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Human activities shape important geographic differences in fish mercury concentration levels

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Fish consumption is a major route of human exposure to mercury (Hg), yet limited understanding of how anthropogenic activities drive geographic variations in fish Hg worldwide hinders effective Hg pollution management. Here we characterized global geographic variations in total Hg (THg) and methylmercury (MeHg), compared THg and MeHg levels between the United States and China, and used a structural equation model to link the geographic variability of MeHg in fish to human activities. Despite previously reported higher Hg emissions in China, Chinese fish have lower THg and MeHg levels than fish in the United States owing to a lower trophic magnification slope, shortened food chains and shorter fish lifespans. The structural equation model revealed strong impacts of human activities on MeHg levels in fish. In the future, China may face elevated MeHg levels in fish with the ongoing recovery of food web ecology, highlighting the importance of local policies.

Mercury (Hg) is a global pollutant that can undergo long-range atmospheric transport[1]. Once it enters the environment, Hg can be microbially methylated to the neurotoxin methylmercury (MeHg)[2]. MeHg biomagnifies along food webs and bioaccumulates to high levels in predatory fish[3], posing high health risks to wildlife and humans. The diminished reproduction and survival of fish-eating birds have been attributed to MeHg poisoning[4]. The Hg, especially MeHg, levels in fish play a central role in human Hg exposure, as fish is the principal nutrient source of high-quality proteins, unsaturated fatty acids and micronutrients[5]. A wide range of fish Hg has been observed in different regions of the world, and understanding the driving forces contributing to the variability in fish Hg accumulation from aquatic ecosystems, especially at a large, geographic scale, is particularly useful for Hg pollution management.

Fish Hg can be directly related to ambient Hg and MeHg concentrations, fish traits and ecology including Hg concentrations at the base of food webs[6], the trophic status of a given system[7] and the food chain length[8], and the trophic magnification efficiency of Hg related to the age, body size and growth rate of individuals[9]. These factors are affected by human activities (consequently by socio-economic development), which are deemed to affect fishery, environmental landscape and Hg biological cycling[3,10]. However, the potential direct and indirect effects of these factors on fish Hg accumulation remain unquantified. Environmental Hg pollution and Hg bioaccumulation in fish have

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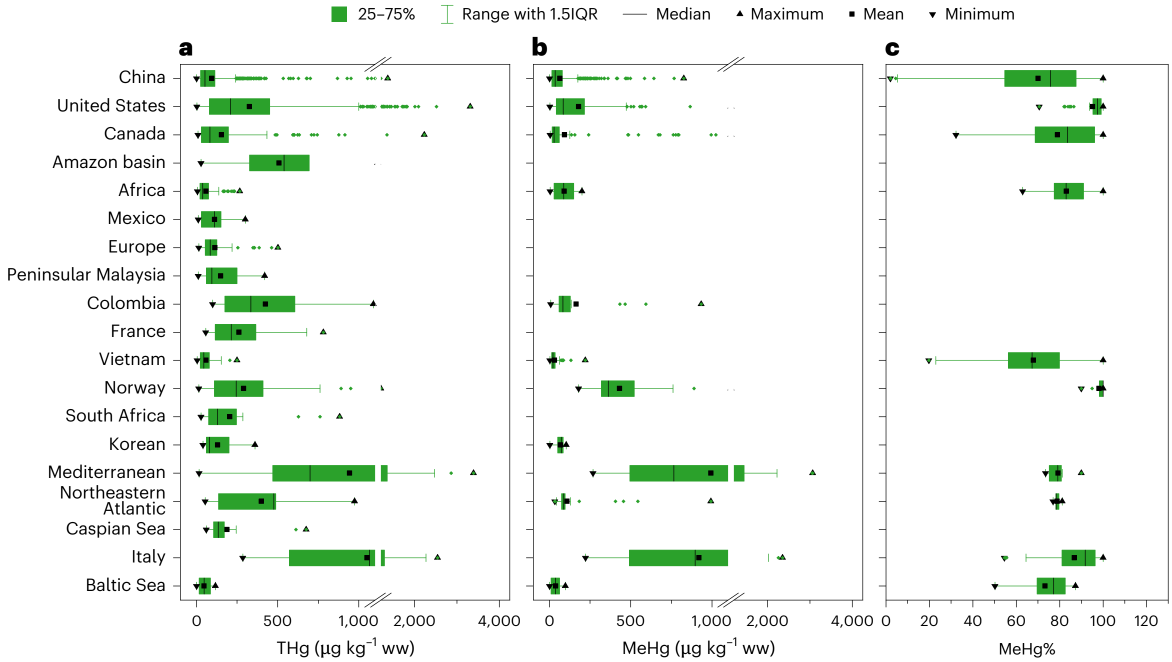

Fig. 1 | Geographic patterns of fish Hg concentrations from different regions worldwide. a–**c**, The datasets contain information on fish THg (**a**), MeHg (**b**) and MeHg% (**c**) concentrations from 315 aquatic systems worldwide, including 86 in China and 106 in the United States. The locations of these aquatic systems are shown in the source data and Extended Data Fig. 1. IQR, interquartile range; ww, wet weight.

been widely observed across the world, but with broad geographic variations[1,9,11], as exemplified by the puzzling patterns of low fish Hg versus elevated Hg levels in environmental matrices in China[1,12,13]. For instance, fish Hg levels in most newly constructed reservoirs in China are relatively lower than those in North America and Europe, whereas water and sediment Hg in these Chinese reservoirs are comparable to or even higher than those in the latter[14].

The global nature of Hg pollution through long-range atmospheric transport and the worldwide health concern of Hg exposure from fish consumption promote international action on Hg emission reduction[15], with the Minamata Convention on Mercury being the concerted effort[1]. To facilitate effective evaluation of the convention, it is imperative to provide comparable monitoring data on the presence and migration of Hg and its compounds in the environment, biota and vulnerable populations. Comparable monitoring data are also essential to assess the effects of Hg on human and ecosystem health, including fish and other wildlife[15]. Given the critical role of aquatic food chains in estimating human and wildlife Hg exposure, a comprehensive data analysis of global fish Hg concentrations is of great importance to public health, in decision-making and particularly in assessing the long-term effectiveness of the Minamata Convention.

The present state of Hg contamination in fish in China has been documented for the past three decades[11,16], while the factors contributing to generally low fish Hg levels have been rarely explicitly evaluated. In this study, we incorporate the geographic patterns of fish Hg on a global scale and quantify the possible factors controlling fish Hg accumulation. We first overview the geographic variations of total Hg (THg), MeHg and the percentage of MeHg to THg (MeHg%) in fish from typical regions worldwide. Then, we conduct a comparative analysis between China and the United States to examine the differences in fish Hg, fish species traits and environmental factors. On these bases, we propose a 'top-down' approach and apply structural equation modelling (SEM) to relate geographical variability in fish MeHg to human activity, the ensuing Hg pollution, food web ecology and watershed characteristics. Thus, factors controlling fish Hg accumulation resulting from human activities are quantified, with the aim of providing implications for policymakers to reduce the Hg exposure risk in humans and wildlife.

Results

Hg concentrations in fish worldwide

THg, MeHg and MeHg% in fish vary widely for a given country or geographic area worldwide (Fig. 1 and Extended Data Fig. 1). The average fish THg and MeHg in many regions are below the fish consumption reference value (500 μg kg^{-1}) in the US Environmental Protection Agency and Food and Drug Administration 2017 guidance. Nevertheless, elevated Hg levels (>500 μg kg^{-1}) can be seen in many cases, but especially in some regions that suffered from historic Hg contaminations of mining activities (Supplementary Note 1). In particular, we take China (Fig. 2) and the United States (Fig. 3) as a comparative case study. The extensive studies on fish Hg accumulation available for these two large countries with similar areas and latitudes allow us to better understand how fish characteristics, Hg pollution, environmental factors and watershed conditions affect fish Hg.

Comparison of fish Hg between China and the United States

When all fish data are combined, the average fish THg and MeHg (92.6 ± 128 μg kg^{-1} and 58.2 ± 83.2 μg kg^{-1}, respectively) in China are significantly ($P < 0.05$) lower than those in the United States (267 ± 233 μg kg^{-1} and 148 ± 211 μg kg^{-1}, respectively) (Fig. 4a,b). For both regions, there are wide ranges of THg (1.05–1,360 μg kg^{-1} in China versus 2.40–3,318 μg kg^{-1} in the United States) and MeHg (0.30–827 μg kg^{-1} in China versus 2.93–1,196 μg kg^{-1} in the United States) as the data cover a variety of fish species (for example, freshwater versus marine, wild versus farmed, herbivore versus carnivore and so on). Even when considering only wild fish (including freshwater and marine fish) owing to the limited Hg data in farmed fish for the United States, the pattern of higher fish Hg in the United States still holds true ($P < 0.05$) (Fig. 4a,b). While there are no differences in marine fish MeHg% (87 ± 13% in China versus 85 ± 18% in the United States) between the two countries, significantly ($P < 0.05$) lower MeHg% in freshwater fish is observed in China (69 ± 23%) than in the United States (94 ± 7.2%) (Fig. 4c). Detailed fish

