Toxicity Reference Values for Polybrominated Diphenyl Ethers: Risk Assessment for Predatory Birds and Mammals from Two Chinese Lakes

Ruiqing Zhang, Jianyang Guo, Fengchang Wu, Yunsong Mu, John P. Giesy, Hong Chang, Xiaoli Zhao, and Chenglian Feng

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R. Zhang
College of Environment and Resources, Inner Mongolia University, Huhhot 010021, Inner Mongolia, China

State Key Laboratory of Environmental Criteria and Risk Assessment, Chinese Research Academy of Environmental Sciences, Beijing 100012, China

J. Guo
State Key Laboratory of Environmental Geochemistry, Institute of Geochemistry, Chinese Academy of Sciences, Guiyang 550002, China

F. Wu (✉) • Y. Mu • H. Chang • X. Zhao • C. Feng
State Key Laboratory of Environmental Criteria and Risk Assessment, Chinese Research Academy of Environmental Sciences, Beijing 100012, China
e-mail: wufengchang@vip.skleg.cn

J.P. Giesy
Department of Veterinary Biomedical Sciences and Toxicology Centre, University of Saskatchewan, Saskatoon, SK, Canada

Zoology Department and Center for Integrative Toxicology, Michigan State University, East Lansing, MI 48824, USA

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1 Introduction

Polybrominated diphenyl ethers (PBDEs) are a class of brominated flame retardants (BFRs), that are widely used in products such as polymer resins, furniture and plastics (Environment Canada 2006). However, PBDEs have been recognized as persistent and bioaccumulative pollutants that can undergo long-range atmospheric transport (De Wit et al. 2006). Based on thresholds for effects of PBDEs on animals under laboratory conditions, PBDEs could accumulate to sufficient concentrations so as to pose a dietary risk to wildlife (Environment Canada 2006; Bureau 2001). Some PBDE congeners can bind to the arylhydrocarbon receptor (AhR), but the binding affinities are very weak or negligible (ATSDR 2004). PBDEs can affect neurobehavioral development, thyroid hormone concentrations in blood plasma, fetal development, reproductive performance, intracellular signaling processes, and also have estrogenic potency (Fernie et al. 2009; Darnerud 2003; ATSDR 2004). Therefore, production and use of some PBDEs technical mixtures have been banned or phased out in most countries (Gao et al. 2009a).

Because PBDEs undergo long-range transport, they have been detected in biota all over the world (Chen and Hale 2010). Their residues have especially appeared in China, where production and use of the PBDEs has occurred in textiles, plastics, and electronics, and where recycling of imported e-waste has distributed them widely in the environment (Wu et al. 2012). Concentrations of PBDEs in Chinese wildlife have generally been greater than those from wildlife from other parts of Asia, comparable to wildlife from Europe, but less than recorded for North American wildlife (Wu et al. 2012). The greatest concentrations of PBDEs in wildlife have been reported at e-waste recycling sites in China, which were greater than those from other regions around the world (Luo et al. 2009). PBDEs are biomagnified in the food web (Wan et al. 2008), where piscivorous species at higher trophic levels in aquatic systems can accumulate concentrations sufficient to be at considerable risk. Unfortunately, no specific criterion has been established against which concentrations of PBDEs in piscivorous species can be assessed for risk.

Most aquatic organisms accumulate greater proportions of the lesser-brominated congeners, with BDE-47 being the dominant congener, followed by BDE-99, BDE-100, and BDE-153 (Meng et al. 2008; Wu et al. 2012; Gao et al. 2009a). Profiles of relative concentrations of individual PBDEs congeners in aquatic organisms were similar to those in the penta-BDE mixture (DE-71), which is a commercial mixture of PBDEs. In DE-71, the percentage content of BDE-47, BDE-99, BDE-100, and BDE-153 are 38%, 49%, 13%, and 5.5%, respectively (La Guardia et al. 2006). DE-71 was extensively used as a flame retardant until its global production and use

was voluntarily discontinued in 2006 (Marteinson et al. 2010). The toxic potency of DE-71 has only been studied in a few top predators, such as mink and American kestrel (Fernie et al. 2009; Zhang et al. 2009).

For this review, the published data that address effects of PBDEs on birds and mammals were assessed. Relevant literature was reviewed and analyzed, and data were selected to determine toxicity reference values (TRVs) for both birds and mammals. These TRVs are based on threshold concentrations of the PBDEs in diet, on allowable daily intake (ADI) values, and on the concentrations in liver of mammals and eggs of birds.

In the present study, Dianchi Lake (DCL) and Tai Lake (TL) are being used as a case assessment of risks posed by PBDEs in fish to wildlife. DCL is the largest plateau lake in Yunnan Province, and TL is the second largest freshwater lake in China (Guo et al. 2012). These lakes are important breeding and wintering areas for migratory waterfowl. However, both areas are polluted with persistent contaminants (Nakata et al. 2005), which pose potential risks of adverse effects to the local fish-eating wildlife. By using TRVs that were estimated to be appropriate from our review, and the actual concentrations at which the PBDEs occur in fish from DCL and TL, a screening-level risk assessment was conducted for fish-eating wildlife.

2 Data Collection and Analysis

2.1 Selection of Toxicity Data

Toxicity threshold values for the PBDEs, expressed as no observed adverse effect levels (NOAEL), or lowest observed adverse effect levels (LOAEL), and based on several endpoints, were derived from toxicity studies that were performed on birds and mammals. These threshold values were determined for wildlife and were based on concentrations of PBDEs in wildlife feed and tissues (liver or eggs). Dietary-based data were converted to average daily intake (ADI) values and were expressed as units of mg ΣPBDE/kg body mass (bm)/day (mg ΣPBDE/kg (bm)/day). ADI values were calculated from body masses and rates of ingestion by the selected surrogate species.

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 was 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); (6) studies that included only acute exposures were not accepted, because they provided no data on chronic, or sub-lethal effects on wildlife.

2.2 Approaches to Develop TRVs

Several approaches can be used for deriving TRVs, such as critical study approach (CSA), averaging method, meta-analyses, or species sensitivity distribution approaches. CSA is based on selecting a critical study for deriving recommended TRVs, which involves finding a technically defensible and definitive study from which a toxicity threshold is bracketed by experimental doses (Blankenship et al. 2008; USEPA 2003). CSA is the primary method for assessing risk to wildlife and for deriving criteria for protection of wildlife (CCME 1998; USEPA 1995a, b, 2003, 2005; Sample and Suter 1993). In the present study, CSA was used to derive TRVs for wildlife that is exposed to PBDEs. The TRVs were derived by using uncertainty factors (UFs), applied either to the LOAEL or to the NOAEL that had been derived in the critical studies. UFs were assigned using the guidance given in Technical Support Document (TSD) for Wildlife Criteria for the Great Lakes Water Quality Initiative (GLWQI) (USEPA 1995c), and GLWQI Criteria Documents for the Protection of Wildlife (USEPA 1995b). Three sources of uncertainty were considered in assigning UF values: (1) interspecies differences in toxicological sensitivity (UF_A), (2) sub-chronic 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 (USEPA 1995c; Newsted et al. 2005). Because of the limit on this method and the quantity of data, some other sources of uncertainty were not considered in the present study, such as sensitivity differences between adult and embryo stages, and the differences in metabolic rate, caloric content of food, and food assimilation efficiency between laboratory and wild species.

Three other methods have been used to develop TRVs, but none of these were applied to derive a TRV for PBDEs in this study. In applying the average method, it must be assumed that each evaluated study is of equal quality and should be weighted equally to calculate an average of the most representative studies. Meta-analysis is a statistical procedure that integrates the results of several independent studies that have compatible data (Egger et al. 1997). Well-conducted meta-analysis can enhance accuracy when estimating levels at which treatments will produce an effect. However, in this approach it is required that the studies utilized were conducted by using standard toxicity protocols (i.e., methodology, exposure routes, exposure duration, etc.). Because the quality of different studies are generally different, the utility of the meta-analysis approach is limited. In the present study, the meta-analysis approach was not used because the toxicity data utilized different effective endpoints. Species sensitivity analysis (SSA) is another approach that has been used to assess ecological risks and to derive environmental quality criteria (Caldwell et al. 2008; Hall et al. 1998; Solomon et al. 1996; Stephan et al. 1985; Wu and Li 2012; Zhang et al. 2012, 2013). A statistical distribution can be developed with SSA, which represents the variation in sensitivity that species display to a contaminant (Posthuma et al. 2002). Because there were too few studies that have been performed on PBDE toxicity on wildlife, none of these three aforementioned methods were used in the present study.

