropped by nearly 75% since the 1990s; legacy sources remain active. General declining trends in Hg concentrations in many rivers between 2000 and 2010, but more seldom after 2010. Offshore surface sediment Hg declining from ~1970s to 2010s, thereafter no clear general decline. Hg in fish trends showed declining, while others showed opposite or stable levels.

Case study AUST-1 showed reductions in total anthropogenic mercury inputs over the past two decades, but no significant temporal declines in mercury concentrations in fish between 2002 and 2024. In comparison, mercury concentrations in deployed oysters either showed no significant change or increased annually by 1.6% to 3.4% over the same period.

Those case studies showed general observation that emission/releases reduction may not be directly reflected in the environmental media, however, there is not clear interpretations in terms of the mechanisms.

Consistency of Answer with Existing Literature (especially the 2018 GMA)

It is difficult to argue consistency between the global assessment and regional case studies.

Evaluation of Confidence

Information shown in the case studies are confident, at least in the context of each case studies. General comments in Answer/Response may need future evaluation of confidence.

Information and Capacity Needs

Long term monitoring and appropriate interpretation are necessary to assess the effectiveness of the convention further.

Linkages among regional case studies and global analysis may also be necessary for further study.

Summary of referred case studies

MEDT-1

Mercury in the Mediterranean Region and the Minamata Convention

Description of the case study

The Mediterranean is a region where the environment and everyday life are deeply connected. With more than 20 countries bordering its shores and nearly half a billion people living in its catchment, it is a place where seafood is not only a key source of food and income but also a cornerstone of culture. Yet this same connection to the sea brings challenges: mercury (Hg) contamination has long been a concern.

Part of the reason lies in the sea's semi-enclosed nature — water circulates slowly, so pollutants tend to remain and build up over time. Another factor is the region's complex mix of natural and human-related mercury sources. Volcanic activity and hydrothermal vents release mercury naturally, while centuries of mining, industrial development, and energy production have added further pressures.

Figure 1. *Mercury hotspots in the Mediterranean*



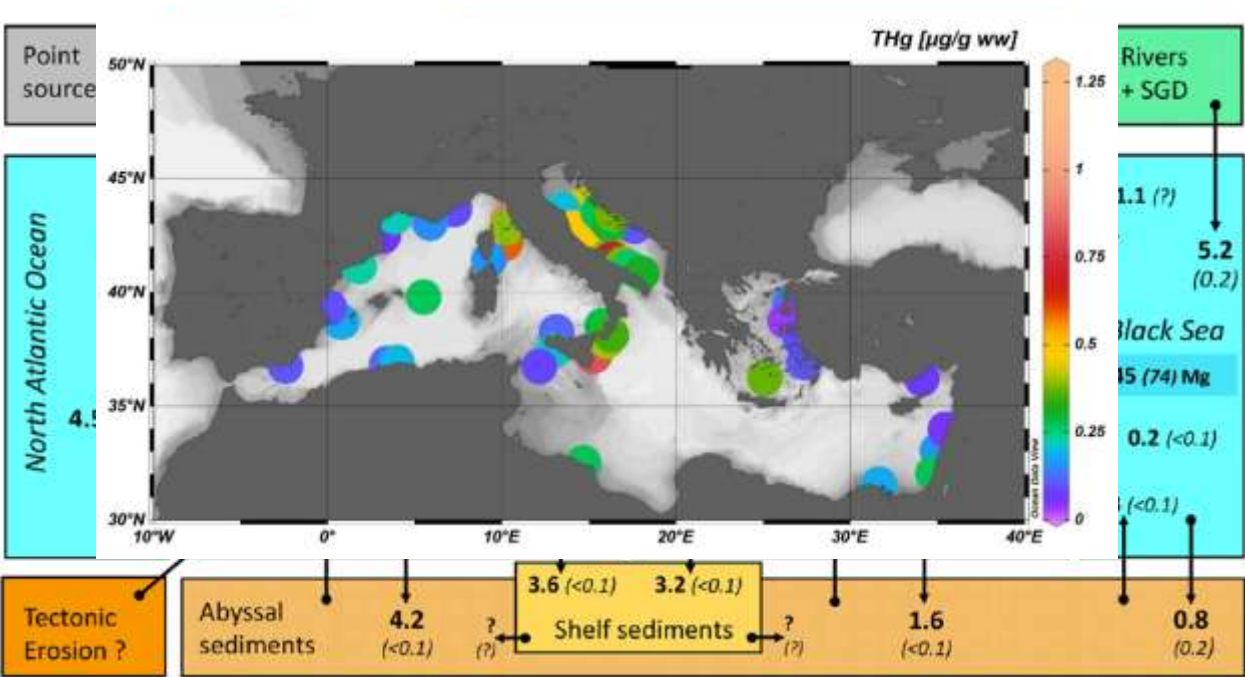
Figure 1 illustrates major Hg contamination hotspots across the Mediterranean region, together with their primary sources of pollution. The distribution highlights the so-called Mediterranean Hg belt, which includes areas affected by historical Hg mining, industrial discharges, and petrochemical activities. Notable sites include the Almadén mine (Spain), historically the world's largest source of mercury, and Idrija (Slovenia), where past mining activities contributed to Hg accumulation in the Gulf of Trieste and the Marano–Grado Lagoon. Additional industrial sources are located along the Rhône River delta and in northern Italy (Livorno, Piombino, Mestre, Ravenna), linked to chlor-alkali production and petrochemical industries. In southern Europe and North Africa, hotspots include Milazzo and Augusta (Sicily), Monteponi (Sardinia), and several coastal sites in Tunisia and Algeria (e.g., Bizerte, Gulf of Gabès, Skikda Bay), where mining, fertilizer production, and petrochemical activities have released substantial mercury loads. Eastern Mediterranean locations, such as Kaštela Bay, the Neretva River delta, Iskenderun Bay, and Haifa Bay, further reflect contamination from chemical industries, agriculture, metallurgy, and fossil fuel use. Collectively, the map emphasizes the widespread but heterogeneous distribution of mercury sources, with legacies of mining and industry continuing to shape Hg inputs to Mediterranean coastal and marine environments.

This comprehensive case study on Mediterranean region includes information on regional intergovernmental agreements supporting mercury monitoring and control in the region, available data in air, multi-model ensemble simulations, seawater, sediments, biota and human biomonitoring. The study summarizes the comprehensive reviews on multi-media monitoring data and their spatial and temporal trends in the region, factors affecting mercury levels and fate, and mass-balance within and to/from outside regions.

Cross-media integration of observed levels/trends

The study reports mass balances of mercury among media and regions in Mediterranean region as summarized in Figure 11 as shown below.

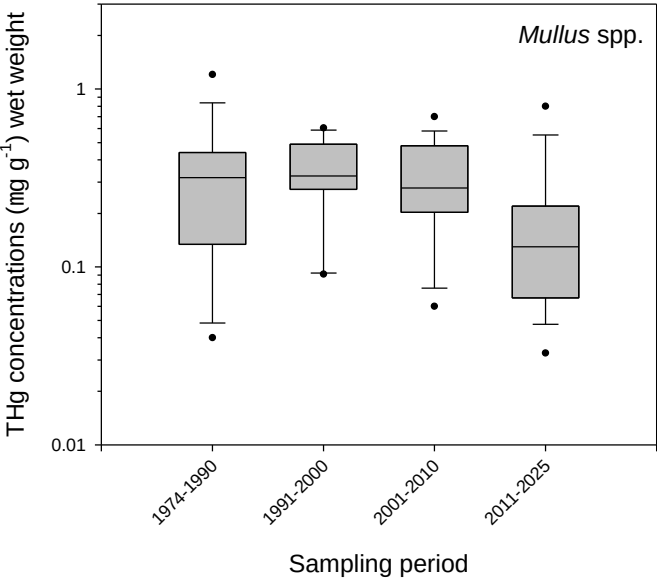
Figure 11. Mediterranean mercury mass balance (after Cossa et al, 2022)



It also report the spatial and temporal trends in biota, as examples in Figure 5 and 6 shown below.

Figure 5. Average THg concentrations (in µg/g wet weight) in fish *Mullus spp.* across the Mediterranean Sea. Data range from 1974 – 2025.

Figure 6. Box-and-whiskers plots (5, 10, 25, 50, 75, 90, and 95th percentile) of the average THg concentrations (in µg/g wet weight) in fish *Mullus spp.* from the Mediterranean Sea, pooled in four time periods.



Mercury levels in biota show apparent spatial distribution within the region, and the temporal trends from the period from 1974-1990 to 2011-2025 as shown in Figure 6.

Relevance to effectiveness evaluation

The Mediterranean is a responsive test bed for Article 22: short effective residence times and tightly coupled compartments allow policy signals to emerge across media. Independent budgets converge on a **net atmospheric source** and **net Atlantic export**, and the Gibraltar sections document **multi-decadal declines** consistent with global emissions controls. Because speciation is explicit (MeHg, DGM), we can link observed **biogeochemical reactivity** to outcomes that matter—**fish Hg** and **human exposure**. As local point sources wane (e.g., chlor-alkali, dental amalgam), **diet becomes the dominant exposure pathway**; thus, **targeted, species/portion-specific advisories** become the practical lever to protect vulnerable groups while preserving seafood benefits. Together, these features make the Mediterranean a **demonstration region** where cross-media indicators can track Convention effectiveness with credibility.

Gaps, limitations and recommendations

Current knowledge on mercury in the Mediterranean is constrained by several significant gaps and limitations. Geographic and trophic coverage remains incomplete, with especially limited data available from the eastern and southern Mediterranean. Time series are scarce, particularly for methylmercury (MeHg) and boundary exchange processes, making it difficult to establish long-term trends and variability. Coordinated human biomonitoring (HBM) efforts are limited across high-consumption communities, restricting the ability to assess exposure patterns in populations at greatest risk.

Submarine groundwater discharge and hydrothermal fluxes remain highly uncertain, yet they may represent important sources of mercury input. Speciation and microbial cycling are not routinely measured, even though these processes are central to mercury transformation and bioavailability. In addition, methodological differences across environmental media hinder cross-integration of datasets and complicate the establishment of coherent regional assessments.

ASGMP-1

Mercury Pollution from Artisanal Gold Mining in Madre de Dios, Peru: A Cross-Media Case Study

Description of the Case Study

Over the last 20 years, ASGM has deforested and degraded nearly 100,000 ha of high-biodiversity rainforest landscapes in the department of Madre de Dios, located in the southern Peruvian Amazon (Caballero Espejo et al., 2018), and created highly degraded landscapes that are pockmarked by thousands of hectares of mining ponds (Gerson et al., 2020). A 2018 study estimated that 181 tons of mercury are released to the region's waterways, soils, and air every year. As of this writing, Madre de Dios is considered the largest hotspot of ASGM activity and ASGM-related mercury pollution in Latin America (Cardo & Vargas, 2017).

This case study illustrates the causal link between mercury emissions from artisanal and small-scale gold mining (ASGM) activities—including amalgam burning in gold shops and mercury releases to soils and waters at mining sites—and elevated mercury levels in air, aquatic ecosystems, and terrestrial biodiversity. The objective is to demonstrate how ASGM contamination manifests across environmental media, with implications for both biodiversity and human exposure.

ASGM and control sites were identified based on the presence and distance of mining activities. Control sites were selected to have very similar features to the impacted sites, but without ASGM. The sites defined as ASGM sites were initially located in rural environments, such as mining ponds formed during mining activity and subsequently abandoned after the cessation of the activity. In urban areas, these sites included gold shops where amalgamated gold is burned, releasing mercury during the process.

The study was performed by The *Centro de Innovación Científica Amazonica* (CINCIA), which is a Peruvian non-profit scientific research centre, developed a framework to characterize the presence, magnitude, and spatial distribution of mercury pollution in and around ASGM sites, between 2017 and 2019. The study includes three topics:

Mercury in abandoned mining ponds: sediment and fish

Mercury around Mining ponds: Bird and bats

Mercury levels in air: Amalgam burning gold shops

Cross-media integration of observed levels/trends

Mining Ponds Assessment

Our results showed that with exception of the bottom sediments samples that didn't show any significant difference between ASGM and control sites (Figs.), all the biota matrix (fish muscle, bird feather and bat fur)

evaluated in and around mining ponds (impacted sites) and oxbows lakes (control sites), showed similar trend : *Mercury levels were consistently and significantly higher in biota from ASGM sites in comparison with control sites (Fig. 1, 2 and 3).* In birds, we reported the highest Hg level reported for bird feather in South America. We recorded in bats from ASGM sites the highest mercury levels in bat fur reported in the Neotropics. (Sayer et al 2023; Pisconte et al 2024; Portillo et al 2023).

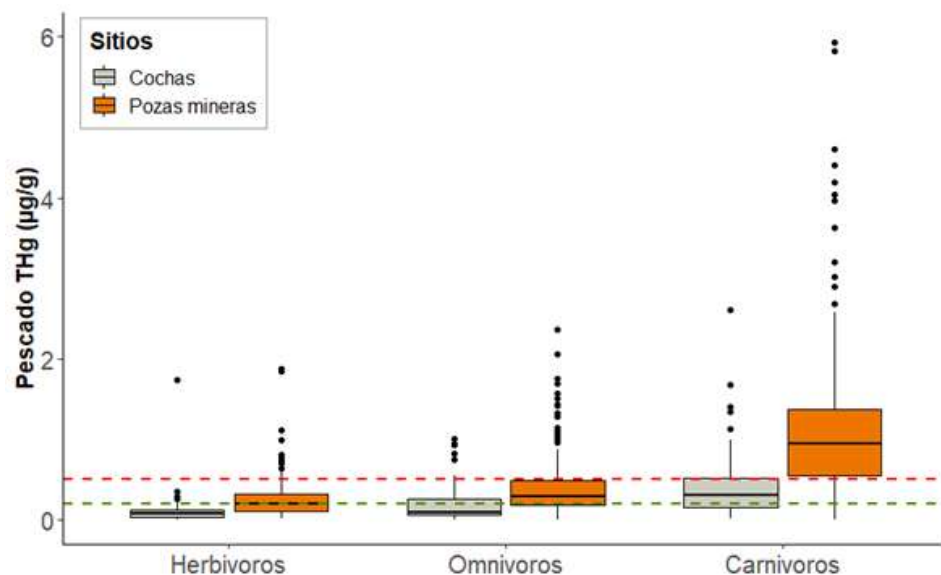


Fig. 1. Mercury concentrations in fish: ASGM ponds vs. oxbow lakes (controls).

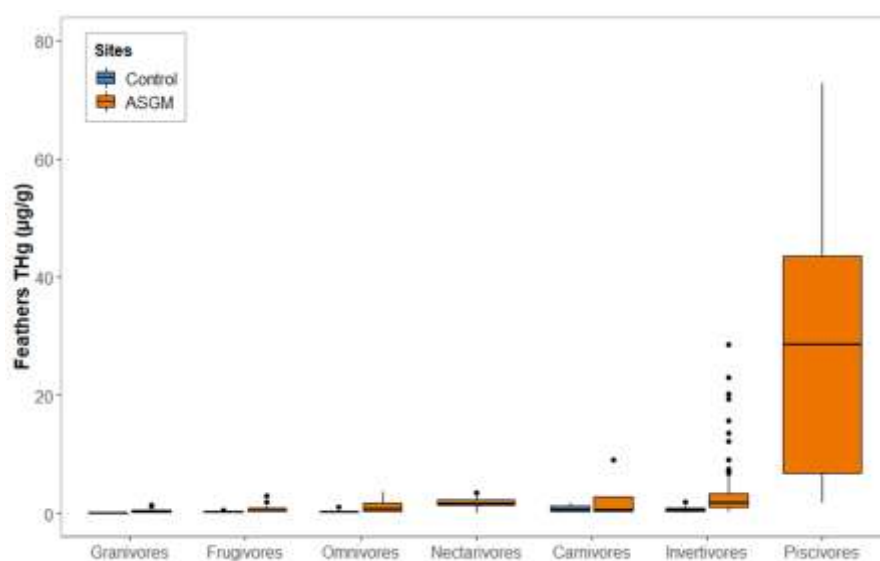


Fig. 2. Mercury concentrations in bird feathers: ASGM ponds vs. oxbow lakes (controls).

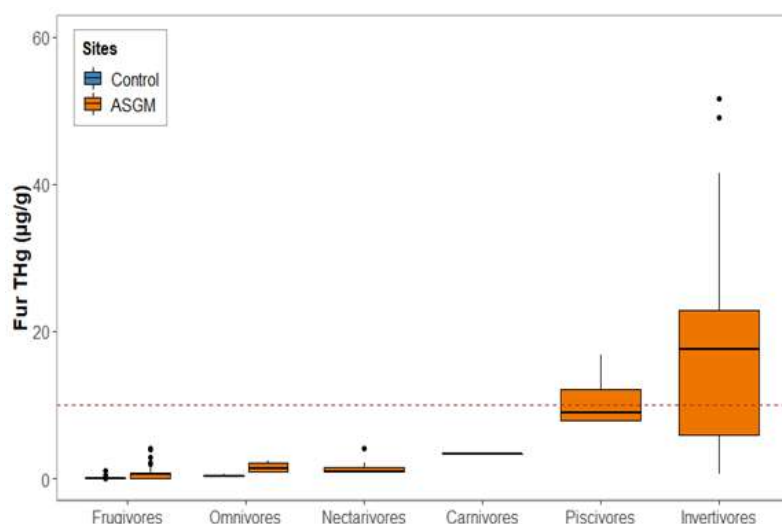


Fig. 3. Mercury concentrations in bat fur: ASGM ponds vs. oxbow lakes (controls).

- **Mercury emissions from gold shops (air assessment)**

Our data showed that there was a spatial variability of Hg emission (Gaseous elemental mercury GEM) in the air in Madre de Dios. Regional GEM concentrations were elevated (1.3 to 11 ng.m³) compared to background levels (< 1 ng.m³) and were even higher in areas near mining and gold shop activity * ~10 to >5000 ng m³). And applying an isotope mixing model, it was estimated that about 70% of the mercury emitted to the air is derived from mining activity, which shows a distinct isotopic signature (Szponar et al 2025).

Taken together, these findings provide strong evidence of cross-media contamination, where mercury emissions from ASGM are simultaneously detectable in air, aquatic food webs, and terrestrial wildlife. The isotopic signatures reinforce that local ASGM is the dominant source, linking emissions directly to observed bioaccumulation.

Relevance to effectiveness evaluation

This case study is one of the few that directly examines mercury contamination from ASGM across multiple environmental matrices. The data provide clear evidence of how mercury from ASGM is distributed throughout mining regions and demonstrate the value of using cross-media indicators to characterize contamination.

These findings are directly relevant to the Minamata Convention's effectiveness evaluation, particularly under Article 7 (ASGM). They show that, despite ongoing national and international commitments, ASGM in Madre de Dios continues to release large amounts of mercury that are traceable across environmental media.

Gaps, Limitations and Recommendations

While this case study provides clear evidence of cross-media mercury contamination from ASGM, several important gaps and limitations should be acknowledged. The monitoring period, conducted between 2017 and 2019, was relatively short and may not fully capture longer-term temporal trends.

Geographically, the study focused exclusively on Madre de Dios, which is the largest ASGM hotspot in Latin America but does not represent the full diversity of mining contexts across the Amazon Basin. Similarly, atmospheric monitoring at gold shops was limited in duration and scope, restricting the ability to generalize results to regional or national scales.

These limitations point to several recommendations. Future work should expand biomonitoring to include human exposure pathways and extend geographic coverage to other ASGM regions in Peru and neighboring countries. Standardizing protocols across media—air, aquatic, and terrestrial biota—would improve comparability across studies and strengthen regional assessments. Finally, the integration of isotope tracing into national mercury monitoring frameworks would provide a powerful tool to attribute sources more precisely and support enforcement of Minamata Convention commitments.

SWED-1**Integrated Analysis of Mercury in Sweden****Description of the Case Study**

Widespread mercury contamination in the biota of Fennoscandia was recognized already in the 1960s, during the same decade as the cause of the Minamata disease was becoming apparent in Japan. The first reports on mercury contamination were from birds (Westermarck, Odsjö and Johnels, 1975), and fish (Berglund, 1971). Local contamination from industry as well as the use of mercury in agriculture were the original concerns, and the first bans on the use of mercury in agriculture came in the latter half of the 1960s. The discovery of unsafe mercury levels in birds and fish hundreds of kilometres from any local source raised concerns about long-range atmospheric transport of nature to remote areas. One manifestation of this concern was the first international conference on mercury as a global pollutant being held in Gävle, Sweden (Lindquist et al., 1991). Further, already in the 1960s, studies conducted in Sweden revealed that mercury in fish was predominantly present as methylmercury (often exceeding 90% of total mercury, Westöö, 1966) and that this compound could form in the environment through biological methylation of inorganic mercury (Jensen and Jernelöv, 1969). These discoveries spurred further studies on mercury's biogeochemical and bioaccumulation processes, research that remains highly active and critically important today. The Swedish concerns about mercury also led to comprehensive monitoring programs for mercury in the environment, and research focused on to determine both the causes of contamination and the effectiveness of countermeasures (Ammar et al. 2024).

This integrated case study centres around the Swedish monitoring program for contaminants in freshwater and marine (Baltic Sea) biota comprising 62 fish Hg time series with the longest time series starting in the late 1960s. By complementing this with long-term records of national mercury emissions, air concentrations, wet deposition, longer term proxies of environmental mercury (tree rings, peat, lake sediments) and mercury in birds, this study addresses the central question for the first effectiveness assessment of the UNEP Minamata Convention: Can reductions in the release of Hg to the environment reduce Hg in the biota?

Cross-media integration of observed levels/trends**1. Mercury emissions and releases, including agriculture**

Swedish emissions of mercury to air have decreased by almost 75% since the 1990s (Naturvårdsverket, 2025). In Europe, large emission reductions occurred in the early 1990s after the large-scale political changes in the former Soviet Union block. These emission reductions are undocumented but led to significant decreases in air concentrations in Sweden (Iverfeldt et al., 1995). In the period 2005 to 2022, emissions of mercury to air from the EU countries continued to decline by 55% (EEA, 2025). The use of mercury in agriculture to protect seeds from fungus was recognized as a source of mercury in birds already in the 1950s. Sweden restricted this use already in 1966, and any use in agriculture was banned by 1990 (Westermarck, Odsjö and Johnels, 1975; Lindquist et al., 1991). Wood processing centres were also recognized as point sources of mercury contamination, since mercury was used in pulp and paper industries and in the production of chemicals in the chlor-alkali process. These uses were also restricted by the 1990s, and Sweden banned all use of mercury and mercury-containing products in 2009, with similar legislation in the other Nordic countries (Hylander and Meili, 2005, Kemi 2025).

Discharges from municipal wastewater treatment plants have also decreased in Sweden since the 1980s. For discharge to water, the releases have decreased with >90% since the early 1990s (29 kg or <0.05 µg/L in 2022), while concentrations for sewage sludge have decreased with >85% since 1987 (0.4 mg/kg dry matter in 2022) (SCB 2022, Naturvårdsverket, 2016).

2. Atmospheric Mercury Deposition

Long-term monitoring by the Swedish Environmental Protection Agency (Naturvårdsverket) shows that atmospheric Hg deposition in rural Sweden has declined by over 50% since the 1990s ([Naturvårdsverket, 2024](#)). This decline is attributed to international emission reductions following the large-scale political and industrial changes in Europe in the early 1990s and continuing with policy driven agreements such as CLRTAP and the EU air quality directives. On a global scale, emission reductions are not as evident and background deposition persists and will continue to constrain the influence of national and regional emission reductions. In addition, remobilization from soil and sediments continues to impact surface waters.

3. Mercury in surface waters

Trends in Hg concentration were evaluated between 2000 and 2010 from 19 watercourses in the Swedish monitoring data (Eklöf et al 2012). Data show no significant trend for most sites (n = 18 of 19), only one site had increasing trends of Hg concentrations. A decade later, trend analyses were conducted for both Swedish

monitoring data and long-term data measure within the International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems (ICP IM). Eklöf et al. (2024) show a decline by average 1.5% in total Hg concentrations in surface waters from 2000 to 2020 in three of the sites. Using generalized additive mixed models (GAMM) it was found that trends were not monotonic. In the first five years (2000–2005) there was a general decline in Hg concentrations in many sites, while in the last five years (2015–2020) increasing trends outnumbered the decreasing trends. These variations are most likely due to other factors that may influence the mobility of mercury such as recovery from acidification, reduced atmospheric deposition, local land use, organic matter transport, and hydrological regimes.

4. Mercury in environmental archives (tree rings, peat and lake sediment).

Environmental archives in the form of tree rings (Peng et al., 2024), tree litter fall (Zhang et al., 2025), and peat profiles (Li et al., 2023) have proven successful at tracking atmospheric mercury concentrations. These archives track both the increase in atmospheric Hg over the Nordic countries during much of the 1900s, and the decline of regional atmospheric Hg levels after they culminated ca 1980, followed by stabilization at a lower level after ca 2000. A study on sediment from a lake in Northern Sweden found steady Hg levels from the 1950s to the end of the record in 2007 (Rydberg et al. 2008), thus deviating from the other terrestrial archives. More extensive work on sediment records is found in Baltic offshore waters near Sweden. These cores show a peak in Hg concentration in the ~1970s followed by declining concentrations to ~2010 (Leipe, 2013). In the last two decades (2003 to 2020), 16 monitoring stations show no clear declining concentrations in offshore sediment along the Swedish coast (Josefsson, 2022). The different environmental archives thus show similar long term trends while the trends deviate in recent decades (decreasing or no trend). Stable mercury isotope measurements of surface sediments (0–2 cm; $n = 29$) from the Baltic Sea and Kattegat indicate that the pool of mercury from recent inputs reflects a variable mixture of sources, dominated by a terrestrial and an industrial end-member, with a smaller contribution from direct atmospheric deposition (Bouchet et al., 2023). Differences in source composition are likely an important factor contributing to the varying recovery rates observed among locations. In general, understanding the source composition of mercury is important for evaluating and comparing time trends across locations.

