Toxicity Reference Values and Tissue Residue Criteria for Protecting Avian Wildlife Exposed to Methylmercury in China

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1 Introduction

Mercury (Hg) is a globally distributed environmental contaminant with both natural and anthropogenic sources. Of the forms and oxidation states of Hg, the organic form, methylmercury (MeHg), is the most biologically available and the most toxic (Scheuhammer et al. 2007). MeHg can be neurotoxic, embryotoxic, and can impair physiological function, particularly by disrupting endocrines (Tan et al. 2009) and altering reproductive behavior (Frederick and Jayasena 2010). Because MeHg can be bioaccumulated and biomagnified through the food web, diet is the major pathway by which vertebrates are exposed (Liu et al. 2008). Species occupying higher trophic levels in aquatic systems are considered to be at the greatest exposure risk, particularly birds at trophic levels 4 or 5. Although concentrations of Hg can exist in surface water at or near historical background concentrations, the concentrations of Hg that exist in wildlife are higher (Liu et al. 2008). Chronic dietary exposure to relatively small, environmentally relevant concentrations of MeHg is sufficient to be accumulated by tissues to concentrations that impair reproduction of birds (Frederick and Jayasena 2010).

Environmental contamination by Hg released from human activities is a major concern in China (Feng 2005). Concentrations of Hg from anthropogenic emissions that are greater than the historical and regional background levels (Zheng et al. 2010) are extensively distributed, and have been detected in surface waters and tissues of birds (Feng 2005; He et al. 2010; Jin et al. 2006; Zhu et al. 2012). However, no specific guidelines, standards, or criteria have been established for the risk that MeHg may pose to wildlife in China. Assessing the risk that MeHg poses to birds in Chinese aquatic systems is urgently needed to support national policy-making decisions. Thus, derivation of Hg wildlife criteria values that apply to the aquatic systems characteristic of China is a primary task of aquatic environmental managers.

Recently, using the tissue residue approach has been recommended for assessing the ecological risk of bioaccumulative contaminants (Sappington et al. 2011; Beckvar et al. 2005; Newsted et al. 2005). Because wildlife regularly consume fish, toxicity reference values (TRVs) that are based on fish tissue concentrations have been developed for 2,3,7,8-tetrachlodibenzo-*p*-dioxin (TCDD), polychlorinated biphenyls (PCB), perfluorooctane sulfonate (PFOS), and cadmium (Cd) (Newsted et al. 2005; Kannan et al. 2000; Blankenship et al. 2008; Stanton et al. 2010). Moreover, concentrations of contaminants in tissues of wildlife, such as blood and feathers (Kahle and Becker 1999; Herring et al. 2009), have been used as exposure indices for risk assessments. Using the cumulatively ingested dose of a chemical from consuming contaminated food (e.g., tissues) is more accurate in that it accounts for bioaccumulation and bioavailability. A direct relationship between toxicity and the consumed (internal) dose can be either measured or predicted (Sappington et al. 2011). Exposure to the internal dose can be expressed on a tissue-specific or whole-body basis. The internal dose can be either measured or predicted from key ratios, such as bioconcentration, -accumulation, and -magnification factors (BCF, BAF, and BMF), respectively, from trophic magnification factors (TMF), or from more

complex pharmacokinetic models. Therefore, TRVs and tissue residue criteria (TRCs) that are based on concentrations of toxicants in tissues are effective for protecting wildlife from the hazards of exposure to pollutants. To illustrate, the critical blood concentration of lead (Pb) has been derived for wildlife by using the tissue residue approach (Buekers et al. 2009), and TRVs were derived for PFOS in avian tissues (e.g., serum and liver) and eggs (Newsted et al. 2005).

Establishing regional criteria is preferred, because species composition, bioaccumulation rates and wildlife diets vary among locations. Canada and the USA have established criteria for assessing potential adverse effects on wildlife from exposure to MeHg. The Canadian Council of Ministers of Environment (CCME) derived wildlife guidelines that were based on concentrations of MeHg in fishes, in which the body mass (bm) and rate of food ingestion by Wilson's storm petrel (*Oceanites oceanicus*) were incorporated (CCME 2000). The U.S. Environmental Protection Agency (US EPA) developed criteria for protecting wildlife that were based on concentrations of MeHg in water, in which the body mass, rate of food ingestion, and BMF for three representative bird species endemic to the North American Great Lakes were used (US EPA 1995b). In addition, by using the Great Lakes Water Quality Initiative (GLWQI), the US EPA developed a set of application factors to account for uncertainties. In these previously developed guidelines and criteria (US EPA 1995b; CCME 2000), the effects of MeHg on reproduction of mallards were used as the critical basis for deriving criteria.

Recently, several new studies of the toxicity of MeHg to birds have become available. The results of these studies suggest that the mallard is not the most sensitive avian species to the effects of MeHg (Heinz et al. 2009, 2010a), and thus may not be representative or protective of other species. Therefore, it was deemed desirable to update the TRV values to reflect the effects of MeHg on birds that consume aquatic biota. Such an update can then be applied to representative species endemic to China. The latest research on the toxicity of MeHg to birds was reviewed, and thresholds of toxicity were derived that were based on concentrations of MeHg in the diet (fish) and on concentrations of MeHg in tissues of birds. Finally, estimates of TRV and TRC values to protect birds in Chinese aquatic systems from the effects of MeHg were developed.

2 Data Collection and Analysis

2.1 Selection of Representative Species in China

According to viewpoints expressed in the technical support document for the GLWQI, the primary basis for selecting representative avian species is exposure to contaminants through aquatic food chains, such as fish-consuming species. The species that experience the greatest exposure are favored as representative avian species (US EPA 1995c). Three species that commonly inhabit Chinese aquatic ecosystems

Table 1 Body mass and rate of food ingestion for three representative species

Species and life history parameters	Value	References
Night heron (*Nycticorax nycticorax*)		
Body mass (kg)	0.706[a, b]	Dunning (1993)
Rate of ingestion (kg/day)	0.239[c]	
Little egret (*Egretta garzetta*)		
Body mass (kg)	0.342[a]	Zamani-Ahmadmahmoodi et al. (2010); Fujita (2003); Zhang and Liu (1991)
Rate of ingestion (kg/day)	0.148[c]	
Eurasian spoonbill (*Platalea leucorodia*)		
Body mass (kg)	2.232	Liu et al. (2003)
Rate of ingestion (kg/day)	0.514[c]	

[a]Geometric mean of the data from different studies
[b]Geometric mean of the values reported (Dunning 1993) and the data from China Digital Science and Technology Museum (http://amuseum.cdstm.cn/AMuseum/dongwu/page/animal_detail_4683.html)
[c]Calculated from the allometric equation (Nagy 2001): $FI = 3.048 \times BW^{0.665}$

were selected as representative species in China. These three species were the night heron (*Nycticorax nycticorax*), little egret (*Egretta garzetta*), and Eurasian spoonbill (*Platalea leucorodia*), all of which consume aquatic prey. These three species are widely distributed in Chinese aquatic ecosystems (Barter et al. 2005), and each has been studied extensively as indicators of environmental pollution and wetlands' health (Zamani-Ahmadmahmoodi et al. 2010; Burger and Gochfeld 1997; Zhang et al. 2006; An et al. 2006). The night heron and Eurasian spoonbill are species regarded as second-grade state-protected animals in China. The little egret and Eurasian spoonbill are species listed in the Convention on International Trade in Endangered Species of Wild Fauna and Flora. Body mass and rates of food ingestion for these three species are summarized in Table 1.

2.2 Selection of Toxicity Data

Information on effects of MeHg on birds has been summarized by the US EPA (US EPA 1995b). Toxicity threshold values for MeHg, expressed as no observed adverse effects levels (NOAEL) or lowest observed adverse effects levels (LOAEL), were derived from several endpoints, and were determined for avian wildlife based on concentrations ingested by eating fish and bird tissues. Dietary-based data were converted to average daily intake (ADI) values and were expressed as units of μg MeHg/g body mass/day (μg MeHg/g (bm)/day). ADI values were calculated from body masses and rates of ingestion by the selected surrogate birds. When rates of food ingestion were not reported in a paper, they were calculated by using the most recent allometric equations (Nagy 2001).

The principles used as the basis for selecting utilizable NOAEL or LOAEL values were as follows (CCME 1998): (1) the study retained suitable control conditions; (2) the study was designed to consider ecologically relevant endpoints, such as reproduction, embryonic development, offspring or survival of adults (F_0), growth,

and other responses; (3) a clear dose–response relationship was demonstrated in the study; (4) the form and dosage of test chemical were reported; (5) the tested chemical was administered via the oral, rather than by other routes (i.e., only the oral route is natural for wildlife in the field); and (6) studies that included only acute exposures were not accepted, because they provided no data on chronic, sublethal effects on wildlife.

2.3 Methods of Deriving TRVs and TRCs

Two methods were used to derive TRVs from dietary or tissue concentrations. These were the critical study approach (CSA) and the species sensitivity distribution (SSD) approach. TRVs that were based on dietary exposure are expressed as daily dietary intake (μg MeHg/g (bm)/day). TRVs that were based on dietary exposure were converted to the corresponding dietary-based TRC values by using body masses and rates of food ingestion by the three representative surrogate species. The TRVs that were based on concentrations of MeHg in tissues of birds do not vary among representative species as a function of body mass and rate of ingestion.

CSA. CSA is the primary method for assessing risk to wildlife and for deriving criteria for protection of wildlife (CCME 1998; US EPA 1995a, b, 2003, 2005; Sample and Suter 1993). This method is used to select the critical study for deriving recommended TRVs, which involves finding a technically defensible, definitive study from which a toxicity threshold is bracketed by experimental doses (Blankenship et al. 2008; US EPA 2003). A series of uncertainty factors (UFs) are applied to LOAEL or NOAEL values that are obtained from the critical study, and these are used to determine the TRVs. A UF is assigned from guidance given in the Technical Support Document (TSD) for Wildlife Criteria for the GLWQI (US EPA 1995c), and in the GLWQI Criteria Documents for the Protection of Wildlife (US EPA 1995b). Three sources of uncertainty are considered in assigning a UF value: (1) interspecies differences in toxicological sensitivity (UF_A), (2) subchronic to chronic extrapolations (UF_S), and (3) LOAEL to NOAEL extrapolations (UF_L). Application factors for each source of uncertainty were assigned values between 1 and 10, based on available information and professional judgment (US EPA 1995c; Newsted et al. 2005).

