Toxicokinetics and the related metabolites in rainbow trout (*Oncorhynchus mykiss*) after exposure to decabromodiphenyl ether

FENG ChengLian, XU YiPing, ZHA JinMiao, LUO Qian, SHAN XiaoQuan & WANG ZiJian*

State Key Laboratory of Environmental Aquatic Chemistry; Research Center for Eco-Environmental Sciences,
Chinese Academy of Sciences, Beijing 100085, China

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Decabromodiphenyl ether (BDE209) is poorly absorbed by mammals, and little information is available on the toxicokinetics of BDE209 and its metabolites in fish. In the present study, rainbow trout (*Oncorhynchus mykiss*) were administered to 100 ng/g and 500 ng/g body wet weight of BDE209 via a single intraperitoneal injection and parent BDE209 and its metabolites were sequentially monitored for 28 days. The results showed that toxicokinetic profiles of BDE209 could be described by the one-compartment model. In the higher dose group (500 ng/g wet weight), the calculated half-life ($t_{1/2}$) and elimination rate (k_e) were 17.7 d and 0.039/d in the liver, and 100.3 d and 0.007/d in the muscle, respectively. Three major methoxylated brominated diphenyl ethers (MeO-BDEs) were detected with 2,2',4,4'-tetrabromo-5-methoxydiphenyl ether (5-MeO-BDE47) being detected in all tissue samples. There was no significant temporal change of 5-MeO-BDE47 concentration in the muscle, whereas an exponential increase was observed in the liver. Therefore, the metabolism rate of BDE209 depended on the administered dose. BDE209 was hardly accumulated in the muscle of rainbow trout, while the liver was a primary metabolic organ. MeO-BDEs were formed via metabolism of BDE209, which probably played a significant role in fish toxicology as a potential indicator.

decabromodiphenyl ether (BDE209), polybrominated diphenyl ethers (PBDEs), methoxylated brominated diphenyl ethers (MeO-BDEs), toxicokinetics, rainbow trout

1 Introduction

Polybrominated diphenyl ethers (PBDEs) are widely used as brominated flame retardants (BFRs), and are ubiquitous environmental contaminants found in both abiotic and biotic environmental samples [1]. It is believed that PBDEs (mainly tetra- to octa-BDEs) are chemically and biologically persistent and lipophilic, and can be accumulated in fatty tissues of organisms. Some of these BDE congeners are even more toxic for animals, causing reproductive and developmental effects, neurobehavioral toxicity, thyroid hormone disruption, immune-toxicity and possibly cancer [2–4]. A mixture of penta- and octa-BDE has been banned

for production and application in Europe (EU) and in other countries, whereas commercial deca-BDE (BDE209) is restricted to be used in electronic devices in the EU because deca-BDE coexists with other PBDEs, notably nona-BDE. Recently, tetra- to hepta-BDEs have been added to the POPs list of the Stockholm Convention [5].

BDE209 is the principal component of commercial deca-BDE, and comprises approximately 80% of the world consumption of PBDEs [6]. BDE209 has extremely low water solubility ($\log K_{ow} \sim 10$), a large size of molecules, and very low bioavailability. Therefore, BDE209 has been considered as an environmentally stable and inert product that cannot be degraded in the environment. Elevated concentrations of BDE209 were detected in biota, sediments, bio-solids and house dust [7–10], in blood and breast milk [11, 12], and in aquatic biota [13].

*Corresponding author (email: wangzj@rcees.ac.cn)

A hot topic about PBDE research is whether they are metabolized to other compounds and what pathway governs their metabolism in most environmental biota. Therefore, understanding the toxicokinetics of the congeners is essential for risk assessment of PBDEs. There are several studies on the toxicology of DE-71, a commercial penta-mixture, and the results showed that it could impair reproductive development and alter thyroid hormone status in rats [14, 15]. In addition, the toxicokinetics of BDE47, BDE99, BDE100, BDE153 and BDE154 were also studied in rats and mice, and the results demonstrated that lipophilic tissues were the preferred uptake sites [16]. The toxicity study of BDE209 in rats with oral administration suggested that BDE209 was hazardous to reproductive health [17]. Overall, most of these studies focused on the toxicokinetics of BDEs in mammals, however, little information was available on the toxicokinetics of BDEs in fish.