5. Mercury in Fish

Diverse temporal trends in fish Hg have been observed since 1965. Åkerblom et al. (2014) looked into >40000 Hg observations from a mix of fish species collected from >2800 Swedish freshwater sites over 50 years. An overall analysis of the normalized dataset showed that the temporal development did not follow any consistent decreases or regional patterns. However, restricting the dataset to locations with long term monitoring, they found an overall decline of ~20 % between the two periods 1965–1990 and 1991–2012. Braaten et al. (2019; 2020) examined temporal trends in Hg concentrations from perch and pike in a pan-Nordic study on freshwater lakes. They found that Hg levels in fish across the Nordic countries had decreased with 20% for pike, 50% for perch between 1970 and 2015 and linked it to reduced atmospheric deposition and changes in water chemistry (e.g., sulphate and DOC). However, they suggested biota Hg recovery lags behind atmospheric decreases due to the remobilization of legacy Hg and continued in-situ methylation. Swedish monitoring of perch Hg concentrations from 27 lakes (reference lakes) across Sweden showed that during the period 2011–2020, 19% of time series were decreasing significantly (3–6%/y, $p < 0.05$), while the remaining lakes showed no significant trend (Faxneld and Soerensen 2022). For the Swedish coastal environment, 30% of 30 fish (herring, perch, cod, eelpout) stations showed significant decreases from the monitoring start year (1972 to 2008) to 2020 while 10% of the stations showed significant increases (Soerensen and Faxneld 2022). For the 2011–2020 period, an equal occurrence (20%) of significantly increasing and decreasing trends were found at the 30 stations. Temporal trends from the Swedish monitoring are shown in Figure 2. Concentration of Hg in monitored perch still exceeds the EU environmental threshold (20 ng/g muscle ww, EQS secondary poisoning) for all lakes and most Baltic Sea fish (Faxneld and Soerensen 2022; Soerensen and Faxneld 2022).

6. Mercury in Birds

Two decadal time series exist for Hg in Swedish birds, one based on white-tailed sea eagle feathers from the area around the central Swedish east coast (1966–2011; Sun et al. 2019) and one based on guillemot eggs from Gotland, a Swedish island in the Baltic Sea (1968–2023; Ammar et al. 2024). Both these time series start around the time when early work by Berg et al. (1966) and Johnels et al. (1979) showed peak Hg concentrations in white-tailed sea eagle, marsh harrier and eagle owls likely due, for a large part, to use of Hg as seed dressing, a practice that was forbidden from 1966. For the white-tailed sea eagle, Sun et al. (2019) found a 70% decrease between 1967 and 2011. For the guillemot eggs, a decrease of 1.9%/y (resulting in a similar total decrease of 70%) was found between 1968 and 2023, while the decrease during the period 2011–2020 was smaller (0.47%/y; Soerensen et al. 2025).

Synthesis and outlook

Figure 1 shows the long term trends of three historic environmental archives from Sweden or the nearby Baltic Sea. While there are minor differences, all three matrices show a similar pattern with low concentrations until the early 1900s where concentrations begin to increase, and peak within a short period between 1960-1975. After the peak, Hg concentrations start to decline and only the tree ring record shows a potential levelling off in the last years of the time series. These long term records present a uniform picture of the anthropogenic influence in Sweden since industrialization.

Since the peak in the 1960s, more biota data have become available to supplement the decreasing trends in the few long term records, and since the 1990s, also data from more volatile compartments such as air and water has become available (Figure 2). These shorter term records do not present a picture as uniform as the long-term environmental archives. Air concentrations, wet deposition, releases from wastewater treatment plants and concentrations in Swedish streams and rivers are all decreasing ~40-60% during the period but with different consistency and variability. These decreases are on a scale similar to the decrease seen during the period from the 1970s to the present day in the historic archives (Figure 1) and the time series on guillemot (bird) eggs (Figure 2). However, we also see that both Swedish freshwater perch and Baltic Sea herring do not follow the same tendency as seen for the other matrices. Here the records show a limited decline in the freshwater environment (<20%, 1995-2023) and no significant decline in the Baltic Sea (marine) environment (Figure 2). These long term fish trends, based on Swedish monitoring data, aligns with the result from the same monitoring data during the recent 2011-2020 period and the results on Swedish freshwater fish from Åkerblom et al. (2014) of a 20% overall decrease between the periods 1965–1990 and 1991-2012. The study by Braaten et al. (2019) shows a different trend with a 50% pan-Nordic decrease in perch (see discussion in section 5 Mercury in fish).

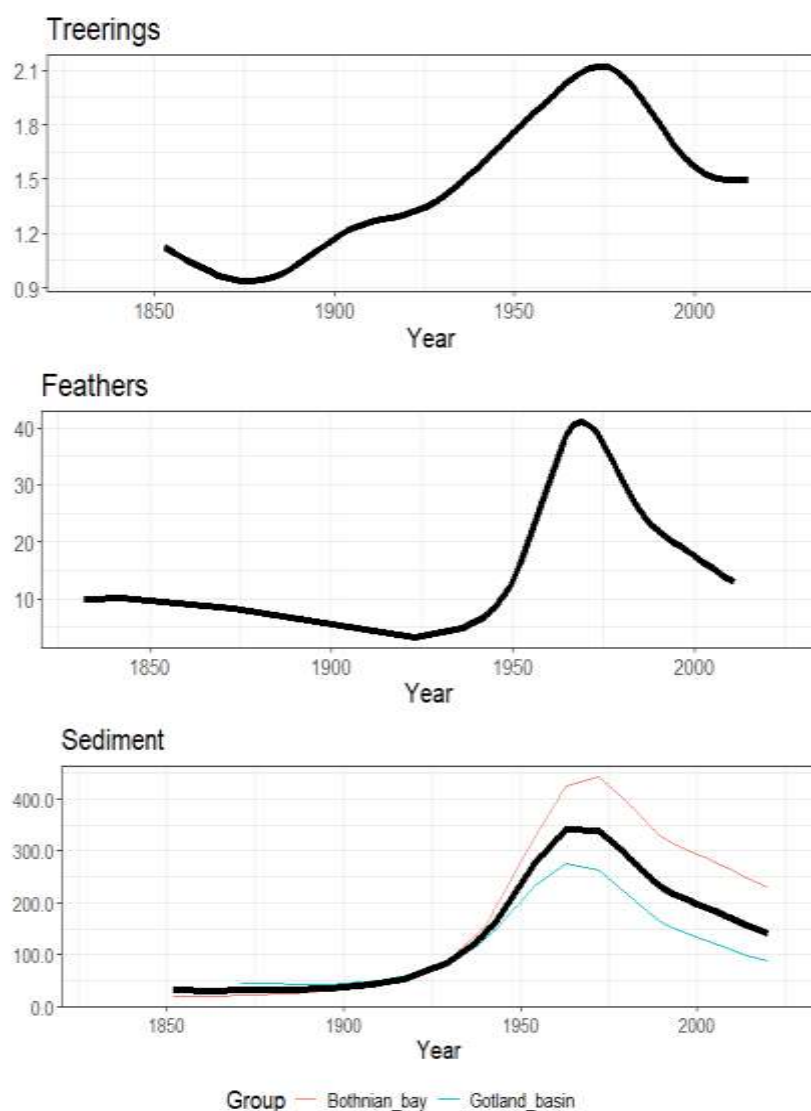


Figure 1. Long-term trend of Swedish environmental mercury observations since 1850 or earlier, collected from tree rings (top panel, Zhang et al., 2025), white tailed sea eagle feathers (middle panel, ug/g), and sediment

(bottom panel, ug/kg). Generalised additive models are used for single series (Hastie and Tibshirani, 1986; Wood, 2017) and hierarchical generalised additive models for plots with several series (Pedersen et al., 2019). Data sources: The trend for bird (white tailed sea eagle) feather is based on two datasets, Berg et al. (1966) covering the period 1832-1965 and Sun et al. (2019; data from figure 2) covering the period 1967-2011; Baltic Sea sediment data is from dated cores in the Bothnian Bay and Gotland Basin presented in Leipe et al. (2013; data from figure 4) covering the period 1871-2009 and surface sediments from Josefsson et al. (2022) covering approximately the same location during the period 2003-2020.

Relevance to effectiveness evaluation

High relevance: Reductions in Hg emissions to the environment are directly associated with decreased Hg burdens in biota. In certain cases—such as the prohibition of Hg use in agriculture (where the link between the Hg source and exposure to biota was direct and short)—measurable effects have been observed within a decade: The response can take much longer and be highly variable in the case of fish higher in the food web where the link between the Hg source and biota exposure is indirect and modulated by multiple processes.

Gaps and Limitations

Extensive as the work in Sweden has been to follow the mercury levels in biota and their causes, as well as the effectiveness of countermeasures, much more remains to be done to define causal linkages and rates. Further studies that cross traditional system boundaries between different media (air, vegetation, soils, water and biota) are of particular value in this regard (Bishop and Eklöf, 2022).

The importance of local legacy point sources (such as sediment contaminated from previous pulp and paper industrial activities) as well as the influence of changing land-use (e.g. forestry) and climate change on the recovery rates remain uncertain. The spread of Hg from different sources and its contribution to the Hg load to aquatic ecosystems and food chains in surrounding environments are challenging to quantify. Environmental change processes can mobilise Hg from soils and influence Hg biogeochemical processes which may cause confounding effects on the recovery.

General conclusions

Sweden and the Baltic Sea region demonstrate that there is a long-term linkage between reduced mercury emissions, exposure and levels in biota. This can provide a model for assessing Hg policy effectiveness, but continued monitoring and mechanistic studies are needed. The legacy Hg pool in soils and sediments, combined with climate-driven shifts in DOC and hydrology, poses challenges for achieving further reductions in aquatic MeHg exposure.

To allow future assessment of trends of Hg in different media, future monitoring activities should be harmonised and designed for comparison and integrated assessment. This would include parameters such as location of monitoring sites, time period and frequency of monitoring as well as methodology for e.g. field sampling and chemical analyses and statistical treatment.

AUST-1

Integrated Analysis of Mercury in the Derwent Estuary, Australia

Description of the Case Study

The Derwent Estuary covers an area of about 200 km² and is an integral part of Tasmania's natural, cultural and economic heritage (Macleod & Coughanowr, 2019) in Australia. It supports diverse urban, industrial, and recreational uses (Macleod & Coughanowr, 2019). Historically, the estuary has been impacted by the operations of a large zinc smelter (in operation since 1917) and a paper mill (established in 1941). Although modern environmental management practices have significantly improved, the legacy of past industrial activity continues to affect the estuary and is likely to persist for decades.

The Derwent Estuary Program, initiated in 1999, plays a central role in advancing the understanding and management of these issues. Cooperative monitoring arrangements with the State Government, industries, local governments and the scientific community have generated a wealth of information on water and sediment quality, seafood safety and estuarine habitats and species (DEP, 2020, 2021). Metal levels in the Derwent sediments, water column, fish and some invertebrates have been monitored for several decades (Bloom & Ayling, 1977; Macleod & Coughanowr, 2019; Ratkowsky et al., 1975; Whitehead, 2013) (Figures 1a, b and c). Additional data on mercury emissions and releases is available through both the Australia's National Pollutant Inventory and industry reports (NPI, 2025; Nyrrstar, 2025).

Cross-media integration of observed levels/trends

Modern control measures

inc metal production in Hobart involves several key stages: ore concentration (crushing and flotation), roasting, leaching, purification, and electrolysis. The zinc ore, primarily a sulfide mineral, is roasted to convert zinc sulfide to zinc oxide, releasing sulfur dioxide (SO₂) and other impurities, including mercury, in the process.

In Tasmania, emissions from the Hobart zinc smelter are regulated under the *Environmental Management and Pollution Control Act (1994)* through an environmental permit issued by Environment Protection Agency of Tasmania (EPA Tasmania, 2024). The facility implements a range of control measures to reduce emissions of both legacy and current contaminants to the Derwent Estuary.

The implementation of modern control measures began in the 1980s, marking a significant shift in environmental management practices. To control air emissions, the zinc smelter uses electrostatic mist precipitators, wet scrubbers, chemical absorption systems, and filters to remove particulate matter and pollutants from the gas stream before atmospheric release (Nyrstar, 2025). These systems are generally effective at capturing particle-bound mercury and some oxidised divalent mercury species, but they are less effective at removing elemental mercury vapour (UNEP, 2021). To address this, the gas scrubbing process at the Hobart zinc smelter also includes a specific mercury removal step, utilising the Boliden-Norzink process which converts elemental mercury into mercury(I) chloride (calomel) via reaction with mercury(II) chloride. The calomel is then treated with sodium hypochlorite to regenerate the scrubbing solution. Minor amounts of excess calomel are stored onsite for reuse when required, or if necessary, stabilised and disposed of with other site mercury bearing wastes.

Mercury containing wastewater is treated via a dedicated mercury removal plant. The process generates mercury filter cake, which is currently managed through chemical stabilisation and disposal at an approved facility (Nyrstar, 2025).

The refinery also generates significant quantities of process wastewater, stormwater and recovered groundwater. These are treated using lime neutralisation, flocculation, reverse osmosis and dilution before being discharged to the Derwent River (Nyrstar, 2025).

Emission and releases

Mercury and its compounds are reported under the Australian National Pollutant Inventory (NPI, 2025). Environmental monitoring indicates that total mercury is primarily released to surface waters, with smaller quantities emitted to the atmosphere (Figure 2). Historically, atmospheric mercury emissions from the smelter, in the absence of effective control measures, were likely a significant source of mercury contamination in the Derwent Estuary.

Between 2007 and 2022, annual mercury emissions to air decreased significantly from 21.3 kg to 1.52 kg reflecting the effectiveness of multiple control measures implemented during this period (Figure 2).

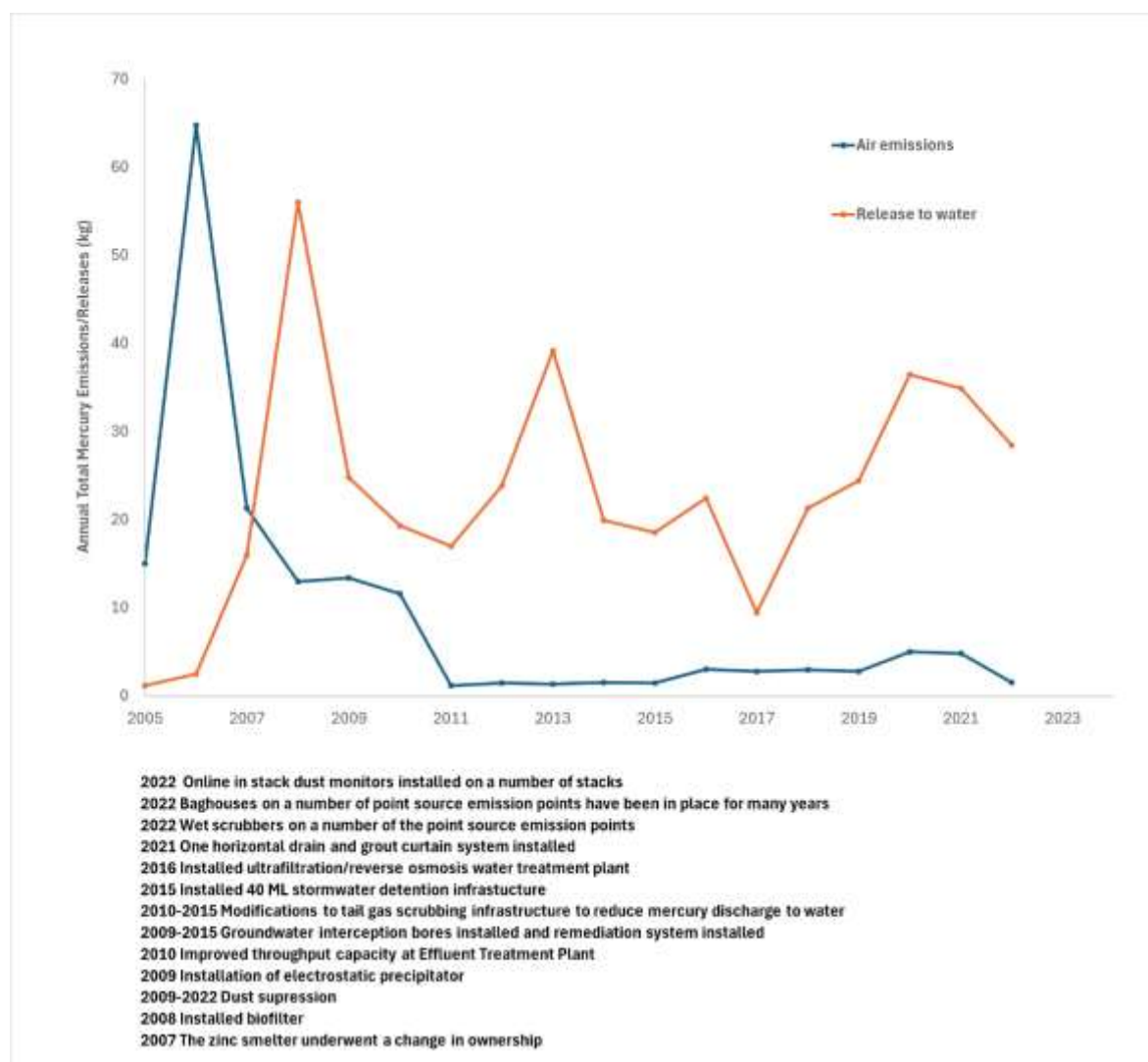


Figure 2. Annual total mercury emissions to the air and releases to the surface water at the Hobart Zinc Smelter. Total air emissions are calculated with a combination of direct measurements and [emission factors](#). Mercury emissions and releases from point sources are based on direct measurements. A monitoring program at the facility is conducted to quantify emissions/releases from multiple point emission/discharge points to determine compliance with Environmental Licence and the National Pollutant Inventory Reporting requirements. Stack monitoring is conducted by an independent NATA accredited organisation. Additionally, two daily 24-hour composite samples taken from discharge outfall (which receives flows from the effluent treatment plant and tail gas scrubber) are analysed for pH, iron, sulphate, copper, cadmium, mercury, lead and zinc and the flow rate is measured for reporting purposes. Sources: [Australia's National Pollutant Inventory data](#) and [Nyrstar Hobart Triennial Public Report 2022-2024](#).

Mercury observations in ambient water and sediments

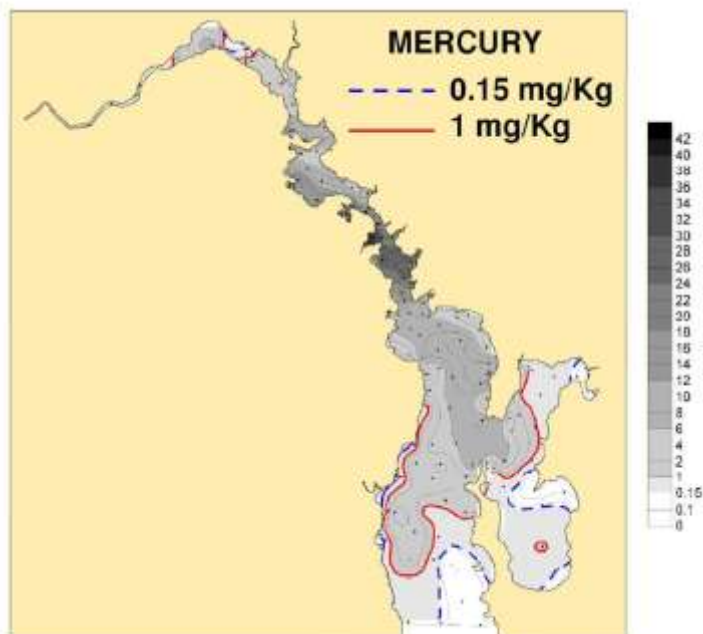


Figure 3. Total Mercury (Hg) distribution in surface sediments from the Derwent Estuary 2011 survey (Whitehead, 2013).

Mercury observations in invertebrates

Deployed oysters at nine impacted sites in the middle estuary accumulated higher concentrations of mercury than those deployed to the control site (Mickey's Bay) and showed the highest loads from sites nearest to the zinc smelter (Nyrstar Wharf) (DEP, 2020, 2021) between 2014 and 2024 (Figure 5a).

Wild oysters and mussels collected in the middle (near the zinc smelter) and lower zones of the Derwent Estuary showed higher concentrations of mercury than those from sites furthest from zinc smelting activities (the D'Entrecasteaux Channel and Mickey's Bay) (Figures 5b and c). Within the estuary, the highest concentrations were typically recorded in the middle estuary and Ralphs Bay, whilst the western shore exhibited the lowest levels. These spatial patterns likely reflect the influence of estuarine circulation, ambient water metal concentrations, proximity to contaminant sources, estuarine processes and longer water and sediment retention times at Ralphs Bay. Sites in the middle estuary are closest to the zinc smelter, whereas marine-influenced regions, such as the western shore, are generally exposed to lower concentrations of metals in the water column.

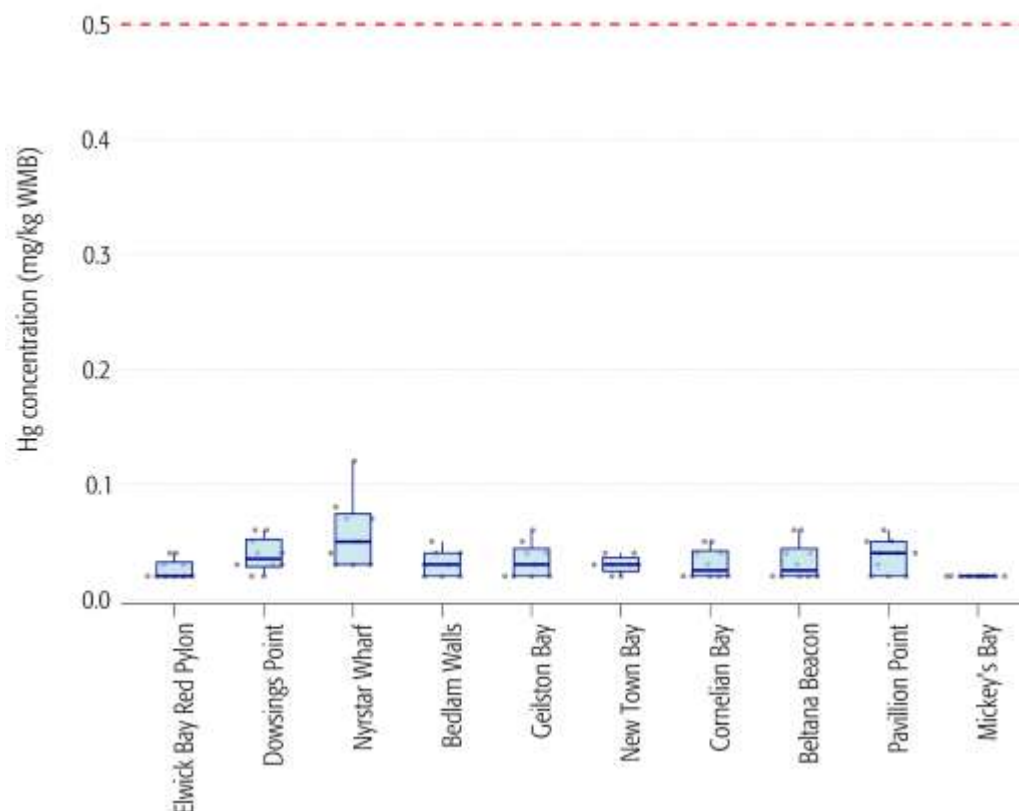


Figure 5a. Total mercury concentrations in surface-deployed Pacific Oysters' soft tissues (2014-2024). Box plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at nine impacted sampling sites in the middle estuary and a control site in Mickey's Bay. Grey dots show the raw data. Horizontal red line represents Food Standards Australia and New Zealand national guideline for mercury (Maximum Permitted Limit; 0.5 mg kg⁻¹). Mercury concentrations are expressed in mg/kg wet matter basis (WMB). Outliers were removed using the interquartile range (IQR) method. Values that were more than 1.5 times the IQR below the first quartile or above the third quartile were excluded.

Mercury observations in fish

Environmental organisms at higher trophic levels are primarily exposed to organic mercury through dietary sources. Sand flathead (*Platycephalus bassensis*) being selected as a bio-indicator of metal contamination because of their high site fidelity, abundance and regular consumption by humans (Jones et al., 2013; Ratkowsky et al., 1975). Sand flathead is a benthic carnivore, most likely exposed to mercury through dietary intake of epibenthic fauna such as crabs, which in turn feed at the sediment surface where methyl mercury can be produced (Jones, 2013). Sand flathead also shows high site fidelity and a relatively small home range (~5-10 km) with limited migration downstream during the spring/summer, presumably for spawning purposes (Tracey et al., 2020).

