

Mercury as a Global Pollutant: Sources, Pathways, and Effects

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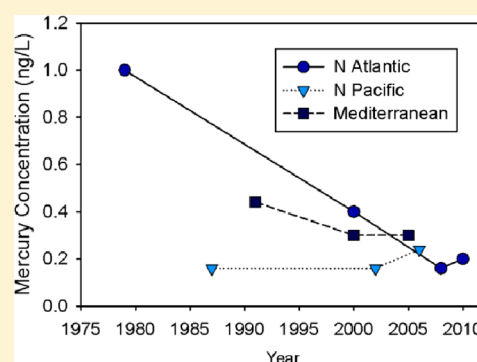
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ABSTRACT: Mercury (Hg) is a global pollutant that affects human and ecosystem health. We synthesize understanding of sources, atmosphere-land-ocean Hg dynamics and health effects, and consider the implications of Hg-control policies. Primary anthropogenic Hg emissions greatly exceed natural geogenic sources, resulting in increases in Hg reservoirs and subsequent secondary Hg emissions that facilitate its global distribution. The ultimate fate of emitted Hg is primarily recalcitrant soil pools and deep ocean waters and sediments. Transfers of Hg emissions to largely unavailable reservoirs occur over the time scale of centuries, and are primarily mediated through atmospheric exchanges of wet/dry deposition and evasion from vegetation, soil organic matter and ocean surfaces. A key link between inorganic Hg inputs and exposure of humans and wildlife is the net production of methylmercury, which occurs mainly in reducing zones in freshwater, terrestrial, and coastal environments, and the subsurface ocean. Elevated human exposure to methylmercury primarily results from consumption of estuarine and marine fish. Developing fetuses are most at risk from this neurotoxin but health effects of highly exposed populations and wildlife are also a concern. Integration of Hg science with national and international policy efforts is needed to target efforts and evaluate efficacy.



INTRODUCTION

The socio-economic consequences of mercury (Hg) contamination and exposure of humans and wildlife to methyl Hg (CH_3Hg) have prompted policies and regulations to limit Hg releases from human activities and to control its transport within and among the atmosphere, freshwaters, the lithosphere, and the ocean at local, regional, and global scales.^{1–4} This synthesis paper is an integration of plenary presentations given at the 2011 meeting of the *International Conference of Mercury as a Global Pollutant* in Halifax, Canada. The overall theme of the conference was “Mercury in Air, Land, Sea and Me”. As such we organize this synthesis around Hg health effects; Hg dynamics in the atmosphere, land and the oceans; and the implications for Hg policy; the topics of the conference plenary sessions. We synthesize the current understanding of Hg and CH_3Hg sources, transport and fate globally, and discuss the potential variation of CH_3Hg exposure to changes in Hg emissions and other environmental perturbations, and impacts on human health and environmental quality. We discuss the most current knowledge on (1) the scale of Hg emission impacts (local-regional-global); (2) the relative importance of geogenic vs anthropogenic emission sources; (3) the recovery response time for ecosystems after emission controls (years-decades-centuries); and (4) the chronic low dose health effects of dietary exposure of CH_3Hg on human populations. We

consider the implications of this knowledge and remaining research gaps for Hg control policies (legally binding or voluntary actions) at regional and global scales.

Mercury is a naturally occurring element, but has been directly mobilized by humans for thousands of years into aquatic and terrestrial ecosystems through mining, the use of Hg in precious metal extraction, its presence as a trace contaminant in many materials (e.g., coal, metal ores), and its use in products (e.g., paint, electronic devices) and by industry (chlor-alkali plants; as a catalyst).^{5–8}

The atmosphere is the foremost transport pathway of Hg emissions, whereas land and ocean processes play an important role in the redistribution of Hg in terrestrial, freshwater, and marine ecosystems and the production of CH_3Hg that drives the major human exposure route, consumption of fish, particularly marine fish. The temporal and spatial scales of Hg transport in the atmosphere and its transfer to aquatic and terrestrial ecosystems depend primarily on its chemical and physical forms. Following emission, elemental Hg ($\text{Hg}(0)$) can be transported long distances before oxidation and removal by

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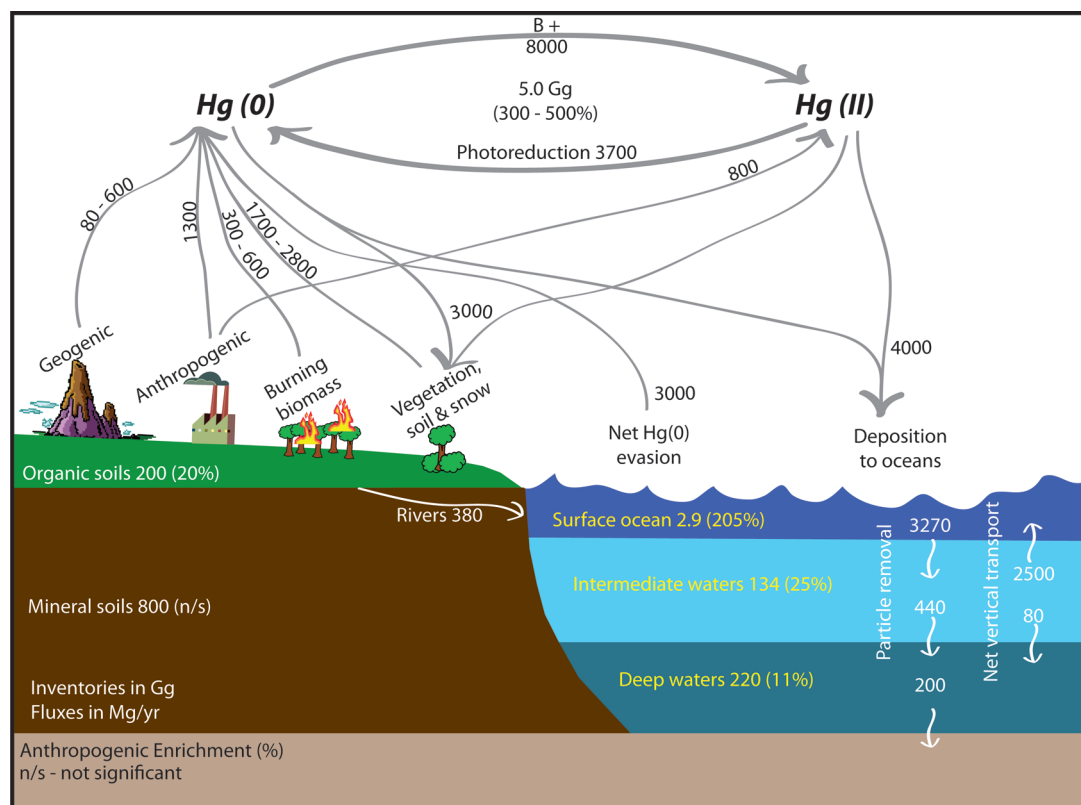


Figure 1. Current estimates of the fluxes and pools of mercury at the Earth's surface derived on this work, and building on previous studies.^{99,140,148,181,183,193} $Hg(II)$ includes both gaseous and particulate forms, plus a negligible contribution (1Mg) from inert particulate mercury. The percentages in brackets are estimated increases in pools and fluxes due to anthropogenic activities over the past 150 years. Fluxes are in $Mg\ yr^{-1}$ and reservoirs are given in Gg.

particle and gas-phase dry deposition or scavenging by precipitation. The atmospheric residence time of $Hg(0)$ is several months to a year.^{3,6,9,10} Mercury can therefore be transported and deposited to remote locations such as the Arctic and Antarctic.¹¹⁻¹⁹ Reactive gaseous Hg (RGM) and particulate bound ionic Hg ($Hg(II)$) have a shorter atmospheric residence time than $Hg(0)$ (atmospheric residence time hours to days), and as a result are generally deposited locally or regionally. Inputs to ecosystems largely occur as $Hg(II)$, while most CH_3Hg is produced within ecosystems.

It is important to distinguish between *primary* and *secondary* emission sources of Hg . Primary sources, both natural and anthropogenic, transfer Hg from long-lived lithospheric reservoirs to the atmosphere. This Hg deposits to land and oceans. Deposited Hg can be reduced to $Hg(0)$ and then re-emitted. Re-emission processes represent *secondary* sources that exchange Hg among surface reservoirs using the atmosphere as a vehicle. Primary sources increase the global pool of Hg in surface reservoirs, while secondary sources redistribute it among and within ecosystems.

Recent global assessments and associated modeling have improved understanding of Hg emissions from both primary and secondary sources,^{6,8,20-22} as well as their atmospheric transport and deposition.^{2,17,23-28} Overall, total global Hg emissions to the atmosphere range from 6500 to 8200 $Mg\ yr^{-1}$, of which 4600 to 5300 $Mg\ yr^{-1}$ are from natural processes and sources (primary geogenic plus secondary emissions). Primary anthropogenic sources release 1900–2900 $Mg\ yr^{-1}$ compared with primary natural (geogenic) inputs of 80–600 $Mg\ yr^{-1}$

(Figure 1). East Asia currently accounts for almost 40% of total primary global anthropogenic emissions of Hg (Figure 2).^{6,23}

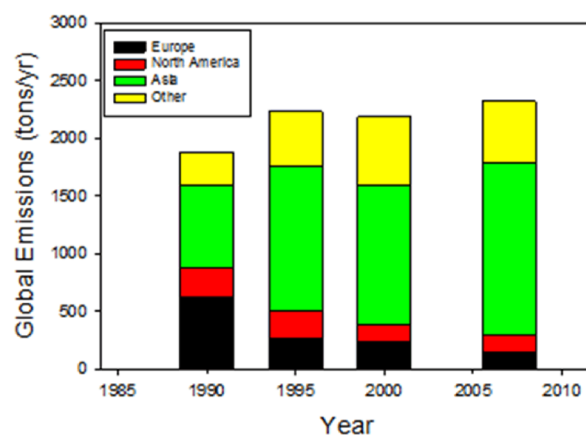


Figure 2. Trends in global emissions of mercury; note emissions in Asia are increasing while Europe and North America are decreasing.^{6,21}

Impacts of Hg have been assessed for three potential future anthropogenic greenhouse gas (GHG) emission scenarios under the Intergovernmental Panel on Climate Change Special Report on Emission Scenarios (IPCC SRES).^{7,8} Under most GHG scenarios, primary Hg emissions will increase unless emission controls are widely implemented. The main driver for increased Hg emissions is the expansion of coal-fired electricity generation in the developing world, particularly Asia, and the

degree to which control technologies are implemented. An evaluation of these scenarios by Sundseth et al.²⁹ confirmed the high socio-economic cost of these projected increases in emissions. They estimate that by 2020, the annual cost of the associated decrease in IQ would reach US \$3.7 billion (2005 dollars).

An important distinction between Hg and most other atmospheric pollutants is that environmental and health impacts are only indirectly related to ambient atmospheric concentrations of Hg(II) or Hg(0). Effects and toxicity result from the net conversion of Hg(II) to the more bioaccumulative CH₃Hg, a process that generally occurs under reducing conditions in wetlands and sediments in watersheds and coastal zones, and in the upper ocean (see below). Exposure to humans and wildlife overwhelmingly occurs by the consumption of CH₃Hg contaminated ocean, freshwater and terrestrial organisms.^{1,30–33} As a result, the impact of Hg on ecosystem health is related not only to the magnitude of regional and global emissions and deposition, but also the potential for watersheds and the oceans to convert Hg(II) to CH₃Hg; of food webs to biomagnify CH₃Hg; and the processes and ecosystem disturbances that alter net methylation and trophic transfer.^{34,35}

Global concern of widespread CH₃Hg contamination has motivated international initiatives to assess current knowledge on atmospheric Hg emissions, transport and its effects on terrestrial and aquatic ecosystems. These include activities of the United Nations Environment Programme (UNEP) (www.unep.org/PDF/PressReleases/GlobalMercuryAssessment2013.pdf) and that of the UNECE-LRTAP Task Force on Hemispheric Transport of Air Pollution (TF HTAP).^{36,37} The UNEP initiative is aimed at decreasing human health and environmental risk from the release of Hg, as well as improving the understanding of international Hg emissions and their transport and fate.^{2,17} An international treaty (Minamata Convention) was signed in January 2013 (www.unep.org/hazardoussubstances/mercury/Negotiations/INC5/INC5PressReleases/tabid/106835/Default.aspx) to control the global release of Hg to the environment. An observational infrastructure system is also being developed to monitor Hg in the atmosphere, and marine and terrestrial ecosystems through international Hg agreements³⁸ involving partnerships among regional Hg monitoring programs. Such information will improve understanding of the global impacts of Hg pollution and help evaluate the effectiveness of national and international policies.

■ EXPOSURE AND HEALTH OF HUMANS AND WILDLIFE

Exposure Pathways and Toxicology. The driver of most environmental Hg and CH₃Hg studies is the perception that human health risk from CH₃Hg exposure has increased with industrialization and the increasing cumulative release of Hg to the environment from anthropogenic sources. CH₃Hg concentrations are elevated in fish and shellfish, particularly piscivorous fish such as tuna. Dietary consumption of marine fish and other seafood is a major route of CH₃Hg exposure among human populations with many populations dependent on fish for food, protein, and nutrients. The contribution of fish to global diets is increasing (17 kg cap⁻¹ yr⁻¹ in 2008).³⁹ In 2007, fish was 15.7% of the global intake of animal protein and 6.1% of all protein consumed. Globally, fish provides >1.5 billion people with ~20% of their average per capita intake of

animal protein, and 3.0 billion with at least 15%. China has the highest total food fish supply (38.4 million tonnes; 32.4% of the global supply) and Japan has the highest per capita consumption (56.7 kg cap⁻¹ yr⁻¹ in 2007).⁴⁰ The widespread contamination of fish by CH₃Hg, therefore, is a threat to human health on a global scale. Further, recent studies from southeastern China suggest that rice grown in soils near artisanal Hg mining and receiving elevated atmospheric Hg deposition may also accumulate elevated CH₃Hg.⁴¹ More research is needed on CH₃Hg in rice as an exposure pathway.

In addition to the global commercial fishery, CH₃Hg exposure can be important at the “local consumer” scale. Local consumers include recreational anglers who eat their catch; people who rely on local marine fish and marine mammals for a majority of their protein and nutrition; immigrant communities who catch their own fish and may have different eating habits such as consuming whole fish instead of fillets; and consumers who prefer to eat local seafood. Indigenous peoples around the world that rely on local fish catch for subsistence living can also be at risk of elevated Hg exposure.⁴² Studies have shown elevated exposure of local populations who consume local fish or animals. Either global Hg sources⁴³ or a combination of global and local sources^{44,45} can be important in such locations.

