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Total mercury in wild fish in Guizhou reservoirs, China

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Abstract

The health hazard of mercury (Hg) compounds is internationally recognized, and the main pathways for methylmercury (MeHg) intake in humans are through consumption of food, especially fish. Given the large releases of Hg to the environment in China, combined with the fast development of hydropower, this issue deserves attention. Provided similar mobilization pathways of Hg in China as seen in reservoirs in North America and Europe one should expect increased Hg contamination in relation to future hydropower reservoir construction in this country. This study presents total Hg (THg) concentrations in wild fish from six Guizhou reservoirs, China. The THg concentrations in fish were generally low despite high background levels in the bedrock and depositions from local point sources. The over all mean ± SD concentration of THg was (0.066 ± 0.078) μg/g (n = 235). After adjusting for among-reservoir variation in THg, there were significant differences in THg among functional groups of the fish, assumed to reflect trophic levels. Predicted THg-concentration ratios, retrieved from a mixed linear model, between the functional groups were 9:4:4:1 for carnivorous, omnivorous, planktivorous and herbivorous fish. This result indicated that MeHg accumulation may prevail even under circumstances with short food chains as in this Chinese water system. No fish exceeded recommended maximum THg limit for human consumption set by World Health Organization and the Standardization Administration of China (0.5 μg/g fish wet weight (ww)). Only six fish (2.5%) exceeded the maximum THg limit set by US Environmental Protection Agency (0.3 μg/g fish ww).

Key words: bioaccumulation; mercury methylation; mixed linear models; aquatic food web; lakes

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Introduction

Elevated mercury (Hg) concentrations in fish are a world-wide environmental concern. High Hg concentrations are found not only in fish living in waters receiving direct discharges of Hg, but also where no direct source exist. Long range atmospheric deposition in combination with transformation and mobilization from catchments and sediments are important contributors to elevated Hg concentrations in fish (Munthe et al., 2007).

Recommended maximum Hg concentrations in food are proxies for total consumption and potential accumulation over time. Therefore, mean daily intake of Hg through relevant food must be taken into account when addressing maximum limits of Hg. Various consumption advisories on fish due to Hg concentration exist, and restricted international standards normally range between 0.3 and 0.5 μg/g wet weight (ww) (WHO, 1990; US EPA, 2001; SAC, 2005).

Fish in hydropower reservoirs in North America and Europe were observed to have elevated Hg concentrations in the late 1970s and early 1980s (Abernathy and Cumbie, 1977; Lodenius et al., 1983). Several-fold increase in fish Hg concentrations was found, especially the first years after reservoir establishment (Verdon et al., 1991; Bodaly et al., 2007). The phenomenon is believed to be related to decomposition of flooded organic matter, providing good conditions for bacterial methylation of inorganic Hg to methylmercury (MeHg) (Lucotte et al., 1999; St Louis et al., 2004). MeHg bioaccumulates in the food web and particularly high concentrations are found in old carnivorous fish.

In China, Hg in the environment has recently received increased attention. Chinese coal fired power plants, metals smelters and other industries release roughly a quarter of the worlds annual total Hg emissions to the atmosphere (about 700 (± 300) tons) (Wu et al., 2006) out of a global estimated total of 1930 tons (UNEP, 2008). In addition, there are numerous hot spots with direct Hg discharge to water, from mine tailings, industrial waste sites, etc. (Jiang et al., 2006; Feng and Qiu, 2008). Chinese industry, particularly in the production of acetaldehyde, is still using considerable amounts of Hg in their processes (Jiang et al., 2006; Feng and Qiu, 2008). On top of all these anthropogenic sources, parts of China have high natural Hg

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releases due to elevated Hg concentration in the bedrock (Feng and Qiu, 2008).

Besides all the Hg releasing sources, another issue of concern is the fast development of the hydropower sector in China. China has plenty of both large and small scale hydropower dams, and new dams are being constructed rapidly (Huang and Yan, 2009). The installed hydropower effect capacity in China in 2007 was 145 GW, which was a doubling over only seven years; a further doubling is planned by 2020 (Huang and Yan, 2009). If the same mobilization of Hg as seen in reservoirs in North America and Europe will happen under typical Chinese natural conditions, there is reason for concern for increased Hg contamination due to hydropower reservoir construction in the future.