SSD. SSD is a statistical distribution representing the variation in sensitivity of species to a contaminant, and can be developed by a statistical or empirical distribution function of response for a sample of species (Posthuma et al. 2002). This method has been used to assess risks to aquatic organisms and for deriving water quality criteria (WQC) for protecting aquatic species (Caldwell et al. 2008; Hall et al. 1998; Solomon et al. 1996; Stephan et al. 1985). However, because data on the toxicity of contaminants to wildlife are often insufficient, the SSD approach to assess wildlife risks has not often been applied. For the analysis presented herein, the SSD was used to determine the concentration of MeHg that would be protective of wildlife. This concentration is the fifth centile (HC_5) of the SSD generated from selected wildlife effects data for MeHg. Data representing the most sensitive endpoint for

each species were selected to construct the SSD. If the duration of exposure was deemed to be insufficient, or if only an unbounded LOAEL was produced in some studies, the data were corrected before fitting the SSD function, by using UF_S or UF_L (US EPA 2005). The basic assumption of the SSD approach is that sensitivity among species can be described by using a specified statistical distribution, such as the normal (Wagner and Løkke 1991; Aldenberg and Jaworska 2000), logistic (Kooijman 1987; Aldenberg and Slob 1993), triangular (Stephan et al. 1985), or Weibull (Caldwell et al. 2008) probability functions, or by using distribution-free, nonparametric methods (Ling 2004; Newman et al. 2000). It was assumed that the toxicity data selected for MeHg are skewed and can be described by using a log-normal distribution (Aldenberg and Jaworska 2000). The ETX2.0 program was used to fit the distribution (Van Vlaardingen et al. 2004). The HC_5 and its two-sided 90% confidence limits, designated as lower limit (LL HC_5) and upper limit (UL HC_5), were calculated. The goodness of fit was tested with the Anderson-Darling and Kolmogorov-Smirnov tests to ensure that the data were log-normally distributed.

3 Review of MeHg Toxicity to Birds

The results of subchronic and chronic toxicity testing on the mallard (*Anas platyrhynchos*), white leghorn chicken (*Gallus domesticus*), ring-necked pheasant (*Phasianus colchicus*), Japanese quail (*Coturnix japonica*), red-tailed hawk (*Buteo jamaicensis*), and zebra finch (*Poephila quttata*) have been summarized in the GLWQI Criteria Documents for the Protection of Wildlife (US EPA 1995b). No additional toxicity information was available for these species except for mallard. Thus, the toxicity threshold concentrations from diet and tissue data for the other five species were used directly (Table 2). Below, a review of the recent relevant toxicity studies is presented.

Mallard (*A. platyrhynchos*). As stated in the GLWQI Criteria Documents (US EPA 1995b), the dietary LOAEL and NOAEL values for mortality and for neurological effects on the mallard were 3.0 and 0.5 μg MeHg/g feed (dry weight, dwt), respectively (Heinz and Locke 1976). Using the average body mass of 1 kg for a mallard (Delnicki and Reinecke 1986), a rate of food ingestion of 0.051 kg dried feed/kg fresh (bm)/day was derived from Nagy's (Nagy 2001) allometric equation for omnivorous birds, rather than the equation (Nagy 1987) that was used in the GLWQI Criteria Documents for Protection of Wildlife. Assuming that the laboratory feed for the mallard consists of 10% water (US EPA 1995b), the rate of food ingestion would be equivalent to 0.057 kg food (wwt)/kg (bm)/day. Corresponding values for LOAEL and NOAEL were calculated to be 0.171 and 0.029 μg MeHg/g (bm)/day. The LOAEL and NOAEL values, expressed as concentrations of MeHg in egg, were, respectively, 0.79 and 5.64 μg MeHg/g (wwt) (Heinz and Locke 1976). Based on multigenerational effects on reproduction, when converted from the dietary values by the use of rates of consumption of food of 0.156 kg food (wwt)/kg (bm)/day (US EPA 1995b), the dietary LOAEL value was 0.078 μg MeHg/g (bm)/day.

Table 2 Summary of subchronic and chronic avian toxicity data for MeHg (μg MeHg/g (bm)/day for ADI; μg MeHg/g (wwt) for diet or tissues)

Species	Media	Toxic effects observed	NOAEL	LOAEL	References
Mallard	Diet	Offspring mortality and neurotoxicity	0.5	3.0	Heinz (1974, 1975, 1976a, b, 1979)
	ADI	Offspring mortality and neurotoxicity	0.029	0.171	
	Egg	Offspring mortality and neurotoxicity	0.79	5.64	
	Diet	Multigenerational exposure, reproduction		0.5	
	ADI	Multigenerational exposure, reproduction		0.078	
	Egg	Multigenerational exposure, reproduction		0.83	
	Liver	Multigenerational exposure, reproduction		1.29	
	Kidney	Multigenerational exposure, reproduction		1.64	
	Breast muscle	Multigenerational exposure, reproduction		0.77	
	Brain	Multigenerational exposure, reproduction		0.55	
	Ovary	Multigenerational exposure, reproduction		0.58	
	Primary feather	Multigenerational exposure, reproduction		9.71	
	Diet	Reproduction, duckling survival	2	4	Heinz et al. (2010a)
	ADI	Reproduction, duckling survival	0.114	0.228	
	Egg	Reproduction, duckling survival	3.7	5.9	
Great egret	Diet	Behavior, growth, biochemistry		0.5	Bouton et al. (1999); Hoffman et al. (2005); Spalding et al. (2000a); Spalding et al. (2000b)
	ADI	Behavior, growth, biochemistry		0.091	
	Blood	Behavior, growth, biochemistry		10.3	
	Liver	Behavior, growth, biochemistry		15.1	
	Brain	Behavior, growth, biochemistry		3.4	
	Kidney	Behavior, growth, biochemistry		8.1	

(continued)

Table 2 (continued)

Species	Media	Toxic effects observed	NOAEL	LOAEL	References
Common loon	Diet	Growth	1.5		Kenow et al. (2003)
	ADI	Growth	0.27		
	Blood	Growth	3.33 μg/mL		
	Diet	Immune function, biochemistry	0.08	0.4	Kenow et al. (2008); Kenow et al. (2007a); Kenow et al. (2007b)
	ADI	Immune function, biochemistry	0.014	0.072	
	Blood	Immune function, biochemistry		1.98	
	Brain	Immune function, biochemistry		0.88	
	Kidney	Immune function, biochemistry		2.29	
	Breast muscle	Immune function, biochemistry		1.23	
	Liver	Immune function, biochemistry		4.03	
	Feather	Immune function, biochemistry		22.03	
American kestrel	Diet	Reproduction		0.3	Albers et al. (2007)
	ADI	Reproduction		0.055	
	Egg	Reproduction		2.00[a]	
	Diet	Immune function		0.23	Fallacara et al. (2011)
	ADI	Immune function		0.043	
	Blood	Immune function		6.12[a]	
	Spleen	Immune function		5.16[a]	
White ibis	Diet	Reproduction		0.05	Frederick and Jayasena (2010)
	ADI	Reproduction		0.010	
	Feather	Reproduction		6.32[a]	
	Blood	Reproduction		0.73[a]	
White leghorn chicken	ADI	Growth		0.29	Fimreite (1970)
	Liver	Growth		3.49	
	ADI	Mortality	0.57	0.86	
	Liver	Mortality	7.25	10.00	
	ADI	Reproduction		0.67	Scott (1977)

Ring-necked pheasant	ADI	Mortality	0.25	0.75	Spann et al. (1972)
	ADI	Reproduction		0.25	
	ADI	Reproduction		0.093	Fimreite (1971)
Japanese quail	ADI	Offspring mortality	0.26	0.52	Eskeland and Nafstad (1978)
Red-tailed hawk	ADI	Neurological effect and mortality	0.49	1.2	Fimreite and Karstad (1971)
Zebra finch	ADI	Neurological effect and mortality	0.88	1.75	Scheuhammer (1988)
	Liver	Neurological effect and mortality		30.5–73	
	Kidney	Neurological effect and mortality		35.5–65	
	Brain	Neurological effect and mortality	7–11	14.1–20	

[a]Total mercury

NOAEL no observed adverse effects level, *LOAEL* lowest observed adverse effects level, *ADI* average daily intake, *bm* body mass, *wwt* wet weight

The corresponding concentrations of MeHg in tissues such as liver, kidney, breast muscle, brain, ovary, and primary feathers, and in eggs are given (geometric mean for three generations) for mallards (Heinz 1979; Table 2).

Reproductive effects of MeHg on mallards were investigated by exposing adults to one of four doses of MeHg (1, 2, 4, or 8 μg MeHg/g (dwt)) (Heinz et al. 2010a). No adverse effects were observed in adults, or on egg fertility or the rate of hatching success. However, at doses of 4 or 8 μg MeHg/g (dwt), survival of ducklings and the number of ducklings produced per female were less than those of untreated controls. Ducklings at 6 days of age from parents fed 4 or 8 μg MeHg/g (dwt) weighed less than the controls. Thus, doses of 2 and 4 μg MeHg/g (dwt) were considered to be the dietary-based NOAEL and LOAEL values, respectively. However, both doses were greater than the LOAEL of 0.5 μg MeHg/g (dwt) (Heinz 1979), which was determined from the results of a multigeneration exposure. The corresponding daily doses, which were converted from the dietary values using a rate of ingestion of food of 0.057 kg/kg (bm)/day, were 0.114 and 0.228 μg MeHg/g (bm)/day, respectively. The NOAEL and LOAEL for Hg concentrations in eggs are provided in Table 2.

When lesser doses of MeHg were injected into mallard eggs, hormesis was observed at 0.05 μg MeHg/g (wwt) (least dose) (Heinz et al. 2011), which agrees with a similar observation, in which mallards were exposed to a single dose of 0.5 μg MeHg/g (dwt) (Heinz et al. 2010b). However, the mean concentration of MeHg in eggs of hens fed MeHg in the diet was 0.81 μg MeHg/g (wwt) (Heinz et al. 2010b), a value greater than the 0.05 μg/g (wwt) of Hg that was injected. The exposure pathway was regarded to be the major reason for the difference. It was suggested that when MeHg was injected into embryos it was more toxic than equivalent concentrations of maternally deposited MeHg (Heinz et al. 2009, 2011). Although the mechanism for this discrepancy is not well understood, it is probably due to a difference in dose vs. dose rate, or from the binding of MeHg, which results from differences in biological activity between the two vectors of exposure. The results of a single dose MeHg exposure (0.5 μg MeHg/g (dwt)) (Heinz et al. 2010b) seemed to contradict the results of the multigeneration, dietary exposure in which the same dose was used. This might result from differences between forms of MeHg and the sources of mallards tested (Heinz et al. 2010b). Thus, additional research is needed on "low-dose" effects of MeHg to the mallard. According to the analysis above, the results from the multigeneration exposure, rather than those from the latest studies (Heinz et al. 2010a, b), should be used as the basis for developing TRVs and TRCs.