2.3 Sample Collection and Quantification of PBDEs

Samples of fish muscle (n = 116) from seven species (viz., crucian carp (*Carassius cuvieri*), topmouth culter (*Erythroculter ilishaeformis*), mongolian culter (*Erythroculter mongolicus*), common carp (*Cyprinus carpio*), bighead carp (*Aristichthys nobilis*), yellow catfish (*Pelteobagrus fulvidraco*), and grass carp (*Ctenopharyngodon idella*)) were collected from TL in 2009. In DCL, samples of fish muscle (n = 76) from five species (viz., crucian carp (*Carassius cuvieri*), sharpbelly (*Hemiculter leucisculus*), silver carp (*Hypophthalmichthys molitrix*), common carp (*Cyprinus carpio*), and bighead carp (*Aristichthys nobilis*)) were collected in 2010. After collection, samples were stored in polyethylene bags, were kept on ice, and then transported immediately to the laboratory, where they were stored at −20 °C until analysis.

PBDE residues in collected fishes were analyzed by using previously described methods (Guo et al. 2007). Briefly, samples were homogenized and Soxhlet extracted with 200 mL of 50% acetone in n-hexane (v/v). Lipid mass was measured for each sample, and then lipids were removed by gel permeation chromatography. The subsequent cleanup and fractionation were performed on multilayer alumina/silica columns. Analysis of PBDEs were performed with a Shimadzu Model 2010 gas chromatograph (GC), coupled with a Model QP 2010 mass spectrometer (MS) (Shimadzu, Japan). MS detections were achieved by using negative chemical ionization set to a selective ion-monitoring mode. Separation for quantification was achieved by using a DB-XLB (30 m × 0.25 mm i.d., 0.25 μm film thickness) capillary column for the lesser-brominated congeners (BDE-28 to BDE-183) and a CP-Sil 13 CB (12.5 m × 0.25 mm i.d., 0.2 μm film thickness) capillary column for the higher brominated congeners (including BDE 203, 206, 207, 208, and 209). Quantification of PBDEs was based on an internal calibration procedure (internal standard ^{13}C-PCB208).

Because wildlife consume whole fish and concentrations in muscle do not accurately represent whole-body concentrations of the PBDEs, whole-body concentrations were estimated for the samples for which only muscle analyses were conducted. According to the results of previous studies, ratios of concentrations in muscle to those in other tissues were 1:1–1:6 (Guo et al. 2008), and the muscle to whole-body concentration ratio was 1:5 (Xian et al. 2008). In our present assessment, the muscle concentration of PBDEs was multiplied by a conversion factor of 5.0, and was converted to concentration in whole-body fish.

3 Derivation of TRVs of PBDEs for Aquatic Mammals

3.1 Toxicity of DE-71 to Mink (*Mustela vison*)

The toxicity of DE-71 to mink was evaluated with emphasis on the following toxic endpoints: reproductive performance and development (Zhang et al. 2009), immunotoxicity (Martin et al. 2007), neurochemistry (Bull et al. 2007), bioaccumulation

and maternal transfer (Zhang et al. 2008). When reproduction and development were evaluated (Zhang et al. 2009), first-year female mink were exposed to DE-71 at four dietary doses (viz., 0, 0.1, 0.5 or 2.5 mg DE-71/kg (wet mass, wm)). Test females were mated to untreated males, and the 3-week old progeny from each group were maintained on their respective treatment diets with the females until they reached approximately 33 weeks of age. Reproductive parameters investigated in this study included the number of females bred, the number that whelped, litter size, and survivability of kits from birth to 6 weeks of age. In addition, circulating thyroid hormone levels and hepatic enzyme activity were investigated. Dams exposed to 2.5 mg DE-71/kg (wm) failed to whelp, while whelping rates were unaffected at doses of 0.1 or 0.5 mg DE-71/kg (wm) in the diet. Implantation sites were observed in 70% of dams fed 2.5 mg DE-71/kg (wm), and one dam in this group was in the latter stages of fetal resorption at the time of examination. It was concluded that failure to whelp largely resulted from toxicity to the fetus (Zhang et al. 2009).

Growth of kits at 6 weeks of age was not affected by any dose of DE-71, and no effects were observed on organ somatic index (Zhang et al. 2009). For juveniles, there were no statistically significant differences in growth between treated and control mink. Liver mass and the liver somatic index of juveniles was significantly greater for mink fed 0.5 mg DE-71/kg (wm) in the diet. Juveniles were more sensitive to the effects of DE-71 on the liver than were their dams, until a dose of 2.5 mg DE-71/kg (wm) was reached. Although treatment-related reproductive and developmental effects were observed, no clear dose-response relationship was evident for either endpoint monitored in this study (Zhang et al. 2009).

A dose-dependent negative relationship was observed between the concentration of DE-71 in the diet and concentration of total triiodothyroxine (TT3) in blood plasma of all age groups, and the effect was statistically significant at 2.5 mg DE-71/kg (wm) in dams and at 0.5 mg DE-71/kg (wm) in juveniles (Zhang et al. 2009). The effect on thyroid hormone level in mink was more sensitive to DE-71 than was reproduction. Therefore, dietary NOAEL and LOAEL values were respectively based on reduced circulating levels of T3 of the 0.1 and 0.5 mg DE-71/kg (wm) dose levels. Corresponding threshold concentrations in liver of exposed juvenile mink were 1.2 and 6.4 mg DE-71/kg (lipid mass, lm), respectively. Treated dams had similar concentrations of PBDEs in the livers of the corresponding treated juveniles, although concentrations in kits were less than those of dams (Zhang et al. 2009).

Moreover, ethoxyresorufin *O*-deethylase (EROD) activity was also determined in this reproductive toxicity study (Zhang et al. 2009). EROD activity was significantly induced in a dose-dependent manner in the livers of juveniles fed 0.1 and 0.5 mg DE-71/kg (wm) (Zhang et al. 2009). In juvenile mink, EROD activity was positively associated with the liver somatic index. EROD activity was more sensitive endpoint. Therefore, 0.1 mg DE-71/kg (wm) dose was considered to be the dietary LOAEL.

The dietary NOAEL and LOAEL mentioned above needs to be converted to ADI-based concentrations. Daily dietary intake of DE-71 in adult females prior to breeding was estimated to be 0, 0.01, 0.05, and 0.25 mg DE-71/kg body mass (bm)/day for the dietary doses of 0, 0.1, 0.5 and 2.5 mg DE-71/kg (wm), respectively

(Bull et al. 2007). Thus, dietary-based NOAEL and LOAEL were 0.01 and 0.05 mg DE-71/kg bm/day for decreased circulating TT3, and dietary-based LOAEL was 0.01 mg DE-71/kg bm/day for induction of EROD activity.