The movement, methylation and bioaccumulation of Hg in different environmental settings are modified by several lake specific factors like hydrology, water quality, trophic structure and temperature (Gilmour and Henry, 1991). Mercury concentrations in fish may vary by an order of magnitude or more in regions with similar atmospheric Hg deposition rates (Munthe et al., 2007). Biophysical control of MeHg uptake in fish includes nature and structure of the food web, stratification of the water column where MeHg accumulates in anoxic bottom waters, near-shore areas of sedimentary MeHg production in large lakes and zones of MeHg production that are disconnected from aquatic food webs (Watras et al., 1994; Herrin et al., 1998; Gorski et al., 2003; Choe et al., 2004; Hammerschmidt and Fitzgerald, 2004; Martin-Doimeadios et al., 2004; Bank et al., 2005; Eckley et al., 2005; Evers et al., 2005). Biological control of MeHg uptake in fish includes food web structure, fish population age structure, growth rates and physiological controls on uptake (Harris and Bodaly, 1998; Lawson and Mason, 1998; Lucotte et al., 1999; Swanson et al., 2003; Simoneau et al., 2005). Chemical controls of MeHg uptake in fish includes MeHg partitioning and complexation, bioavailability of MeHg to lower trophic levels, direct or indirect inhibition by higher molecular weight dissolved organic matter, particulate matter and pH (Lange et al., 1993; Back and Watras, 1995; Tsui and Wang, 2004; Chen et al., 2005). Carbon in soil may also stimulate bacterial methylation, facilitating the accumulation and biomagnification in aquatic food webs (Bodaly et al., 1984; Paterson et al., 1998; Guimaraes et al., 2000; Hylander et al., 2006). In temperate and boreal ecosystems the Hg concentration in fish is usually positively correlated to fish size, age, trophic level and dissolved organic carbon (DOC) in the water, but negatively correlated to water pH and trophic condition of the lake (Driscoll et al., 1994). In tropical ecosystems the variation often reflex the high complexity of the food chains and the Hg concentration in fish may not be that easily explained. Several studies have shown that tropical fish have a higher bioaccumulation capacity than high latitude fish species (Oliveira et al., 1996, 2000). Although elimination of Hg from fish is favoured by high temperature, rates of methylation could also be greater (Trudel and Rasmussen, 1997). In a recent study, the Hg input-output budgets of two reservoirs in Guizhou Province, south-western China, were estimated (Feng et al., 2009b). It was shown that the reservoirs act as net sinks for total Hg (THg), but net sources of MeHg. The MeHg production however seems moderately low compared to results from North America.

A few scattered studies available suggest low concentrations of Hg in fish in China (Jin et al., 2006; Zhang et al., 2007; Chen et al., 2008; Cheung et al., 2008; Chung et al., 2008; Li et al., 2009). Based on knowledge from North America and northern Europe, higher concentrations in Chinese fish, especially from the reservoirs may be expected. But there has so far been very little focus on Hg methylation and mobilization in reservoirs. Given the large releases of Hg to the environment in China in combination with the fast development of hydropower, the issue may however deserve some attention.

The health hazards of mercury compounds have been internationally recognized for quite some time. The main pathways for methyl mercury intake in humans are through the consumption of food, especially fish (Clarkson, 1993). But as fish consumption also is an excellent source of high-quality proteins and omega-3 fatty acids, low in saturated fat, health authority advices should be well balanced between risks and benefits. Our primary objective in this article was to present Hg concentrations in wild fish from Guizhou reservoirs, China. We also wanted to explore the variation observed in relation to fish ecology and water chemistry.

1 Materials and methods

1.1 Study area

Guizhou Province (24°30′–29°13′N, 103°01′–109°30′E; Fig. 1) is located in southwestern China with an average elevation of 1100 m above sea level and a land area of 176,000 km^2. Its climate represents a typical subtropical humid monsoon with an average