Great egret (*Ardea alba*). Several different effects, including behavior (Bouton et al. 1999), survival, growth and accumulation in tissues (Spalding et al. 2000a), histology, neurology, and immunology (Spalding et al. 2000b) from exposure to MeHg, were addressed in three sladies, in which juvenile great egrets were exposed to two dietary doses of 0.5 or 5 μg MeHg/g (wwt) for 12 weeks. Severe ataxia was observed in individuals fed the greater dose, whereas the lesser dose produced effects on activity, a tendency to seek shade, and motivation to hunt prey (Bouton et al. 1999). After 9 weeks of exposure, appetite and mass declined significantly in both dosed groups (Spalding et al. 2000a). Adverse effects, related to immune function, were observed in individuals fed the lesser dose, whereas individuals fed the greater dose exhibited adverse effects on tissues related to immune and nerve functions

(Spalding et al. 2000b). In a study of the biochemical effects of MeHg (Hoffman et al. 2005), only activities of the enzyme aspartate aminotransferase (AST) in blood plasma, and thiobarbituric acid-reactive substances (TBARS) in liver of individuals fed the lesser dose were significantly greater than those of the control group. However, there were significant changes in activities of glutathione peroxidase (GSH-Px), aspartate aminotransferase (AST), lactate dehydrogenase, and in the concentrations of uric acid, total protein, and inorganic phosphorus in tissues of individuals fed the greater dose. Therefore, on the basis of effects observed, the dietary LOAEL value for the great egret was determined to be 0.5 μg MeHg/g (wwt). The corresponding LOAEL values were based on concentrations in plasma, liver, brain, and kidney, and are presented in Table 2. Using an average bm of 1.0 kg for great egret reported by Rumbold et al. (2008), and a rate of intake of food of 0.181 kg/day estimated from an allometric equation for wading birds (Kushlan 1978; US EPA 1993), the dietary LOAEL was calculated to be 0.091 μg MeHg/g (bm)/day.

Common loon (*Gavia immer*). The effects of MeHg on growth (Kenow et al. 2003), behavior (Kenow et al. 2010), immune function (Kenow et al. 2007a), and the biochemical index (Kenow et al. 2008) of juvenile common loons were investigated during which individuals were exposed for 15 weeks. Neither adverse effects on growth or survival occurred in loons that were exposed to three doses, 0.1, 0.5, or 1.5 μg MeHg/g (wwt) (Kenow et al. 2003), nor behavioral effects at doses of 0.08, 0.4, or 1.2 μg MeHg/g (wwt) (Kenow et al. 2010). However, adverse effects on immune function (Kenow et al. 2007a) and effects related to oxidative stress and altered glutathione metabolism (Kenow et al. 2008) occurred at 0.4 and 1.2 μg MeHg/g (wwt). Using a bm of 4.67 kg for common loon adults (Barr 1996; Dunning 1993), a food intake rate of 0.839 kg/day was derived from the allometric equation for carnivorous birds (Nagy 2001). Thus, the dietary-based NOAEL value, based on growth effects of the common loon, was 0.27 μg MeHg/g (bm)/day (1.5 μg MeHg/g (wwt)). The NOAEL and LOAEL values were based on dietary exposure and effects on immune function and biochemistry, and were, respectively, 0.014 μg MeHg/g (bm)/day (0.08 μg MeHg/g (wwt)) and 0.072 μg MeHg/g (bm)/day (0.4 μg MeHg/g (wwt)). The corresponding concentration of MeHg in blood that was associated with growth effects was 3.33 μg MeHg/mL (Kenow et al. 2003). In Table 2, we show the LOAEL values for blood, brain, kidney, breast muscle, liver, and feathers (geometric mean of Hg in feather at different sites); these were based on immune and biochemical-function effects. The corresponding NOAEL values were not available (Kenow et al. 2007b).

White ibis (*Eudocimus albus*). Juvenile white ibises were exposed to dietary MeHg at three doses of 0.05, 0.1, or 0.3 μg MeHg/g (wwt), and their foraging behavior and efficiency (Adams and Frederick 2008), survival (Frederick et al. 2011), and breeding behavior (Frederick and Jayasena 2010) were examined. Hormetic effects were observed on foraging efficiency at doses of 0.05 and 0.1 μg MeHg/g (wwt), and the effects from exposure to the 0.3 μg MeHg/g (wwt) group were similar to that of the controls. However, no clear dose–response relationship was demonstrated in the Adams and Frederick (2008) study. Therefore, we did not further utilize this study for deriving TRVs. Exposure to MeHg at these three doses did not affect the survival of white ibises (Frederick et al. 2011). The effect on breeding behavior was investigated

by exposing white ibises to these three doses of MeHg over a 3-year period (Frederick and Jayasena 2010). The effects produced were increases in male–male pairing behavior, dose-related reductions in key courtship behaviors for males, and fewer eggs laid (Frederick and Jayasena 2010). In addition, productivity per nest by heterosexual males was significantly less than that of controls. These reproductive effects could be related to changes in endocrine function, because concentrations of estradiol and testosterone were altered in birds exposed to all three doses of MeHg (0.05, 0.1 and 0.3 μg MeHg/g (wwt)) (Jayasena et al. 2011). In another study, endocrine function of white ibises was affected by relatively small concentrations of MeHg. These small exposures might have changed reproductive behavior and altered mate choice in males, and may have reduced reproductive success so as to finally influence population numbers. Studies of wild birds (Heath et al. 2005) also suggest that exposure to Hg may result in fewer birds nesting or more nest abandonment from subacute effects on hormone systems. From these studies, the dietary LOAEL value for a reproductive endpoint in white ibises exposed to MeHg was established to be 0.05 μg MeHg/g (wwt), and the corresponding Hg tissue residue concentrations were 6.32 μg total Hg (THg)/g (wwt) (geometric mean of values during 2006 and 2008) in blood, and 0.73 μg THg/g (wwt) in feathers (Frederick and Jayasena 2010). Concentrations of THg in blood and feathers were derived primarily from MeHg food residue information. Using a white ibis bm of 0.869 kg (geometric mean) (Dunning 1993), and a food ingestion rate of 0.182 kg/day (21% of bm) (Kushlan 1977), the corresponding LOAEL was derived to be 0.010 μg MeHg/g (bm)/day.

American Kestrel (*Falco sparverius*). American kestrels were exposed to 0.3, 0.7, 1.2, 1.7, or 2.2 μg MeHg/g (wwt), and parameters of their reproduction were measured (Albers et al. 2007). Adverse effects on reproduction were observed in birds at all doses. At a dose of 0.3 μg MeHg/g (wwt), the number of fledglings and the percent of nestlings fledged were lesser than that of controls and were lesser yet at the greater doses (i.e., 0.7 or 1.2 μg MeHg/g (wwt)). Total failure of fledging was observed at 1.7 μg MeHg/g (wwt). Thus, 0.3 μg MeHg/g (wwt) was considered to be a LOAEL for reproductive effects in American kestrels. The LOAEL, based on concentrations in eggs, was 2.00 μg MeHg/g (wwt). Using a bm of 0.119 kg and a food ingestion rate of 0.022 kg/day (geometric means) (Dunning 1993; Yáñez et al. 1980; US EPA 1993), a dietary-based LOAEL value of 0.055 μg MeHg/g (bm)/day was calculated.

Effects of MeHg on immune function and hematology were determined for adult male American kestrels exposed to 0.23 or 1.5 μg MeHg/g (wwt) for a period of 13 weeks (Fallacara et al. 2011). Suppression of immune function occurred at both doses. Adult kestrels were more sensitive to the effects of MeHg for immune function than for reproduction. The 0.23 μg MeHg/g (wwt) dose was assigned as the dietary LOAEL value for immunotoxic effects. Using bm and rate of ingestion of food, this LOAEL was converted to an ADI of 0.043 μg MeHg/g (bm)/day. The corresponding concentrations of total Hg in blood and spleen after exposure are shown in Table 2. The toxic effects observed in the foregoing studies for birds are summarized in Table 2, and the LOAEL and NOAEL values, expressed as ADI, are shown in Fig. 1. Reproduction, immune-function, and effects on biochemistry were more sensitive to MeHg exposure than were behavior, mortality, and neuropathology effects. The reproductive, immune

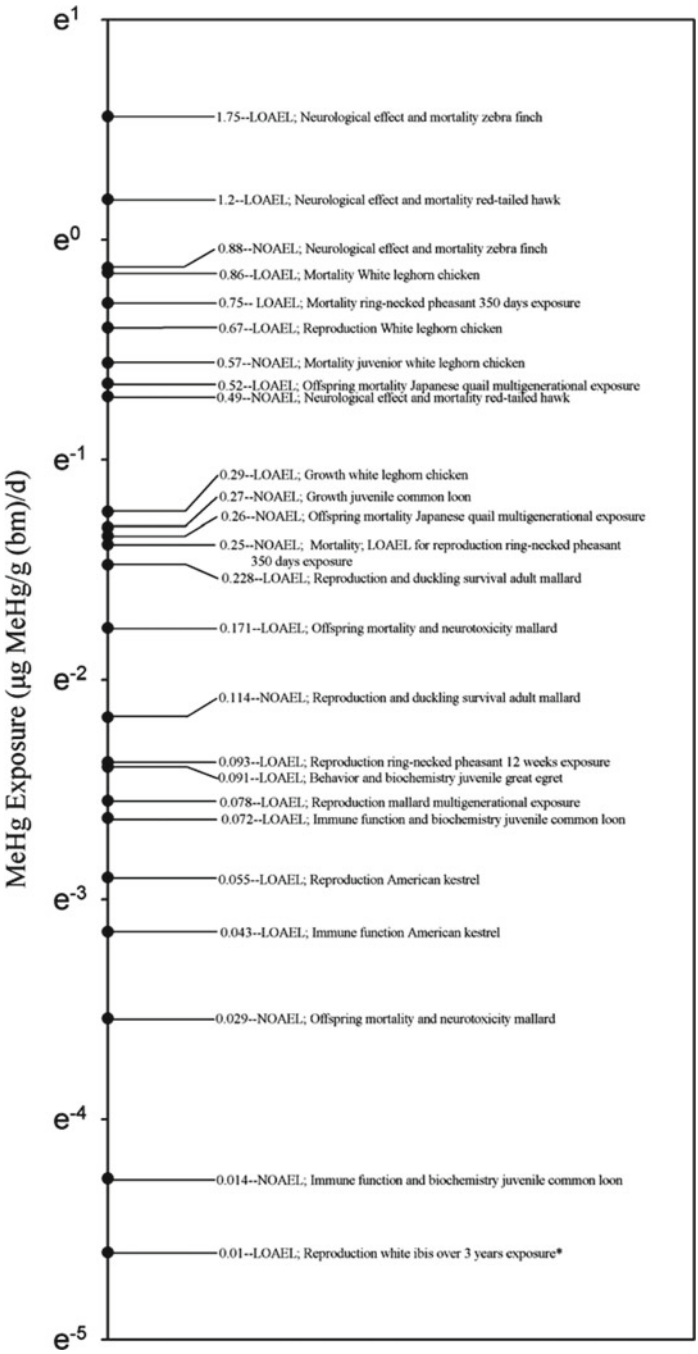

Fig. 1 Toxicity thresholds for avian species dietary exposure to MeHg expressed as the average daily intake (ADI). The *asterisk* indicates that the data are critical for deriving the criteria values. *NOAEL* no observed adverse effects level, *LOAEL* lowest observed adverse effects level, *bm* body mass. See Table 2 for data set

function, and biochemistry effects occurred over a dose range of 0.01–0.67 μg MeHg/g (bm)/day. The threshold for other adverse effects was 0.029 μg MeHg/g (bm)/day. White ibises, juvenile common loons, mallards, and American kestrels are species that are relatively more sensitive than other species. Although all effects were concentration dependent, the endpoint that was most biologically relevant and sensitivity to it varied among bird species. The most sensitive endpoint effect for MeHg exposure in any species was reproductive productivity of the white ibis (Frederick and Jayasena 2010).