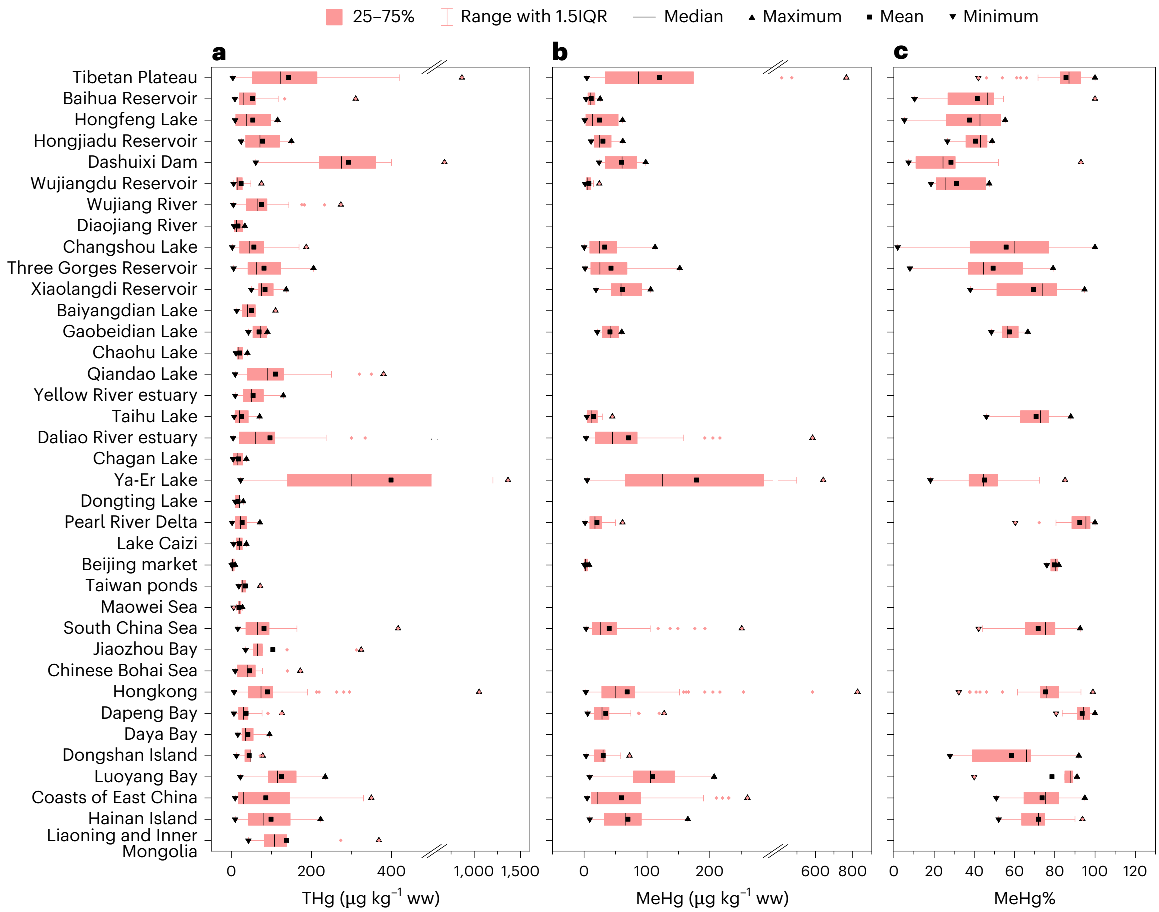

Fig. 2 | Geographic patterns of fish Hg concentrations from different regions of China. a–c, The datasets contain information on fish THg (**a**), MeHg (**b**) and MeHg% (**c**) concentrations from 86 aquatic systems in China. Detailed characteristics of fish Hg in China are presented in Supplementary Notes 2 and 3, in which an in-depth analysis of the effect of dietary behaviours (Extended Data Fig. 2), the intra- and interspecies differences in fish Hg (Extended Data Fig. 3) and fish farming (Extended Data Fig. 4) on Hg accumulation is given. The locations of these aquatic systems are shown in the source data and Extended Data Fig. 1c. IQR, interquartile range; ww, wet weight.

Hg characteristics of China and the United States are presented in Supplementary Notes 2 and 3 (Extended Data Figs. 2–4) and 4, respectively.

Differences in food web ecology between China and the United States

In China, the trophic level (TL) values of wild fish range from 1.58 to 4.90 for freshwater and from 1.50 to 4.43 for marine systems, respectively. In the United States, the TL values of freshwater and marine fish are in the range of 1.53–5.56 and 2.30–4.83, respectively. They both have a longer food chain in freshwater sites ($P < 0.05$). The median TL values in China (3.22 for freshwater and 2.71 for marine species) are significantly lower than those in the United States (3.60 for freshwater and 3.60 for marine species; Extended Data Fig. 5).

Trophic magnification slopes (TMS) of both THg and MeHg in China (0.37 ± 0.09 and 0.43 ± 0.10) are significantly lower than those in the United States (0.49 ± 0.06 and 0.49 ± 0.06) (Fig. 5). Overall, these significant and positive TMS suggest that both THg and MeHg are biomagnified through the food webs in China and the United States. The estimated concentrations of THg and MeHg at the base of the food webs in China (primary consumer, TL = 2) are 86.7 ± 1.21 μg kg^{-1} dry weight (dw) and 57.2 ± 1.22 μg kg^{-1} dw, respectively, which are also significantly lower than those in the United States (101 ± 1.34 μg kg^{-1} dw and 93.1 ± 1.35 μg kg^{-1} dw) (Fig. 5). These results are in line with the field reports of zooplankton Hg in China and the United States (Supplementary Table 1). The estimated ages of the most frequently caught fish in China are in the range of 1 to 3 years, with a few collected from the Tibetan Plateau being older than 10 years. In the United States, however, the estimated ages of the most collected fish are older than 3 years, with a substantial fraction being over 15 years (Extended Data Fig. 6).

Effect of fish farming on fish Hg accumulation

The prevalent aquaculture in China provides a unique opportunity to assess the effect of fish farming on fish Hg levels. For freshwater sites, significantly higher THg, MeHg and MeHg% are found in wild fish than in farmed fish ($P < 0.01$), while farmed fish contain higher THg and MeHg than wild ones do for marine species (Extended Data Fig. 4). The variation in fish food sources and food web structures between freshwater and marine systems may contribute to these differences (Supplementary Note 5). Notably, distinct Hg patterns emerge in freshwater versus marine systems, suggesting that the impacts of fish farming on Hg accumulation in fish vary according to the regional aquaculture environments.

Factors impacting fish Hg levels

We propose a top-down approach to account for the geographical variability in fish Hg (Fig. 6a), in which the modulators affecting fish Hg accumulation are classified into tiers. The top tier is the socio-economic development status of a country (region), which determines the watershed (hydrosphere) landscape and Hg emission in the region (Supplementary Note 6). Combining with the data we obtained, we further construct a structural equation model to explain the influencing factors

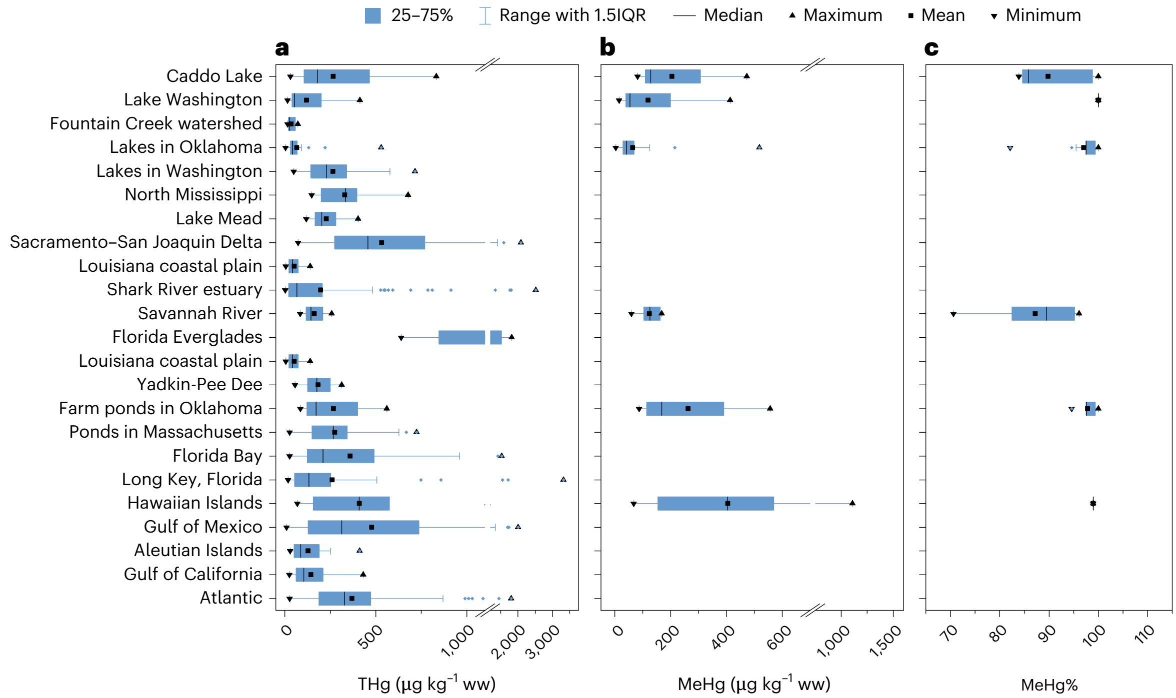

Fig. 3 | Geographic patterns of fish Hg concentrations from different regions of the United States. a–**c**, The datasets contain information on fish THg (**a**), MeHg (**b**) and MeHg% (**c**) concentrations from 106 aquatic systems in the United States. Detailed characteristics of fish Hg in the United States are presented in Supplementary Note 4. The locations of these aquatic systems are shown in the source data and Extended Data Fig. 1b. IQR, interquartile range; ww, wet weight.

of fish MeHg accumulation (Fig. 6b). The SEM path network fits reasonably well, with $P \leq 0.001$ for all paths and 0.049 as the standardized root mean squared residual. Fish size is the strongest factor (standardized path coefficient (β) = 0.63) that directly impacts fish Hg levels. It is closely related to the trophic level of fish (β = 0.24), water trophic status (that is, oligotrophic, mesotrophic and eutrophic; β = −0.13) and watershed type (β = −0.47), which in turn are governed largely by human activity (β = −0.36, 0.61 and 0.14, respectively). Trophic levels of fish also significantly positively impact fish MeHg levels (β = 0.11). Besides, this model suggests that ambient Hg (Supplementary Table 2) is exacerbated by human activity (β = 0.88) but has a negative correlation (β = −0.49) with fish MeHg levels. Under the modulating of eutrophication, a positive correlation is observed between the water trophic status–ambient Hg interactions (β = 0.21).

Discussion

The paradox of Hg in environments and fish accumulation

The anomalously low levels of THg and MeHg found in Chinese fish are paradoxical to the elevated environmental Hg in China[1,12,13,16]. The gaseous and particulate Hg concentrations in the atmosphere in urban areas of China are reported to be up to five times and two orders of magnitude higher, respectively, compared with the corresponding settings in North America and Europe[17]. Despite large variations in Hg concentrations in aquatic ecosystems depending on the aquatic feature type (for example, river, lake and coastal) or even within a specific system, the water and sediment THg concentrations in China are generally higher than those in the United States, especially in urban settings[18,19]. For instance, water THg in major Chinese rivers can be two to three times higher than those for large rivers elsewhere in the world[20,21]. Large-scale surveys suggest that the average THg concentrations in lake, river and coastal sediments are also higher in China, compared with those in the United States and Canada[12,19]. Furthermore, the MeHg% in a considerably large proportion of Chinese fish is even <50% (refs. 11,13), which is much different from the commonly reported MeHg consisting of more than 85% of THg in fish worldwide[22]. This could be due to the lower trophic level and shorter lifespan of Chinese fish, as MeHg retention increases with trophic position and age[23].

