Review

Species-specific dechlorane plus isomer fractionation during bioaccumulation: phenomenon and potential mechanisms

Ke-Lan Guan¹², Hong-Ying Liu³, Xiao-Jun Luo¹

¹State Key Laboratory of Organic Geochemistry and Guangdong Key Laboratory of Environmental Resources Utilization and Protection, Guangzhou Institute of Geochemistry, Chinese Academy of Sciences, Guangzhou 510640, Guangdong, China.
²University of Chinese Academy of Sciences, Beijing 100049, China.
³College of Chemistry and Chemical Engineering, Hubei University, Wuhan 430062, Hubei, China.

Correspondence to: Dr. Xiao-Jun Luo, State Key Laboratory of Organic Geochemistry and Guangdong Key Laboratory of Environmental Resources Utilization and Protection, Guangzhou Institute of Geochemistry, Chinese Academy of Sciences, Guangzhou 510640, Guangdong, China. E-mail: luoxiaoj@gig.ac.cn

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Abstract

The occurrence and behavior of dechlorane plus (DP), an additive chlorinated flame retardant, have been intensively studied since it was identified in 2006. The commercial products of DP are a mixture of two stereoisomers: syn-DP and anti-DP. Stereoselective bioaccumulation of DP isomers in biota was reported in field monitoring and laboratory experiments. This review summarizes stereoselective bioaccumulation of DP in biota samples and provides the potential mechanisms for this stereoselective bioaccumulation. Stereoselective enrichment of syn-DP was widely observed in fish, whereas selective enrichment of anti-DP was mainly found in some birds. This species-specific stereoselective enrichment of DP might reflect that two different types of DP isomer fractionation occurred in bioaccumulation between ectotherms and endotherms. Anti-DP is more readily metabolized through biotransformation in all animals. However, a preferential excretion of anti-DP in fish and syn-DP in birds was observed based on the available data. Both processes determine the DP isomer fractionation in bioaccumulation. A direct comparison in DP composition between biological samples and commercial products was conducted for most studies to determine the occurrence of stereoselective DP enrichment, which may lead to underestimating the potential stereoselective enrichment of DP in organisms. The factors which affected the DP isomer composition in organisms included the tissues or organs used, DP concentration, organisms’ trophic levels occupied, and sex. Inconsistent results were obtained considering the effects of these influence factors. The
underlying cause of these inconsistent results is unclear based on present data. Further research on DP biotransformation and interactions between DP and biomacromolecule is needed.

**Keywords:** Dechlorane plus, bioaccumulation, stereoselective enrichment, species-specific

**INTRODUCTION**

Dechlorane plus (DP), also called bis(hexachlorocyclopentadiene)cyclooctane, is a type of chlorinated additive flame retardant, which has been widely used in textiles, paints, circuit boards, and especially in the plastic polymers of electrical appliances, such as carbons, wires, computer connectors, etc. DP makes up 10%-35% of the components in some commercial polymer products [1].

As an additive chlorinated flame retardant, DP is inevitably released into the environment through production, usage, and recycling [1]. DP was first identified in sediment and fish samples from the Great Lakes of North America in 2006 [2]. Since then, DP has been widely reported in environmental and biological matrices worldwide, indicating that it is widespread in the environment [3,4]. According to toxicity research, oral exposure to DP can lead to hepatic oxidative damage, perturbations of metabolism, and signal transduction for male mice [5]. A high concentration and extended exposure to DP induced oxidative damage and neurotoxicity on earthworms (*Eisenia fetida*) [6]. DP has been identified as a Substance of Very High Concern by the European Chemicals Agency (ECHA) and is being reviewed for addition to the list of the Stockholm Convention on Persistent Organic Pollutants [7].

Intensive research has been conducted over the past decade on the occurrence and behavior of DP in environment, biota, and humans [4,8-12]. Species-specific stereoselective enrichment of DP isomers was observed in previous field monitoring and laboratory experiments. However, the stereoselective enrichment of DP in organisms and its potential mechanisms are rarely discussed comprehensively. This study aims to review the current knowledge about DP bioaccumulation; summarize research results, including species-specific stereoselective enrichment in different organisms and existing problems in the current studies; and guide future research. The keywords “dechlorane plus” and “organisms” were used to perform a literature search in the Web of Science. Literature reporting the change of DP isomer composition in the environmental matrix is also included, and 80 articles are covered in this review.