The immunotoxicity of DE-71 to ranch-grown sub-adult (20 weeks old) mink was studied, in which mink were exposed to one of four doses of 0, 1, 10 or 100 mg DE-71/kg (wm) for 9 weeks (Martin et al. 2007). Because unexpected growth effects were encountered in mink exposed to 100 mg DE-71/kg (wm), this group was switched to a diet containing 5 mg DE-71/kg (wm) for the duration of their exposure. Immune function was monitored by measuring the response of antibodies to T lymphocyte-dependent antigen, levels of keyhole limpet hemocyanin (KLH) conjugated to dinitrophenol (DNP), running phytohemagglutinin (PHA)-induced skin test and measuring hematological parameters. No significant differences were observed in the PHA-induced skin response for any of the groups; however, mink fed 5 or 10 mg DE-71/kg (wm) exhibited significantly greater anti-DNP-KLH antibody production relative to the control mink. A moderate degree of periarteriolar lymphatic sheath development and occasional germinal centers were scattered throughout the spleens from control animals and mink exposed to 1 mg DE-71/kg (wm). Significantly, greater development of germinal center and B-cell hyperplasia were observed in spleens of mink exposed to 10 mg DE-71/kg (wm). The number of germinal centers was positively associated with the liver concentration of ∑PBDEs. The hematocrit in mink from the two greatest exposure groups (5 and 10 mg DE-71/kg, (wm)) was significantly less than that of the control and 1 mg DE-71/kg (wm) mink groups, and was negatively correlated with ∑PBDE levels in the liver. The percentage of neutrophils were significantly greater, while percentages of lymphocytes were significantly less in mink fed 5 or 10 mg of DE-71/kg (wm). EROD activity in liver microsomes was significantly induced in livers of mink fed DE-71. EROD was induced approximately 20-fold greater in liver microsomes from mink fed 1 mg DE-71/kg (wm) than in the controls, 22-fold greater in the 5 mg DE-71/kg (wm) group, and least in the livers of mink fed 10 mg DE-71/kg (wm). For all treatments, there was a significant and positive association between EROD activity and the ∑PBDE concentration in livers. Based on these results, the lowest dose (viz., 1 mg DE-71/kg (wm)) was selected as the dietary LOAEL for effects on immune function, and the estimated daily intake dose was 0.079 mg DE-71/kg bm/day. The corresponding tissue-based LOAEL was 5.067 mg DE-71/kg (lm).

Effects of DE-71 on the nervous system, focusing on cholinergic parameters in the cerebral cortex of ranch mink have been studied (Bull et al. 2007). Adult female mink were exposed via the diet during and exposure extended to *in utero*, lactational, and subsequent dietary exposure of the offspring. Dietary doses to adult females were 0, 0.1, 0.5 or 2.5 mg DE-71/kg (wm). No significant effects of exposure to DE-71 were observed on the following: muscarinic acetylcholine receptor (mAChR) or nicotinic acetylcholine receptor (nAChR) binding activity, cholinesterase (ChE) activity, or acetylcholine (ACh) concentration in the cerebral cortex of adult females, 6-week-old kits, or 27-week-old juveniles. The ChE activity in blood plasma of adult females fed 2.5 mg DE-71/kg (wm) was threefold greater than for all other treatments. However, because ChE in blood plasma is synthesized in liver,

the increased ChE activity in blood plasma may have resulted from the effects of DE-71 on liver function, rather than on neurochemistry, a conclusion that is supported by other evidence (Bull et al. 2007). A NOAEL of 2.5 mg DE-71/kg (wm) was inferred for neurotoxicity in mink. The corresponding concentration of PBDEs in cerebral cortex of adult female mink was 88 ng PBDEs/g (wm).

Several measurement endpoints for effects of DE-71 on mink were examined in the studies reviewed above, and we needed to identify the most appropriate endpoint among these for deriving the TRV. EROD activity in liver is a common biomarker of exposure to environmental inducers of CYP1A1, such as those for which effects are modulated through the AhR. However, the PBDEs have been confirmed to be non-dioxin-like compounds and are incapable of inducing EROD activity (Sanders et al. 2005; Peters et al. 2004; Talsness 2008). Dioxin-like toxicity observed in studies of commercial mixtures of PBDEs under laboratory conditions were attributed to potential contaminants such as brominated biphenyls, dioxins or dibenzofurans in mixtures of PBDEs (Brown et al. 2004; Sanders et al. 2005). Effects of the polybrominated dibenzo-*p*-dioxins and dibenzofurans (PBDD/Fs) are similar to those of their chlorinated analogues (polychlorinated dibenzo-*p*-dioxins and dibenzofurans, PCDD/Fs) (Birnbaum et al. 2003; Behnisch et al. 2003). Thus, EROD activity in liver cannot be used as an appropriate critical effect endpoint for exposure to the PBDEs. Reduced concentrations of circulating TT3 was the second most sensitive endpoint for mink exposed to the PBDEs (Table 1). Similar findings were also observed in studies, in which thyroid hormone concentrations were correlated with the PBDE levels of at least some congeners (Leijs et al. 2012; Tomy et al. 2004; Hallgren and Darnerud 2002). However, AhR-active compounds, such as 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) have also been shown to influence thyroid hormone metabolism (Leijs et al. 2012; Boas et al. 2006). Before toxicity values, based on thyroid effects, can be used for deriving TRVs, it is necessary to eliminate effects from any AhR-active contaminants or impurities in commercial mixtures of PBDEs. Seo et al. (1995) exposed weanling rats to TCDD by gestational and lactational pathways, and reported that changes in thyroid hormone status at weaning are not among the more sensitive effects of perinatal exposure to coplanar PCB congeners or TCDD. Similar results were observed in a toxicity study of ranch mink exposed to dietary TCDD (Martin et al. 2006); no effect was observed in the TT3 and free triiodothyronime (T3) concentrations in blood plasma of the kits. Results of mink acute toxicity studies indicated that the concentrations of bound and free T3 and thyroxine (T4) were lower in mink exposed to 2.5 μg TCDD/kg bm than those in controls (Hochstein et al. 1988). In another study, T4 concentrations in mink fed dietary 0.01 or 0.1 ng TCDD/g (and survived 125 days) were not significantly different from those of controls (Hochstein et al. 1998). Based on the feed consumption and body weight of mink after 1 week exposure (Hochstein et al. 1998), the two doses, 0.01 and 0.1 ng TCDD/g feed, can be converted to values of 0.534 and 4.91 ng TCDD/kg bm/day, respectively. To preclude immunological effects of brominated dioxins and furans as co-contaminants in diet, Martin et al. (2007) analyzed a sample of the DE-71 technical material, and none of the dioxin and furan isomers were detected (detection limits of most congeners, <0.03 ppb). In addition, Sanders et al. (2005) analyzed concentrations of PBDDs and PBDFs in commercial DE-71 and

Table 1 NOAEL and LOAEL values for DE-71 in laboratory studies with mink

Effective endpoint	PBDEs concentration	Reference
Decreased TT3 concentration in plasma		
Dietary NOAEL	0.1 mg/kg (wm)	Zhang et al. (2009)
Dietary LOAEL	0.5 mg/kg (wm)	
Daily dose NOAEL	0.01 mg/kg (bm)/day	
Daily dose LOAEL	0.05 mg/kg (bm)/day	
Liver NOAEL	1.2 mg/kg (lm)	
Liver LOAEL	6.4 mg/kg (lm)	
EROD activity in liver		
Dietary LOAEL	0.1 mg/kg (wm)	Zhang et al. (2009)
Daily dose LOAEL	0.01 mg/kg (bm)/day	
Liver LOAEL	1.2 mg/kg (lm)	
Immunotoxicity effect		
Dietary LOAEL	1 mg/kg (wm)	Martin et al. (2007)
Daily dose LOAEL	0.079 mg/kg (bm)/day	
Liver LOAEL	5.067 mg/kg (lm)	
Neurotoxicity parameters in cerebral cortex		
Dietary NOAEL	2.5 mg/kg (wm)	Bull et al. (2007)
Daily dose NOAEL	0.25 mg/kg (bm)/day	
Cerebral cortex NOAEL	88 ng/g (wm)	

NOAEL no observed adverse effects level, *LOAEL* lowest observed adverse effects level, *bm* body mass, *PBDEs* polybrominated diphenyl ethers, *TT3* total triiodothyroxine, *wm* wet mass, *lm* lipid mass

determined the total concentrations of them to be 72.13 ng/g. Hanari et al. (2006) also measured PBDF levels in DE-71, and found 257 ng/g, while PBDD congeners were not detected (<100–200 ng/g). Assuming the total concentration of PBDD/Fs in DE-71 to be 257 ng/g, the concentration of PBDD/Fs in feed for 0.1 and 0.5 mg DE-71/kg (wm) treatments would be 7.213 and 36.065 pg PBDD/Fs/kg feed, respectively. The corresponding average daily dietary intake of PBDD/Fs would be 0.7213 and 3.6065 pg PBDD/Fs/kg (bm)/day. These doses of PBDD/Fs in the feed were several orders of magnitude less than the toxicity threshold values reported above (Hochstein et al. 1998). Thus, it can be assumed that the observed effects on thyroid hormone in mink (Zhang et al. 2009) were attributed primarily to the effects of PBDE exposure. Thyroid hormones in mammals play critical roles in reproductive physiology, cellular differentiation, growth and metabolic regulation. It is appropriate to derive TRVs for PBDEs by using the toxicity threshold values that are based on effects on thyroid hormone.