Lower TMS and baseline Hg reduce fish Hg accumulation

The differences observed in THg TMS and MeHg TMS between China and the United States are consistent with those reported in a previous study that compiled 205 aquatic food webs worldwide and found high variations in TMS values across aquatic systems[9]. As fish obtain Hg burdens predominantly from diets[6], the lower baseline Hg may serve as a driving factor for lower fish Hg in China, as supported by previously well-corroborated positive relationships between fish and baseline organism Hg (ref. 24). Hg concentrations in baseline organisms are determined by multiple factors, of which the availability and methylation potential of Hg in environmental matrices (for example, sediments and water), as well as other associated physicochemical factors of a given system (for example, water pH, temperature and dissolved organic matter), are key determinants[7]. In some circumstances, in which the habitat-related differences in Hg methylation vary greatly, the role of baseline Hg levels may override the effect of trophic levels on fish[25].

Shortened food chain lengths reduce fish Hg in China

The trophic positions of fish species play a crucial role in Hg accumulation. The lower fish TL values of China than those of the United States are indicative of significantly shorter lengths of food chains in Chinese aquatic systems relative to the United States, contributing to the lower fish Hg in China. Fish Hg increasing with the relative trophic level is consistent with the known fact of Hg biomagnification in food webs[9], and therefore, the length of food chains in a given system exerts powerful effects on Hg accumulation, especially in top predators. The effect of food chain length on Hg bioaccumulation is also evident when comparing wild freshwater fish with marine species in China.

The relatively shorter food chains in China can be attributed to the declined mean trophic level in most regions, caused by overfishing and

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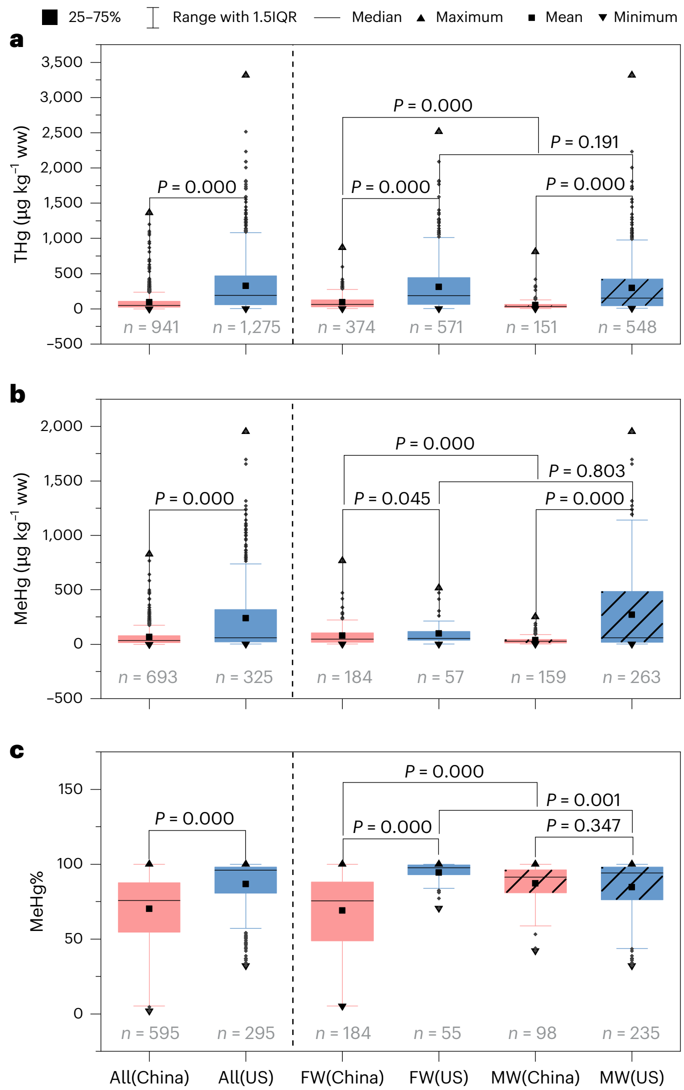

Fig. 4 | Comparison of fish Hg concentrations between China and the United States. a–c, Concentrations of THg (**a**), MeHg (**b**) and MeHg% (**c**) for all fish (All), freshwater wild fish (FW) and marine wild fish (MW) across China and the United States. Detailed sample sizes are provided in the source data; *n* represents the number of data points per graph. A Mann–Whitney *U* test was performed to compare fish Hg levels. Based on Monte Carlo analysis (two sided), a *P* value less than 0.05 indicates statistical significance. IQR, interquartile range; ww, wet weight.

environmental pollution (Supplementary Note 7)[26,27], and the imbalance of whole-ecosystem primary production because of eutrophication (Supplementary Note 8).

Shorter-lived, fast-growing fish accumulate less Hg

The significantly shorter lifespans (age or consequently body size) in China than those in the United States are consistent with the observed Hg patterns, suggesting the role of exposure duration in fish Hg accumulation. Indeed, the importance of age on the trophic transfer of Hg has been evidenced by both field and laboratory studies[28], and positive correlations are frequently observed between Hg accumulation and fish size[6,29].

In addition, for species at a given age and prey intake, individuals that have faster growth will have lower Hg concentrations than the slow-growing ones, as a higher growth rate would dilute Hg concentrations in fish bodies, a well-known somatic growth dilution (SGD) effect[30]. The faster growth rates of fish (cm yr^{-1}) observed in China compared with those in the United States imply that Hg in Chinese fish should be more affected by the SGD effect. The current finding clearly suggests that the faster growth rates play a role in the observed lower fish Hg in China, agreeing well with previous studies on the SGD effect of Hg accumulation[31].

SEM interpreting geographical variations in fish Hg

Human activity strongly impacts fish MeHg levels by exacerbating ambient Hg and environmental eutrophication, and/or by altering fish traits and characteristics. While the United States is a well-developed country socio-economically, China is considered a developing country but has experienced rapid economic and social changes over the past two decades. These changes have resulted in alterations in land use and landscape, rapid establishment and expansion of manufacturing facilities, increased Hg emission and environmental deterioration (for example, eutrophication of water bodies). For instance, the manufacturing output of China began to overtake that of the United States in 2008[32], resulting in greater Hg emissions[33] and deposition fluxes[17] in China than in the United States. However, atmospheric Hg deposition only partially reflects the bioavailable Hg (especially MeHg) in the ambient environment. Other factors also affect bioavailable MeHg, including runoff Hg inputs, Hg methylation potential and watershed-specific characteristics[2]. Consequently, despite human activity exacerbating Hg emissions, ambient Hg is negatively correlated with fish MeHg. This intriguing result corresponds to the paradox of lower fish Hg but elevated environmental Hg levels in China[1,12,13].

The recent rapid development in China also leads to environmental deterioration as exemplified by eutrophication occurring in approximately 75% of major lakes in China[34]. A survey of 240 US lakes in 2007 finds that only 23% of them are eutrophic or hypereutrophic[35]. Generally, eutrophication decreases MeHg in fish through bloom dilution[36], but could also promote MeHg production through the role of algal organic matter, thereby enhancing fish MeHg accumulation[37,38], as observed here. In the future, if appropriate variables that related to MeHg production (ultimately bioavailable MeHg) of each ecosystem are simultaneously accessible, they could be incorporated into the SEM path networks, and the pathways of ambient Hg on fish MeHg levels would probably be better illustrated.

The fish traits and characteristics (that is, lower food base Hg, shortened food chain length and enhanced growth rate) contributing to the lower Hg concentrations in China are related to hydrosphere landscape and watershed characteristics associated with socio-economic development status. Overfishing and eutrophication could have contributed to the declined size and trophic levels of fish (hence shortened food chain lengths) and enhanced fish growth rates in China[26,27]. The lower Hg in baseline organisms may be attributed to watershed characteristics and eutrophication. Watershed land-use activities (for example, reservoir creation and urbanization) could alter Hg mobility and Hg methylation, affecting Hg accumulation in biota[39]. In China, owing to climate change and the construction of dams and reservoirs, the area of marshes decreased by 4.8% from 2000 to 2015[40], while the areas or the representation by saltwater and freshwater wetlands did not statistically change[41]. In the United States, however, the national estimates of wetland areas did not statistically change between 2004 and 2009, nor did the representation by saltwater and freshwater components, although silviculture, rural development, urban development and agriculture resulted in the gain or loss of some types of wetland[41]. The decreasing areas of marsh wetlands in China[40], the hotspots for MeHg production and accumulation[42], could reduce the MeHg pool available to baseline organisms. The relatively higher eutrophication in Chinese aquatic systems could reduce Hg accumulation in fish owing to the bio-dilution effect of primary producers (baseline organisms)[43], although contrary effects may also be present[37].

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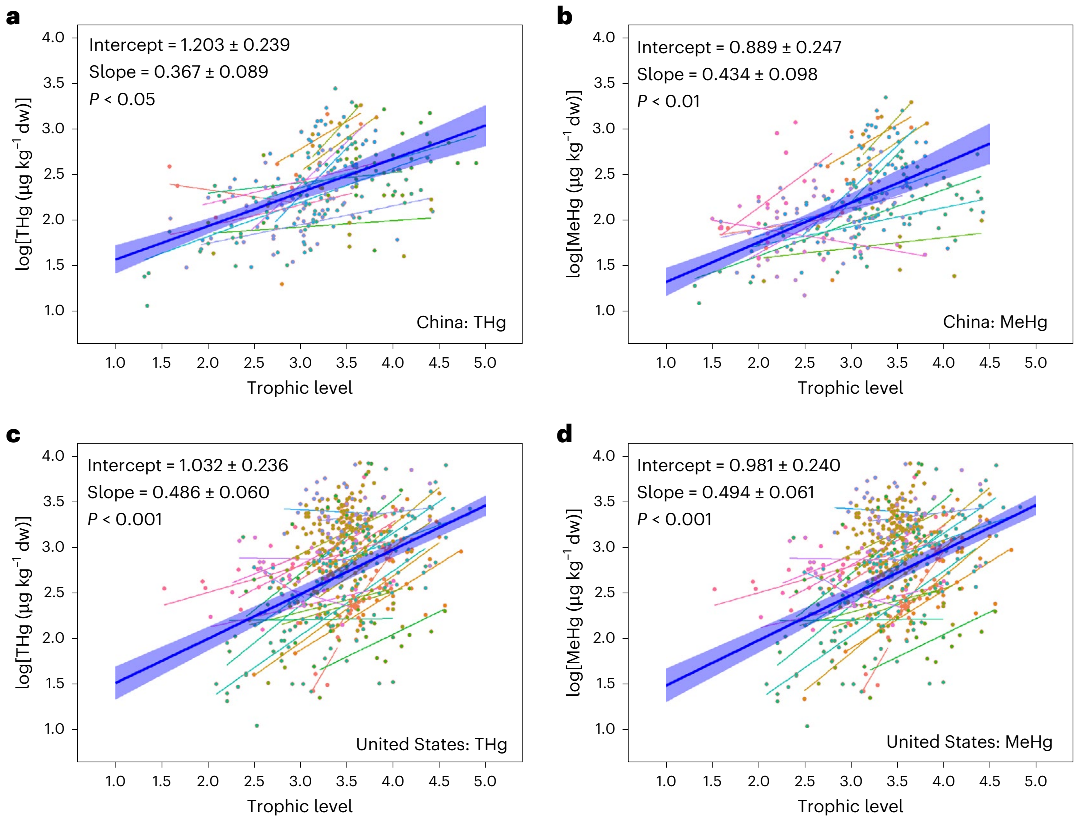