4 Derivations of TRVs and TRCs

CSA. As indicated above, the study on the reproductive effects of MeHg in white ibises (Frederick and Jayasena 2010) was the most appropriate critical study, and the results of that study were used for deriving the TRV and TRC. Both juveniles and adults were exposed over 3 years and three breeding seasons in the critical study. LOAEL values were based on reproductive effects in white ibis, and in particular on male–male pairing behavior and on fewer eggs laid. The LOAEL values were expressed as both dietary (ADI) and residue concentrations in feathers and blood (viz., respectively, 0.010 μg MeHg/g (bm)/day, 6.32 μg THg/g (wwt), and 0.73 μg THg/g (wwt)). Based on reproductive effects of MeHg on white ibises and on characteristics of avian predators, three uncertainty factors were considered as follows: (1) an interspecies uncertainty factor, (2) a LOAEL to NOAEL uncertainty factor, and (3) a subchronic to chronic uncertainty factor. An overall uncertainty factor of 2 was assigned to account for data gaps (US EPA 1995b, c; Table 3).

Table 3 Assignment of uncertainty factors for derivation of avian wildlife toxicity reference values (TRVs) for MeHg

Uncertainty factors	Notes
Interspecies uncertainty factor (UF_A)	The data selected to determine avian toxicity reference values were from a reproductive study of white ibis, which is a piscivorous wading bird in the same order with the three representative birds. Because the white ibis is the most sensitive of the species reviewed in the present study, $UF_A = 1.0$
LOAEL to NOAEL (UF_L)	The data from the critical study is a LOAEL based on a reproduction endpoint in a multiple year exposure, but not a NOAEL. However, the difference between the LOAEL and control was only 13.2% for the decreases in egg productivity. Taken together with the ratio of LOAEL and NOAEL in other studies, the $UF_L = 2.0$
Subchronic to chronic uncertainty (UF_S)	The critical study was conducted over 3 years, covered juvenile stage and 3 breeding seasons, evaluated reproductive behavior and productivity, which are considered ecologically most relevant, thus $UF_S = 1.0$
Overall uncertainty factor (UF) for TRVs	$UF = 1 \times 2 \times 1 = 2$

NOAEL no observed adverse effects level, *LOAEL* lowest observed adverse effects level

Table 4 MeHg toxicity reference values (TRVs) and tissue residue criteria (TRCs) for representative avian species based on average daily intake (ADI) or diet, feathers, and blood

	LOAEL	TRV	TRC
ADI, ng MeHg/g (wwt) for TRC, ng MeHg/g (bm)/day for LOAEL and TRV	10	5.0	15.47
Feather, μg THg/g (wwt)	6.32	3.16	3.16
Blood, μg THg/g (wwt)	0.73	0.365	0.365

LOAEL lowest observed adverse effects level, *bm* body mass, *wwt* wet weight

Table 5 Toxicity thresholds for species sensitivity distribution curve fitting (μg MeHg/g (bm)/day)

Species	Exposure period	Reported value		Converted NOAEL	References
		LOAEL	NOAEL		
Mallard	3 years	0.078		0.039[a]	Heinz (1974, 1975, 1976a, b, 1979)
Great egret	12 weeks	0.091		0.0455[a]	Bouton et al. (1999), Hoffman et al. (2005), Spalding et al. (2000a), Spalding et al. (2000b)
Common loon	15 weeks		0.014	0.014	Kenow et al. (2008), Kenow et al. (2007a), Kenow et al. (2007b)
American kestrel	13 weeks	0.043		0.0215[a]	Fallacara et al. (2011)
White ibis	3 years	0.01		0.005[a]	Frederick and Jayasena (2010)
White leghorn chicken	21 days	0.29		0.0145[a, b]	Fimreite (1970)
Ring-necked pheasant	12 weeks	0.093		0.0465[a]	Fimreite (1971)
Japanese quail	6 weeks		0.26	0.26	Eskeland and Nafstad (1978)
Red-tailed hawk	12 weeks		0.49	0.49	Fimreite and Karstad (1971)
Zebra finch	76 days		0.88	0.88	Scheuhammer (1988)

[a]The value derived by applying a LOAEL to NOAEL factor of 2.0

[b]The value derived by applying a factor of 10 to account for the uncertainty in establishing a NOAEL from short time (<1 month) (Jongbloed et al. 1996)

NOAEL no observed adverse effects level, *LOAEL* lowest observed adverse effects level, *bm* body mass

TRVs based on ADI and on concentrations of MeHg in feathers and blood were 5.0 ng MeHg/g (bm)/day, 3.16 μg THg/g (wwt), and 0.365 μg THg/g (wwt), respectively (Table 4). Based on the dietary exposure of the representative avian species, the TRC values for the night heron, little egret, and Eurasian spoonbill were 14.77, 11.55, and 21.71 ng MeHg/g (wwt), respectively; the geometric mean was 15.47 ng MeHg/g (wwt). TRCs, based on concentrations of THg in feathers and blood were the same as the corresponding TRVs (Table 4).

SSD. NOAEL and LOAEL values selected for constructing the SSD were corrected by applying a LOAEL to NOAEL factor, or a subchronic to chronic factor (Table 5). On the basis of the function describing the toxicity thresholds of all species (Table 5),

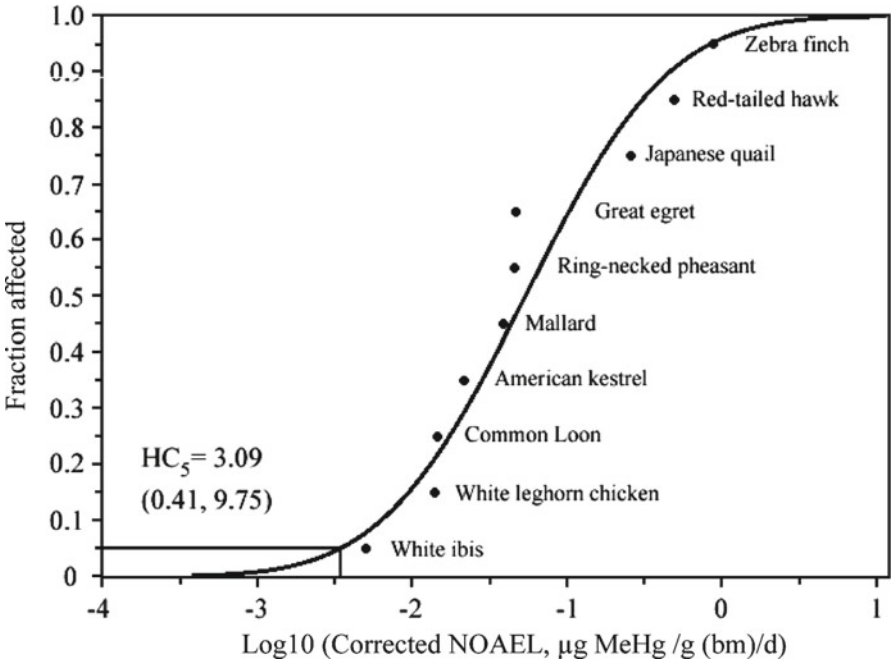

Fig. 2 The distribution of species sensitivity for the avian toxicity data of MeHg. Fifth centile of species sensitivity distribution (HC_5, ng MeHg/g (bm)/day) is presented with two-sided 90% confidence limits given in *parenthesis*. The goodness-of-fit test by the use of the Anderson-Darling and Kolmogorov-Smirnov tests was accepted. The *black dots* represent the toxicity data, and the *solid curve* represents the log-normal distribution. *NOAEL* no observed adverse effects level, *bm* body mass. See Table 5 for data set

the HC_5 was predicted to be 3.09 ng MeHg/g (bm)/day, which was defined as the dietary-based TRV (Fig. 2). Based on dietary exposure, TRCs for the night heron, little egret, and Eurasian spoonbill were 9.13, 7.14, and 13.42 ng MeHg/g (wwt), respectively, and the geometric mean was 9.56 ng MeHg/g (wwt).

5 Reasonableness of TRVs and TRCs

Deriving TRVs and TRCs for birds is always limited by the paucity of toxicological study results. To assess the risk posed by MeHg to birds in China, the TRVs and TRCs, based on concentrations of MeHg in tissues of fish and birds, were derived by applying two methods and incorporating the most recent toxicological data. These criteria values provide points of reference for concentrations of MeHg in aquatic life and birds, and can be used directly in the tissue residue approach to ecological risk assessment. To judge the reasonableness of protective guidelines derived by the two methods, they were compared to criteria developed by others.

CSA. In the criteria documents of the US EPA (US EPA 1995b) and CCME (CCME 2000), effects on reproduction of mallard (Heinz 1974, 1975, 1976a, b, 1979) were used as the most appropriate results for deriving wildlife criteria values. The sensitivity of the reproductive endpoint of mallard to MeHg was similar to that of the reproductive and immune endpoints of American kestrels, but less than that of the immune and biochemical endpoints of juvenile common loons, and the reproductive endpoint of white ibises (Fig. 1). Thus, it was concluded that reproduction of the mallard is relatively insensitive to the effects of MeHg (Heinz et al. 2010a). This conclusion was confirmed by using a study in which the eggs of 26 bird species were injected with MeHg (Heinz et al. 2009). Only one species, the double-crested cormorant (*Phalacrocorax auritus*), was less sensitive than the mallard (Heinz et al. 2009). Therefore, basing the TRVs and TRCs on the reproduction study results in white ibises provides more protection to avian wildlife than basing them only on the results of MeHg effects on the mallard.