3.2 Recommended TRVs of PBDEs for Aquatic Mammals

Effects of PBDE exposure on mink from several toxicity studies were compared in Table 1. With a reasonable assumption that the observed effects on thyroid hormone in mink were attributed primarily to PBDEs, a NOAEL of 0.01 mg DE-71/kg (bm)/day,

based on effects of thyroid hormone in mink (Zhang et al. 2009), was used to derive the dietary-based TRV. According to the TSD for Wildlife Criteria for the GLWQI (USEPA 1995c) and GLWQI Criteria Documents for the Protection of Wildlife (USEPA 1995b), uncertainty factors from three sources were applied to derive the TRVs for effects of PBDEs on mink. The first source of uncertainty was associated with interspecies extrapolation. Mink were the only piscivorous mammals for which PBDE effects were reported, which prevented determining interspecies sensitivity to PBDEs. However, as fish-eating wildlife species, mink are capable of great trophic magnification, thereby rendering themselves more susceptible to accumulating persistent organic pollutants (Heaton et al. 1995; Tillitt et al. 1995). Mink is often used as a representative wildlife criteria species, and as a relevant animal model for environmental risk assessment of aquatic wildlife (Blankenship et al. 2008; USEPA 1995c; Giesy and Kannan 1998; Millsap et al. 2004). Thus, using mink as a surrogate species to derive TRV values is reasonable for protecting other aquatic mammals against potentially hazardous PBDEs exposure. In addition, comparing the threshold doses given for the present study to results from rodent studies shows that DE-71 caused effects on reproduction and thyroid in mink at concentrations lower than those that caused similar effects in rodents (Zhang et al. 2009; Talsness et al. 2008; Zhou et al. 2002). This suggests that mink are more sensitive than rodents. Therefore, the UF_A was set to 1.0. The second source of uncertainty was associated with LOAEL to NOAEL extrapolation. Because the NOAEL was identified in the critical study, a UF_L greater than 1.0 is not necessary. The third source of uncertainty was associated with extrapolation from results of sub-chronic exposure to chronic exposure. In the study of effects on reproduction and development of mink (Zhang et al. 2009), adult females were exposed to DE-71 following dietary exposure through pregnancy and nursing, and the offspring were then exposed to DE-71 following *in utero*, lactational, and dietary exposure until 33 weeks of age. This exposure scenario is environmentally relevant, and covers the sensitive life stages of mink. The accumulation, disposition and metabolism behavior of DE-71 in mink (Zhang et al. 2008) indicated that the study of toxicity to mink could be accepted as a chronic exposure study. Thus, an UF of >1.0 was deemed to be unnecessary. An overall uncertainty factor of 1.0 was assigned to account for data gaps in deriving TRVs, by using the results of thyroid effects in mink (Zhang et al. 2009). The TRVs, based on dietary and ADI PBDEs concentrations, were 0.1 mg DE-71/kg feed (wm) and 0.01 mg DE-71/kg (bm)/day, respectively.

Threshold values from PBDE concentrations in liver, were used to derive a tissue-based TRV. Using the cumulative ingested dose of a chemical from consuming contaminated food (e.g., tissues) provides a better assessment of exposure, since it accounts for bioaccumulation and bioavailability. Moreover, monitoring studies usually detect concentrations of contaminants in specific body tissue of animals. Therefore, TRVs that are based on concentrations of toxicants in tissues of wildlife are effective for protecting wildlife from the hazards of pollutants exposure. On the basis of results in thyroid hormone of mink (Zhang et al. 2009) and the overall UF value presented above, a tissue-based TRV for liver PBDEs concentration was derived to be 1.2 mg/kg (lm).

4 Derivation of TRVs of PBDEs for Aquatic Birds

4.1 Toxicity of DE-71 to Birds

American Kestrel (*Falco sparverius*)

A series of toxicity studies were conducted in which the American kestrel (*Falco sparverius*) was fed DE-71, after which reproduction and development endpoints were measured (Fernie et al. 2008, 2009; Marteinson et al. 2010, 2011; Sullivan et al. 2010). Adult American kestrels were exposed through their diet to control, 0.3, or 1.6 mg DE-71/kg (wm) for 75 days each year (Fernie et al. 2008). Exposure began 21 days prior to pairing and continued through courtship, egg laying, and incubation periods in each year of 2005 and 2006, or until the first chick hatched. Kestrel nestlings were exposed only *in ovo* by direct maternal transfer to DE-71. Mean concentrations of ΣPBDE in eggs of kestrel were 3.01, 288.60, and 1,130.59 ng/g (wm), respectively, for the control, lesser and greater doses, and seven major congeners (BDE-28, -47, -100, -99, -154, -153, and -183) accounted for 69%, 96%, and 94% of ΣPBDE, respectively (Fernie et al. 2008).

During the experiment, the reproductive behavior of pairs (e.g., mate choice, pair-bonding, copulation and food transfer) were assessed (Fernie et al. 2008). Compared to controls, timing, duration and frequency of courtship behavior of kestrels were altered by DE-71 in both exposure groups. For the same exposure scenarios, reproductive success and egg quality of kestrels exposed to DE-71 were evaluated (Fernie et al. 2009). Compared to controls, egg laying was significantly delayed for kestrels from the two exposure groups, and was increasingly delayed with increasing concentrations of BDE-153, BDE-154, BDE-28 and BDE-17. Eggs laid by kestrels from the greater exposure group were significantly smaller and had lower mass than did those laid by kestrels from controls and the lower exposure groups. Moreover, adverse effects on eggshell qualities occurred that were attributed to DE-71 exposure (Fernie et al. 2009). Compared to controls, the eggshells from the greater exposure were significantly thinner, and those from the lesser exposure had significantly lower mass. Eggshell thickness was significantly and negatively associated with the measured PBDE concentration. Poorer fertility, hatching, and fledging success were observed for pairs of kestrels fed the greater dose compared to control pairs, and fledging success was modestly reduced as concentrations of BDE-153 increased.

Retinol is involved in the reproduction and development of birds. Thus, concentrations of retinol in blood plasma were measured in adult American kestrels and nestlings exposed to DE-71 (Sullivan et al. 2010). Concentrations of retinol in plasma of adult females fed the lesser dose were significantly less than those in the controls, and were negatively correlated with *in ovo* BDE-153 concentrations. No significant effect on the plasma retinol level of the adult males was observed, but the retinol concentrations of the males exposed to greater dose were negatively correlated with the ΣPBDE and BDE-100 concentrations. For nestlings exposed to lesser

in ovo concentrations of DE-71, concentrations of retinol in blood plasma were significantly less than those from control nestlings, but were not correlated with the *in ovo* PBDE concentrations.

In a further study, the reproductive success and behavior of male American kestrels exposed to DE-71 as embryos were assessed (Marteinson et al. 2010). The F1 progeny of the F0 kestrels exposed via the diet (Fernie et al. 2008) were never exposed directly via the diet, but only during the 28-days embryonic period via direct maternal transfer to the egg. At 1 year of age, male kestrels that had been exposed *in ovo* to three exposures were paired with unexposed females. Throughout courtship, there were fewer copulations by all *in ovo* exposed males and fewer male-calls made by greater-exposure males, and which were negatively associated with the males' embryonic exposure to concentrations of ΣPBDEs as well as individual congeners. Compared to controls, the greater-exposure males spent less time in their nest boxes, and the amount of time decreased significantly with increasing embryonic exposure to BDE-99 and -100. Moreover, 43% of female kestrels from greater -exposure pairs failed to lay eggs, while all other pairs laid complete clutches. The female kestrels paired with males exposed *in ovo* to either of two doses of DE-71 laid significantly smaller clutches and smaller eggs, and these eggs were less fertile, compared to controls. The fertility and number of eggs in clutches were both strongly and negatively correlated with *in ovo* exposure concentrations of ΣPBDEs, as was the individual congeners. All of these reproductive parameters were influenced by the frequencies of courtship behaviors from males.