**ISOMER COMPOSITION OF DP COMMERCIAL PRODUCTS AND ITS CHANGE IN ENVIRONMENT**

DP was developed by Hooker Chemical (presently OxyChem) in the 1960s as a substitute for Mirex, and its annual production is estimated to be 450-4500 tons. In China, Jiangsu Anpon Electrochemical Co., Ltd has manufactured 300-1000 tons of DP annually since 2003 [1]. DP commercial products contain two stereoisomers (syn- and anti-DP, Figure 1); the ratio of syn- and anti-DP is approximately 1:3; i.e., the fraction of anti-DP \[ f_{anti} = anti-DP/(anti-DP + syn-DP) \] is 0.75 [2].

Most of the commercial product and analytical standard \( f_{anti} \) values measured by researchers ranged from 0.7 to 0.8 [Table 1], which are consistent with the theoretical ratio (0.75). A low ratio such as 0.65 for commercial products by OxyChem [14] and 0.60 for products from Anpon Electrochemical were also reported [15]. Differences in the \( f_{anti} \) values may be related to the composition of different production batches.
Table 1. $f_{anti}$ values of technical DP mixtures

| Products                        | $f_{anti}$ | References          |
|---------------------------------|------------|---------------------|
| OxyChem                         | 0.75-0.80  | Hoh et al., 2006[2] |
|                                 | 0.75       | Qiu et al., 2007[11]|
|                                 | 0.65       | Tomy et al., 2007[14]|
| Anpon Electrochemical           | 0.60       | Wang et al., 2010[13]|
|                                 | 0.75, 0.78 | Luo et al., 2013[10]|
|                                 | 0.70       | Wu et al., 2010[6]  |
| Cambridge Isotope Laboratories  | 0.74-0.76  | Zhu et al., 2007[7] |
|                                 | 0.75-0.77  | Gauthier and Letcher, 2009[18]|
|                                 | 0.75       | Kang et al., 2010[9]|

Figure 1. Chemical structure of DP isomers.

Information on DP environment behavior remains scarce. A significant change in $f_{anti}$ was observed after long-distance transport. Möller et al. observed that $f_{anti}$ decreased from 0.63 to 0.37 with increasing transport distance during transport from Europe to the Arctic and Antarctica through sea and air[20]. Stereoselective photodegradation of anti-DP through UV light during long-range transport was proposed as the main cause for this alternation. Zheng et al. compared the DP composition in indoor dust collected from an e-waste site and two control areas (rural and urban) and found that the indoor dust in the e-waste recycling workshops had an average $f_{anti}$ value of 0.54, which was significantly lower than indoor dust from the residences in the rural (0.76) and urban control areas (0.70)[21]. The $f_{anti}$ value of indoor dust from residences living in the e-waste site (0.66) was between that of the e-waste recycling workshops and control area. The ratios of dechlorination product of anti-DP (anti-Cl$_1$-DP) to anti-DP were also provided in this study. The average ratio in the indoor dust from the e-waste workshops (0.014) was one order of magnitude higher than that (0.0012) in the indoor dust from the residences, indicating a stereoselective degradation of anti-DP in e-waste recycling process.

The above results indicate that the composition of DP can be influenced by environmental processes. Thus, a direct comparison in DP composition between organisms and industrial products is unreasonable for determining the occurrence of stereoselective accumulation. The result is reliable when the $f_{anti}$ values of the environmental matrix and food fed to organisms are simultaneously provided.

ISOMER COMPOSITION OF DP IN BIOTA SAMPLES

Regarding the DP stereoselective bioaccumulation in organisms, three categories are reported in the literature: syn-DP enrichment, anti-DP enrichment, and no clear stereoselective enrichment. The literature is summarized as follows according to the standard of whether a conclusion was given by the authors of the study.
Syn-DP enrichment in organisms