Fig. 5 | TMS, baseline (the intercept) and primary consumer (TL_2) Hg concentration of China and the United States. a–d, Linear mixed models were used to estimate the TMS and baseline Hg concentration in China (**a** and **b**) and the United States (**c** and **d**) at the national scale, with trophic level as the fixed factor and sampling site as the random factor ($n = 935$). Different coloured points and lines represent data from different watersheds. The intercept and slope are present as the estimated mean ± s.d. derived from linear mixed models. The blue solid line represents the predicted fit for each nation, and the blue shaded area indicates 95% confidence level.

Managing global Hg issue considering regional variability

From the differences in the socio-economic development processes and distinct geographical variability in Hg between China and the United States, it is noteworthy that China may face a dilemma of elevated fish MeHg and ecological risks with the ongoing recovery of food web ecology, despite concurrent Hg emission reduction. With the rapid socio-economic development of China, these ecosystem restoration efforts may generate conducive environments for MeHg production, alter hydrology and primary productivity, and increase trophic lengths and fish sizes, facilitating MeHg bioaccumulation[44]. This situation may be an analogue to the Hg dilemma currently present in the Florida Everglades in the United States, where the entire ecosystem (over 10,000 km^2) is under fish consumption advisory owing to excessive Hg levels from the system's propensity for Hg methylation and bioaccumulation despite the nearly 90% decrease in the local Hg emission[45,46]. Similar attention should be given to other developing countries worldwide and other toxic compounds with bioaccumulative effects, for example, persistent organic pollutants[47].

While managing Hg contamination as a global issue, it is imperative to consider regional variabilities. On the one hand, a concerted international effort towards Hg emission control is beneficial to managing global Hg pollution, as with the case of the Minamata Convention. On the other hand, regional action may play a remarkable role in abating the environmental, ecological and health impacts of Hg. As the United States has a significantly lower Hg emission than China does, domestic policies and actions such as watershed management and dietary advice could contribute strongly to local benefits associated with Hg exposure and risk management, complementing the measures on Hg emission control. A previous assessment of the potential human-health-related economic benefits of Hg control has suggested that domestic actions are important for protecting the populations consuming mainly local freshwater fish in the United States, although global actions on Hg emission control exert greater benefits for the US population as a whole[48]. For China, Hg emission control may be a priority as global Hg controls associated with the Minamata Convention have been suggested to have large benefits in Asia[48]. Other policies, such as those on fishery culture and watershed management, also need to be considered for their effect on Hg contamination and bioaccumulation. These measures, such as ecosystem restoration and dam construction, could alter hydrology, primary productivity, species assemblages and food web structure in a watershed, hence affecting Hg mobilization in the environment and along the food chain[44]. In addition, it is imperative to direct attention towards the frequent regional fish consumers, as the MeHg exposure risk to humans is not determined solely by fish MeHg levels but also by the amount of fish consumed[49].

It is also imperative to manage Hg pollution in the context of climate change, as Hg emissions may come from the same major carbon-emission sources (for example, fossil fuel combustion)[10]. In fact, China's recent implementation of carbon-neutrality policies in response to the Paris Agreement on climate change has contributed to the reduction of Hg emissions[50]. Aside from the perspective of simultaneous control of carbon and Hg emissions, climate-change-driven alterations on the environment and biological communities have profound effects on Hg cycling and bioaccumulation, the understanding of which plays an important role in developing policy options addressing both global issues.

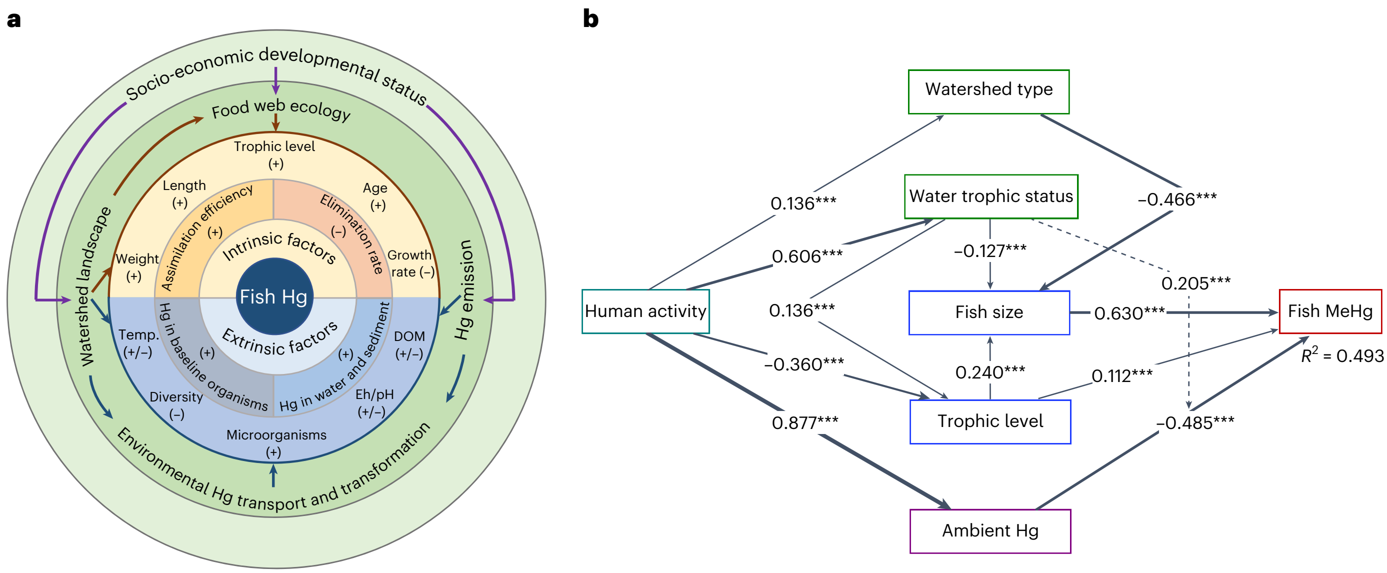

Fig. 6 | Factors affecting the Hg bioaccumulation process in fish. a, A top-down approach accounting for the geographical variability in fish Hg. These can be grouped into (1) intrinsic factors, such as assimilation efficiency and Hg elimination rate, and (2) extrinsic factors, affecting Hg levels in fish prey, including Hg in water, sediments and baseline organisms. The plus and minus symbols denote the enhancing or reducing effect on fish Hg concentrations, respectively. Eh represents the oxidation–reduction potential. **b**, SEM showing the direct or indirect effects of human activity (UDF 1), food web ecology (fish size and trophic level), watershed characteristics (watershed type and water trophic status) and ambient Hg (Hg deposition) on fish Hg levels. The models incorporate 704 fish samples from 20 Chinese and 20 American aquatic systems, with the Hg data being unnormalized by fish size or species. Solid arrows to fish MeHg represent direct pathways; others indicate indirect pathways. Dashed arrows show modulating effects. Arrow thickness signifies the relationship strength when significant, and no arrow indicates non-significant effects. Adjacent numbers are significant path coefficients. R^2 represents explained variance; ***$P < 0.001$ (two-sided t-test).

Methods

Data acquisition

We constructed an observation-based database of fish Hg from 315 aquatic systems worldwide, including 86 in China and 106 in the United States. This database was based on the extraction of research observations from peer-reviewed publications, including THg, MeHg and MeHg% in fish and ancillary parameters linked to Hg accumulation (Source Data). We first extracted Hg data and fish species from research articles (ending April 2019) indexed in the Web of Science (http://apps.webofknowledge.com) and the China Knowledge Resource Integrated Database (https://www.cnki.net/), using the following keywords: Mercury, Fish, Biomagnification, Nitrogen isotope, Mercury AND China, Mercury AND United States. Published articles were included in the final datasheet if they met the following criteria: (1) the study must be an original research on the Hg bioaccumulation in fish; (2) clear analytical methods for THg or MeHg in fish were provided; and (3) original data could be extracted from tables and figures.

We further restricted our analysis to data on wild species when investigating the factors that influence Hg accumulation in fish, owing to the limited farmed fish from regions other than China. From each of these studies, we compiled information on THg, MeHg and MeHg% values for sampled fish when available. Commonly, raw data were collected as much as possible, and if not, average concentrations were used. If the data were shown in graphs, the GetData Graph Digitizer (v2.20) was applied to extract the related data. All the fish Hg concentrations were based on wet weight (ww) unless stated otherwise, and an empirical coefficient of 4.7 was applied to convert dry-weight-based Hg concentrations in the literature if there were no data for the moisture content[51]. The calculated MeHg% values that exceeded 100% owing to propagation of analytical errors in the separate analysis of THg and MeHg were capped at 100% for plotting and statistical analysis.

We also gathered other available information linked to Hg accumulation from the retrieved publications, including the food web ecology (for example, species, age, length, weight, nitrogen isotope, trophic level and dietary habits of individuals), sampling size, the watershed landscape (for example, watershed type and water trophic status) and ambient Hg levels. Due to limited data, we used annual growth length (length/age) to denote fish growth rate, despite the potential bias brought by species differences.