Numerous studies and databases exist on total Hg or CH₃Hg concentrations in fish.⁴ As Hg in the muscle of piscivorous fish is predominately CH₃Hg,⁴⁶ total Hg measurement is often used as a surrogate for CH₃Hg in fish. A comprehensive review of both gray and published literature of Hg concentrations in 36 species of fish ($n = 34\,300$) showed that (1) fish such as tuna and swordfish have the highest CH₃Hg concentrations; (2) mackerel and sardines from the Mediterranean Sea contain more CH₃Hg compared to other harvesting areas; (3) mollusks and crustaceans have relatively low CH₃Hg concentrations (below 100 ng g⁻¹);⁴⁷ and (4) within-species variability in concentration often exceeds the interspecies variability.⁴⁸ Note that fish Hg data are limited for most of the world, highlighting the need for local data to inform exposure assessments as is being done by international organizations.⁴⁹

The toxicology of Hg and the environmental health implications of CH₃Hg exposure have been extensively reviewed.^{50,51} Recent research has advanced the understanding of the dose–response relationship between exposure to CH₃Hg and its toxic effects. Information has been derived from observations of the CH₃Hg poisoning incident in Minamata, Japan, where more than 2000 people consumed CH₃Hg contaminated fish (total Hg concentrations >10 µg g⁻¹) and suffered from neurological disorders collectively known as Minamata Disease.⁵² Pregnant women in Minamata had exposure levels 27 times higher than reference areas,⁵³ equivalent to 40 µg g⁻¹ in hair. In comparison, the mean hair Hg concentrations among women in the U.S. was 0.2 µg g⁻¹ in 1999–2000⁵⁴ and the mean maternal hair concentration of the Hg exposure cohort in the Faroe Islands was 4.27 µg g⁻¹ in 1986–1987.⁵⁵ In Minamata, fetuses exposed to CH₃Hg through the placenta displayed severe cerebral palsy-like symptoms, while their mothers had mild or no manifestations of poisoning. The high levels of pre- or postnatal exposure to CH₃Hg were found to cause long-term psychiatric symptoms in adults including impairment of intelligence and mood, and behavioral dysfunction.⁵⁶

Current global Hg exposure is at levels much lower than conditions at Minamata. However, there is convincing evidence

of adverse neurological/neurodevelopmental outcomes in infants and young children associated with CH₃Hg exposure during fetal development as a result of maternal fish consumption during pregnancy. Three major epidemiological studies of neurobehavioral development in children have been conducted in populations consuming fish/seafood with no known local industrial contamination: (1) the Seychelles Islands (median maternal hair concentration at $6.6 \mu\text{g g}^{-1}$ ($n = 779$);⁵⁷ (2) the Faroe Islands (mean maternal hair $4.27 \mu\text{g g}^{-1}$ ($n = 900$);⁵⁵ and (3) the New Zealand study (mean maternal hair concentration at $8.3 \mu\text{g g}^{-1}$ ($n = 38$)).⁵⁸ Adverse neurodevelopment effects were identified in the Faroe Islands at a benchmark dose (BMDL) of $12\text{--}15 \mu\text{g g}^{-1}$ Hg in maternal hair. In contrast, the New Zealand study found a BMDL at $4\text{--}6 \mu\text{g g}^{-1}$ Hg and for the Seychelles Islands at $17\text{--}25 \mu\text{g g}^{-1}$ Hg.⁵⁹

Long-term study of the Seychelles Child Development Study cohort at age 17 years found no consistent pattern of adverse associations between prenatal MeHg exposure and detailed domain specific neurocognitive and behavioral testing.⁶⁰ On the other hand, impact of prenatal CH₃Hg exposure on neurobehavioral function was found to be multifocal and permanent among the children of the Faroe cohort as similar associations between CH₃Hg exposure and deficits in motor, attention, and verbal tests were found in children at age 7 years and 14 years.⁶¹ Sustained attention of the 14 year old children was found to be the neuropsychological domain that is particularly vulnerable to developmental CH₃Hg exposure, indicating probable underlying dysfunction of the frontal lobes.⁶² The discrepancies among studies may partly be explained by variability in long chained polyunsaturated fatty acids (LCPUFA) in the fish consumed which counter the adverse effects of CH₃Hg.⁶³ A follow up cohort study of 229 mother-child pairs in the Seychelles Child Development Nutrition Study found that maternal fish consumption during pregnancy exposed the fetus simultaneously to CH₃Hg and LCPUFA which had significant but opposite effects on child development at 9 and 30 months of age.⁶⁴ Moreover, at 5 years of age, the positive association of LCPUFA remained significant while the adverse associations between MeHg exposure and developmental outcomes was no longer observed.⁶⁵ These results demonstrate the importance confounding effects of the maternal nutritional status in determining the risk of prenatal CH₃Hg exposure.

Using the data from these three studies, a dose-response relationship between maternal hair Hg concentrations and child IQ was established suggesting a loss of 0.18 IQ points (95% confidence interval, -0.378 to -0.009) for each $\mu\text{g g}^{-1}$ increase of maternal hair Hg.⁶⁶ Based on the three studies, the U.S. Environmental Protection Agency (EPA) derived a reference dose (RfD) for CH₃Hg of $0.1 \mu\text{g kg body weight}^{-1} \text{d}^{-1}$ as the daily intake likely to preclude appreciable risk of deleterious effects during a lifetime, assuming a total uncertainty factor (UF) of 10 to account for variability and uncertainty.⁵⁹ Between 3 and 15% of women of child bearing age in the U.S.⁶⁷ and 27.7% in Korea⁶⁸ exceed this RfD indicating the scale of the public health problem.

Risks and Benefits of Fish Consumption. A review of epidemiological studies on effects of low levels of CH₃Hg exposure (maternal hair Hg less than $4 \mu\text{g g}^{-1}$) conducted in Poland, Spain, Korea, France, the U.S., Austria, Canada, and the United Kingdom found possible effects on fetal growth among susceptible subgroups and on infant growth in the first two years of life.⁶⁹ There was also evidence of early childhood

neurological effects at low levels of prenatal CH₃Hg exposure that may differ by age, gender, and timing of exposure. The authors found no conclusive evidence on effects on the risk of cardiovascular disease or immune functions.

The dose response relationship between low dose CH₃Hg exposure and effects in adults is not as clear. The lowest observable effect was reported in the Tapajós River basin (Brazil), a major Amazon tributary impacted by traditional gold mining since the mid 1980s. Alteration of motor performance and visual dysfunction was found in adults whose mean Hg hair concentration was above $10 \mu\text{g g}^{-1}$.⁷⁰ Interestingly, it appeared that selenium from dietary sources could mitigate the adverse effects of CH₃Hg on motor functions in some Amazon populations.⁷¹

Health effects such as cardiac function from CH₃Hg exposure have received recent attention. Mechanistic evidence and experimental animal toxicological studies suggest that CH₃Hg can be a risk factor for cardiovascular disease (CVD).⁷² However, epidemiological evidence is inconclusive.⁷³ Roman et al.⁷⁴ considered the current epidemiological literature sufficiently robust to support the development of a dose-response function between CH₃Hg exposure and acute myocardial infarction. However, results of two major cohort studies in the U.S. found no evidence of any clinically relevant adverse effects of Hg exposure on coronary heart disease, stroke, or total CVD disease in adults.⁷⁵ The public health implications are potentially significant, as CVD is the leading cause for mortality in most developed countries. For example, the expected monetary value of the annual health benefits associated with decreases in fatal heart attacks due to a 10% decrease in Hg exposure of the U.S. population would be 4 times higher than that associated with IQ gains.⁷⁶ Therefore, it is important to improve characterization of the potential linkage between CH₃Hg exposure and the risk of CVD.

Nutritional benefits are an important consideration in risk assessment of fish consumption.⁷⁷ Essential polyunsaturated fatty acids, such as docosahexaenoic acid (DHA), confer benefits to brain and visual system development in infants and reduce risks of certain forms of heart disease in adults which mitigate against CH₃Hg effects. Advisories issued with the aim of decreasing Hg exposure can also result in substantial reductions in healthful DHA.⁷⁸ Facing this dilemma of risk and benefit of fish consumption, the FAO and WHO held an Expert Consultation Workshop in 2011.⁴⁸ Using multiple national databases of Hg and fatty acids concentrations in commercial fish species, they compared the benefits of n-3 polyunsaturated fatty acids with the risks of CH₃Hg among women of childbearing age, concluding that maternal fish consumption lowers the risk of suboptimal neurodevelopment in their offspring compared with the offspring of women not eating fish in most circumstances evaluated. This integrative approach of benefits and risks assessment has been successfully applied to estimate optimal fish consumption level in The Netherlands,⁷⁹ Finland,⁸⁰ and Sweden.⁸¹ Given the nutritional benefits of fish, it is important to continue to decrease Hg sources with a goal of achieving fish Hg concentrations for commonly consumed species of $<0.3 \mu\text{g g}^{-1}$. In the interim, it is important to work with public health agencies to improve fish consumption advisories and communication strategies to minimize risks and maximize benefits from consumption of low-Hg fish.

Hg Exposure to Wildlife. While concern of exposure of CH₃Hg is focused on human health, research is increasingly demonstrating that elevated CH₃Hg also impacts the health of

fish and other wildlife. Studies in the mid-1990s suggested that concentrations of CH_3Hg in the range of $5\text{--}10\ \mu\text{g g}^{-1}$ (wet weight) were necessary for sublethal and lethal effects on fish.⁸² However, recent studies have shown that whole body concentrations at $0.3\ \mu\text{g g}^{-1}$ and muscle tissue concentrations of $0.5\ \mu\text{g g}^{-1}$ compromise fish reproduction, embryonic development, change biochemical process, and cause damage to cells and tissues.⁸³ These levels are in the range commonly reported for top predator fish⁸⁴ and suggest that the health of fisheries may be impaired due to CH_3Hg contamination.

Elevated concentrations of CH_3Hg have been observed in piscivorous birds (common loon, *Gavia immer*;⁸⁵ wading birds;⁸⁶ bald eagles, *Haliaeetus leucocephalus*⁸⁷) and mammals (mink, *Mustela vison*; river otter, *Lontra Canadensis*⁸⁸). Although a relatively insensitive species, there is considerable understanding of Hg effects on the common loon. Exposure to elevated CH_3Hg causes hormonal changes, impairment of motor skill and decreases in reproduction. As a result, a wildlife criterion of $3.0\ \mu\text{g g}^{-1}$ has been established for common loon blood.⁸⁹ This blood concentration value is comparable to that observed in eastern North America.^{85,90} In addition to piscivores, insectivorous songbirds and bats, particularly those associated with wetland habitats, have been shown to have elevated Hg.^{91,92} Using the Carolina wren (*Thryothorus ludovicianus*) as a model system of CH_3Hg effects of Hg songbirds, Jackson et al.⁹³ found that nesting success (i.e., the ability to fledge at least one offspring) decreased as the parents CH_3Hg exposure increased, with a 10% or more nest failure when females had blood Hg of $0.7\ \mu\text{g g}^{-1}$, 20% failure at blood Hg of $1.2\ \mu\text{g g}^{-1}$ and 30% failure at blood Hg of $1.7\ \mu\text{g g}^{-1}$.

Fish eating mammals such as mink are affected by Hg exposure at dietary concentrations as low as $0.1\ \mu\text{g g}^{-1}$. Concentration-dependent decreases in N-methyl-D-aspartic acid (NMDA) receptors were found in the basal ganglia, cerebellum, brain stem, and occipital cortex of wild and captive mink in Canada.⁹⁴ Similar effects on NMDA receptors were found in the brain stem of polar bears (*Ursus maritimus*).⁹⁵ High Hg concentrations were reported in the brain of other apex predators in the Arctic ecosystem such as beluga whale (*Delphinapterus leucas*).⁹⁶ These findings are of physiological and ecological concern because they demonstrate that at ambient concentrations, Hg can affect the health of key species of wildlife in diverse ecosystems.

AIR

The atmosphere is a major pathway for the biogeochemical cycle of Hg. Background atmospheric concentrations typically range from 1.3 to $1.7\ \text{ng m}^{-3}$ in the Northern Hemisphere and from 1.1 to $1.3\ \text{ng m}^{-3}$ in the Southern Hemisphere.^{9,13,17,97} Data from lake sediments, peat cores, and ice cores all indicate that atmospheric Hg deposition has increased globally by about a factor of 3 since preindustrial times (~ 1850), with larger increases in regions receiving elevated deposition from regional sources.^{13,98} On a global scale net atmospheric Hg deposition is determined by the overall emissions, whereas deposition at the local scale is controlled by atmospheric processes and the speciation of regional and local emissions. Understanding these processes is therefore critical for the development of accurate predictive models that inform global Hg policy. Models of atmospheric Hg must accurately depict four basic processes: emissions, transport, redox chemistry, and deposition.

Emissions. Estimates of Hg emissions to the atmosphere are depicted in Figure 1, based on Holmes et al.⁹⁹ Primary

emissions include natural geogenic sources (volcanism and erosion; estimated at $500\ \text{Mg a}^{-1}$) and anthropogenic sources (fuel combustion, waste incineration, and mining; $2100\ \text{Mg a}^{-1}$). Primary emission sources due to human activity have increased over geogenic sources by a factor of 2–15, which is larger than the aforementioned factor of 3 average increase in Hg deposition (Figure 1). This discrepancy reinforces the fact that the Earth system (including recycling through surface reservoirs) is not in steady-state, and that Hg in surface reservoirs will continue to increase even if future anthropogenic emissions remain constant.¹⁰⁰

Noncombustion processes emit Hg as $\text{Hg}(0)$, but combustion processes (including fuel use, waste incineration, open fires) emit about half of their Hg as $\text{Hg}(\text{II})$.²² This partitioning of emissions is of considerable importance because the emitted $\text{Hg}(\text{II})$ can be deposited regionally downwind of the source, resulting in a local deposition enhancement in contrast to the global influence associated with emissions of $\text{Hg}(0)$. Emission inventories also generally include a particulate component of Hg ($\text{Hg}(\text{p})$) in addition to $\text{Hg}(0)$ and RGM. $\text{Hg}(\text{p})$ is best viewed as part of $\text{Hg}(\text{II})$, with gas-particle partitioning of $\text{Hg}(\text{II})$ occurring in the atmosphere on the basis of local particle concentrations and temperature.¹⁰¹

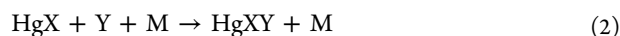
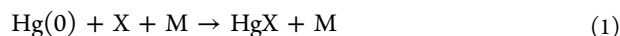
Transport. Hg transport in the atmosphere is rapid compared to other environmental reservoirs, enabling the dispersal and exchange of Hg in the global environment. The time scale for global mixing of the troposphere is about 1 year, limited principally by exchange of air between the two hemispheres. Surface air observations of total gaseous Hg (TGM) show little variability in general, and data from ship cruises indicate 30% higher concentrations in the northern than in the southern hemisphere. The observed atmospheric variability of TGM implies an atmospheric lifetime of 0.5–1 year.^{99,102} While $\text{Hg}(0)$ emitted to the atmosphere is transported efficiently on a hemispheric scale and may be deposited anywhere in its hemisphere of origin, transport between hemispheres is less efficient.¹⁰³ Aircraft observations show that $\text{Hg}(0)$ is well-mixed through the depth of the troposphere, as would be expected since vertical mixing in the troposphere occurs on a time scale of about a month. Concentrations decrease sharply above the tropopause,¹⁰⁴ implying rapid oxidation of $\text{Hg}(0)$ in the lower stratosphere.

Redox Chemistry of Atmospheric Hg. There is considerable uncertainty regarding the atmospheric redox chemistry of Hg. Until about 10 years ago, the conventional view based on experiments and expressed in atmospheric models was that oxidation of $\text{Hg}(0)$ would principally occur by OH and O_3 , and that reduction of $\text{Hg}(\text{II})$ would occur by reaction with $\text{HO}_2(\text{aq})$ in clouds. However, recent laboratory experiments and thermodynamic and kinetic constraints indicate that these reactions are unlikely to be significant in the atmosphere.¹⁰⁵ This understanding has led to a fundamental reassessment of the chemical processes involved in $\text{Hg}(0)/\text{Hg}(\text{II})$ redox chemistry.

Atmospheric observations place some constraints on the processes involved. The diurnal cycle of RGM in surface air, especially in the marine boundary layer (MBL), indicates that the oxidation of $\text{Hg}(0)$ to $\text{Hg}(\text{II})$ must be photochemical.^{106,107} Observations of a weak summer minimum in $\text{Hg}(0)$ concentrations at northern midlatitudes are consistent with this photochemical sink. Mercury depletion events (MDEs) are frequently observed in the Arctic boundary layer in spring.¹⁸ There is good evidence that MDEs are due to rapid oxidation

of Hg(0) by Br atoms released by photochemical reactions following the debromination of sea salt, either in the aerosol or at sea ice surfaces.¹⁰⁸ Similar MDEs have been observed in surface air over the Dead Sea.¹⁰⁹

The halogen atoms are potentially important Hg(0) oxidants in the atmosphere because the overall oxidation reactions are exothermic, involving a two-step process:^{105,110,111}



where $\text{X} \equiv \text{Cl}, \text{Br}, \text{I}$ is a halogen atom and Y is another radical, either a halogen atom or another species such as OH, OH₂, or NO₂.¹¹¹ Both reactions 1 and 2 are three-body processes. The binding energies of the Hg(I) compounds produced in reaction 1 are in the order $\text{HgCl} > \text{HgBr} > \text{HgI}$.¹¹⁰ Cl is thus a particularly efficient oxidant, but its tropospheric concentration is low because reaction with methane results in stable HCl. Oxidation by I atoms may be limited by the stability of HgI.

