Environ Sci Pollut Res (2014) 21:168–176
DOI 10.1007/s11356-013-2110-2

ENVIRONMENTAL QUALITY BENCHMARKS FOR PROTECTING AQUATIC ECOSYSTEMS

Comparison of species sensitivity distributions for species from China and the USA

Xiaonan Wang · Zhenguang Yan · Zhengtao Liu · Cong Zhang · Weili Wang · Handong Li

Received: 2 May 2013 / Accepted: 28 August 2013 / Published online: 8 September 2013

Abstract China has recently commenced water quality criteria (WQC) research using the species sensitivity distribution (SSD) method; however, it is difficult to obtain sufficient native species toxicity data for thousands of contaminants. In this study, the feasibility of using non-native toxicity data in deriving native WQC was analyzed. We constructed SSDs based on acute toxicity data of species from China and the USA for eight priority pollutants, and compared the sensitivities of different taxonomic groups between the two countries. The results showed that the SSD method of log-logistic distribution fit the toxicity data of different taxa well. The comparison of sensitivity distribution and hazardous concentration for 5 % of the species and 50 % of the species showed that there was no significant difference between Chinese and American taxa. It could be feasible to use toxicity data from the USA to provide a temporary way to protect organisms in China in emergency situations or for management of priority pollutants when native toxicity data are lacking.

Keywords China · USA · Species sensitivity distribution · Native species · Water quality criteria · Acute toxicity data · Priority pollutant

Introduction

The species sensitivity distribution (SSD) method is widely used in ecological risk assessment procedures (Solomon et al. 1996; Versteeg et al. 1999) and the development of water quality criteria (ANZECC and ARMCANZ 2000; Wheeler et al. 2002). One of the purposes of SSD analysis is to determine the environmental concentration of a toxicant that protects most species in the environment. Usually, a point estimate of the hazardous concentration for 5 % of species (known as the HC5) or the 95 % protection level (Van Straalen and Van Rijn 1998) is used for this purpose. SSDs are constructed by fitting a cumulative distribution function to a plot of species toxicity data against rank-assigned percentiles (Van Straalen and Denneman 1989; Aldenberg and Slob 1993; Wheeler et al. 2002). The function applied in the SSD model of Europe and the USA is often log-normal (Wagner and Løkke 1991; European Commission 2011) or log-logistic (Aldenberg and Slob 1993), while that in Australia and New Zealand is the Burr type III function (Shao 2000). From each of these models, the HC5 value is calculated; it is known as the final acute value or final chronic value in the USA (Suter 2002). SSDs are dependent upon available datasets and can differ in type of distribution, taxonomic diversity, and sample size (Wheeler et al. 2002; Maltby et al. 2005). Only a substantial amount of

Responsible editor: Philippe Garrigues

Electronic supplementary material The online version of this article (doi:10.1007/s11356-013-2110-2) contains supplementary material, which is available to authorized users.

X. Wang · Z. Yan (✉) · Z. Liu (✉) · W. Wang · H. Li
State Key Laboratory of Environmental Criteria and Risk Assessment, State Environmental Protection Key Laboratory of Ecological Effect and Risk Assessment of Chemicals, Chinese Research Academy of Environmental Sciences, An Wai Da Yang Fang 8, Chaoyang District, Beijing 100012, China
e-mail: zgyan@craes.org.cn
e-mail: liuzt@craes.org.cn

X. Wang
College of Water Sciences, Beijing Normal University, Beijing 100875, China

C. Zhang
School of Environment, Beijing Normal University, Beijing 100875, China

toxicity data from several taxonomic groups can result in a robust HC5. Therefore, recommendations of minimum sample sizes necessary for meaningful HC5s of different distribution methodology vary in different countries (Feng et al. 2012). Eight species representative of diverse taxa have been considered a sufficient number to derive water quality criteria in the USA (Stephen et al. 1985). Similar taxa requirements have also been adopted in the water quality guidelines of other countries (ANZECC and ARMCANZ 2000; ECB 2003; CCME 2007; Van Vlaardingen and Verbruggen 2007; European Commission 2011).

In general, SSD curves are constructed using laboratory-derived toxicity data for species. However, it is difficult to obtain such toxicity data since a number of toxicity tests are limited by test procedure, species availability, time, and expense, especially for threatened and endangered species. Therefore, for most new and existing substances, this type of data is lacking (Sijm et al. 2001). Furthermore, in most countries, SSD curves and HC5 values are used to derive water quality criteria for toxicants based on local species data or site-specific data (Stephen et al. 1985; ANZECC and ARMCANZ 2000; Yin et al. 2003a, b). The potential use of non-native toxicity data for handling local problems is controversial, and leaves one to question whether criteria based on species from one geographical region provide appropriate protection for species in a different region (Davies et al. 1994). However, this argument cannot be resolved appropriately in large part due to the paucity of toxicity data applicable for local species and the lack of studies on such problems. Therefore, it is important to investigate whether it is feasible to use toxicity data of non-native species to extrapolate water quality criteria to protect native species.

In China, systematic water quality criteria studies have drawn increasing concern in the past 5 years, and SSD methods with HC5 values have been used to derive water quality criteria for limited toxicants with an emphasis on using native Chinese species (Jin et al. 2011; Yan et al. 2012; Yang et al. 2012). However, the comparison of SSDs based on native Chinese species and species from other geographic areas is rarely studied.