When F1 progeny of dietary exposed F0 individuals (Fernie et al. 2008) were used, effects of embryonic exposure to DE-71 on the male kestrel reproductive tract and associated endocrinology were assessed (Marteinson et al. 2011). At 1 year of age, the *in ovo*-exposed male kestrels in the three exposure groups were paired with unexposed females and allowed to complete one reproductive cycle (Marteinson et al. 2011). One year later, males in the unpaired state were euthanized at 2 years of age during the fertile period, and concentrations of thyroid hormone and testosterone in blood plasma, sperm numbers and testis mass and histology were assessed. Lower testosterone concentrations appeared in blood plasma of males exposed to the greater concentration of DE-71 at the time the first egg was laid by females. However, there were no alterations observed for thyroid function in these adult kestrels during the breeding period, and concentrations of testosterone and thyroid hormone were not associated with *in ovo* exposure concentration of PBDEs. Compared to controls, numbers of sperm on the perivitelline layer of the first egg were greater for the two treatment groups. Males fed the greater dose had a greater gonadosomatic index and a heavier right testis than controls. Testis mass was positively associated with *in ovo* concentrations of ΣPBDEs and congeners, BDE-100, -47, -85, and -183. Males exposed to the greater concentration had more seminiferous tubules containing lumen than did controls, and in proportion to the total number of tubules, males exposed to the lesser concentration had more tubules within the lumen than did controls. Mean percent of tubules containing final spermatids were 43% for males exposed to the greater dose, while they were 53% and 59% in controls and those exposed to the lesser dose, respectively. The proportion of tubules containing

final spermatids positively correlated with *in ovo* exposure to BDE-47, -85, -49, and -28 (Marteinson et al. 2011). These changes in reproductive physiology of males American kestrels might reduce the reproductive success of these birds.

Based on reproduction, a LOAEL of 0.3 mg/kg (wm), as DE-71 was inferred. No food consumption or body mass data were reported for the adult birds in these dietary exposure studies. By assuming a body mass of 0.119 kg and a food ingestion rate of 0.022 kg/day (geometric means) (Dunning 1993; Yáñez et al. 1980; USEPA 1993), a dietary-based LOAEL value of 0.055 mg DE-71/kg (bm)/day was calculated. The corresponding concentration of ΣPBDEs in kestrel eggs from the lesser-exposure group was considered to be the tissue-based LOAEL in eggs; this LOAEL value was 288.6 ng ΣPBDEs/g (wm).

Because DE-71 is a commercial PBDE mixture with the presence of some impurities (Hanari et al. 2006), it is important to preclude effects caused by co-contaminants in the toxicity studies to ensure that the TRV accurately and reasonably reflects only the effect of PBDEs on animals. It should be noted that none of the 11 brominated dioxins and furans was detected in the DE-71 mixture that was used in these toxicity studies (Fernie et al. 2008). Thus, the effect of dioxin-like compounds as impurities in the DE-71 mixture was precluded for these studies. Unfortunately, total-α-hexabromocyclododecane (HBCD) was detected in the eggs of kestrels exposed to DE-71 by diet, and its presence in the diet of the birds occurred unintentionally (Fernie et al. 2009; Sullivan et al. 2010). Concentrations of HBCD in kestrel eggs from the controls and two treatment groups of 0.3 or 1.6 mg DE-71/kg (wm) were 0.002, 3.27, and 15.61 ng/g (wm), respectively (Fernie et al. 2009). In those toxicity studies reviewed above, thickness of eggshells of kestrels was inversely proportional to dose of HBCD in the diet (Fernie et al. 2009). Concentrations of retinol in blood plasma of adult males (Sullivan et al. 2010), and frequency of courtship behaviors, clutch size and fertility in kestrels exposed *in ovo* by direct maternal transfer (Marteinson et al. 2010) were negatively correlated with *in ovo* concentrations of HBCD. However, in the study with only HBCD exposure, eggshell thickness, egg fertility and reproductive success (Fernie et al. 2011), and copulation frequency (Marteinson et al. 2012) were not affected and not correlated with *in ovo* concentrations of HBCD. The *in ovo* concentration of HBCD in kestrels exposed to the lesser concentration of DE-71 in previous studies was approximately 50 times less than the HBCD concentration (163.5 ng/g (wm)) in the study, in which eggs were exposed to HBCD only. Thus, in the present assessment, the effects from HBCD can be precluded, and it is assumed that the reproductive effects in kestrels were attributed primarily to PBDEs.

Egg Injection Studies with Birds

There are several possible routes of exposure when performing toxicity studies. These include oral, subcutaneous injection, dermal, inhalation, and egg injection. When screening toxicity data for deriving wildlife TRVs, results from oral-administration exposures are preferred (Stanton et al. 2010; USEPA 2005).

For the study of embryo development and nestling growth, the common method of exposure is via the diet of adult females with maternal transfer to eggs. However, dietary exposure of adult female birds may induce changes in their food intake and incubation behaviors, and further affect growth of both embryo and nestling unintentionally. Direct injection of a contaminant into an egg avoids many of these issues and allows one to directly assess the contaminant's effect on embryo development (Winter et al. 2013). When there is sufficient information to show that the maternally transferred concentration elicits comparable toxicity to that from egg injection, then the results from egg injection studies can be used to derive TRVs (USEPA 2005).

The effects of PBDE injected into eggs were tested in several toxicity studies. A series of egg injection studies were conducted to evaluate effects on growth (Fernie et al. 2006), immunomodulation (Fernie et al. 2005a), thyroid, retinol and oxidative stress (Fernie et al. 2005b) in American kestrels. In these studies, the chemical was injected into the air cell of eggs with safflower oil (control group) or BDE-47, -99, -100, and -153 dissolved in safflower oil (18.7 μg ΣPBDEs/egg (wm), or 1.5 μg ΣPBDEs/g egg (wm)) after the eggs had been incubated for 19 days. Nestlings were dosed daily via oral gavage with the same mixture of PBDEs (15.6 ng/g bm/day) for 29 days. The relative concentrations of BDE congeners measured in PBDE mixtures were as follows: 56.4% (BDE-47); 27.2% (BDE-99); 24.8% (BDE-100); and 0.6% (BDE-153); these proportions approximated current concentrations in Great Lakes herring gulls. Nestlings exposed to PBDEs gained weight more quickly and ate more food than did the controls, and food consumption was positively associated with concentrations of ΣPBDEs in their bodies, especially for BDE-100 (Fernie et al. 2006). A 14% greater response to PHA was observed that might be biologically important in PBDE-exposed birds (Fernie et al. 2005a). A lower antibody-mediated response was observed and was positively associated with concentrations of BDE-183. There were also histological changes in spleen, bursa and thymus, and negative associations between the spleen somatic index and ΣPBDEs, and the bursa somatic index and BDE-47 (Fernie et al. 2005a). Concentrations of plasma T4, plasma retinol, and hepatic retinol and retinyl palmitate concentrations were less in birds exposed to the PBDEs, and the first three parameters were negatively correlated with concentrations of the following individual PBDE congeners: BDE-47, BDE-99, and BDE-100 (Fernie et al. 2005b). Hepatic oxidative stress was also induced in PBDEs-exposed birds. No difference was observed in T3 concentrations between groups. However, only one dose among these three studies was the same, and a threshold of toxicity could not be obtained for derivation of TRVs.

Chickens (*Gallus gallus*), mallards (*Anas platyrhynchos*) and American kestrels were exposed to DE-71 (0.01, 0.1, 1, 10, or 20 μg DE-71/g egg (wm)) by air cell injection, and embryonic survival, piping, hatching success, and sublethal biochemical, endocrine, and histological endpoints were measured (McKernan et al. 2009). Dose-dependent decreases in piping and hatching success of kestrels were observed at doses of 1, 10, and 20 μg DE-71/g egg (wm), with significant differences occurring at 10 and 20 μg DE-71/g egg (wm), while there were no effects on survival,

Table 2 NOAEL and LOAEL values for DE-71 in laboratory studies with American kestrels

Effective endpoint	PBDEs concentration	Reference
Reproductive effects in multi-generations		
Dietary LOAEL	0.3 mg/kg (wm)	Fernie et al. (2008, 2009), Marteinson et al. (2010, 2011), Sullivan et al. (2010)
Daily dose LOAEL	0.055 mg DE-71/kg (bm)/day	
Egg LOAEL	288.6 ng ΣPBDEs/g (wm)	
Piping and hatching success		
Egg-injection LOAEL	1,800 ng ΣPBDEs/g (wm)	McKernan et al. (2009)
Egg-injection NOAEL	180 ng ΣPBDEs/g (wm)	

piping, and hatching success of chickens or mallards at any PBDEs dose. American kestrels were more sensitive to effects of DE-71 on piping and hatching success than were mallards and chickens. Thus, tissue-based NOAEL and LOAEL in egg were 1 and 10 μg DE-71/g egg (wm), respectively.