Holmes et al.¹¹² suggested that Br atoms could provide the dominant global atmospheric oxidant for Hg(0). Known sources of Br atoms to the troposphere include photolysis and oxidation of bromoform (CHBr₃), dimethylbromide (CH₂Br₂), and methyl bromide (CH₃Br), as well as debromination of sea salt aerosol. CHBr₃ and CH₂Br₂ are emitted to the marine biosphere, while CH₃Br has both biogenic and anthropogenic sources. This ensemble of sources produce inorganic bromine (Br₂), which then cycles among its different forms in a manner that is well established from earlier studies of stratospheric chemistry.¹¹³ Unlike HCl, HBr has a lifetime of only a few days against oxidation by OH, providing a sustained pool of reactive Br. Satellite observations indicate a tropospheric background of 0.5–2 ppt BrO¹¹⁴ and this can be reproduced in models including the above sources.^{115,116} Holmes et al.¹¹² found that the implied Br atom concentrations yielded a global atmospheric lifetime of 0.5–1 years for Hg(0). However, the kinetics of the two-step oxidation of Hg(0) by Br have recently been revised^{111,117} and these revisions need to be incorporated in atmospheric Hg models.

Even more uncertainty is associated with atmospheric reduction of Hg(II). Indeed, there is enough uncertainty in the rate of Hg(0) oxidation for global model simulations to account for observational constraints even without reduction of Hg(II).⁹⁹ The depletion of Hg(0) above the tropopause¹⁰⁴ indicates that oxidation cannot take place only by a gas-phase photochemical reaction. Reduction could occur by a condensed-phase photochemical process in lower tropospheric clouds, although there is no evidence to support this process. Observations of a diurnal cycle of Hg(0) in Antarctica by Dommergue et al.¹¹⁸ offer indirect evidence of photochemical reduction of Hg(II) on snow surfaces.

There is some evidence for rapid Hg(II) reduction taking place in power plant plumes. Atmospheric observations in plumes and in polluted regions show much lower Hg(II)/Hg(0) concentration ratios than inferred from emission inventories,^{101,119,120} and observed atmospheric deposition patterns do not show the regional enhancements that would be expected from power plant Hg(II) emissions.¹²¹ The mechanism for reduction is unknown. It cannot be a simple first-order process since there is no indication of rapid Hg(II) reduction in the background atmosphere.

Deposition. Mercury can be removed from the atmosphere by wet or dry deposition. Wet deposition involves scavenging of

gas-phase and aerosol-phase Hg(II). Dry deposition involves surface uptake of both Hg(0) and Hg(II).¹²²

Observations of the diurnal cycle of Hg(0) and RGM on land imply rapid nighttime removal by dry deposition.¹²³ Such data suggest that dry deposition of Hg(0) is a major atmospheric sink of Hg, although the underlying mechanisms and its reversibility remain poorly characterized and quantified. The rapid dry deposition of RGM can be understood in terms of its high water solubility. Dry deposition of Hg(p) depends on particle size, being slow for submicrometer particles but fast for supermicrometer particles. Observations indicate that Hg(p) in continental air is mostly in the submicrometer size range, while Hg(p) in marine air is mostly in the supermicrometer size range of sea salt particles.^{124,125} Efficient uptake of RGM by sea salt particles is expected due to formation of HgCl₃[−] and HgCl₄^{2−} complexes.¹²⁶

Wet deposition provides an efficient sink for Hg(II). However, there is uncertainty regarding the efficiency of scavenging by cold precipitation (snow and ice). Field observations indicate that snow does not effectively scavenge RGM.¹²⁷ Wet deposition flux observations from the Mercury Deposition Network (MDN) in the eastern U.S. show a winter minimum that has been attributed to inefficient scavenging of RGM.¹⁰¹ By contrast, aerosol particles are expected to be scavenged by snow more effectively than by rain.¹²⁸

The phase partitioning of Hg(II) thus has important implications for removal by deposition. Observations of RGM and Hg(p) suggest that Hg(II) behaves as a semivolatile compound, partitioning between the gas and aerosol phases in a manner dependent on local aerosol concentration and composition and ambient temperature.^{129,130} As limiting cases, Hg(II) is mainly in the gas phase in warm environments with low aerosols and mainly in the particle phase in cold environments with high aerosols.¹⁰¹ Efficient uptake of Hg(II) by sea salt particles makes sea salt deposition a major process for delivery of Hg to the surface ocean.¹²⁶

Source-Receptor Relationships and Long-Term Trends. The atmospheric behavior of Hg affects source-receptor relationships. Early work focused on Hg deposition “hot spots” associated with sources upwind. Very high Hg deposition rates in Florida in the 1990s were attributed to local sources from waste incineration.¹³¹ However, high deposition remained even after those sources were controlled. It is now well established that the maximum in Hg wet deposition observed along the Gulf Coast of the U.S. in summer is due to scavenging of the Hg(II)-rich pool in the upper troposphere by deep convection.^{121,132,133} Thus the maxima in Hg wet deposition in the southeast U.S. does not appear to be associated with local sources of Hg(II).

Sediment cores indicate a 3-fold increase in total Hg atmospheric deposition from the mid-19th century to present,¹³⁴ but with substantial variability indicative of local and regional impacts.⁹⁸ Longer records show evidence of human influence extending back several millennia.^{135,136} Amos et al.¹⁰⁰ thus estimated that present-day Hg deposition associated with past and current anthropogenic emissions is enhanced 7-fold globally relative to preanthropogenic values.

Compilations of atmospheric observations from ship cruises and long-term monitoring sites around the Atlantic suggest decreasing trends in atmospheric Hg(0) in recent decades.^{102,123,137,138} These decreasing trends appear at odds with global inventories indicating constant or increasing primary anthropogenic emissions (Figure 2).^{22,139} One possible

explanation is a decline in secondary terrestrial Hg emissions¹⁰² or changes in ocean evasion. For example, Mason et al.¹⁴⁰ suggest that subsurface concentrations in the North Atlantic have decreased markedly since the 1980s (see below), which should drive decreases in evasion from the ocean surface. Soerensen et al.¹⁴¹ modeled the impact of the resulting decrease in Hg(0) evasion from the North Atlantic, demonstrating that it could explain the observed atmospheric decline and its spatial variability. Possible explanations for the decrease in North Atlantic subsurface Hg include a decrease in near-field Hg(II) deposition from sources in North America and Europe,¹⁴⁰ changes in the Hg(0) oxidation rate in the MBL, and decreases in Hg product use and in wastewater and riverine inflows to coastal zones.¹⁴¹ Recent decreases measured in estuarine sediment cores and in coastal watershed Hg mass balances in the North Atlantic suggest that coastal inputs have decreased dramatically.^{142–144} A better understanding of coastal Hg inputs to the open ocean and their historical trends is needed, particularly for the North Atlantic and the Arctic Ocean where the relatively large riverine contributions are thought to be a significant Hg source.¹⁴⁵

■ LAND AND FRESHWATERS

Processes Controlling Hg Dynamics on Land and Freshwaters. Atmospheric Hg deposition is the dominant input of Hg to most watersheds.¹⁴⁶ Weathering of Hg-bearing minerals in soil also contributes natural Hg inputs. Note legacy or ongoing Hg supply from mining, industrial activities, or wastewater can overwhelm inputs from atmospheric deposition and natural sources resulting in localized areas of elevated contamination.^{35,147}

Atmospheric inputs of Hg readily partition with living and detrital biomass influencing global and local Hg dynamics and ultimately its bioavailability in terrestrial and freshwater ecosystems. There are three mechanisms by which inputs of Hg to terrestrial ecosystems are released as secondary emissions. Following atmospheric deposition, Hg can be reduced from foliage, soil, surface waters or snowpack, and evaded back to the atmosphere.¹⁴⁸ Second, Hg(0) entering the stomata of foliage is subsequently deposited to soil as litterfall Hg inputs. This input, supplemented by wet and dry Hg(II) deposition and throughfall, is immobilized by soil organic matter probably through binding by reduced sulfur functional groups.^{149,150} Conceptually detrital organic matter and associated Hg has been viewed as fractions of varying reactivity and residence times (i.e., from months, years, decades, to centuries^{148,150}). As these “fractions” of soil organic matter are mineralized, a byproduct of organic matter “respiration” is Hg(0), which is evaded back to the atmosphere. Globally, these two pathways resupply from 1700 to 2800 Mg yr⁻¹ of Hg back to the atmosphere (Figure 1). Third, Hg is emitted to the atmosphere from biomass burning (300–600 Mg yr⁻¹; Figure 1²⁰) and other nonpoint sources. Overall, a considerable fraction of the Hg deposited to land is evaded back to the atmosphere as Hg(0) (Figure 1) and can be subjected again to transport, oxidation and deposition.

The balance of Hg input to terrestrial ecosystems largely accumulates in soil organic matter. Reconstructions of past Hg emissions suggest that although deposition has increased by about a factor of 3, soil Hg pools have increased by only about 20%, and like the ocean, are not at steady-state with respect to current emissions.^{22,100,148} The fraction of soil organic matter that degrades rapidly is largely associated with surface soil and a

relatively small Hg reservoir (~10–15 Gg). This pool is thought to respond rapidly to changes in Hg emissions. In contrast, slow decaying (~50 Gg) and highly recalcitrant (~250 Gg) soil Hg pools are large reservoirs that turnover slowly. These soil pools are important legacy reservoirs of historical Hg emissions, which return Hg to the atmosphere over long time scales. On a global basis, a small fraction of soil Hg is transported by riverine fluxes largely in association with dissolved or particulate organic matter (Figure 1), but this can be an important pathway to coastal areas and to certain portions of the open oceans.^{43,151,152}

Considerable research on terrestrial and freshwater ecosystems has focused on (1) transport of Hg(II) via drainage waters to reducing zones where net CH₃Hg production is high; (2) the net methylation of Hg(II); and (3) the trophic transfer of CH₃Hg. A host of characteristics and disturbances can accelerate or diminish each of these processes. Watershed characteristics such as size, topography and elevation, land cover, and use influence the net deposition of Hg to the land surface and transport to the aquatic ecosystems.^{153,154} Hydrologic characteristics and conditions regulate the fluvial transport of Hg(II) from uplands to reducing zones, which exhibit high rates of net methylation and from which large quantities of CH₃Hg are supplied to downstream and adjacent aquatic environments.¹⁵²

A variety of factors alter net rates of CH₃Hg in methylation zones such as wetlands, riparian zones and surface water sediments, including temperature, hydrology, and hydrologic perturbations (i.e., changes in discharge, hydroperiod, reservoir management), the supply of Hg(II) in a bioavailable form, labile organic carbon, and reducing conditions.³⁵ The formation of CH₃Hg is largely microbially mediated by SO₄²⁻¹⁵⁵ and to a lesser extent Fe^{156,157} reducing bacteria. Thus, the supply of electron acceptors to zones of methylation impact rates of methylation. Inputs of oxygen and NO₃⁻ slow net CH₃Hg supply either by the suppression of SO₄²⁻ reduction or maintaining oxidizing conditions that allow for the formation of Fe and Mn oxides and the partitioning of Hg to these surfaces.¹⁵⁸ At low concentrations of SO₄²⁻, further addition of SO₄²⁻ stimulates production of CH₃Hg until a maximum value of CH₃Hg⁺ production is reached.^{155,159} At higher concentrations, further addition of SO₄²⁻ diminishes production of CH₃Hg due to the formation of charged mercuric sulfide complexes that are less available and/or the immobilization of Hg(II) due to the formation of HgS_(s).¹⁶⁰ Note that ecosystem accumulation of CH₃Hg is the net effect of the production of CH₃Hg by methylation and the loss by demethylation. Therefore any processes that increase methylation or diminish demethylation will increase the accumulation of CH₃Hg.

Recent research has demonstrated the complex and pivotal role of dissolved organic matter (DOM) in net CH₃Hg production.¹⁶¹ DOM can serve as a Hg “carrier” facilitating transport and has the ability to chelate Hg(II) altering bioavailability. DOM can also stabilize or destabilize colloids altering transport, and inhibit or promote metal precipitation thereby altering the immobilization by HgS_(s) formation, through the formation of Hg-containing nanoparticles.^{162,163} Furthermore, CH₃Hg can be produced from Hg associated with colloids.^{164,165} Under reducing conditions, low concentrations of DOM can reduce Hg(II) to Hg(0).¹⁶⁶ However with increasing DOM, binding of Hg(II) limits the production of Hg(0).

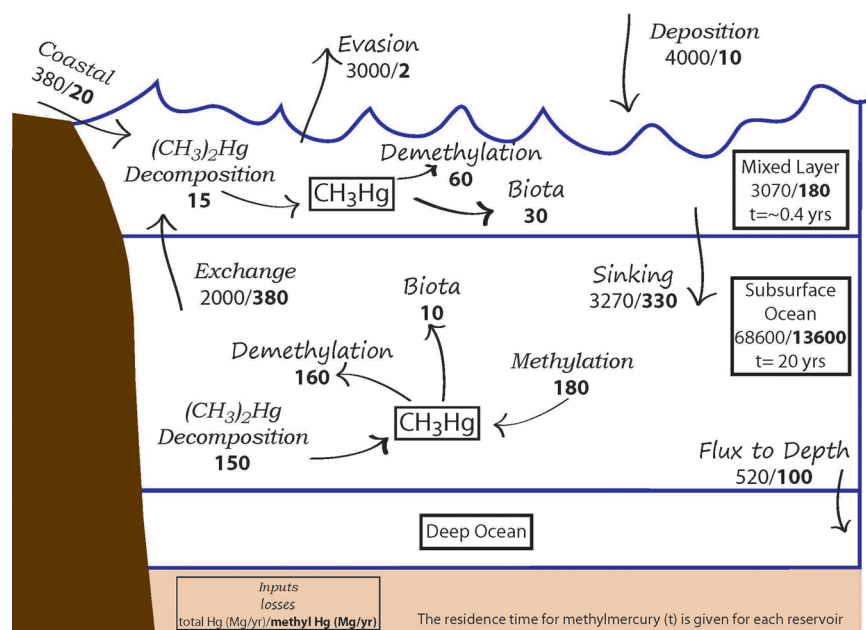


Figure 3. Overall budget for the sources and losses of total and methylated (in bold) mercury to the mixed layer and the subsurface ocean (defined as waters above the permanent thermocline) using data and information discussed throughout the paper. Fluxes are in $Mg\ yr^{-1}$ and reservoirs are given in Mg.