As for BDE209, its metabolism in fish indicated that BDE209 could be transformed to debrominated diphenyl ethers. There are numerous previous studies about biotic and abiotic debromination of BDE209. Metabolic debromination of BDE209 occurred in rats, cows and birds depending on species [18–21]. Stapleton *et al.* [22] conducted an *in vivo* study in common carp and rainbow trout, revealing that debromination of BDE209 metabolism could also occur in fish. A previous study about BDE209 in rainbow trout indicated that BDE209 could undergo biological debromination [23]. Furthermore, debromination of BDE209 by juvenile carp following dietary exposure was also observed, and several hexa- through octa-BDE metabolites were found to be more bioavailable than BDE209 itself [24]. In addition to debrominated diphenyl ether metabolites, methoxylated brominated diphenyl ethers (MeO-BDEs) were also regarded as the BDE metabolites in organisms [25–28]. However, the majority of those studies attributed the MeO-BDEs in biota samples to naturally biogenetic processes in marine sponges and algae [29], while very limited studies about MeO-BDEs metabolites of PBDEs were available. The relationship between MeO-BDE metabolites and the BDE toxicokinetics remained unclear.

The aim of the present study was to investigate (i) the toxicokinetics of BDE209 in rainbow trout (*Oncorhynchus mykiss*) and (ii) the metabolism of debrominated diphenyl ethers and MeO-BDE in the liver and muscle. The data could provide scientific supports for future research of BDE209 toxicology in fish.

2 Materials and methods

2.1 Standards and reagents

Commercial deca-BDE (BDE209: purity $\geq 98\%$) used for the exposure experiment was kindly provided by Dalian Institute of Chemical Physics (DICP), Chinese Academy of

Sciences (CAS), China. The BDE and MeO-BDE standards were purchased from Wellington Laboratories (Ontario, Canada). BDE mixture standards include 27 BDE congeners (BDE3, 7, 15, 17, 28, 47, 49, 66, 71, 77, 85, 99, 100, 119, 126, 138, 153, 154, 156, 183, 184, 191, 196, 197, 206, 207, 209). MeO-BDE mixture standards include four tetra-MeO-BDEs and four penta-MeO-BDEs: 2,2',4,4'-tetrabromo-5-methoxydiphenyl ether (5-MeO-BDE47), 2,2',4,4'-tetrabromo-6-methoxydiphenyl ether (6-MeO-BDE47), 2,2',4,5'-tetrabromo-4'-methoxydiphenyl ether (4'-MeO-BDE49), 2,3',4,5'-tetrabromo-2'-methoxydiphenyl ether (2'-MeO-BDE68), 2,2',4,4',5-pentabromo-5'-methoxydiphenyl ether (5'-MeO-BDE99), 2,2',4,4',6-pentabromo-5'-methoxydiphenyl ether (5'-MeO-BDE100), 2,2',4,5,5'-pentabromo-4'-methoxydiphenyl ether (4'-MeO-BDE101), and 2,2',4,5',6-pentabromo-4'-methoxydiphenyl ether (4'-MeO-BDE103). ^{13}C -labeled PBDE mixtures (^{13}C -labeled BDE-3, 15, 28, 47, 99, 153, 154, 183, 197, 207 and 209) were used to optimize extraction and cleanup procedures for recovery and sensitivity tests of BDE congeners. ^{13}C -labeled 2,2',4,4'-tetrabromo-6-methoxydiphenyl ether (6-MeO-BDE47) and ^{13}C -labeled 2,2',4,4',6-pentabromo-6'-methoxy diphenyl ether (6-MeO-BDE100) were used as recovery surrogates for MeO-BDE congeners. ^{13}C -labeled-2,2',3,4,4',5'-hexabromodiphenyl ether (^{13}C -BDE138) was used as an internal standard for analysis. All solvents used for extraction and clean-up of the extract, including dichloromethane (DCM), hexane and acetone, were purchased from J. T. Baker (HPLC grade).