The control region (Mickey's Bay) mercury concentrations in fish (0.20 ± 0.02 mg kg⁻¹) were significantly lower than mercury concentrations in fish from within the Derwent Estuary (0.49 ± 0.01 mg kg⁻¹) between 1975 and 2011 (Jones et al., 2013).

In 2025, the Derwent Estuary Program scientists plotted current mercury concentrations in fish (Figure 5d). Mercury concentrations in fish from the control site (Mickey's Bay) were lower than those measured in fish collected from within the Derwent Estuary during 2014–2024. The observed spatial variability in mercury concentrations across impacted sites is likely influenced by differences in fish growth rates (Jones, 2013) and potentially greater mercury bioavailability and longer water and sediment retention times in Ralphs Bay (Ralphs Bay Spit and Richardson Beach) (Jones et al., 2014).

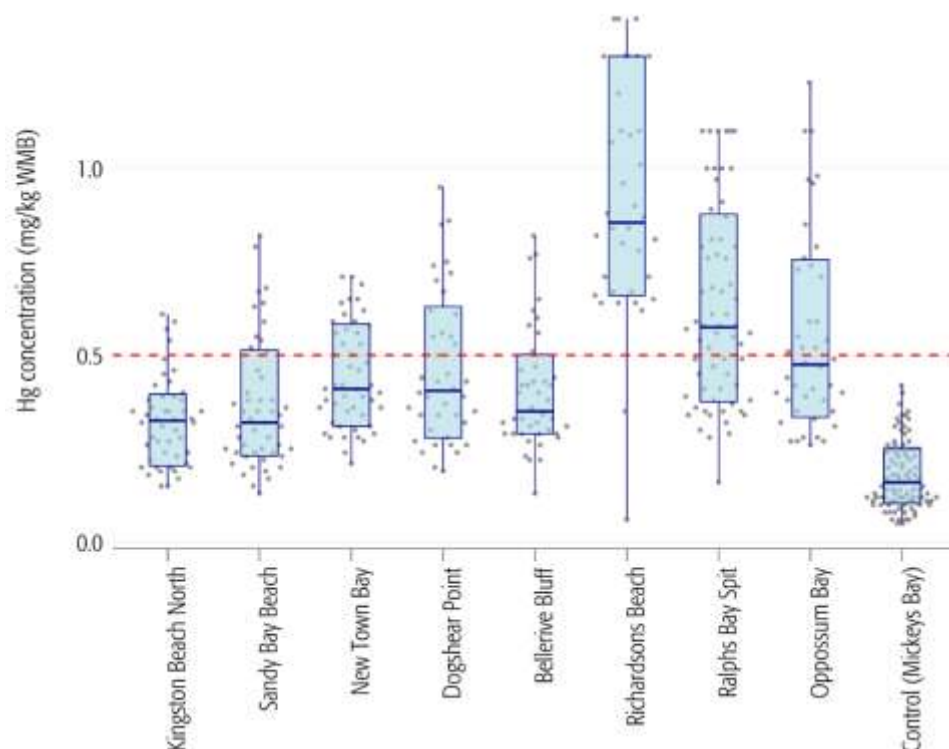


Figure 5d. Total mercury concentrations in fish (flathead) muscle tissues (2014-2024). Box plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at eight impacted sampling sites and a control site in Mickey's Bay. Grey dots show the raw data. Horizontal red line represents Food Standards Australia and New Zealand national guideline for mercury (Maximum Permitted Limit; 0.5 mg kg⁻¹). Mercury concentrations are expressed in mg/kg wet matter basis (WMB). Outliers were removed using the interquartile range (IQR) method. Values that were more than 1.5 times the IQR below the first quartile or above the third quartile were excluded. 2018 data points were removed because the skin was left on and data was erroneous. Mercury concentrations were not length-standardised.

Key Findings

The Derwent Estuary case highlights complex and persistent mercury contamination issues. Modern control measures have substantially reduced mercury emissions from industrial activities over recent decades. However, elevated mercury concentrations in fish and invertebrates at impacted sites, compared to control sites, indicate that historical contamination remains a major source of exposure.

Metal concentrations in all biota were consistently lower at the control site (Mickey's Bay) compared to impacted sites within the Derwent Estuary. Mercury spatial patterns in biota likely reflect the influence of proximity to contaminant sources, estuarine circulation, and estuarine processes. For example, elevated mercury concentrations in biota from Ralphs Bay, despite its greater distance from the zinc smelter, suggest additional environmental factors influencing mercury bioavailability and bioaccumulation in this region. There were some differences in temporal trends between deployed oysters and resident benthic fish, likely reflecting distinct exposure pathways and bioaccumulation mechanisms. Despite reductions in total anthropogenic mercury inputs over the past two decades, no significant temporal decline in mercury concentrations in fish has been observed between 2002 and 2024. In comparison, mercury concentrations in deployed oysters showed either no significant changes or increased annually by 1.6% to 3.4% over 2004-2024. Although these concentrations in deployed oysters remain well below thresholds of concern for human health, further investigation and predictive modelling are required to better understand the processes driving these observed changes.

Overall, the results suggest that significant reductions in anthropogenic mercury inputs achieved to date have not translated into corresponding decreases in mercury concentrations in fish and oysters. This highlights the complexity of mercury cycling processes and interactions within the Derwent Estuary, as well as the need for long-term monitoring and targeted management strategies.

Relevance to effectiveness evaluation

This case study provides a rare and detailed examination of mercury cycling within an estuarine environment impacted by modern and historical zinc smelting activities. The site's current environmental permit sets discharge and emission limits for mercury and mandates ongoing monitoring of estuarine water, sediment, and biota. This enables timely and effective responses to any increases in environmental mercury exposure.

Over the past two decades, modern control measures have effectively reduced mercury emissions. However, elevated mercury concentrations in fish and invertebrates at impacted sites suggest that historical contamination, resulting from the absence of earlier regulatory controls, remains a significant source of exposure.

These findings are directly relevant to the effectiveness evaluation under the Minamata Convention, particularly Articles 8 and 9, which address emissions and releases. The metal production and processing sector is a large anthropogenic source of mercury emissions and releases globally, and its contribution is likely to grow (UNEP, 2021). To mitigate associated mercury risks, the adoption of best available techniques (BAT) and best environmental practices (BEP) is essential to help minimise mercury emissions and releases. Without appropriate control measures, elevated mercury levels in impacted environment and biota are likely to persist for multiple decades, posing potential ecological and human health risks.

Gaps and Limitations

First, sediment coring could offer valuable insights into mercury deposition rates and help assess the effectiveness of mercury reduction measures implemented over last few decades.

Second, key factors influencing methylation potential and bioaccumulation remain poorly understood. A strong correlation between total mercury and selenium in both sediments and fish suggests that selenium may play a significant role in modulating the ecological impacts of mercury contamination (Jones, 2013; Jones et al., 2014). This highlights the importance of including selenium dynamics in future scientific assessments of mercury methylation and bioaccumulation.

Third, integrating isotope tracing within the Derwent Estuary could yield critical information on mercury sources, transport pathways, speciation and bioavailability, thereby enhancing our understanding of mercury cycling in estuarine systems.

Finally, predictive models that link changes in anthropogenic inputs and environmental factors to mercury species accumulation in the food web are needed to evaluate the effectiveness of pollution control measures. Such models would support evidence-based decision-making and guide future remediation and management efforts.

10.1.4 How have observed trends changed over the available record and how do those changes compare to the timeline of the Convention?

No information is available on this question.

10.1.5 How do temporal trends of Hg levels observed in species that display different degrees of site fidelity (fish, invertebrates, plants, mammals, birds) vary with emissions/release data in impacted areas by point sources?

Information Used (including applicable indigenous and traditional knowledge)

There is no straightforward information to answer this question.

A case study. IATT-1 “Integrated Analysis Study of Mercury in Tropical Tuna Species” provided by Rober Mason, et al., United States, introduced trends in tropical tuna species in the global ocean.

We show the outcomes of the case study as an example of how this question could be answered in the context of the case study.

IATT-1

Integrated Analysis Study of Mercury in Tropical Tuna Species

How do temporal trends of Hg levels observed in species that display different degrees of site fidelity (fish, invetebrates, plants, mammals, birds) vary with emissions/release data in impacted areas by point sources?

Case study Focus: Tropical tuna species in the global ocean

Information used. The focus of the study is tropical tuna species – skipjack (*Katsuwonus pelamis*), yellowfin (*Thunnus albacares*) and bigeye (*T. obesus*) globally but with a focus on the Pacific Ocean.

Monitoring and modeling information on global, regional and local scales were used in the study.

Methodology. Integrated analysis of temporal trends that will aim to investigate the factors that influence MeHg levels in seawater food web and how that impacts the MeHg concentration in tuna, and potential human exposure.

Answer/response. The study analyzed Hg levels in tropical tuna species and possible impact of site fidelity to the levels and trends.

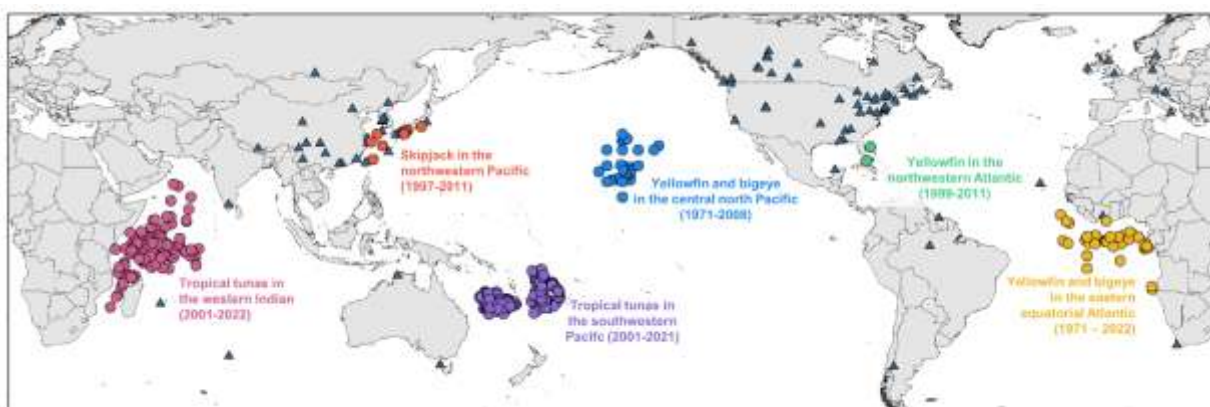


Figure 1. Spatial distribution of tropical tunas analyzed for Hg (colored circles) and atmospheric Hg level observation sites (blue triangles), adapted from Médiéu et al. (2024).

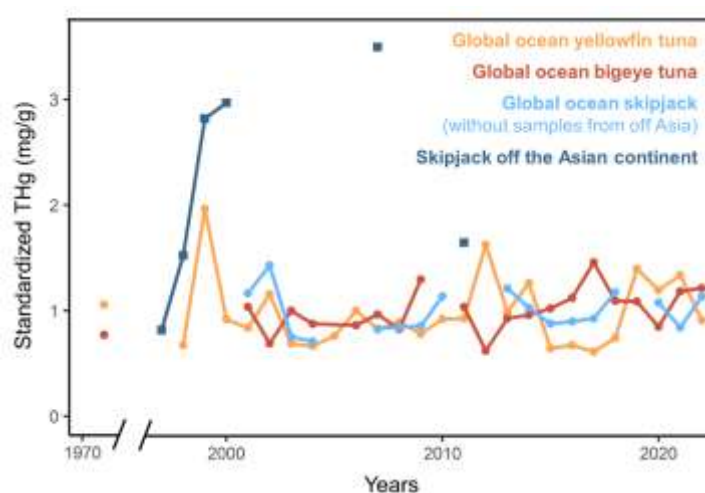


Figure 2. Temporal variability of standardized Hg concentrations (mg/g) in tropical tunas: yellowfin (orange), bigeye (red), and skipjack (blue), adapted from Médiéu et al. (2024). The colored dots represent average annual concentrations measured in the global ocean, except in the northwestern Pacific, off Asia. In that area, average annual concentrations are represented by dark blue squares.

Figure 1 shows spatial distribution tropical tuna analyzed in the study, and Figure 2 shows temporal variability of standardized Hg concentration categorized by tuna species and selected regions. The analysis of the tuna data suggest that in regions of significant regional anthropogenic inputs such as the northwest Pacific Ocean close to Asia that the impact on fish concentrations are evident and can be demonstrated but for many open ocean

regions more removed from point source inputs that a changing anthropogenic signal can be masked by other factors such as mixing of ocean waters and Hg transport with ocean currents.

10.2 Process Understanding and Model Evaluation

10.2.1 Organization of chapter 10.2

The organization of chapter 10.2 is different from chapter 10.1 and 10.4. The questions in chapter 10.2 is directly directed to the outcomes of MCHgMAP project described below.

10.2.2 What data are available?

The Multi-Compartment Mercury Modelling and Analysis Project (MCHgMAP) is an international scientific initiative designed to inform the Convention's effectiveness evaluation through multi-model assessment and attribution of geospatial variation and temporal trends in mercury pollution. MCHgMAP combines atmospheric, ocean, terrestrial, and multi-media mass balance models, and promotes collaboration between emissions reporting, monitoring, and modelling communities to improve data and inform global policy decisions. A comprehensive description of the multi-model experiment programme was recently published in the white paper (<https://gmd.copernicus.org/articles/18/2747/2025/gmd-18-2747-2025.html>).

Given the limited geographical coverage of existing measurement networks, which are globally uneven and mostly restricted to continental regions, model simulations can provide a more comprehensive picture of global Hg level distribution. Multi-model simulations can be used to develop spatial patterns of Hg concentrations and deposition, track their changes over the period 2010–2020, and explain these in terms of the contributions from different emission sources (anthropogenic, geogenic, secondary) and the influence of environmental drivers on Hg processes.

The project's multi-model ensemble consists of eight contemporary models of different types, operated by various institutions (Table 1).

Table 1. Multi-model ensemble of the project

<i>Model</i>	<i>Institution</i>
Atmospheric models	
GEM-MACH-Hg	Environment and Climate Change Canada (Canada)
GEOS-Chem	Massachusetts Institute of Technology (USA)
WACCM	Institute of Physical Chemistry Blas Cabrera (Spain)
GLEMOS ⁽¹⁾	MSC-E, Jožef Stefan Institute (Slovenia)
Ocean models	
ICON-MERCY	HEREON (Germany)
MITgcm	Tulane University (USA)
Terrestrial model	

2D air-land Hg exchange model Lamar University (USA),
Institute of Geochemistry, CAS (China)

Multi-media mass balance model

GBBM University Grenoble Alpes, CNRS (France),
Harvard University (USA)

⁽¹⁾ Did not participate in the current round of the model assessment.

The simulations covered the period 2010–2020 and included several emission scenarios designed to analyse and attribute historical Hg trends to changes in primary anthropogenic, geogenic, and secondary emissions, as well as other environmental drivers (Table 2).

Table 2. *Simulated emission scenarios*

<i>Scenario</i>	<i>Description</i>	<i>Purpose</i>
BASE	Full time series of anthropogenic emissions (EDGAR v8.1), geogenic, and secondary emissions	Reproduce geospatial variation and historical trends
MOSAIC	Anthropogenic EDGAR emissions for China replaced with the national P-CAME inventory	Test alternative emissions trend data for China
NATURE	Anthropogenic emissions fixed at 2010; time-variable geogenic and secondary emissions	Isolate the effect of non-anthropogenic sources and drivers on temporal trends
AGSM	Emissions from ASGM fixed at 2010; time-variable geogenic and secondary emissions	Isolate the effect of ASGM emissions on temporal trends
SCAv4	Trends in MOSAIC emissions are scaled for Europe, North America, and East Asia to better match observations	Estimate the decreases in emissions that are consistent with observed trends

The main results of the assessment are outlined below, organised according to the policy questions.

10.2.3 How consistent are the observed levels, temporal trends and spatial patterns of Hg levels in air (TGM, GEM, Wet Dep, GOM, PBM) with estimates from current mechanistic models at the global scale?

No straightforward information is available on this question, as the speciated levels of Hg from TGM to PBM are currently unavailable. However, the spatial patterns and temporal trends of total Hg levels are argued in later sections of the section 10.2.

10.2.4 What are the spatial patterns of Hg levels in air across the globe?

MA-1

Information Used.

The analysis of the geospatial patterns of Hg atmospheric concentration and deposition levels is based on multi-model simulations for the period 2010–2020. The study presents average results from the atmospheric chemical transport model ensemble (GEM-MACH-Hg, GEOS-Chem, WACCM; see Table 1) for the BASE scenario (Table 2). The simulated spatial patterns were evaluated against a compilation of global-scale observations of Hg(0) concentration and Hg wet deposition submitted to the OESG.

Methodology

Global patterns of Hg air concentrations and deposition were modelled for the period 2010–2020 using an ensemble of chemical transport models. The simulations incorporated the anthropogenic Hg emissions inventory EDGAR v8.1, as well as a variety of natural and secondary sources, including geogenic, volcanic, wildfire emissions, and terrestrial re-emissions. Atmosphere–ocean exchange was simulated by coupling atmospheric and ocean models with a harmonised parameterisation of air–water exchange. The simulated and observed spatial distributions of Hg(0) concentrations and Hg wet deposition, as well as simulated total (wet + dry) Hg deposition, were analysed by averaging over 2010–2020 to minimise the effect of interannual variability.

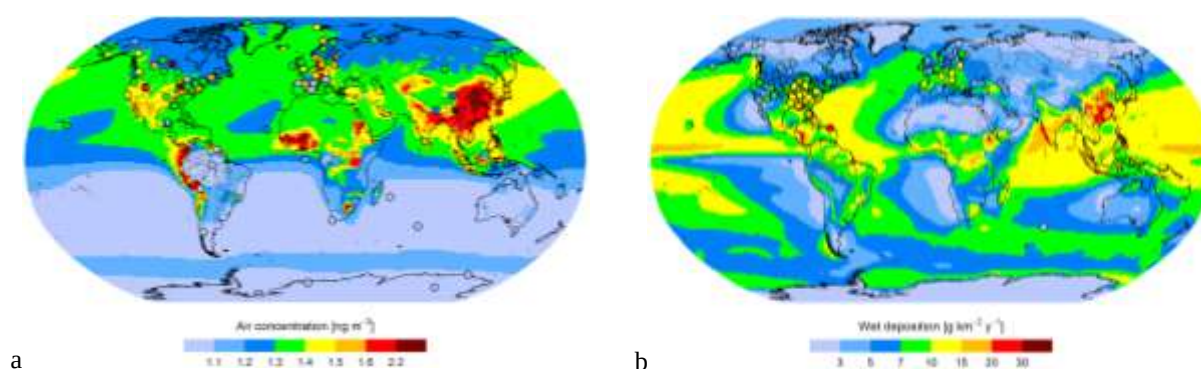
Answer

The simulation results demonstrated strong latitudinal variations in Hg(0) concentrations, driven primarily by the distribution of anthropogenic emissions. The background concentration levels vary from 1 ng m^{-3} in high latitudes of Southern Hemisphere to above 2.2 ng m^{-3} in industrial regions of the Northern Hemisphere and the Tropics. Elevated concentrations are estimated in major industrial regions and ASGM areas (Figure 4a). It generally agrees with available observations in various parts of the globe. However, multiple concentration hotspots, particular in the Tropics, are not covered with observational data.

The simulation results demonstrate strong latitudinal variations in Hg(0) concentrations, driven primarily by the distribution of anthropogenic emissions. Background levels range from about 1 ng m^{-3} at high latitudes of the Southern Hemisphere to above 2.2 ng m^{-3} in industrial regions of the Northern Hemisphere and the tropics. Elevated concentrations are estimated in major industrial regions and ASGM areas (Figure 4a), generally in agreement with available observations across different parts of the globe. However, multiple concentration hotspots, particularly in the tropics, are not covered by observational data.

Hg wet deposition is characterised by distinct spatial variability and is influenced by the patterns of primary and secondary emissions, precipitation, and atmospheric chemistry (Figure 4b). Simulations predict elevated Hg deposition with precipitation over the oceans and heavily industrialised regions, with fluxes exceeding $15 \text{ g km}^{-2} \text{ y}^{-1}$. The modelling results capture the spatial variation of observed Hg wet deposition in the limited regions covered by observational data, reproducing spatial gradients in Europe and North America, whereas most other regions remain without measurements.

Unlike wet deposition, total atmospheric Hg deposition is strongly elevated over forested regions due to vegetation uptake (Figure 4c). According to model estimates, high Hg deposition occurs over tropical forests in South America and Africa, as well as in industrial regions of East Asia, Europe, and North America.



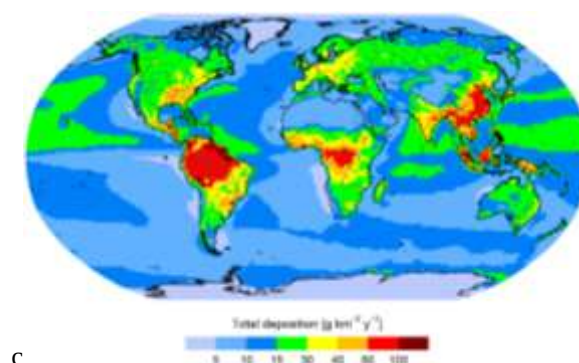


Figure 4. Spatial distribution of the multi-model ensemble mean Hg(0) concentration (a), Hg wet deposition (b), and total Hg deposition (c), averaged over the period 2010–2020. Coloured circles indicate measured values using the same colour palette.

Consistency with Existing Literature

The simulated spatial patterns of Hg(0) concentrations and Hg wet deposition are generally consistent with the results of GMA2018, despite the differences in simulated periods. Similar modelling results have also been reported in other recent publications (Shah et al., 2021; Zhou et al., 2021; Zhang & Zhang, 2022; Fu et al., 2024). The present modelling results suggest higher Hg terrestrial deposition in the tropics due to re-evaluated Hg(0) dry deposition to forests, which is in line with recent estimates by Zhou et al. (2021) and Feinberg et al. (2022).

Confidence

There is high confidence in simulated Hg(0) levels and Hg wet deposition in the northern mid-latitudes, particularly in North America, Europe, and East Asia, where most measurement sites are located. Confidence is lower in the tropics and the Southern Hemisphere. Observational coverage is especially lacking in tropical regions, which show the highest simulated atmospheric deposition, as well as over the oceans.

Information and Capacity Needs.

Additional Hg(0) observations are needed in regions with potentially high and uncertain emissions (e.g., the tropics) to better evaluate modelling results. This gap is being partially addressed through the recent deployment of passive samplers in several understudied regions. More widely distributed measurements of Hg wet deposition, together with reliable Hg(II) observations, are also required to improve the characterisation of atmospheric chemistry.

10.2.5 How consistent are the observed levels and spatial patterns of Hg in air and deposition with estimates from current mechanistic models across global regions?

MA-2

Information used

The model-ensemble simulations of Hg(0) concentrations and Hg wet deposition were evaluated against a compilation of long-term observations submitted to the OESG.

Methodology

Annual mean modelled Hg(0) concentrations and Hg wet deposition from the BASE scenario (Table 2) were compared with ground-based measurements for the period 2010–2020 (Figure 5). The comparison included only annual mean observations with data coverage exceeding six months.

Answer

The model ensemble reproduces observed Hg levels reasonably well, showing unbiased average concentrations and wet deposition, acceptable spatial correlation, and most model-to-measurement differences within a factor

of two. The simulated variability of Hg(0) is lower than observed, with models underestimating higher values mainly at the beginning of the period and overestimating lower values toward the end. Models also tend to underestimate high wet deposition fluxes, likely due to insufficient spatial resolution.

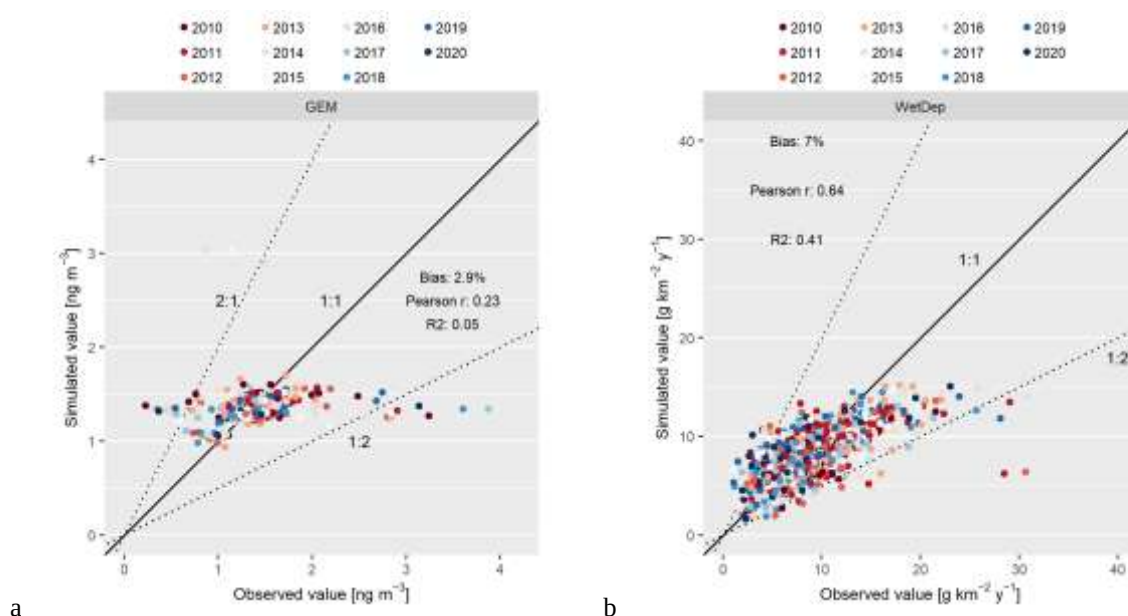


Figure 5. Scatter plots comparing modelled and observed annual Hg(0) air concentrations (a) and Hg wet deposition (b) for the period 2010-2020

Consistency with Existing Literature

General conclusions on the model performance and uncertainty levels are consistent with GMA 2018. However, the measurement dataset used in this study is more extensive and covers multi-year period.