Several factors have been shown to enhance trophic transfer of CH_3Hg . Mercury burdens increase with the age and size of individuals within given fish populations, in part due to the slower rates of elimination and longer exposure in larger individuals, and due to the consumption of higher trophic level foods by older and larger individuals.⁸² Mercury concentrations in top predator fish are higher in food webs with longer chain lengths and less omnivory.¹⁶⁷ Both experimental and field studies show that nutrient enrichment diminishes CH_3Hg bioaccumulation in phytoplankton via the biodilution under algal bloom conditions.¹⁶⁸ Mercury concentrations in zooplankton also decrease with increasing zooplankton densities that, in turn, are correlated with lower CH_3Hg concentrations in fish.¹⁶⁹ Growth dilution in fish, also under conditions of high productivity and food availability, may also result in lower CH_3Hg concentrations.¹⁷⁰ Many studies have shown that fish Hg concentrations increase with decreases in pH.^{84,171} Most importantly the magnitude of fish Hg concentrations appears to be set at the base of the food chain and maintained through to top predators.¹⁷²

Spatial and Land Cover Patterns. Land cover and land use appear to be critical controllers of Hg in terrestrial and freshwater ecosystems. For example, Denkenberger et al.¹⁵⁴ synthesized rates of $Hg(0)$ evasion across land cover types, finding elevated rates from agricultural and pasture lands, which are characterized by greater incident solar radiation and rates of soil organic matter turnover, and lower values in forests. Rates of net Hg deposition can be enhanced under forest cover due to greater precipitation quantity coupled with enhanced scavenging of atmospheric Hg by the forest canopy and lower evasive losses.^{149,154} These patterns suggest that in regions remote from local sources that net Hg deposition would generally be greater under forest cover than agricultural land cover. Obrist et al.¹⁵⁰ found that soil Hg concentrations across the continental U.S. were not strongly related to Hg deposition but rather increased with soil C, latitude, precipitation quantity and clay content. They hypothesize

that over large spatial scales soil Hg accumulation is strongly linked with organic C accumulation. These spatial and land cover patterns are generally consistent with fish Hg concentrations. Fish Hg concentrations for a given region are typically higher in waters draining areas with greater forest and wetland cover, and lower in waters draining agricultural lands.^{33,34}

Temporal Patterns. There are few comprehensive studies of long-term response of ecosystems to Hg deposition, which has limited our quantitative understanding of the time-scale and extent of ecosystem response to changes in atmospheric Hg deposition.³⁵ Multidecadal insight has been mostly obtained through paleolimnological studies involving lake sediment cores and time series of fish Hg concentrations. Sediment studies show enrichment in Hg deposition for approximately the past century with the extent of increases in Hg deposition from preanthropogenic values decreasing with increasing latitude from the midlatitudes ($41-50^\circ N$)¹⁷³. Studies have indicated that remote regions receiving Hg inputs from largely global sources of Hg generally show long-term increases in Hg inputs. In contrast, in regions experiencing recent controls on elevated Hg emissions have been decreases in sediment Hg deposition over recent decades.⁹⁸

Fish Hg measurements are available over shorter time scales but in some cases extend back 40 years or more. Generally these studies have shown declines in fish Hg concentrations consistent with regional declines in Hg deposition,¹⁷⁴ although patterns of changes in fish Hg have been linked with other disturbances including land use, nutrient supply, climate change effects, and decreases in acid deposition.³³ Drevnick et al.¹⁷⁵ observed decreases in fish Hg in lakes on Isle Royale in Lake Superior, which they attributed to decreases in atmospheric SO_4^{2-} deposition. Decreases in acid deposition could also drive decreases in fish Hg due to increases in surface water pH.¹⁷¹ In contrast, recent studies in Scandinavia report increases in fish Hg associated with increases in DOM resulting from watershed recovery from decreases in acid deposition.^{176,177} These

variable spatial and temporal patterns indicate that while controlling Hg inputs are essential to decrease Hg exposure, the response of biota will be highly variable in space and time due to the residence time of Hg associated with processing through detrital organic pools, and watershed processes/disturbances that affect Hg transport, net methylation and trophic transfer.

OCEANS

Ocean Hg Transport and Long-Term Changes.

Empirical and modeling studies^{16,99,178–183} provide a reasonable constraint on external Hg inputs to oceans, including the magnitude of air-sea exchange of Hg (wet and dry deposition and gas evasion). Atmospheric deposition is the dominant input of Hg(II) to the open oceans (3600 Mg yr⁻¹; Figures 1,3).

Over 80% of the Hg deposited in marine ecosystems is reemitted to the atmosphere (as Hg(0) predominantly, but some as (CH₃)₂Hg) (Figures 1,3), increasing the residence time of Hg cycling through the reservoirs of the surface biosphere.^{148,181} Net biotic and photochemical reduction of Hg(II) (photochemical oxidation and reduction, as well as biological reduction occur in surface waters) drives evasion of the Hg(0)^{181,184–187} While evasion may decrease the pool of potentially bioavailable Hg(II) for methylation and bioaccumulation over the short-term, model calculations suggest that because of the overall recycling of Hg (deposition and re-emission) between both the surface ocean and the land and the atmosphere, the ocean is the dominant long-term sink for primary anthropogenic Hg emissions to the atmosphere.^{22,100}

An evaluation of the cumulative impact of historical anthropogenic emissions from the last century on ocean Hg concentrations and distribution suggests an increase in the surface ocean (~200% increase on average) and subsurface waters (~25% increase¹⁴⁰). In contrast, the effects of historical emissions on the deep ocean are small (~10%) and are confined to recent deep water formation, such as the deep North Atlantic (due to sinking North Atlantic Deep Water) and the deeper waters of the Southern Ocean (where Antarctic Deep Water sinks from the surface). If one further considers inputs from historic precious metal mining in the Americas and elsewhere dating back from the 1500s,^{22,188} the anthropogenic impact of Hg on the surface and intermediate ocean waters is greater, increasing by a factor of 2–3. Time series data indicate an enrichment in Hg concentrations in the upper waters of the North Atlantic Ocean with the first reliable measurements in the 1980s, followed by decreases in recent years¹⁴⁰ (Figure 4).

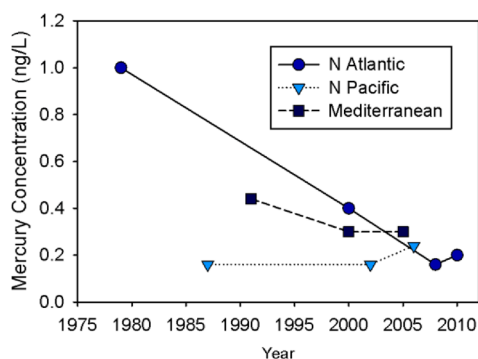


Figure 4. Long-term trends in total Hg concentrations in the surface waters of the oceans and Mediterranean Sea.¹⁴⁰

Similar decreases have also been reported for the Mediterranean.¹⁸⁹ In contrast, in the Pacific there have been increases in Hg concentrations in the upper waters since 2000.^{32,190} These data confirm that Hg in the upper ocean responds on a decadal time scale to changes in the atmosphere, and reflects regional signals associated with anthropogenic inputs. The recent trends in ocean water column Hg appear consistent with decreasing Hg emissions in North America and Europe and rapidly increasing emissions in Asia^{6,8} (Figure 2).

Methylated Hg Dynamics. In contrast to an improved understanding of the dynamics of Hg inputs to the oceans, there is limited information on how methylated Hg concentrations (both CH₃Hg and (CH₃)₂Hg; collectively termed ΣCH₃Hg) have responded to these changes. Nevertheless, past and ongoing changes in the extent and distribution of anthropogenic emissions and Hg inputs to ocean waters^{6,8,22,191} have and will lead to changes in ocean fish concentrations.^{183,192,193} Ocean circulation patterns and the presence of a subsurface zone of maximum methylation¹⁴⁰ contribute to lags between the onset of changes in atmospheric Hg deposition and the response of fish CH₃Hg concentrations in the open ocean.

Concentrations of ΣCH₃Hg are low in ocean surface waters and increase with depth, reaching a maximum in the subsurface/subthermocline waters (>100–1000 m) depending on location. Concentrations of ΣCH₃Hg are generally low in the deep ocean. The higher concentrations of ΣCH₃Hg are found in zones of the upper ocean where sinking organic matter is rapidly remineralized.¹⁹⁴ Open ocean profiles are generally consistent with the formation of ΣCH₃Hg in conjunction with the decomposition of organic matter.^{32,195–198} Stable Hg isotope incubation experiments in the Arctic also indicate that methylation of Hg(II) occurs in the water column producing both CH₃Hg and (CH₃)₂Hg.¹⁹⁹ In sum, field observations and modeling demonstrate that in situ methylation, rather than external inputs, is the predominant source of CH₃Hg that accumulates in ocean fish consumed by humans.

The formation of ΣCH₃Hg in the open ocean contrasts with patterns of freshwater and coastal ecosystems (see Land and Freshwaters above). First, while measurable (CH₃)₂Hg has generally been found in open ocean studies, it is rarely detected in coastal and freshwater ecosystems. Moreover, while CH₃Hg is generally formed in oxygen deficient or anoxic zones in coastal and freshwaters, this is not a prerequisite for formation in the open ocean.^{140,160}

A recent mass balance suggests riverine/coastal inputs of CH₃Hg are small relative to the production of ΣCH₃Hg within the ocean water column¹⁴⁰ (Figure 3). Riverine inputs of CH₃Hg from the coastal zone to offshore waters are approximately 20 Mg yr⁻¹ (range 14–24 Mg yr⁻¹) with about 30 Mg yr⁻¹ (20–40 Mg yr⁻¹) deposited into the coastal zone due to net particulate settling.¹⁴⁰ The return flux of dissolved CH₃Hg due to diffusional and advective transport from coastal (shelf and slope) sediments is estimated to be on the same order as the net depositional input (range 4–40 Mg yr⁻¹).^{140,200} Groundwater inputs are not considered substantial given the generally low concentrations of CH₃Hg and the estimated water flux.^{140,201}

The major ocean sink for ΣCH₃Hg is photochemical and biotic demethylation (Figure 3). Estimates of demethylation of ΣCH₃Hg in the surface waters (<50 m) and in the subsurface waters (50–1000 m) are 60 and 160 Mg yr⁻¹, respectively, based on an assessment of the literature.^{184,185,187,198,202} While

these values are average rates, demethylation rate constants vary by a factor of 5 depending on incident solar radiation and other factors. The net removal of $\sum \text{CH}_3\text{Hg}$ to deep ocean sediments is a secondary sink ($<20 \text{ Mg yr}^{-1}$; $<20\%$ of the inputs to the deep ocean; Figure 3). The rate of decay of $\sum \text{CH}_3\text{Hg}$ even in deeper, colder intermediate depth waters is sufficiently rapid that both CH_3Hg and $(\text{CH}_3)_2\text{Hg}$ should not persist over the time scales of decades (i.e., the time scale of offshore water mass transport), suggesting continued in situ formation in subsurface ocean waters in concert with organic matter degradation.^{197,198,203}

Mercury in Marine Biota. As with freshwaters, CH_3Hg concentrations in ocean fish generally track trophic position and age, but important differences are evident in concentrations for the same species found in different ocean basins. Generally, patterns in fish Hg concentrations are consistent with variations in total Hg concentrations among oceans, although there are differences in the fraction of Hg as $\sum \text{CH}_3\text{Hg}$ across marine ecosystems.^{31,32,195,204}

Feeding depth and location are important considerations. Many commercial and noncommercial fish species that feed in subsurface waters to 600 m have higher CH_3Hg levels than species of similar trophic status feeding at the surface. Positive correlations have been found between the concentrations of CH_3Hg in piscivorous fish and marine birds and their feeding depth or the depth of their prey.^{204–209} Studies using stable isotopes of Hg, C, and N in marine fish indicate that while estuarine and coastal fish may obtain CH_3Hg generated within the coastal watersheds and sediments, offshore ocean fisheries derive CH_3Hg from the open ocean.^{210,211}

As for freshwater ecosystems, it is unlikely that future CH_3Hg concentrations in ocean fish will solely be controlled by changes in atmospheric Hg emissions and deposition to the oceans. The future impacts of changing climate on ocean Hg deposition, transformations and bioaccumulation of CH_3Hg will likely be manifested through changes in (1) the efficiency of net reduction in surface waters and net methylation in the subsurface; (2) the rate of gas exchange driven by changes in physical characteristics; (3) the rate of oxidation of $\text{Hg}(0)$ in the atmosphere; and/or (4) changes in wet deposition patterns.^{212,213} Moreover, changes in Hg removal from the surface ocean by particle scavenging as a result of changes in productivity also could impact the extent of re-emission to the atmosphere, and therefore net methylation. The impacts of climate change, eutrophication,²¹⁴ and other secondary factors (e.g., changes in species composition, overfishing) on the net transfer of atmospheric Hg deposition into the aquatic food chain as CH_3Hg are often ignored in model projections of future changes in the global Hg cycle. However, these secondary effects will likely be important drivers of future Hg exposure.

■ CONCLUDING REMARKS

Policy Implications. The major policy implications from this synthesis of existing Hg science are (1) atmospheric emissions and deposition dominate anthropogenic Hg inputs and controlling these inputs remain an important priority for national and international policy; (2) releases to land and water are very high in some locales and more information is needed on the extent to which these inputs circulate globally and contaminate global fisheries via re-emissions and transport from nearshore to offshore waters; (3) human consumption of marine fish and shellfish is a globally important exposure

pathway and should be more fully integrated into cost-benefit analyses; (4) health effects to humans beyond IQ decrement and at low levels of exposure, and the widespread effects to wildlife suggest the need for expanding the end points evaluated in risk assessments; and (5) effectiveness evaluation for both national policy and the international Hg treaty should include not only fish, but also supporting measurements of Hg deposition, water column $\text{Hg}/\text{CH}_3\text{Hg}$, and sediment $\text{Hg}/\text{CH}_3\text{Hg}$, for a range of systems and geographic locations.

Mercury is a local, regional, and global pollutant; while we have focused on Hg pollution at the global scale, it is important to emphasize that Hg also has regional and local dimensions. Regional/local releases and/or exposure of the local consumer and wildlife can be mitigated by strategies that are geared toward the characteristics of a particular location. In parallel with discussions to limit and manage Hg releases through an international treaty, there is a critical need for coordination of local, national and international health and environmental agencies to develop (1) strategies and technologies to limit releases and exposure from local/regional sources;^{4,45,215,216} and (2) quantitative risk-benefit assessment paradigms for local/regional fish consumption to protect local populations.

Monitoring and Effectiveness Evaluation. Documenting the response of Hg in ecosystems following the implementation of Hg policies and management activities and the confounding effects of other actions that affect Hg dynamics (e.g., nutrient control, controls on atmospheric S and N deposition, land use and climate change, changes in water management) at both local and global scales is a critical element of the international Hg treaty and most national policies.^{4,17,217,218}

To support evaluation of these national and international policies, there is a need for coordinated and detailed global, multimedia Hg monitoring network (air, water, sediment and soils, and biota). Better coordination among current and future measurement campaigns should focus less on a particular media or ecosystem type (e.g., air, water) and more on cross boundary transport, such as land–atmosphere exchange or riverine transport to the coastal ocean.^{15,97,219} There have been few studies of changes in Hg transport and contamination following recent industrialization of developing countries which are likely to have increasing future emissions.^{6,8,220} There is also a critical need for observations in understudied areas such as the Southern Hemisphere, the major oceans, and within the free troposphere.

Further model development will be required to validate processes and hypothesis testing; probe the quality of empirical observations; and evaluate future scenarios of the impacts of changing Hg emissions and/or global change on the levels of CH_3Hg in aquatic organisms. While monitoring is useful to assess trends and validate models, it often fails to elucidate the processes and factors which influence the net conversion of $\text{Hg}(\text{II})$ into CH_3Hg . A quantitative understanding of pathways and mechanisms that affect the transport of $\text{Hg}(\text{II})$ from sources to ecosystems, the conversion of $\text{Hg}(\text{II})$ to CH_3Hg , and its bioaccumulation in food webs is fundamental to evaluating and managing human and wildlife health risks. Therefore, scientific synthesis should be periodically supported.

Future Research Priorities. While research over the past few decades has provided considerable insight on Hg dynamics and effects ranging from local to global scales, this synthesis and the associated literature have identified several focused questions for future Hg research:

- What is the redox chemistry of atmospheric Hg and how does it determine source-receptor relationships between emissions and deposition?
- To what extent and how rapidly will changes in anthropogenic Hg emissions at one location affect local, regional and global concentrations of Hg and ultimately CH₃Hg in fish elsewhere?
- How do climatic variability and anthropogenic disturbances (e.g., climate change, population increases, perturbations to food chains, changes in other air pollutants, land disturbance) affect CH₃Hg concentrations in fish and other top predators?
- What are the low dose chronic effects of CH₃Hg on the fish eating populations?

These questions should continue to drive future Hg research that will improve understanding of the response to anticipated controls on anthropogenic Hg emissions and releases.

