Critical Review

Mercury Exposure and Toxicological Consequences in Fish and Fish-Eating Wildlife from Anthropogenic Activity in Latin America

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ABSTRACT

Despite the risk of significant adverse toxicological effects of Hg to humans and wildlife, Hg use in anthropogenic activities, and artisanal small-scale gold mining (ASGM) in particular, is widespread throughout Latin America. However, there are few research and monitoring studies of Hg toxicity in fish and fish-eating wildlife in Latin America compared to North America. In the present paper, we reviewed the literature from published articles and reports and summarized and assessed data on Hg in fish from 10,391 individuals and 192 species sampled across Latin America. We compared fish Hg levels with toxicity reference values (TRVs) for fish and dietary TRVs for fish-eating wildlife. We determined that fish, piscivorous birds, and other wildlife are at risk of Hg toxicity. We observed a large disparity in data quantity between North and Latin America, and identified regions requiring further investigation. In particular, future biomonitoring and research should focus on exposure of wildlife to Hg in Peru, Chile, Uruguay, the eastern and northern regions of Brazil, Venezuela, Ecuador, and Colombia. We also discuss Hg risk assessment methodological issues and recommend that future evaluations of Hg risk to wildlife must collect key physiological variables, including age, body size, and ideally Hg:to-Se molar ratios. Integr Environ Assess Manag 2021;17:13–26. © 2020 Environment and Climate Change Canada. Integrated Environmental Assessment and Management published by Wiley Periodicals LLC on behalf of Society of Environmental Toxicology & Chemistry (SETAC)

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INTRODUCTION

Mercury is a nonessential toxic metal emitted to the environment from a variety of natural sources, including particulate and gaseous release from volcanic emissions, erosion of soil and rock, and fluxes from marine systems (Pirrone and Mason 2009). Humans have released Hg to the environment since at least 1400 BCE by using Hg to extract Au and other precious metals (Cook et al. 2009; Telmer and Veiga 2009; Outridge et al. 2018). Industrial and agricultural use and release of Hg increased throughout the industrial revolution and continued into the mid-20th century (Pirrone et al. 2010; Streets et al. 2019). In the late 1950s, following the crises caused by mass human Hg poisonings in Minimata Bay, Japan, and in Iraq, many countries began to identify and regulate point sources of Hg release (Bakir et al. 1973; Sherbin 1979; Tsuda et al. 2009; Normile 2013). However, net anthropogenic Hg emissions have continued to increase, primarily from fossil fuel burning and particularly from the sharp rise in use of coal for electricity in expanding Asian economies (Streets et al. 2009, 2019; AMAP/UNEP 2019). Global atmospheric Hg is composed of 3 times more Hg of anthropogenic than natural origin (AMAP/UNEP 2019).

The use of Hg in artisanal small-scale gold mining (ASGM) continues to be a major anthropogenic source in tropical regions of the globe (Telmer and Veiga 2009; AMAP/UNEP 2019). Many of the sites affected by ASGM are biodiversity hotspots and are located within protected areas (Gandiwa and Zisadza-Gandiwa 2012; Asner and Tupayachi 2017). When Au is extracted through amalgamation, most Hg is lost to the atmosphere as vapor, and the remainder drains directly into streams and rivers. Mercury vapor is toxic to Au miners and local residents (da Silva et al. 2012; Siregar and Prayogo 2017). Inorganic Hg (IHg) that enters the aquatic environment, via wet or dry deposition, can then be methylated by microbial activity in sediments and converted to methylmercury (MeHg), increasing its bioavailability, toxicity, and biomagnification potential (Wiener et al. 2003). Artisanal small-scale Au mining also involves deforestation and soil removal, which can result in habitat loss and movement of Hg through aquatic and terrestrial food
Poorly regulated ASGM in Latin America threatens some of the most biodiverse regions of the world, including the Andean mountains (Bax et al. 2019), the Amazon basin (Álvarez-Berrios and Mitchell Aide 2015; Asner and Tuppyachi 2017; Espejo et al. 2018), and the Chocó biogeographic region (Palacios-Torres et al. 2018). For instance, nearly 100,000 ha of rainforest were deforested by ASGM between 1985 and 2007 in the department of Madre de Dios in Peru (Espejo et al. 2018), and Hg pollution has been on the rise over the last 100 y (Kumar et al. 2018; Moreno-Brush et al. 2018), posing a risk to ecological and human health of local and downstream communities (Ashe 2012; Diringer et al. 2015).

During the 19th century, Hg amalgamation was employed throughout western North America. The majority of those mines are now abandoned, and the remainder use other extraction processes, adhering to environmental regulations set out in the Minimata Convention on Mercury (UNEP 2013). Thus, North American Au mines are sites of “historical” Hg emissions. Previous Hg research has focused on the impacts of “historical” Au mining to wildlife in North America (Ackerman et al. 2016; Eagles-Smith, Ackerman et al. 2016; Eagles-Smith, Wiener et al. 2016). In contrast to historical gold mining, ASGM operations are unorganized, unregulated, and a current source of Hg emission (Telmer and Veiga 2009; AMAP/UNEP 2019). Despite environmental regulations, ASGM operations in the southern hemisphere increased rapidly in the 1970s, fueled by a suite of socioeconomic conditions, including poverty, unemployment, and lack of affordable alternative technologies (Veiga et al. 2005; Telmer and Veiga 2009; WWF 2018; AMAP/UNEP 2019). Of the estimated 2220 t of Hg emitted globally in 2015 from human activities, South America produced 18%, of which ASGM contributed 70% (340 t). In contrast, North America produced only 2% of global Hg emissions, of which ASGM contributed none (AMAP/UNEP 2019). In 2015, South America alone accounted for 53% of the estimated 1220 t of Hg released globally from ASGM to aquatic and terrestrial environments (AMAP/UNEP 2019).