However, not all the PBDEs in these injected doses were responsible for the effects on piping and hatching success (McKernan et al. 2009). Uptake tests with PBDEs into kestrel eggs indicated that 18% of the administered dose was absorbed into kestrel embryos (McKernan et al. 2009, 2010). Based on this information, the tissue-based NOAEL and LOAEL were converted to 0.18 and 1.8 μg DE-71/g egg (wm), respectively. The NOAEL of 0.18 μg DE-71/g egg (wm), based on an egg injection study, is comparable to the LOAEL value of 0.29 μg ΣPBDEs/g (wm) in embryonic exposure of PBDEs transferred from mother birds. The hatching success of kestrel eggs dosed with 0.18 μg DE-71/g via injection is also comparable to that of lesser doses in eggs in a maternally-transferred exposure study (i.e., 60% vs. 56%) (Fernie et al. 2009). In McKernan et al. (2009) study, EROD activity was also assayed, and the EROD activity was induced only in chicken hatchlings at doses of 1, 10, or 20 μg DE-71/g egg (wm). However, there wasn't sufficient information from maternally-transferred exposure study to identify this results. Thus, the egg injection NOAEL value from the kestrel study, based on piping and hatching success rather than the EROD activity, was used as the basis for deriving the tissue-based TRV for PBDE.

4.2 Recommended TRVs of PBDEs for Aquatic Birds

Several studies, in which DE-71 produced effects on birds, were systematically reviewed and analyzed (Table 2). The LOAEL, based on measurement endpoints involving reproduction, was 0.3 mg/kg (wm) (0.055 mg DE-71/kg (bm)/day) from a series of studies on American kestrels (Fernie et al. 2008, 2009; Marteinson et al. 2010, 2011; Sullivan et al. 2010), and was selected as the basis for calculating a dietary-based avian TRV. Three uncertainty factors were considered for use with this LOAEL: interspecies differences in sensitivity (UF_A), subchronic to chronic extrapolations (UF_S), and LOAEL to NOAEL extrapolations (UF_L). The American kestrel is a terrestrial ecosystem predator that has been used as a model species for

investigating the effects of methylmercury (Albers et al. 2007; Fallacara et al. 2011) and several organic pollutants on various measurement endpoints (Marteinson 2011).

According to results of an egg injection study, American kestrel was more sensitive to PBDEs injected into eggs than were chicken or mallard (McKernan et al. 2009). Therefore, a UF_A of 1.0 was selected. In the series of studies on kestrels, multi-generational effects of reproduction were evaluated, and the effects from maternal transferred exposure were also investigated. Duration of exposure of adult birds was 75 days in each year of 2005 and 2006, and the period throughout pairing, courtship, egg laying, and incubation, was a relative sensitive life stage. An uncertainty factor of 1.0 was assigned to account for subchronic to chronic extrapolations. Because a LOAEL value was identified rather than a NOAEL value was identified from these studies, a UF_L of greater than 1.0 was applied. According to the results of these studies, several reproductive parameters were affected, but compared to controls the decrease of eggshell thickness in the lesser-exposure kestrels was only approximately 1%, which was still far from the magnitude of eggshell thinning that would render kestrels incapable of maintaining a stable population (Fernie et al. 2009). The reduced hatching success in lesser-exposed kestrels was approximately 13% of that in controls. Thus, a UF_L of 3.0 was a reasonable value for the extrapolation. An overall UF of 3.0 was assigned to account for data gaps in deriving TRVs by using the results in kestrels. Thus, TRVs, based on diet and ADI, were 0.1 mg DE-71/kg (wm) and 0.018 mg DE-71/kg (bm)/day, respectively.

Using this overall UF of 3.0, the tissue-based LOAEL of 288.6 ng ΣPBDEs/g (wm) in eggs was converted to 96.2 ng ΣPBDEs/g (wm) as the corresponding dietary exposure NOAEL value in eggs. This dietary exposure NOAEL of 96.2 ng ΣPBDEs/g (wm), and the egg-injection NOAEL of 180 ng ΣPBDEs/g (wm) from the McKernan et al. study (2009) (Table 2) were both used to derive the tissue-based TRV in the egg for aquatic species in the present study, and the tissue-based TRV in the egg were estimated as the geometric mean of these two values. Therefore, the tissue-based TRV in the egg is 131.6 ng ΣPBDEs/g (wm). The lipid content in egg of American kestrel was not reported in the toxicity studies reviewed above. The concentrations of PBDEs based on wet weight and lipid weight of egg were reported in McKernan et al. study (2009), and a lipid content of 5.6% was estimated for kestrel egg. On this lipid weight basis, the tissue-based TRV was converted to 2.35 μg ΣPBDEs/g (lm).

To assess the possible risk posed by PBDEs to fish-eating wildlife in China, the dietary- and tissue-based TRVs for DE-71 were derived for protecting birds and mammals that might eat fish (Table 3). Information on the toxicity of DE-71 to mink and American kestrel was available. Therefore, these two species were used as the surrogates for other birds and mammals, respectively. These TRVs provide points of reference for concentrations of PBDEs measured in fish and fish-eating wildlife, and can be used in the tissue residue approach to ecological risk assessment. Threshold values, based on concentrations of PBDEs in diets were compared between mink and American kestrel (Fig. 1). The LOAEL, based on reproduction of American kestrel was comparable to the LOAEL value that was based on thyroid effects in mink. The ADI-based TRV, from effects on mink thyroid, is one fold less

Table 3 PBDE toxicity reference values (TRVs) for aquatic mammals and birds based on dietary, ADI, liver, and egg toxic doses

	Mammal			Bird		
	NOAEL	UF	TRV	LOAEL	UF	TRV
Dietary, mg DE-71/kg (wm)	0.1	1	0.1	0.3	3	0.1
ADI, mg DE-71/kg (bm)/day	0.01	1	0.01	0.055	3	0.018
Tissue, μg ΣPBDEs/g (lm)[a]	1.2	1	1.2	5.15[b]	3	2.35[c]

ADI average daily intake, *UF* uncertainty factor

[a] For mammals the tissue is liver, and the tissue for birds is egg

[b] This value was derived by using the LOAEL of 288.6 ng ΣPBDEs/g (wm) in eggs and the estimated lipid weight of 5.6%

[c] This TRV was estimated as the geometric mean of the egg-injection NOAEL of 180 ng ΣPBDEs/g (wm) and the dietary exposure NOAEL of 96.2 ng ΣPBDEs/g (wm), and the NOAEL of 96.2 ng ΣPBDEs/g (wm) was derived by LOAEL of 288.6 ng ΣPBDEs/g (wm) in eggs and the UF of 3.0

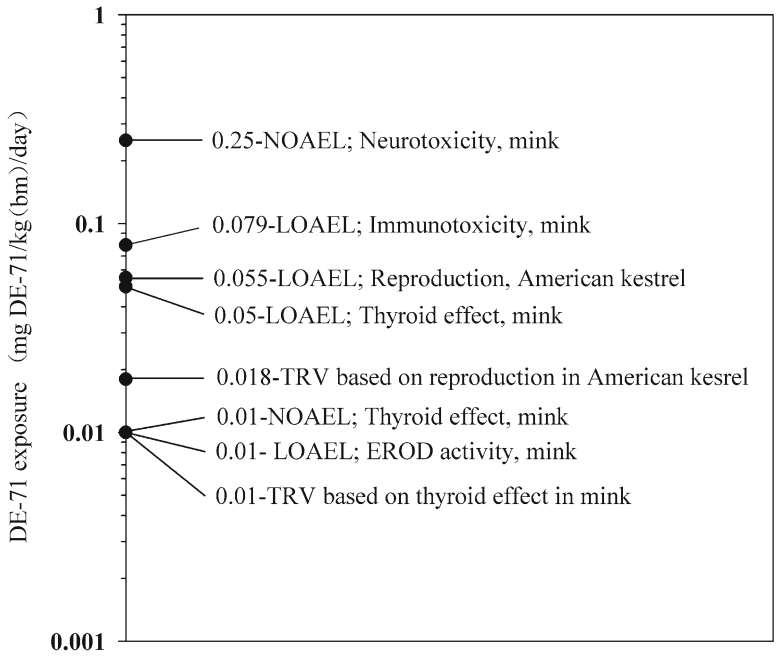

Fig. 1 Toxicity thresholds for wildlife exposed to DE-71 via diet; expressed as average daily intake (ADI). *NOAEL* no observed adverse effects level, *LOAEL* lowest observed adverse effects level, *bm* body mass. See Tables 1, 2, and 3 for data set

than the TRV that was based on reproduction in the American kestrel, which incorporated a uncertainty factor of 3.0. Therefore, the dietary-based TRV of 0.01 mg/kg (bm)/day (or 0.1 mg DE-71/kg (wm)) is appropriate to protect fish-eating wildlife.