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Notes

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REFERENCES

- (1) Pirrone, N.; Mahaffey, K. R. *Dynamics of Mercury Pollution on Regional and Global Scales*; Springer Publishers: New York, 2005; p 744.
- (2) Pirrone, N.; Mason, R. *Mercury Fate and Transport in the Global Atmosphere: Emissions, Measurements and Models*; Springer: New York; p 637.
- (3) Selin, N. E.; Wu, S.; Nam, K. M.; Reilly, J. M.; Paltsev, S.; Prinn, R. G.; Webster, M. D. Global health and economic impacts of future ozone pollution. *Environ. Res. Lett.* **2009**, *4*, 9.
- (4) Schmeltz, D.; Evers, D. C.; Driscoll, C. T.; Artz, R.; Cohen, M.; Gay, D.; Haeuber, R.; Krabbenhoft, D. P.; Mason, R.; Morris, K.; Wiener, J. G. MercNet: A national monitoring network to assess responses to changing mercury emissions in the United States. *Ecotoxicology* **2011**, *20*, 1713–1725.
- (5) Pirrone, N.; Keeler, G. J.; Nriagu, J. O. Regional differences in worldwide emissions of mercury to the atmosphere. *Atmos. Environ.* **1996**, *30* (17), 2981–2987.
- (6) Pirrone, N.; Cinnirella, S.; Feng, X.; Finkelman, R. B.; Friedli, H. R.; Leaner, J.; Mason, R.; Mukherjee, A. B.; Stracher, G. B.; Streets, D.

- G.; Telmer, K. Global mercury emissions to the atmosphere from anthropogenic and natural sources. *Atmos. Chem. Phys.* **2010**, *10* (13), 5951–5964.
- (7) Pacyna, J. M.; Pacyna, E. G.; Aas, W. Changes of emissions and atmospheric deposition of mercury, lead, and cadmium. *Atmos. Environ.* **2009**, *43*, 117–127.
- (8) Streets, D. G.; Zhang, Q.; Wu, Y. Projections of global mercury emissions in 2050. *Environ. Sci. Technol.* **2009**, *43* (8), 2983–2988.
- (9) Sprovieri, F.; Pirrone, N.; Ebinghaus, R.; Kock, H.; Dommergue, A. A review of worldwide atmospheric mercury measurements. *Atmos. Chem. Phys.* **2010**, *10*, 8245–8265.
- (10) Schroeder, W. H.; Munthe, J. Atmospheric mercury—An overview. *Atmos. Environ.* **1998**, *32*, 809–822.
- (11) Ebinghaus, R.; Kock, H. H.; Temme, C.; Einax, J. W.; Lowe, A. G.; Richter, A.; Burrows, J. P.; Schroeder, W. H. Antarctic springtime depletion of atmospheric mercury. *Environ. Sci. Technol.* **2002**, *36* (6), 1238–1244.
- (12) Lindberg, S. E.; Brooks, S. B.; Lin, C.-J.; Scott, K. J.; Landis, M. S.; Stevens, R. K.; Goodsite, M.; Richter, A. The dynamic oxidation of gaseous mercury in the Arctic atmosphere at polar sunrise. *Environ. Sci. Technol.* **2002**, *36*, 1245–1256.
- (13) Lindberg, S.; Bullock, R.; Ebinghaus, R.; Engstrom, D.; Feng, X. B.; Fitzgerald, W.; Pirrone, N.; Prestbo, E.; Seigneur, C. A synthesis of progress and uncertainties in attributing the sources of mercury in deposition. *Ambio* **2007**, *36* (1), 19–32.
- (14) Sprovieri, F.; Pirrone, N.; Hedgecock, I. M.; Landis, M. S.; Stevens, R. K. Intensive atmospheric mercury measurements at Terra Nova Bay in Antarctica during November and December 2000. *J. Geophys. Res., [Atmos.]* **2002**, *107* (D23), 4722.
- (15) Sprovieri, F.; Pirrone, N.; Landis, M.; Stevens, R. K. Oxidation of gaseous elemental mercury to gaseous divalent mercury during 2003 polar sunrise at Ny-Alesund. *Environ. Sci. Technol.* **2005**, *39*, 9156–9165.
- (16) Hedgecock, I. M.; Pirrone, N.; Trunfio, G. A.; Sprovieri, F. Integrated mercury cycling, transport, and air-water exchange (MECAWEx) model. *J. Geophys. Res., [Atmos.]* **2006**, *111*, D20302.
- (17) Pirrone, N.; Hedgecock, I. M.; Sprovieri, F. New Directions: Atmospheric mercury, easy to spot and hard to pin down: Impasse? *Atmos. Environ.* **2008**, *42*, 8549–8551.
- (18) Steffen, A.; Douglas, T.; Amyot, M.; Ariya, P.; Aspmo, K.; Berg, T.; Bottenheim, J.; Brooks, S.; Cobbett, F.; Dastoor, A.; Dommergue, A.; Ebinghaus, R.; Ferrari, C.; Gardfeldt, K.; Goodsite, M. E.; Lean, D.; Poulain, A. J.; Scherz, C.; Skov, H.; Sommar, J.; Temme, C. A synthesis of atmospheric mercury depletion event chemistry in the atmosphere and snow. *Atmos. Chem. Phys.* **2008**, *8* (6), 1445–1482.
- (19) Durnford, D.; Dastoor, A.; Figueras-Nieto, D.; Ryjkov, A. Long range transport of mercury to the Arctic and across Canada. *Atmos. Chem. Phys. Discuss.* **2010**, *10*, 6063–6086.
- (20) Friedli, H. R.; Arellano, A. F.; Cinnirella, S.; Pirrone, N. Initial estimates of mercury emissions to the atmosphere from global biomass burning. *Environ. Sci. Technol.* **2009**, *43* (10), 3507–3513.
- (21) Pacyna, E. G.; Pacyna, J. M.; Steenhuisen, F.; Wilson, S. Global anthropogenic mercury emission inventory for 2000. *Atmos. Environ.* **2006**, *40*, 4048–4063.
- (22) Streets, D. G.; Devane, M. K.; Lu, Z.; Bond, T. C.; Sunderland, E. M.; Jacob, D. J. All-time releases of mercury to the atmosphere from human activities. *Environ. Sci. Technol.* **2011**, *45* (24), 10485–10491.
- (23) AMAP/UNEP. Technical Background Report to the Global Atmospheric Mercury Assessment. Arctic Monitoring and Assessment Programme/UNEP Chemicals Branch, 2008. http://www.chem.unep.ch/mercury/Atmospheric_Emissions/Technical_background_report.pdf.
- (24) Dastoor, A. P.; Davignon, D. Global mercury modelling at Environment Canada. In *Mercury Fate and Transport in the Global Atmosphere*; Pirrone, N., Mason, R., Eds.; Springer: New York, 2009; pp 519–532.
- (25) Jaegle, L.; Strode, S. A.; Selin, N. E.; Jacob, D. J. The Geos-Chem mode. In *Mercury Fate and Transport in the Global Atmosphere*; Pirrone, N., Mason, R., Eds.; Springer: New York, 2009; pp 533–545.

- (26) Jung, G.; Hedgecock, I. M.; Pirrone, N. The ECHMERT model. In *Mercury Fate and Transport in the Global Atmosphere*; Pirrone, N., Mason, R., Eds.; Springer: New York, 2009; pp 547–569.
- (27) Seigneur, C.; Vijayaraghavan, K.; Lohman, K.; Levin, L. The AER/EPRI global chemical transport model for mercury (CTM-HG). In *Mercury Fate and Transport in the Global Atmosphere*; Pirrone, N.; Mason, R., Eds.; Springer: New York, 2009; pp 589–602.
- (28) Travnikov, O.; Ilyin, I. The EMEP/MSC-E mercury modeling system. In *Mercury Fate and Transport in the Global Atmosphere*; Pirrone, N., Mason, R., Eds.; Springer: New York, 2009; pp 571–587.
- (29) Sundseth, K.; Pacyna, J. M.; Pacyna, E. G.; Munthe, J.; Belhaj, M.; Astrom, S. Economic benefits from decreased mercury emissions: Projections for 2020. *J. Clean. Prod.* **2010**, *18*, 386–394.
- (30) Sunderland, E. M.; Dalziel, J.; Heyes, A.; Branfireun, B. A.; Krabbenhoft, D. P.; Gobas, F. A. P. C. Response of a macrotidal estuary to changes in anthropogenic mercury loading between 1850 and 2000. *Environ. Sci. Technol.* **2010**, *44*, 1698–1704.
- (31) Cossa, D.; Averty, B.; Pirrone, N. The origin of methylmercury in open Mediterranean waters. *Limnol. Oceanogr.* **2009**, *54* (3), 837–844.
- (32) Sunderland, E. M.; Krabbenhoft, D. P.; Moreau, J. W.; Strode, S. A.; Landing, W. M. Mercury sources, distribution, and bioavailability in the North Pacific Ocean: Insights from data and models. *Global Biogeochem. Cycles* **2009**, *23*, GB2010 DOI: 10.1029/2008GB003425.
- (33) Evers, D. C.; Wiener, J. G.; Driscoll, C. T.; Gay, D. A.; Basu, N.; Monson, B. A.; Lambert, K. F.; Morrison, H. A.; Morgan, J. T.; Williams, K. A.; Soehl, A. G. *Great Lakes Mercury Connections: The Extent and Effects of Mercury Pollution in the Great Lakes Region*, Report BRI 2011-18; Biodiversity Research Institute: Gorham, Maine, 2011.
- (34) Driscoll, C. T.; Han, Y.-J.; Chen, C. Y.; Evers, D. C.; Lambert, K. F.; Holsen, T. M.; Kamman, N. C.; Munson, R. K. Mercury contamination in forest and freshwater ecosystems in the Northeastern United States. *Bioscience* **2007**, *57* (1), 17–28.
- (35) Munthe, J.; Bodaly, R. A.; Branfireun, B. A.; Driscoll, C. T.; Gilmour, C. C.; Harris, R.; Horvat, M.; Lucotte, M.; Malm, O. Recovery of mercury-contaminated fisheries. *Ambio* **2007**, *36* (1), 33–44.
- (36) AMAP/UNEP. *Technical Background Report for the Global Mercury Assessment 2013*. ; Arctic Monitoring and Assessment Programme, Oslo, Norway/UNEP Chemicals Branch: Geneva, Switzerland, 2013.
- (37) Pirrone, N.; Keating, S. T. *Hemispheric Transport of Air Pollution, Part B: Mercury*; Air Pollution Studies No. 18; UN Economic Commission for Europe: Geneva, Switzerland, 2010.
- (38) GEO. [Group on Earth Observations] *Report on Progress Beijing Ministerial Summit: Observe, Share, Inform*, 5 November 2010; GEO Secretariat: Geneva, Switzerland, 2011.
- (39) FAO. Food and Agriculture Organization of the United Nations. *The State of World Fisheries and Aquaculture—2010*; FAO Statistics and Information Service of the Fisheries and Aquaculture Department: Rome, 2010.
- (40) FAO. *Fishery and Aquaculture Statistics*; Food and Agriculture Organization of the United Nations: Rome, 2011; p 415.
- (41) Meng, B.; Feng, X.; Qiu, G.; Liang, P.; Li, P.; Chen, C.; Shang, L. The process of methylmercury accumulation in rice (*Oryza sativa* L.). *Environ. Sci. Technol.* **2011**, *45*, 2711–2717.
- (42) Chan, H. M.; Receveur, O. Mercury in the traditional diet of indigenous peoples in Canada. *Environ. Pollut.* **2000**, *110* (1), 1–2.
- (43) Kirk, J. L.; Lehnher, I.; Andersson, M.; Braune, B. M.; Chan, L.; Dastoor, A. P.; Durnford, D.; Gleason, A. L.; Loseto, L. L.; Steffen, A.; St. Louis, V. L. Mercury in Arctic marine ecosystems: Sources, pathways and exposure. *Environ. Res.* **2012**, *119*, 64–87.
- (44) Lincoln, R. A.; Shine, J. P.; Chesney, E. J.; Vorhees, D. J.; Grandjean, P.; Senn, D. B. Fish consumption and mercury exposure among louisiana recreational anglers. *Environ. Health Perspect.* **2011**, *119* (2), 245–251.
- (45) Davis, J. A.; Looker, R. E.; Yee, D.; Marvin-Di Pasquale, M.; Grenier, J. L.; Austin, C. M.; McKee, L. J.; Greenfield, B. K.; Brodberg, R.; Blum, J. D. Reducing methylmercury accumulation in the food webs of San Francisco Bay and its local watershed. *Environ. Res.* **2012**, *119*, 3–26.
- (46) Grieb, T. M.; Driscoll, C. T.; Gloss, S. P.; Schofield, C. L.; Bowie, G. L.; Porcella, D. B. Factors affecting mercury accumulation in fish in the Upper Michigan Peninsula. *Environ. Toxicol. Chem.* **1990**, *9* (7), 919.
- (47) Sioen, I.; Van Camp, J.; Verdonck, F. A. M.; Van Thuyne, N.; Willems, J. L.; De Henauw, S. W. J. How to use secondary data on seafood contamination for probabilistic exposure assessment purposes? Main problems and potential solutions. *Hum. Ecol. Risk Assess.* **2007**, *13* (3), 632–657.
- (48) FAO/WHO. *Report of the Joint FAO/WHO Expert Consultation on the Risks and Benefits of Fish Consumption*, 25–29 January 2010, Report # 979; Food and Agriculture Organization of the United Nations and World Health Organization: Rome, 2011.
- (49) WHO. *Report of the WHO Working Group on Data*; World Health Organization, GEMS/Food Programme: Geneva, 2010; p 24, <http://www.who.int/foodsafety/chem/gems/en/index.html>.
- (50) Clarkson, T. W.; Magos, L. The toxicology of mercury and its chemical compounds. *Crit. Rev. Toxicol.* **2006**, *36* (8), 609–662.
- (51) Grandjean, P.; Satoh, H.; Murata, K.; Eto, K. Adverse effects of methylmercury: Environmental health research implications. *Environ. Health Perspect.* **2010**, *118* (8), 1137–45.
- (52) Harada, Y. Congenital (or fetal) Minamata disease. In *Minamata Disease. Study group of Minamata Disease*; Japan: Kumamoto University, 1968; pp 93–118.
- (53) Sakamoto, M.; Murata, K.; Tsuruta, K.; Miyamoto, K.; Akagi, H. Retrospective study on temporal and regional variations of methylmercury concentrations in preserved umbilical cords collected from inhabitants of the Minamata area, Japan. *Ecotoxicol. Environ. Saf.* **2010**, *73* (6), 1144–1149.
- (54) McDowell, M. A.; Dillon, C. F.; Osterloh, J.; Bolger, P. M.; Pellizzari, E.; Fernando, R.; de Oca, R. M.; Schober, S. E.; Sinks, T.; Jones, R. L.; Mahaffey, K. R. Hair mercury levels in US children and women of childbearing age: Reference range data from NHANES 1999–2000. *Environ. Health Perspect.* **2004**, *112* (11), 1165–1171.
- (55) Grandjean, P.; Weihe, P.; White, R. F.; Debes, F.; Araki, S.; Yokoyama, K.; Murata, K.; Sorensen, N.; Dahl, R.; Jorgensen, P. J. Cognitive deficit in 7-year-old children with prenatal exposure to methylmercury. *Neurotoxicol. Teratol.* **1997**, *19* (6), 417–428.
- (56) Yorifuji, T.; Tsuda, T.; Inoue, S.; Takao, S.; Harada, M. Long-term exposure to methylmercury and psychiatric symptoms in residents of Minamata, Japan. *Environ. Int.* **2011**, *37* (5), 907–913.
- (57) Davidson, P. W.; Myers, G. J.; Cox, C.; Axtell, C.; Shamlaye, C.; Sloane-Reeves, J.; Cernichiari, E.; Needham, L.; Choi, A.; Wang, Y.; Berlin, M.; Clarkson, T. W. Effects of prenatal and postnatal methylmercury exposure from fish consumption on neurodevelopment development study. *JAMA, J. Am. Med. Assoc.* **1998**, *280*, 701–707.
- (58) Kjellstrom, T.; Kennedy, P.; Wallis, S.; Mantell, C. *Physical and Mental Development of Children with Prenatal Exposure to Mercury from Fish. Stage 1. Preliminary Tests at Age 4*, Report 3080; National Swedish Environmental Protection Board : Solna, Sweden, 1986.
- (59) Rice, D. C.; Schoeny, R.; Mahaffey, K. Methods and rationale for derivation of a reference dose for methylmercury by the US EPA. *Risk Anal.* **2003**, *23* (1), 107–115.
- (60) Davidson, P. W.; Cory-Slechta, D. A.; Thurston, S. W.; Huang, L. S.; Shamlaye, C. F.; Gunzler, D.; Watson, G.; van Wijngaarden, E.; Zareba, G.; Klein, J. D.; Clarkson, T. W.; Strain, J. J.; Myers, G. J. Fish consumption and prenatal methylmercury exposure: Cognitive and behavioral outcomes in the main cohort at 17 years from the Seychelles child development study. *Neurotoxicology* **2011**, *32* (6), 711–717.
- (61) Debes, F.; Budtz-Jorgensen, E.; Weihe, P.; White, R. F.; Grandjean, P. Impact of prenatal methylmercury exposure on neurobehavioral function at age 14 years. *Neurotoxicol. Teratol.* **2006**, *28* (5), 536–547.
- (62) Julvez, J.; Debes, F.; Weihe, P.; Choi, A.; Grandjean, P. Sensitivity of continuous performance test (CPT) at age 14 years to

developmental methylmercury exposure. *Neurotoxicol. Teratol.* **2010**, *32* (6), 627–632.