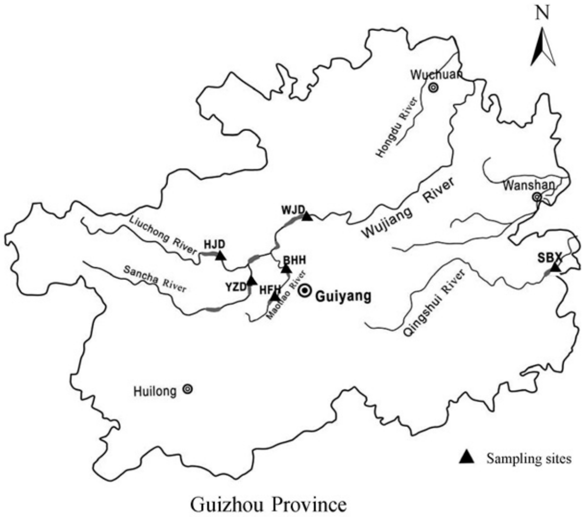

Fig. 1 Sampling sites in Guizhou Province, China. Baihua (BHH), Hongfeng (HFH), Yinzidu (YZD), Hongjiadu (HJD), Sanbanxi (SBX) and Wujiangdu (WJD) reservoirs.

annual temperature of 15°C and an annual precipitation of 1100–1400 mm. Guizhou has a typically karstic topography and the bedrocks in the province are mainly limestone and dolomite. Guizhou is the most abundant region for water resources in China due to distinct climate conditions. With the implementation of the "Go West" policy, a great number of large reservoirs have been/are being constructed along rivers in this region (e.g., Wujiang River, Xijiang River, Beipanjiang River and Yuanjiang River) for hydropower production.

Six reservoirs were selected to collect fish samples (Fig. 1). Baihua reservoir (BHH) was seriously contaminated with Hg by Guizhou Organic Chemical Plant which used metallic mercury as catalyser to produce acetic acid (Yan et al., 2008). For the other five reservoirs, no direct point source of mercury contamination is identified. BHH and Hongfeng (HFH) reservoirs are located on Maotiao River, which is a branch of Wujiang River. Yinzidu (YZD) and Hongjiadu (HJD) reservoirs are located on Sanca River, Liucong River, respectively, which are two main tributaries of Wujiang River at the upper reach. Wujiangdu (WJD) reservoir is located on the mainstream of Wujiang River. Sanbanxi (SBX) Reservoir is located on Qingshuijiang River, which flows to Yuanjiang River. A summary of characteristics of these reservoirs is listed in Table 1. BHH, HFH and WJD are old reservoirs with ages more than 30 years, while SBX, HJD and YZD are newly built reservoirs. The trophic status varies from oligotrophic to eutrophic (Table 1).

1.2 Fish sampling and analysis

Fish were collected during 2007–2009 from both professional fishermen at local markets and by cooperation with game fishermen. The number of fish species and the number of specimens sampled for each species were limited by availability of fish during the sampling campaign. All fish were visually inspected for fin and body deformations to avoid farmed fish in the samples. No stocking programs were known in these systems. The collected fish were stored alive in barrels with water and air purge, or dead on ice in freeze-boxes until sampling at the lab. The fish collected (Table 2) may not represent all species in the reservoirs, but the most abundant ones should be included. In total, 235 fish samples were collected from, but not evenly distributed between the reservoirs. The weight and length of the fish were measured before a sample of the dorsal muscle was removed and stored frozen for Hg analysis.

All samples were analyzed for THg, by digesting the muscle tissue in HNO_3:H_2SO_4 ratio of 7:3 (*V/V*) at 95°C for 3 hr before measuring by CVAFS (Pfeil and Stalvey, 2004; Yan et al., 2005). Quality control consisted of using duplicates, method blanks, and standard reference material. Blank spikes and duplicates were taken regularly (> 10% of samples) throughout each sampling process. As fish standard reference material, NRCC-TORT-2 was used. THg concentrations in muscle were reported in μg/g fresh weight. THg average concentration in the reference material was (0.28 ± 0.02) μg/g, which is comparable with the certified value (0.27 ± 0.06) μg/g.

In order to explore functional relationships between observed THg levels and fish ecology, the fish species were categorized by food preferences into four ecological functional groups (carnivorous (CV), omnivorous (OV), planktivorous (PV) and herbivorous (HV)). Four species, accounting for 23% of the samples taken, were carnivorous fish, four species (55%) were omnivorous, two species (10%) were planktivorous and two species (12%) were herbivorous fish. The most frequent species collected among the carnivorous were catfish (*Parasilurus asotus*), which accounted for 12% of the total number of captures. Of the omnivorous, planktivorous and herbivorous fish, the species most frequently collected were common carp (*Cyprinus carpio*), bighead carp (*Aristichthys nobilis*) and grass carp (*Ctenopharyngodon idellus*), which accounted for 40%, 9% and 10% of the total number of captures, respectively.