In the United States and Canada, the effects of Hg bioaccumulation, biomagnification, and toxicity to wildlife have been studied extensively. Fish are useful and effective indicators of relative MeHg exposure within food webs and toxicological risk to fish themselves, humans, and other fish-eating wildlife (Depew et al. 2012; Elliott et al. 2015; Eagles-Smith, Wiener et al. 2016). In North America, many studies examined fish Hg concentrations (Weech et al. 2004; Kamman et al. 2005; Monson et al. 2011; Sandheinrich et al. 2011; Depew et al. 2012, 2013; Wiener et al. 2012; Chetelat et al. 2015; Scheuhammer et al. 2016) and assessed the risk of Hg to avian piscivores (Weech et al. 2003, 2006; Scheuhammer et al. 2008; Seewagen 2010; Guigueno et al. 2012; Ackerman et al. 2016; Eagles-Smith, Wiener et al. 2016; Jackson et al. 2016). Bird Hg data has been summarized for North America by well-funded activities such as the Western Mercury Initiative (Ackerman et al. 2016; Jackson et al. 2016), the Arctic Contaminants Program (Mallory and Braune 2012), and other programs for both piscivores (Evers et al. 2011) and passerines (Jackson et al. 2015) in the Great Lakes and western and northern regions of the United States and Canada. Despite recent increases in ASGM and subsequent Hg emissions in South America, Hg biomonitoring programs for birds and marine mammals are sparse in that region (AMAP/UNEP 2019). Most Hg research in South America has focused on human health risks from mining (Malm 1998; Eisler 2004; Cordy et al. 2011; Herman and Grace 2014; Veiga et al. 2014; Hsu-Kim et al. 2018; WWF 2018), whereas studies of Hg risk to wildlife are scarce (Lacher and Goldstein 1997; Eisler 2004; Moreno-Brush et al. 2018).

In the present synthesis, we summarized the available data on Hg contamination of fish in Latin America. We reviewed the literature from published articles and reports to extract mean Hg concentrations in fish for each species and site. We compiled Hg concentrations from 10,391 fish, in 96 unique sites across 12 countries. The primary goal of the present assessment is to describe patterns in fish Hg concentrations in Latin America to 1) determine whether fish and birds are at risk of Hg toxicity using established toxicity reference values (TRVs) and 2) identify regions with insufficient data on Hg contamination of South and Central American wildlife. We synthesized the results of fish Hg studies to assess the risk for piscivorous wildlife species, primarily birds, in Latin America. The present paper highlights the data quantity disparity between North America and Latin America, and identifies geographic regions that require baseline data collection and further research. We also discuss Hg risk assessment methodological issues and provide recommendations for future studies.

METHODS

Data acquisition: Literature review

We conducted a literature review of peer-reviewed journal articles and published reports documenting Hg concentrations in fish in Latin America. Although many industrial activities release Hg to the atmosphere, the majority of point source Hg pollution in Latin America is from ASGM activities, the locations of which, as illegal operations, are not well documented. Therefore, the present review encompasses some studies that do not focus primarily on ASGM as a source of Hg because ASGM Hg emissions are likely contributing to background levels. Only studies conducted up until 2019 were selected. We conducted all Web searches in Google Scholar using all combinations of the following key search terms: “fish,” “wildlife” and “mercury,” “toxicity,” “gold mining” and “South America,” and “Latin America.” Citations were traced through further examination of the literature cited in the papers generated from the initial search. Two of the authors conducted independent and separate Web searches as a quality control measure. When possible, mean Hg concentrations for multiple species and locations within the
same study were kept separate. When a single publication reported multiple mean Hg concentrations, we included means for each unique species and location measured in the study. Mean Hg concentrations were visually approximated from figures when values were not reported in a table or text. The median was used as a proxy when the mean was not reported. When they were available, we compiled measures of mean weight (g), length (cm), age, and Se in fish sampled for Hg concentration. Species trophic position was inferred from the source article whenever possible. For articles that did not include a description of trophic position, an Internet search was conducted and scientific sources were preferentially selected; however, due to the high diversity of tropical species, high-quality sources were not always available. Thus caution should be exercised when considering trophic position of some rare or unstudied species. When global positioning system (GPS) coordinates were not reported, study site descriptions (e.g., village and lake names) and any maps or figures provided were used to approximate latitudes and longitudes using Google Earth. We included only original research studies, excluding those without sufficient description of methodology and analysis. Methods and analyses were carefully reviewed to ensure consistent and comparable quality assurance–quality control (QAQC) practices, and any data that did not meet these standards were excluded. Some local government data sets were located but were not included in the present study because the QAQC methods used could not be verified. One study (Alho and Vieria 1997) was not included because it did not present original data and methods.

Data transformations and assumptions

We examined data from each source for completeness and standardized all Hg concentrations (total Hg [THg] µg g⁻¹), fish lengths (cm), and fish weights (g). Given that most Hg in upper trophic level biota is MeHg, we used THg as an effective measure of MeHg in biota (Scheuhammer et al. 2007) and all further use of “Hg” refers to this measure. All Hg concentrations were reported as either muscle tissue, flesh tissue, or fillet, which we assumed synonymous to boneless, skinless fillet. Most studies report THg and MeHg concentrations on a fresh or wet weight basis, whereas others report dry weight concentrations. Because 96% of the data were reported on a wet weight basis, we converted all dry weight values to wet. Of the studies that reported dry weights, no moisture content data were available. We converted dry weight Hg concentrations to wet weight Hg concentrations using the mean tissue-specific moisture content (76%) determined by Eagles-Smith, Ackerman et al. (2016) derived from 6594 paired measures of fish moisture content and dry weight Hg. This estimate aligns well with South American measurements for fish tissue water contents of 76% to 63% in red fish (da Silva et al. 2008) and 72% to 80% in a variety of Central American fish (Elliott et al. 2015). We converted dry weights to wet weight using this formula:

\[ \text{Wet wt} = \text{Dry wt} \times [1 - (76\% / 100)]. \]