In addition, we estimated atmospheric Hg concentration (that is, surface Hg(0), Hg(II) and TGM) and annual Hg deposition (that is, Hg(0) dry deposition, Hg(II) wet and dry deposition) using the GEOS-Chem global atmospheric chemistry model. The current standard version of the model for Hg and related mechanisms were described elsewhere[52]. We developed the urban distance factor (UDF) to represent the influence of human activities on fish Hg accumulation, as the emission, transport and transformation of Hg; watershed characteristics; and food web structure were largely anthropogenically controlled. The UDF was estimated based on the population (P) and distances (D) between the sampling sites and potential source cities (Supplementary Table 3)[53], and details on the calculation process and the final UDF database are provided in Supplementary Table 4 and Note 9.

Calculation of trophic magnification factor through food chains

Higher THg and MeHg concentrations have been generally observed in predators at higher TLs in food webs. Stable nitrogen isotopes ($\delta^{15}N$) could be a powerful tool used to explore the biomagnification of Hg through food webs, with the heavy isotope (^{15}N) increasing from prey to predator[54,55]. For an individual system, the Hg biomagnification potential could be reflected by the TMS, which was characterized as a simple linear relationship between Hg concentrations and TL (or $\delta^{15}N$):

$$\log_{10}[\mathrm{Hg}] = \mathrm{TL} \times b_1 + a_1 \quad (1)$$

$$\text{or } \log_{10}[\mathrm{Hg}] = \delta^{15}\mathrm{N} \times b_2 + a_2 \quad (2)$$

where the slope b_1 or b_2 (TMS) was the measure of the biomagnification of Hg through the entire food webs and the intercept (a_1 or a_2) was typically representative of the Hg concentration at the base of food webs (primary producer). A positive TMS (TMS > 0) indicated Hg bioaccumulation along food chains. In the case of equation (1), TL can be transferred from $\delta^{15}N$ using the following equation[54,55]:

$$TL_{consumer} = (\delta^{15}N_{consumer} - \delta^{15}N_{baseline})/\Delta^{15}N + \lambda \quad (3)$$

where $TL_{consumer}$ was the TL of a given consumer, λ was the TL of the baseline organism ($\lambda = 1$ for the primary producer and $\lambda = 2$ for the primary consumer), and $\delta^{15}N_{consumer}$ and $\delta^{15}N_{baseline}$ were the $\delta^{15}N$ values of the given and the baseline organism, respectively. $\Delta^{15}N$ was the trophic discrimination factor for $\delta^{15}N$, which was frequently given as 3.4‰ in the literature.

In addition, the degree of Hg biomagnification was also advantageously characterized as the trophic magnification factor (TMF), which represented the increase in Hg concentrations from one TL to the next averaged throughout the food webs. The TMF can be calculated based on the TMS[9]:

$$TMF = 10^{b_1} \quad (4)$$

$$\text{or } TMF = 10^{b_2 \times \Delta^{15}N} \quad (5)$$

where b_1 and b_2 were the TMS values calculated from equations (1) and (2), respectively.

In this study, the method based on TL was applied to characterize the degree of Hg biomagnification, as it corrected for the baseline variation in $\delta^{15}N$ that could occur among systems as a result of human inputs of nitrogen from wastewaters or agriculture[56]. We further constructed linear mixed-effect models (LMMs) to estimate the TMS and baseline Hg concentrations of China and the United States at the national scale for comparison, with trophic level as the fixed factor and sampling site as the random factor. This was because we were not interested in the response of specific aquatic systems, but rather in the overall response of a nation. We also considered random slopes and intercepts, as well as the interactions between random effects, for the impact on fish Hg. Model selection was based on the Akaike information criterion, with the lowest Akaike information criterion being the best (Supplementary Table 5). Then, Hg at trophic level 2 (THg_{TL_2} and $MeHg_{TL_2}$), which was designated as primary consumer, was estimated using the slope of the best LMMs. In data collection, we prioritized the studies in which TL and Hg concentrations were locally presented and the TLs were calculated from the $\delta^{15}N$ values using equation (3), with GetData (v2.20) being used for data extraction from the images if necessary. The data would not be considered if the $\delta^{15}N_{baseline}$ was indistinct, as the TL of an organism was intrinsically linked to the baseline species.

Statistical analysis and modelling

Statistical analyses and figure visualization were done using SPSS 19.0, Origin 2018, R version 4.2.0, SmartPLS 4.0 and ArcGIS 10.6. The intra- and interspecies differences in fish Hg were analysed using rank-transformed one-way ANOVA, owing to the non-normal distribution (Kolmogorov–Smirnov test) and non-homogeneity of the variances (Levene test). All other comparisons were conducted using the Mann–Whitney *U* test. For the estimations of TMS and baseline Hg concentrations, simple linear regressions and LMMs were fitted using the function lmer() from the package 'lmerTest (version 3.1-3)' of R (ref. 57) based on the $\log_{10}$-transformed THg or MeHg concentrations and the respective trophic levels of fish.

We further used SEM to construct the quantitative impact of a key variable on Hg bioaccumulation. Integrating fish Hg, UDF, food web ecology, watershed landscape and ambient Hg data from different sources, the final dataset included a total of 704 fish samples, from 20 Chinese and 20 American aquatic systems. We did not normalize the Hg data by fish size or species, unlike previous studies[58]. This is due to the high variability of Hg levels within a species or even the same-sized individuals, influenced by factors such as diet, age, sex, trophic position and the environment. Normalizing could mask other ecological or geographic impacts on Hg in fish. Due to the limited ambient Hg data collected, we applied the annual Hg deposition data instead, although the bioavailable Hg could be more directly relevant to MeHg accumulation. Based on our current knowledge of potential factors affecting fish Hg accumulation, we first established an a priori model summarizing the possible pathways that directly and/or indirectly impacted fish Hg levels (Extended Data Fig. 7). Furthermore, all indexes were standardized into the *Z*-scores, and the data matrix was fitted to the SEM model using the partial least square estimation method by SmartPLS version 4.0. We used *P* values (at least <0.05) and the standardized root mean squared residual (at least <0.08) as the criteria for the evaluation of SEM fit[59]. β of the SEM path network reflected the relative importance or strength of association of different factors, with an absolute β value closer to 1 representing a stronger impact. In the SEM, fish size (fish length) was the most proximate predictor and was assumed to have only a direct effect on fish Hg levels. We chose fish length over weight as the measure of body size because of the more complete dataset for fish length, although bias resulting from various body shapes between fish species might occur. Moreover, fish length and weight were not independent measures of fish size, which affect Hg levels equally in most cases[60]. Besides fish size, the other predictors were tested for both direct and indirect effects on fish Hg levels. To simplify path models and their interpretations, we assigned values to water trophic status, where 1, 2 and 3 represented oligotrophic, mesotrophic and eutrophic, respectively. We also assigned values to watershed type considering the watershed area, water depth and flow velocity, with 1 for marine; 2 for lakes, reservoirs and marshes; and 3 for rivers. Owing to the limited sample size of farmed fish, only wild species were incorporated into the SEM dataset.

Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this article.

Data availability

The authors declare that all data supporting the findings of this study are available within the article and Supplementary Information. Source data are provided with this paper.

Code availability

The LMM code is available from the corresponding author Yongguang Yin (ygyin@rcees.ac.cn) upon request.

References

1. Wang, F. et al. How closely do mercury trends in fish and other aquatic wildlife track those in the atmosphere?—Implications for evaluating the effectiveness of the Minamata Convention. *Sci. Total Environ.* **674**, 58–70 (2019).
2. Lin, C. C., Yee, N. & Barkay, T. in *Environmental Chemistry and Toxicology of Mercury* (eds Liu, G. et al.) 155–191 (Wiley, 2012).
3. Buck, D. G. et al. A global-scale assessment of fish mercury concentrations and the identification of biological hotspots. *Sci. Total Environ.* **687**, 956–966 (2019).
4. Lavoie, R. A., Kyser, T. K., Friesen, V. L. & Campbell, L. M. Tracking overwintering areas of fish-eating birds to identify mercury exposure. *Environ. Sci. Technol.* **49**, 863–872 (2015).
5. Undeland, I., Ellegård, L. & Sandberg, A. S. Fish and cardiovascular health. *Scand. J. Nutr.* **48**, 119–130 (2004).