1.3 Statistics

Two THg outliers (one crucian carp and one blunt snout bream) were excluded from the dataset due to questionable levels. Nevertheless, if these data were taken into account, no change in major conclusions would be expected. Prior to each statistical analysis, groups with $n < 10$ were excluded. Common carp is the only species sampled evenly among reservoirs, and is the only species included when comparing THg among reservoirs. This among-reservoir analysis was conducted by fitting an ANCOVA where ln(THg) was predicted as function of reservoir ID and ln(fish length).

In other analyses where unique species effects and functional group effects were explored, reservoir was

Table 1 Characteristics of reservoirs in Guizhou Province, China

Reservoir	Location	Area (km^2)	Volume (m^3)	Max depth (m)	Mean depth (m)	Retention time (yr)	Altitude (m)	pH	TP (mg/L)	Trophic status	Chl-*a* (mg/m^3)
BHH	106°27′E, 26°35′N	14.5	1.91×10^9	45	12.5	0.102	1189	7.15–8.90	0.074	Eutrophic	18.80
HFH	106°20′E, 26°24′N	32.2	6.01×10^9	45	10.5	0.325	1233	7.20–8.70	0.047	Eutrophic	29.00
WJD	106°46′E, 27°19′N	48	21.4×10^9	95	44.6	0.145	760	7.20–9.00	0.034	Mesotrophic-eutrophic	11.78
SBX	n.a.	79.56	37.5×10^9	n.a.	n.a.	n.a.	400	7.00–8.00	0.300	Eutrophic	n.a.
YZD	104°18′E, 26°08′N	13.9	5.27×10^9	95	80	0.120	1086	7.20–8.60	0.018	Oligotrophic-mesotrophic	3.70
HJD	105°51′E, 26°53′N	80.5	49.5×10^9	110	61.5	1.04	1140	7.10–7.90	0.028	Oligotrophic-mesotrophic	2.25

TP: total phosphorous; Chl-*a*: chlorophyll-*a*; n.a.: not available.

Table 2 Fish species and their functional feeding groups with corresponding THg-concentrations, Guizhou Province, China

Scientific name	Common name	Environment	Functional group	Food items	THg (μg/g) Mean	SD	Median	Range	Fish number	Percentage* (%)
Parasilurus asotus	Catfish	Demersal	Carnivorous	Small fish, insects, crustaceans, plankton, rotting flesh and plants	0.117	0.109	0.092	0.012–0.445	28	12
Opsariichthys bidens	Chinese hooksnout carp	Benthopelagic	Carnivorous	Invertebrates, fish	0.114	0.091	0.072	0.029–0.350	13	6
Erythroculter ilishaeformis	Predatory carp	Benthopelagic	Carnivorous	Insects, crustaceans and fish	0.133	0.058	0.130	0.048–0.231	10	4
Micropterus salmonides	Largemouth black bass	Benthopelagic	Carnivorous	Crustaceans, insects, fish, crayfish and frogs	0.043	0.020	0.037	0.027–0.065	3	1
			Carnivorous total		**0.116**	**0.094**	*0.100*	**0.012–0.445**	**54**	**23**
Cyprinus carpio	Common carp	Benthopelagic	Omnivorous	Aquatic insects, crustaceans, annelids, mollusks, weed and tree seeds, wild rice, aquatic plants and algae	0.045	0.058	0.021	0.002–0.320	94	40
Carassius carassius	Crucian carp	Demersal	Omnivorous	Plants, insect larvae and plankton	0.095	0.057	0.093	0.016–0.228	29	12
Hemiculter bleekeri bleekeri	Minnow	Benthopelagic	Omnivorous	Plants and detritus, insects	0.311	–	–	–	1	0
Oreochromis mossambicus	Tilapia	Benthopelagic	Omnivorous	Algae, phytoplankton, aquatic macrophytes, zooplankton, small insects, shrimps, earthworms, fish, detritus and sediment	0.009	0.004	0.008	0.005–0.014	4	2
			Omnivorous total		**0.057**	**0.065**	**0.028**	**0.002–0.320**	**128**	**54**
Aristichthys *nobilis*	Bighead carp	Benthopelagic	Planktivorous	Zooplankton	0.069	0.093	0.028	0.007–0.315	22	9
Hypophthalmichthys molitrix	Silver carp	Benthopelagic	Planktivorous	Phytoplankton and zooplankton	0.022	0.022	0.022	0.006–0.038	2	1
			Planktivorous total		**0.065**	**0.090**	**0.028**	**0.006–0.315**	**24**	**10**
Megalobrama amblycephala	Blunt snout bream	Benthopelagic	Herbivorous	Plant material	0.011	0.007	0.009	0.003–0.024	6	3
Ctenopharyngodon idellus	Grass carp	Demersal	Herbivorous	Higher aquatic plants and submerged grasses, detritus, insects and other invertebrates	0.010	0.005	0.009	0.002–0.020	23	10
			Herbivorous total		**0.010**	**0.005**	**0.009**	**0.002–0.024**	**29**	**12**
			Over all total		**0.066**	**0.078**	**0.028**	**0.002–0.445**	**235**	**100**