Toxicity assessment

Apart from the relatively few focused field studies, assessment of the toxicity and risk of Hg in wildlife typically involves comparison of MeHg concentrations in a population of interest with one or more TRVs. The TRVs derived from peer-reviewed scientific literature may indicate an exposure level shown to be without deleterious effects, a low level of adverse effects (LOAE), or other thresholds, such as a 20% effect concentration (EC20), in the species of interest (Fuchsman et al. 2017). The LOAE and EC20 levels are common TRVs in wildlife toxicological studies. However, the majority of research on Hg effects in wildlife comes from diet studies and dosing experiments in North America. Effects on fish reproduction and survival, calculated from a multispecies dose–response curve, were observed at a low magnitude of effect (5.5%) at 0.20 µg Hg g⁻¹ (Beckvar et al. 2005; Dillon et al. 2010) and at an EC20 of 1.04 µg Hg g⁻¹ (Dillon et al. 2010; Fuchsman et al. 2016). The LOAE thresholds for Hg-related reproductive impairment in birds range from 0.2 to >1.4 µg Hg g⁻¹ in diet (Fuchsman et al. 2017). We converted TRVs to muscle concentration equivalent, by dividing the whole body concentration by 0.74 as described in Eagles-Smith, Ackerman et al. (2016), where TRVs were reported as whole body concentration to compare with the Latin American data. Many raptors, such as ospreys and eagles, preferentially consume muscle tissue, whereas other fish-eating species, including herons, cormorants, and loons, consume fish whole. Mercury whole body burden is often higher because it contains organ tissue, such as liver, which accumulates toxins. Because not all birds consume the whole fish, we chose muscle tissue Hg threshold as the more conservative measure of toxicity in piscivorous birds.

To determine whether fish in Latin America are at risk of Hg toxicity, we plotted fish Hg concentrations from each unique combination of species, site, and year, along with 2 critical Hg thresholds in fish muscle (0.2 µg Hg g⁻¹ low magnitude effect TRV and 1.04 µg Hg g⁻¹ EC20 TRV; Dillon et al. 2010). Latin American fish Hg concentrations were also compared against extensive reviews of fish Hg concentrations in North America (Eagles-Smith, Ackerman et al. 2016).

To determine whether birds are at risk of Hg toxicity from fish in Latin America, we took a subset of all freshwater fish of prey size and weight for piscivorous birds (<40 cm or <500 g) and plotted fish Hg along with critical diet thresholds associated with reproductive effects in common loons (0.2 µg Hg g⁻¹ wet wt; Depew et al. 2012) and the lower and upper range (0.45–1.4 µg Hg g⁻¹ wet wt) for large birds (794–5500 g; Fuchsman et al. 2017). Fish size metrics were derived from average length and weight of osprey forage fish (Elliott et al. 2015). We also plotted median Hg concentrations for a subset of 15 fish species in Latin America for which we had a large enough sample size (>10 unique site–year means) and compared them against toxicity thresholds for fish and piscivorous birds. Median was chosen...
instead of arithmetic mean because the data do not fit a normal distribution (Rizzo et al. 2011).

We grouped Latin American fish Hg concentrations into low (≤0.2 µg Hg g⁻¹; below critical diet threshold for common loons), medium (0.2–0.4 µg Hg g⁻¹; above critical diet threshold for common loon and below the lower range for large birds), and high (≥0.4 µg Hg g⁻¹; above the critical diet threshold for large birds) and mapped the locations of fish Hg for each bin. Maps were created using the "ggmap" package and all analyses were performed in the statistical software program R (version 3.6.0) (RStudio Team 2016). Our analysis allowed us to determine whether fish in Latin America exceed TRV thresholds for fish and avian piscivores and to identify geographic hot spots where levels exceed the thresholds and regions with insufficient data where further study is required.

RESULTS

We extracted fish Hg concentrations (µg/g wet wt) from 45 studies published from 1988 to 2019 that cumulatively measured 192 unique fish species and 10,391 individual fish in 12 countries across Latin America. Data collection occurred during 1986 to 2017 (Table 1). Countries with the fewest species studied include Ecuador, Paraguay, Nicaragua, Costa Rica, and Bolivia (Table 1). The majority of research took place in Brazil, and in particular the Pantanal region, with Hg measurements in 97 fish species from 20 separate studies and data ranging from 1986 to 2017 (Table 1). Spatial fish Hg across Latin America is shown in Figure 1 with Hg concentrations binned into low, medium, and high on the basis of Hg thresholds in prey fish associated with risk of reproductive impairment in piscivorous birds (low = 0–0.19 µg Hg g⁻¹ [Depew et al. 2012]; medium = 0.2–0.39 µg Hg g⁻¹ [Depew et al. 2012]; and high ≥0.4 µg Hg g⁻¹ [Fuchsman et al. 2017]). Medium risk of Hg toxicity to piscivorous birds from consumption of prey fish (≥0.2 µg Hg g⁻¹) is analogous to risk of Hg toxicity to fish themselves (Depew et al. 2012). Other species assessed for Hg toxicity in Latin America, which are beyond the scope of the present review, include terrestrial birds (further described in Supplemental Data file) (Hylander et al. 1994; Klekowski et al. 1999; Lama et al. 2011; Townsend et al. 2013; Guedron et al. 2017; Burger et al. 2018; Di Marzio et al. 2018), giant otters (Fonseca et al. 2005), dolphins (Mosquera Guerra et al. 2019), turtles (Schneider et al. 2010), and caimans (Aula et al. 1994).