A dietary-based TRC of 33 ng MeHg/g (wwt) has been derived by CCME for Wilson's storm petrel from study results on mallard without applying UFs (CCME 2000). If a UF of 2.0 is applied to this value, the TRC for Wilson's storm petrel would be 16.5 ng MeHg/g (wwt), which is similar to the value derived in the present evaluation. However, the TRV given by the CCME would become 15.5 if a UF of 2.0 is applied. This value is three times greater than the TRV of 5.0 ng MeHg/g (bm)/day derived in the present assessment. The difference is because Wilson's storm petrel has a greater ingestion of food (FI):BM ratio of 0.94. The species with the greatest FI:BW ratio has the greatest potential exposure to contaminants. Hence, the selection of representative species is important for deriving a TRC.

A NOAEL of 0.014 μg MeHg/g (bm)/day for effects on immune function and other biochemical effects in juvenile common loons is similar to the value derived for the white ibis. When this value was used to derive a TRV, and a UF_L of 1.0, a UF_S of 1.0 and a UF_A of 3.0 were considered; the resulting avian dietary-based TRV would be 4.7 ng MeHg/g (bm)/day, which is similar to the value derived from the MeHg toxicity to white ibis. The UF_L was set to 1.0 because the common loons study provided a NOAEL rather than a LOAEL (Kenow et al. 2007a). According to pharmacokinetic studies of absorption and elimination of MeHg in common loon chicks (Fournier et al. 2002), the result of a study of common loons, with a duration of 15 weeks, was accepted as being a chronic exposure study. Thus, a UF_S of >1.0 was deemed to be unnecessary (Kenow et al. 2007a). Although the study by Kenow et al. (2007a) did not provide information on the most ecologically relevant endpoint, which is reproduction, the common loon is a piscivorous wading bird and is the second most sensitive avian species to MeHg. Applying a UF_A of 10 is likely to be overly conservative. Therefore, according to the UF_A used in GLWQI Criteria Document (US EPA 1995b), an intermediate value of 3.0 was used for the UF_A.

SSD. SSD is an effective method to represent the variation in sensitivity to chemicals among species. A specified effect level, such as the proportion of species expected to respond to a particular exposure for a specific measurement endpoint, can be determined so that most species are protected. SSDs have been used to assess risk

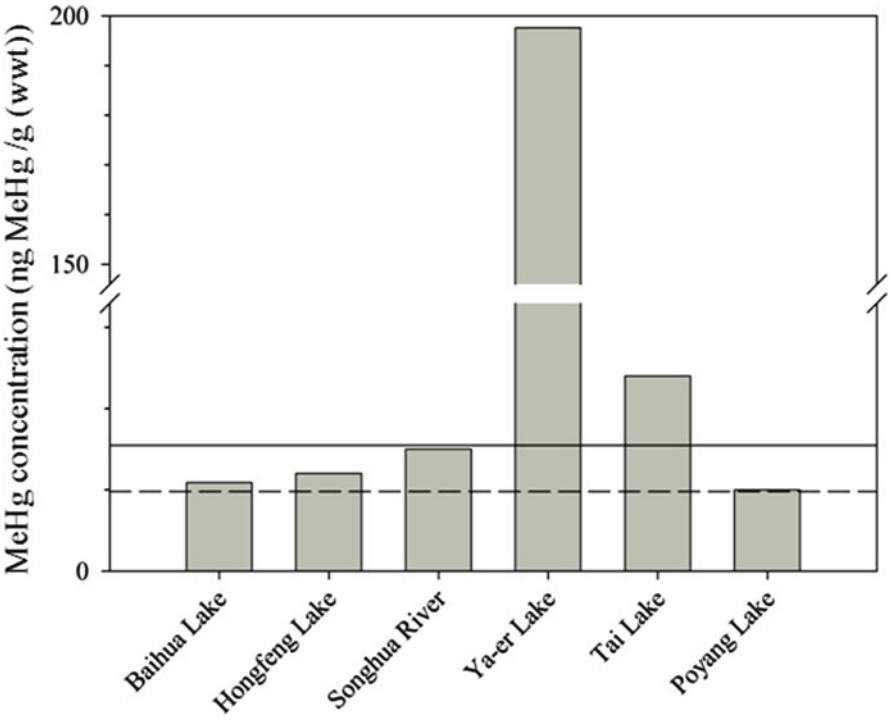

Fig. 3 MeHg concentrations found in fish from different aquatic systems in China, compared to the dietary-based TRC values from critical study approach (CSA) and species sensitivity distribution (SSD) approach. The *solid* and *dashed lines* represent the TRC values from CSA and SSD, respectively. *wwt* wet weight. Baihua Lake (Yan et al. 2008), Hongfeng Lake (He et al. 2010), Songhua River (Zhu et al. 2012), Ya-er Lake (Jin et al. 2006) Tai Lake, and Poyang Lake (Zhang et al. 2006)

and develop water quality criteria for aquatic species (Caldwell et al. 2008; Schuler et al. 2008), but have had limited application for wildlife because of the dearth of toxicity data for wildlife (Awkerman et al. 2008). In some studies, SSDs have been used to derive quality criteria to protect top predators from residues in soils (Jongbloed et al. 1996; Traas et al. 1996). By incorporating interspecies toxicity correlation models, SSDs were developed for wildlife from toxicity data on 23 chemicals (Awkerman et al. 2008). SSDs created for 15 or more wildlife species could give accurate results, whereas data for approximately 7 species can be used to provide only an adequate estimate for some combinations of chemicals and species (Awkerman et al. 2008). In the present study, information for ten species was used to construct the SSD for MeHg, with the log-normal function providing an accepted fit to the distribution, which was tested by use of the Anderson-Darling and Kolmogorov-Smirnov tests. The dietary TRC of 9.56 ng MeHg/g (wwt), derived by SSD, was slightly less than the value of 15.47 ng MeHg/g (wwt) that resulted from

CSA, but the results were similar. Because this assessment, the results of which are reported here, utilized data for ten species, the assessment resolution was no better than 10% (1 of 10). However, an HC_5 was interpolated, which previous work has shown is similar to the threshold value for effects as determined in multispecies tests. This interpolation makes sense, because not all of a compound is environmentally relevant and effects on animals are dependent on the dose and dose rate. Furthermore, animals have the ability to repair damage, and thereby they exhibit some resilience. For these reasons the HC_5 is regarded as being a reasonable surrogate for the adverse effect threshold at the community and ecosystem levels of organization (Giesy et al. 1999).

6 Comparison to Ambient Concentrations in Tissues

To judge the reasonableness of protective guidelines derived by the two methods, and to determine the potential for MeHg to cause adverse effects on fish-eating birds, the derived TRCs were compared to MeHg concentrations measured during monitoring of aquatic environments in China. Concentrations of MeHg in fish tissues, and THg in birds of some Chinese aquatic systems, were collected from the literature. Concentrations of THg have been reported for prey of little egrets in Tai Lake and Poyang Lake, and the geometric means given were 0.24 and 0.10 μg THg/g (dwt), respectively (Zhang et al. 2006). According to a conservative assumption that vertebrates have >50% of the THg as MeHg in muscle (Albers et al. 2007; Eisler 2000), and that the moisture content is 80%, the estimated MeHg concentrations in the prey of little egrets in Tai Lake and Poyang Lake would be 0.024 and 0.01 μg MeHg/g (wwt), respectively. Concentrations of MeHg in fish from Baihua Lake (Yan et al. 2008), Hongfeng Lake (He et al. 2010), Songhua River (Zhu et al. 2012), and Ya-er Lake (Jin et al. 2006) were also available for comparison (Fig. 3). All recorded concentrations of MeHg in fish were greater than the dietary-based TRC value of 9.56 ng MeHg/g (wwt), which was derived by using the SSD. Concentrations of Hg in fish from the Songhua River and Ya-er Lake were significantly greater than this TRC value ($p < 0.05$ by t-test), while concentrations of Hg in fish from Baihua, Hongfeng, and Poyang Lakes approached this TRC value (Fig. 3). Concentrations of MeHg found in fishes of Ya-er Lake (Jin et al. 2006) and prey of little egrets in Tai Lake were greater than the dietary-based TRC of 15.47 ng MeHg/g (wwt) derived by CSA, especially that of Ya-er Lake, which was significantly different from the TRC ($p < 0.05$ by t-test). Therefore, there is a significant risk of MeHg causing adverse effects on avian wildlife populations in the Songhua River, and Ya-er and Tai Lakes, but a lesser risk in Baihua, Hongfeng, and Poyang Lakes. These results are consistent with the pollution characteristics of these water bodies. The Songhua River (Zhu et al. 2012), and Ya-er (Jin et al. 2006) and Tai Lakes are known to be more polluted than Poyang Lake (Zhang et al. 2006). Although both Baihua and Hongfeng Lakes were polluted by Hg from chemical plants in southwest China, the special characteristics of the water environment (e.g., alkaline

water, eutrophication, low-dissolved organic carbon, and short food chain) limited accumulation of Hg in fish (He et al. 2010; Yan et al. 2008).

Currently, no data for concentrations of MeHg in blood or feathers of birds in China are available for comparison to the developed criteria. However, studies have been conducted to determine the total concentrations of Hg (THg) in bird feathers, including species located in Poyang, Tai Lakes, the Pearl River Delta (Zhang et al. 2006), Hong Kong and Szechuan (Burger and Gochfeld 1993). Nearly 100% of the Hg that is found in feathers is in the form of MeHg (Herring et al. 2009; Thompson and Furness 1989; Kim et al. 1996). The range of concentrations of THg in feathers was 0.26–4.1 μg THg/g (dwt). The TRC, based on concentrations of THg in feathers that were determined in the present meta-analysis, was expressed as μg THg/g (fresh weight basis). Because information on the feather moisture content of wild birds was not available to convert dry mass to fresh mass, some mallard feathers were collected and dried for 12 h in an oven at 80 °C. The range of moisture content found in mallard feathers was 14–20%. Thus, it was assumed that the moisture content of scapular feather of white ibis was approximately 20%. Given that value, the TRC, based on concentrations expressed on a dry weight basis, would be 3.95 μg THg/g (dwt). Only the concentration of Hg in feathers of little egret from Au Tau in Hong Kong (Connell et al. 2001) exceeded the TRC (Fig. 4). This result is in agreement with a previous risk assessment, in which Hg probably had an adverse effect on the breeding success of the little egret at this site (Connell et al. 2001). Although the concentration of Hg in little egret prey at Tai Lake exceeded both dietary-based TRCs, the level of Hg found in little egret feathers was less than the feather-based TRC. We speculated that this may be due to absorption and metabolism of Hg in the little egret. In conclusion, the TRC values reported in this study can be used as indicators for screening-level risk assessment of avian wildlife in Chinese aquatic systems.