*: Fish with this functional feeding group

included as a random effect to account for spatial effects related to local conditions. In order to account for heteroscedasticity, both individual length and THg values were ln(transformed in these analyses). The ln(transformed fish length values) were then standardized (hereafter st_length) at species level by subtracting each ln(length observation) with the respective species mean value of ln(length) and then dividing by the species standard deviation of ln(length). This operation enabled analysing size effects at relevant species-specific size scales. Estimates of various group effects (species, functional group and/or reservoir) were assessed by fitting mixed-effects models (Pinheiro and Bates, 2000) to the data. The general model structure was as follows (Eq. (1)):

$$\ln(y)_{ijk} = \alpha_0 + \alpha_j + \beta_j l_{ij} + a_{jk} + \varepsilon_{ijk} \quad (1)$$

where, α_0 is the global intercept, α_j is group-specific intercepts, β_j is group-specific slopes of individual standardized length (l_i) on THg (y) regressions, a_{jk} is the random effect coefficient (i.e., variance) of the location (k) within-group (j) effect, and ε_{ijk} is the normally distributed error variance. In order to explore further eventual variation embedded in the reservoir variation component of the mixed-model analyses, residual from these models were fitted the reservoir characteristics variables total phosphorous (TP) and age of reservoir to reveal eventual signals of eutrophication and ageing effects. All analyses were performed using the software JMP version 7.0.1 (JMP, 2007).

2 Results

Results on THg in 12 species of fish (n = 235) from six reservoirs in Guizhou Province, China, are presented in Table 2. The over all mean ± SD concentration of THg in fish muscle was (0.066 ± 0.078) μg/g (n = 235). Among the fish species collected, the minimum mean concentration of THg was observed in Tilapia (*Oreochromis mossambicus*), an omnivorous species (Table 2). The maximum mean concentration was observed in the omnivorous species minnow (*Hemiculter bleekeri bleekeri*). Of the total number of samples analyzed, no fish exceeded recommended maximum THg limit for human consumption (0.5 μg/g fish wet weight (ww)) set by the World Health Organization (WHO, 1990) and also by the Standardization Administration of China (SAC, 2005). Only six fish (2.5%) exceeded the maximum THg limit (0.3 μg/g fish ww) set by US Environmental Protection Agency (US EPA, 2001). Of these six fish one catfish, two bighead carp and one common carp were collected from SBX, one chinese minnow was collected from HJD and one Chinese hooksnout carp was collected from WJD reservoir (Fig. 2).

The mixed models fitted to estimate total mercury from individual size and either fish species or functional group, revealed that more than 50% of the explained variation could be attributed to among-location variation (Table 3). Model 1, which explained 58% of the total variation in THg, indicated that most species have differentiated slopes