Toxicological assessment

Our data reveal that 49% of the fish in the present review exceed the low magnitude effect threshold of 0.2 µg Hg g⁻¹ and 10% exceeded the EC20 of 1.04 µg Hg g⁻¹ (Figure 2). Highest Hg levels were measured in 38 species in Brazil (1.04–2.84 µg Hg g⁻¹), 16 species in Colombia (1.60–4.50 µg Hg g⁻¹), 7 species in Bolivia (1.08–2.86 µg Hg g⁻¹), 1 species in French Guiana (1.11 µg Hg g⁻¹), 2 species in Suriname (1.15–1.38 µg Hg g⁻¹), 3 species in Ecuador (1.39–1.60 µg Hg g⁻¹), and 1 species in Peru (1.40 µg Hg g⁻¹).

Mercury levels in freshwater fish were reported with corresponding fish weight or length for 176 individuals. Those measurements were available only from studies conducted in Argentina, Brazil, Bolivia, Colombia, and Suriname. Figure 3 compares fish of suitable prey size for most birds (<40 cm or <500 g) against toxicity thresholds in prey fish for avian piscivores. Fifty-seven percent of this subset were above the 0.2 µg Hg g⁻¹ threshold that corresponds to MeHg levels in prey fish associated with a reproductive impairment in wild adult loons (Depew et al. 2012), 27% were above the 0.4 µg Hg g⁻¹ lower threshold value for large birds, and 2% exceeded the 1.4 µg Hg g⁻¹ upper threshold value for large birds (Fuchsman et al. 2017). Highest Hg levels in fish of prey size ranged from 1.5 to 2.3 µg Hg g⁻¹ in 1 omnivorous species measured in Bolivia and in 2 piscivorous species in Colombia and Brazil.

Median Hg concentrations for 15 fish species are plotted in Figure 4. Seven predatory species exceeded the 0.4 µg Hg g⁻¹ threshold for piscivorous birds, and 2 species (Plagioscion squamosissimus and Pellona castelnaena) exceeded 0.2 µg Hg g⁻¹, the threshold for fish and low-end threshold for piscivorous birds. Trophic position of fish species below the 0.2 µg Hg g⁻¹ threshold included 2 predator, 4 omnivore, 1 herbivore, and 1 detritivore.

Methodological issues

Data quality and completeness is a concern when assessing risk of Hg toxicity in South American wildlife. To facilitate meaningful interpretation of Hg concentration in wildlife, Hg should be measured along with Se and some measure of age, weight, or size. Methylmercury bioaccumulates in biological tissues and biomagnifies at higher trophic levels. Fish exposed to contaminants for long periods have higher concentrations than do individuals less exposed, such that concentrations in a mature fish will exceed those of an adolescent. Fish size (length or weight) are common proxies for age because these measurements are inexpensive and easy to obtain (Pfeiffer et al. 1993; Stow 1995; Dorea et al. 2004; Bhavsar et al. 2008). Selenium, found naturally in aquatic and terrestrial systems, can help to protect against the toxicity of MeHg, and correlations between Hg and Se concentrations have been reported in livers and kidneys of several fish-eating wildlife species, including common loons (Gavia immer) and bald eagles (Haliaeetus leucocephalus) (Schuhammer et al. 2008). Many ecotoxicology studies reviewed here did not measure Se-to-Hg molar ratios in biological samples, despite the potential for Se to affect interpretation of Hg’s toxic effects (Henny et al. 2002; Sampaio da Silva et al. 2013; Burger et al. 2018).

We compiled 763 unique fish species—location—date Hg means from 45 studies in Latin America, of which 17% lacked a corresponding mean length, weight, or age measurement. Where researchers recorded corresponding measures of fish length, weight, or age, those values were not reported in published text, tables, or figures in 73% of all instances. When fish Hg was compared between areas of
## Table 1. Fish Hg concentrations from studies in Latin America published between 1988 and 2019

| Country   | Species (n) | Individuals sampled (n) | Sites (n) | Species-site-year (n) | Sources                                                                 | Years of sampling          |
|-----------|-------------|-------------------------|-----------|----------------------|------------------------------------------------------------------------|-----------------------------|
| Brazil    | 97          | 5434                    | 34        | 346                  | Martinelli et al. 1988; Malm et al. 1990; Reuther 1994; Castilhos et al. 1998; dos Santos et al. 2000; Leady and Gottgens 2001; Uryu et al. 2001; Barbosa et al. 2003; Dorea et al. 2004; Bastos et al. 2006, 2007; Hylander et al. 2006; Azevedo et al. 2009; 2019; Kasper et al. 2009; Beltran-Pedreros et al. 2011; da Silva et al. 2012, 2013; de Souza Azevedo et al. 2012; Bourdineaud et al. 2015; Rocha et al. 2015; Venturieri et al. 2017 | 1986–1988, 1990–2005, 2007–2017 |
| Argentina | 7           | 563                     | 15        | 105                  | Arribere et al. 2008; Rizzo et al. 2011; Arcagni et al. 2017            | 2007, 2010, 2011            |
| Colombia  | 35          | 1832                    | 7         | 81                   | Olivero and Solano 1998; Marrugo et al. 2007; Marrugo-Negrete et al. 2008; Olivero-Verbel and Caballero-Gallardo 2013; Olivero-Verbel et al. 2016; Palacios-Torres 2018 | 1996, 2003–2012, 2014, 2016 |
| Bolivia   | 6           | 832                     | 8         | 55                   | Maurice-Bourgoin et al. 2000; Pouilly et al. 2012; Monroy et al. 2014; Guedron et al. 2017 | 1991–1996, 1998–2001, 2007, 2010–2016 |
| French Guiana | 35     | 703                     | 1         | 36                   | Richard et al. 2000                                                 | 1999                        |
| Suriname  | 15          | 626                     | 8         | 34                   | Mol et al. 2001; Ouboter et al. 2012                                  | 1997–2007, 2002–2010        |
| Ecuador   | 6           | 136                     | 2         | 33                   | Tarras-Wahlberg et al. 2001; Araújo and Cedeño-Macias 2016; de la Torre et al. 2016 | 1996–1999, 2013            |
| Mexico    | 15          | 128                     | 16        | 31                   | Elliott et al. 2015                                                  | 1998–2001, 2000–2003        |
| Peru      | 2           | 36                      | 5         | 19                   | Gutleb et al. 1997                                                   | 1990–1993                   |
| Paraguay  | 3           | 50                      | 8         | 11                   | Hylander et al. 1994                                                  | 1992, 1998–2001             |
| Nicaragua | 4           | 37                      | 5         | 7                    | Elliott et al. 2015                                                  | 2007                        |
| Costa Rica| 5           | 14                      | 3         | 5                    | Elliott et al. 2015                                                  | 2007                        |
| Total     | —           | 10 391                  | 115       | 763                  | —                                                                      | —                           |