In the present study, eight priority pollutants both in China and the USA were selected due to the lack of suitable native ecotoxicology data and the priority management of priority pollutants in China. The pollutants were arsenic (As(III)), chromium (Cr(VI)), mercury (Hg), cadmium (Cd), nitrobenzene, 2,4,6-trichlorophenol, 2,4-dichlorophenol, and parathion. In addition, comparative studies on SSDs of different taxa between China and the USA were carried out. The aims of the study were (1) to determine the differences of sensitivity of each taxonomic group between China and the USA and (2) to discuss whether toxicity data of species from the USA can be used in deriving criteria to protect species in China. This study could provide useful information for site-specific risk assessment and environmental management.

Materials and methods

Data collection

The published acute toxicity data of As(III), Cr(VI), Hg, Cd, nitrobenzene, 2,4,6-trichlorophenol, 2,4-dichlorophenol, and parathion were collected from the ECOTOX database (http://cfpub.epa.gov/ecotox); United States Environmental Protection Agency water quality criteria document for As(III), Cr(VI), Hg, Cd, and parathion (US EPA 1996); China National Knowledge Infrastructure (http://www.cnki.net); and other sources. Species were selected based on whether they are (1) native to China or (2) introduced for economic reasons and now widely exist in China. The same principles were used for American species toxicity data when there was no water quality criteria document. All data were screened and analyzed according to guidelines for water quality criteria for aquatic life (Stephen et al. 1985). Toxicity data were limited to acute lethal concentration (LC50) and effective concentration (EC50) values from studies with exposure periods of 48 h for cladocerans and 96 h for others. Test organisms were categorized as either invertebrates or vertebrates, and each group was analyzed separately, including Chinese and/or American species.

Data analysis

Many cumulative distribution functions have been used to fit SSDs (Erickson and Stephan 1988; Wagner and Løkke 1991; Aldenberg and Jaworska 2000; Van der Hoeven 2001; Chen 2004; Hose and Van den Brink 2004). In this study, in order to make the comparisons feasible and statistically meaningful, just one method was used. The log-logistic distribution was used since it often fits the toxicity data well (Kooijman 1987; Newman et al. 2000; Wheeler et al. 2002; Feng et al. 2012). The equation is as follows:

$$y = 1\Big/\Big(1 + \exp\Big((P1 - x)\Big/P2\Big)\Big)$$

where y is the cumulative probability of species, defined as (the order of the data point)/(1+total number of data points), x is the mean of the $\log_{10}$-transformed LC50 or EC50 values, $P1$ is the parameter representing the intercept, and $P2$ is the parameter representing the slope of the curve.

The distribution model was fitted to toxicity data points and evaluated using the chi-square goodness-of-fit

Table 1 Number of data values and goodness-of-fit of different taxonomic groups for eight toxicants

Toxicant	Taxonomic group	*n*	Adj-R^2	*p*
As(III)	Total Chinese species	14	0.94	<0.01
	Chinese invertebrate	8	0.83	<0.05
	Chinese vertebrate	6	0.94	<0.01
	Total American species	16	0.91	<0.01
	American invertebrate	9	0.88	<0.01
	American vertebrate	7	0.93	<0.01
Cr(VI)	Total Chinese species	29	0.94	<0.01
	Chinese invertebrate	16	0.93	<0.01
	Chinese vertebrate	13	0.98	<0.01
	Total American species	34	0.85	<0.01
	American invertebrate	17	0.94	<0.01
	American vertebrate	17	0.98	<0.01
Hg	Total Chinese species	47	0.97	<0.01
	Chinese invertebrate	25	0.99	<0.01
	Chinese vertebrate	22	0.97	<0.01
	Total American species	33	0.98	<0.01
	American invertebrate	25	0.98	<0.01
	American vertebrate	8	0.94	<0.01
Cd	Total Chinese species	49	0.99	<0.01
	Chinese invertebrate	32	0.98	<0.01
	Chinese vertebrate	17	0.95	<0.01
	Total American species	50	0.94	<0.01
	American invertebrate	30	0.94	<0.01
	American vertebrate	20	0.89	<0.01
Nitrobenzene	Total Chinese species	20	0.88	<0.01
	Chinese invertebrate	8	0.83	<0.05
	Chinese vertebrate	12	0.82	<0.01
	Total American species	13	0.94	<0.01
	American invertebrate	6	0.96	<0.01
	American vertebrate	7	0.69	<0.10
2,4,6-Trichlorophenol	Total Chinese species	16	0.98	<0.01
	Chinese invertebrate	8	0.95	<0.01
	Chinese vertebrate	8	0.95	<0.01
	Total American species	16	0.97	<0.01
	American invertebrate	6	0.94	<0.01
	American vertebrate	10	0.96	<0.01
2,4-Dichlorophenol	Total Chinese species	17	0.96	<0.01
	Chinese invertebrate	6	0.88	<0.05
	Chinese vertebrate	11	0.96	<0.01
	Total American species	13	0.93	<0.01
	American invertebrate	5	0.85	<0.10
	American vertebrate	8	0.91	<0.01
Parathion	Total Chinese species	32	0.96	<0.01
	Chinese invertebrate	18	0.95	<0.01
	Chinese vertebrate	14	0.94	<0.01
	Total American species	38	0.93	<0.01
	American invertebrate	23	0.96	<0.01
	American vertebrate	15	0.97	<0.01

n number of data values of each taxonomic group, *Adj-R^2* adjusted coefficient of determination (R^2), *p* is the significance level of the adjusted coefficient of determination (R^2)

test with the adjusted coefficient of determination R^2 (Adj-R^2) in the software OriginLab 8.0 (USA, Origin Lab Company).