Information and Capacity Needs.

Additional Hg(0) observations are needed in regions with potentially high and uncertain emissions (e.g., the tropics). More extensive measurements of Hg wet deposition, along with reliable Hg(II) observations, are also required to improve the characterisation of atmospheric chemistry.

10.2.6 What is the spatial representativeness of current levels and temporal trends of Hg observed in air?

MA-3

Figure 8 shows the spatial distribution of modeled annual mean concentrations of atmospheric elemental mercury Hg(0), aggregated by IPCC regions, with current monitoring sites indicated by white dots. Atmospheric models provide crucial added value by offering spatially continuous estimates across regions where direct observations are lacking or sparse — particularly in the tropics, southern latitudes, and parts of Asia and Africa. This allows us to assess the representativeness of existing measurements and to identify geographic gaps where observations are most needed. Combining models and observations enables a more robust evaluation of current Hg⁰ levels and supports efforts to track progress under the Minamata Convention.

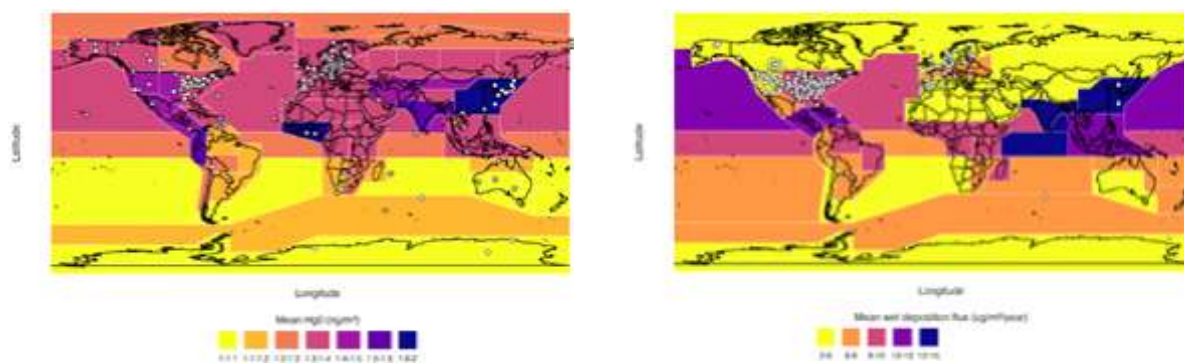


Figure 8. Spatial distribution of modeled annual mean concentrations of atmospheric elemental mercury ($\text{Hg}(0)$; top) and wet deposition flux (bottom), aggregated by IPCC regions. White dots indicate active monitoring sites during the 2010–2020 period, with only officially submitted data to the OESG. Grey dots denote sites where passive samplers have been deployed more recently (measurements initiated in 2018 or later), providing valuable new observations in previously under-monitored regions.

10.2.7 How consistent are the anthropogenic emissions inventories/inventory data and current mechanistic models with spatial patterns and temporal trends of Hg levels observed in air on the global scale?

MA-4

Information used

To analyze observed trends in atmospheric Hg, we use the dataset submitted to the OESG which includes air concentrations of Hg(0) and wet deposition measurements. The overall dataset is composed of 67 monitoring stations for GEM observations (61 in Northern Hemisphere, 6 in Southern Hemisphere) and 107 monitoring stations for wet deposition observations. In terms of the simulations, the model ensemble mean trends from three scenarios with different anthropogenic emissions assumptions were analyzed. The scenarios are: 1) BASE, using the EDGARv8.1 global emissions inventory; 2) MOSAIC, which substitutes Chinese emissions in EDGARv8.1 with the national inventory from P+CAME; and 3) SCAv4, which assumes scaled anthropogenic emissions trends in the MOSAIC inventory in North America, Europe, and Asia in order to better match observed trends.

Methodology

Trends in observational and modelled time series were analyzed using several different methodologies. In general, we apply the approaches recommended by the Tropospheric Ozone Assessment Report for analyzing multi-site measurement data (Chang et al., 2023). To derive overall trends for regions, we analyzed time series with linear mixed effect models (Bates et al., 2015) to aggregate the information from multiple sites within the same region. To analyze the trend at each grid cell of the model simulations or at individual measurement stations, we analyzed data using quantile regression of median trends (QR50).

Answer

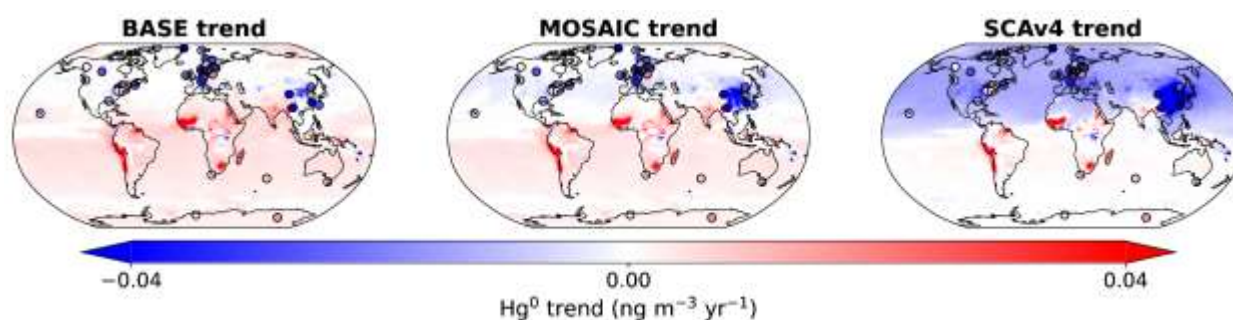


Figure 10. Spatial distribution of modelled Hg(0) trends over 2010–2020, compared to available measurements in the OESG datasets. Ensemble mean model simulations and observations (sites > 8 years) are deseasonalized and analyzed with quantile regression for the median trends in monthly mean values. Only grid boxes with trends significantly different from zero ($\text{trend} < 2 \times \text{trend error}$) are shown in colour.

We compare the trends from different modelled scenarios to available observation stations in two ways: as maps (Fig. 10) and as summarized time series for regions (Fig. 11). Observational trends in Hg(0) are declining over all Northern Hemisphere regions for the 2010–2020 time period and are stable or slightly increasing in the Southern Hemisphere. The BASE and MOSAIC simulations underestimate the decline in Hg(0) at sites in the Arctic, North America, Europe, and East Asia. The SCAv4 simulation, assuming stronger regional emissions declines for Europe, North America, and East Asia, shows much closer agreement with observational data than the simulations based on the EDGARv8.1 and P+CAME inventories (BASE and MOSAIC). Trends in the Southern Hemisphere are generally well captured in all simulations. This suggests that regional declines in anthropogenic emissions stronger than EDGARv8.1 and associated declines in secondary land and ocean emissions is the best explanation for the decline in atmospheric Hg concentrations in NH.

Comparisons of simulations with wet deposition measurements are less conclusive. The wet deposition in regions of North America and Europe is found to decline in the observations (Fig. 12). The BASE model simulations with EDGAR generally underestimate the observed decline in wet deposition, yet interannual variability patterns are well captured, likely due to the dependence on precipitation variability. Agreement with observed wet deposition trends is slightly improved in the SCAv4 simulation (Fig. 12), but this scenario still underestimates the observed decline in deposition in Eastern North America and Europe.

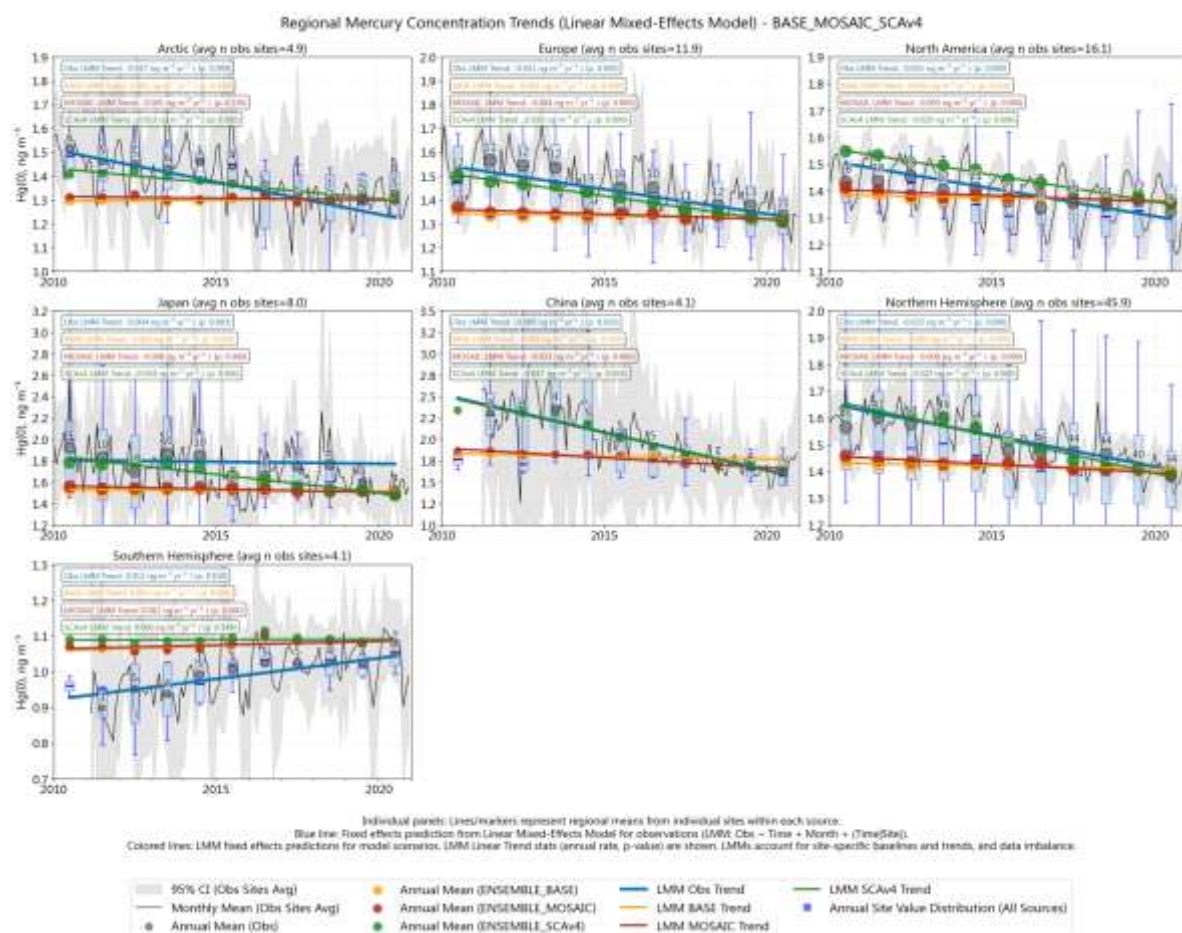


Figure 11. Regional trends in Hg(0) calculated using Linear Mixed-Effects modelling. Trends are calculated for both observations and simulations.

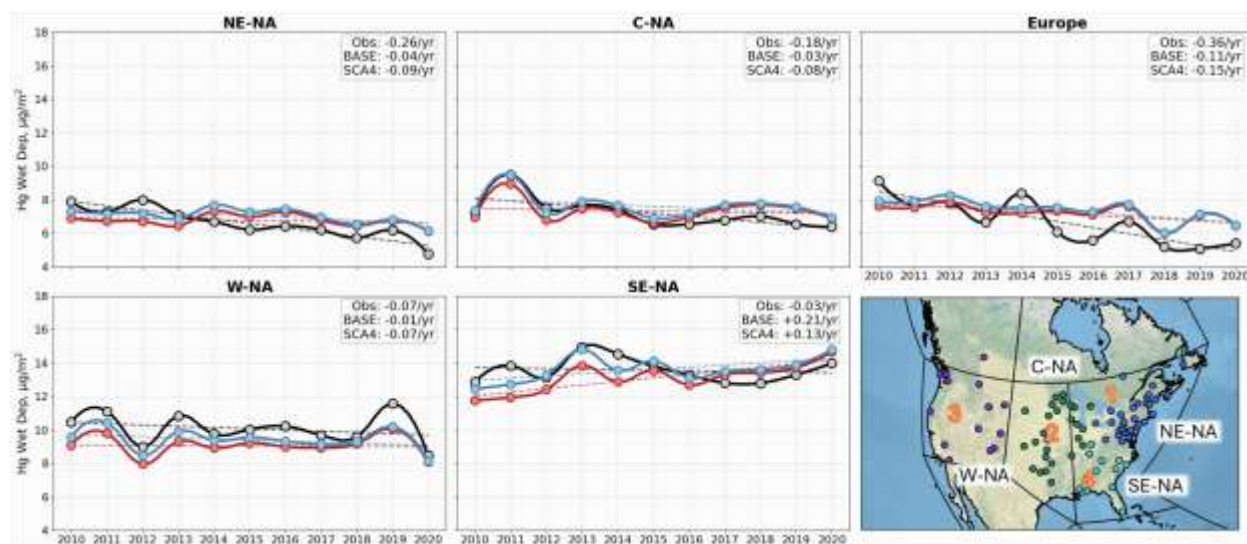


Figure 12. Hg wet deposition trends in different regions (abbreviations: NA–North America, NE–Northeast, C–Central, W–West, SE–Southeast) in observations and two simulations (BASE and SCAv4).

Consistency with Existing Literature

This analysis is generally consistent with previous studies that compared modelled Hg trends based on available emissions inventories with observed trends. Both Zhang et al. (2016) and Feinberg et al. (2024) also suggest that stronger declines in anthropogenic emissions compared to what is shown in inventories are necessary to explain available Hg(0) measurements. The 2018 Global Mercury Assessment did not focus on a trend estimation for

atmospheric Hg, as less site data was available globally. However, they identified general decreases in Hg measurements in Canadian and European stations. The SCAv4 scenario requires an additional $\sim 450 \text{ Mg yr}^{-1}$ to be emitted at the beginning of the analysis period (2010–2012), compared to the BASE scenario. This emissions total is small compared to the uncertainties in the emissions of Hg from product use and disposal (Horowitz et al., 2014; Streets et al., 2019), ASGM (Streets et al., 2019), and re-emissions of legacy anthropogenic Hg (Amos et al., 2013; Feinberg et al., 2024). Thus the assumptions of the SCAv4 scenario, which best matches observations, could still be consistent with our current knowledge of emissions uncertainties.

Confidence

The Northern Hemisphere midlatitudes are fairly well-covered by measurement stations for the 2010–2020 period, given the observation sites in North America, the Arctic, Europe, and East Asia. Nevertheless, other regions have very few measurement stations or none at all: e.g., continental regions in the Southern Hemisphere, the tropics and subtropics, Eurasia/Central Asia, and the Middle East. Thus, the assessment of model and observed trends for these regions is more uncertain. The observed declines are robust to choice of trend technique, as illustrated here with two different methods (quantile regression and linear mixed effects modelling). Numerous sensitivity simulations with other potential scenarios for changing emissions were conducted in one of the atmospheric Hg models (GEM-MACH-Hg) to rule out other explanations for the observed trends. We also expect that trends in Hg(0) sinks (vegetation uptake and oxidation) are unlikely to substantially impact the trends over this period 2010–2020, as elaborated in a following section of this report (MA-9) and the discussion in Feinberg et al. (2024).

Information and Capacity Needs

A long-term monitoring network in well-distributed sites would be invaluable to ensure better coverage of global air Hg trends and to improve attribution of changes in air Hg concentrations to primary and secondary emissions in all regions. The development of top-down modelling techniques to derive Hg emissions based on ambient measurements would improve the quantification of trend drivers. Some advances in this direction have been conducted for Hg (Angot et al., 2021; Denzler et al., 2017), yet this inverse modelling methodology is more developed for other pollutants such as chlorofluorocarbons (CFCs), which has aided in the detection of emissions changes (Rigby et al., 2019). A better understanding is required in terms of the gaps between bottom-up emissions inventories (such as EDGAR v8.1) and the emissions trends implied from observations. To this end, there should be efforts to improve the accounting of potentially overlooked anthropogenic sources in inventories (e.g., waste), as well as the re-emissions trends from legacy contaminated sites.

10.2.8 How sensitive are levels of Hg in the air to various Hg processes across the globe? How sensitive are levels of Hg in the air to various primary and secondary emissions and releases and environmental conditions across the globe?

MA-5

Information used

Multi-model ensemble MCHgMAP simulations of the BASE scenario were used to analyse possible sensitivity of Hg levels to variety of factors including anthropogenic and secondary emissions, environmental parameters (precipitation, wind speed, temperature, etc.) and surface characteristics (land cover, LAI).

Methodology

A multi-factor correlation analysis was conducted to assess the spatial co-variation of Hg air concentrations and deposition with anthropogenic and environmental factors. The analysis was performed using average Hg levels and potential drivers across the IPCC reference land and ocean regions (Figure 5).

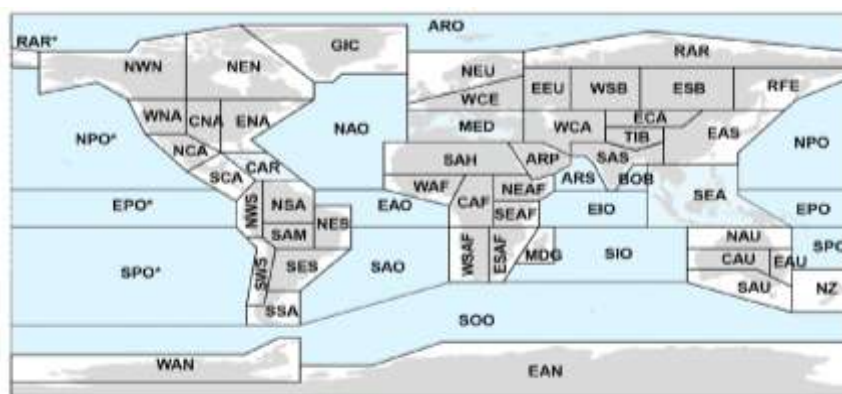


Figure 5. IPCC reference land and ocean regions used in the analysis (Iturbide et al., 2020)

The statistical analysis revealed a significant positive correlation between Hg(0) concentrations and anthropogenic emissions, and a weaker correlation with other terrestrial emissions (geogenic, volcanic, wildfire) and re-emissions (Figure 6). Oxidised Hg(II) correlates positively only with anthropogenic emissions, while Hg dry deposition also shows a strong correlation with vegetation cover (leaf area index). Both Hg(0) and dry deposition correlate negatively with wind speed and Br concentration, one of the primary Hg oxidants. Wet deposition correlates positively with various emissions and meteorological parameters (precipitation and air temperature) but negatively with snow and ice cover.

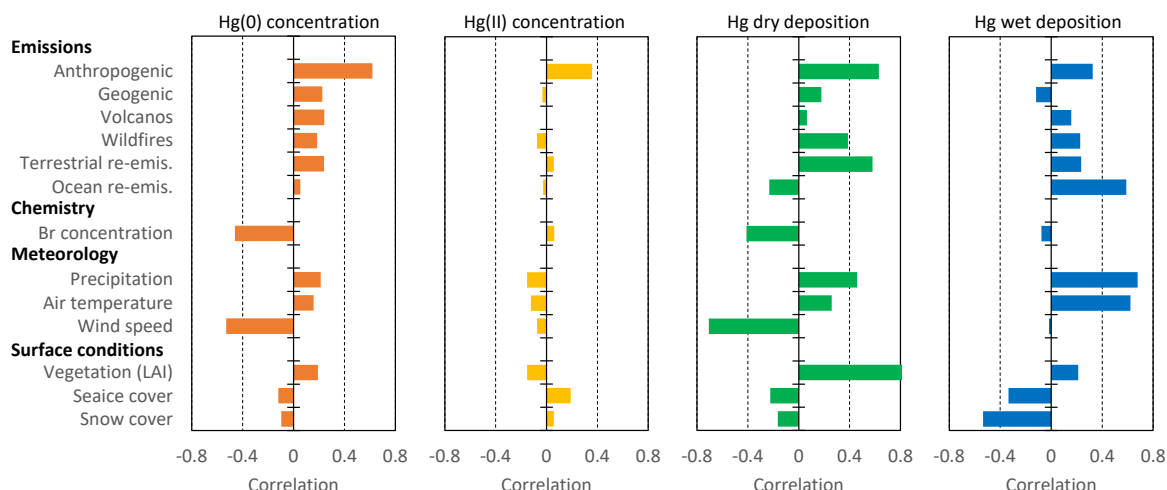


Figure 6. Pearson correlations of Hg concentrations and deposition across IPCC regions with emissions and environmental factors

Qualitatively, the results of the analysis are illustrated in Figures 7 and 8.

Answer

High Hg concentrations and deposition are driven by multiple factors, including emissions, precipitation, and vegetation cover. Hotspots of elevated Hg concentrations are linked to regions with high anthropogenic and/or secondary emissions (e.g., East, South, and Southeast Asia; West Africa; and northwestern South America) (Figure 7). Atmospheric Hg deposition occurs via surface uptake (dry deposition) and precipitation scavenging (wet deposition); dry deposition dominates over land, while wet deposition prevails over the oceans. Deposition is strongly influenced by surface and environmental conditions, with the highest levels observed in vegetated regions with high precipitation (Figure 8). Many regions with high simulated Hg concentrations remain underrepresented in observational datasets.

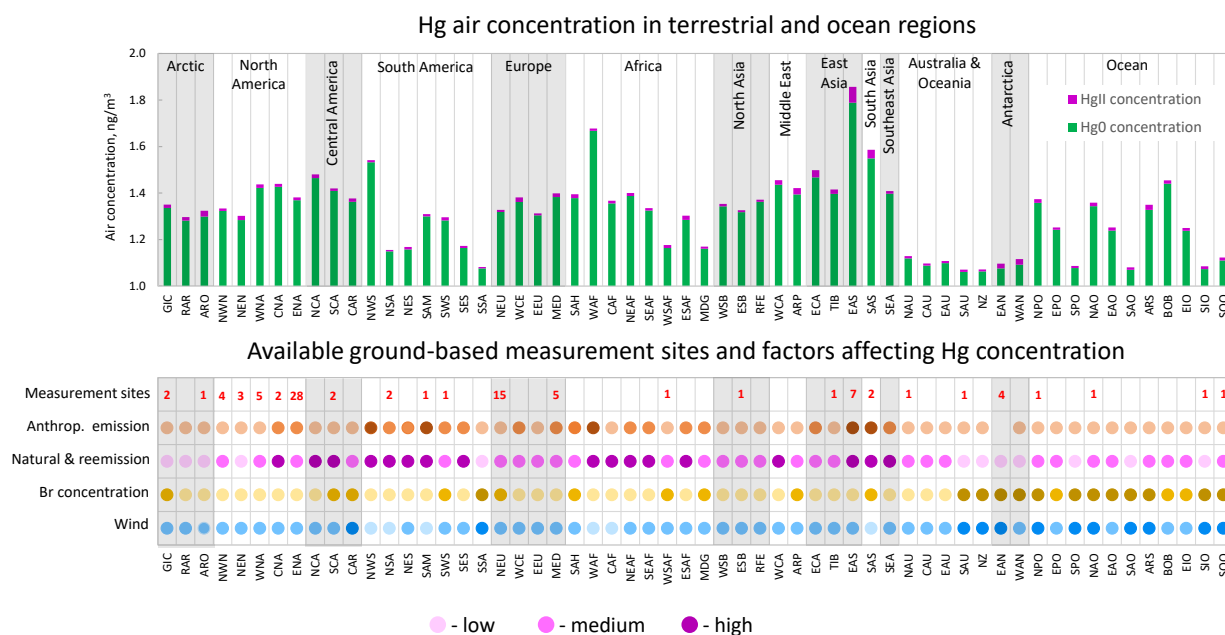


Figure 7. Average atmospheric Hg concentrations across IPCC regions, complemented by a qualitative analysis of factors influencing these levels and the availability of ground-based Hg(0) measurement sites.