*Countries are listed along with number of unique species measured, total number of individuals sampled, sites sampled, number of unique combinations of species-site-year, referenced sources, and years of data collection.
presumed high and low or no Hg contamination, the bioaccumulative effects of age or size on fish Hg concentration was not controlled for in 70% of measures. Of the fish sampled in South America for Hg toxicity, 96% did not measure corresponding Se levels.

DISCUSSION

Toxicological assessment

Mercury in fish. Many fish samples from Latin America exceeded the adverse effects TRVs for muscle Hg concentration and avian dietary Hg; thus fish, fish-eating birds, and other wildlife in Latin America could be at risk of Hg toxicity. From the small subset of data for which fish size was measured, 57% of forage fish were above the TRV for common loon (0.2 µg Hg g\(^{-1}\); Depew et al. 2012), 27% exceeded the lower range TRV for large birds (0.4 µg Hg g\(^{-1}\); Fuchsm et al. 2017), and 2% exceeded the upper range (1.4 µg Hg g\(^{-1}\); Fuchsm et al. 2017). Median Hg levels from a subset of the data found predatory fish are most at risk for toxic levels of Hg, which is consistent with other findings that Hg biomagnifies up the food web (Barbosa et al. 2003; Eisler 2004; Sampaio da Silva et al. 2013). These results suggest that not only are fish at risk of Hg toxicity in Latin America, but fish-eating birds are also at risk of Hg toxicity from consumption of prey fish. Studies that have measured Hg concentrations in avian piscivores have found Hg levels in exceedance of recognized toxicity benchmarks (LOAE: 0.2 µg Hg g\(^{-1}\) and moderate risk: 1.0 µg Hg g\(^{-1}\); Ackerman et al. 2016). In Bolivia, bird Hg levels found in association with Au mining activities were high in Andean coots (0.174 µg THg g\(^{-1}\) in blood) and white-tufted grebes (2.684 µg THg g\(^{-1}\) in blood; Guedron et al. 2017). Not only are strictly piscivorous bird species at risk of Hg exposure from mining activities, but those that feed on fish carcasses as well. Biomagnification in turkey vulture (Di Marzio et al. 2018), crested caracara, and black vulture feathers also reflected Hg contamination in respective sampling areas of Brazil and Patagonia (Hylander et al. 1994; Di Marzio et al. 2018). In a review of fish THg in the western United States and Canada, 30% of individual fish samples, totaling 96 310, exceeded the United States Environmental Protection Agency (USEPA) fish tissue criterion of 0.3 µg/g (Eagles-Smith, Ackerman et al. 2016). In comparison, 49% of fish samples (763 mean species–year–site; Table 1) in our study exceeded the low magnitude effects of 0.2 µg Hg g\(^{-1}\) TRV, 10% exceeded the EC20 of 1.04 µg Hg g\(^{-1}\), and concentrations ranged from 0.092 to 4.50 µg Hg g\(^{-1}\). Although there are likely biases based on variation in species sampled and other factors such as habitat type and bioenergetics in some countries, fish Hg concentrations are higher in Latin America than in North America, yet research on Hg toxicity in Latin American wildlife is scarce. Although forage fish in Latin America are preyed upon by piscivorous birds, such as

![Figure 1](https://example.com/figure1.png)

Figure 1. Fish Hg concentrations synthesized from all available Latin American literature. Mercury data (µg THg g\(^{-1}\) wet wt) collected from 1986 to 2017. Data sorted according to Hg thresholds in fish associated with low risk of reproductive impairment in piscivorous birds and toxicity in fish (0–0.19 µg Hg g\(^{-1}\); Depew et al. 2012) (A); risk of reproductive impairment in wild adult loons (0.2–0.39 µg Hg g\(^{-1}\); Depew et al. 2012) (B); and risk of reproductive impairment in large birds (>0.4 µg Hg g\(^{-1}\); Fuchsm et al. 2017) (C).

![Figure 2](https://example.com/figure2.png)

Figure 2. Average THg levels (µg Hg g\(^{-1}\) wet wt) in muscle tissue of freshwater fish from Latin America compiled from 45 studies and data collected from 1986 to 2017. The dashed lines correspond to the MeHg toxicity thresholds in adult fish of 0.2 µg Hg g\(^{-1}\) (Beckvar et al. 2005; Dillon et al. 2010) and 1.04 µg Hg g\(^{-1}\) (Dillon et al. 2010; Fuchsm et al. 2016), corresponding to potential sublethal effects in fish. THg = total mercury; TRV = toxicity reference value.
concentrations in South America (Tarras 2001). The most recent data on present review. Although 1992 (Hylander et al. 1994), nearly 3 decades prior to the Ecuador, yet these 6 species have been assessed for Hg toxicity for Nicaragua (Mol et al. 2001), toothed whales (Caceres-Saez et al. 2018), giant otters (Gutleb et al. 1997), jaguars (Junior et al. 2018), and caimans (Vieira et al. 2011). As such, other wildlife populations in Latin America may also be at risk of Hg toxicity from fish consumption.