Statistical analyses of the difference of species sensitivity distributions for total species, invertebrates, or vertebrates between China and the USA were compared using the two-sample Kolmogorov–Smirnov test and Mann–Whitney test in the SPSS software (SPSS 20.0 for Windows). The two-sample Kolmogorov–Smirnov test (K–S test) and Mann–Whitney test (M–W test) are nonparametric methods that can be used to test whether two samples came from the same distribution, and have been used to compare the difference between SSDs in previous studies (Maltby et al. 2005; Jin et al. 2011, 2012). Moreover, hazardous concentrations for 5 % (HC5) and 50 % of the species (HC50) were calculated and compared between Chinese and American taxa.

Table 2 Comparison between different taxonomic groups from China and the USA using the two-sample Kolmogorov–Smirnov test and Mann–Whitney test

Taxa	Toxicant	ks	*p* (K–S test)	*p* (M–W test)
As(III)	Total	0.830	0.497	0.570
	Invertebrate	0.543	0.930	0.815
	Vertebrate	0.642	0.804	0.628
Cr(VI)	Total	1.059	0.212	0.264
	Invertebrate	0.834	0.490	0.345
	Vertebrate	0.835	0.488	0.805
Hg	Total	0.693	0.723	0.494
	Invertebrate	0.849	0.468	0.210
	Vertebrate	0.606	0.857	0.801
Cd	Total	0.581	0.889	0.685
	Invertebrate	0.984	0.288	0.269
	Vertebrate	1.106	0.173	0.044
nitrobenzene	Total	0.411	0.996	0.924
	Invertebrate	0.526	0.945	0.622
	Vertebrate	0.401	0.997	0.837
2,4,6-trichlorophenol	Total	0.707	0.699	0.423
	Invertebrate	0.309	1.000	1.000
	Vertebrate	0.738	0.648	0.360
2,4-dichlorophenol	Total	0.436	0.991	0.732
	Invertebrate	0.298	1.000	1.000
	Vertebrate	0.514	0.955	0.600
parathion	Total	0.953	0.324	0.147
	Invertebrate	1.113	0.168	0.109
	Vertebrate	0.423	0.994	0.847

ks is a test statistic used to determine the significance level *p* (K–S test), $p>0.05$ means the difference between distributions is not significant; *p* (M–W test) represents the significance level, $p>0.05$ means the difference between distributions is not significant

Results

Toxicity data and SSD construction

As shown in Table 1, we collected a total of 14, 29, 47, 49, 20, 16, 17, and 32 acute toxicity values for Chinese species that were divided into invertebrate and vertebrate taxa for As(III), Cr(VI), Hg, Cd, nitrobenzene, 2,4,6-trichlorophenol, 2,4-dichlorophenol, and parathion, respectively. Moreover, 16, 34, 33, 50, 13, 16, 13, and 38 acute toxicity values for the American taxa were found for the respective toxicants from ECOTOX, water quality criteria (WQC) documents, and other literature. The organisms included fish, amphibians, planktonic crustaceans, benthic crustaceans, insects, annelids, and so on (see Electronic supplementary material (ESM)).

The results indicated that the log-logistic distribution fit the data points of most taxonomic groups well, with Adj-R^2 of different taxonomic groups both in China and the USA from 0.82 to 0.99 ($p<0.01$). However, the distribution did not fit the nitrobenzene data for American vertebrates or 2,4-dichlorophenol for American invertebrates ($p>0.05$; Table 1, Fig. 2).

Comparison of SSDs for total Chinese and American species

In the present study, SSDs based on the total species were compared between Chinese and American taxa (Tables 2 and 3, Fig. 1). The results showed that compared with the SSDs of total American taxa, the SSDs of Cr(VI) and Hg, and SSD curves of As(III) and nitrobenzene below 0.20 (HC20) for Chinese species were shifted to the left, which indicated that Chinese species were more sensitive (Fig. 1). For nitrobenzene, the lower tails of both curves did not fit well, and species appeared to have similar sensitivity above 25 % of the affected species. On the contrary, the SSDs for American species were shifted to the left compared with the SSDs of Cd, 2,4,6-trichlorophenol, 2,4-dichlorophenol, and parathion for total Chinese taxa (Fig. 1). The comparison showed that HC5 values of total Chinese species were similar to those of total American species (difference, −146.87 to 64.83 %; Table 3) except Cr(VI) and parathion. The HC50 values of total Chinese species were similar to those of total American species for all eight pollutants (difference, −217.63 to 79.73 %; Table 3). Results of the two-sample Kolmogorov–Smirnov test (ks=0.411–1.059, $p=0.212$–0.996) and Mann–Whitney test ($p=0.109$–1.000) showed that the sensitivity distributions for total Chinese species and American species were not significantly different for any of the eight toxicants.