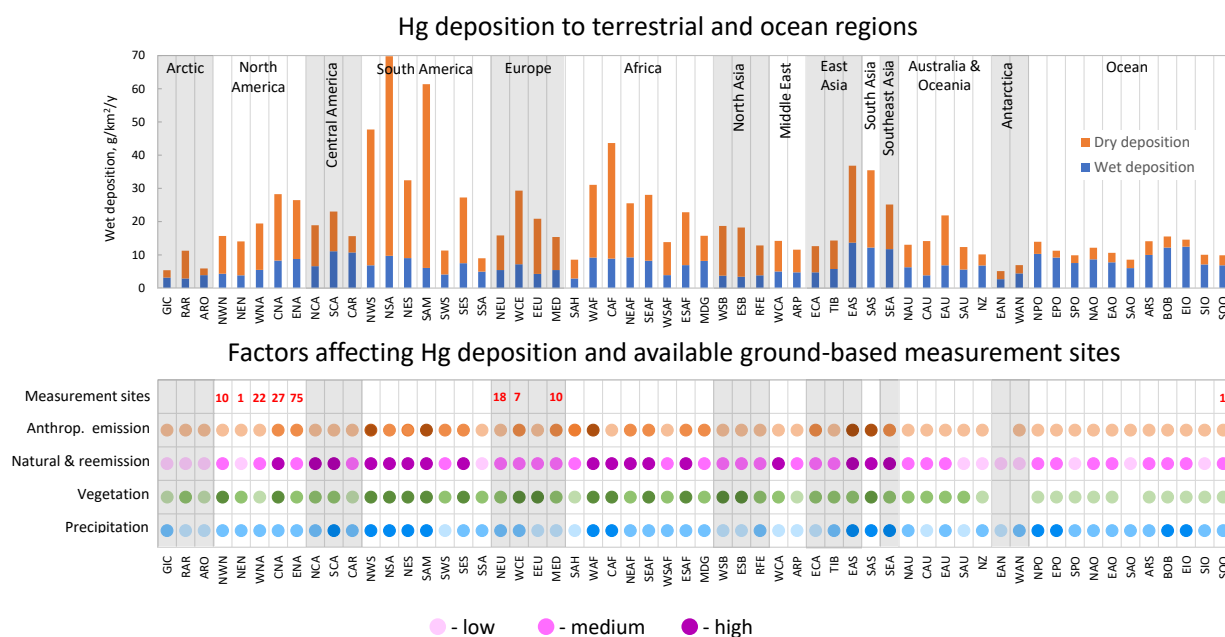


Figure 8. Average atmospheric Hg concentrations across IPCC regions, complemented by a qualitative analysis of factors influencing these levels and the availability of ground-based Hg wet deposition measurement sites

Consistency with Existing Literature.

Information and Capacity Needs.

A more extensive analysis of the effects of driving factors and processes on Hg levels is needed, involving sensitivity simulations, assessments of atmospheric chemistry, and the application of more advanced statistical approaches (e.g., Positive Matrix Factorization).

10.2.9 What are the spatial patterns and temporal trends of Hg levels in the ocean across the globe?

How sensitive are levels of Hg in the ocean to various Hg processes across the globe?

What is the spatial representativeness of current levels and temporal trends of Hg observed in ocean?

MA-6

Information used

To assess spatial patterns and temporal trends of Hg in the global ocean, we used model simulations from the MCHgMAP ensemble (section 3.2 Table 1). Observational data from surface ocean research studies were used to evaluate the models and are further used to compare to the BASE simulation (see section 3.2.9). Three model simulation scenarios were analyzed (section 3.2 Table 2):

1. **BASE**, representing realistic time-variable anthropogenic (EDGAR v8.1) geogenic, and secondary emissions for 2010–2020.
2. **STEPDECLINE**, simulating a theoretical sudden global 20% reduction in anthropogenic emissions after 2015.
3. **NATURE**, anthropogenic emissions fixed at 2010 levels with time-variable geogenic and secondary emissions to isolate natural variability from anthropogenic influences.
4. **Observations** of total and elemental Hg in the ocean from the marine Hg database compiled for this report (see emissions and releases).

Methodology

Temporal trends and spatial patterns were analyzed using the ensemble mean of the two ocean model simulations (Fig. 8). Model confidence was assessed based on the spread of individual models and comparison to observations of total and elemental Hg (Fig. 9). Surface ocean sensitivity to anthropogenic emissions was evaluated by comparing the BASE and STEPDECLINE scenarios, while the relative impact of natural variability and anthropogenic emissions was assessed by comparing BASE and NATURE simulations. Observational Hg data (total Hg and Hg⁰) were used to evaluate the spatial representativeness of modeled Hg concentrations. Trends were analysed using 10 years of model output ranging from 2010 to 2020.

Answer

The global surface ocean exhibits well-defined total Hg spatial patterns. Anthropogenic influences are evident in the Northern Hemisphere (0.65 pM) versus the Southern Hemisphere (0.55 pM). On a north-south gradient the general distribution exhibits higher concentrations in polar regions and at the equatorial interhemispheric convergence zone (ITCZ) while there are much lower values in the mid latitudes between 20° and 40°. These large scale patterns are mainly driven by upwelling and downwelling and follow general ocean circulation patterns. However, it is also seen that the highest Hg concentrations are found near major anthropogenic source regions namely the IPCC ocean regions northern Indian Ocean (SAS), South China Sea (EAS), eastern Pacific (SEA). Observational coverage remains limited, but where available, the spatial representativeness of the modeled Hg distribution is in good agreement (Tables 3,4,5).

- **Upwelling regions** (e.g., Benguela, Equator) and areas near major anthropogenic sources show the highest Hg concentrations (0.7-1.0 pM).
- **Subtropical gyres** in both hemispheres (20°-40° latitude) exhibit lower concentrations (0.2 -0.4 pM).
- **Subpolar regions** (50°-70° latitude) show increasing gradients (0.5-0.7 pM) due to vertical mixing of Hg-enriched deeper waters.
- **Permanent sea-ice regions** display elevated Hg (0.6-0.8 pM), as Hg(0) evasion is limited.

Over 2010–2020, surface ocean Hg levels were largely stable, with a slight decreasing but insignificant trend after 2015 ($\sim 0.5\%$ per annum, or 3.8 fM a^{-1}). The tropics showed no detectable long-term trend at all (Section 3.4 Figure 3). The natural inter annual variability seems driven by well known ocean oscillations such as ENSO. ENSO-driven natural variability can explain a significant portion of interannual changes: the correlation between annual ENSO index and modeled inter-annual Hg changes is high ($R = 0.84$), accounting for up to 70% of the observed variance. The NATURE scenario confirmed that natural variability dominates changes over the period. We are not able to link temporal changes in ocean Hg concentrations to changes in anthropogenic emission during the study period.

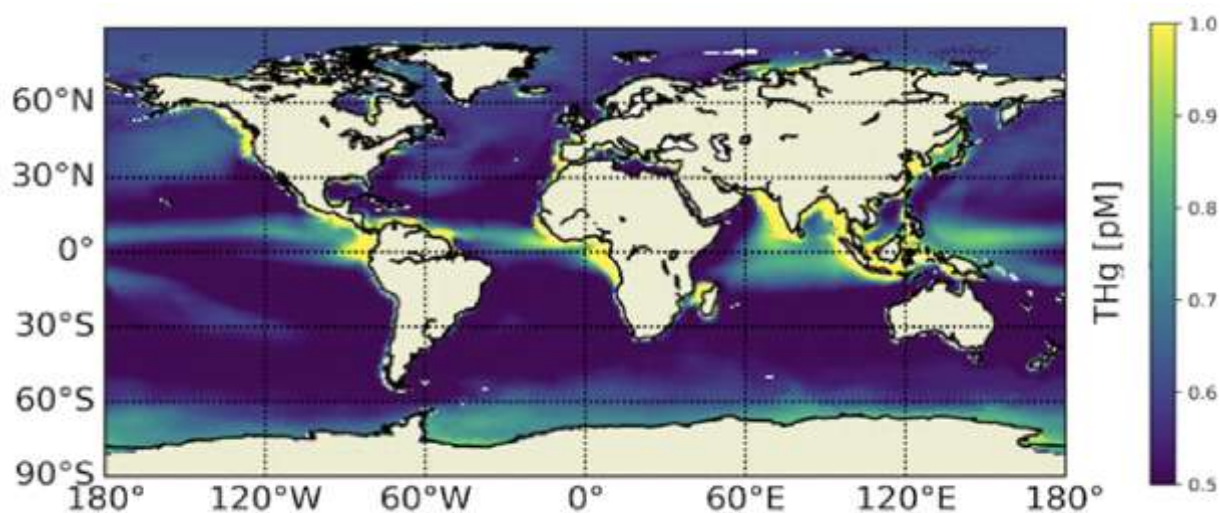


Figure 8: Average modeled spatial distribution of total Hg in the surface ocean 2010–2020. Highest Hg concentrations are found in upwelling regions and near major anthropogenic source regions.



Figure 9: Location of surface observations or profiles (multiple observations with depth).

Modeled sensitivity tests indicate the surface ocean responds strongly and immediately to atmospheric deposition: a 20% reduction in atmospheric inputs in the STEPDECLINE scenario resulted in a 15% reduction in surface Hg, with a new equilibrium reached within less than 3 years. It should be noted that more localized reduction signals will propagate slower across the globe than the uniform global reduction introduced in this model scenario. Finally, riverine and coastal inputs influence Hg concentrations mainly on regional scales. Anthropogenic emission changes (**BASE**) during 2010–2020 produced no detectable global signal in either of the marine Hg models.

Consistency with existing literature

These results are consistent with prior studies of global marine Hg cycling. Spatial patterns align with known oceanographic features, and the dominant role of atmospheric deposition in surface ocean Hg is supported by previous model-observation comparisons (Zhang et al., 2019, Rosati et al., 2020, Bieser et al., 2024)

Confidence

- High confidence: spatial patterns of Hg and the importance of atmospheric input.
- Low confidence: temporal trends due to limited observational coverage and strong natural variability, especially in the tropics, Southern Hemisphere midlatitudes, and poorly monitored subpolar regions.

Information and capacity needs

- Expansion of surface ocean observations, particularly in under-sampled regions, to better constrain models.
- Recurring observations on fixed transects to better identify long term trends.
- Continued development of marine Hg models and intercomparison studies to reduce uncertainties and improve predictions of responses to emission changes.
- Better data on Hg fluxes from land such as coastal erosion and groundwater inflow. More data on geogenic emissions from hydrothermal vents.

10.2.10 Key Messages / Policy Relevance

- **Spatial patterns:** Highest Hg concentrations occur in upwelling regions, polar areas, and near major anthropogenic sources; subtropical gyres show lower levels.
- **Temporal trends:** Surface ocean Hg levels were largely stable 2010–2020; Natural interannual variability, particularly ENSO, dominates observed inter-annual changes.
- **Sensitivity:** Surface ocean Hg is highly sensitive to atmospheric deposition; a 20% reduction in inputs leads to a ~15% decline in under 3 years. Land-based inputs have only localized effects.
- **Modeling contribution:** Ocean models are essential to quantify spatial and temporal Hg patterns, assess system sensitivity to emissions, and disentangle natural versus anthropogenic drivers and to identify the fraction of legacy versus geogenic emissions.
- **Confidence and gaps:** High confidence in spatial distribution and atmospheric influence; low confidence in long-term trends; expanded observations and model development are critical.

10.2.11 What is the spatial representativeness of current levels and temporal trends of Hg observed in ocean?

Where are we missing observations, where does the model help us, where are there interesting features in areas we do not have observations?

10.2.12 How consistent are the observed levels, spatial patterns, and temporal trends of Hg observed in ocean with estimates from current mechanistic models across global regions?

10.2.13 What is the spatial representativeness of current levels and temporal trends of Hg observed in ocean?

MA-7

Information used

The analysis is based on the BASE case model ensemble global Hg distribution and the database of ocean observations described in “link to report on releases”.

- **BASE**, representing realistic time-variable anthropogenic (EDGAR v8.1) geogenic, and secondary emissions for 2010–2020.
- **Observations** of total and elemental Hg in the ocean from the marine Hg database compiled for this report (see emissions and releases).

Methodology

Spatial representativeness and current levels of Hg were evaluated by comparison of modeled and observed Hg species. To quantify the agreement between model and observations we used statistical methods described in Bieser et al. (2024). Due to the sparse nature of ocean observations a trend analysis of the dataset is not feasible. Further, the models did not find a significant trend (MA-11).

Answer

Recent modeling efforts to simulate total mercury (Hg), elemental mercury (Hg⁰), and methylated mercury (MeHg) in the global ocean demonstrate a reasonable degree of accuracy with available observations but also highlight key areas where data gaps and model limitations remain.

Simulated total Hg concentrations align with observational data within an average error margin of 50%, meaning model outputs typically lie within a factor of two of the measurements (Figure 10, Table 3). These models are able to reproduce the broad latitudinal gradients of total Hg and Hg⁰ concentrations, particularly in tropical and mid-latitude regions. However, in polar regions, models fall short in capturing the high variability and elevated values observed in situ (Figures 10 and 11), indicating regional shortcomings in current process understanding or model representation.

In regions with significant anthropogenic emissions, such as East Asia, peak Hg concentrations heavily skew observational averages. The concentration of observations in these areas may distort the evaluation of model skill on a global scale, as they disproportionately influence mean values.

Data availability remains a major constraint on model evaluation. For example, in the Indian Ocean, only a single cruise dataset is currently available. Furthermore, the sporadic and campaign-based nature of ocean sampling precludes the robust identification of long-term trends in observational records. There is a significant spatial and temporal sampling bias, with the majority of observations limited to few regions and a strong seasonal clustering.

For dissolved elemental mercury (Hg⁰), the model reproduces observed concentrations within an average error of 65% (Figure 11, Table 4), indicating moderate confidence in the model's treatment of surface ocean redox processes. In contrast, the largest discrepancies are found for methylated mercury species (Table 5). At the ocean surface, model errors exceed 100%, largely driven by the model's limited ability to reproduce monomethyl mercury (MMHg) concentrations. However, in deeper ocean layers where dimethyl mercury (DMHg) dominates, model performance improves significantly, suggesting a better representation of deep-ocean methylation processes.

Confidence

- Moderate to high confidence for total Hg and Hg⁰ concentrations
- Lower confidence in polar regions
- Low confidence for MMHg concentrations

Information and capacity needs

- More detailed model intercomparison studies are necessary
- More observations are needed to properly evaluate the models
- Sophisticated methods of measuring air-sea exchange of Hg in the environment

Table 3. Statistical analysis of total Hg based on vertical profiles

	observation	model	bias	error	Fac x2	N
	median	median				
surface (0-50m)	0.67 pM	0.5 pM (0.38-0.65)	-9%	52%	75%	313
intermediate ocean (50-1000m)	0.77 pM	0.60 pM (0.37-0.85)	-13%	31%	98%	291
deep ocean (>1000m)	0.85 pM	0.63 pM (0.39-0.90)	-22%	27%	95%	206

Table 4. Statistical analysis of Hg(0) in the surface ocean

	observation	model	bias	error	Fac x2	N
	median	median				
Hg(0) surface (0-50m)	14.5 pg/L	14.4 pg/L	16%	65%	57%	1562

Table 5. Statistical analysis of methylated Hg based on vertical profiles

	observation	model	bias	error	Fac x2	N
	median	median				
surface (0-50m)	97 fM	240 pM (80-380)	-122%	146%	37%	432
intermediate ocean (50-1000m)	220 pM	344 pM (55-350)	-23%	50%	91%	342
deep ocean (>1000m)	280 pM	320 pM (54-363)	-12%	32%	97%	155

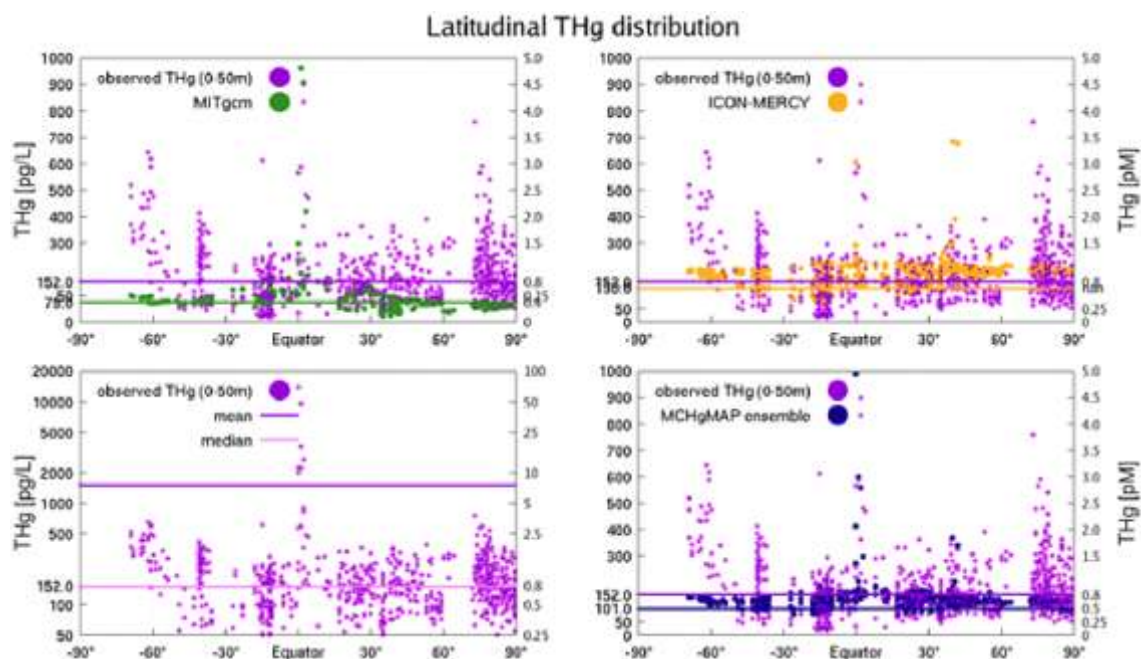


Figure 10. Latitudinal distribution of modeled and observed total Hg concentrations in the surface ocean.

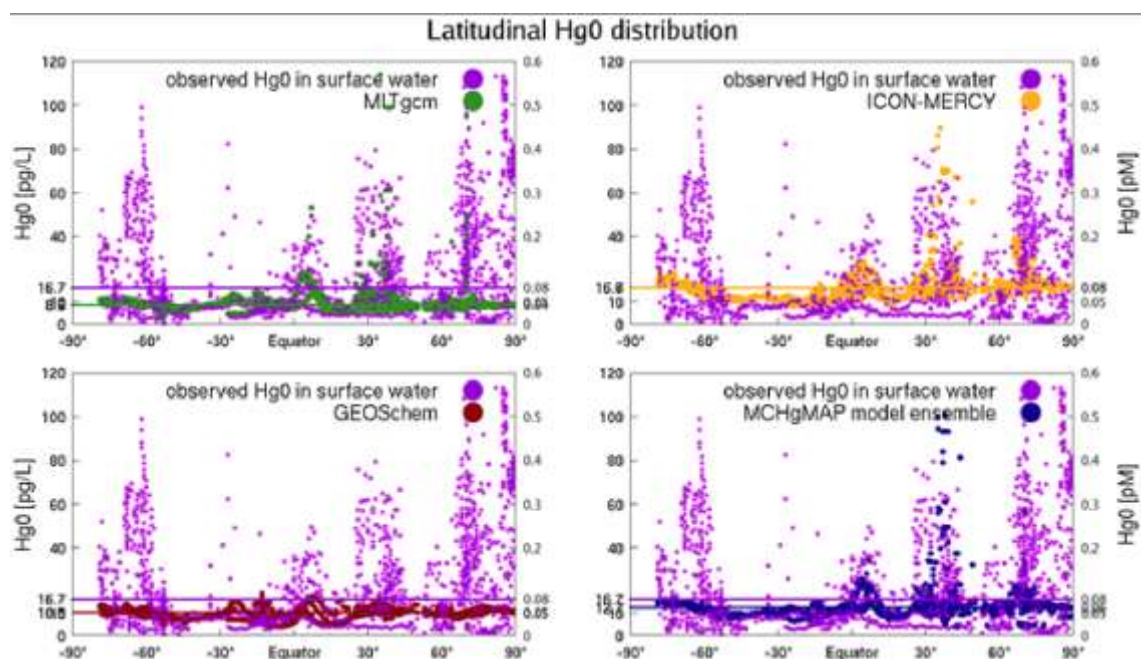


Figure 11. Latitudinal distribution of modeled and observed total Hg concentrations in the surface ocean.

10.2.14 What are the current levels and temporal trends of secondary Hg emissions from oceans across the globe?

What is the contribution of changes in environmental conditions to changes in secondary Hg emissions from global oceans over the period circa 2010 to present across the globe?

MA-8

Information**used**

To assess secondary Hg emissions from the ocean, we used the BASE case model simulations from the MCHgMAP ensemble of ocean models.

Methodology

Oceanic re-emissions were calculated by the ocean models using atmospheric boundary conditions for atmospheric Hg concentrations and deposition fluxes. Air-sea exchange was calculated following the harmonized MCHgMAP set of equations (Dastoor et al., 2025). Trends were analysed using 10 years of model output ranging from 2010 to 2020.

Answer

Model results indicate that during the period 2010–2020, the global ocean was a net emitter of elemental mercury to the atmosphere. Specifically, the annual gross Hg evasion flux from the ocean surface was estimated at approximately **4,300 Mg a⁻¹**. This emission flux was offset by atmospheric deposition, estimated at **3,700 Mg a⁻¹**, leading to a **net total upward flux of about 600 Mg a⁻¹** from the ocean into the atmosphere. Given the inherent uncertainty of current models and input data, these values indicate an ocean close to equilibrium with the atmosphere.

While the modeled evasion fluxes are somewhat higher than previously reported estimates, such as those included in the Global Mercury Assessment 2018, they remain within the range of existing literature. The increase in estimated evasion is attributed to improved model resolution and updated parameterizations of air-sea exchange processes. Further, this is the first time the Hg air-sea exchange was calculated from the side of the ocean models.

Notably, the modeling exercise did not identify any statistically significant changes in the magnitude of secondary Hg emissions from the global ocean over the ten-year study period. The modeled fluxes remained relatively stable year-to-year, suggesting that any emerging trends may only become apparent over longer temporal scales. Therefore, it is currently not possible to establish a clear trend in secondary oceanic mercury emissions for the period 2010–2020 based on available model results.

The contribution of environmental variability, such as changes in sea surface temperature, stratification, primary productivity, and atmospheric Hg concentrations, to changes in secondary Hg emissions was indirectly examined through the model outputs. However, the relatively short duration of the modeling period and the lack of significant temporal trends in the emission forcing data limit the capacity to robustly attribute changes in oceanic emissions to environmental drivers.

Although the current modeling approach is not fully coupled, that is, it exchanges fields with atmospheric models which are run separately, it provides a useful approximation of the air-sea exchange dynamics. However, this setup limits the system's responsiveness to rapidly changing environmental inputs and feedback mechanisms.

Uncertainties, Confidence, and Research Needs

There is **medium confidence** in the total air-sea exchange estimates at the global scale, based on consistency across different model runs and general agreement with existing literature. We estimate the uncertainty to be in the range of 1000 Mg a⁻¹. However, the system is currently so close to equilibrium, that it is not clear whether the ocean actually is a source or a sink for Hg on a global scale.

However, **confidence is lower in polar regions**, where data coverage is sparse, and environmental conditions differ substantially from lower latitudes. This contributes to a higher degree of uncertainty in those regions' contribution to global Hg cycling.

Addressing these limitations requires the development and implementation of **fully coupled atmosphere-ocean or Earth system models** capable of simulating feedbacks between mercury cycling, climate variability, and oceanic biogeochemistry. Such models would provide more robust projections of future mercury fluxes and allow for a better understanding of how environmental change may alter the ocean's role in the global mercury budget.

10.2.15 Key Messages / Policy Relevance

In summary, global oceans emitted between **300 and 900 Mg a⁻¹** of net Hg(0) to the atmosphere during the period 2010 to 2020, acting as a consistent net source of mercury. This is not in agreement with current estimates from

atmospheric models which identify the ocean as a net sink of Hg. Based on current knowledge the uncertainty in the air-sea exchange flux of Hg is in the range of **1000 Mg a⁻¹**.

No significant temporal trends were observed in the model simulations during this timeframe. Environmental drivers such as atmospheric deposition and surface ocean processes undoubtedly influence secondary Hg emissions, but the short time span and limitations of the current model framework constrain our ability to quantify their contributions. Future work must prioritize long-term observational records and the development of fully coupled Earth system models to advance understanding in this critical area.

Table 6. Modeled surface ocean Hg fluxes

	Annual flux
Net Hg(0) evasion flux from the surface ocean to the atmosphere	-4300 Mg a ⁻¹
Atmospheric Hg deposition flux	3700 Mg a ⁻¹
Net total Hg upwards flux from the surface ocean into the atmosphere	-600 Mg a ⁻¹
Net total Hg downwards flux from the surface ocean to the intermediate ocean	300 Mg a ⁻¹
Net total Hg inflow through rivers	900 Mg a ⁻¹

10.2.16 MA-14: How consistent are the observed levels, temporal trends and spatial patterns of Hg levels in air, water, soil with estimates from current multi-media mechanistic models?

MA-9

Information Used

- Multi-media global box model (described in Geyman et al., 2025).
- Post-2000 anthropogenic emissions from EDGAR v8.1 (BASE inventory) and EDGAR v8.1 combined with P+CAME (MOSAIC inventory).
- Pre-2000 releases from Streets et al. (2019).
- Fluxes and Hg mass balance from atmospheric (WACCM, GEOS-Chem, GEM-MACH-Hg) and ocean (MITgcm, ICON) models.

Methodology

- Set of box model simulations for different combinations of models and emissions inventory.

Answer

- BASE and MOSAIC inventories show increasing anthropogenic emissions post-2000.
- This results in a corresponding increase in the modeled atmospheric Hg mass over the period.