Mercury risk assessment studies target different study species based on specific research objectives and logistic barriers. For instance, in Central America, fish Hg was measured in prey species for fish-eating birds to inform the Hg risk to migratory osprey through diet (Elliott et al. 2015). Therefore, Nicaraguan fish measurements are for benthic invertivores and omnivores, species that occupy relatively low trophic positions and thus have lower biomagnification potential. Because trophic position and Hg concentration are positively correlated, piscivorous fish in Nicaragua, which occupy high trophic positions and thus possess greater biomagnification potential, may contain higher Hg concentrations than the current species measured. However, no measurements on carnivorous fish in this region are reported in the scientific literature.

The paucity of research on Hg risk to wildlife in Latin America in terms of number of species assessed for Hg risk and the timing of research, such as during a wet or dry season, is a concern. For instance, few fish species have been assessed for Hg risk in Peru (n = 2), Paraguay (n = 3), Nicaragua (n = 4), Costa Rica (n = 5), and Ecuador (n = 6). Only 6 species have been assessed for Hg toxicity for Ecuador, yet these fish contained some of the highest Hg concentrations in South America (Tarras-Wahlberg et al. 2001). The most recent data on fish in Paraguay are from 1992 (Hylander et al. 1994), nearly 3 decades prior to the present review. Although fish Hg is described for 35 species in French Guiana, those data were collected in 1999, 2 decades prior to the present review. Recent reports suggest ASGM is still on the rise in South America (AMAP/UNEP 2019), and data collected pre-2000s may not represent current environmental conditions. Continued monitoring is required for proper assessment of modern Hg risk to wildlife. Artisanal small-scale Au mining operates on an illegal basis throughout South America, and there is no comprehensive record of their existing locations. The maps in Figure 1 reveal large gaps in our knowledge of Hg risk to wildlife. These knowledge gaps are particularly concerning in regions with known or potential ASGM activity. Unstudied ASGM and Hg deposition hotspots in Latin America should also be a research priority going forward. In contrast to the 45 publications synthesized in the present paper, Hg exposure threats to wildlife in North America have been extensively studied and were recently subject to a comprehensive synthesis and summarized in several review papers (Ackerman et al. 2016; Eagles-Smith, Wiener et al. 2016; Jackson et al. 2016; Fuchsman et al. 2017). Due to the high rates of ASGM Hg emissions in Latin America, wildlife may be at greater risk of Hg toxicity, thus data-deficient regions of Latin America should be a priority for future research and monitoring.

Toxicity assessment issues

In a review of TRV thresholds in birds, Fuchsman et al. (2017) outline several issues inherent in wildlife and ecological Hg risk assessments. Although these issues are important to consider when assessing the results of dose–response experiments, in their review of observed fish Hg in Latin America the most pertinent issues related to Hg risk

Figure 3. Mercury concentrations (µg Hg g\(^{-1}\) wet wt) in fish muscle tissue, and corresponding fish weight (A) and length (B) from field studies in Latin America for prey fish of size and weight for piscivorous birds (<40 cm or <500 g). Dashed lines show toxicity thresholds. The 0.2 µg Hg g\(^{-1}\) threshold corresponds to MeHg levels in prey fish associated with a reproductive impairment in wild adult loons (Depew et al. 2012) and the 0.4 µg Hg g\(^{-1}\) and 1.4 µg Hg g\(^{-1}\) thresholds correspond to the lower and upper range values for large birds (794–5500 g), respectively (Fuchsman et al. 2017). THg = total mercury, TRV = toxicity reference value.
evaluation identified by Fuchsman et al. (2017) are causality and extrapolation.

In ecological field studies, the cause of any observed effect or outcome can be difficult to determine with a high degree of probability. Identification of the exact source of environmental contamination in an ecosystem or organism can be extremely challenging. Several Latin American studies claim to measure Hg in fish from sites contaminated by ASGM activities (Leady and Gottgens 2001; Barbosa et al. 2003; Dorea et al. 2004; Hylander et al. 2006; de Souza Azevedo et al. 2012; Sampaio da Silva et al. 2013; Monroy et al. 2014). Mercury from Au mining entering an aquatic ecosystem, either directly or through atmospheric deposition, is dependent on habitat-specific factors such as hydrology. Hydrologic factors such as natural seepage and flooding (Hylander et al. 2006) influence methylation rates and bioaccumulation in food chains. However, the many different fates, pathways, and physical processes governing Hg deposition and methylation complicate the linkage between Hg released during Au mining activities and Hg accumulated in individual animals. Regardless of the contaminant’s source, knowledge of Hg risk to wildlife is critical. However, causality is important for management and conservation initiatives. In the present summary, as in many studies here synthesized, we assume the majority of Hg contamination in South America is from ASGM although other anthropogenic activities (industrial production, fuel combustion, waste treatment), as well as natural sources, also contribute to global Hg emissions (AMAP/UNEP 2019). Thus, we recommend further focused research to investigate whether fish Hg levels are directly impacted by ASGM.