2.2 Experimental design

Juvenile rainbow trout (*Oncorhynchus mykiss*), approximately 200 mm in length, 100 g in weight, were purchased from a breeding farm. Fish were cultured in four 250-liter glass tanks (40 fish/tank). Aeration was performed throughout the experiments to maintain oxygen saturation. These fish were maintained in de-chlorinated tap water (using an activated carbon filter) at a constant temperature ($15 \pm 2^\circ\text{C}$), with a photoperiod of 16:8 h (light:dark), which was close to their optimal temperature range in the natural environment. Fish were acclimated for a week prior to the beginning of the experiments.

BDE209 was dissolved in corn oil to make the stock concentration of 0.25 g/L. Two treatment groups and one control group were used in the experiments. The rainbow trout in the control group were not exposed to BDE209 and only exposed to corn oil using intraperitoneal injection. Rainbow trout in the treatment groups were exposed to 500 ng/g fresh wet weight of BDE209 for the high-dose group (HG), and to 100 ng/g fresh body wet weight of BDE209 for the low-dose group (LG) using single intraperitoneal injection. During the whole experimental period (28 d), fish were fed with commercial dry fish food (Tetra Pond sticks, Germany) once a week. Care should be taken that the food

was consumed within 20–40 s by the fish.

2.3 Sampling and pretreatment

To avoid contamination from the gastrointestinal tract, fish were starved for 24–48 h before sampling. One fish from each tank was sampled on days 1, 2, 4, 7, 14, 21 and 28, and each sampling procedure was performed in triplicate. Fish mass and length were measured and then the fish were killed by a blow to the head. The fish muscle (skin removed) and liver were separated from the whole body and stored at $-80\text{ }^{\circ}\text{C}$ prior to analysis.

The muscle and liver samples were extracted and cleaned up as described elsewhere [30]. The muscle (skin removed) samples were freeze-dried, and subsequently dried fish muscle was homogenized in a stainless steel blender. The water content in the muscle was determined using gravimetric analysis. About 2 g of dried fish muscle was mixed with 4 g of anhydrous sodium sulfate that was cleaned with DCM and activated at $140\text{ }^{\circ}\text{C}$ for 24 h in advance. Meanwhile, each liver sample (~ 1 g wet weight) was homogenized with 4 g anhydrous sodium sulfate in porcelain mortar. The surrogate standards of ^{13}C -labeled BDE mixture and ^{13}C -labeled MeO-BDE mixture were added, and then the muscle and liver samples were extracted with 100 mL of DCM/acetone for 24 h in Soxhlet extraction apparatus. Afterwards, the sample extracts were concentrated with a rotary evaporator and transferred to a glass bottle with 6 mL of hexane. The extracts were then treated with concentrated sulfuric acid to remove the bulk of lipids. Afterwards, two steps of further cleanup process were carried out. First, the extracts were transferred to a silica gel column containing 4 g of anhydrous sodium sulfate, 10 g of silica gel impregnated with sulfuric acid (30%) and 2 g of active silica gel with the order from top to bottom. Hexane (30 mL) was used for preliminary elution, and then the eluted solution was collected. Second, the collected elution from the first column was loaded onto an alumina column containing 4 g of anhydrous sodium sulfate and 8 g of acid alumina with the order from top to bottom. Then 30 mL of hexane/DCM (7:3) was used to elute this column, and the eluted solution was collected and concentrated in a gentle nitrogen evaporation apparatus at room temperature. Finally, the sample was adjusted to 100 μL of hexane containing 500 $\mu\text{g/L}$ internal standard of ^{13}C -labeled BDE138. All the samples were concentrated in Agilent vials and stored in a fridge at $-80\text{ }^{\circ}\text{C}$ prior to the analysis by GC-MS. To prevent photodegradation, the samples remained wrapped with aluminum foil during the entire process.