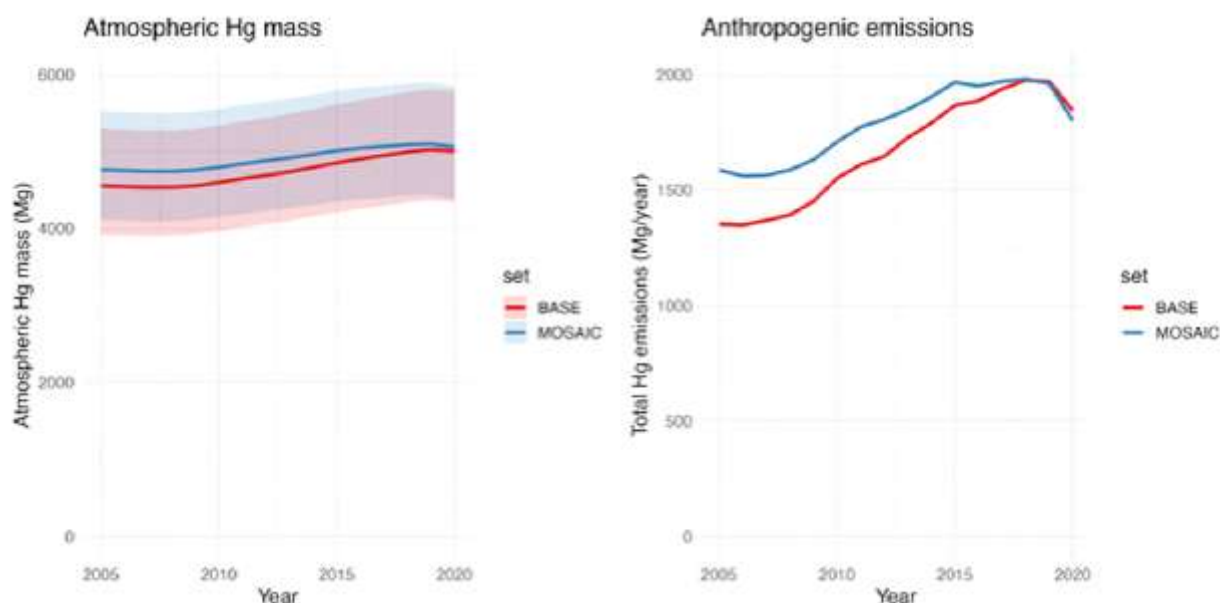


Figure 20. Atmospheric Hg trends and emissions from 2005 to 2020 based on global box model simulations. Left panel: Modeled atmospheric Hg mass using two emission inventories – BASE (EDGAR v8.1) and MOSAIC (EDGAR v8.1 and P+CAME) – shown as ensemble envelopes from different combinations of atmospheric and oceanic models. Right panel: Corresponding anthropogenic Hg emissions from the BASE and MOSAIC inventories. The increase in emissions in both inventories leads to an increase in modeled atmospheric Hg mass, contrasting with observational trends showing stable or declining levels, particularly in the Northern Hemisphere.

Consistency with Existing Literature. The increasing atmospheric Hg mass is not fully consistent with observational trends that show a decline in the Northern Hemisphere and stable or slightly increasing levels in the Southern Hemisphere. However, since the model is global and not hemispherically resolved, direct comparison with hemispheric trends is not possible. The discrepancy may also reflect known limitations in emission inventories, particularly underestimation of emission reductions in key regions (e.g., North America, Europe), and uncertainties in natural and legacy sources.

Confidence. High confidence in the internal consistency of the box model results: increasing emissions lead to increasing atmospheric Hg mass, as expected. However, confidence in the absolute trends is limited by uncertainties in the emission inventories used as input, particularly regarding regional trajectories and recent changes in anthropogenic sources.

Information and Capacity Needs

- Hemispherically or regionally resolved box models to better evaluate consistency with observed spatial trends.
- Improved constraints on post-2000 anthropogenic and legacy emissions.
- Better representation of atmospheric sinks and re-emission processes, informed by observational data and process-based studies.

10.2.17 How consistent are the observed levels, temporal trends and spatial patterns of Hg levels in oceans with estimates from current mechanistic models at the global scale?

Currently no information is available on this question.

10.2.18 How sensitive are levels and trends of Hg in the air and ocean to various primary and secondary emissions and releases and environmental conditions across the globe?

Currently no information is available on this question.

10.2.19 How sensitive are levels and trends of Hg in the air and ocean to various Hg processes across the globe?

Currently no information is available on this question.

10.2.20 How consistent are the observed levels, temporal trends and spatial patterns of Hg levels in biota with estimates from current mechanistic models of biotic exposure?

Currently no information is available on this question.

10.2.21 How consistent are the observed levels, temporal trends and spatial patterns of Hg levels in humans with estimates from current mechanistic models of human exposure?

Currently no information is available on this question.

10.2.22 How consistent are the observed levels, temporal trends and spatial patterns of Hg levels in air, water, soil with estimates from current multi-media mechanistic models?

Currently no information is available on this question.

10.3 Process and Source Attribution for Air and Oceans

10.3.1 What data are available?

The organization of chapter 10.3 is different from chapter 10.1 and 10.4. The questions in chapter 10.3 is directly directed to the outcomes of MCHgMAP project.

10.3.2 What is the contribution of primary anthropogenic emissions and releases to Hg levels observed in the air (GEM, Wet Dep, Dry Dep)?

Currently no information is available on this question.

10.3.3 What are the fractional contributions of changes in primary anthropogenic and natural Hg emissions and environmental conditions to changes in Hg levels observed in air and deposition over the period circa 2010 to present across global regions?

MA-10

Information used

We use the BASE and NATURE scenarios from MCHgMAP model ensemble simulations to calculate the contribution of natural and anthropogenic emissions changes to atmospheric Hg trends (Hg(0) and wet deposition).

Methodology

The BASE simulation includes anthropogenic emissions and environmental forcing conditions varying between 2010–2020, while the NATURE simulation assumes anthropogenic Hg emissions fixed at 2010 values but with all other factors varying. We compared the trends in the NATURE simulation with the BASE simulation to identify the role of environmental conditions to changes in Hg levels and deposition. The natural contribution is defined as the trend in the NATURE simulation, while the anthropogenic contribution is defined as the BASE trend – NATURE trend.

Answer

In the case of Hg wet deposition (Fig. 13), the impacts of natural drivers are found to be more significant than the contribution of anthropogenic trends, at least for the changes assumed in the BASE scenario. This is due to the influence of precipitation patterns on Hg wet deposition (Fig. 13) over the short (11 year) analysis period. On the other hand, for atmospheric Hg(0) the influence of natural drivers on the trends over land is generally minor, especially compared to anthropogenic changes (Fig. 14). Anthropogenic drivers in trends are found to be the most important, especially for locations where observations sites are currently located.

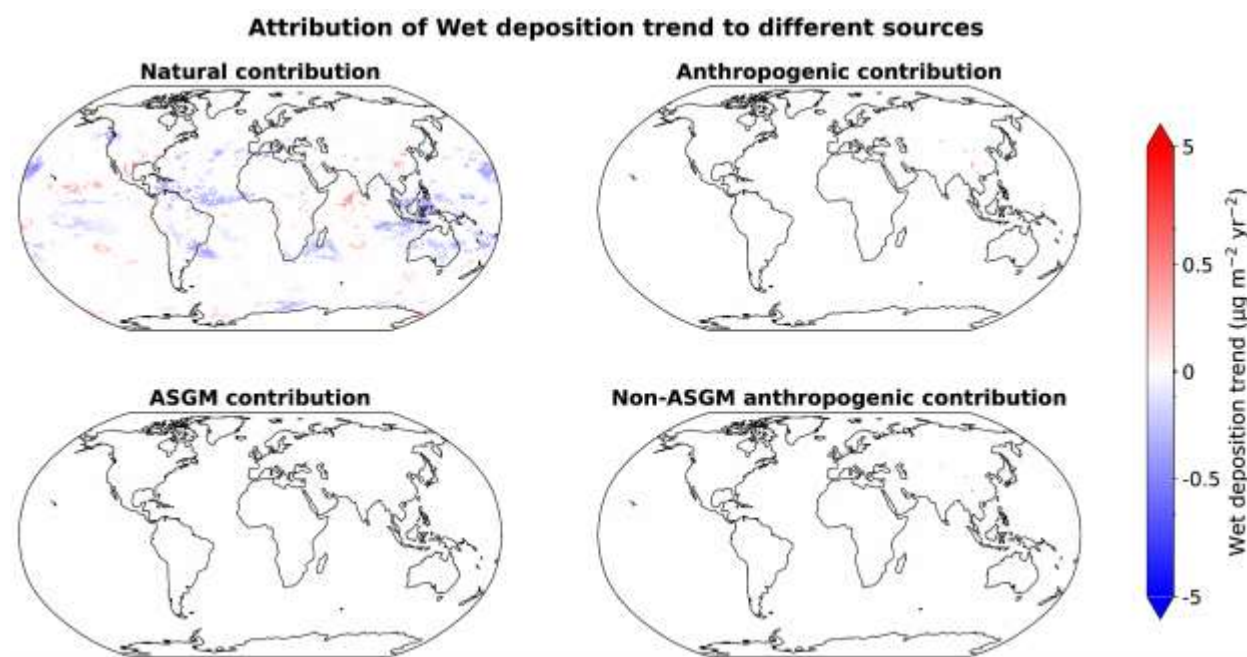


Figure 13. Spatial distribution of modelled wet deposition trend contributions from different drivers over 2010–2020. The anthropogenic contribution is calculated using the EDGARv8.1 emissions dataset. Ensemble mean model simulations and observations (sites > 8 years) are deseasonalized and analyzed with quantile regression for the median trends in monthly mean values. Only grid boxes with trends significantly different from zero ($\text{trend} < 2 \times \text{trend error}$) are shown in colour.

Consistency with existing literature

The 2018 Global Mercury Assessment did not include an assessment of model trends, and hence conducting such an assessment with a multi-model ensemble is a novel advance. The finding of this report that atmospheric Hg(0) trends are generally much more sensitive to emissions changes compared to wet deposition, which is more sensitive to meteorological trends, agrees with a previous model study that focused on the Great Lakes region (Giang et al., 2018)

Confidence

We have considered the potential for other environmental factors to explain the declining atmospheric Hg trends. Many climate change processes are most likely to lead to increasing atmospheric Hg rather than decreasing atmospheric Hg, for example: release of Hg from melting permafrost, increased Hg(0) ocean evasion due to warming (treated in models), decreased sea ice coverage allowing for more Hg(0) evasion (treated in models), and enhanced wildfire emissions (treated in models). One study has hypothesized that the vegetation sink of Hg may have increased, leading to lower Hg(0) concentrations (Jiskra et al., 2018); however, we find that the role of vegetation uptake on atmospheric Hg trends is minor in the 2010–2020 period, as several models consider the interannual variability in vegetation and its impact on Hg(0) dry deposition. Similarly, the changes in the oxidation of Hg(0) are unlikely to explain the trends over such a short time period, and they would lead to increases in Hg wet deposition and reactive mercury, which isn't found in the observations. Thus, we are reasonably confident that the declines in atmospheric Hg observations can be explained by changes in anthropogenic emissions and consequent re-emissions, rather than changes in environmental conditions.

Information and Capacity Needs

To improve our attribution of atmospheric Hg trends to natural or anthropogenic factors, several advances would be needed both on the modelling and measurement fronts. In terms of modelling, further development of coupled earth system models to better represent natural processes and their interactions with the Hg cycle would be critical for analyzing the impacts of environmental drivers. Long-term observations in new areas that are potentially more impacted by natural variability (tropics, ocean-influenced, Arctic, and Antarctic) would improve confidence in our understanding of trends. In addition, the assessment of re-emissions trends is limited

by gaps in the observations of soil and ocean concentrations and the associated fluxes, including a lack of measured time series in these media.

10.3.4 What is the fractional contribution of changes in primary anthropogenic Hg emissions to changes in secondary Hg emissions from soils over the period circa 2010 to present across the globe?

MA-11

Information Used

- Multi-media global box model (described in Geyman et al., 2025).
- Post-2000 anthropogenic emissions from EDGAR v8.1.
- Pre-2000 releases from Streets et al. (2019).
- Fluxes and Hg mass balance from atmospheric (WACCM, GEOS-Chem, GEM-MACH-Hg) and ocean (MITgcm, ICON) models.

Methodology

- Baseline simulation with full anthropogenic emissions (pre- and post-2000).
- Constant anthropogenic emissions at the 2010 level for the entire 2010–2020 period.
- Calculate the difference in secondary emissions between the baseline and the constant anthropogenic emission simulation over 2010–2020. This difference isolates the change in secondary emissions attributable to the changes in anthropogenic emissions after 2010.

$$\Delta Secondary_{2010-2020} = Secondary_{baseline} - Secondary_{constant\ emissions}$$

- Calculate the total change in secondary emissions over 2010–2020 in the baseline simulation:

$$Total\ change_{2010-2020} = Secondary_{2020} - Secondary_{2010}$$

Then, compute the fractional contribution of changes in primary emissions as:

$$Fractional\ contribution = \frac{\Delta Secondary_{2010-2020}}{Total\ change_{2010-2020}}$$

Answer

- Approximately 76% of the increase in soil Hg re-emissions over 2010–2020 is attributable to changes in primary anthropogenic emissions during the period, as captured by the difference between the Baseline and Fixed emission scenarios.

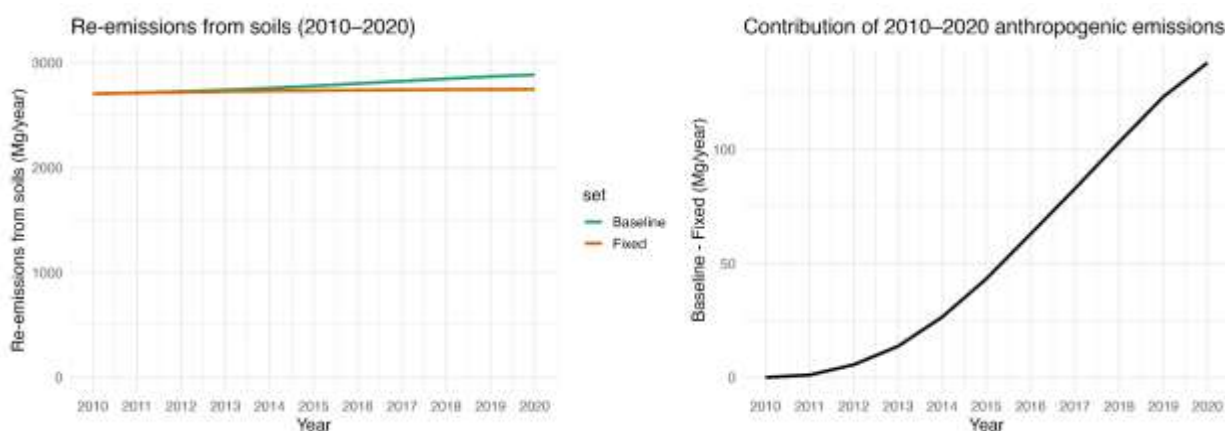


Figure 21. Annual Hg re-emissions from soils for two global box model simulation sets: one with time-varying anthropogenic emissions (Baseline) and one with emissions fixed at 2010 levels (Fixed). The right panel shows the difference between the two simulations, quantifying the impact of post-2010 changes in anthropogenic

emissions on soil re-emissions. Over the period 2010–2020, approximately 76% of the observed increase in soil re-emissions is attributable to these emission changes.

Confidence. While the model results are physically consistent and align with expectations that increasing emissions drive increased re-emissions, the confidence in the magnitude of the contribution is limited by uncertainties in the anthropogenic emission inventories themselves, especially regarding post-2010 trends (see question 13).

Information and Capacity Needs. Further refinement of anthropogenic emission inventories and are needed to enhance confidence of secondary emission trends.

10.3.5 What is the fractional contribution of changes in primary anthropogenic Hg emissions and releases and environmental conditions to changes in secondary Hg emissions and releases over the period circa 2010 to present across the globe?

MA-12

Information Used: Same as above.

Methodology: Same as above.

Answer

- Approximately 86% of the increase in soil and ocean Hg re-emissions over 2010–2020 is attributable to changes in primary anthropogenic emissions during the period, as captured by the difference between the Baseline and Fixed emission scenarios.

Confidence. It is important to emphasise that the global box model does not explicitly include changes in environmental conditions (such as climate change effects on soil temperature, moisture, or other ecosystem processes) that can also influence secondary Hg emissions. Therefore, this fractional contribution reflects only the impact of changes in anthropogenic emissions and releases, not environmental changes.

Information and Capacity Needs. Further work requires incorporation of environmental condition changes into modeling frameworks to better partition contributions from anthropogenic emission changes and climate or environmental variability.

10.3.6 What is the global mass balance of Hg budgets between emissions and releases, levels and movements in the environment? (using air, ocean, soil, and mass balance models)

MA-13

Information Used

- Multi-model ensemble simulations of the BASE scenarios from atmospheric and ocean models.

Methodology

- Uncertainty intervals showing the inter-model range of simulated Hg fluxes and reservoir masses.

Answer

- Current levels are shaped not only by ongoing anthropogenic emissions (~25% of annual emissions) but also by substantial legacy re-emissions from soils and oceans (~70% of annual emissions).
- Reservoir estimates confirm that the atmosphere holds only a small fraction of total environmental Hg (<0.5%), while soils and the deep ocean act as major sinks (1185–1340 Gg).
- Modeling is essential to resolve the full budget –especially by quantifying re-emissions and transfers that are not directly observable – and to assess how past and present emissions influence current and future Hg levels.

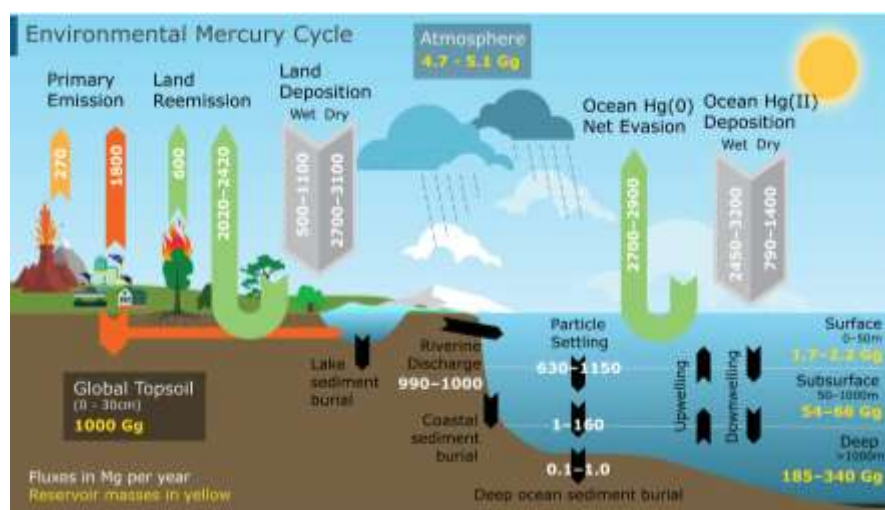


Figure 22. Global Hg mass balance. Primary emissions (both natural and anthropogenic) are shown in orange while legacy re-emissions from soils and oceans are shown in green

Consistency with Existing Literature

- Comparison with other recent publications (Outridge et al., 2018; Sonke et al., 2023).
- Relatively consistent with both studies. The main difference is that land re-emissions are twice that of past studies based on new terrestrial Hg modeling.

Confidence

- Relatively good agreement between models.

Information and Capacity Needs

- Ongoing debate as to the relative contribution of Hg(0) and Hg(II) to deposition to oceans (Song et al., 2025; Jiskra et al., 2021), affecting the magnitude of Hg(0) ocean–atmosphere gas exchange.
- Need in situ flux measurements to constrain atmosphere–surface exchange rates.

10.3.7 In each world region, how much of the contribution of primary anthropogenic emissions to changes in Hg levels observed in air over the period circa [2010] to present are due to emissions within the region versus outside the region?

Currently no information is available on this question.

10.3.8 How are anthropogenic Hg emissions and releases expected to change in future under different scenarios of implementation of the Minamata and other Conventions across global regions?

MA-14

Information Used

- GAINS V.4.0.3 model implementation from 2024
- Global Mercury Assessment 2018 emission factors
- Minamata Convention provisions and National Action Plans
- World Energy Outlook 2022 energy trends

Methodology

- Projection of speciated Hg emissions into the future, using global energy scenarios (see Fig. 23).
- Quantification of global clean air policy (PM 2.5, SO₂) and climate policy (aligning with the World Energy Outlook 2022 projections) (see Table 7).
- Quantification of Minamata Convention provision ‘option space’.

Modelling data is available from the GAINS model on a global level, until 2050, in 5-year steps. Its sectoral scope for Hg is comparable to that of the Global Mercury Assessment ‘18, with the addition of transport sectors, various industrial processes and domestic coal combustion. The GAINS model represents the majority of Hg emission sources in 270 sector-activity combinations. These combinations belong to the aggregated sector groups of artisanal and small-scale gold mining, combustion in power plants, industry and other sectors, industrial processes, the waste sector, transport, as well Hg-specific sectors like cremation, Hg mining and VCM production.

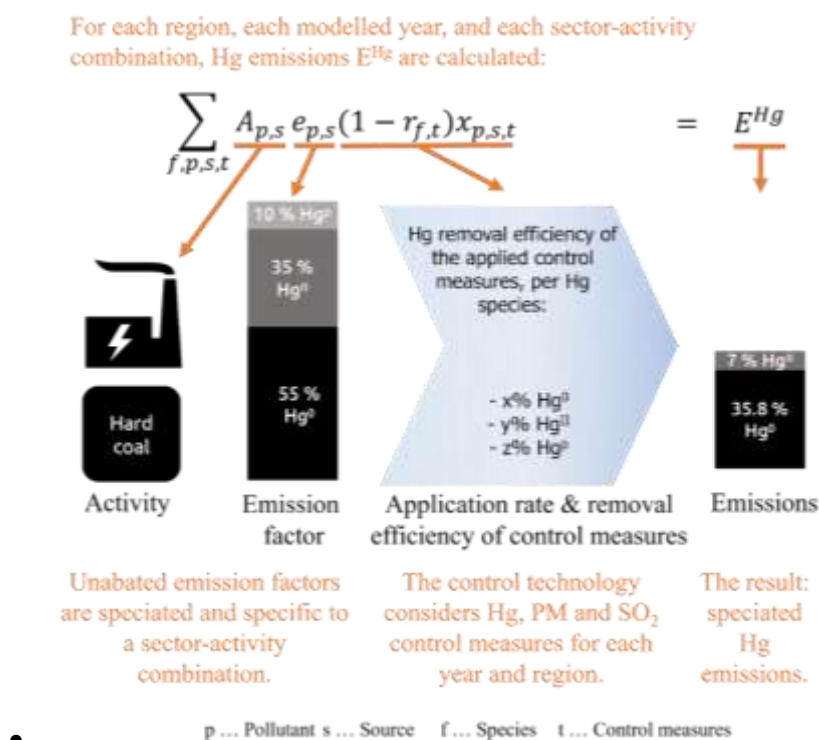


Figure 23. The GAINS model methodology of calculating future Hg emissions, based on the activity, emission factor, and control technology application of an anthropogenic Hg emission source. Source: Brocza et al. 2024, DOI: 10.5194/acp-24-7385-2024.

With regards to its ability to represent the MC provisions, most but not all sources in Annex D can be explicitly represented in GAINS (see Table 8). Annex B sources are only partially covered by the modelling and those modelled have clear Hg reduction targets and phase-out dates, which are modelled in the baseline scenarios. Trajectories of Annex C (ASGM) are defined by the percentage reduction targets as defined in the MC National Action Plans. Annex D sources (except incineration of specialized waste) are well represented in modelling and thus can be analysed on the source-level. Article 8 does not require a phase-out of these activities but instead mandates the use of Best Available Techniques (BAT) and Best Environmental Practices (BEP) for new sources. Their definition is quite flexible and parties to the MCM must themselves formulate appropriate steps to reduce Hg emissions from existing sources. Such steps may include the quantification of existing emissions, co-benefit reductions from other air pollution controls or Hg-specific measures, but parties may choose to install dedicated Hg control equipment. Different scenarios were designed to visualize the “option space” of the MC emission reductions, with relation to a ‘Current legislation’ scenario (01_BAS_CLE), a scenario where all regions maximize their ‘traditional’ air pollution control of PM_{2.5} and SO₂ abatement technologies, which in turn maximizes co-benefits for Hg (03_BAS_coMFR), a scenario which takes into account the National Action Plans on ASGM (02_BAS_MINA) for relevant countries, and, as an end point for technical Hg control, a scenario which computes emissions when maximum feasible reduction of Hg is achieved by technical means, disregarding current policy or cost (04_BAS_HgMFR). Three energy scenarios delineate how energy policy affects Hg emissions for the energy and related industry sectors.

Table 7.

Scenario name	Energy policy	Clean air policy (Hg, PM _{2.5} , SO ₂)	Minamata Convention
00_BAS_NOC	Baseline scenario, following WEO 2022 State Policies	Hypothetical baseline of unabated emissions. No PM, SO ₂ or Hg controls implemented.	