(63) Lynch, M. L.; Huang, L. S.; Cox, C.; Strain, J. J.; Myers, G. J.; Bonham, M. P.; Shamlaye, C. F.; Stokes-Riner, A.; Wallace, J. M. W.; Duffy, E. M.; Clarkson, T. W.; Davidson, P. W. Varying coefficient function models to explore interactions between maternal nutritional status and prenatal methylmercury toxicity in the Seychelles Child Development Nutrition Study. *Environ. Res.* **2011**, *111* (1), 75–80.

(64) Stokes-Riner, A.; Thurston, S. W.; Myers, G. J.; Duffy, E. M.; Wallace, J.; Bonham, M.; Robson, P.; Shamlaye, C. F.; Strain, J. J.; Watson, G.; Davidson, P. W. A longitudinal analysis of prenatal exposure to methylmercury and fatty acids in the Seychelles. *Neurotoxicol. Teratol.* **2011**, *33*, 325–328.

(65) Strain, J. J.; Davidson, P. W.; Thurston, S. W.; Harrington, D.; Mulhern, M. S.; McAfee, A. J.; van Wijngaarden, E.; Shamlaye, C. F.; Henderson, J.; Watson, G. E.; Zareba, G.; Cory-Slechta, D. A.; Lynch, M.; Wallace, J. M.; McSorley, E. M.; Bonham, M. P.; Stokes-Riner, A.; Sloane-Reeves, J.; Jancius, J.; Wong, R.; Clarkson, T. W.; Myers, G. J. Maternal PUFA status but not prenatal methylmercury exposure is associated with children's language functions at age five years in the Seychelles. *J. Nutr.* **2012**, *142* (11), 1943–1949.

(66) Axelrad, D. A.; Bellinger, D. C.; Ryan, L. M.; Woodruff, T. J. Dose-response relationship of prenatal mercury exposure and IQ: An integrative analysis of epidemiologic data. *Environ. Health Perspect.* **2007**, *115* (4), 609–615.

(67) Mahaffey, K.; Clickner, R.; Jeffries, R. Adult women's blood mercury concentrations vary regionally in USA: Association with patterns of fish consumption (NHANES 1999–2004). *Environ. Health Perspect.* **2009**, *117*, 47–53.

(68) Kim, N. S.; Lee, B. K. Blood total mercury and fish consumption in the Korean general population in KNHANES III, 2005. *Sci. Total Environ.* **2010**, *408* (20), 4841–4847.

(69) Karagas, M. R.; Choi, A. L.; Oken, E.; Horvat, M.; Schoeny, R.; Kamai, E.; Cowell, W.; Grandjean, P.; Korrick, S. Evidence on the human health effects of low-level methylmercury exposure. *Environ. Health Perspect.* **2012**, *120* (6), 799–806.

(70) Nevado, J. J. B.; Martin-Doimeadios, R. C. R.; Bernardo, F. J. G.; Moreno, M. J.; Herculano, A. M.; do Nascimento, J. L. M.; Crespo-Lopez, M. E. Mercury in the Tapajos River basin, Brazilian Amazon: A review. *Environ. Int.* **2010**, *36* (6), 593–608.

(71) Lemire, M.; Fillion, M.; Frenette, B.; Passos, C. J. S.; Guimaraes, J. R. D.; Barbosa, F.; Mergler, D. Selenium from dietary sources and motor functions in the Brazilian Amazon. *Neurotoxicology* **2011**, *32* (6), 944–953.

(72) Furieri, L. B.; Fiorelli, M.; Junior, R. F. R.; Bartolomé, M. V.; Fernandes, A. A.; Cachoeiro, V.; Lahera, V.; Salaices, M.; Stefanon, I.; Vassallo, D. V. Exposure to low mercury concentration in vivo impairs myocardial contractile function. *Toxicol. Appl. Pharmacol.* **2011**, *255* (2), 193–9.

(73) Chan, H. M.; Egeland, G. M. Fish consumption, mercury exposure, and heart diseases. *Nutr. Rev.* **2004**, *62* (2), 68–72.

(74) Roman, H. A.; Walsh, T. L.; Coull, B. A.; Dewailly, E.; Guallar, E.; Hattis, D.; Marien, K.; Schwartz, J.; Stern, A. H.; Virtanen, J. K.; Rice, G. Evaluation of the cardiovascular effects of methylmercury exposures: Current evidence supports development of a dose-response function for regulatory benefits analysis. *Environ. Health Perspect.* **2011**, *119* (5), 607–614.

(75) Mozaffarian, D.; Shi, P. L.; Morris, J. S.; Spiegelman, D.; Grandjean, P.; Siscovick, D. S.; Willett, W. C.; Rimm, E. B. Mercury exposure and risk of cardiovascular disease in two U.S. cohorts. *New Engl. J. Med.* **2011**, *364* (12), 1116–1125.

(76) Rice, G. E.; Hammitt, J. K.; Evans, J. S. A probabilistic characterization of the health benefits of reducing methyl mercury intake in the United States. *Environ. Sci. Technol.* **2010**, *44* (13), S216–S224.

(77) Mahaffey, K. R.; Sunderland, E. M.; Chan, H. M.; Choi, A. L.; Grandjean, P.; Marien, K.; Oken, E.; Sakamoto, M.; Schoeny, R.; Weihe, P.; Yan, C. H.; Yasutake, A. Balancing the benefits of n-3

polyunsaturated fatty acids and the risks of methylmercury exposure from fish consumption. *Nutr. Rev.* **2011**, *69* (9), 493–508.

(78) Shimshack, J. P.; Ward, M. B. Mercury advisories and household health trade-offs. *J. Health Econ.* **2010**, *29* (5), 674–685.

(79) Hoekstra, J.; Hart, A.; Owen, H.; Zeilmaker, M.; Bokkers, B.; Thorgilsson, B.; Gunnlaugsdottir, H. Fish, contaminants and human health: Quantifying and weighing benefits and risks. *Food Chem. Toxicol.* **2012**, *54*, 18–29.

(80) Leino, O.; Karjalainen, A. K.; Tuomisto, J. T. Effects of docosahexaenoic acid and methylmercury on child's brain development due to consumption of fish by Finnish mother during pregnancy: A probabilistic modeling approach. *Food Chem. Toxicol.* **2011**, *54*, 50–58.

(81) Strom, S.; Helmfrid, I.; Glynn, A.; Berglund, M. Nutritional and toxicological aspects of seafood consumption—An integrated exposure and risk assessment of methylmercury and polyunsaturated fatty acids. *Environ. Res.* **2011**, *111* (2), 274–280.

(82) Wiener, J. G.; Spry, D. J. Toxicological significance of mercury in freshwater fish. In *Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations*; Beyer, W. N.; Heinz, G. H.; Redmon-Norwood, A. W., Eds.; Lewis Publishers: Boca Raton, FL, 1996.

(83) Sandheinrich, M. B.; Wiener, J. G. Methylmercury in freshwater fish—Recent advances in assessing toxicity of environmentally relevant exposures. In *Environmental Contaminants in Biota: Interpreting Tissue Concentrations*, 2nd; Beyer, W. N.; Meador, J. P., Eds.; CRC Press/Taylor and Francis: Boca Raton, FL, 2011; pp 169–190.

(84) Kamman, N. C.; Chalmers, A.; Clair, T. A.; Major, A.; Moore, R. B.; Norton, S. A.; Shanley, J. B. Factors influencing mercury in freshwater surface sediments of northeastern North America. *Ecotoxicology* **2005**, *14*, 101–111.

(85) Yu, X.; Driscoll, C. T.; Montesdeoca, M.; Evers, D.; Duron, M.; Williams, K.; Schoch, N.; Kamman, N. C. Spatial patterns of mercury in biota of Adirondack, New York lakes. *Ecotoxicology* **2011**, *20* (7), 1543–1554.

(86) Frederick, P. C.; Spalding, M. G.; Sepulveda, M. S.; Williams, G.; Nico, L.; Robins, R. Exposure of egret (*Ardea albus*) nestlings to mercury through diet in the Everglades ecosystem. *Environ. Toxicol. Chem.* **1999**, *18*, 1940–1947.

(87) Bowerman, W. W.; Roe, A. S.; Gilbertson, M.; Best, D. A.; Sikarskie, J. G.; Summer, C. L. Using bald eagles to indicate the health of the Great Lakes environment. *Lake Reservoir Manage.* **2002**, *7*, 183–187.

(88) Yates, D. E.; Mayack, D. T.; Munney, K.; Evers, D. C.; Major, A.; Kaur, T.; Taylor, R. J. Mercury levels in mink (*Mustela vison*) and river otter (*Lontra canadensis*) from northeastern North America. *Ecotoxicology* **2005**, *14*, 263–274.

(89) Evers, D. C.; Lane, O. P.; Savoy, L.; Goodale, W. Assessing the Impacts of Methylmercury on Piscivorous Wildlife Using a Wildlife Criterion Value Based on the Common Loon, 1998–2003, Gorham Report BRI 2004-05 submitted to the Maine Department of Environmental Protection; BioDiversity Research Institute: Gorham, ME, 2004.

(90) Evers, D. C.; Han, Y.-J.; Driscoll, C. T.; Kamman, N. C.; Goodale, W. M.; Lambert, K. F.; Holsen, T. M.; Chen, C. Y.; Clair, T. A.; Butler, T. J. Biological mercury hotspots in the Northeastern United States and Southeastern Canada. *Bioscience* **2007**, *57* (1), 1–15.

(91) Edmonds, S. T.; Evers, D. C.; O'Driscoll, N. J.; Mettke-Hofmann, C.; Powell, L. L.; Cristol, D.; McGann, A. J.; Armiger, J. W.; Lane, O.; Tessler, D. F.; Newell, P. Geographic and seasonal variation in mercury exposure of the declining Rusty Blackbird. *Condor* **2010**, *112* (4), 789–799.

(92) Evers, D. C.; Jackson, A. K.; Tear, T. H.; Osborne, C. E. Hidden risk: Mercury in terrestrial ecosystems of the Northeast; Biodiversity Research Institute: Gorham, ME, 2012.

(93) Jackson, A. K.; Evers, D. C.; Etterson, M. A.; Condon, A. M.; Folsom, S. B.; Detweiler, J.; Schmerfeld, J.; Cristol, D. A. Mercury exposure affects the reproductive success of a free-living terrestrial songbird, the Carolina wren (*Thryothorus ludovicianus*). *Auk* **2011**, *128* (4), 759–769.