2.4 Instrumental analysis

GC-MS analysis was performed on an Agilent 6890 gas chromatograph (GC) connected to an Agilent 5975 network mass spectrometer (MS). GC-MS-MS was performed on the

Varian CP3800/300 system to calibrate the data. For the determination of mono- to hexa-BDE congeners and MeO-BDEs, a DB-5ms capillary column ($30\text{ m} \times 0.25\text{ mm i.d.} \times 0.25\text{ }\mu\text{m}$ film thickness) was used. The temperature program was from $110\text{ }^{\circ}\text{C}$ (held for 1 min) to $240\text{ }^{\circ}\text{C}$ (held for 2 min) at $15\text{ }^{\circ}\text{C/min}$, and then from $240\text{ }^{\circ}\text{C}$ to $265\text{ }^{\circ}\text{C}$ (held for 25 min) at $2\text{ }^{\circ}\text{C/min}$ using the splitless injection mode during 1 min, with injection volume of 1 μL . For the determination of hepta- to deca-BDEs, a DB-5ms capillary column ($15\text{ m} \times 0.25\text{ mm i.d.} \times 0.1\text{ }\mu\text{m}$ film thickness) was used. The temperature program was from $150\text{ }^{\circ}\text{C}$ (held for 1 min) to $325\text{ }^{\circ}\text{C}$ (held for 5 min) at $17\text{ }^{\circ}\text{C/min}$ using the pulsed splitless injection mode until 1.5 min, purge flow 50 mL/min at 2 min, with injection volume of 1 μL .

Electron ionization (EI) and electron-capture negative ionization (ECNI) were used. For EI, the operating conditions were as follows: the ion source temperature was $300\text{ }^{\circ}\text{C}$, the quadrupole temperature was $150\text{ }^{\circ}\text{C}$, the transfer line was at $300\text{ }^{\circ}\text{C}$, the scan range was 200–1000 amu, and electron energy was 70 eV. Ion peaks ($[\text{M}]^{+}$ or $[\text{M}-\text{Br}_2]^{+}$) were selected. For ECNI, helium and methane were used as carrier gas (constant column flow 2 mL/min) and reaction gas (6 Torr), respectively, and the multiplier voltage was set at 1250 V. BDE209 was quantified by monitoring ions m/z 486.6 and 488.5. All the analytes were identified on the basis of their retention time, relative to authentic standards in the selected ion monitoring (SIM) mode.

BDE209 is a fully-brominated diphenyl ether. Because of its high degree of substitution, it is not stable at high temperatures in the injector and on the GC column [31]. Moreover, the compound is sensitive to UV light and strongly adsorbs to any kind of surfaces. The limit of detection (LOD) was defined as the concentration that would result in a signal-to-noise ratio of 3, and the detection limit for BDE209 was approximately 1.25 ng/g fish wet weight. LODs of BDEs and MeO-BDEs varied from congener to congener based on the amount of the sample and the instrument sensitivity, and ranged from 0.15 to 1.05 ng/g. Additionally, one procedural blank was run with every batch of 6–10 samples to assess the potential sample contamination. The laboratory blank analysis demonstrated that concentrations of BDEs and MeO-BDEs were below the LOD, which means that the samples were free from contamination. The recovery of surrogate standards in spiked blanks was found to be 70.9% to 120.5%, which was in agreement with the EPA standard.

2.5 Statistical analysis

Statistical analysis of data was performed with SPSS software (SPSS 13.0 for Windows). Spearman rank correlation was used to examine the strength of associations between parameters. Kruskal-Wallis nonparametric tests were used to compare the difference among groups. One way analysis of variance (ANOVA) was performed on all the experi-

mental data. Probability values less than 0.05 were considered as statistically significant.

3 Results

During the whole experimental period, the fish body mass, liver mass and hepatic somatic index (HSI, mass of liver/body mass $\times 100\%$) of the juvenile rainbow trout were measured. A comparison of body mass, liver mass and HSI between the control and the treated fish was made, revealing no significant differences among these groups (data not shown). Therefore, no growth calibrations need to be considered in this toxicokinetic study.