01_BAS_CLE	scenario trends (IEA, 2022). Global energy demand growth met mostly by renewables; share of fossil fuels in global energy mix falls to < 75 % by 2030, 60 % by 2050.	Current legislation of Hg, PM and SO ₂	Annex B implemented Annex C: no NAP implementation Annex D: no additional ambition to status quo;
02_BAS_MINA	Global energy-related CO ₂ emissions plateau at 37 Gt and fall to 32 Gt in 2050, leading to 2.5 °C global average temperature rise by 2100.	Current legislation of PM and SO ₂	Annex B implemented Annex C: NAP implementation Annex D: no additional ambition to status quo;
03_BAS_coMFR		Maximum technically feasible reduction of PM and SO ₂	Annex B implemented Annex C: no NAP implementation Annex D: ambitious implementation through maximizing of co-benefit Hg reduction
04_BAS_HgMFR		Maximum technically feasible reduction of PM and SO ₂ , as well as Hg across all sectors (including those not covered by MC)	Annex B implemented Annex C: no NAP implementation Annex D: ambitious implementation through maximizing Hg-specific pollution control application
05_CLIM1_CLE	Climate Policy scenario (CLIM1): Adapted from WEO 2022 Announced Pledges (AP) scenario (IEA, 2022). Demand for all fossil fuels declines by 2030. CO ₂ emissions fall to 12 Gt in 2050, leading to 1.7 °C global average temperature rise by 2100.	Current legislation of Hg, PM and SO ₂	Annex B implemented Annex C: no NAP implementation Annex D: no additional ambition to status quo;
06_CLIM2_CLE	Net Zero scenario (CLIM2): Adapted from WEO 2022 Net Zero Emissions scenario (IEA, 2022). CO ₂ emissions fall to 23 Gt in 2030 before	Current legislation of Hg, PM and SO ₂	Annex B implemented Annex C: no NAP implementation Annex D: no additional ambition to status quo;

07_CLIM2_MFR	reaching 0 Gt in 2050, leading to < 1.5 °C in 2100.	Maximum technically feasible reduction of PM and SO ₂ , as well as Hg across all sectors (including those not covered by MC)	Annex B implemented Annex C: no NAP implementation Annex D: ambitious implementation through maximizing Hg-specific pollution control application

Table 8. Scenario definition with regards to clean air policy, energy policy and Minamata Convention (MC) provisions.

Source and process categories	Convention provisions	GAINS representation
Extraction and use of fuels/energy sources		
Coal combustion in power plants	Article 8, Annex D	COMB_POWER
Coal combustion in coal fired industrial boilers	Article 8, Annex D	COMB_IND
Primary (virgin) metal production		
Gold (and silver) extraction with mercury amalgamation processes	Article 7, Annex C	ASGM
Gold extraction and initial processing by other methods	Article 8, Annex D	GOLD
Zinc extraction and initial processing	Article 8, Annex D	NFME
Copper extraction and initial processing	Article 8, Annex D	NFME
Lead extraction and initial processing	Article 8, Annex D	NFME
Primary (virgin) metal production		
Cement and clinker production	Article 8, Annex D	CEM
Intentional use of mercury in industrial processes		
Chlor-alkali production with mercury-technology	Article 5, Annex B	OTHER (Subsector 'PR_CSP')
Vinyl chloride monomer production with mercury catalyst	Article 5, Annex B	OTHER (Subsector OTHER_HG)
Acetaldehyde production with mercury catalyst	Article 5, Annex B	not represented
Other production of chemicals and polymers with mercury	Article 5, Annex B	not represented
Waste incineration		
Incineration of municipal/general waste	Article 8, Annex D	OTHER (WASTE)
Incineration of hazardous waste	Article 8, Annex D	not represented
Incineration of medical waste	Article 8, Annex D	not represented
Sewage sludge incineration	Article 8, Annex D	not represented
Other emission sources		
E.g. transport emissions, various industrial processes, domestic coal burning	not covered	NOT_COVERED

Answer

Expected development of Hg emissions until 2050. The 01_BAS_CLE scenarios project stagnant Hg emissions until 2050, considering the uncertainty inherent in different sectors. Despite emission decreases from the power sectors - due to a lower carbon intensity in power generation, as well as lower Hg emission intensity following stringent Chinese policy -, Increased waste generation and the inherent uncertainty in ASGM emissions compensate this. Considering the 02_BAS_MINA scenario and 03_BAS_coMFR, it can be seen that the projected emission decrease from the implementation of the NAPs that have been published until August 2024 is comparable to total achievable emission reductions achievable from the implementation of co-benefit controls globally, at around -18% relative to 2015 emissions (Fig. 8).

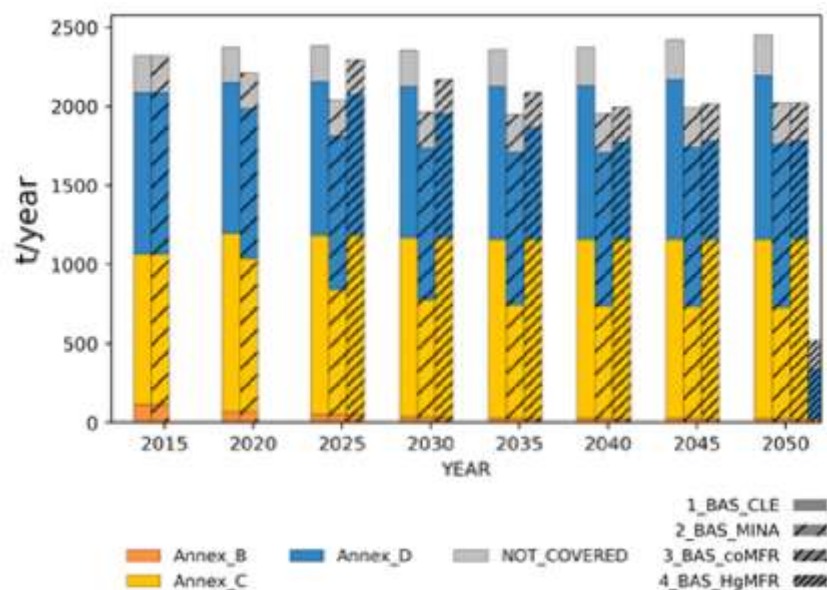


Figure 24. Global anthropogenic Hg emission scenarios following the BAS energy scenario, from 2015 - 2050 (t/year). Emissions are split according to the Minamata Convention (MC) Annex that they are subject to. 'NOT_COVERED' signifies emissions computed by GAINS that are not covered by the MC, as listed in Table 8.

Regional differences. Regional analyses show that the world regions are markedly different in their total emissions and main emission sources, thus, their emission trends will depend on vastly different policy and emission drivers. In Africa, Central and South America as well as Southeast Asia, the notoriously ill-constrained ASGM emissions make up the majority of emissions, so the energy transition present in climate scenarios barely impacts these regions in the present modelling. Similarly, the present modelling indicates that the macroeconomic developments in the Middle East and North America have a strong impact on Other emissions, which are dominated by landfill emissions. This is another region where the modelling uncertainty is high. Emissions in Europe, North America, China and India are dominated by sectors with lower uncertainties in emission factors and activity levels, thus there is a high level of confidence in total emission levels and a clear relationship between energy policy and air pollution control technology choices and Hg emission levels exists. In China, emissions are projected to decline already in the baseline scenario, driven by decarbonisation and the stringent application of PM and SO₂ controls. Conversely, Indian Hg emissions from fossil fuel combustion (COMB_POWER) are expected to rise despite lower Hg emissions from the power sector, as other industry-related emissions and waste emissions are projected to rise. Under more stringent climate pathways, and considering that ambitious air pollution control leading to co-benefits is already mandated by current legislation coming into force in the next decade, Hg emissions are expected to decouple further from CO₂ emissions.

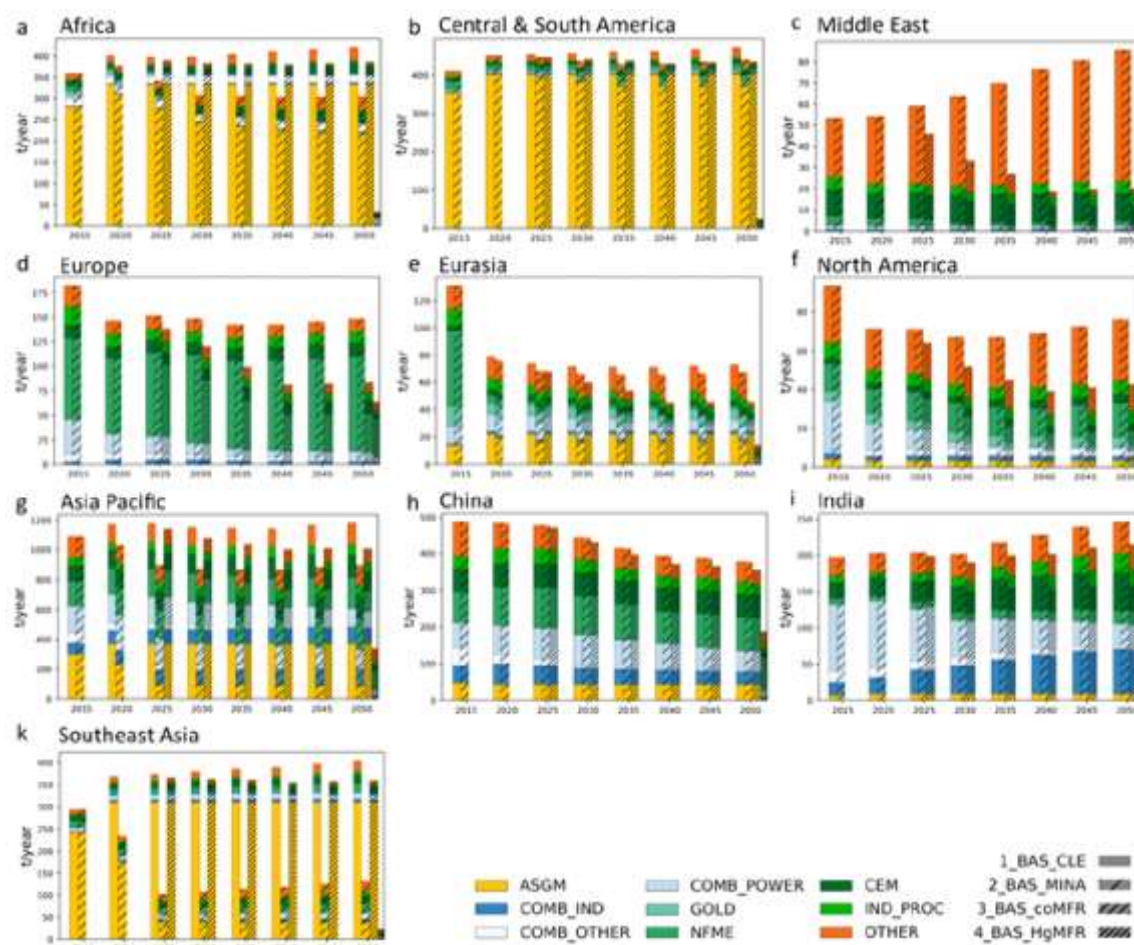


Figure 25. Regional anthropogenic Hg emission scenarios following the BAS energy scenario, from 2015 - 2050 (t/year).

Consistency of answer with existing literature

Geyman et al. (2024) have explored anthropogenic Hg emissions until 2300, including Hg deposition on land and water (releases) under different SSP pathways. This analysis uses a different approach to inferring the future emission intensity of different emission sources, which does not reference policy such as the MC in an explicit way, but infers that different world regions approach the world region with the lowest emission intensity in 2010 of a certain sector by 2100. (Geyman et al. 2024, DOI: 10.1029/2023EF004231). Similarly to this study, Geyman et al. find that “decoupling of Hg and CO₂ emission sources occurs under low-to mid-range scenarios, though contributions from artisanal and small-scale gold mining remain uncertain.”.

Confidence

- High confidence in emission intensity trajectories and the effect of policy on Hg emission sources related to fossil fuel and biomass combustion, as well as industry sectors.
- High uncertainties related to emission intensities and future development of drivers of ASGM, as well as waste sectors.

Information and Capacity Needs

While Hg emissions from combustion sources are well defined through detailed energy scenarios and numerous studies on Hg emission factors and the effect of different control technologies, the same level of detail in currently available scenarios is not available for several important industry sectors, the waste sector, and ASGM, where the Hg levels in the raw materials and Hg releases, as well as the effects of different abatement technologies are highly variable under local conditions and modelled at a lower level of detail that might not aid policy decisions.

There is a need for capacity building in Hg waste modelling, better representing regional Hg levels, as well as the end-of-life modelling of Hg-containing products, which would aid the modelling representation of MC Annex B source emissions.

Further, currently available policy scenarios do not fully take into account the circularity aspect of the global bioeconomy, which could lead to lower emission intensities e.g. in NFME production and also to improved waste management and subsequently lower Hg emissions. Such scenarios would further aid the policy discussion.

10.3.9 How are secondary Hg emissions and releases from biomass burning, soils and ocean expected to change in future under changing anthropogenic Hg emissions and releases and environmental conditions across global regions?

Currently no information is available on this question.

10.3.10 What are the fractional contributions of primary (anthropogenic and natural) and secondary Hg emissions and releases to current Hg levels observed in the air and ocean across global regions?

Currently no information is available on this question.

10.3.11 How have the fractional contributions of primary (anthropogenic and natural) and secondary Hg emissions and releases to levels of Hg observed in the air and ocean changed over the period circa 2010 to present across global regions?

Currently no information is available on this question.

10.3.12 What is the fractional contribution of anthropogenic Hg emissions influenced by the Convention to current Hg levels observed in the air and ocean across global regions?

Currently no information is available on this question.

10.3.13 How has the fractional contribution of anthropogenic Hg emissions influenced by the Convention to Hg levels observed in the air and ocean changed over the period circa 2010 to present across global regions?

What is the fractional contribution of changes in anthropogenic Hg emissions influenced by the Convention to changes in Hg levels observed in air and deposition over the period circa 2010 to present across global regions?

MA-15

Information used

We use the BASE, NATURE, and ASGM scenarios from MCHgMAP model ensemble simulations to calculate the contribution in anthropogenic emissions influenced by the Convention to atmospheric Hg(0) trends.

Methodology

The BASE simulation includes anthropogenic emissions and environmental forcing conditions varying between 2010–2020, the NATURE simulation assumes anthropogenic Hg emissions fixed at 2010 values but with all other factors varying, the ASGM simulation includes ASGM levels fixed at 2010 values but all other anthropogenic sources varying. The natural contribution is defined as the trend in the NATURE simulation, the anthropogenic contribution is defined as the BASE simulation trend – NATURE simulation trend, the ASGM contribution is defined as the BASE simulation trend – ASGM simulation trend, and the non-ASGM anthropogenic contribution is defined as the ASGM simulation trend – NATURE simulation trend.

Answer

Model experiments strongly suggest that during the timeline of the Minamata Convention air Hg concentrations have declined in NH (in the regions where observations are available, NA, EU, Arctic, EA), due to the decline in primary anthropogenic plus associated secondary land and ocean emissions (by about 300 Mg yr^{-1}), showing the effectiveness of the Convention. This is evidenced by the agreement of the SCAv4 simulation with atmospheric observations (Figs. 10 and 11). We have conducted attribution simulations (ASGM, NATURE) to assess the drivers of trends, yet these simulations are based on the BASE inventory that does not capture the observed decrease in $\text{Hg}(0)$ concentrations. Hence, it is difficult to make conclusions about specific sources. Nevertheless, based on these experiments we can see that ASGM sources dominate the anthropogenic trend contribution between 2010–2020 in the BASE simulation with EDGAR v8.1 emissions (Fig. 14). The ASGM emissions potentially explain the increasing $\text{Hg}(0)$ in the Southern Hemisphere which is observed in atmospheric measurements (Figs. 10 and 11). The non-ASGM anthropogenic contribution is consistent with the direction of observed trends in the Northern Hemisphere (Fig. 14), although the overall BASE trend still underestimates the decline in the observations (Figs. 10 and 11). The next steps for this analysis would be to conduct further source attribution experiments based on the SCAv4 simulation and to better understand the discrepancies between the available bottom-up inventories and emissions trends derived from available $\text{Hg}(0)$ measurements.

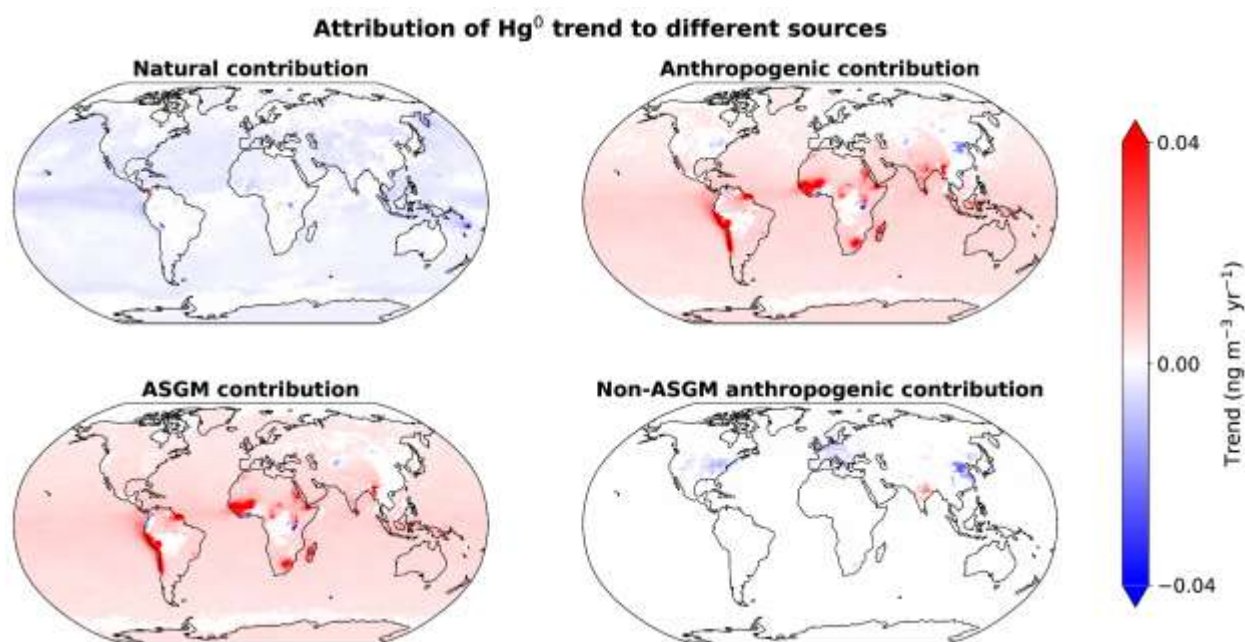


Figure 14. Spatial distribution of modelled $\text{Hg}(0)$ trend contributions from different drivers over 2010–2020. The anthropogenic contribution is calculated using the EDGARv8.1 emissions dataset. Ensemble mean model simulations and observations (sites > 8 years) are deseasonalized and analyzed with quantile regression for the median trends in monthly mean values. Only grid boxes with trends significantly different from zero ($\text{trend} < 2 \times \text{trend error}$) are shown in colour.

Consistency with existing literature

The mercury literature includes many examples of source apportionment centered on a specific analysis year (Lei et al., 2013; Chen et al., 2014; Dastoor et al., 2015; De Simone et al., 2017; De Simone et al., 2020). For the most part, this source attribution was done using geographic source regions rather than source sectors. In the

2018 Global Mercury Assessment, source attribution was conducted on both a geographical as well as sectoral basis. Again, the focus was on a single year (2015). Therefore, the current approach to calculate sectoral contributions to Hg(0) trends between 2010 and 2020 has limited analogues in the available literature, although it is consistent with the strong ASGM contribution to the Southern Hemisphere and tropics (UNEP, 2019). The SCAv4 scenario showing that observed trends are consistent with an emissions decrease of $\sim 300 \text{ Mg yr}^{-1}$ is consistent with a previous analysis that found that anthropogenic emissions between 2005 and 2020 must have declined by at least 140 Mg yr^{-1} in the Northern Hemisphere to match observed trends (Feinberg et al., 2024).

Confidence

The contributions of ASGM emissions to Hg(0) trends are highly uncertain, since long-term observation sites are not located near ASGM regions. In addition, the available emissions inventories are not consistent with observed Hg(0) trends (see comparison of observations to BASE simulation in Fig. 10). The SCAv4 scenario is just one potential scenario of many that could match the observed trends (Feinberg et al., 2024), and it is not clear which sources need to be adjusted or added in the EDGAR v8.1 inventory to explain the gap between inventories and measurements. Thus, it remains challenging to attribute the decreasing Hg(0) and wet deposition trends to specific source sectors or regions.

Information and Capacity Needs

In the near term, additional simulations could be conducted to assess the source contributions to trends on a sectoral basis relevant to the Minamata Convention (e.g., Annex D sources). These simulations could be analyzed to understand the potential roles of specific sources covered by the Minamata Convention as drivers of trends. However, the refinement of bottom-up emissions inventory data would also be crucial; specifically ensuring that bottom-up emissions inventories are consistent with available observations of atmospheric Hg trends. In addition, to further understand the role of the Minamata Convention in Hg emissions trends, it would be necessary to create emissions scenarios that represent a Minamata-avoided case (i.e., with air pollution control technology implemented for SO₂ and NO_x but no specific Hg measures). The uncertainty of these assessments would be further constrained by expanding long-term atmospheric Hg monitoring efforts, especially in regions strongly affected by ASGM sources as this is currently lacking.

10.3.14 What are the fractional contributions of changes in primary anthropogenic and natural Hg emissions and releases, and environmental conditions to changes in Hg levels observed in the air and ocean over the period circa 2010 to present across global regions?

Currently no information is available on this question.

10.3.15 What is the fractional contribution of changes in anthropogenic Hg emissions influenced by the Convention to changes in Hg levels observed in the air and ocean over the period circa 2010 to present across global regions?

Currently no information is available on this question.

10.3.16 In each world region, how much of the contribution of primary anthropogenic Hg emissions to changes in Hg levels observed in air over the period circa 2010 to present are due to emissions within the region versus outside the region?

Currently no information is available on this question.

10.3.17 How are Hg levels in the air and ocean expected to change in future under different scenarios of implementation of the Minamata and other Conventions across global regions?

Currently no information is available on this question.

10.3.18 How are Hg levels in the air and ocean expected to change in future under changing environmental conditions across global regions?

Currently no information is available on this question.

10.4 Process and Source Attribution for Biota, Humans, and Other Media

10.4.1 What is the contribution of changes in primary anthropogenic emissions and releases to changes in Hg levels observed in biota over the period circa 2010 to present? (local environment, lower trophic levels,)

Information Used (including applicable indigenous and traditional knowledge)

There is no straightforward information to answer this question.

Two case studies. IATT-2 “Integrated Analysis Study of Mercury in Tropical Tuna Species” provided by Rober Mason, et al., United States, and GL-1 “Great Lakes Regional Case Study” provided by Mark Burton and Celia Chen, United States introduced trends in tropical tuna species in the global ocean and cross media observation of trends and patterns in Great Lakes, respectively.

We show the outcomes of the case studies as examples of how this question could be answered in the context of the case studies.

Summary of referred case studies

IATT-2

Integrated Analysis Study of Mercury in Tropical Tuna Species

Information used and methodology

Global monitoring data for ocean tuna, emissions, and modeling data.

Answer/response

In the global ocean, the stability of Hg concentrations in tropical tunas in all oceans between 1971 and 2022 was revealed by the Médiéu et al. (2024) study, which contrasts with the significant decrease in global primary anthropogenic Hg emission, concentration and deposition trends (AMAP and UNEP, 2019). While surface waters rapidly equilibrate with the atmosphere, deeper oceanic waters have a longer residence time, so that Hg accumulated there takes longer to be redistributed to shallower waters where tunas live and feed. Data from models included in the current OESG analysis of ocean data can be used to further examine the notion of changes in concentration over time. The trends since 2010 in the mean MeHg (MMHg in Figure 3) concentrations for the tropical Pacific Ocean surface waters (30°N to 30°S) show no obvious trend. While there is seasonal variability and a large spread in the modeled data, there is no consistent trend in concentration over the time period (Bieser, pers. comm.). Such modelling illustrated that even if Hg emissions into the atmosphere were drastically reduced, it would take almost 10 years to detect a fall in Hg concentrations in surface waters, and approximately 25 years in subsurface waters

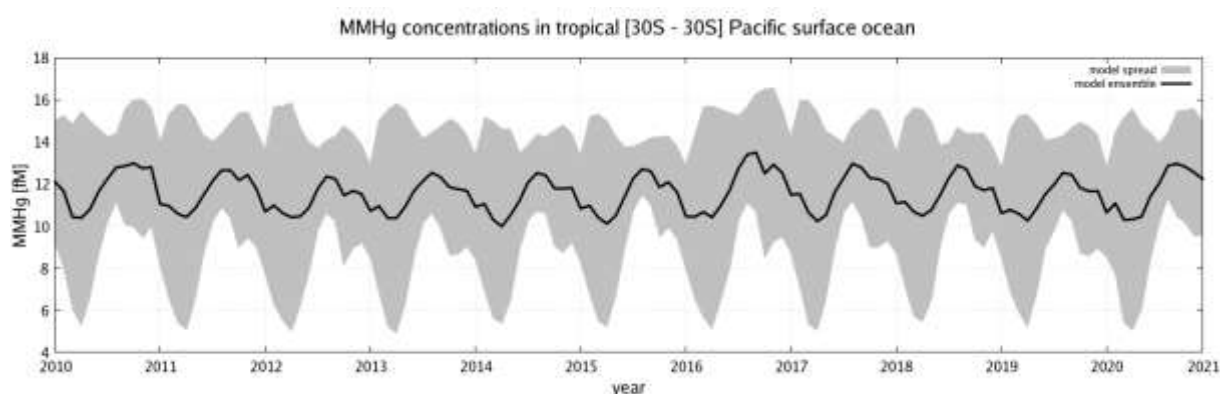


Figure 3: Model output for surface layer methyl-mercury (MMHg) concentrations (fM) for the period 2010 to 2021 for the tropical regions of the Pacific Ocean (30°N to 30°S; Bieser, pers. comm.)