5 Comparison to Ambient Tissue Concentrations

Concentrations of PBDEs in livers of aquatic mammals found dead were collected to examine the reasonableness of the threshold values derived in the present study. The geometric mean concentration of total PBDEs in livers of Eurasian otter found dead in England and Wales in 2010 was 50.56 ng/g (wm) with a range of 3.001–717.8 ng/g (wm), and the BDE-47, -153, -100, and -99 were the predominant congeners (Walker et al. 2012). Assuming a lipid content of 5% in livers of Eurasian otter (Kannan et al. 2000), the concentration of total PBDEs in livers of Eurasian otter would be 1,011 ng/g (lm) with a range of 60.02–14,356 ng/g (lm). The mean concentration of total PBDEs, in livers of adult female sea otters found freshly dead or dying between 1992 and 2002 along the central California coast, was 2,200 ng/g (lm) with a range of 10–26,800 ng/g (lm) (Kannan et al. 2007). The mean concentration of total PBDEs in liver of harbor porpoises stranded on the North Sea coast of Belgium between 1997 and 2000 was 2,290 ng/g (lm), with a range of 410–5,810 ng/g (lm) (Covaci et al. 2002). All concentrations of PBDEs in livers of mammals mentioned above were comparable with, or greater than the tissue-based TRV of 1,200 ng/g (lm) that was derived in the present study. The toxicological evidence of exposure to PBDEs in these mammals were uncertain, and links between liver PBDE concentrations and health effects in these mammals can't be established. However, the results indicated that the PBDEs in liver still may potentially affect health, which supports this tissue-based TRV.

Concentrations of PBDEs in wild aquatic biota, including invertebrates and fishes, were compiled and compared to the dietary-based TRV values to evaluate the current risk in the Chinese environment. Concentrations of total PBDEs in wild aquatic species from the Yangzi River (Gao et al. 2009b), Baiyangdian Lake (Hu et al. 2010), Bohai Bay (Wan et al. 2008), and a reservoir in Longtang Town (Wu et al. 2008) were available. The geometric mean of concentrations of PBDEs in several species was calculated and compared for each region. Concentrations of PBDEs in aquatic species from the first three regions were less than the dietary-based TRV of 100 ng DE-71/g (wm), while the geometric mean of concentration of PBDEs in dietary items from the reservoir in Longtang Town was much greater than this TRV value (Fig. 2). These results are consistent with the known contamination status of these water bodies. The reservoir in Longtang Town is known to be polluted by crude e-waste recycling activities (Wu et al. 2008). Thus, PBDEs in these Longtang Town reservoir aquatic species may be sufficiently concentrated to pose an adverse risk to the wild fish-eating birds and mammals.

Concentrations of PBDE in eggs from wild aquatic birds in China were compared to the tissue-based TRVs for egg (Fig. 3). All concentrations of total PBDEs in bird eggs collected from Hong Kong, Xiamen, Quanzhou (Lam et al. 2007), and the Yellow River Delta (Gao et al. 2009a) were lower than the tissue-based TRV of 2.35 μg ΣPBDEs/g (lm). Concentrations of PBDEs in bird eggs might not have affected reproductive function in birds collected from these regions. This comparison suggests that the TRV values reported in this study can serve as indicators for screening-level risk assessment of piscivorous species in other Chinese aquatic systems.

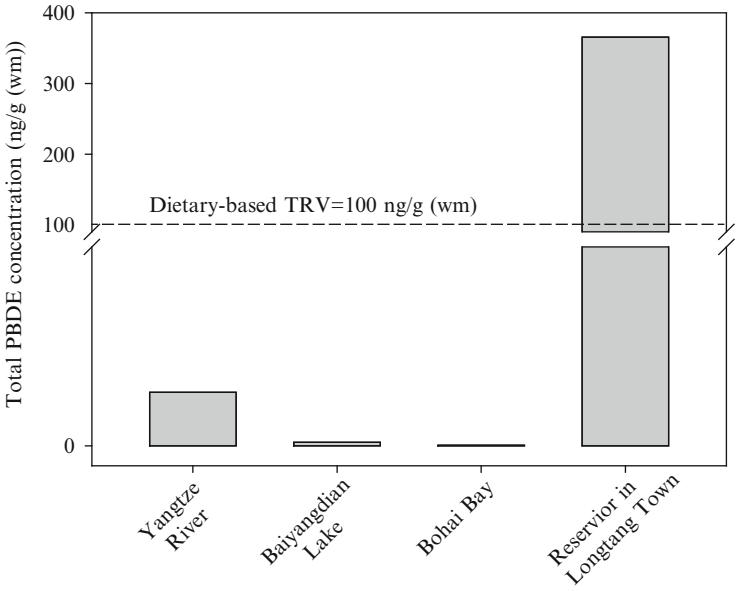

Fig. 2 Comparison of the dietary-based TRV to reported concentrations of PBDEs in wild aquatic species from different regions in China. *TRV* toxicity reference value, *wm* wet mass. Yangzi River (Gao et al. 2009b), Baiyangdian Lake (Hu et al. 2010), Bohai Bay (Wan et al. 2008), and Reservoir in Longtang Town (Wu et al. 2008)

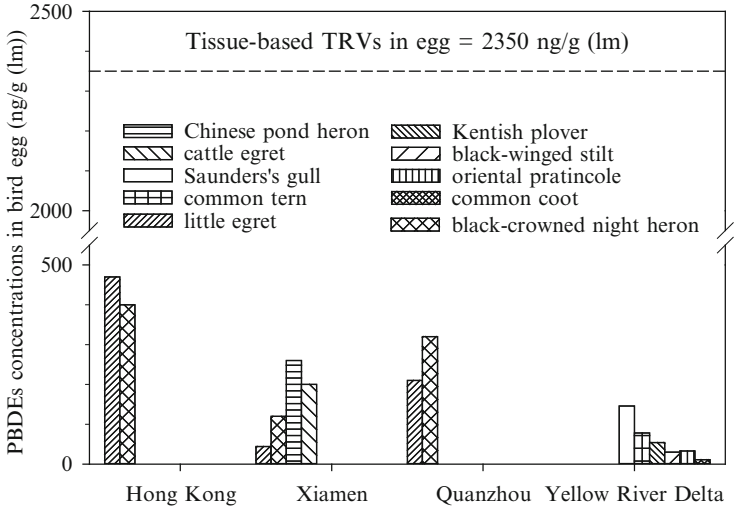

Fig. 3 Comparison of the tissue-based TRV in eggs to reported concentrations of PBDEs in wild aquatic avian species from different regions in China. *TRV* toxicity reference value, *wm* wet mass. Hong Kong, Xiamen, Quanzhou (Lam et al. 2007), and Yellow River Delta (Gao et al. 2009a)

6 Ecological Risk Assessment of PBDEs in DCL and TL

Relative congener concentration profiles of PBDEs in fishes from DCL and TL were similar to those found in DE-71; the dominant congener was BDE-47, followed by BDE-28, BDE-100, BDE-154, BDE-99, and BDE-153 (Fig. 4). The main difference in the congener profiles in fishes and DE-71 was the greater amount of BDE-17 and BDE-28 in wild fish. ΣPBDE concentrations (sum of the detected congeners) estimated in whole-body fish from DCL and TL ranged from 199 to 7,040 and 79 to 3,018 ng/kg (wm), respectively (Table 4 and Fig. 5). A screening-level risk assessment was conducted by comparing concentrations of ΣPBDE in fishes to the dietary-based TRV of 0.1 mg DE-71/kg (wm) presented in this report. Maximum concentrations of PBDEs in fishes from DCL and TL were two orders of magnitude less than the dietary-based TRV of 0.1 mg DE-71/kg (wm), which suggests that current PBDE levels would not exceed the threshold for adverse effects to piscivorous species in DCL and TL.