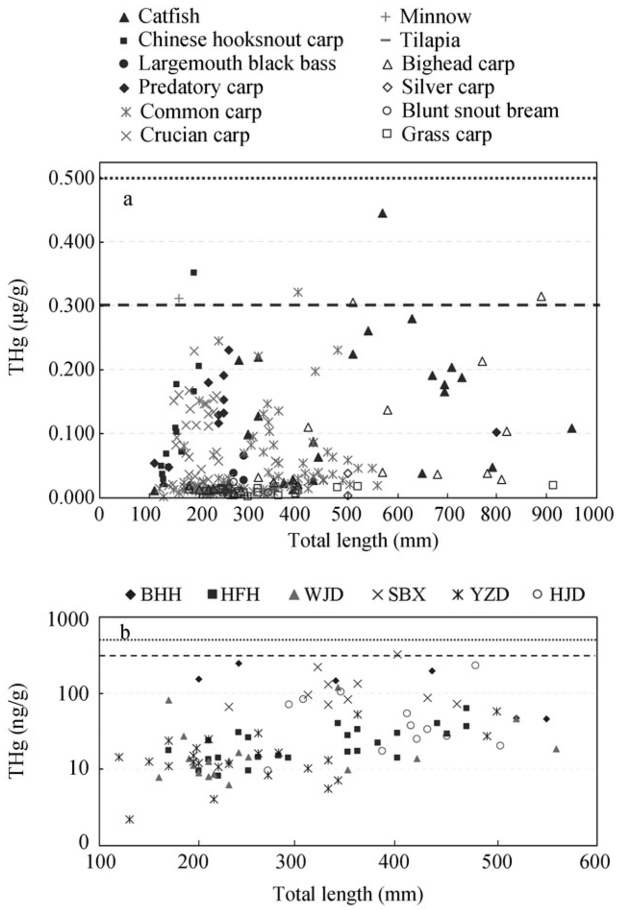

Fig. 2 Individually THg-concentrations in fish muscle plotted against total length. (a) n = 235, all fish; (b) n = 94, common carp. Dotted line shows the maximum limit for human consumption of total mercury (0.5 μg/g) recommended by the World Health Organization (WHO, 1990) and the Standardization Administration of China (SAC, 2005). Dashed line shows the standard limit of total mercury (0.3 μg/g) set by US EPA, based on a total consumption of 17.5 grams of fish per person per day (US EPA, 2001).

and intercepts for the individual size vs. THg regressions. However, when re-scaling the st_length to real scale, the Chinese hooksnout carp and grass carp tended to have higher (3.7 on real scale, remark that Chinese hooksnout carp have low mean ln(length)) and lower (0.4) slope values than the other species, respectively. The same was the case for model 2, which explained 51.1% of the total variation, where the functional group planktivorous seemed to have a higher slope (1.2 on real scale) than the other groups. There was good support in the data for differentiated intercepts among the functional groups in their size vs. THg-regression lines. Post-hoc Tuckey-Kramer HSD tests showed that carnivorous fish have significantly higher THg-concentration than fish belonging to the herbivorous group. The least-square mean (± S.E.) THg-levels, adjusted to individual length of 284 mm, were estimated from the model: carnivorous = 0.088 ± 0.013, omnivorous = 0.040 ± 0.004, planktivorous = 0.036 ± 0.005 and herbivorous = 0.009 ± 0.001. From this the THg mean ratio could be assessed as 9:4:4:1, for the carnivorous, omnivorous, planktivorous and herbivorous fish.

In order to explore further what reservoir attributes that could explain some of the location-level variation included in the model 2, we fitted an ANCOVA model using model 2 residuals as response to a fully factorial model of reservoir age group (old vs. new, Table 1) and TP as predictors. The model explained 27% of the residual variation and showed that for both reservoir age-groups the residual THg-values increased with TP, but significantly more so for old reservoirs ($p_{\text{reservoir age*TP}} < 0.0001$, slope$_{\text{old}}$ = 19.2, slope$_{\text{new}}$ = 5.0).

Another ANCOVA model fitted to explore among-reservoir differences in length-adjusted THg concentrations in common carp revealed that the basins had similar slopes (0.78 ± 0.23, s.e.) but differentiated intercepts (Table 4). The reservoir rank in least square means (μg/g) were as follows (adjusted to st_length = 0, i.e. at mean ln(length)): YZD: 0.0157 ± 0.0005 (A); WJD: 0.0179 ± 0.0007 (BC); HFH: 0.0200 ± 0.0007 (BC); HJD: 0.0325 ± 0.0019 (B); SBX: 0.0957 ± 0.0087 (D). Same letters provided in the parentheses indicate pairs of reservoir that are not significantly different (Tuckey-HSD post hoc contrasts test).