Another issue common to toxicity assessments, and which was observed throughout the present review, is extrapolation. Single-species TRV extrapolation ignores important differences in body weight (Sample et al. 2014; Fuchsman et al. 2017), feeding guild, and MeHg tolerance adaptations between species (Eagles-Smith et al. 2009; Robinson et al. 2011). Body weight affects paracellular absorption and oxidative stress resistance in birds, which suggests that size could affect MeHg exposure–response relationships. Oxidative stress is a symptom of MeHg toxicity. In tropical systems, birds tend to be larger, live longer, metabolize slower, and appear more resistant to oxidative stress than temperate system birds (Jimenez et al. 2014). Due to greater oxidative stress resistance, large birds may be more tolerant to MeHg toxicity. Small animals have fast metabolic and contaminant elimination rates. Feeding guild might also indicate MeHg toxicity resistance in birds. Piscivores may have adapted greater Hg tolerance, via demethylation capacity, for example, due to exposure to high MeHg via the greater bioaccumulation potential in fish, their obligate prey (Thompson and Furness 1989; Henny et al. 2002; Eagles-Smith et al. 2009). These considerations are particularly important for toxicity assessments in Latin America due to the very high biodiversity of fish and birds. Additionally, other factors such as interspecific differences in cerebral metabolism and toxicodynamics of MeHg and Se
suggest that common loons may be more susceptible to MeHg toxicity than are bald eagles (Scheuhammer et al. 2008). Many avian risk assessments use loon Hg TRVs, which suggests some of these toxicity risks are underestimated and others overestimated (Elliott et al. 2015; Fuchsman et al. 2017). We chose to assess the Hg risk to birds with TRVs for common loons and the lower and upper ranges for large birds (794–5500 g; Fuchsman et al. 2017). Our confidence in the range of TRVs for large birds is greater than that of common loon (see Fuchsman et al. 2017). Common loon Hg TRV was included for comparison between Latin America and North America because it is the predominant reference value in the avian risk assessment literature (Depew et al. 2012; Ackerman et al. 2016). However, future assessments should consider body size, feeding guild, and known evolutionary adaptations specific to the species of interest before choosing a Hg TRV. Furthermore, most toxicity studies have been conducted on temperate species. We assume here that South American and tropical bird species will respond similarly to North American species. Given that previous studies have demonstrated species-specific responses to Hg exposure, we acknowledge that TRVs may vary between North and South American species.

**Methodological issues**

We identified several issues with the literature available on fish Hg in Latin America. The most serious issues are failure to consider fish weight, length or age, and the Hg-to-Se molar ratio during data collection and analysis research phases. We also acknowledge the deficiency that our assessment of Hg risk to fish and fish-eating birds does not account for species, age, and Se due to deficiencies in the data.

Age is widely recognized as an important predictor of Hg concentration in fish because the length of time spent in contaminated environments, and thus bioaccumulation potential, is greater in older individuals (Dorea et al. 2004; Azevedo et al. 2009; Coelho et al. 2010). Standardization of Hg bioaccumulation factors in fish significantly improves the comparability across samples (Dorea et al. 2004; Depew et al. 2013; Skudder Eikenberry et al. 2015). Without considering fish age, and the associated bioaccumulation bias, fish Hg interpretation becomes ambiguous and comparison among studies uninformative. Fish weight and length are highly correlated with age and yet 21% of the studies did not measure fish age, weight, or length. In studies that did measure fish age, weight, or length, 76% did not factor those parameters into their assessment of the Hg measures. This is problematic for determining fish Hg toxicity, as well as the risk to other wildlife that consume them. In fish, organism size can also affect bioaccumulation distinctly from age-related exposure time. For omnivorous species, larger fish will eat larger prey, potentially at a higher trophic level and thus greater biomagnification risk, compared to smaller fish (Borga et al. 2004). Body size measurements could explain some of the interspecific Hg variability observed between sites and years.

Mercury and Se have an antagonistic effect in biological tissues. In fish, Se increases demethylation of MeHg to IHg in the intestine and prompts the uptake and elimination of the resulting IHg (Wang and Wang 2017). Selenium not only prevents but also reduces MeHg bioaccumulation and its effects (Ralston and Raymond 2010; Klimstra et al. 2012). This ameliorative interaction may confound risk assessment interpretation and cross-study comparisons. Despite the importance of Se’s “protective effect” in biota, many studies do not include Se-to-Hg molar ratios in their analyses. In our Latin American Hg summary, <1% of fish data included measurement of Se concentrations (Sampaio da Silva et al. 2013). Despite widespread demonstration of Se’s ameliorative effect on Hg toxicity (Peterson et al. 2009), Se itself is also toxic to animals at high levels. Furthermore, the ability to ameliorate toxicity may differ between organisms (Scheuhammer et al. 2008; Burger et al. 2018). For example, the Se–MeHg interaction both positively and negatively affected zebrafish reproduction (Penglase et al. 2014), and forms of Se can be highly toxic to waterbirds (Ohlendorf et al. 1986). Although much is still unknown about the synergistic effects of Se and Hg, in individuals and species, it is clear that Se is an important factor in Hg risk assessment. Therefore, studies of Hg effects and risk assessments in wildlife should measure Se-to-Hg molar ratios and consider these interactions during analysis.