Hoh et al. first reported the occurrence and composition of DP in sediments and fish of the Great Lakes of North America in 2006[31]. The average $f_{\text{anti}}$ of fish was found to be 0.60, which was significantly lower than that of sediments ($P < 0.001$), indicating a selective enrichment of syn-DP in fish. Subsequently, numerous studies have reported that, compared to anti-DP, syn-DP is preferentially accumulated in fish worldwide [Table 2], although a few studies reported inconsistent results. For example, the anti-isomer was reported to be dominant in walleye and goldeye in Lake Winnipeg; the $f_{\text{anti}}$ in Lake Michigan fish ($0.82 \pm 0.15$) reported by Guo et al. was higher than that of the technical DP mixture. Süähring et al. reported an evident enrichment of syn-DP in eels ($f_{\text{anti}} = 0.4-0.48$) in German rivers, which was consistent with that of Hoh et al.[2,14,21,26]. However, another study on the bioaccumulation and maternal transfer of DP in the European eels from Ems and Schlei Rivers reported an anti-DP enrichment in eels[39]. The former study predominantly analyzed yellow eels (no mature), while the latter used mature silver eels. The difference in the observed $f_{\text{anti}}$ patterns can be attributed to the difference in maturation. The selective maternal transfer of syn-DP to egg and gonad tissues was observed in silver eels, and hence the proportion of anti-DP in the mother’s body increased.

The $f_{\text{anti}}$ values used for comparison with those of fish included both $f_{\text{anti}}$ values in sediments where fish were collected and in technical DP mixtures. The $f_{\text{anti}}$ values in sediments reported in the literature are consistent with those reported in the technical DP mixtures, which indicate no obvious stereoselective degradation of DP in the sediments. Thus, a direct comparison between fish, which are bottom-dwelling, sedentary, and feed on benthic organisms, and technical DP mixtures can provide a reliable result on the stereoselective accumulation of syn-DP in fish.

In addition to fish, some aquatic organisms at high trophic levels, such as seals, water snakes, dolphins, whales, and water birds, were also reported to show selective enrichment of syn-DP [Table 2]. However, the $f_{\text{anti}}$ values used for comparison were those of sediment or industrial products, but not their prey. Generally, fish are the main prey of these high trophic level organisms, and it is not credible to draw a syn-DP enrichment conclusion from these organisms because of the low $f_{\text{anti}}$ that they may acquire from their prey.

Some other studies also reported a syn-DP enrichment in organisms. However, after carefully checking the data, these statements were found to be incorrect. For example, the $f_{\text{anti}}$ value of oysters collected in the coastal area of Dalian, China, was 0.55, which is lower than that of industrial products; this was considered a selective enrichment of syn-DP[40]. However, the value of the surrounding sediment was 0.56, which was similar to that in the oysters. Thus, no stereoselective accumulation occurred in this species. Na et al. collected alga, limpet, starfish, gammarid, krill, cod, penguin, seal, and skua samples in Fildes Peninsula in Antarctica and found that the $f_{\text{anti}}$ value of DP ranged from 0.23 to 0.53[41]. They believed that these lower $f_{\text{anti}}$ values than those of industrial products implied a selective removal of anti-DP or selective enrichment of syn-DP during long-distance migration. However, compared with the previously reported $f_{\text{anti}}$ value (0.35) of the surrounding seawater, there was no obvious tendency of syn-DP selective enrichment in organisms.

Amphibians and terrestrial organisms were also reported to enrich syn-DP selectively. Wu et al. collected frogs in an e-waste recycling area in Qingyuan, Guangdong Province[42]. The $f_{\text{anti}}$ values in muscle (0.65), liver (0.58), and egg (0.53) were significantly lower than the average $f_{\text{anti}}$ value of its prey item (insects: 0.73), indicating a selective enrichment of syn-DP in frogs. Venier et al. collected serum samples from four American pet dogs and found that their $f_{\text{anti}}$ value was $0.61 \pm 0.08$, which was lower than that in the dog food ($0.76 \pm 0.02$), implying a slight syn-DP enrichment[43]. Given the limited number of samples, the authors warned that the conclusion could not be over interpreted. Chen et al. collected biological samples (plants,
Table 2. Enrichment of syn-DP in organisms reported in the literature