6. Ward, D. M., Nislow, K. H., Chen, C. Y. & Folt, C. L. Rapid, efficient growth reduces mercury concentrations in stream-dwelling Atlantic salmon. *Trans. Am. Fish. Soc.* **139**, 1–10 (2010).
7. Hammerschmid, C. R. & Fitzgerald, W. F. Methylmercury in freshwater fish linked to atmospheric mercury deposition. *Environ. Sci. Technol.* **40**, 7764–7770 (2016).
8. Cossa, D. et al. Influences of bioavailability, trophic position and growth on methylmercury in hakes (*Merluccius merluccius*) from Northwestern Mediterranean and Northeastern Atlantic. *Environ. Sci. Technol.* **46**, 4885–4893 (2012).
9. Lavoie, R. A. et al. Biomagnification of mercury in aquatic food webs: a worldwide meta-analysis. *Environ. Sci. Technol.* **47**, 13385–13394 (2013).
10. Eagles-Smith, C. A. et al. Modulators of mercury risk to wildlife and humans in the context of rapid global change. *Ambio* **47**, 170–197 (2018).
11. Wang, X. & Wang, W. The three 'B' of fish mercury in China: bioaccumulation, biodynamics and biotransformation. *Environ. Pollut.* **250**, 216–232 (2019).
12. Lin, Y., Vogt, R. & Larssen, T. Environmental mercury in China: a review. *Environ. Toxicol. Chem.* **31**, 2431–2444 (2012).
13. Cheng, H. & Hu, Y. Understanding the paradox of mercury pollution in China: high concentrations in environmental matrix yet low levels in fish on the market. *Environ. Sci. Technol.* **46**, 4695–4696 (2012).
14. Zhou et al. Causes of low mercury levels in fish from the Three Gorges Reservoir, China. *J. Hazard. Mater.* **464**, 132930 (2024).
15. UNEP. *Minamata Convention on Mercury: Text and Annexes*. www.mercuryconvention.org (2013).
16. Zhang, H. et al. Decreasing mercury levels in consumer fish over the three decades of increasing mercury emissions in China. *Eco. Environ. Health* **1**, 46–52 (2022).
17. Fu, X. W., Feng, X. B., Sommar, J. & Wang, S. F. A review of studies on atmospheric mercury in China. *Sci. Total Environ.* **421**, 73–81 (2012).
18. Yu, C. et al. Mercury and methylmercury in China's lake sediments and first estimation of mercury burial fluxes. *Sci. Total Environ.* **770**, 145338 (2021).
19. Fleck, J. A. et al. Mercury and methylmercury in aquatic sediment across western North America. *Sci. Total Environ.* **568**, 727–738 (2016).
20. Liu, M. D. et al. Rivers as the largest source of mercury to coastal oceans worldwide. *Nat. Geosci.* **14**, 672–677 (2021).
21. Liu, M. D. et al. Observation-based mercury export from rivers to coastal oceans in East Asia. *Environ. Sci. Technol.* **55**, 14269–14280 (2021).
22. Perrot, V., Landing, W. M., Grubbs, R. D. & Salters, V. J. M. Mercury bioaccumulation in tilefish from the northeastern Gulf of Mexico 2 years after the Deepwater Horizon oil spill: insights from Hg, C, N and S stable isotopes. *Sci. Total Environ.* **666**, 828–838 (2019).
23. Lescord, G. L., Johnston, T. A., Branfireun, B. A. & Gunn, J. M. Percentage of methylmercury in the muscle tissue of freshwater fish varies with body size and age and among species. *Environ. Toxicol. Chem.* **37**, 2682–2691 (2018).
24. Yoshino, K. et al. Food sources are more important than biomagnification on mercury bioaccumulation in marine fishes. *Environ. Pollut.* **262**, 113982 (2020).
25. Chumchal, M. M. et al. Habitat-specific differences in mercury concentration in a top predator from a shallow lake. *Trans. Am. Fish. Soc.* **137**, 195–208 (2008).
26. Zhang, M. et al. Trophic level changes of fishery catches in Lake Chaohu, Anhui Province, China: trends and causes. *Fish. Res.* **131–133**, 15–20 (2012).
27. Du, J. et al. Impacts of fishing on the marine mean trophic level in Chinese marine area. *Acta Ecol. Sin.* **35**, 83–88 (2015).
28. Braune, B. M. Mercury accumulation in relation to size and age of Atlantic herring (*Clupea harengus harengus*) from the southwestern Bay of Fundy, Canada. *Arch. Environ. Contam. Toxicol.* **16**, 311–320 (1987).
29. Dang, F. & Wang, W. X. Why mercury concentration increases with fish size? Biokinetic explanation. *Environ. Pollut.* **163**, 192–198 (2012).
30. McManamay, R. A. et al. Scaling mercury biodynamics from individuals to populations: implications of an herbivorous fish on mercury cycles in streams. *Freshwater Biol.* **64**, 815–831 (2019).
31. Wang, R. & Wang, W. X. Contrasting mercury accumulation patterns in tilapia (*Oreochromis niloticus*) and implications on somatic growth dilution. *Aquat. Toxicol.* **114**, 23–30 (2012).
32. UNIDO *Industrial Statistics Database (INDSTAT): 2-Digit Level of ISIC Code (Revision 3). United Nations Industrial Development Organization*. https://stat.unido.org/data/table (2021).
33. UN Environment. *Global Mercury Assessment 2018. UN Environment Programme, Chemicals and Health Branch Geneva, Switzerland* (2019).
34. Li, S. An approach to accelerating innovative development of the lake science in China. *Bull. Chin. Acad. Sci.* **21**, 399–405 (2006).
35. Bachmann, R. W., Hoyer, M. V. & Canfield, D. E. The extent that natural lakes in the United States of America have been changed by cultural eutrophication. *Limnol. Oceanogr.* **58**, 945–950 (2013).
36. Pickhardt, P. C. et al. Algal blooms reduce the uptake of toxic methylmercury in freshwater food webs. *Proc. Natl Acad. Sci. USA* **99**, 4419–4423 (2002).
37. Lei, P. et al. Mechanisms of algal biomass input enhanced microbial Hg methylation in lake sediments. *Environ. Int.* **126**, 279–288 (2019).
38. Soerensen, A. L. et al. Eutrophication increases phytoplankton methylmercury concentrations in a coastal sea—a Baltic Sea case study. *Environ. Sci. Technol.* **50**, 11787–11796 (2016).
39. Chen, C. Y. et al. A critical time for mercury science to inform global policy. *Environ. Sci. Technol.* **52**, 9556–9561 (2018).
40. Xu, W. H. et al. Hidden loss of wetlands in China. *Curr. Biol.* **29**, 3065–3071 (2019).
41. Dahl, T. E. *Status and Trends of Wetlands in the Coastal Watersheds of the Conterminous United States, 2004 to 2009* (US Fish & Wildlife Service, Fisheries and Habitat Conservation, 2009).
42. Paranjape, A. R. & Hall, B. D. Recent advances in the study of mercury methylation in aquatic systems. *FACETS* **2**, 85–119 (2017).
43. Sharpley, A. Managing agricultural phosphorus for water quality: lessons from the USA and China. *J. Environ. Sci.* **26**, 1770–1782 (2014).
44. Walters, D. M. et al. Food web controls on mercury fluxes and fate in the Colorado River, Grand Canyon. *Sci. Adv.* **6**, eaaz4880 (2020).
45. Liu, G. et al. Legacy and fate of mercury and methylmercury in the Florida Everglades. *Environ. Sci. Technol.* **45**, 496–501 (2011).
46. Rumbold, D. in *Mercury and the Everglades. A Synthesis and Model for Complex Ecosystem Restoration* Vol. 2 (eds Pollman, C. et al.) 49–86 (Springer Nature, 2019).
47. Govaerts, A. et al. Distribution and bioaccumulation of POPs and mercury in the Ga-Selati River (South Africa) and the rivers Gudbrandsdalslagen and Rena (Norway). *Environ. Int.* **121**, 1319–1330 (2018).
48. Giang, A. & Selin, N. E. Benefits of mercury controls for the United States. *Proc. Natl Acad. Sci. USA* **113**, 286–291 (2016).
49. Wang, Y. et al. A review of studies on the biogeochemical behaviors of mercury in the Three Gorges Reservoir, China. *Bull. Environ. Contam. Toxicol.* **102**, 686–694 (2019).
50. Feng, X. B. et al. Mercury pollution in China: implications on the implementation of the Minamata Convention. *Environ. Sci. Process Impacts* **24**, 634–648 (2022).

51. Fry, B. & Chumchal, M. M. Mercury bioaccumulation in estuarine food webs. *Ecol. Appl.* **22**, 606–623 (2012).
52. Shah, V. et al. Improved mechanistic model of the atmospheric redox chemistry of mercury. *Environ. Sci. Technol.* **55**, 14445–14456 (2021).
53. Li, A. et al. Polybrominated diphenyl ethers in the sediments of the Great Lakes. 4. Influencing factors, trends, and implications. *Environ. Sci. Technol.* **40**, 7528–7534 (2006).
54. Clayden, M. G. et al. Mercury biomagnification through food webs is affected by physical and chemical characteristics of lakes. *Environ. Sci. Technol.* **47**, 12047–12053 (2013).
55. Jardine, T. D., Kidd, K. A. & Fisk, A. T. Applications, considerations and sources of uncertainty when using stable isotope analysis in ecotoxicology. *Environ. Sci. Technol.* **40**, 7501–7511 (2006).
56. Borga, K. et al. Trophic magnification factors: considerations of ecology, ecosystems and study design. *Integr. Environ. Asses. Manag.* **8**, 64–84 (2012).
57. Bates, D., Machler, M., Bolker, B. M. & Walker, S. C. Fitting linear mixed-effects models using lme4. *J. Stat. Softw.* **67**, 1–48 (2015).
58. Eagles-Smith, C. A. et al. Mercury in western North America: a synthesis of environmental contamination, fluxes, bioaccumulation and risk to fish and wildlife. *Sci. Total Environ.* **568**, 1213–1226 (2016).
59. Pollman, C. D. Mercury cycling in aquatic ecosystems and trophic state-related variables—implications from structural equation modeling. *Sci. Total Environ.* **499**, 62–73 (2014).
60. Thomas, S. M. et al. Climate and landscape conditions indirectly affect fish mercury levels by altering lake water chemistry and fish size. *Environ. Res.* **188**, 109750 (2020).

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Author contributions

Conceptualization: G.L., Y.Y. and Y.C. Investigation: Y.X. and Y.Y. Funding acquisition: Y.X., Y.Y., Y.C. and G.J. Writing—original draft: Y.X. Writing—review and editing: G.L., Y.Y., Y.L., D.W., Y.C. and G.J.

Competing interests

The authors declare no competing interests.

Additional information

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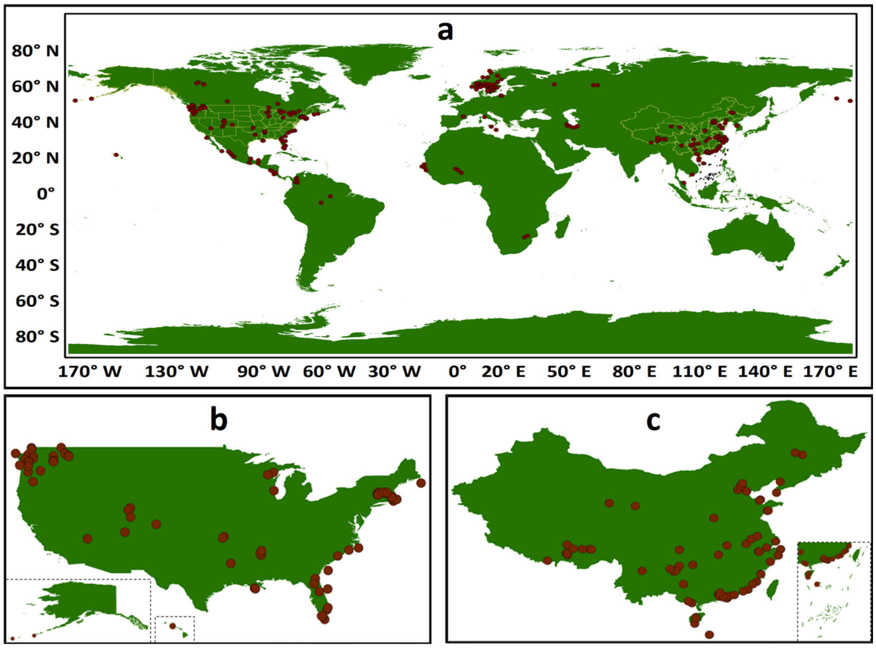

Extended Data Fig. 1 | Map of the site locations of peer-reviewed studies. **a**, Worldwide, **b**, the US and **c**, China. Administrative boundary data of worldwide and China are supported by the Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences (http://www.resdc.cn). Administrative boundary data of the US are supported by Natural Earth (https://www.naturalearthdata.com). The list of papers reviewed and the detailed sample points are present in the Source Data.