**Other factors to consider**

Although several intrinsic and extrinsic factors affect Hg bioaccumulation and biomagnification in food webs, in the present review we discuss only hydrology and its potential effects on Latin American fish Hg concentration (for a complete summary of factors affecting Hg bioaccumulation, see Eagles-Smith et al. 2018). Biogeochemical conditions in aquatic environments regulate the rate of methylation and MeHg production (Gilmour et al. 1992, 2013; Hall et al. 2008), and water chemistry is important to metal bioaccumulation (Mason et al. 2000). There is a wide variety of aquatic habitat types in South America, and these ecosystems experience a wide range in precipitation throughout the year. In those environments, the season (dry or wet) in which fish sampling occurs is significant because flooding may temporarily alter biogeochemical components of the system, such as O content, pH, and prey availability (Oliveiro and Solano 1998; Leady and Gottgens 2001; Uryu et al. 2001; Dorea et al. 2004; Marrugo-Negrete et al. 2008; Azevedo et al. 2009; Costa et al. 2009; Barletta et al. 2012; da Silva et al. 2012; Ouboter et al. 2012; Guedron et al. 2017; Obrist et al. 2018). Many Latin American bird species are seasonal migrants, whose movement depends on fluctuating water levels. Changes in prey availability from seasonal migration, enabled by precipitation during the wet season, may influence Hg bioavailability and accumulation in predatory species (Hylander et al. 2006; Costa et al. 2009; Barletta et al. 2010). Reservoir lake formation is another hydrologic environmental change that influences MeHg production. For example, reservoir construction at a
Brazilians hydroelectric dam resulted in increased Hg concentrations in piscivorous fish 3 y postflooding (Hylander et al. 2006). An increase in C and decrease in O content after flooding increased decomposition, creating anoxic conditions and enhanced methylation (Hylander et al. 2006). Contrarily, in systems where the conditions for methylation are not present, MeHg in fish and other components of aquatic food webs may not reflect actual total Hg loadings. It is important that future studies consider habitat factors, such as hydrology, including natural flood cycles and artificial flooding, when assessing Hg toxicity in Latin America. Additionally, the extent of Hg exposure in terrestrial wildlife using riparian habitats in polluted regions of Latin America remains to be evaluated. Similarly to fish and aquatic birds, exposure may also be higher in terrestrial species using tropical flood plain regions, which are important in the production and bioaccumulation of MeHg (Guimarães et al. 2000). Though beyond the scope of the present work, Hg in riparian terrestrial food webs of Latin America is an important topic requiring more research (for further information, refer to Supplemental Data file).

CONCLUSIONS

Recommendations

There are many challenges with determining whether Hg risk to wildlife in Latin America is greater now than in the past because of the inconsistency in the data, such as missing years, few repeated measurements of sites over time, countries with missing information, and sampling period variation between wet and dry seasons. However, well-designed monitoring studies can reveal how anthropogenic Hg emissions influence bioaccumulation patterns in wildlife and inform targeted mitigation measures. Because Hg emissions have continued to increase for the past 2 decades, long-term wildlife Hg toxicity monitoring is an important area for future research.

Using fish to assess landscape-scale spatiotemporal variation in Hg availability within food webs is an effective bioassessment tool for scientists and resource managers. However, accurate interpretation and assessment of fish Hg risk can be complicated and/or undermined by variability in fish age, size, species (Peterson et al. 2007; Walters et al. 2010), and landscape and habitat factors that alter methylation rate, such as hydrologic patterns (Shanley et al. 2012; Drenner et al. 2013). Standardized and uniform fish Hg measurements will increase accuracy of risk assessments and facilitate future spatiotemporal comparison between studies (Depew et al. 2013).

We strongly recommend that any future Hg research include measures of fish age and size to account for bioaccumulation and to facilitate risk assessment and comparison across species, space, and time. We also strongly suggest that Se-to-Hg molar ratios be included in wildlife toxicity assessments due to Se’s ameliorative effects on Hg toxicity. Sampling during both wet and dry season has proven to account for large variability in fish Hg levels in previous studies. Therefore, we also urge researchers to consider habitat factors like hydrology when designing field studies.

For comparison between areas of suspected high and low Hg contamination, we recommend mixed-effects models to analyze these data. Mixed models allow the above factors (age, size, Se:Hg, season, site, year) to vary as random effects so true patterns in fish Hg toxicity may be examined. Adding site as a random effect can take into account variation in methylation rates due to in-situ biogeochemical environmental characteristics that influence methylation and thus bioavailability.

Extrapolation of toxicity data to new contexts and comparison over time and space are inherent in ecological risk assessment. However, it is important to recognize there are similarities and differences among studies, species, and sites. It is our intent to provide recommendations for future research and monitoring to improve our understanding of Hg risk to wildlife in Latin America and to provide more comprehensive and usable data. The present review suggests that fish and fish-eating wildlife in many areas of Latin America are at risk of Hg toxicity. Future Hg work should focus on areas of South America where information on the risk to wildlife is currently inadequate. In particular, Peru, Chile, Uruguay, the eastern and northern regions of Brazil, Venezuela, Ecuador, and Colombia require further investigation. More research and monitoring is required to confirm whether birds, fish-eating wildlife, and riparian terrestrial wildlife are at risk of Hg exposure, and to what extent ASGM contributes to Hg bioaccumulation in wildlife in Latin America.

Finally, to our knowledge, the present review is the first attempt to summarize the data on Hg in fish, and risk to fish-eating wildlife, from Central and South America. We recognize the limitations of our exercise. Ideally in future, resources can be obtained to do a more comprehensive search for data from other sources, particularly government agencies, and also NGOs, indigenous groups, and industry in the manner of the Western Mercury Initiative that resulted in the comprehensive assessments of Hg exposure in North America (Ackerman et al. 2016; Jackson et al. 2016). Additionally, we would encourage other researchers to engage with colleagues in Latin America to undertake focused and well-designed studies, especially in areas with the most pressing lack of data. Regarding ongoing and widespread ASGM in Latin America, ultimately what is needed is support and resources for the kind of socioeconomic measures, along with enforcement capability and sociopolitical will, necessary to help redress the imbalance between the need for income in impoverished communities and protection of some of the most biologically diverse systems on the planet (Miserendino et al. 2013; Saldarriaga-Isaza et al. 2013).

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SUPPLEMENTAL DATA
Mercury in riparian terrestrial food webs.

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