3 Discussion

In general, concentrations of THg in the samples showed rather low levels (Table 2, Fig. 2). No fish exceeded the guidelines (maximum 0.5 μg/g fish ww) of WHO (1990) nor SAC (2005) (Fig. 2). Only six of the 235 fish (2.5%) exceeded the maximum THg limit set by US EPA (0.3 μg/g fish ww) (US EPA, 2001), based on a total consumption of 17.5 grams of fish per person per day. Other studies have observed the same, such as Li et al. (2009) detected mean value 0.063 μg/g (n = 228), where only one fish exceeded the WHO guideline; Chung et al. (2008) reported medianwhole fish of 0.063 μg/g (n = 280) with a range 0.030–1.370 μg/g, where only three fish exceeded the WHO guideline; Qiu et al. (2009) observed the mean 0.29 μg/g (n = 12) with a range 0.061–0.68 μg/g, where only one fish exceeded the WHO guideline.

In this study, fish species were categorized by food preferences into four ecological functional groups (carnivorous, omnivorous, planktivorous and herbivorous fish). The functional groups sort into three levels according to THg, where the carnivores and herbivores were significantly different (Table 3). Omnivorous and planktivorous fish were overlapping between the other two. The mean THg-concentration ratios between the four functional groups were 9:4:4:1 for carnivorous, omnivorous, planktivorous and herbivorous fish. Hg levels are usually higher in muscles of older and larger fish than in those of younger specimens, as a consequence of longer time for bioaccumulation (Dixon and Jones, 1994; Joiris et al., 1995; Da Silva et al., 2005; Mirlean et al., 2005). From the literature we find that fast growing fish have a potential to have lower Hg concentration relative to size through growth dilution (Harris and Bodaly, 1998; Simoneau et al., 2005; Dittman and Driscoll, 2009; Jardine et al., 2009). The Chinese hooksnout carp (*Opsariichthys bidens*) has the steepest

Table 3 Parameter estimates and corresponding effect tests and fit statistics for two mixed models fitted to estimate the natural logarithm of total mercury in fish muscle from individual size and either fish species (Model 1) or functional group (Model 2) accounting for the random effect of location.

Parameter estimates				Effect test					Random effect		
Group	Term/level	Estimate	S.E.	Source	DF	DF-den	F-value	p-value	Term	% of variance	R^2_{adj}
Model 1	Intercept	–3.05	0.16	Species	6	21.7	3.79	0.010	location(species)	58.0	0.77
Species	Bighead carp	–0.16	0.36	St_length	1	178.2	31.42	< 0.0001			
	Catfish	0.63	0.33	Species*st_length	6	186.6	2.38	0.031			
	Chinese hooksnout carp	0.57	0.42								
	Common carp	–0.32	0.30								
	Crucian carp	0.40	0.33								
	Grass carp	–1.64	0.40								
	St_length*	0.40	0.07								
Species	Bighead carp*st_length	0.27	0.18								
	Catfish*st_length	0.19	0.13								
	Chinese hooksnout carp*st_length	0.21	0.17								
	Common carp*st_length	–0.17	0.09								
	Crucian carp*st_length	–0.31	0.15								
	Grass carp*st_length	–0.26	0.19								
Model 2	Intercept	–3.38	0.18	Functional group	3	15.3	5.71	0.008	Location(group)	51.1	0.69
Functional group											
	Piscivores	0.97	0.30	St_length	1	196.7	21.87	< 0.0001			
	Planktivores	0.15	0.33	Group*st_length	3	205.0	5.14	0.002			
	Herbivores	–1.24	0.35								
	St_length	0.35	0.08								
Functional group											
	Piscivores*st_length	0.17	0.10								
	Planktivores*st_length	0.33	0.17								
	Herbivores*st_length	–0.28	0.15								

Random effects are provided as percentage of explained variance. Both models were highly significant ($p < 0.0001$). Metrics for estimation of st_length are as follows. * st_length = ln(length) – mean(ln(length))/s.d.(ln(length)) for each species. Mean(ln(length) ± s.d.(ln(length)): bighead carp, 5.97 ± 0.55; catfish, 6.07 ± 0.50; Chinese hooksnout carp, 5.04 ± 0.16; common carp, 5.67 ± 0.36; crucian carp, 5.27 ± 0.19; grass carp, 5.73 ± 0.33; predatory carp, 5.48 ± 0.51.