7 Evaluation of Uncertainties

Describing and assessing uncertainty is an important part of deriving and applying TRVs and TRCs (US EPA 1998). Application of qualitative and quantitative expressions of uncertainty compensate for deficiencies in knowledge concerning the accuracy of test results and the data gaps related to the extrapolation of toxicity data among species. If uncertainty factors are identified and are defensible, a more accurate estimate of TRVs and TRCs for protecting wildlife can be achieved. However, when uncertainty factors are applied, they are meant to make criteria more protective without being overly protective; the resulting TRVs and TRCs should be considered to be protective more than predictive of adverse effects under field conditions. According to the US EPA GLI Methodology and GLWQI technical support document for wildlife criteria, three sources of uncertainty were considered for CSA in this study. The first source of uncertainty was associated with interspecies extrapolation. Based on a comparison with other species for which LOAEL and NOAEL values are

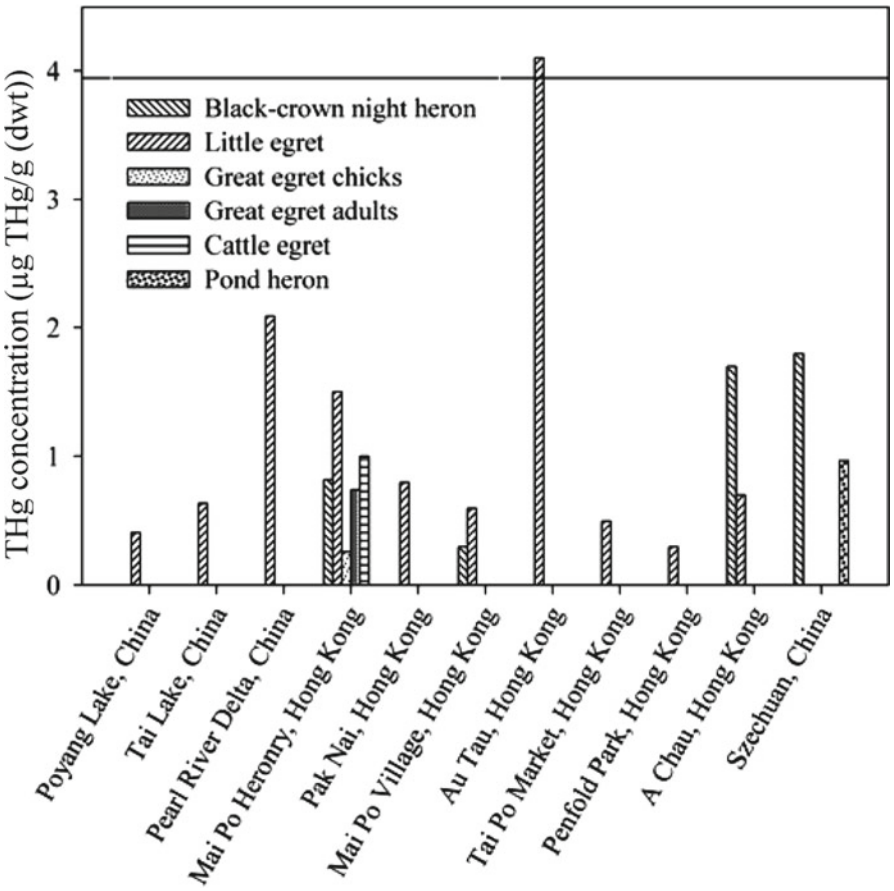

Fig. 4 Total mercury concentrations (THg) found at different sites for bird feathers in China. The *horizontal solid line* represents the toxicity reference criterion (TRC) for feathers. *dwt* dry weight. Poyang, Tai Lakes, and Pearl River Delta (Zhang et al. 2006), Hong Kong and Szechuan (Burger and Gochfeld 1993)

presented (Table 2), the white ibis is the most sensitive species. The results of a study, in which MeHg was injected into eggs, indicates that white ibis is one of the species that are most sensitive to injected MeHg (Heinz et al. 2009). In addition, the results of tests done under laboratory conditions are available from ten species of birds; the white ibis is a piscivorous wading bird that is in the same order as the three representative birds. Therefore, an UF_A of 1.0 was reasonable for interspecies extrapolation of toxicity data to protect other avian species. The second source of uncertainty was associated with LOAEL to NOAEL extrapolation. The LOAEL selected for deriving TRVs was identified from the least exposure dose of 0.05 μg MeHg/g (wwt) in the white ibis study. Compared with the controls, the loss of productivity for this dose of

0.05 μg MeHg/g (wwt) over the 3-year test period was only 13.2% (Frederick and Jayasena 2010). This LOAEL is similar to the NOAEL for common loons exposed to MeHg in the diet (Fig. 1). This LOAEL was similar to the threshold for effects of MeHg on white ibises. In addition, the range of LOAEL to NOAEL ratios for other species was 1.5–6. Thus, a UF_L of 2.0 was used in the extrapolation of the LOAEL to NOAEL. The third source of uncertainty was associated with extrapolation from results of subchronic exposures to chronic exposures. In the white ibis reproductive study, individuals were initially exposed to MeHg when birds were 90 days old, and exposure continued over 3 years, covering three breeding seasons. The value therefore needs no adjustment to cover longer exposure periods. A value of 1.0 was assigned for the UF_S, which could protect the wildlife against chronic effects.

When the SSD approach was used, two sources of uncertainty were explicitly considered, i.e., the relationship between the LOAEL and NOAEL and between subchronic and chronic exposures. Considering the LOAEL to NOAEL ratios of species, a value of 2.0 was assigned to the LOAEL to NOAEL correction factor. A factor of 10 was used to account for the uncertainty in establishing a NOAEL from short-term exposure (<1 month) (Jongbloed et al. 1996). Although the SSD is useful in assessing the range of sensitivities among species during derivation of water quality criteria, much attention has been directed towards extrapolation from laboratory tests to field conditions (Forbes and Forbes 1993; Smith and Cairns 1993). Other uncertainty factors have been used to correct NOAEL data from laboratory tests, and have been used to account for differences in metabolic rate, caloric content of food, and food assimilation efficiency between laboratory and wild species (Traas et al. 1996). Moreover, some studies used a statistical procedure, which is more scientifically defensible, to estimate uncertainty factors so as to obtain precise uncertainty factors and criteria (Calabrese and Baldwin 1994; Dourson and Parker 2007; Gaylor and Kodell 2000). Thus, more research is needed for addressing the suitability of TRVs and TRCs that are derived from laboratory species for protecting avian wildlife in the field.

8 Summary

MeHg is the most biologically available and toxic form of mercury, and has the potential to bioaccumulate and biomagnify as it moves up the food chain. These characteristics result in MeHg exposure to avian wildlife at high trophic levels that can produce adverse effects. The toxicity of MeHg to birds was reviewed, and using available data, TRVs and TRCs were derived for protecting birds in China. The TRV and TRC values were based on concentrations of MeHg in diet (or fish tissue based) and tissues of birds. Two methods were applied to derive TRVs from concentrations in the diet or in tissues. These were the CSA and SSD approaches. Results of published studies show that reproductive productivity of white ibis was the most sensitive endpoint for MeHg exposure, and study results on white ibises were used for deriving the TRV and TRC values, which included applying a UF of 2.0. For the

SSD approach, data for ten species were used to construct the SSD for MeHg, and to calculate the dietary-based TRV and TRC values. Using the CSA approach, the TRV was based on MeHg in the diet and was derived as 5.0 ng MeHg/g (bm)/day; for feathers and blood, the TRVs were 3.16 μg THg/g (wwt), and 0.365 μg THg/g (wwt), respectively. The corresponding TRCs were 15.47 ng MeHg/g (wwt), 3.16 μg THg/g (wwt) and 0.365 μg THg/g (wwt), respectively. The dietary-based TRV and TRC derived by SSD were 3.09 ng MeHg/g (bm)/day and 9.56 ng MeHg/g (wwt), respectively. However, bird tissue residue-based criteria were not available because insufficient MeHg effects data existed to construct an SSD for birds. We compared the criteria derived in our study to those developed by others, and concluded that our results provided more reasonable protection to Chinese avian wildlife. By comparing the criteria values we calculated to actual MeHg levels in fish and bird tissues, we concluded that these criteria values are useful indicators for screening-level risk assessments of avian wildlife in Chinese aquatic systems. The results of this meta-analysis might therefore have important implications for assessing the risk of Hg exposure to birds and for environmental management in China and in other regions. Moreover, because humans and top avian wildlife consumers are at the same trophic level, these criteria may also be used as a reference for human health risk assessment. The diet of birds consists of aquatic species from different trophic levels. However, the structure of the food web for avian wildlife and the environmental factors that affect their exposure to MeHg vary among aquatic systems. Therefore, further research results are needed on the food web structure of avian wildlife in Chinese aquatic systems to provide more insight into what constitutes adequate protection for avian wildlife.

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References

Adams EM, Frederick PC (2008) Effects of methylmercury and spatial complexity on foraging behavior and foraging efficiency in juvenile white ibises (*Eudocimus albus*). Environ Toxicol Chem 27(8):1708–1712. doi:10.1897/07-466.1

Albers PH, Koterba MT, Rossmann R, Link WA, French JB, Bennett RS, Bauer WC (2007) Effects of methylmercury on reproduction in American kestrels. Environ Toxicol Chem 26(9):1856–1866. doi:10.1897/06-592r.1

Aldenberg T, Jaworska JS (2000) Uncertainty of the hazardous concentration and fraction affected for normal species sensitivity distributions. Ecotoxicol Environ Saf 46(1):1–18. doi:10.1006/eesa.1999.1869

Aldenberg T, Slob W (1993) Confidence limits for hazardous concentrations based on logistically distributed NOEC toxicity data. Ecotoxicol Environ Saf 25(1):48–63

An W, Hu J, Yao F (2006) A method of assessing ecological risk to night heron, *Nycticorax nycticorax*, population persistence from dichlorodiphenyltrichloroethane exposure. Environ Toxicol Chem 25(1):281–286. doi:10.1897/05-043r.1

Awkerman JA, Raimondo S, Barron MG (2008) Development of species sensitivity distributions for wildlife using interspecies toxicity correlation models. Environ Sci Technol 42(9):3447–3452. doi:10.1021/es702861u

Barr JF (1996) Aspects of common loon (*Gavia immer*) feeding biology on its breeding ground. Hydrobiologia 321(2):119–144. doi:10.1007/bf00023169