3.1 Toxicokinetics of BDE209

Toxicokinetic profiles of BDE209 in rainbow trout were illustrated in Figure 1. Toxicokinetics of BDE209 in both the muscle and liver were fitted to the one-compartment model with extravascular administration using the equation shown below:

$$C_t = \frac{k_a F D}{V(k_a - k_e)} (e^{-k_e t} - e^{-k_a t})$$

where C_t is BDE209 concentration in the fish at time t , k_a is the first-order absorption rate constant, k_e is the first-order elimination rate constant, F is bioavailability, D is the dose, and V is the volume of distribution. The biological half-life ($t_{1/2}$) was calculated as $0.693/k_e$.

It was found that throughout the whole experiment, BDE209 went through the absorption and the elimination process. For LG, BDE209 reached the highest concentration on day 7, whereas for HG, BDE209 reached the highest concentration on day 4. For the same treatment group, BDE209 concentrations in the muscle were much higher than those in the liver, which indicated that BDE209 could

Figure 1 Toxicokinetic profiles of BDE209 in rainbow trout. (muscle-LG = low-dose group in muscle tissue; liver-LG = low-dose group in liver tissue; muscle-HG = high-dose group in muscle tissue; liver-HG = high-dose group in liver tissue).

metabolize rapidly in the liver.

3.2 Metabolites of BDE209

The debrominated diphenyl ether metabolites including mono- to nona-BDEs in fish tissues were also detected throughout the whole experiment. The temporal changes of lower brominated diphenyl ethers (low-BDEs, mono- to tri-BDEs, including BDE3, 7, 15, 17, 28), medium brominated diphenyl ethers (medium-BDEs, tetra- to hepta-BDEs, including BDE47, 49, 66, 71, 77, 85, 99, 100, 119, 126, 138, 153, 154, 156, 183, 184, 191) and higher brominated diphenyl ethers (high-BDEs, octa- to nona-BDEs, including BDE196, 197, 206, 207) were also investigated (Figure 2). In the muscle, low-BDEs were seldom detected, only BDE28 was detected after 21 d in both LG and HG, and the concentration was below 1 ng/g wet weight. No significant difference was observed between LG and HG, while the highest medium-BDEs were detected, including BDE47, 49, 66, 71, 77, 85, 99, 100, 119, 126. Peak concentrations of medium-BDEs were detected on day 21. After that, the concentrations decreased during the monitored time. The

Figure 2 Metabolite concentrations of debrominated diphenyl ether in tissues of rainbow trout. low-BDEs-LG = lower brominated diphenyl ethers (mono- to tri-BDEs) in the low-dose group; low-BDEs-HG = lower brominated diphenyl ethers (mono- to tri-BDEs) in the high-dose group; medium-BDEs-LG = medium brominated diphenyl ethers (tetra- to hepta-BDEs) in the low-dose group; medium-BDEs-HG = medium brominated diphenyl ethers (tetra- to hepta-BDEs) in the high-dose group; high-BDEs-LG = higher brominated diphenyl ethers (octa- to nona-BDEs) in the low-dose group; high-BDEs-HG = higher brominated diphenyl ethers (octa- to nona-BDEs) in the high-dose group.

concentration of high-BDEs was lower than that of medium-BDEs, with BDE206 and BDE207 being detected. The concentration of BDE206 was a little higher than that of BDE207. In the liver, the concentrations of the debrominated diphenyl ether metabolites were much higher than those in the muscle. High-BDEs had the highest concentrations (including BDE196, 197, 206, 207 being detected), followed by medium-BDEs (including BDE47, BDE49, BDE66, BDE71, BDE77, BDE183, BDE184, BDE191). Whereas low-BDEs, including BDE3, 7, 15, 28, were detected with the lowest concentration and no significant differences were observed between LG and HG. In LG, the debrominated diphenyl ether posed a slight increasing trend after day 14, while in HG the debrominated diphenyl ethers posed a slight decreasing trend.