- (94) Basu, N.; Scheuhammer, A. M.; Bursian, S. J.; Elliott, J.; Rouvinen-Watt, K.; Chan, H. M. Mink as a sentinel species in environmental health. *Environ. Res.* **2007**, *103* (1), 130–144.
- (95) Basu, N.; Scheuhammer, A. M.; Sonne, C.; Letcher, R. J.; Born, E. W.; Dietz, R. Is dietary mercury of neurotoxicological concern to wild polar bears (*Ursus Maritimus*)? *Environ. Toxicol. Chem.* **2009**, *28* (1), 133–140.
- (96) Lemes, M.; Wang, F. Y.; Stern, G. A.; Ostertag, S. K.; Chan, H. M. Methylmercury and selenium speciation in different tissues of beluga whales (*Delphinapterus leucas*) from the Western Canadian Arctic. *Environ. Toxicol. Chem.* **2011**, *30* (12), 2732–2738.
- (97) Dommergue, A.; Sprovieri, F.; Pirrone, N.; Ebinghaus, R.; Brooks, S.; Courteaud, J.; Ferrari, C. P. Overview of mercury measurements in the Antarctic troposphere. *Atmos. Chem. Phys.* **2010**, *10*, 3309–3319.
- (98) Drevnick, P. E.; Engstrom, D. R.; Driscoll, C. T.; Balogh, S. J.; Kamman, N. C.; Long, D. T.; Muir, D. G. C.; Parsons, M. J.; Rolffhus, K. R.; Rossmann, R.; Swain, E. B. Spatial and temporal patterns of mercury accumulation in lacustrine sediments across the Laurentian Great Lakes region. *Environ. Pollut.* **2012**, DOI: 10.1016/j.envpol.2011.05.025.
- (99) Holmes, C. D.; Jacob, D. J.; Corbitt, E. S.; Mao, J.; Yang, X.; Talbot, R.; Slemr, F. Global atmospheric model for mercury including oxidation by bromine atoms. *Atmos. Chem. Phys.* **2010**, *10* (24), 12037–12057.
- (100) Amos, H. M.; Jacob, D. J.; Streets, D. G.; Sunderland, E. M. Legacy impacts of all-time anthropogenic emissions on the global mercury cycle. *Nat. Geosci.* **In Press**.
- (101) Amos, H. M.; Jacob, D. J.; Holmes, C. D.; Fisher, J. A.; Wang, Q.; Yantosca, R. M.; Corbitt, E. S.; Galarneau, E.; Rutter, A. P.; Gustin, M. S.; Steffen, A.; Schauer, J. J.; Graydon, J. A.; St Louis, V. L.; Talbot, R. W.; Edgerton, E. S.; Zhang, Y.; Sunderland, E. M. Gas-particle partitioning of atmospheric Hg(II) and its effect on global mercury deposition. *Atmos. Chem. Phys.* **2012**, *12* (1), 591–603.
- (102) Slemr, F.; Brunke, E.-G.; Ebinghaus, R.; Kuss, J. Worldwide trend of atmospheric mercury since 1995. *Atmos. Chem. Phys.* **2011**, *11*, 4779–4787.
- (103) Corbitt, E. S.; Jacob, D. J.; Holmes, C. D.; Streets, D. G.; Sunderland, E. M. Global source-receptor relationships for mercury deposition under present-day and 2050 emissions scenarios. *Environ. Sci. Technol.* **2011**, *45* (24), 10477–10484.
- (104) Talbot, R.; Mao, H.; Scheuer, E.; Dibb, J.; Avery, M. Total depletion of Hg⁰ in the upper troposphere – lower stratosphere. *Geophys. Res. Lett.* **2007**, *34*, L23804 DOI: 10.1029/2007GL031366.
- (105) Hynes, A. J.; Donohoue, D. L.; Goodsite, M. E.; Hedgecock, I. M. Our current understanding of major chemical and physical processes affecting mercury dynamics in the atmosphere and at the air-water/terrestrial interfaces. In *Mercury Fate and Transport in the Global Atmosphere*; Mason, R., Pirrone, N., Eds.; Springer: New York, 2009; pp 427–457.
- (106) Hedgecock, I. M.; Pirrone, N. Mercury and photochemistry in the marine boundary layer-modelling studies suggest the *in situ* production of reactive gas phase mercury. *Atmos. Environ.* **2001**, *35* (17), 3055–3062.
- (107) Selin, N. E.; Jacob, D. J.; Park, R. J.; Yantosca, R. M.; Strobe, S.; Jaeglé, L.; Jaffe, D. Chemical cycling and deposition of atmospheric mercury: Global constraints from observations. *J. Geophys. Res.* **2007**, *112* (D2), D02308.
- (108) Mao, H.; Talbot, R. W.; Sive, B. C.; Youn Kim, S.; Blake, D. R.; Weinheimer, A. J. Arctic mercury depletion and its quantitative link with halogens. *J. Atmos. Chem.* **2011**, *65*, 145–170.
- (109) Obrist, D.; Tas, E.; Peleg, M.; Matveev, V.; Fain, X.; Asaf, D.; Luria, M. Bromine-induced oxidation of mercury in the mid-latitude atmosphere. *Nat. Geosci.* **2011**, *4* (1), 22–26.
- (110) Goodsite, M. E.; Plane, J. M. C.; Skov, H. A theoretical study of the oxidation of Hg⁰ to HgBr₂ in the troposphere. *Environ. Sci. Technol.* **2004**, *38* (6), 1772–1776.
- (111) Dibble, T. S.; Zelig, M. J.; Mao, H. Thermodynamics of reactions of ClHg and BrHg radicals with atmospherically abundant free radicals. *Atmos. Chem. Phys. Discuss.* **2012**, *12*, 10271–10279.
- (112) Holmes, C. D.; Jacob, D. J.; Yang, X. Global lifetime of elemental mercury against oxidation by atomic bromine in the free troposphere. *Geophys. Res. Lett.* **2006**, *33*, (20).
- (113) NASA JPL.[NASA Jet Propulsion Laboratory]. Ocean Bottom Pressure Data from GRACE Sheds Light on Ocean, Climate, 2009. <http://grace.jpl.nasa.gov/>.
- (114) Theys, N.; Van Roozendaal, M.; Hendrick, F.; Yang, X.; De Smedt, I.; Richter, A.; Begoin, M.; Errera, Q.; Johnston, P. V.; Kreher, K.; De Mazière, M. Global observations of tropospheric BrO columns using GOME-2 satellite data. *Atmos. Chem. Phys. Discuss.* **2011**, *11*, 1791–1811.
- (115) Yang, X.; Cox, R. A.; Warwick, N. J.; Pyle, J. A.; Carver, G. D.; O'Connor, F. M.; Savage, N. H. Tropospheric bromine chemistry and its impacts on ozone: A model study. *J. Geophys. Res.* **2005**, *110* (D23311), 18.
- (116) Parrella, J. P.; Jacob, D. J.; Liang, Q.; Zhang, Y.; Mickley, L. J.; Miller, B.; Evans, M. J.; Yang, X.; Pyle, J. A.; Theys, N.; Van Roozendaal, M. Tropospheric bromine chemistry: Implications for present and pre-industrial ozone and mercury. *Atmos. Chem. Phys. Discuss.* **2012**, *12* (15), 6723–6740.
- (117) Goodsite, M. E.; Plane, J. M. C.; Skov, H. Correction to A theoretical study of the oxidation of Hg-0 to HgBr₂ in the troposphere. *Environ. Sci. Technol.* **2012**, *46*, S262–S262.
- (118) Dommergue, A.; Barret, M.; Courteaud, J.; Cristofanelli, P.; Ferrari, C. P.; Galle, H. Dynamic recycling of gaseous elemental mercury in the boundary layer of the Antarctic Plateau. *Atmos. Chem. Phys. Discuss.* **2012**, *12*, 11,027–11,036.
- (119) Edgerton, E. S.; Hartsell, B. E.; Jansen, J. J. Mercury speciation in coal-fired power plant plumes observed at three surface sites in the southeastern US. *Environ. Sci. Technol.* **2006**, *40* (15), 4563–4570.
- (120) Lohman, K.; Seigneur, C.; Edgerton, E.; Jansen, J. Modeling mercury in power plant plumes. *Environ. Sci. Technol.* **2006**, *40* (12), 3848–3854.
- (121) Zhang, Y.; Jaeglé, L.; van Donkelaar, A.; Martin, R. V.; Holmes, C. D.; Amos, H. M.; Wang, Q.; Talbot, R.; Artz, R.; Brooks, S.; Luke, W.; Holsen, T. M.; Felton, D.; Miller, E. K.; Perry, K. D.; Schmeltz, D.; Steffen, A.; Tordon, R.; Weiss-Penzias, P.; Zsolway, R. Nested-grid simulation of mercury over North America. *Atmos. Chem. Phys. Discuss.* **2012**, *12*, 2603–2646.
- (122) Zhang, L.; Wright, L. P.; Blanchard, P. A review of current knowledge concerning dry deposition of atmospheric mercury. *Atmos. Environ.* **2009**, *43*, S853–S864.
- (123) Mao, H.; Talbot, R. Long-term variation in speciated mercury at marine, coastal and inland sites in New England: Part I Temporal Variability. *Atmos. Chem. Phys. Discuss.* **2012**, *12*, 5099–5112.
- (124) Malcolm, E. G.; Keeler, G. J.; Landis, M. S. The effects of the coastal environment on the atmospheric mercury cycle. *J. Geophys. Res.* **2003**, *108* (D12), ACH 4–1–ACH 4–9.
- (125) Feddersen, D.; Talbot, R.; Mao, H.; Sive, B. C. Aerosol size distribution of atmospheric mercury in marine and continental atmospheres. *Atmos. Chem. Phys. Discuss.* **2012**, *12*, 10899–10909.
- (126) Holmes, C. D.; Jacob, D. J.; Mason, R. P.; Jaffe, D. A. Sources and deposition of reactive gaseous mercury in the marine atmosphere. *Atmos. Environ.* **2009**, *43*, 2278–2285.
- (127) Mao, H.; Talbot, R.; Hegarty, J.; Koerner, J. Long-term variation in speciated mercury at marine, coastal and inland sites in New England: Part II relationships with atmospheric physical parameters. *Atmos. Chem. Phys. Discuss.* **2012**, *12*, 4181–4206.
- (128) Wang, X.; Zhang, L.; Moran, M. D. On the discrepancies between theoretical and measured below-cloud particle scavenging coefficients for rain - a numerical investigation using a detailed one-dimensional cloud microphysics model. *Atmos. Chem. Phys. Discuss.* **2011**, *11*, 11859–11866.
- (129) Rutter, A. P.; Schauer, J. J. The effect of temperature on the gas-particle partitioning of reactive mercury in atmospheric aerosols. *Atmos. Environ.* **2007**, *41* (38), 8647–8657.

- (130) Rutter, A. P.; Schauer, J. J. The impact of aerosol composition on the particle to gas partitioning of reactive mercury. *Environ. Sci. Technol.* **2007**, *41* (11), 3934–3939.
- (131) Dvonch, J. T.; Graney, J. R.; Marsik, F. J.; Keeler, G. J.; Stevens, R. K. An investigation of source–receptor relationships for mercury in south Florida using event precipitation data. *Sci. Total Environ.* **1998**, *213* (1–3), 95–108.
- (132) Selin, N. E.; Jacob, D. J. Seasonal and spatial patterns of mercury wet deposition in the United States: Constraints on the contribution from North American anthropogenic sources. *Atmos. Environ.* **2008**, *42* (21), 5193–5204.
- (133) Guentzel, J. L.; Landing, W. M.; Gill, G. A.; Pollman, C. D. Processes influencing rainfall deposition of mercury in Florida. *Environ. Sci. Technol.* **2001**, *35*, 863–873.
- (134) Biester, H.; Bindler, R.; Martinez-Cortizas, A.; Engstrom, D. R. Modeling the past atmospheric deposition of mercury using natural archives. *Environ. Sci. Technol.* **2007**, *41* (14), 4851–4860.
- (135) Roos-Barracough, F.; Martinez-Cortizas, A.; Garcia-Rodeja, E.; Shotyk, W. A 14 500 year record of the accumulation of atmospheric mercury in peat: Volcanic signals, anthropogenic influences and a correlation to bromine accumulation. *Earth Planet. Sci. Lett.* **2002**, *202* (2), 435–451.
- (136) Cooke, C. A.; Balcom, P. H.; Biester, H.; Wolfe, A. P. Over three millennia of mercury pollution in the Peruvian Andes. *Proc. Natl. Acad. Sci. U.S.A.* **2009**, *106* (22), 8830–8834.
- (137) Cole, A. S.; Steffen, A. Trends in long-term gaseous mercury observations in the Arctic and effects of temperature and other atmospheric conditions. *Atmos. Chem. Phys.* **2010**, *10* (10), 4661–4672.
- (138) Cole, A. S.; Steffen, A.; Poissant, L.; Pilote, M.; Pfaffhuber, K. A.; Berg, T. Trends in atmospheric mercury concentrations in the Northern Hemisphere: Why is the Arctic different? *Paper presented at the 10th International Conference on Mercury as a Global Pollutant*, Halifax, Canada. 2011.
- (139) Wilson, S.; Munthe, J.; Sundseth, K.; Kindbom, K.; Maxson, P.; Pacyna, J.; Steenhuisen, F. *Updating Historical Global Inventories of Anthropogenic Mercury Emissions to Air*, AMAP Technical Report 3; Arctic Monitoring and Assessment Programme: Oslo, 2010.
- (140) Mason, R. P.; Choi, A. L.; Fitzgerald, W. F.; Hammerschmidt, C. R.; Lamborg, C. H.; Soerensen, A. L.; Sunderland, E. M. Mercury biogeochemical cycling in the ocean and policy implications. *Environ. Res.* **2012**, *119*, 101–117.
- (141) Soerensen, A. L.; Jacob, D. J.; Streets, D. G.; Witt, M. L. I.; Ebinghaus, R.; Mason, R. P.; Andersson, M.; Sunderland, E. M. Multi-decadal decline of mercury in the North Atlantic atmosphere explained by changing subsurface seawater concentrations. *Geophys. Res. Lett.* **2012**, *39*, L21810.
- (142) Fitzgerald, W. F.; O'Connor, J. S. Mercury Cycling in the Hudson/Raritan River Basin. Prepared for the "Industrial Ecology, Pollution Prevention and the NY/NJ Harbor" Project of the New York Academy of Sciences. 2001.
- (143) Balcom, P. H.; Hammerschmidt, C. R.; Fitzgerald, W. F.; Lamborg, C. H.; O'Connor, J. S. Seasonal distributions and cycling of mercury and methylmercury in the waters of New York/New Jersey Harbor estuary. *Mar. Chem.* **2008**, *109*, 1–17.
- (144) Sunderland, E. M.; Amirbahman, A.; Burgess, N. M.; Dalziel, J.; Harding, G.; Jones, S. H.; Kamai, E.; Karagas, M. R.; Shi, X.; Chen, C. Y. Mercury sources and fate in the Gulf of Maine. *Environ. Res.* **2012**, *119*, 27–41.
- (145) Fisher, J. A.; Jacob, D. J.; Soerensen, A. L.; Amos, H. M.; Steffen, A.; Sunderland, E. M. Riverine source of Arctic Ocean mercury inferred from atmospheric observations. *Nat. Geosci.* **2012**, *5*, 499–504.
- (146) Fitzgerald, W. F.; Engstrom, D. R.; Mason, R. P.; Nater, E. A. The case for atmospheric mercury contamination in remote areas. *Environ. Sci. Technol.* **1998**, *32* (1), 1–7.
- (147) Kocman, D.; Horvat, M.; Pirrone, N.; Cinnirella, S. Contribution of contaminated sites to the global mercury budget. *Environ. Res.* **In Review**.
- (148) Smith-Downey, N. V.; Sunderland, E. M.; Jacob, D. J. Anthropogenic impacts on global storage and emissions of mercury from terrestrial soils: Insights from a new global model. *J. Geophys. Res.* **2010**, *115*.
- (149) Demers, J. D.; Driscoll, C. T.; Fahey, T. J.; Yavitt, J. B. Mercury cycling in litter and soil in different forest types in the Adirondack region, New York, USA. *Ecol. Appl.* **2007**, *17* (5), 1341–1351.
- (150) Obrist, D.; Johnson, D. W.; Lindberg, S. E.; Luo, Y.; Hararuk, O.; Bracho, R.; Battles, J. J.; Dail, D. B.; Edmonds, R. L.; Monson, R. K.; Ollinger, S. V.; Pallardy, S. G.; Pregitzer, K. S.; Todd, D. E. Mercury distribution across 14 U.S. forests. Part I: Spatial patterns of concentrations in biomass, litter, and soils. *Environ. Sci. Technol.* **2011**, *45*, 3974–3981.
- (151) Chen, C. Y.; Driscoll, C. T.; Lambert, K. F.; Mason, R. P.; Rardin, L.; Schmitt, C. V.; Serrell, N. S.; Sunderland, E. M. *Sources to Seafood: Mercury Pollution in the Marine Environment*; Dartmouth College Toxic Metals Superfund Research Program: Hanover, NH, 2012.
- (152) Dittman, J. A.; Shanley, J. B.; Driscoll, C. T.; Aiken, G. R.; Chalmers, A. T.; Towse, J. E.; Selvendiran, P. Mercury dynamics in relation to dissolved organic carbon concentration and quality during high flow events in three northeastern U.S. streams. *Water Resour. Res.* **2010**, *46*, W07522 DOI: 10.1029/2009WR008351.
- (153) Grigal, D. F. Inputs and outputs of mercury from terrestrial watersheds: A review. *Environ. Rev.* **2002**, *10*, 1–39.
- (154) Denkenberger, J. S.; Driscoll, C. T.; Branfireun, B. A.; Eckley, C. S.; Cohen, M.; Selvendiran, P. A synthesis of rates and controls on elemental mercury evasion in the Great Lakes Basin. *Environ. Pollut.* **2012**, *161*, 291–298.
- (155) Gilmour, C. C.; Henry, E. A.; Mitchell, R. Sulfate stimulation of mercury methylation in freshwater sediments. *Environ. Sci. Technol.* **1992**, *26*, 2281–2287.
- (156) Fleming, E. J.; Mack, E. E.; Green, P. G.; Nelson, D. C. Mercury methylation from unexpected sources: Molybdate-inhibited freshwater sediments and an iron-reducing bacterium. *Appl. Environ. Microbiol.* **2006**, *72* (1), 457–464.
- (157) Kerin, E. J.; Gilmour, C. C.; Roden, E.; Suzuki, M. T.; Coates, J. D.; Mason, R. P. Mercury methylation by dissimilatory iron-reducing bacteria. *Appl. Environ. Microbiol.* **2006**, *72* (12), 7919–7921.
- (158) Todorova, S. G.; Driscoll, C. T.; Matthews, D. A.; Effler, S. W.; Hines, M. E.; Henry, E. A. Evidence for regulation of monomethyl mercury by nitrate in a seasonally stratified, eutrophic lake. *Environ. Sci. Technol.* **2009**, *43* (17), 6572–6578.
- (159) Branfireun, B. A.; Bishop, K.; Roulet, N. T.; Granberg, G.; Nilsson, M. Mercury cycling in boreal ecosystems: The long-term effect of acid rain constituents on peatland pore water methylmercury concentrations. *Geophys. Res. Lett.* **2001**, *28* (7), 1227–1230.
- (160) Benoit, J. M.; Gilmour, C.; Heyes, A.; Mason, R. P.; Miller, C. Geochemical and biological controls over methylmercury production and degradation in aquatic ecosystems. In *Biogeochemistry of Environmentally Important Trace Elements*, ACS Symposium Series #835; Chai, Y., Braids, O. C., Eds.; American Chemical Society: Washington, DC, 2003; pp 262–297.
- (161) Aiken, G. R.; Gilmour, C. C.; Krabbenhoft, D. P.; Orem, W. Dissolved organic matter in the Florida Everglades: Implications for ecosystem restoration. *Crit. Rev. Environ. Sci. Technol.* **2011**, *41* (S1), 217–248.
- (162) Deonarine, A.; Hsu-Kim, H. Precipitation of mercuric sulfide nanoparticles in NOM-containing water: Implications for the natural environment. *Environ. Sci. Technol.* **2009**, *43* (7), 2368–2373.
- (163) Gerbig, C. A.; Kim, C. S.; Stegemeier, J. P.; Ryan, J. N.; Aiken, G. R. Formation of nanocolloidal metacinnabar in mercury-DOM-sulfide systems. *Environ. Sci. Technol.* **2011**, *45* (21), 9180–9187.
- (164) Zhang, T.; Kim, B.; Leyard, C.; Reinsch, B. C.; Lowry, G. V.; Deschusses, M. A.; Hsu-Kim, H. Methylation of mercury by bacteria exposed to dissolved, nanoparticulate, and microparticulate mercuric sulfides. *Environ. Sci. Technol.* **2012**, *46*, 6950–6958.