| Regions | Organisms | Reference | Reference |
|---------|-----------|-----------|-----------|
| Lake Ontario | Trout: 0.44-0.58 (1979-2004) | Sediment: 0.76-0.86 (1980-2004) | Shen et al., 2011[22] |
| Lake Ontario | Fish: 0.65±0.06 | Technical DP mixture: 0.75 | Guo et al., 2017[23] |
| Superior | Fish: 0.63±0.07 | | |
| Erie | Fish: 0.60±0.07 | | |
| Lake Michigan | Fish: 0.82±0.15 | | |
| Iberian river basins in Spain | Fish: 0.48 ± 0.16 | Sediment: 0.64-0.80 | Santín et al., 2013[24] |
| France | catfish: 0.60 ± 0.12 | Industrial products: 0.75 | Malak et al., 2018[25] |
| River Rhine | Eels: 0.04-0.48 | Industrial products: 0.75 | Sühring et al. 2013[26] |
| E-waste site in Qingyuan, South China | Fish: 0.14-0.68 | Sediment: 0.72 | Wu et al., 2010[26] |
| The Pearl Rivers, China | Mud carp: 0.60 | Sediment: 0.77 | He et al. 2014[27] |
| | Nile tilapia: 0.59 | | |
| | Plecostomus: 0.70 | | |
| The Pearl Rivers, China | Mud carp: 0.52 | Technical products: 0.65-0.80 | Sun et al., 2016[28] |
| | Nile tilapia: 0.57 | | |
| | Plecostomus: 0.65 | | |
| The Yellow River Delta, China | Fish and shellfish: 0.56-0.60 | Technical products: 0.65-0.80 | Zhang et al. 2020[29] |
| Daling river in northeastern China | Fish: 0.53 | Water: 0.72 | Wang et al., 2012[30] |
| | | Sediment: 0.75 | |
| | | Reed: 0.73 | |
| E-waste site in Guiyu, South China | Fish: 0.56-0.68, average of 0.65 | Surrounding soil: 0.67-0.83, average of 0.76. | Tao et al. 2015[31] |
| Urban river in South Korea | Fish: 0.67 ± 0.06, | Technical products: 0.75 | Kang et al. 2010[32] |
| Japanese market | Fish: 0.62 ± 0.05 | Technical products: 0.75 | Kakimoto et al. 2012[33] |
| The coast of Conception, Chile | Fish: 0.47-0.61 | Technical products: 0.75 | Barón et al., 2013[34] |
| San Francisco Bay, USA | Seals: 0.43-0.49 | Sediment: 0.65 | Klosterhaus et al. 2012[35] |
| E-waste site in Qingyuan, South China | Water snake: 0.41 | Sediment: 0.72 | Wu et al. 2010[36] |
| Southern European waters | Short-beaked common dolphin: 0.37 | Technical products: 0.75 | Barón et al., 2015[35] |
| | Bottlenose dolphin: 0.49 | | |
| | Long-finned pilot whale: 0.49 | | |
| E-waste Site in Qingyuan, South China | Five water birds: 0.34-0.61 | Technical products: 0.75 | Zhang et al., 2011[36] |
| DP production plant, China | Human serum and hair: 0.51-0.61 | Technical products: 0.68 | Zhang et al. 2013[37] |
| Minzu University of China | Human hair: 0.727 | Dust in house: 0.78-0.79 | Chen et al. 2019[38] |

Anti-DP enrichment in organisms

insects, birds, reptiles, and mammals) and abiotic samples (air, water, and soil) at two sites in Xilingol, Inner Mongolia[44]. They found that the $f_{anti}$ of plant samples (0.68) were lower than those in water (0.72), soil (0.73), and air (0.71), showing syn-DP enrichment. The $f_{anti}$ in ectotherms, including lizards, toads, and snakes (0.44-0.61), was significantly lower than that of the surrounding environmental matrix, indicating a selective enrichment of syn-DP.

Two studies reported possible syn-DP enrichment in human samples. Zhang et al. collected human serum and hair samples from a DP production plant and its surrounding areas in China[37]. The $f_{anti}$ value of serum and hair of workers was between 0.54 and 0.61, lower than the $f_{anti}$ value of factory products (0.68). Chen et al. analyzed the composition of DP in hair, dormitory dust, and classroom dust of students of Minzu University of China and found that the $f_{anti}$ value in hair (0.727) was significantly lower than that in dormitory dust (0.791) and classroom dust (0.783), indicating a syn-DP enrichment in hair[38].