These criteria values for PBDEs can be used to protect wild piscivorous species and to assess ecological risk associated with increasing PBDEs concentrations in Chinese surface waters. Note that these criteria were derived from the toxicity of

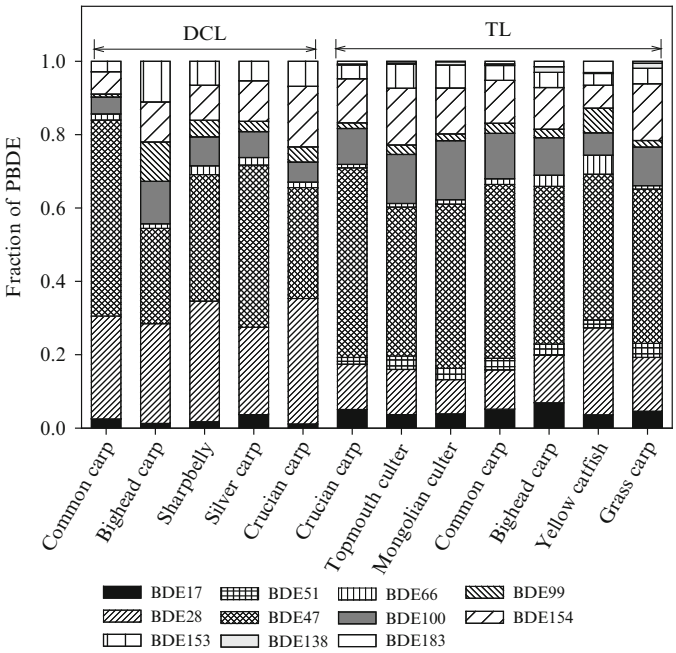

Fig. 4 Congener profile of PBDEs in wild fish from Dianchi Lake (DCL) and Tai Lake (TL)

Table 4 PBDE concentrations (ng/kg wm) in wild fish from two Chinese lakes (DCL and TL)

Species	Number of sample	PBDE concentration		
		Range	Median	Mean
TL				
Crucian carp (*Carassius cuvieri*)	26	147–1,591	554	571
Topmouth culter (*Erythroculter ilishaeformis*)	20	392–2,095	1,097	1,131
Mongolian culter (*Erythroculter mongolicus*)	4	983–3,018	1,285	1,643
Common carp (*Cyprinus carpio*)	24	128–887	405	456
Bighead carp (*Aristichthys nobilis*)	27	79–971	436	437
Yellow catfish (*Pelteobagrus fulvidraco*)	6	241–1,053	487	534
Grass carp (*Ctenopharyngodon idella*)	9	142–336	236	228
DCL				
Crucian carp (*Carassius cuvieri*)	19	199–1,042	390	423
Sharpbelly (*Hemiculter leucisculus*)	16	226–1,858	611	661
Silver carp (*Hypophthalmichthys molitrix*)	20	274–2,788	862	994
Common carp (*Cyprinus carpio*)	12	535–1,290	800	825
Bighead carp (*Aristichthys nobilis*)	9	331–7,040	643	1,691

DCL Dianchi Lake, *TL* Tai Lake

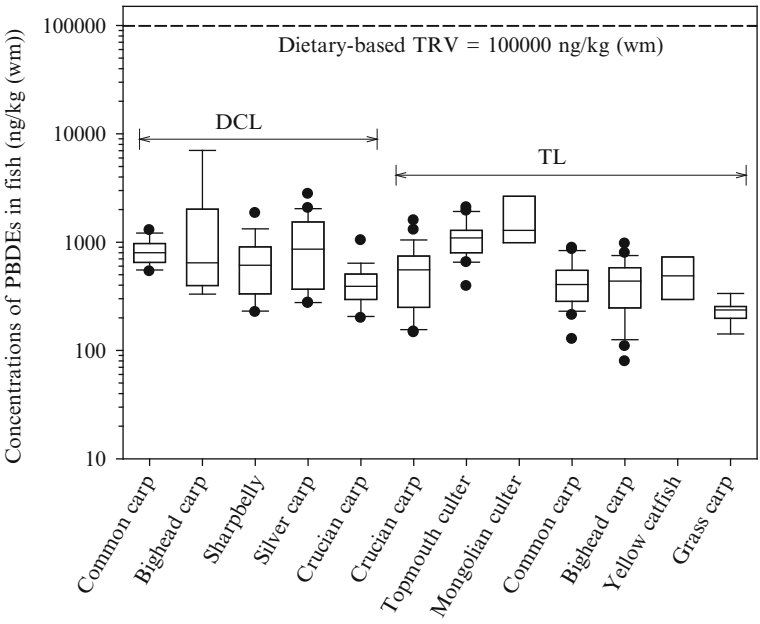

Fig. 5 Comparison of dietary-based TRV to concentrations of PBDEs in wild fish from Dianchi Lake (DCL) and Tai Lake (TL). *TRV* toxicity reference value, *wm* wet mass

DE-71, which is one of the commercial PBDE products and consists of several lesser brominated congeners. At present, though the lesser brominated diphenyl ethers were the predominant congeners in most aquatic organisms, some more highly brominated congeners, such as BDE-209, were also detected in tissues of species at higher trophic levels (Gao et al. 2009a; Lam et al. 2007). The criteria values presented in this study cannot provide information for these compounds. Concentrations of PBDEs in whole-body fish were derived by using a conversion factor and muscle concentrations, a degree of uncertainty was generated by the data gap related to the conversion of concentrations between tissues. Additionally, the diet of piscivorous birds and mammals consists of aquatic species from different trophic levels, and there are some interspecific differences in the structure of the diet. Thus, when these criteria values are used to assess ecological risk at a specific-site, information on food web structure must also be considered.

7 Summary

PBDEs are persistent organic pollutants, and have the capability to produce adverse effects on organisms. Aquatic piscivorous species at higher trophic levels have the greatest exposure risk. Information on the toxic potency of a commercial PBDE mixture, DE-71, to mink and American kestrel was reviewed, and dietary- and tissue-based TRVs were derived and evaluated for ecological risk assessment of aquatic piscivorous species inhabiting wetland areas in China. The effect on mink thyroid function was identified as the most appropriate and protective endpoint for deriving the TRVs for mammals. The TRV was based on dietary exposure, and was 0.1 mg DE-71/kg (wm) or 0.01 mg DE-71/kg (bm)/day (ADI); for liver of mammals, the TRV was 1.2 mg ΣPBDEs/kg (lm). For birds, reproductive effects on American kestrels were used to derive the TRVs, in which an overall UF of 3.0 was used. The TRV was based on dietary exposure, and was 0.1 mg DE-71/kg (wm) or 0.018 mg DE-71/kg (bm)/day (ADI); for eggs of birds, the TRV was 2.35 μg ΣPBDEs/g (lm). Reported concentrations of PBDEs in livers of aquatic mammals found dead, and in fish and bird eggs from Chinese wetland areas were compiled and compared to the corresponding criteria values. Results indicated that TRV values reported in this study can be used as indicators for screening-level risk assessment of piscivorous species in Chinese aquatic systems. Furthermore, based on monitoring concentrations of PBDEs in fishes from two lakes (DCL and TL) in China and the dietary-based TRV of 0.1 mg DE-71/kg (wm), a screening-level risk assessment of PBDEs was performed for predatory birds and mammals. The results suggest that concentrations of PBDEs in these two areas would not be expected to cause any adverse effects on the local fish-eating wild birds and mammals.

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