Comparison of SSDs for Chinese and American taxonomic groups

In this study, SSDs based on invertebrates were compared between Chinese and American taxa, and similar comparisons were conducted for vertebrates (Tables 2, 3; Fig. 2). Compared with the SSDs of As(III), Cr(VI), Hg, Cd, and nitrobenzene for American invertebrates, the SSDs for Chinese species were shifted slightly to the left (Fig. 2a). For vertebrates, SSDs for Chinese species for As(III), Cr(VI), Hg, and parathion were also shifted to the left (Fig. 2b). The comparison showed that HC5 and HC50 values of Chinese invertebrates were very close to those of American invertebrates, except for parathion (Table 3). As for vertebrates, HC5 and HC50 values between the two countries were similar except HC5 for nitrobenzene and Cd. Additionally, the sensitivity distributions for Chinese and American invertebrates were not significantly different for any of the eight toxicants (K–S test: ks=0.298–1.113, $p=$ 0.168–1.000; M–W test: $p=0.109$–1.000). The difference for vertebrates was also not significant (K–S test: ks=0.401–0.835, $p=0.488$–0.997; M–W test: $p=0.360$–0.847).

Discussion

In this study, through the goodness-of-fit test, we found that the log-logistic distribution fit the toxicity data of different taxonomic groups in China and the USA well (Adj-R^2: 0.82–0.99, $p<0.01$; Table 1). Previous studies also concluded that log-logistic distribution often fit the toxicity data best (Kooijman 1987; Newman et al. 2000; Wheeler et al. 2002). However, it does not work when the toxicity data are insufficient (Table 1).

In the present study, species sensitivity distributions and HC5 values of invertebrates were compared with vertebrates in both China and the USA (Table 3, Fig. 2). Sensitivity of invertebrate groups to As(III), Cr(VI), Hg, Cd, 2,4-dichlorophenol, and parathion was higher than that of vertebrates in both China and the USA. However, sensitivity of invertebrate groups was lower than that of vertebrate groups to nitrobenzene and 2,4,6-trichlorophenol. This was similar with previous studies that reported the need to derive SSDs for taxonomic groups separately for toxicants due to the different and specific toxic modes of action (Maltby et al. 2002); these studies showed a significant difference in the sensitivity of vertebrate and invertebrate groups for atrazine, diuron, and 2,4-D. On the contrary, there was no significant difference in the sensitivity of vertebrate and invertebrate groups for simazine (Maltby et al. 2002). The results indicated that As(III), Cr(VI), Hg, Cd, 2,4-dichlorophenol, and parathion might have similar toxic modes of action on invertebrates and vertebrates, while nitrobenzene and 2,4,6-trichlorophenol might have different toxic modes of action. The reason for this difference requires further investigation. Therefore, the sensitivity of different taxonomic groups and the toxic mode of action of toxicants should be taken into account when deriving site-specific water quality criteria or assessing the ecological risk.

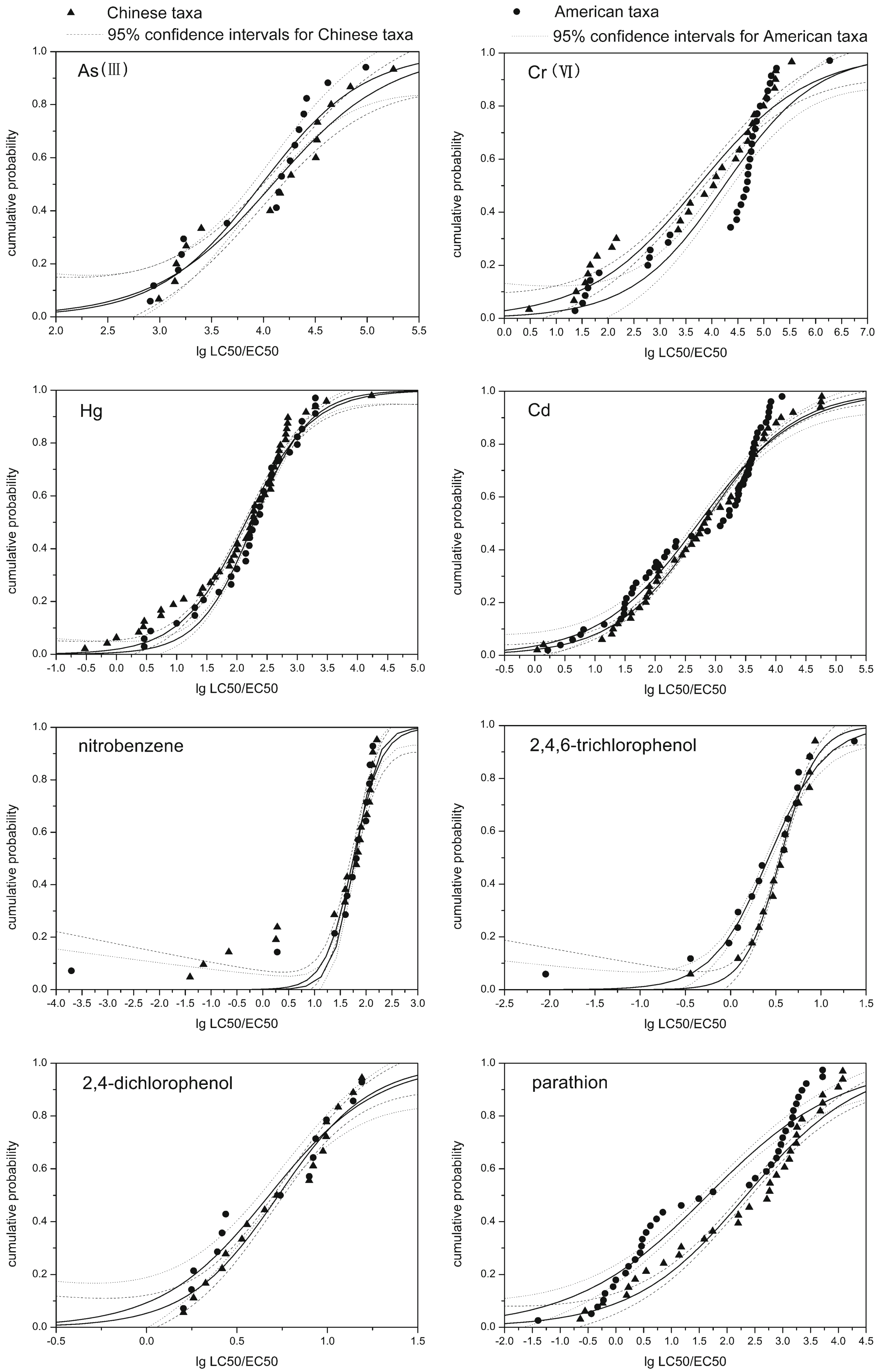
In this study, the SSDs of As(III), Cr(VI), Hg, and nitrobenzene for total Chinese taxa were shifted to the left of those