Table 4 Parameter estimates and corresponding effect tests and fit statistics for ANCOVA model testing for among-location differences in length-adjusted THg values in common carp.

	Parameter estimates			Effect test				
	Term/level	Estimate	S.E.	Source	DF	DF-den	F-value	p-value
Group	Intercept	–8.00	1.29	ln(length)	1	5.2	11.93	0.0010
	ln(length)	0.78	0.23	Location	4	23.6	13.56	< 0.0001
Location	HFH	–0.34	0.13					
	HJD	0.15	0.17					
	SBX	1.23	0.18					
	WJD	–0.45	0.15					

Model fit statistics: $R^2_{adj} = 0.54$, $F_{5,87} = 19.8$, $p < 0.0001$.

slope between THg and length amongst all the species in our study. Chinese hooksnout carp is a relatively small-sized fish (typically 15–20 cm) feeding on small fishes and insects. It forms large populations in reservoirs because it reproduces quickly and is a strong predator on small-sized fishes stocked in reservoirs (Li and Xu, 1995). Because of relatively low growth potential it should be less sensitive to growth dilution than other carnivorous species in China. As Chinese hooksnout carp is a widespread cyprinid species of eastern Asia, it may be a suitable indicator species for future studies of Hg bioaccumulation processes in Chinese water systems. The fact that this species is not subjected to harvesting makes it even more valuable as an indicator species as it will have a high likelihood of achieving high ages.

Guizhou reservoirs have sub-tropical characteristics and the food-chains seem to be relatively short compared to what we typically find in European and North-American reservoirs. The water quality ranges from oligotrophic-mesotrophic to eutrophic, with chlorophyll-a, total phosphorous and pH ranging 2.25–29.00 mg/m^3, 0.018–0.300 mg/L and 7.00–9.00 respectively (Table 1). When comparing length-adjusted THg levels in common carp from different reservoirs (Table 4), individuals from SBX had significant higher THg than the other reservoirs, while fish from YZD had the lowest (significantly lower than SBX and HJD). Both SBX and YZD are young reservoirs (Table 1) and the observed difference in THg concentrations in fish can hence not be explained by reservoir age alone. Both have high pH (> 7.0), therefore, pH can also not explain the difference in THg concentrations in fish. YZD has Chl-*a* concentration of 3.70 mg/m^3, far below the levels in the old reservoirs (Table 1) (SBX has no available data on Chl-*a*). This may be interpreted as reservoirs with low nutrient input (low TP) may have lower potential for elevated Hg in fish. Indeed, the subsequent residual analysis of model 2 (Table 3) showed that the residual THg (i.e., after accounting for fish size and functional group composition) increased with increasing TP and most pronounced so for old reservoirs. This may again be linked to areas in or around the reservoir having conditions for MeHg production.

According to the water quality reported in Guizhou reservoirs (Table 1), Hg methylation is expected to be lower compared to North-American and European conditions, since the pH is high, DOC concentration low and there are very few wetland areas in the catchments. Recent mass balance studies, however, show that Guizhou reservoirs also are net sources of MeHg (Feng et al., 2009b), although possibly to a smaller extent than typical for North-American and European conditions (Larssen, 2010). Rice paddy fields may be important areas for methylation due to reducing and/or anoxic conditions in the flooded soil during the growth season of the rice (Feng et al., 2008).

In summary, our results show significant differences in THg among different functional groups, assumed to reflect different trophic levels, indicating MeHg accumulation even in the short food chains present here. Since we lack data on MeHg/THg-ratio, it is difficult to assess bioaccumulation processes properly. In addition to MeHg, data on stable isotopes of N and C are also required in order to describe more in detail individual trophic levels and carbon sources in the food webs. Nevertheless, the concentration of Hg in fish in Guizhou reservoirs is in general low. Hg exposure from consumption of this fish should therefore be a minor concern. An important unknown factor, however, is possible future changes in methylation rates and fish concentrations with continued development of hydropower in the province. It has been indicated that the methylation rate may increase with age of the reservoirs in Guizhou (Feng et al., 2009a, 2009b). Since Chinese reservoirs may behave differently with respect to Hg mobilization compared to the much studied cases in North America and Europe, further study to rise the understanding of the bioaccumulation processes in Chinese reservoirs is needed (Larssen, 2010).

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