Anti-DP enrichment in organisms

Organisms reported to enrich anti-DP were mainly birds. Zheng et al. analyzed absorption and tissue distribution of DP in chickens raised in an e-waste recycling site[45]. Soil in the yard was found to be the main source of DP in the chickens. The elevated $f_{anti}$ in chickens (muscle, liver, brain, and fat: 0.64-0.65)
compared with the soil (0.52) implied a stereoselective enrichment of anti-DP in chickens. No significant difference in \( f_{\text{anti}} \) values was observed in chyme, intestinal contents, and feces \((P > 0.05)\), indicating no stereoselective absorption of DP isomers during gastrointestinal absorption. Sun et al. found that the \( f_{\text{anti}} \) value in muscle and liver for three bird species (light-vented bulbul, 0.80 ± 0.017; long-tailed shrike, 0.78 ± 0.01; and oriental magpie-robin, 0.75 ± 0.01) was higher than in dust samples (0.70) in the Pearl River Delta\(^{46}\). In the Pearl River Delta, the eggs of three terrestrial birds (light-vented bulbul, 0.80-0.91; yellow-bellied prinias, 0.77-0.94; and dark green white-eyes, 0.79-0.83) showed higher \( f_{\text{anti}} \) than that of commercial mixtures and dust\(^{47}\).

In the southwestern Mediterranean, the \( f_{\text{anti}} \) value for yellow-legged gulls ranged from 0.73 to 0.85, with an average of 0.81, whereas the \( f_{\text{anti}} \) value for Audouin’s gulls ranged from 0.70 to 1.00, with an average of 0.82\(^{48}\). A slight enrichment of anti-DP was anticipated in both species. The \( f_{\text{anti}} \) values ranged 0.71-0.78 in penguin samples and 0.69-0.89 in skua samples from King George Island in Antarctica, which were slightly higher than those of commercial mixtures\(^{50}\). In addition, Kim et al. reported the \( f_{\text{anti}} \) of DP in organisms of King George Island in Antarctica\(^{50}\). The \( f_{\text{anti}} \) values of limpets and Antarctic cod were 0.68 ± 0.24 and 0.57 ± 0.11, respectively. Gentoo and Chinstrap penguin samples had \( f_{\text{anti}} \) values of 0.74 ± 0.20 and 0.65, respectively, while the Antarctic icefish and the south polar skua had \( f_{\text{anti}} \) values of 0.71 and 0.79, respectively. The \( f_{\text{anti}} \) values could have decreased because of diasteromerization during long-range transportation; however, because of bioaccumulation, the \( f_{\text{anti}} \) values in biota samples, even in remote regions, were observed to be similar to or higher than those in commercial mixtures, indicating enrichment of anti-DP\(^{11}\).

An enrichment of anti-DP was also reported in human samples. Chen et al. collected the serum and hair of workers in e-waste recycling workshops and observed that the average \( f_{\text{anti}} \) value in the serum (0.65) was higher than that in the hair (0.47) of the workers and the dust (0.54) in the workshops, indicating a selective enrichment of anti-DP in the human body\(^{51}\). Yan et al. also reported a similar result. The \( f_{\text{anti}} \) values in the serum were 0.67 ± 0.07 for female samples and 0.63 ± 0.06 for male samples\(^{52}\). These values were higher than those found in human hair (0.55 ± 0.11) and dust samples (0.54) in the workshops\(^{21}\).

Chen et al. found that ectotherms, including lizards, toads, and snakes, stereoselectively enriched syn-DP when compared to the surrounding environmental matrix (Section 2.1)\(^{44}\). However, endotherms such as birds [Cuckoo (0.70) and swallow (0.72)] and weasels (0.66) exhibited elevated \( f_{\text{anti}} \) values when compared with their prey [insect (0.66) and mouse (0.58)], indicating an anti-DP enrichment. This is an intriguing study, which provided some insight into the stereoselective bioaccumulation of DP.

**Unclear DP stereoselective enrichment in organisms**

In addition to the selective enrichment of syn/anti-DP, some studies also reported no obvious selective enrichment of DP in organisms [Table 3]. Rjabova et al. reported that anti-DP has a significantly higher detection frequency than does syn-DP in salmon collected in the Baltic Sea, while samples containing both isomers showed \( f_{\text{anti}} \) values ranging from 0.56 to 0.94, with an average value of 0.71, which was close to the \( f_{\text{anti}} \) value of technical DP mixtures\(^{53}\).