Table 3 HC5 and HC50 values of different taxonomic groups and comparisons between Chinese and American taxa

Toxicant	Taxonomic group	HC5[a]	Difference (%)	HC50[a]	Difference (%)
As(III)	Total Chinese species	264.05	−26.97	12,295.32	21.97
	Total American species	335.29		9.593.34	
	Chinese invertebrate	28.04	−151.03	5.717.42	34.39
	American invertebrate	70.39		3.751.11	
	Chinese vertebrate	5,949.86	−41.21	23.016.54	19.65
	American vertebrate	8,402.05		18.492.69	
Cr(VI)	Total Chinese species	4.15	−905.06	5.478.86	−217.63
	Total American species	41.71		17,402.83	
	Chinese invertebrate	0.39	−5.12	317.12	−315.17
	American invertebrate	0.41		1,316.59	
	Chinese vertebrate	8,014.93	−193.58	63,794.02	9.22
	American vertebrate	2,3530.48		57,912.19	
Hg	Total Chinese species	3.52	−146.87	139.95	−36.86
	Total American species	8.69		191.54	
	Chinese invertebrate	0.49	−630.61	64.50	−165.45
	American invertebrate	3.58		171.22	
	Chinese vertebrate	29.87	−140.91	250.81	10.75
	American vertebrate	71.96		223.84	
Cd	Total Chinese species	4.39	54.44	644.40	19.31
	Total American species	2.00		519.96	
	Chinese invertebrate	3.07	9.44	235.30	−112.83
	American invertebrate	2.78		500.80	
	Chinese vertebrate	384.17	99.85	3,836.45	86.68
	American vertebrate	0.57		510.67	
Nitrobenzene	Total Chinese species	8.86	−53.38	56.51	−6.93
	Total American species	13.59		60.43	
	Chinese invertebrate	11.45	−57.11	51.01	−9.76
	American invertebrate	17.99		55.99	
	Chinese vertebrate	4.48	95.98	57.30	38.28
	American vertebrate	0.18		35.36	
2,4,6-Trichlorophenol	Total Chinese species	0.91	64.83	3.56	25.84
	Total American species	0.32		2.64	
	Chinese invertebrate	1.06	12.26	3.49	−6.30
	American invertebrate	0.93		3.71	
	Chinese vertebrate	0.51	80.39	3.48	45.11
	American vertebrate	0.10		1.91	
2,4-Dichlorophenol	Total Chinese species	0.96	34.37	5.36	11.19
	Total American species	0.63		4.76	
	Chinese invertebrate	0.44	22.72	3.88	−11.85
	American invertebrate	0.34		4.34	
	Chinese vertebrate	1.52	53.94	6.37	19.93
	American vertebrate	0.70		5.10	
Parathion	Total Chinese species	0.22	90.90	222.37	79.73
	Total American species	0.02		45.07	
	Chinese invertebrate	0.01	−900.00	32.23	90.69
	American invertebrate	0.10		3.00	
	Chinese vertebrate	132.86	−99.36	984.60	−7.97
	American vertebrate	264.88		1,063.11	

[a] The unit of As(III), Cr(VI), Hg, Cd, parathion is microgram per liter, the unit of nitrobenzene; 2,4,6-trichlorophenol, and 2,4-dichlorophenol is milligram per liter. The difference was calculated by the function $(HC5_{\text{Chinese taxa}} - HC5_{\text{American taxa}})/HC5_{\text{Chinese taxa}} \times 100\ \%$

for total American taxa, resulting in lower HC5 and HC50 values for Chinese taxa, except for HC50 of As(III) (Table 3, Fig. 1). The results were in accordance with previous studies that showed HC5 values derived from tests with European species were lower than those derived from tests with North American species (Maltby et al. 2002). For nitrobenzene


Chinese taxa
95% confidence intervals for Chinese taxa
American taxa
95% confidence intervals for American taxa
As(Ⅲ)
Cr(Ⅵ)
Hg
Cd
nitrobenzene
2,4,6-trichlorophenol
2,4-dichlorophenol
parathion
cumulative probability
lg LC50/EC50