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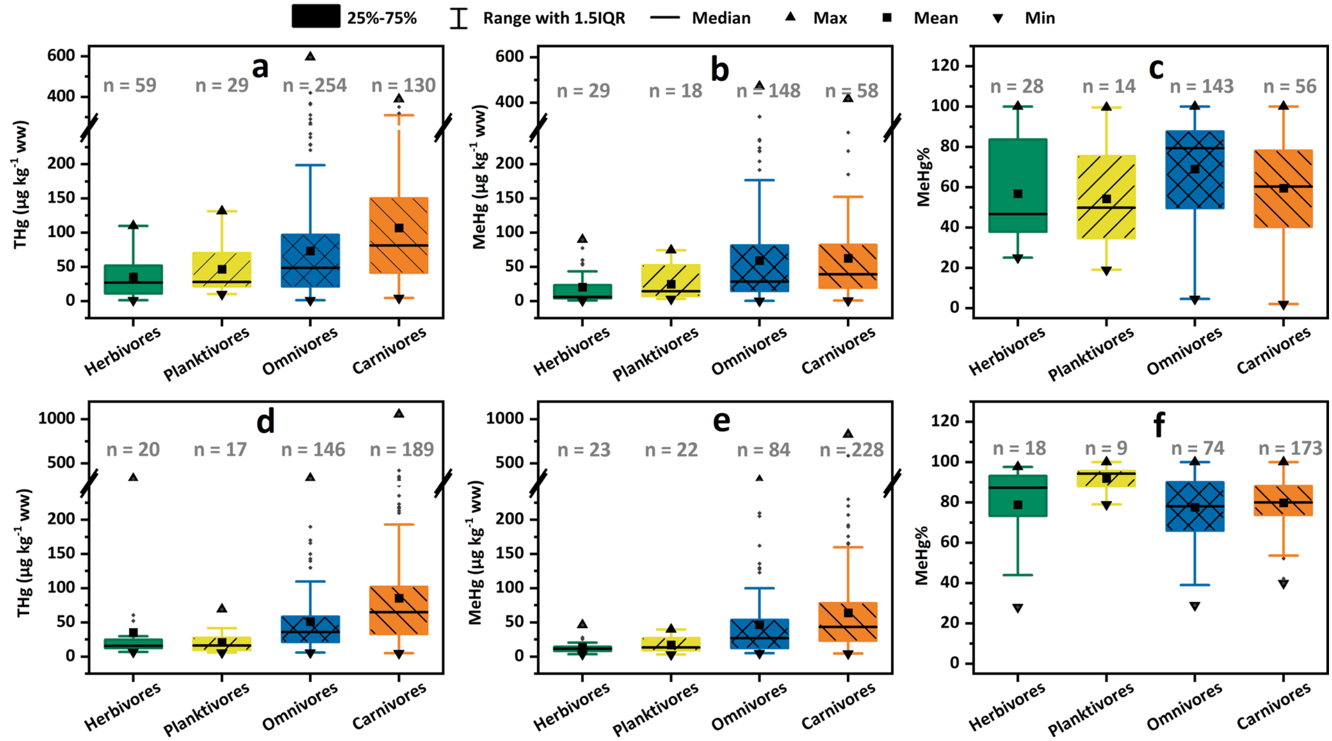
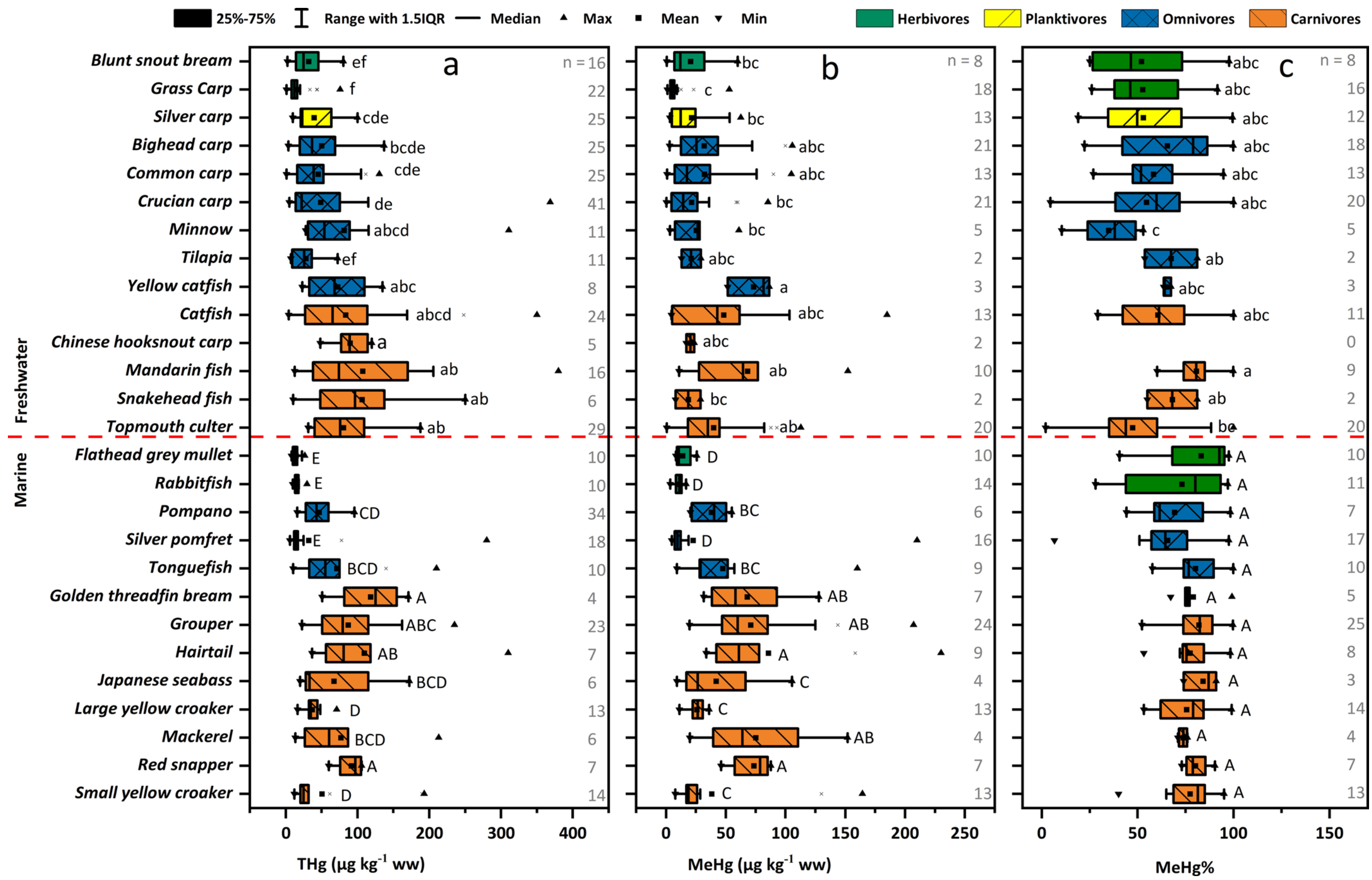


Extended Data Fig. 2 | THg and MeHg concentrations (μg kg^{-1} ww), as well as MeHg% in fish of China with different dietary behaviors. a–c, Freshwater fish, **d–f**, marine fish. The detailed sample sizes are provided in the Source Data, and "n" represents the number of data points per graph. Green: herbivores, yellow: planktivores, blue: ominivores, and orange: carnivores.



Extended Data Fig. 3 | Hg concentrations of different fish species in China. **a**, THg (μg kg^{-1} ww), **b**, MeHg (μg kg^{-1} ww), **c**, MeHg%. The difference analysis of THg, MeHg and MeHg% among fish species was conducted by Rank transformed one-way ANOVA, owing to the non-normal distribution (Kolmogorov-Smirnov test, K-S test) and non-homogeneity of the variances (Levene test). Categories that share common letter do not differ significantly (two-sided tests): lower case letters are for comparison of Hg among freshwater species and capital letters are for comparison of Hg among marine fish species. The detailed sample sizes are provided in the Source Data, and "n" represents the number of data points per graph.

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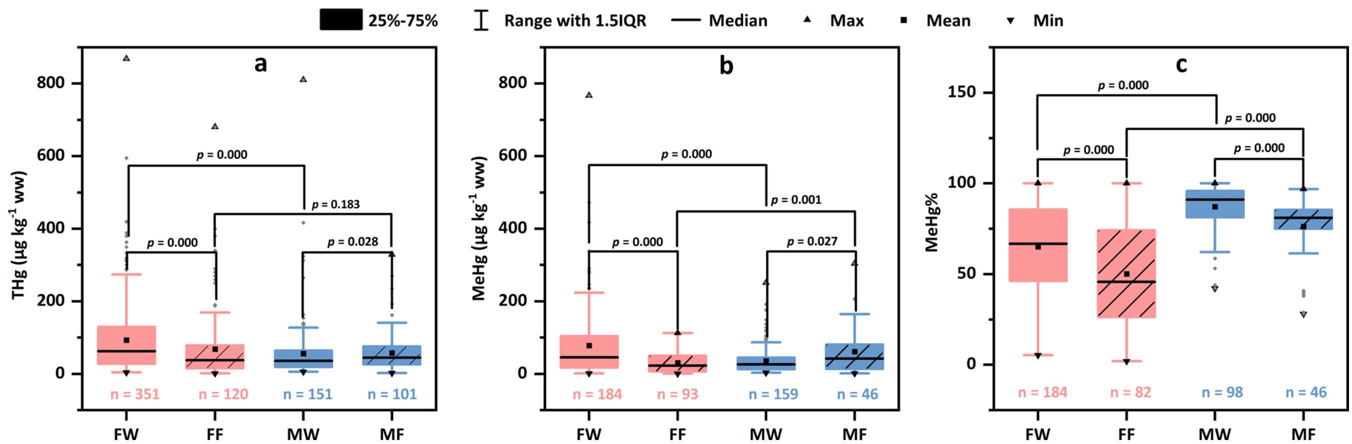

Extended Data Fig. 4 | Hg concentrations in wild and farmed fish across China. **a**, THg (μg kg^{-1} ww), **b**, MeHg (μg kg^{-1} ww), **c**, MeHg%. Data are the comprehensive presentation of Fig. 2. FW and FF are short for freshwater wild and farmed fish, and MW and MF are short for marine wild and farmed fish, respectively. The detailed sample sizes are provided in the Source Data, and "n" represents the number of data points per graph. A Mann-Whitney U test was performed to compared fish Hg levels between groups. Based on Monte Carlo analysis (two-sided), a *p*-value less than 0.05 indicates statistically significance.