Several fish-feeding bird samples exhibited similar DP composition as DP commercial products, and it was considered that there was no selective enrichment of stereoisomers of DP. Since the preferential enrichment of syn-DP was found in fish, the selective enrichment of anti-DP appeared to be the reason for the \( f_{\text{anti}} \) ratio of fish-eating birds being close to or higher than that of industrial products. For example, Mo et al. found that the \( f_{\text{anti}} \) values of kingfishers in an e-waste recycling region were similar to those of technical DP.
mixtures (0.60-0.80)\cite{57}. However, the fish had $f_{\text{anti}}$ values ranging from 0.18 to 0.47 in this region. An enrichment of $\textit{anti}$-DP was expected in kingfishers when compared with fish. This was also expected in other fish-feeding birds considering the stereoselective enrichment of $\textit{syn}$-DP in fish.

Chicken eggs ($f_{\text{anti}}$: 0.63-0.74) collected from an e-waste recycling site in Qingyuan were also expected to exhibit unclear DP stereoselective enrichment\cite{56}. However, the average $f_{\text{anti}}$ values in the surrounding soil and dust samples were 0.52 and 0.54, respectively\cite{21,45}. Compared with these samples, an enrichment of $\textit{anti}$-DP in chicken eggs was evident. Raptors collected in Beijing, China, were found to have similar $f_{\text{anti}}$ values (average: 0.79) compared with those of their prey (0.75- 0.77)\cite{59}. Terrestrial passerines collected from a nature reserve in South China exhibited a wide range (0.34-0.97) of $f_{\text{anti}}$ values\cite{58}. It was difficult to determine the stereoselective enrichment occurrence for these terrestrial birds.

Considering human samples, determining selective enrichment was impossible because no information on the isomer composition of DP in the exposure source is available. Overall, the credibility of the conclusion based on a direct comparison between organisms and technical DP mixture is questionable. The potential stereoselective enrichment of DP in organisms may be underestimated if organisms stereoselectively enrich one DP isomer but their prey enriches the other one.

**FACTORS INFLUENCING THE DP FRACTIONATION DURING BIOACCUMULATION**

The following factors can affect DP composition measured in the organisms discussed above.

(1) **Tissue or organ**: Values of $f_{\text{anti}}$ were found to be tissue- or organ-specific in a given species. Zhang \textit{et al.} reported that $\textit{anti}$-DP was preferentially accumulated in the brain compared to the liver and muscle for mud carp and snakehead, suggesting that the $\textit{anti}$-isomer can cross the blood-brain barrier of fish and has

| Areas | Organisms | $f_{\text{anti}}$ | Reference $f_{\text{anti}}$ | Reference |
|-------|-----------|-------------------|-----------------------------|-----------|
| Daugava and Venta rivers | Baltic wild salmon: 0.71 (0.56-0.94) | Technical DP mixture | Rjabova \textit{et al.}, 2016\cite{53} |
| Great Lakes of North America | Eggs of herring gulls: 0.69 ± 0.08 | Technical DP mixture | Gauthier and Letcher \textit{et al.}, 2009\cite{18} |
| Madrid in Spain Doñana National Park | White stork eggs: 0.64 | Technical DP mixture | Muñoz-Arnanz \textit{et al.}, 2011\cite{54} |
| St. Lawrence River, Canada | Ring-billed gull liver: 0.72 | Technical DP mixture | Gentes \textit{et al.}, 2012\cite{55} |
| An e-waste site in Qingyuan, South China | Kingfishers: 0.65 (0.56-0.72) | Fish: 0.18-0.47 | Mo \textit{et al.}, 2013\cite{57} |
| A nature reserve in South China | Terrestrial passerines: 0.34-0.97 | / | Peng \textit{et al.}, 2015\cite{56} |
| Beijing, China | Scops owl: only anti-DP detected or 0.60-0.80 | / | Chen \textit{et al.}, 2013\cite{59} |
| Beijing, China | Common kestrel: 0.79 | Sparrow: 0.75 | Yu \textit{et al.}, 2013\cite{60} |
| E-waste site in Guiyu, South China | Chicken egg: 0.74-0.76 | Brown rat: 0.77 | Zeng \textit{et al.}, 2016\cite{61} |
| Southeastern and Southern Coast of Brazil | Franciscana Dolphin: 0.71 ± 0.16 | Technical DP mixture | de la Torre \textit{et al.}, 2012\cite{62} |
| E-waste site in Guiyu, South China | Human serum: 0.58 ± 0.11 | / | Ren \textit{et al.}, 2009\cite{63} |
| Kingston and Sherbrooke, Canada | Human milk: 0.67 | Technical DP mixture | Siddique \textit{et al.}, 2012\cite{64} |
| France | Human serum: 0.75 (0.65-0.86) | Technical DP mixture | Brasseur \textit{et al.}, 2014\cite{65} |
high affinity to the brain\cite{66}. Peng et al. found that the \( f_{\text{anti}} \) values in the heart and eggs of Chinese sturgeon (0.58 and 0.65, respectively) were significantly lower than those in the liver (0.72) and muscle tissue (0.72)\cite{67}. Zheng et al. investigated the tissue distribution of DP in chickens in an e-waste area and found elevated \( f_{\text{anti}} \) values in the fat, brain, and liver (0.65, 0.64, and 0.64, respectively) compared with other tissues (0.54-0.59)\cite{67}. Wu et al. found that frog eggs exhibited remarkably lower \( f_{\text{anti}} \) (0.53) than frog muscle (0.58) and liver (0.65), indicating preferential transfer of syn-DP during maternal transmission\cite{42}. This was consistent with the lower \( f_{\text{anti}} \) value in Chinese sturgeon eggs than in other tissues.