◀ **Fig. 1** Species sensitivity distribution of total species from China and the USA for As(III), Cr(VI), Hg, Cd, nitrobenzene, 2,4,6-trichlorophenol, 2,4-dichlorophenol, and parathion

(Fig. 1), the lower tail of both curves did not fit well, but species appeared to have similar sensitivity above 25 % of the affected species. Moreover, there were almost two times more available data for Chinese species than American species, and this might cause the difference in sensitivity. It was opposite, however, for Cd, 2,4,6-trichlorophenol, 2,4-dichlorophenol, and parathion, and the HC5 and HC50 values for Chinese taxa were higher (Fig. 1). Although the HC5 and HC50 values between Chinese and American taxa were different, they were within an order of magnitude for the two countries except HC5 for Cr(VI) and parathion (Table 3). Dyer et al. (2008) and Feng et al. (2012) reported that HC5 values within an order of magnitude were acceptable in deriving WQC. The order of magnitude differences of HC5 for Cr(VI) and parathion were mainly due to the most sensitive species: *Diaphanosoma brachyurum* for Cr(VI) in China (ESM Table S3) and *Orconectes nais* for parathion in the USA (ESM Table S16). Moreover, there was no statistically significant difference in the sensitivity distributions for total species between China and the USA for any of the eight toxicants (K–S test: ks=0.411–1.059, p=0.212–0.996; M–W test: p=0.109–1.000; Table 2). Jin et al. (2012) and Feng et al. (2013) found there was no significant difference in SSDs for Chinese and American (non-Chinese) species for pentachlorophenol and zinc, and this was in accordance with our study. Moreover, studies conducted by Hose and Van den Brink (2004) also showed no significant difference in SSDs between Australian and non-Australian organisms exposed to endosulfan. In addition, studies reported similar sensitivities among North American and European taxa with different geographic distributions (Maltby et al. 2002). Other studies also found that there was no significant difference in the acute

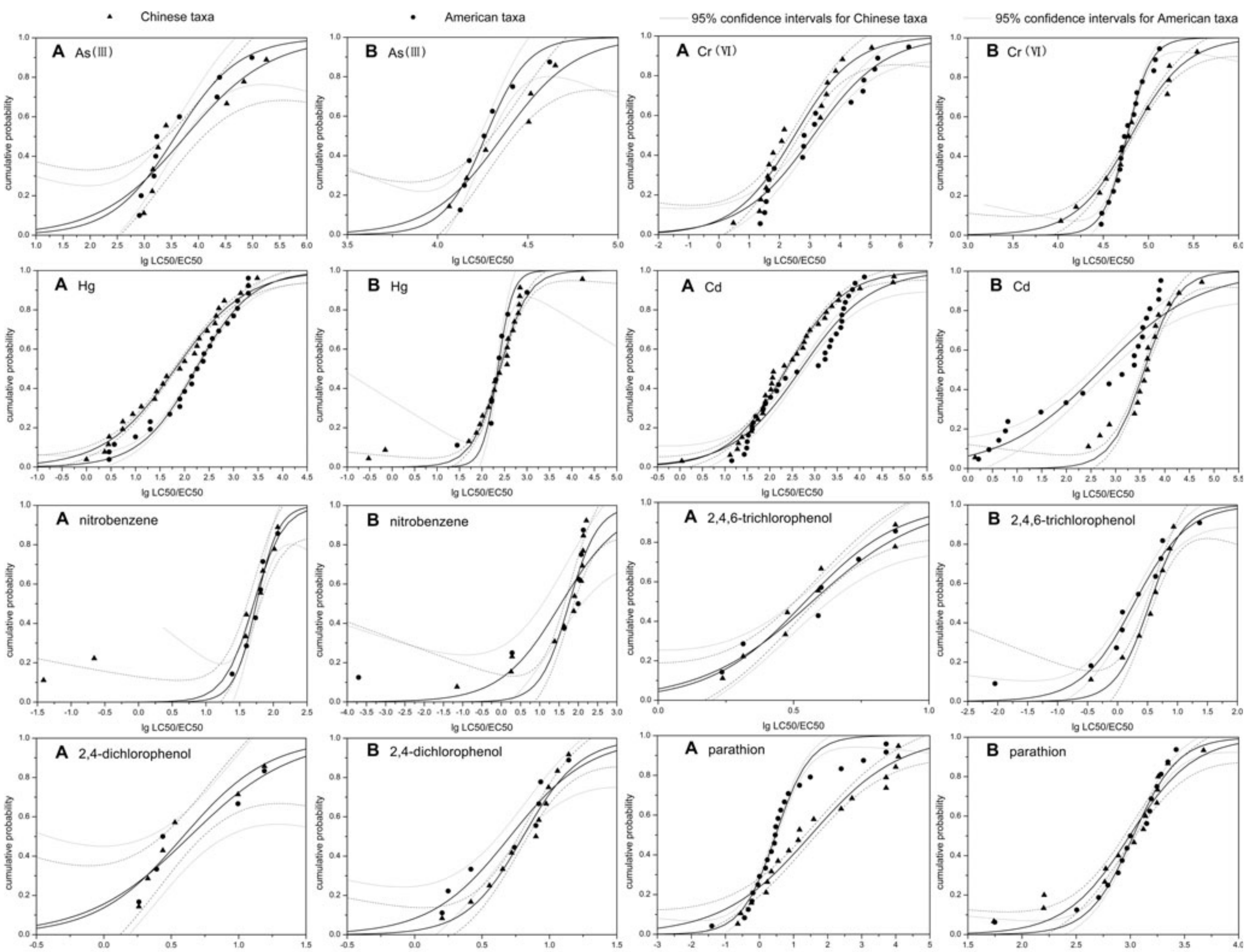


Fig. 2 Species sensitivity distribution (SSD) of different taxonomic groups from China and the USA. *A* The SSD curve derived from Chinese and American invertebrates for As(III), Cr(VI), Hg, Cd, nitrobenzene, 2,4,6-trichlorophenol, 2,4-dichlorophenol, and parathion; *B* the SSD curve derived from Chinese and American vertebrates for the same toxicants

toxicity of carbaryl, lindane, or malathion on temperate or tropical fish (Dyer et al. 1997).