Among the eight MeO-BDE congeners, three MeO-BDEs (5-MeO-BDE47, 4'-MeO-BDE49 and 5'-MeO-BDE100) were detected in fish samples of the treatment groups. While in the samples from the control group, no MeO-BDEs were detected, suggesting that the detected MeO-BDE congeners were from BDE209 metabolism. It is noteworthy that 5-MeO-BDE47 was detected in all fish tissue samples (Figure 3). In the muscle, two MeO-BDE congeners (5-MeO-BDE47 and 4'-MeO-BDE49) were detected, and the concentrations of 5-MeO-BDE47 were slightly higher than those of 4'-MeO-BDE49. Statistical data showed that there was no significant difference in the concentrations of the two congeners between the LG and HG groups ($p > 0.05$, significant at the 95% confidence level). Two congeners in the muscle had a similar trend in the temporal change during the monitored time. In the liver, two MeO-BDE congeners (5-MeO-BDE47 and 5'-MeO-BDE100) were detected, but the concentration levels were much higher than those in the muscle. For 5'-MeO-BDE100, there was no significant difference between the LG and HG groups. However, 5-MeO-BDE47 concentrations were observed exponentially accumulated, which were fitted to first-order exponential growth model. The predicted values of the exponential growth rate constants (k_b) in LG and HG were 0.076/d and 0.289/d, respectively. It was found that the MeO-BDE metabolites in the liver were accumulated more rapidly in HG than in LG.

4 Discussion

4.1 Toxicokinetics of BDE209

The metabolism of BDE209 in the environment has become a serious concern and merits more investigations. Kierkegaard *et al.* [23] preliminarily studied the uptake of BDE209 in caged rainbow trout with dietary exposure, and the results showed that the uptake of BDE209 was slow but measurable. Several lower brominated PBDEs were also measured, suggesting that BDE209 could be metabolized by debromination. In addition, the accumulation and debromination of

Figure 3 Temporal change of MeO-BDEs in tissues of rainbow trout. 5-MeO-BDE47-LG=2,2',4,4'-tetrabromo-5-methoxydiphenyl ether in the low-dose group; 4'-MeO-BDE49-LG = 2,2',4,5'-tetrabromo-4'-methoxydiphenyl ether in the low-dose group; 5-MeO-BDE47-HG = 2,2',4,4'-tetrabromo-5-methoxydiphenyl ether in the high-dose group; 4'-MeO-BDE49-HG = 2,2',4,5'-tetrabromo-4'-methoxydiphenyl ether in the high-dose group; 5'-MeO-BDE100-LG=2,2',4,4',6-pentabromo-5'-methoxydiphenyl ether in the low-dose group; 5'-MeO-BDE100-HG = 2,2',4,4',6-pentabromo-5'-methoxydiphenyl ether in the high-dose group.

BDE209 in juvenile carp were assessed by Stapleton *et al.* [24]. However, there is a lack of information on the toxicokinetics of BDE209 in previous studies. The present study extensively investigated the toxicokinetics of BDE209 in rainbow trout. BDE209 metabolism in the liver and muscle both followed the one-compartment model with extravascular administration, which was similar to the toxicokinetics of BDE47 in Japanese Medaka [32]. Toxicokinetic model-predicted values of the absorption and elimination rate (k_a , k_e), peak concentration (C_{max}) and peak time (T_{max}), and biological half-life ($t_{1/2}$) for the muscle and liver of rainbow trout are listed in Table 1. Both the absorption and elimination rates in the liver were higher than those in the muscle for the same dose group, suggesting that the liver might have a higher metabolic activity. Half-lives of BDE209 in the liver were shorter than those in the muscle, especially for HG, half-lives of BDE209 in the liver and muscle were found to be 17.7 d and 100.3 d, respectively.