(2) Trophic level: Wu et al. first reported that \( f_{\text{anti}} \) values decreased up the trophic ladder in the aquatic food chain\cite{16}. Subsequently, a significantly negative correlation between \( f_{\text{anti}} \) values and the trophic level of organisms in aquatic and terrestrial food chains in the same region was also reported\cite{16,68}. A significantly negative linear relationship between \( f_{\text{anti}} \) fraction and stable nitrogen isotope ratio (\( \delta^{15} \)N) of biological individuals was also observed in three terrestrial birds collected in different regions of the Pearl River Delta\cite{46}. In a food chain composed of seven species of aquatic organisms collected from Huai’an in Jiangsu province of China, Wang et al. found that shrimp occupied the lowest trophic level with the highest \( f_{\text{anti}} \) value of 0.81, while snakes occupied the highest trophic level with the lowest \( f_{\text{anti}} \) value of 0.51\cite{69}. There was a significantly negative linear correlation between \( f_{\text{anti}} \) values and the trophic level of organisms. However, an inconsistent result was observed in the food chain of Fildes Peninsula by Na et al.: the \( f_{\text{anti}} \) first increased and then decreased with increasing trophic levels. The \( f_{\text{anti}} \) value decreased through the food chains composed of ectotherms, while the value initially decreased before increasing in the food chains composed of endotherms\cite{41,44}.

(3) DP concentration: Mo et al. found that there was a significant negative relationship between DP composition and its concentration in kingfishers and fish and proposed that DP concentration could be an important factor influencing isomeric fraction\cite{57}. A similar phenomenon was also found in terrestrial bird eggs and muscles in the Pearl River Delta. However, \( f_{\text{anti}} \) values increased with increasing DP burdens in muscle and liver in Eurasian sparrowhawk, which is in contrast with the above results\cite{46,58,59}.

(4) Gender: Yan et al. collected serum samples from 33 male and 37 female workers in e-waste recycling plants\cite{52}. Samples from female workers exhibited significantly higher DP concentrations (mean: 265 ng/g lw) and \( f_{\text{anti}} \) values (median: 0.70) than those from the males (121 ng/g lw, and 0.64, respectively). Additionally, the ratios of anti-Cl\(_{11}\)-DP to anti-DP were remarkably higher in males (mean: 0.017) than in females (mean: 0.010). Lower DP loading and lower \( f_{\text{anti}} \) values in males can be ascribed to males having stronger metabolic ability for DPs than females and anti-DP having a higher metabolism capacity in organisms. Studies on maternal transfer of DP in Chinese sturgeons\cite{67}, eels\cite{39}, and frogs\cite{42} revealed that syn-DP more readily transferred to eggs than does anti-DP. This may cause higher \( f_{\text{anti}} \) values in females than in males. Wu et al. verified this hypothesis in their work, wherein the observed concentration of DP in female frogs was significantly lower than that in male frogs, while the \( f_{\text{anti}} \) value was higher than that in male frogs\cite{42}.