Barter M, Cao L, Chen L, Lei G (2005) Results of a survey for waterbirds in the lower Yangtze floodplain, China, in January-February 2004. Forktail 21:1–7

Beckvar N, Dillon TM, Read LB (2005) Approaches for linking whole-body fish tissue residues of mercury or DDT to biological effects thresholds. Environ Toxicol Chem 24(8):2094–2105. doi:10.1897/04-284r.1

Blankenship AL, Kay DP, Zwiernik MJ, Holem RR, Newsted JL, Hecker M, Giesy JP (2008) Toxicity reference values for mink exposed to 2,3,7,8-tetrachlodibenzo-p-dioxin (TCDD) equivalents (TEQs). Ecotoxicol Environ Saf 69(3):325–349. doi:10.1016/j.ecoenv.2007.08.017

Bouton SN, Frederick PC, Spalding MG, McGill H (1999) Effects of chronic, low concentrations of dietary methylmercury on the behavior of juvenile great egrets. Environ Toxicol Chem 18(9):1934–1939. doi:10.1002/etc.5620180911

Buekers J, Steen Redeker E, Smolders E (2009) Lead toxicity to wildlife: derivation of a critical blood concentration for wildlife monitoring based on literature data. Sci Total Environ 407(11):3431–3438. doi:10.1016/j.scitotenv.2009.01.044

Burger J, Gochfeld M (1993) Heavy metal and selenium levels in feathers of young egrets and herons from Hong Kong and Szechuan, China. Arch Environ Contam Toxicol 25(3):322–327. doi:10.1007/bf00210724

Burger J, Gochfeld M (1997) Heavy metal and selenium concentrations in feathers of egrets from Bali and Sulawesi, Indonesia. Arch Environ Contam Toxicol 32(2):217–221. doi:10.1007/s002449900178

Calabrese EJ, Baldwin LA (1994) A toxicological basis to derive a generic interspecies uncertainty factor. Environ Health Perspect 102(1):14–17

Caldwell DJ, Mastrocco F, Hutchinson TH, Länge R, Heijerick D, Janssen C, Anderson PD, Sumpter JP (2008) Derivation of an aquatic predicted no-effect concentration for the synthetic hormone, 17α-ethinyl estradiol. Environ Sci Technol 42(19):7046–7054. doi:10.1021/es800633q

CCME (1998) Protocol for derivation of canadian tissue residue guidelines for the protection of wildlife that consume aquatic biota. Canadian Council of Ministers of the Environment, Winnipeg

CCME (2000) Canadian tissue residue guidelines for the protection of wildlife consumers of aquatic biota: Methylmercury. Canadian Council of Ministers of the Environment, Winnipeg

Connell DW, Wong BSF, Lam PKS, Poon KF, Lam MHW, Wu RSS, Richardson BJ, Yen YF (2001) Risk to breeding success of ardeids by contaminants in Hong Kong: evidence from trace metals in feathers. Ecotoxicology 11(1):49–59. doi:10.1023/a:1013745113901

Delnicki D, Reinecke KJ (1986) Mid-winter food use and body weights of mallards and wood ducks in Mississippi. J Wildl Manage 50(1):43–51

Dourson ML, Parker AL (2007) Past and future use of default assumptions and uncertainty factors: default assumptions, misunderstandings, and new concepts. Hum Ecol Risk Assess 13(1):82–87. doi:10.1080/10807030601105480

Dunning JB (1993) CRC handbook of avian body masses. CRC Press I LLC, Boca Raton, FL

Eisler R (ed) (2000) Handbook of chemical risk assessment: Health hazards to humans, plants, and animals, Volume 1: Metals. Lewis Publishers, Boca Raton, FL

Eskeland B, Nafstad I (1978) The modifying effect of multiple generation selection and dietary cadmium on methyl mercury toxicity in Japanese quail. Arch Toxicol 40(4):303–314

Fallacara DM, Halbrook RS, French JB (2011) Toxic effects of dietary methylmercury on immune function and hematology in American kestrels (*Falco sparverius*). Environ Toxicol Chem 30(6):1320–1327. doi:10.1002/etc.494

Feng X (2005) Mercury pollution in China—an overview. In: Pirrone N, Mahaffey KR (eds) Dynamics of mercury pollution on regional and global scales: atmospheric processes and human exposures around the world. Springer US, New York, pp 657–678. doi:10.1007/0-387-24494-8_27

Fimreite N (1970) Effects of methyl mercury treated feed on the mortality and growth of leghorn cockerels. Can J Anim Sci 50(2):387–389. doi:10.4141/cjas70-058

Fimreite N (1971) Effects of dietary methylmercury on ring-necked pheasants, with special reference to reproduction. Canadian Wildlife Service, Ottawa, Occasional paper, no. 9. Department of the Environment

Fimreite N, Karstad L (1971) Effects of dietary methyl mercury on red-tailed hawks. J Wildl Manage 35:293–300

Forbes TL, Forbes VE (1993) A critique of the use of distribution-based extrapolation models in ecotoxicology. Funct Ecol 7(3):249–254

Fournier F, Karasov WH, Kenow KP, Meyer MW, Hines RK (2002) The oral bioavailability and toxicokinetics of methylmercury in common loon (*Gavia immer*) chicks. Comp Biochem Physiol A Mol Integr Physiol 133(3):703–714. doi:10.1016/s1095-6433(02)00140-x

Frederick P, Jayasena N (2010) Altered pairing behaviour and reproductive success in white ibises exposed to environmentally relevant concentrations of methylmercury. Proc R Soc B 278:1851–1857. doi:10.1098/rspb.2010.2189

Frederick P, Campbell A, Jayasena N, Borkhataria R (2011) Survival of white ibises (*Eudocimus albus*) in response to chronic experimental methylmercury exposure. Ecotoxicology 20(2):358–364. doi:10.1007/s10646-010-0586-9

Fujita M (2003) Head bobbing and the body movement of little egrets (*Egretta garzetta*) during walking. J Comp Physiol A 189(1):53–58. doi:10.1007/s00359-002-0376-9

Gaylor DW, Kodell RL (2000) Percentiles of the product of uncertainty factors for establishing probabilistic reference doses. Risk Anal 20(2):245–250. doi:10.1111/0272-4332.202023

Giesy JP, Solomon KR, Coats JR, Dixon KR, Giddings JM, Kenaga EE (1999) Chlorpyrifos: ecological risk assessment in North American aquatic environments. Rev Environ Contam T 160:1–129

Hall LW, Scott MC, Killen WD (1998) Ecological risk assessment of copper and cadmium in surface waters of Chesapeake Bay watershed. Environ Toxicol Chem 17(6):1172–1189. doi:10.1002/etc.5620170626

He T, Wu Y, Pan L, Feng X (2010) Distribution of mercury species and their concentrations in fish in hongfeng reservoir. Journal of Southwest University (Natural Science Edition) 32(7):78–82

Heath JA, Frederick PC, Karasov W (2005) Relationships among mercury concentrations, hormones, and nesting effort of white ibises (*Eudocimus albus*) in the Florida Everglades. The Auk 122(1):255–267

Heinz G (1974) Effects of low dietary levels of methyl mercury on mallard reproduction. Bull Environ Contam Toxicol 11(4):386–392. doi:10.1007/bf01684947

Heinz G (1975) Effects of methylmercury on approach and avoidance behavior of mallard ducklings. Bull Environ Contam Toxicol 13(5):554–564. doi:10.1007/bf01685179

Heinz GH (1976a) Methylmercury: second-generation reproductive and behavioral effects on mallard ducks. J Wildl Manage 40(4):710–715

Heinz GH (1976b) Methylmercury: second-year feeding effects on mallard reproduction and duckling behavior. J Wildl Manage 40:82–90

Heinz GH (1979) Methylmercury: reproductive and behavioral effects on three generations of mallard ducks. J Wildl Manage 43:394–401

Heinz G, Locke LN (1976) Brain lesions in mallard ducklings from parents fed methylmercury. Avian Dis 20(1):9–17

Heinz G, Hoffman D, Klimstra J, Stebbins K, Kondrad S, Erwin C (2009) Species differences in the sensitivity of avian embryos to methylmercury. Arch Environ Contam Toxicol 56(1):129–138. doi:10.1007/s00244-008-9160-3

Heinz G, Hoffman D, Klimstra J, Stebbins K (2010a) Reproduction in mallards exposed to dietary concentrations of methylmercury. Ecotoxicology 19(5):977–982. doi:10.1007/s10646-010-0479-y

Heinz GH, Hoffman DJ, Klimstra JD, Stebbins KR (2010b) Enhanced reproduction in mallards fed a low level of methylmercury: an apparent case of hormesis. Environ Toxicol Chem 29(3):650–653. doi:10.1002/etc.64

Heinz G, Hoffman D, Klimstra J, Stebbins K, Kondrad S, Erwin C (2011) Hormesis associated with a low dose of methylmercury injected into mallard eggs. Arch Environ Contam Toxicol 62(1):141–144. doi:10.1007/s00244-011-9680-0

Herring G, Gawlik DE, Rumbold DG (2009) Feather mercury concentrations and physiological condition of great egret and white ibis nestlings in the Florida Everglades. Sci Total Environ 407(8):2641–2649. doi:10.1016/j.scitotenv.2008.12.043

Hoffman DJ, Spalding MG, Frederick PC (2005) Subchronic effects of methylmercury on plasma and organ biochemistries in great egret nestlings. Environ Toxicol Chem 24(12):3078–3084. doi:10.1897/04-570.1

Jayasena N, Frederick PC, Larkin ILV (2011) Endocrine disruption in white ibises (*Eudocimus albus*) caused by exposure to environmentally relevant levels of methylmercury. Aquat Toxicol 105(3–4):321–327. doi:10.1016/j.aquatox.2011.07.003

Jin L, Liang L, Jiang G, Xu Y (2006) Methylmercury, total mercury and total selenium in four common freshwater fish species from Ya-Er Lake, China. Environ Geochem Health 28(5):401–407. doi:10.1007/s10653-005-9038-5

Jongbloed R, Traas T, Luttik R (1996) A probabilistic model for deriving soil quality criteria based on secondary poisoning of top predators: II. Calculations for dichlorodiphenyltrichloroethane (DDT) and cadmium. Ecotoxicol Environ Saf 34(3):279–306

Kahle S, Becker PH (1999) Bird blood as bioindicator for mercury in the environment. Chemosphere 39(14):2451–2457. doi:10.1016/s0045-6535(99)00154-x

Kannan K, Blankenship AL, Jones PD, Giesy JP (2000) Toxicity reference values for the toxic effects of polychlorinated biphenyls to aquatic mammals. Hum Ecol Risk Assess 6(1):181–201. doi:10.1080/10807030091124491