In our paper, the SSDs based on Chinese and American invertebrates were compared. Results showed that the SSDs of Cr(VI), Hg, and nitrobenzene for Chinese invertebrates were shifted to the left in relation to American invertebrates, and resulted in lower HC5 and HC50 values. The SSDs of As(III) and parathion showed lower HC5 but higher HC50 values for Chinese invertebrates. As for vertebrates, the SSDs of parathion showed lower HC5 and HC50 values for Chinese vertebrates, while SSDs of As(III), Cr(VI), and Hg showed lower HC5 but higher HC50 values. It was opposite for nitrobenzene, 2,4,6-trichlorophenol, and 2,4-dichlorophenol, resulting in higher HC5 and HC50 values for Chinese vertebrates. However, the differences of HC5 and HC50 values between the two countries were small and within an order of magnitude except HC5 values of vertebrates for Cd and nitrobenzene. Previous studies reported that the variation of HC5 values within an order of magnitude were acceptable in deriving WQC (Dyer et al. 2008; Feng et al. 2012). The order of magnitude differences of HC5 values were mainly due to the most sensitive species, *Oncorhynchus mykiss*, for nitrobenzene in the American vertebrate group (ESM Table S10), and there were many toxicity data from Salmonidae fishes that were very sensitive to Cd in the American vertebrate group (ESM Table S8). Moreover, there was no statistically significant difference in the sensitivity distributions for invertebrates and vertebrates between China and the USA for any of the eight toxicants (invertebrates: K–S test: ks=0.298–1.113, p=0.168–1.000; M–W test: p=0.109–1.000; vertebrates: K–S test: ks=0.401–1.106, p=0.173–0.997; M–W test: p=0.360–0.847) except the sensitivity distribution of vertebrates for Cd (Table 2). This was mainly because these cadmium-sensitive Salmonidae fishes are native species in the USA but non-native species in China (ESM Tables S7 and S8). These findings were in accordance with previous studies that showed no significant difference was observed in SSDs of invertebrate arthropods between Europe and America or between temperate and tropical areas for chlorpyrifos and fenitrothion (Maltby et al. 2002). In addition, studies conducted by Hose and Van den Brink (2004) also showed that no significant difference occurred in SSDs of arthropods and fish between Australian and non-Australian countries exposed to endosulfan.

Conclusion

In this study, the acute SSDs of total species, invertebrates, and vertebrates from China and the USA for eight priority pollutants were constructed, and the differences of species sensitivities were compared between the two countries. The results showed that the log-logistic distribution fit the toxicity data of different taxonomic groups in China and the USA well, and indicated that As(III), Cr(VI), Hg, Cd, 2,4-dichlorophenol, and parathion might have similar toxic modes of action on invertebrates and vertebrates, while nitrobenzene and 2,4,6-trichlorophenol might have different toxic modes of action. Comparison of the SSDs and HC5 and HC50 values showed that there was no significant difference between Chinese and American species. The use of toxicity data from another place could be feasible in emergencies or other situations. This finding provides useful information in site-specific water quality criteria derivation and risk assessment management.

Acknowledgments This work was financially supported by the National Science and Technology Project of Water Pollution Control and Abatement of China (grant no. 2012ZX07501-003-06), the Revolution Startup Special Project of Chinese Research Academy of Environmental Sciences (grant no. 2011GQ-02, 2012YSKY18), and the Program of Environmental Protection Commonweal Research (grant no. 2011467054).

References

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

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

ANZECC, ARMCANZ (2000) Australian and New Zealand guidelines for fresh and marine water quality. National Water Quality Management Strategy Paper No 4 ANZECC and ARMCANZ, Canberra

CCME (2007) A protocol for the derivation of water quality guidelines for the protection of aquatic life. Canadian Council of Ministers of the Environment, Winnipeg

Chen L (2004) A conservative, nonparametric estimator for the 5th percentile of the species sensitivity distributions. J Stat Plan Infer 123(2):243–258

Davies PE, Cook LSJ, Goenarso D (1994) Sublethal responses to pesticides of several species of Australian freshwater fish and crustaceans and rainbow trout. Environ Toxicol Chem 13(8):1341–1354

Dyer SD, Belanger SE, Carr GJ (1997) An initial evaluation of the use of Euro/North American fish species for tropical effects assessments. Chemosphere 35(11):2767–2781

Dyer SD, Versteeg DJ, Belanger SE, Chaney JG, Raimondo S, Barron MG (2008) Comparison of species sensitivity distributions derived from interspecies correlation models to distributions used to derive water quality criteria. Environ Sci Technol 42(8):3076–3083