GL-1

Great Lakes Regional Case Study

“Great Lakes Regional Case Study” shows integrated cross media observation of trends and patterns in Great Lakes.

Information used

Atmospheric mercury emission data in US (USEPA (2021))

Deposition data from MDN in Great Lakes basin.

Fish monitoring data from US and Canada

Methodology

Statistical analysis of trends, comparison of trends across media and lakes

Answer/response

The decline in atmospheric mercury emissions from anthropogenic sources in the United States.

Mercury deposition patterns do not exhibit corresponding monotonic declines consistent with local to regional scale trends in mercury emissions.

A significant decreasing trend in annual mercury deposition was found in the Lake Ontario basin, while a significant positive trend was detected in the Lake Huron basin (**Figure 1**, $p < 0.05$). No significant trends were observed across the Lake Erie, Lake Michigan, and Lake Superior basins.

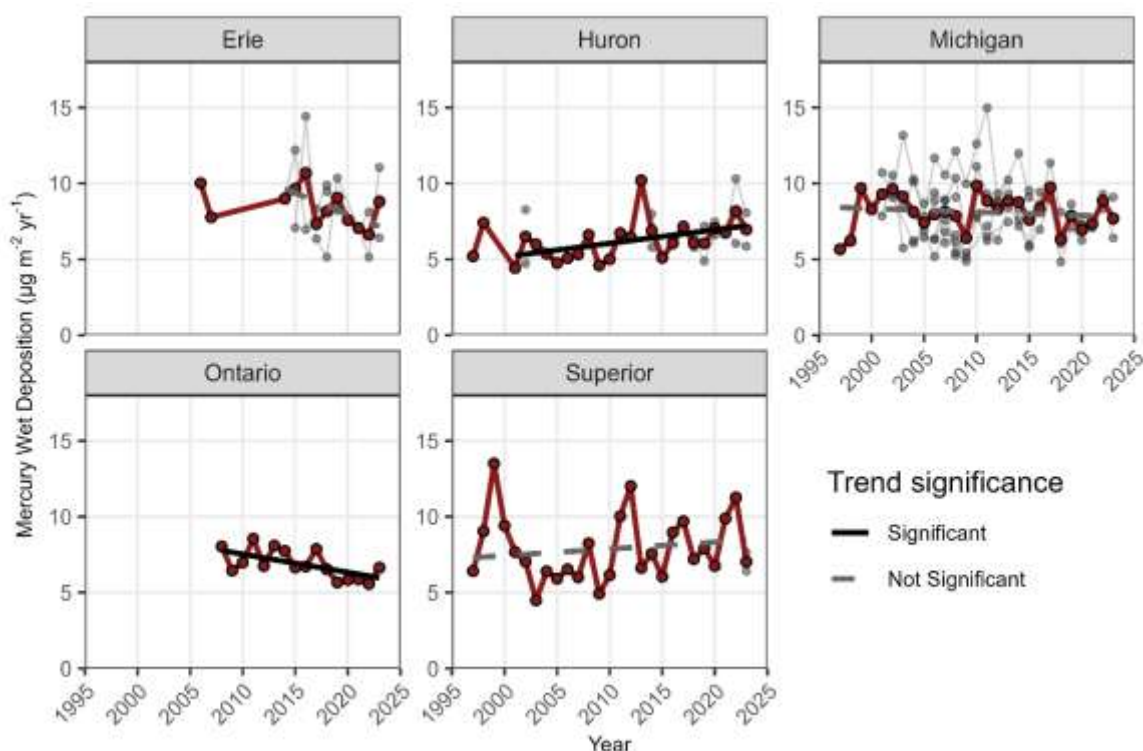


Figure 1. Annual mercury deposition for MDN sites in the Great Lakes basins from 1997-2023. Light gray lines and points are individual MDN stations within each Great Lake basin. Red lines and points are the mean annual mercury deposition for all the stations within the basin. Trend lines were derived using the Mann–Kendall test and Sen’s slope estimator; solid black lines denote statistically significant trends ($p < 0.05$), and dashed gray lines indicate non-significant trends. Trends were calculated and displayed only for the range of consecutive years with measurements

Mercury concentrations in fish muscle varied significantly across lakes and through time, with multiple significant interactions between year, lake, and species, highlighting the spatial and temporal complexity of mercury trends in the Great Lakes (**Figure 2**).

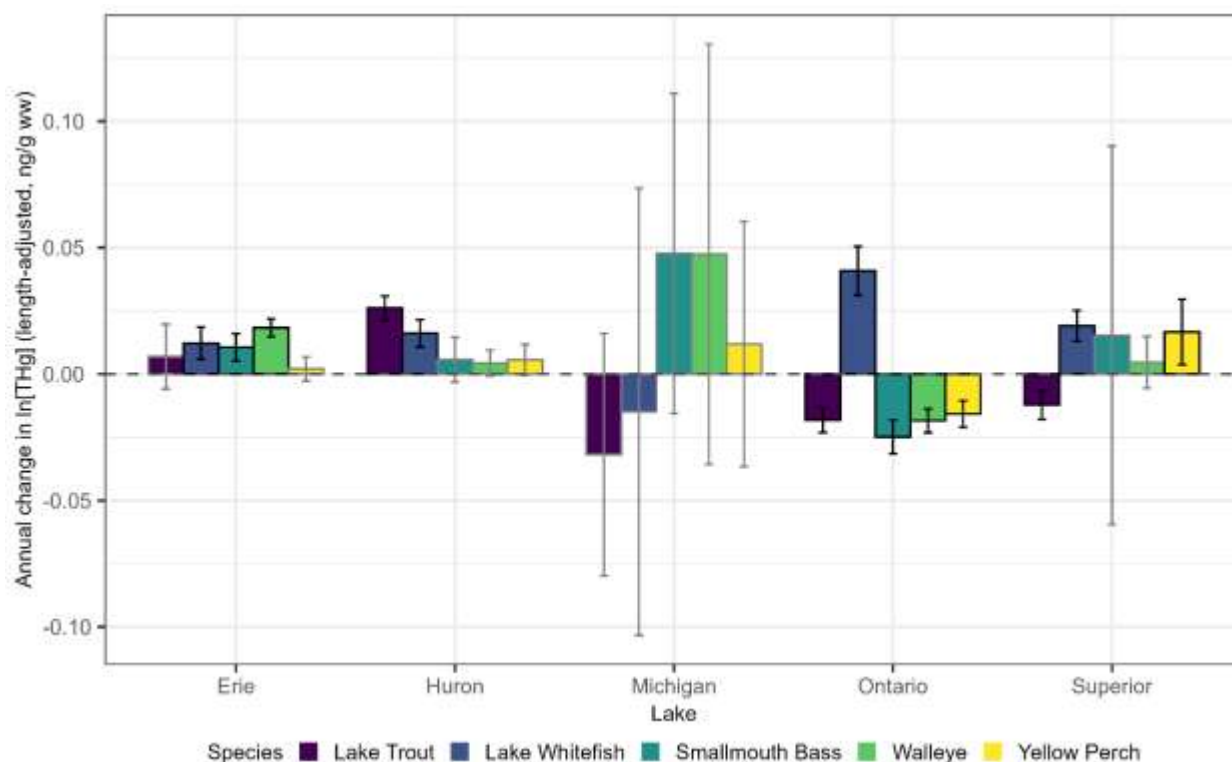


Figure 2. Estimated annual trends in length-adjusted total mercury concentrations (ln[THg], ng/g muscle wet weight) in five fish species from the five Great Lakes. Temporal trends were estimated using a linear mixed-effects model with fixed effects for Year, Lake, Species, and their interaction, and including sampling site as a random effect. Bars show model-estimated slopes for each combination of species and lakes with lines for 95% confidence intervals. Black outlines denote statistically significant trends ($p < 0.05$), while gray outlines denote non-significant trends. Only the five most-sampled species were modelled

Relevance to effectiveness evaluation

The decline of atmospheric emissions of mercury in the region, and nations, does not appear to be mirrored in changes in mercury deposition in each of the Great Lake basins or in the temporal trends in concentrations in fish tissue which decrease, increase, or do not change depending on the species and the lake. This suggests that there is very little linkage between atmospheric concentrations and fish tissue, which has also been documented in the literature (Lepak et al. 2025). This makes it more difficult to decipher the effectiveness of decreases in mercury emissions and releases in other compartments besides the atmosphere.

Gaps, limitations and recommendations

Based on this analysis and past studies on the Great Lakes region, cross-media integration of Hg data is complex given the different scales at which mercury is emitted, transported, and deposited as well as the differences in biogeochemical and ecological characteristics of each lake. Gaps and limitations exists in technical aspects of sampling protocols, and also complicated mechanisms among media and species in Great Lakes to hinder establishment of simple cross-media integration.

10.4.2 What is the contribution of changes in anthropogenic, natural and legacy mercury in Hg levels observed in humans over the period circa 2010 to present?

No information is available on this question.

10.4.3 What is the contribution of anthropogenic, natural, or legacy sources to the spatial and temporal trends in soil, sediments, surface and ocean waters?

Information Used (including applicable indigenous and traditional knowledge)

There is no straightforward information to answer this question. A regional case study may give part of answer on the question in the context of regional study.

Methodology

MEDT-2 “Mercury in the Mediterranean Region and the Minamata Convention” provided by Milena Horvat, et al., Slovenia, introduced comprehensive summary of cross-media monitoring data on air, water and sediments, biota, and human biomonitoring together with the discussion of cross-media integration. As the study showed spatial trend of various media in the region and temporal trend in some monitoring examples, we could have implications to consider further about the possible contribution of various sources to the pattern of mercury in the region.

Answer/Response

Case study MEDT-2 may not give direct answer to the question, however, spatial pattern and temporal trends well studied in the region would be the basis to argue contribution of anthropogenic, natural, or legacy sources to the levels in the region in future.

Consistency of Answer with Existing Literature (especially the 2018 GMA)

It is difficult to argue consistency between the global assessment and regional case studies.

Evaluation of Confidence

Information shown in the case studies are confident, at least in the context of each case studies. General comments in Answer/Response may need future evaluation of confidence.

Information and Capacity Needs

Long term monitoring and appropriate interpretation are necessary to assess the effectiveness of the convention further.

Linkages among regional case studies and global analysis may also be necessary for further study.

Summary of referred case study

MEDT-2

Description of the case study

The Mediterranean is a region where the environment and everyday life are deeply connected. With more than 20 countries bordering its shores and nearly half a billion people living in its catchment, it is a place where seafood is not only a key source of food and income but also a cornerstone of culture. Yet this same connection to the sea brings challenges: mercury (Hg) contamination has long been a concern.

Part of the reason lies in the sea’s semi-enclosed nature — water circulates slowly, so pollutants tend to remain and build up over time. Another factor is the region’s complex mix of natural and human-related mercury sources. Volcanic activity and hydrothermal vents release mercury naturally, while centuries of mining, industrial development, and energy production have added further pressures.

Figure 1. *Mercury hotspots in the Mediterranean*

Figure 1 illustrates major Hg contamination hotspots across the Mediterranean region, together with their primary sources of pollution. The distribution highlights the so-called Mediterranean Hg belt, which includes areas affected by historical Hg mining, industrial discharges, and petrochemical activities. Notable sites include the Almadén mine (Spain), historically the world's largest source of mercury, and Idrija (Slovenia), where past mining activities contributed to Hg accumulation in the Gulf of Trieste and the Marano–Grado Lagoon. Additional industrial sources are located along the Rhône River delta and in northern Italy (Livorno, Piombino, Mestre, Ravenna), linked to chlor-alkali production and petrochemical industries. In southern Europe and North Africa, hotspots include Milazzo and Augusta (Sicily), Monteponi (Sardinia), and several coastal sites in Tunisia and Algeria (e.g., Bizerte, Gulf of Gabès, Skikda Bay), where mining, fertilizer production, and petrochemical activities have released substantial mercury loads. Eastern Mediterranean locations, such as Kaštela Bay, the Neretva River delta, Iskenderun Bay, and Haifa Bay, further reflect contamination from chemical industries, agriculture, metallurgy, and fossil fuel use. Collectively, the map emphasizes the widespread but heterogeneous distribution of mercury sources, with legacies of mining and industry continuing to shape Hg inputs to Mediterranean coastal and marine environments.

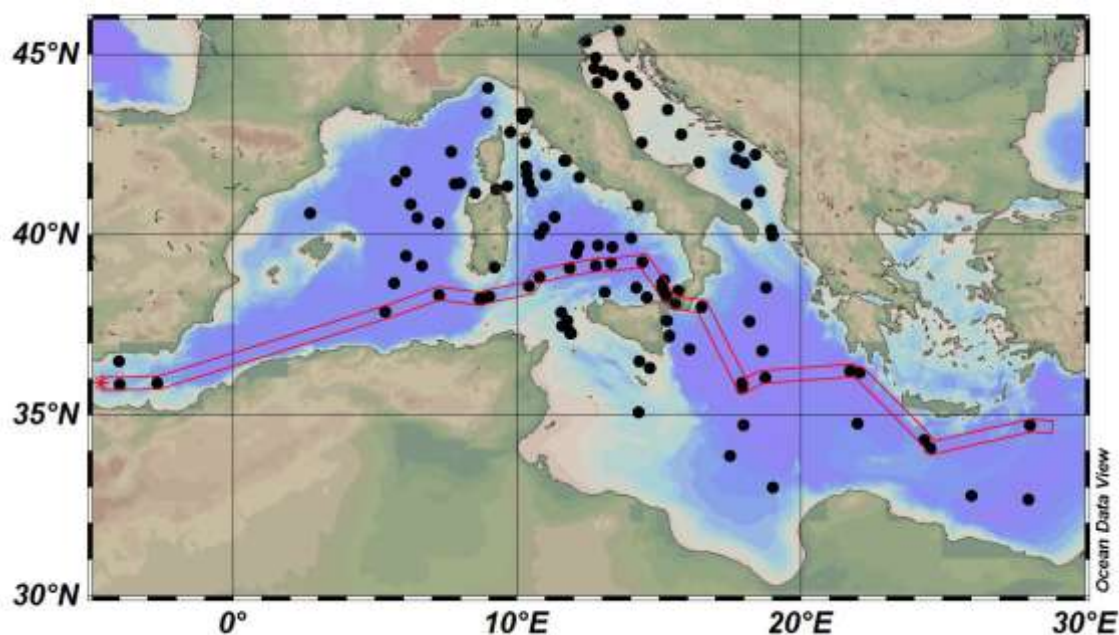
This comprehensive case study on Mediterranean region includes information on regional intergovernmental agreements supporting mercury monitoring and control in the region, available data in air, multi-model ensemble simulations, seawater, sediments, biota and human biomonitoring. The study summarizes the comprehensive reviews on multi-media monitoring data and their spatial and temporal trends in the region, factors affecting mercury levels and fate, and mass-balance within and to/from outside regions.

Cross-media integration of observed levels/trends

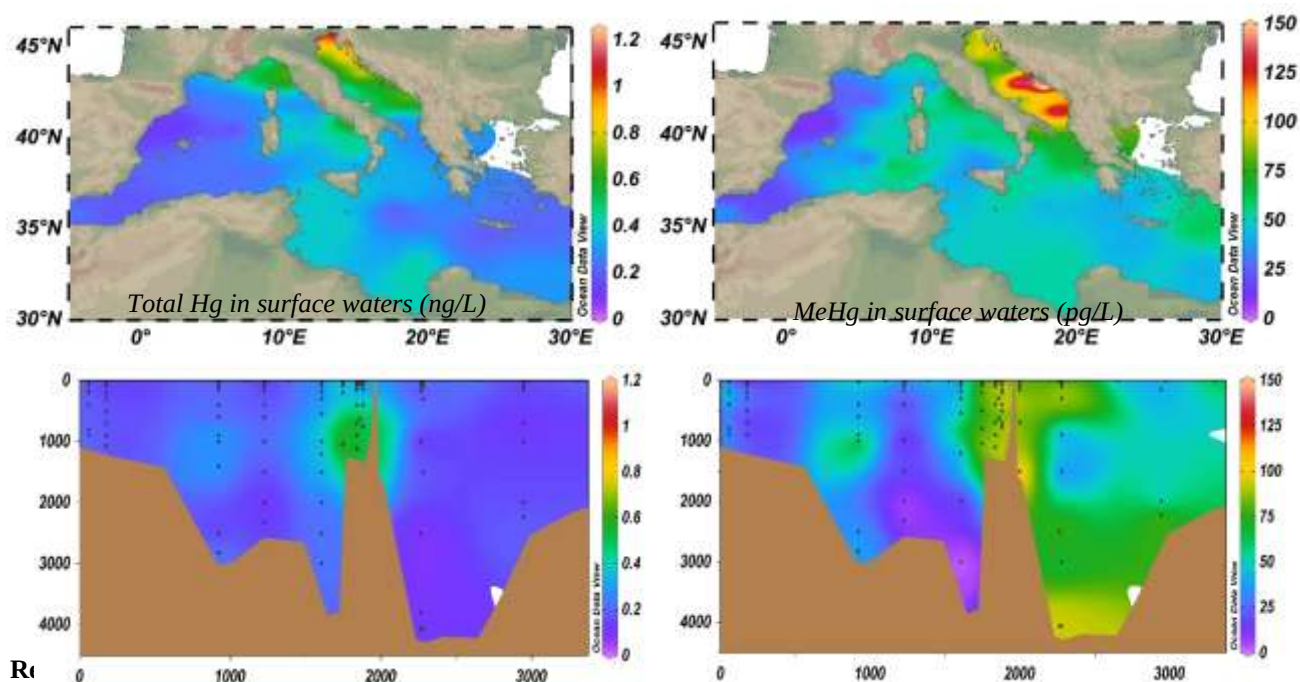
The study reports mass balances of mercury among media and regions in Mediterranean region as summarized in Figure 11 in the report (see Appendix-?).

The report argues the spatial distribution of the total and methyl mercury levels in the region, as shown in the Figure 3 below.

Figure 3: Spatial distribution of total mercury (THg) and methylmercury (MeHg) in Mediterranean seawater during the CASP ship campaigns (2000–2015), highlighting hotspots in the Gulf of Trieste, Adriatic, and western basin (Kotnik et al., 2014).



Note: Sampling locations of surficial and vertical profiles. All GMOS cruises together 2000 – 2015. The red line shown in Figure 3 marks a representative transect where vertical depth profiles of total mercury (THg) and methylmercury (MeHg) were collected. These depth profiles are presented in the following figures, providing detailed insight into how concentrations



R₁

The Mediterranean is a responsive test bed for Article 22: short effective residence times and tightly coupled compartments allow policy signals to emerge across media. Independent budgets converge on a **net atmospheric source** and **net Atlantic export**, and the Gibraltar sections document **multi-decadal declines** consistent with global emissions controls. Because speciation is explicit (MeHg, DGM), we can link observed **biogeochemical reactivity** to outcomes that matter—**fish Hg** and **human exposure**. As local point sources

wane (e.g., chlor-alkali, dental amalgam), **diet becomes the dominant exposure pathway**; thus, **targeted, species/portion-specific advisories** become the practical lever to protect vulnerable groups while preserving seafood benefits. Together, these features make the Mediterranean a **demonstration region** where cross-media indicators can track Convention effectiveness with credibility.

Gaps, limitations and recommendations

Current knowledge on mercury in the Mediterranean is constrained by several significant gaps and limitations. Geographic and trophic coverage remains incomplete, with especially limited data available from the eastern and southern Mediterranean. Time series are scarce, particularly for methylmercury (MeHg) and boundary exchange processes, making it difficult to establish long-term trends and variability. Coordinated human biomonitoring (HBM) efforts are limited across high-consumption communities, restricting the ability to assess exposure patterns in populations at greatest risk.

Submarine groundwater discharge and hydrothermal fluxes remain highly uncertain, yet they may represent important sources of mercury input. Speciation and microbial cycling are not routinely measured, even though these processes are central to mercury transformation and bioavailability. In addition, methodological differences across environmental media hinder cross-integration of datasets and complicate the establishment of coherent regional assessments.

10.4.4 How can the spatial and temporal trends in soil, sediment, surface and ocean waters be attributed to changes in anthropogenic sources of mercury at the global, regional and local scale over the observed period?

No information is available to answer this question.

10.4.5 What is the contribution of trends in drivers of human exposure related to implementation of the Convention to the trends observed in Hg levels in humans over the period circa 2010 to present?

No information is available to answer this question.

10.4.6 What is the contribution of environmental processes and pressures (including climate, habitat changes) to the trends observed in Hg levels in biota over the period circa 2010 to present (does not include anthropogenic inputs)?

Information Used (including applicable indigenous and traditional knowledge)

There is no straightforward information to answer this question.

A case study. IATT-3 “Integrated Analysis Study of Mercury in Tropical Tuna Species” provided by Rober Mason, et al., United States, introduced information of tropical tunas and albacore in the Pacific Ocean.

We show the outcomes of the case study as an example of how this question could be answered in the context of the case study.

IATT-3

Integrated Analysis Study of Mercury in Tropical Tuna Species

Case study: Tropical tunas and albacore in the Pacific Ocean

Information used

Variety of studies regarding Hg levels in tuna and environmental, physiological and oceanological parameters.

Answer/response

Mercury concentrations significantly differ among tropical tunas and albacore (*T. alalunga*), following the general pattern: bigeye > albacore > yellowfin and skipjack (Médieu et al., 2023). Tuna Hg concentrations also vary spatially, as highlighted regionally in the eastern Pacific (Ferriss and Essington, 2011) and the western Indian (Chouvelon et al., 2017), and across 12 locations in the global ocean (Nicklisch et al., 2017). These recent broad-scale and transdisciplinary studies lead to the hypothesis that the spatial variability in tuna Hg concentrations is mainly explained by both tuna foraging depth and the variability of MeHg formation and bioavailability in the ocean. By revealing the importance of ecological and biogeochemical processes, these studies support the assumption that climate change will likely affect MeHg concentrations in marine food webs.

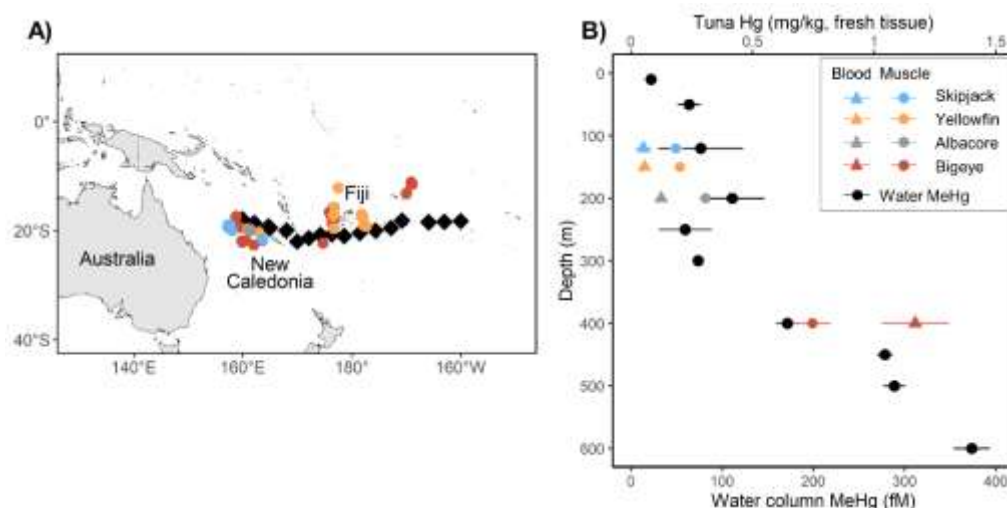


Figure 5. The importance of tuna foraging depth in the context of marine methylmercury (MeHg) depth profiles in the southwestern Pacific Ocean. A) Location of seawater (black diamonds) and tuna (colored dots) blood and muscle samples used to investigate the link between tuna Hg concentrations and dissolved MeHg levels in water. B) Relationship between total Hg concentrations (mg/kg, fresh tissue) in tuna species exhibiting different foraging depths, and concentration-depth profile of dissolved MeHg (fM), concentration-depth profiles were averaged from all locations), adapted from Barbosa et al., (2022).

10.4.7 What is the contribution of trends in drivers of human exposure unrelated to implementation of the Convention to the trends observed in Hg levels in humans over the period circa 2010 to present?

No information is available to answer this question.

10.4.8 How can the spatial and temporal trends in soil concentrations be attributed to changes that are not influenced by the Convention?

No information is available to answer this question.

10.4.9 How can the spatial and temporal trends in sediments concentrations be attributed to changes that are not influenced by the Convention?

No information is available to answer this question.

10.4.10 How can the spatial and temporal trends in surface concentrations be attributed to changes that are not influenced by the Convention?

No information is available to answer this question.