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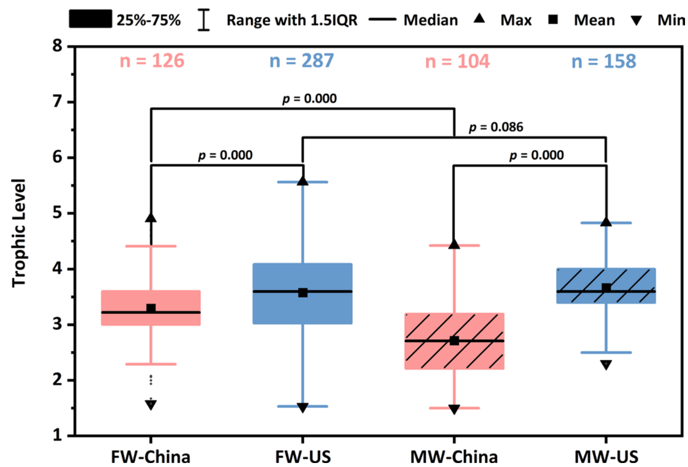

Extended Data Fig. 5 | Trophic levels of fish observed in China and the US. FW and MW are short for freshwater and marine wild fish, respectively. The detailed sample sizes are provided in the Source Data, and "n" represents the number of data points per graph. A Mann-Whitney U test was performed to compared fish TL between groups. Based on Monte Carlo analysis (two-sided), a *p*-value less than 0.05 indicates statistically significance.

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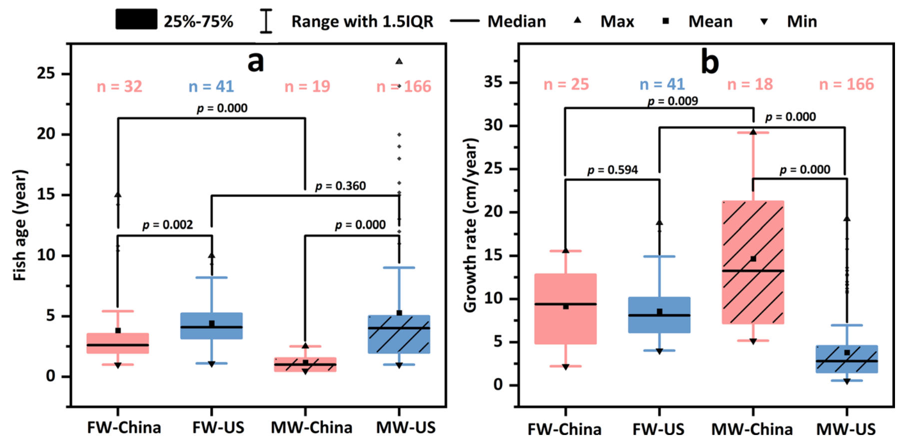

Extended Data Fig. 6 | The growth traits of wild fish across China and the US collected. a, Fish age, **b**, growth rate (cm/year). FW and MW are short for freshwater and marine wild fish, respectively. The detailed sample sizes are provided in the Source Data, and "n" represents the number of data points per graph. A Mann-Whitney U test was performed to compared fish age and growth rate between groups. Based on Monte Carlo analysis (two-sided), a *p*-value less than 0.05 indicates statistically significance.

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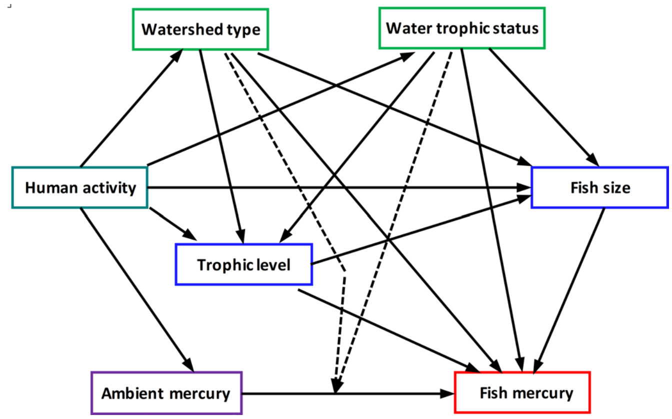

Extended Data Fig. 7 | The hypothesized fish mercury accumulation model summarizing pathways that directly and/or indirectly mediated mercury levels in fish. Solid arrows pointing straight to fish mercury without any intermediate factors are the predicted direct pathways, while others are the predicted indirect pathways. Dotted arrows represent the modulating effect.

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☒	☐	For Bayesian analysis, information on the choice of priors and Markov chain Monte Carlo settings
☒	☐	For hierarchical and complex designs, identification of the appropriate level for tests and full reporting of outcomes
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Our web collection on statistics for biologists contains articles on many of the points above.

Software and code

Policy information about availability of computer code

Data collection	GetData Graph Digitizer (v2.20) was used for data collection if the data were displayed in graphs.
Data analysis	SPSS 19.0, Origin 2018, R 4.2.0, lmerTest Package (Ver. 3.1-3), SmartPLS 4.0, and ArcGIS 10.6 were used for data analysis.

For manuscripts utilizing custom algorithms or software that are central to the research but not yet described in published literature, software must be made available to editors and reviewers. We strongly encourage code deposition in a community repository (e.g. GitHub). See the Nature Portfolio guidelines for submitting code & software for further information.

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The authors declare that all the data supporting the findings of this study are available within the paper and its supplementary information file.

Human research participants

Policy information about studies involving human research participants and Sex and Gender in Research.

Reporting on sex and gender	This information has not been collected.
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Recruitment	This information has not been collected.
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Ecological, evolutionary & environmental sciences study design

All studies must disclose on these points even when the disclosure is negative.

Study description	In China, the present state of Hg contamination in fish has been well documented for the past three decades, but the underlying factors that directly and indirectly affect fish Hg accumulation remains poorly understood and quantified. This knowledge gap is particularly evident when it comes to identifying the specific contributors to the anomalously low levels of Hg found in fish in China. In this study, therefore, we incorporate the geographic patterns of fish Hg on a global scale and quantify the possible factors controlling fish Hg accumulation, with the aim to provide implications for policy makers to reduce Hg exposure risks in humans and wildlife.
Research sample	This study extracted existing data from the publications (ending April 2019) indexed in the Web of Sciences (WOS, http://apps.webofknowledge.com) and China Knowledge Resource Integrated Database (CNKI, https://www.cnki.net/), including Hg concentrations (THg, MeHg and MeHg%) in fish and other available information linked to Hg accumulation from the retrieved publications, including the food web ecology (e.g., species, age, length, weight, nitrogen isotope, trophic level, and dietary habits of individuals), sampling size, the watershed landscape (e.g. watershed type and water nutritional status), as well as ambient Hg levels. In addition, we estimate atmospheric Hg concentration (i.e., surface Hg(0), Hg(II) and TGM) and annual Hg deposition (i.e., Hg(0) dry deposition, Hg(II) wet and dry deposition) using the GEOS-Chem global atmospheric chemistry model. We develop urban distance factor (UDF) to represent the influence of human activities on fish Hg accumulation, since the emission, transportation, and transformation of Hg, watershed characteristics and food webs are largely anthropogenically controlled.
Sampling strategy	In this study, we did not precalculate the sampling-size needed. Instead we extracted all available data (which were collected from 1993 to 2017) reported in Web of Science, including fish Hg concentrations and other ancillary parameters from 315 aquatic systems worldwide (86 in China and 106 in the US), sufficient for statistical analysis.
Data collection	We constructed an observation-based database of fish Hg from 315 aquatic systems worldwide, including 86 in China and 106 in the US. This database was based on the extraction of research observations from the peer-reviewed publications, including fish THg, MeHg and MeHg% in fish and ancillary parameters linked to Hg accumulation (Source Data). We first extracted Hg data and fish species from research articles (ending April 2019) indexed in the Web of Sciences (WOS, http://apps.webofknowledge.com) and China Knowledge Resource Integrated Database (CNKI, https://www.cnki.net/), using the keywords: Mercury, Fish, Biomagnification, Nitrogen isotope, Mercury AND China, Mercury AND United States. Published articles which met the following criteria were included in the final datasheet: 1) the study must be original research on the Hg bioaccumulation in fish; 2) clear analytical methods for THg or MeHg in fish were provided; 3) original data could be extracted from tables and figures. We further restricted our analysis to data on wild species when investigating the factors that influence Hg accumulation in fish, owing to the limited farmed fish from regions other than China. From each of these studies, we compiled the information on THg, MeHg, and MeHg% values for sampled fish when available. Commonly, raw data were collected as much as possible, and if not, average concentrations were used. If the data were displayed in graphs, the GetData Graph Digitizer (v2.20) was applied to extract the related data.
Timing and spatial scale	The data, collected from 1993 to 2017 and covering China, the US and other regions worldwide, were extracted from publications indexed by Web of Sciences (WOS, http://apps.webofknowledge.com) and China Knowledge Resource Integrated Database (CNKI, https://www.cnki.net/).
Data exclusions	The data would not be considered if the δ15N baseline was indistinct when characterizing the degree of Hg biomagnification, since the trophic level of an organism is intrinsically linked to the baseline species.

Reproducibility | The findings of study are based on existing data and not on experimental measurement. Therefore, this is not relevant to this study.

Randomization | The findings of study are based on existing data and not on experimental measurement. Therefore, this is not relevant to this study.

Blinding | The findings of study are based on existing data and not on experimental measurement. Therefore, this is not relevant to this study.

Did the study involve field work? ☐ Yes ☒ No

Reporting for specific materials, systems and methods

We require information from authors about some types of materials, experimental systems and methods used in many studies. Here, indicate whether each material, system or method listed is relevant to your study. If you are not sure if a list item applies to your research, read the appropriate section before selecting a response.

Materials & experimental systems

n/a	Involved in the study
☒	☐ Antibodies
☒	☐ Eukaryotic cell lines
☒	☐ Palaeontology and archaeology
☒	☐ Animals and other organisms
☒	☐ Clinical data
☒	☐ Dual use research of concern

Methods

n/a	Involved in the study
☒	☐ ChIP-seq
☒	☐ Flow cytometry
☒	☐ MRI-based neuroimaging