ECB (2003) Technical guidance document on risk assessment part II. Institute for Health and Consumer Protection, European Chemicals Bureau, Europe, 20418

Erickson RJ, Stephan CE (1988) Calculation of the final acute value for water quality criteria for aquatic organisms. National Technical Information Service, Springfield VA. PB88-214994

European Commission (2011) WFD-CIS guidance document no. 27 technical guidance for deriving environmental quality standards. Office for Official Publications of the European Communities, Luxembourg: 204pp

Feng CL, Wu FC, Dyer SD, Chang H, Zhao XL (2013) Derivation of freshwater quality criteria for zinc using interspecies correlation estimation models to protect aquatic life in China. Chemosphere 90(3):1177–1183

Feng CL, Wu FC, Zhao XL, Li HX, Chang H (2012) Water quality criteria research and progress. Sci China Earth Sci 55(6):882–891

Hose GC, Van den Brink PJ (2004) Confirming the species-sensitivity distribution concept for endosulfan using laboratory, mesocosm, and field data. Arch Environ Contam Toxicol 47(4):511–520

Jin X, Zha J, Xu Y, Giesy JP, Wang Z (2012) Toxicity of pentachlorophenol to native aquatic species in the Yangtze River. Environ Sci Pollut Res 19(3):609–618

Jin X, Zha J, Xu Y, Wang Z, Kumaran SS (2011) Derivation of aquatic predicted no-effect concentration (PNEC) for 2,4-dichlorophenol: comparing native species data with non-native species data. Chemosphere 84(10):1506–1511

Kooijman SALM (1987) A safety factor for LC_{50} values allowing for differences in sensitivity among species. Water Res 21(3):269–276

Maltby L, Blake N, Brock TCM, Van den Brink PJ (2002) Addressing interspecific variation in sensitivity and the potential to reduce this source of uncertainty in ecotoxicological assessments. DEFRA project code PN0932. London (UK): UK Department for Environment, Food and Rural Affairs, 22pp

Maltby L, Blake N, Brock TCM, Van den Brink PJ (2005) Insecticide species sensitivity distributions: importance of test species selection and relevance to aquatic ecosystems. Environ Toxicol Chem 24(2): 379–388

Newman MC, Ownby DR, Mézin LCA, Powell DC, Christensen TRL, Lerberg SB, Anderson BA (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

Shao Q (2000) Estimation for hazardous concentrations based on NOEC toxicity data: an alternative approach. Environmetrics 11(5):583–595

Sijm D, de Bruijn J, Crommentuijn T, van Leeuwen K (2001) Environmental quality standards: endpoints or triggers for a tiered ecological effect assessment approach? Environ Toxicol Chem 20(11):2644–2648

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

Stephen CE, Mount DI, Hansen DJ, Gentile JR, Chapman GA, Brungs WA (1985) Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. United States Environmental Protection Agency, Office of Research and Development, Washington DC

Suter GW II (2002) North American history of species sensitivity distributions. In: Postuma L, Suter GW II, Traas TP (eds) Species sensitivity distributions in ecotoxicology. Lewis, Boca Raton, FL, USA, pp 11–17

US EPA (1996) 1995 Updates: water quality criteria documents for the protection of aquatic life in ambient water. United States Environmental Protection Agency, Office of Water, EPA-820-B-96-001,Washington DC

Van der Hoeven N (2001) Estimating the 5-percentile of the species sensitivity distributions without any assumptions about the distribution. Ecotoxicology 10(1):25–34

Van Straalen NM, Van Rijn JP (1998) Ecotoxicological risk assessment of soil fauna recovery from pesticide application. Rev Environ Contam Toxicol 154:83–141

Van Straalen NM, Denneman CAJ (1989) Ecotoxicological evaluation of soil quality criteria. Ecotoxicol Environ Saf 18(3):241–251

Van Vlaardingen PLA, Verbruggen EMJ (2007) Guidance for the derivation of environmental risk limits within the framework of 'International and national environmental quality standards for substances in the Netherlands' (INS). Bilthoven, The Netherlands: National Institute for Public Health and the Environment (RIVM) Report (601782001): 146pp

Versteeg DJ, Belanger SE, Carr GJ (1999) Understanding single-species and model ecosystem sensitivity: data-based comparison. Environ Toxicol Chem 18(6):1329–1346

Wagner C, Løkke H (1991) Estimation of ecotoxicological protection levels from NOEC toxicity data. Water Res 25(10):1237–1242

Wheeler JR, Grist EPM, Leung KMY, Morritt D, Crane M (2002) Species sensitivity distributions: data and model choice. Mar Pollut Bull 45(1–12):192–202

Yan ZG, Zhang ZS, Wang H, Liang F, Li J, Liu HL, Sun C, Liang LJ, Liu ZT (2012) Development of aquatic life criteria for nitrobenzene in China. Environ Pollut 162:86–90

Yang SW, Yan ZG, Xu FF, Wang SR, Wu FC (2012) Development of freshwater aquatic life criteria for tetrabromobisphenol A in China. Environ Pollut 169:59–63

Yin D, Hu S, Jin H, Yu L (2003a) Deriving freshwater quality criteria for 2,4,6-trichlorophenol for protection of aquatic life in China. Chemosphere 52(1):67–73

Yin D, Jin H, Yu L, Hu S (2003b) Deriving freshwater quality criteria for 2,4-dichlorophenol for protection of aquatic life in China. Environ Pollut 122(2):217–222