- (165) Graham, A. M.; Aiken, G. R.; Gilmour, C. C. Dissolved organic matter enhances microbial mercury methylation under sulfidic conditions. *Environ. Sci. Technol.* **2012**, *46*, 2715–2723.
- (166) Gu, B. H.; Bian, Y. R.; Miller, C. L.; Dong, W. M.; Jiang, X.; Liang, L. Y. Mercury reduction and complexation by natural organic matter in anoxic environments. *Proc. Natl. Acad. Sci. U.S.A.* **2011**, *108* (4), 1479–1483.
- (167) Stemberger, R. S.; Chen, C. Y. Fish tissue metals and zooplankton assemblages of northeastern US lakes. *Can. J. Fish. Aquat. Sci.* **1998**, *55*, 339–352.
- (168) Pickhardt, P. C.; Folt, C. L.; Chen, C. Y.; Klaue, B.; Blum, J. D. Algal blooms reduce the uptake of toxic methylmercury in freshwater food webs. *Proc. Natl. Acad. Sci. U.S.A.* **2002**, *99*, 4419–4423.
- (169) Chen, C. Y.; Folt, C. L. High plankton biomass reduces mercury biomagnification. *Environ. Sci. Technol.* **2005**, *39*, 115–121.
- (170) Essington, T. E.; Houser, J. N. The effect of whole-lake nutrient enrichment on mercury concentration in age-1 yellow perch. *Trans. Am. Fish. Soc.* **2003**, *132*, 57–68.
- (171) Dittman, J. A.; Driscoll, C. T. Factors influencing changes in mercury concentrations in yellow perch (*Perca flavescens*) in Adirondack lakes. *Biogeochemistry* **2009**, *93* (3), 179–196.
- (172) Chasar, L. C.; Scudder, B. C.; Stewart, A. R.; Bell, A. H.; Aiken, G. R. Mercury cycling in stream ecosystems. 3. Trophic dynamics and methylmercury bioaccumulation. *Environ. Sci. Technol.* **2009**, *43* (8), 2733–2739.
- (173) Muir, D. C. G.; Wang, X.; Yang, F.; Nguyen, N.; Jackson, T. A.; Evans, M. S.; Douglas, M.; Köck, G.; Lamoureux, S.; Pienitz, R.; Smol, J. P.; Vincent, W. F.; Dastoor, A. Spatial trends and historical deposition in mercury in Eastern and Northern Canada inferred from lake sediment cores. *Environ. Sci. Technol.* **2009**, *43*, 4802–4809.
- (174) Monson, B. A.; Staples, D. F.; Bhavsar, S. P.; Holsen, T. M.; Schrank, C. S.; Moses, S. K.; McGoldrick, D. J.; Backus, S. M.; Williams, K. A. Spatiotemporal trends of mercury in walleye and largemouth bass from the Laurentian Great Lakes region. *Ecotoxicology* **2011**, *20* (7), 1555–1567.
- (175) Drevnick, P. E.; Canfield, D. E.; Gorski, P. R.; Shinneman, A. L. C.; Engstrom, D. R.; Muir, D. C. G.; Smith, G. R.; Garrison, P. J.; Cleckner, L. B.; Hurley, J. P.; Noble, R. B.; Otter, R. R.; Oris, J. T. Deposition and cycling of sulfur controls mercury accumulation in Isle Royale fish. *Environ. Sci. Technol.* **2007**, *41* (21), 7266–7272.
- (176) Hongve, D.; Haaland, S.; Riise, G.; Blakar, I.; Norton, S. Decline of acid rain enhances mercury concentrations in fish. *Environ. Sci. Technol.* **2012**, *46* (5), 2490–2491.
- (177) Åkerblom, S.; Nilsson, M.; Yu, J.; Ranneby, B.; Johansson, K. Temporal change estimation of mercury concentrations in northern pike (*Esox lucius* L.) in Swedish lakes. *Chemosphere* **2012**, *86* (5), 439–445.
- (178) Gardfeldt, K.; Sommar, J.; Ferrara, R.; Ceccarini, C.; Lanzillotta, E.; Munthe, J.; Wangberg, I.; Lindqvist, O.; Pirrone, N.; Sprovieri, F.; Pesenti, E.; Stromberg, D. Evasion of mercury from coastal and open waters of the Atlantic Ocean and the Mediterranean Sea. *Atmos. Environ.* **2003**, *37*, S73–S84.
- (179) Laurier, F.; Mason, R. Mercury concentration and speciation in the coastal and open ocean boundary layer. *J. Geophys. Res.-Atmos.* **2007**, *112*, (D6).
- (180) Laurier, F. J. G.; Mason, R. P.; Whalin, L.; Kato, S. Reactive gaseous mercury formation in the North Pacific Ocean's marine boundary layer: A potential role of halogen chemistry. *J. Geophys. Res., [Atmos.]* **2003**, *108* (D17), Art. # 4529.
- (181) 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. An improved global model for air-sea exchange of mercury: High concentrations over the North Atlantic. *Environ. Sci. Technol.* **2010**, *44* (22), 8574–8580.
- (182) Sprovieri, F.; Pirrone, N.; Mason, R. P.; Andersson, M. Spatial and temporal trends of over-water, air-surface exchange, surface and deep water mercury measurements. In *Mercury Fate and Transport in the Global Atmosphere*; Pirrone, N., Mason, R. P., Eds.; Springer: Dordrecht, 2009; pp 323–379.
- (183) Sunderland, E. M.; Mason, R. P. Human impacts on open ocean mercury concentrations. *Global Biogeochem. Cycles* **2007**, *21*, GB4022 DOI: 10.1029/2006GB002876.
- (184) Gardfeldt, K.; Sommar, J.; Stromberg, D.; Feng, X. B. Oxidation of atomic mercury by hydroxyl radicals and photoinduced decomposition of methylmercury in the aqueous phase. *Atmos. Environ.* **2001**, *35* (17), 3039–3047.
- (185) Whalin, L.; Kim, E. H.; Mason, R. Factors influencing the oxidation, reduction, methylation and demethylation of mercury species in coastal waters. *Mar. Chem.* **2007**, *107* (3), 278–294.
- (186) Whalin, L. M.; Mason, R. P. A new method for the investigation of mercury redox chemistry in natural waters utilizing deflatable Teflon (R) bags and additions of isotopically labeled mercury. *Anal. Chim. Acta* **2006**, *558* (1–2), 211–221.
- (187) Zhang, T.; Hsu-Kim, H. Photolytic degradation of methylmercury enhanced by binding to natural organic ligands. *Nature Geosci.* **2010**, *3* (7), 473–476.
- (188) Hudson, R. J. M.; Gherini, S. A.; Watras, C. J.; Porcella, D. B. Modeling the biogeochemical cycle of mercury in lakes: The mercury cycling model (MCM) and its application to the MTL study lakes. In *Mercury Pollution: Integration and Synthesis*; Watras, C. J., Huckabee, J. W., Eds.; Lewis Publishers: Boca Raton, FL, 1994; pp 473–523.
- (189) Rajar, R.; Cetina, M.; Horvat, M.; Zagar, D. Mass balance of mercury in the Mediterranean Sea. *Mar. Chem.* **2007**, *107* (1), 89–102.
- (190) Laurier, F. J. G.; Mason, R. P.; Gill, G. A.; Whalin, L. Mercury distributions in the North Pacific Ocean - 20 years of observations. *Mar. Chem.* **2004**, *90* (1–4), 3–19.
- (191) Hudson, R.; Gherini, S.; Fitzgerald, W.; Porcella, D. Anthropogenic influences on the global mercury cycle: A model-based analysis. *Water, Air, Soil Pollut.* **1995**, *80*, 265–272.
- (192) Selin, N. E.; Sunderland, E. M.; Knights, C. D.; Mason, R. A. Sources of mercury exposure for US seafood consumers: Implications for policy. *Environ. Health Perspect.* **2010**, *118* (1), 137–143.
- (193) Selin, N. E.; Jacob, D. J.; Yantosca, R. M.; Strode, S.; Jaegle, L.; Sunderland, E. M. Global 3-D land-ocean-atmosphere model for mercury: Present-day versus preindustrial cycles and anthropogenic enrichment factors for deposition. *Global Biogeochem. Cycles* **2008**, *22*, (2).
- (194) Knauer, G. A.; Martin, J. H. Primary productivity and carbon-nitrogen fluxes in the upper 1500 m of the northeast Pacific. *Limnol. Oceanogr.* **1981**, *26*, 181–186.
- (195) Cossa, D.; Heimburger, L.-E.; Lannuzel, D.; Rintoul, S. R.; Butler, E. C. V.; Bowie, A. R.; Averty, B.; Watson, R. J.; Remenyi, T. Mercury in the southern ocean. *Geochim. Cosmochim. Acta* **2011**, *75*, 4037–4052.
- (196) Heimburger, L. E.; Cossa, D.; Marty, J. C.; Migon, C.; Averty, B.; Dufour, A.; Ras, J. Methyl mercury distributions in relation to the presence of nan- and picoplankton in an oceanic water column (Ligurian Sea, North-western Mediterranean). *Geochim. Cosmochim. Acta* **2010**, *74*, 5549–5559.
- (197) Mason, R. P.; Fitzgerald, W. F. The distribution and biogeochemical cycling of mercury in the equatorial Pacific Ocean. *Deep Sea Res., Part II* **1993**, *40* (9), 1897–1924.
- (198) Mason, R. P.; Sullivan, K. A. The distribution and speciation of mercury in the South and equatorial Atlantic. *Deep Sea Res., Part II* **1999**, *46*, 937–956.
- (199) Lehnher, I.; St Louis, V. L.; Hintelmann, H.; Kirk, J. L. Methylation of inorganic mercury in polar marine waters. *Nat. Geosci.* **2011**, *4* (5), 298–302.
- (200) Hollweg, T. A.; Gilmour, C. C.; Mason, R. P. Mercury and methylmercury cycling in sediments of the mid-Atlantic continental shelf and slope. *Limnol. Oceanogr.* **2010**, *55* (6), 2703–2722.
- (201) Lamborg, C. H.; Von Damm, K. L.; Fitzgerald, W. F.; Hammerschmidt, C. R.; Zierenberg, R. Mercury and monomethylmercury in fluids from Sea Cliff submarine hydrothermal field, Gorda Ridge. *Geophys. Res. Lett.* **2006**, *33*, (17).
- (202) Monperrus, M.; Tessier, E.; Amouroux, D.; Leynaert, A.; Huonnic, P.; Donard, O. F. X. Mercury methylation, demethylation

and reduction rates in coastal and marine surface waters of the Mediterranean Sea. *Mar. Chem.* **2007**, *107*, 49–63.

(203) Mason, R. P.; Rolffhus, K. R.; Fitzgerald, W. F. Mercury in the North Atlantic. *Mar. Chem.* **1998**, *61*, 37–53.

(204) Ferriss, B. E.; Essington, T. E. Regional patterns in mercury and selenium concentrations of yellowfin tuna (*Thunnus albacares*) and bigeye tuna (*Thunnus obesus*) in the Pacific Ocean. *Can. J. Fish. Aquat. Sci.* **2011**, *68* (12), 2046–2056.

(205) Choy, C. A.; Popp, B. N.; Kaneko, J. J.; Drazen, J. C. The influence of depth on mercury levels in pelagic fishes and their prey. *Proc. Natl. Acad. Sci. U.S.A.* **2009**, *106*, 13865–13869.

(206) Magalhaes, M. C.; Costa, V.; Menezes, G. M.; Pinho, M. R.; Santos, R. S.; Monteiro, L. R. Intra- and inter-specific variability in total and methylmercury bioaccumulation by eight marine fish species from the Azores. *Mar. Pollut. Bull.* **2007**, *54* (10), 1654–1662.

(207) Monteiro, L.; Costa, V.; Furness, R.; Santos, R. Mercury concentrations in prey fish indicate enhanced bioaccumulation in mesopelagic environments. *Mar. Ecol.: Prog. Ser.* **1996**, *141*, 21–25.

(208) Monteiro, L. R.; Furness, R. W. Accelerated increase in mercury contamination in North Atlantic mesopelagic food chains as indicated by time series of seabird feathers. *Environ. Toxicol. Chem.* **1997**, *16*, 2489–2493.

(209) Vo, A. T. E.; Bank, M. S.; Shine, J. P.; Edwards, S. V. Temporal increase in organic mercury in an endangered pelagic seabird assessed by century-old museum specimens. *Proc. Natl. Acad. Sci. U.S.A.* **2011**, *108* (18), 7466–7471.

(210) Gehrke, G. E.; Blum, J. D.; Slotton, D. G.; Greenfield, B. K. Mercury isotopes link mercury in San Francisco Bay forage fish to surface sediments. *Environ. Sci. Technol.* **2011**, *45*, 1264–1270.

(211) Senn, D. B.; Chesney, E. J.; Blum, J. D.; Bank, M. S.; Maage, A.; Shine, J. P. Stable Isotope (N, C, Hg) Study of Methylmercury Sources and Trophic Transfer in the Northern Gulf of Mexico. *Environ. Sci. Technol.* **2010**, *44* (5), 1630–1637.

(212) Macdonald, R. W.; Wang, F. Y.; Stern, G.; Outridge, P. The overlooked role of the ocean in mercury cycling in the Arctic. *Mar. Pollut. Bull.* **2008**, *56* (12), 1963–1965.

(213) Mason, R. P.; Pirrone, N.; Hedgecock, I.; Suzuki, N.; Levin, L. Conceptual overview. In *Hemispheric Transport of Air Pollution 2010: Part B: Mercury*, Air Pollution Studies #18; Pirrone, N., Keating, T., Eds.; United Nations: New York and Geneva, 2010; pp 1–26.

(214) Driscoll, C. T.; Chen, C. Y.; Hammerschmidt, C. R.; Mason, R. P.; Gilmour, C. C.; Sunderland, E. M.; Greenfield, B.; Lamborg, C. H. Nutrient supply and mercury dynamics in marine ecosystems: A conceptual model. *Environ. Res.* **2012**, *119*, 118–131, DOI: 10.1016/j.envres.2012.05.002.

(215) Matthews, D. A.; Babcock, D. B.; Nolan, J. G.; Prestigiacomo, A. R.; Effler, S. W.; Driscoll, C. T.; Todorova, S.; Kuhr, K. M. Whole-lake nitrate addition for control of methylmercury in Onondaga Lake, NY. *Environ. Res.*

(216) Lambert, K. F.; Evers, D. C.; Warner, K. A.; King, S. L.; Selin, N. E. Integrating mercury science and policy in the marine context: Challenges and opportunities. *Environ. Res.* **2012**, *119*, 132–142.

(217) Harris, R.; Krabbenhoft, D. B.; Mason, R.; Murray, M. W.; Reash, R.; Saltman, T. *Ecosystem Responses to Mercury Contamination: Indicators of Change*. SETAC; CRC Press: Boca Raton, FL, 2007.

(218) Mason, R. P.; Abbott, M. L.; Bodaly, R. A.; Bullock, J., O. R.; Driscoll, C. T.; Evers, D.; Lindberg, S. E.; Murray, M.; Swain, E. B. Monitoring the response to changing mercury deposition. *Environ. Sci. Technol.* **2005**, *39*, 15–22A.

(219) Sprovieri, F.; Hedgecock, I. M.; Pirrone, N. An investigation of the origins of reactive gaseous mercury in the Mediterranean marine boundary layer. *Atmos. Chem. Phys.* **2010**, *10* (8), 3985–3997.

(220) Feng, X.; Li, P.; Qiu, G.; Wang, S.; Li, G.; Shang, L.; Meng, B.; Jiang, H.; Bai, W.; Li, Z.; Fu, X. Human exposure to methylmercury through rice intake in mercury mining areas, Guizhou Province, China. *Environ. Sci. Technol.* **2008**, *42*, 326–332.