Microsomal fractions have been previously used to investigate the biotransformation of organic compounds by fish [33, 34]. Liver microsomes of rainbow trout and common carp incubated with BDE209 confirmed a debromination

Table 1 Predicted values of toxicokinetic parameters by the one-compartment model

Tissue	Dose group	k_a (1/d) ^{a)}	k_e (1/d) ^{b)}	C_{max} (ng/g) ^{c)}	T_{max} (d) ^{d)}	$t_{1/2}$ (d) ^{e)}
Muscle	LG ^{f)}	0.129	0.009	106.9	1.15	75.2
	HG ^{g)}	92.161	0.007	791.7	4.13	100.3
Liver	LG ^{f)}	22.892	0.014	128.9	3.22	50.1
	HG ^{g)}	429.924	0.039	568.4	4.04	17.7

a) First-order absorption rate constant (k_a); b) first-order elimination rate constant (k_e); c) peak concentration of BDE209 (C_{max}); d) time of peak concentration occurrence (T_{max}); e) half-life ($t_{1/2}$); f) the low-dose group (LG); g) the high-dose group (HG).

pathway of BDE209 metabolism, which was supported by *in vivo* experiments [22]. The biological parameters such as the half-life ($t_{1/2}$) of BDE209 derived from the model were in agreement with the previous studies in fish (Table 2). A comparison of $t_{1/2}$ among BDE209 and the lower brominated BDE in different species indicated that BDE209 was metabolized at a relatively higher rate than lower brominated BDE congeners, especially in mammals.

4.2 Metabolites of BDE209 in rainbow trout

The debrominated diphenyl ether metabolites of BDE209 have been extensively studied among different species, including terrestrial and marine animals [6]. In the present study, debrominated diphenyl ether concentration levels in the liver were higher than those in the muscle tissue, which was in accordance with the previous studies. The comparison of debrominated diphenyl ether congeners in rainbow trout with other species showed that there was a little difference of debrominated diphenyl ether metabolites among different species [27]. Such difference of debrominated diphenyl ether metabolites might depend on the exposure route and tested species [39].

Environmental research on MeO-BDEs has thus far provided an appreciable body of evidence suggesting that several of these compounds occur naturally in marine environments, and they can also be formed from O-methylation of hydroxylated PBDEs (OH-PBDE) [40]. Previous studies

about the MeO-BDEs mostly focused on the environmental samples collected from the natural environmental sites or the contaminated sites [41, 42]. In contrast to the previous research, the present study identified and quantified the MeO-BDE metabolites in fish after exposure to BDE209 under experimental conditions. The results showed among the detected MeO-BDE metabolites, 5-MeO-BDE47 showed a significant accumulation trend in the liver throughout the experiment, which is probably a significant indicator for BDE209 toxicology in fish. While in the muscle, no significant difference was observed throughout the experiment. The different distribution patterns of MeO-BDE metabolites in rainbow trout represented different pathways and different metabolic abilities of BDE209 in the muscle and liver. We preliminarily concluded that the liver was a primary metabolic organ of BDE209 in rainbow trout.

5 Conclusions

Toxicokinetic profiles of BDE209 in rainbow trout were fitted to one-compartment model. The predicted half-life values indicated that BDE209 was eliminated more rapidly in the liver than in the muscle. BDE209 was eliminated at a higher rate than lower brominated BDE congeners. Concentration levels of debrominated diphenyl ether and MeO-BDE metabolites in the liver were higher than those in the muscle, suggesting that the metabolic ability of the liver

Table 2 Comparison of half-lives ($t_{1/2}$) between BDE209 and lower brominated BDEs in different test species

Species	Class	Tissue/Organ	Compound	$t_{1/2}$ (d)	Ref.
European starlings	Aves	blood	BDE209	11–18	[21]
Rats	Mammalia	plasma	BDE209	2.5	[35]
Gray seal	Mammalia	blood	BDE209	8–13	[36]
Human	Mammalia	serum	BDE209	15	[37]
Lake trout	Osteichthyes	whole body	BDE209	26	[38]
Rainbow trout	Osteichthyes	liver	BDE209	18	this study
Human	Mammalia	serum	BDE183	94	[37]
Juvenile carp	Osteichthyes	liver	BDE154	35	[24]
Lake trout	Osteichthyes	whole body	BDE154	139	[38]
Lake trout	Osteichthyes	whole body	BDE47	346	[38]

was higher than that of the muscle. An increasing trend of 5-MeO-BDE47 in the liver might be an indicator for fish toxicology of BDE209. Further research is needed to thoroughly understand the metabolism pathway of BDE209 in rainbow trout.

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