Kenow KP, Gutreuter S, Hines RK, Meyer MW, Fournier F, Karasov WH (2003) Effects of methyl mercury exposure on the growth of juvenile common loons. Ecotoxicology 12(1):171–181. doi:10.1023/a:1022598525891

Kenow KP, Grasman KA, Hines RK, Meyer MW, Gendron-Fitzpatrick A, Spalding MG, Gray BR (2007a) Effects of methylmercury exposure on the immune function of juvenile common loons (*Gavia immer*). Environ Toxicol Chem 26(7):1460–1469. doi:10.1897/06-442r.1

Kenow KP, Meyer MW, Hines RK, Karasov WH (2007b) Distribution and accumulation of mercury in tissues of captive-reared common loon (*Gavia immer*) chicks. Environ Toxicol Chem 26(5):1047–1055. doi:10.1897/06-193r.1

Kenow KP, Hoffman DJ, Hines RK, Meyer MW, Bickham JW, Matson CW, Stebbins KR, Montagna P, Elfessi A (2008) Effects of methylmercury exposure on glutathione metabolism, oxidative stress, and chromosomal damage in captive-reared common loon (*Gavia immer*) chicks. Environ Pollut 156(3):732–738. doi:10.1016/j.envpol.2008.06.009

Kenow KP, Hines RK, Meyer MW, Suarez SA, Gray BR (2010) Effects of methylmercury exposure on the behavior of captive-reared common loon (*Gavia immer*) chicks. Ecotoxicology 19(5):933–944. doi:10.1007/s10646-010-0475-2

Kim EY, Murakami T, Saeki K, Tatsukawa R (1996) Mercury levels and its chemical form in tissues and organs of seabirds. Arch Environ Contam Toxicol 30(2):259–266. doi:10.1007/bf00215806

Kooijman SALM (1987) A safety factor for LC50 values allowing for differences in sensitivity among species. Water Res 21(3):269–276. doi:10.1016/0043-1354(87)90205-3

Kushlan JA (1977) Population energetics of the American white ibis. Auk 94(1):114–122

Kushlan JA (1978) Feeding ecology of wading birds. In: Sprunt A, Ogden J, Winckler S (eds) Wading birds, vol 7. National Audubon Society Research Report, New York, pp 249–296

Ling C (2004) A conservative, nonparametric estimator for the 5th percentile of the species sensitivity distributions. J Stat Plan Inference 123(2):243–258. doi:10.1016/s0378-3758(03)00148-4

Liu J, Wang D, Sun R (2003) Growth and development of homeothermy in nestlings of Eurasian spoonbill (*Platalea eucorodia*). Zoological Res 24(4):249–253

Liu G, Cai Y, Philippi T, Kalla P, Scheidt D, Richards J, Scinto L, Appleby C (2008) Distribution of total and methylmercury in different ecosystem compartments in the Everglades: implications for mercury bioaccumulation. Environ Pollut 153(2):257–265. doi:10.1016/j.envpol.2007.08.030

Nagy KA (1987) Field metabolic rate and food requirement scaling in mammals and birds. Ecol monogr 57:112–128

Nagy K (2001) Food requirements of wild animals: predictive equations for free-living mammals, reptiles, and birds. Nutr Abstr Rev Ser B 71:21R–31R

Newman MC, Ownby DR, Mézin LCA, Powell DC, Christensen TRL, Lerberg SB, Anderson B-A (2000) Applying species-sensitivity distributions in ecological risk assessment: assumptions of distribution type and sufficient numbers of species. Environ Toxicol Chem 19(2):508–515. doi:10.1002/etc.5620190233

Newsted JL, Jones PD, Coady K, Giesy JP (2005) Avian toxicity reference values for perfluorooctane sulfonate. Environ Sci Technol 39(23):9357–9362. doi:10.1021/es050989v

Posthuma L, Suter GW, Traas TP (2002) Species sensitivity distributions in ecotoxicology. CRC Press LLC, Boca Raton, FL

Rumbold D, Lange T, Axelrad D, Atkeson T (2008) Ecological risk of methylmercury in Everglades National Park, Florida, USA. Ecotoxicology 17(7):632–641. doi:10.1007/s10646-008-0234-9

Sample B, Suter D (1993) Toxicological benchmarks for wildlife. ORNL Oak Ridge National Laboratory (US), Oak Ridge

Sappington KG, Bridges TS, Bradbury SP, Erickson RJ, Hendriks AJ, Lanno RP, Meador JP, Mount DR, Salazar MH, Spry DJ (2011) Application of the tissue residue approach in ecological risk assessment. Integr Environ Assess Manag 7(1):116–140. doi:10.1002/ieam.116

Scheuhammer A (1988) Chronic dietary toxicity of methylmercury in the zebra finch, *Poephila guttata*. Bull Environ Contam Toxicol 40(1):123–130

Scheuhammer AM, Meyer MW, Sandheinrich MB, Murray MW (2007) Effects of environmental methylmercury on the health of wild birds, mammals, and fish. Ambio 36(1):12–19

Schuler L, Hoang T, Rand G (2008) Aquatic risk assessment of copper in freshwater and saltwater ecosystems of South Florida. Ecotoxicology 17(7):642–659. doi:10.1007/s10646-008-0236-7

Scott M (1977) Effects of PCBs, DDT, and mercury compounds in chickens and Japanese quail. Fed Proc 36:1888–1893

Smith EP, Cairns J (1993) Extrapolation methods for setting ecological standards for water quality: statistical and ecological concerns. Ecotoxicology 2(3):203–219. doi:10.1007/bf00116425

Solomon KR, Baker DB, Richards RP, Dixon KR, Klaine SJ, La Point TW, Kendall RJ, Weisskopf CP, Giddings JM, Giesy JP, Hall LW, Williams WM (1996) Ecological risk assessment of atrazine in North American surface waters. Environ Toxicol Chem 15(1):31–76. doi:10.1002/etc.5620150105

Spalding MG, Frederick PC, McGill HC, Bouton SN, McDowell LR (2000a) Methylmercury accumulation in tissues and its effects on growth and appetite in captive great egrets. J Wildl Dis 36(3):411

Spalding MG, Frederick PC, McGill HC, Bouton SN, Richey LJ, Schumacher IM, Blackmore C, Harrison J (2000b) Histologic, neurologic, and immunologic effects of methylmercury in captive great egrets. J Wildl Dis 36(3):423

Spann J, Heath R, Kreitzer J, Locke L (1972) Ethyl mercury p-toluene sulfonanilide: lethal and reproductive effects on pheasants. Science 175(4019):328

Stanton B, de Vries S, Donohoe R, Anderson M, Eichelberger JM (2010) Recommended avian toxicity reference value for cadmium: justification and rationale for use in ecological risk assessments. Hum Ecol Risk Assess 16(6):1261–1277. doi:10.1080/10807039.2010.526499

Stephan CE, Mount DI, Hansen DJ, Gentile JH, Chapman GA, Brungs WA (1985) Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. US EPA, Washington DC

Tan SW, Meiller JC, Mahaffey KR (2009) The endocrine effects of mercury in humans and wildlife. Crit Rev Toxicol 39(3):228–269. doi:10.1080/10408440802233259

Thompson DR, Furness RW (1989) The chemical form of mercury stored in South Atlantic seabirds. Environ Pollut 60(3–4):305–317. doi:10.1016/0269-7491(89)90111-5

Traas T, Luttik R, Jongbloed R (1996) A probabilistic model for deriving soil quality criteria based on secondary poisoning of top predators: I. model description and uncertainty analysis. Ecotoxicol Environ Saf 34(3):264–278

US EPA (1993) Wildlife exposure factors handbook, vol I. US EPA, Office of Research and Development, Washington DC, EPA/600/R-93/187

US EPA (1995a) Final water quality guidance for the great lakes. Fed Regist 60(56): 15366–15425

US EPA (1995b) Great lakes water quality initiative criteria documents for the protection of wildlife, US EPA. Office of Water, Washington DC

US EPA (1995c) Great lakes water quality initiative technical support document for wildlife criteria, US EPA. Office of Water, Washington DC

US EPA (1998) Guidelines for ecological risk assessment, US EPA. Office of Research and Development, Washington DC

US EPA (2003) Attachment 4–5. Ecological Soil Screening Levels (Eco-SSLs) standard operating procedure SOP No.6: Derivation of wildlife toxicity reference value (TRV). OWSER directive 92857–55. US EPA, Washington DC

US EPA (2005) Science advisory board consultation document. proposed revisions to aquatic life guidelines: tissue-based criteria for "bioaccumulative" chemicals. US EPA, Office of Water, Washington DC

Van Vlaardingen PLA, Traas TP, Wintersen AM, Aldenberg T (2004) ETX 2.0. A program to calculate hazardous concentrations and fraction affected, based on normally distributed toxicity Data. National Institute for Public Health and the Environment, Bilthoven, Netherlands

Wagner C, Løkke H (1991) Estimation of ecotoxicological protection levels from NOEC toxicity data. Water Res 25(10):1237–1242. doi:10.1016/0043-1354(91)90062-u

Yan H, Feng X, Liu T, Shang L, Li Z, Li G (2008) Present situation of fish mercury pollution in heavily mercury-contaminated Baihua reservoir in Guizhou. Chin J Ecol 27(8):1357–1361

Yáñez JL, Núñez H, Schlatter RP, Jaksi FM (1980) Diet and weight of American kestrels in central Chile. Auk 97(3):629–631

Zamani-Ahmadmahmoodi R, Esmaili-Sari A, Savabieasfahani M, Bahramifar N (2010) Cattle egret (*Bubulcus ibis*) and little egret (*Egretta garzetta*) as monitors of mercury contamination in Shadegan Wetlands of south-western Iran. Environ Monit Assess 166(1):371–377. doi:10.1007/s10661-009-1008-4

Zhang L, Liu Z (1991) Ecological study on the breeding habits of little egret. J Shanxi Univ 14(2):202–208

Zhang Y, Ruan L, Fasola M, Boncompagni E, Dong Y, Dai N, Gandini C, Orvini E, Ruiz X (2006) Little egrets (*Egretta garzetta*) and trace-metal contamination in wetlands of China. Environ Monit Assess 118(1):355–368. doi:10.1007/s10661-006-1496-4

Zheng W, Kang S, Feng X, Zhang Q, Li C (2010) Mercury speciation and spatial distribution in surface waters of the Yarlung Zangbo River, Tibet. Chin Sci Bull 55(24):2697–2703. doi:10.1007/s11434-010-4001-y

Zhu H, Yan B, Cao H, Wang L (2012) Risk assessment for methylmercury in fish from the Songhua River, China: 30 years after mercury-containing wastewater outfalls were eliminated. Environ Monit Assess 184(1):77–88. doi:10.1007/s10661-011-1948-3