**POTENTIAL MECHANISMS REVEALED BY LABORATORY EXPOSURE EXPERIMENT**

A series of laboratory experiments have been conducted to explore the mechanism of DP fractionation during bioaccumulation.

**Exposure experiments in fish**

Tomy et al. first exposed juvenile rainbow trout to DP isomers through their diet for 49 days (uptake phase) before feeding them with untreated food for 112 days (depuration phase), were conducted by L\cite{70}. Syn- and
anti-isomer loads increased during the entire uptake phase and did not reach plateau during the uptake phase. The uptake rates for syn-DP and anti-DP were 0.045 ± 0.005 and 0.018 ± 0.002 nmol/day, respectively, while the measured half-lives were 53.3 ± 13.1 days for syn-isomer and 30.4 ± 5.7 days for anti-isomer. These results verify syn-DP selective accumulation in the fish species. No suspected degradation products, including dechlorinated, hydroxylated, methoxylated, and methyl sulfone metabolites, were detected in the liver samples, indicating that in vivo biotransformation of DP could hardly produce these metabolites.

Common carp was exposed to DP industrial products and DP isomers successively in two studies[71,72]. Despite differences in exposure and depuration times, sampling intervals, and sample collection, both studies found higher absorption efficiency of anti-DP in the gastrointestinal tract but significantly lower assimilation efficiency of anti-DP, indicating the stereoselective metabolism of anti-DP in fish. DP tissue distribution was a dynamic process. During the uptake phase, the DP concentration in liver was remarkably higher than those in other tissues, indicating a selective accumulation of DP in the liver. However, in the depuration period, the highest elimination rate of DP was found in the liver, which subsequently reduced the concentration gap between liver and other tissues. The liver showed preferential enrichment of antiisomer, intestine and muscle preferred syn-DP, and the whole fish exhibited selective accumulation of syn-DP. Tang et al. observed an increasing f_anti value trend in the feces of common carp along with a decreasing trend of f_anti value in the whole fish during the depuration phase, indicating that selective excretion of anti-DP was likely the primary reason for the low f_anti values[72].

These three DP exposure experiments confirmed the selective enrichment of syn-DP in fish. Despite no direct evidence of selective degradation of anti-DP, there was clear evidence of selective excretion of anti-DP. Therefore, the selective excretion of anti-DP should be an important factor for enriching syn-isomer in fish, but selective degradation of anti-DP also cannot be ruled out.

**Exposure experiment in bird and mammal**

Sprague-Dawley rats[73] and male common quails[74] were consecutively exposed to commercial DP for 90 days at 0, 1, 10, and 100 mg/kg/day doses and endured 45 days of depuration to investigate the DP composition in the liver, muscle, and serum. The studies found that the liver had the highest DP concentration among all tissues, implying that DP preferred to accumulate in the liver. DP composition was found to be dose dependent. The f_anti values (0.7) in the low-exposure group (1 mg/kg/day) were close to those in DP industrial products, while the f_anti values (0.16-0.34) in the high-exposure groups (10 and 100 mg/kg/day), were significantly lower than DP industrial products. The activity of erythromycin N-demethylase (ERND) and the antioxidant enzyme catalase significantly increased in high-exposure groups[74]. The activity of ERND is mainly CYP3A-dependent, and forms of CYP3A are among the most abundant and important xenobiotic-metabolizing CYP enzymes, mediating the metabolism of numerous xenobiotics[74]. Therefore, high-dose DP exposure might induce a high metabolism rate of DP, while anti-DP is more prone to metabolizing[4], which might be the reason for these dose-dependent results. During the depuration phase for Sprague-Dawley rats, the f_anti values in the liver and muscle increased compared to the end of the uptake phase, implying a selective elimination of syn-DP during the depuration period.

In ovo exposure and in vivo exposure to DP of chicken eggs and hens, respectively, were conducted by Li et al.[75]. The results of in ovo exposure experiment indicate that approximately 12% and 28% of the absorbed syn- and anti-isomers, respectively, were eliminated during egg hatching, resulting in a relative enrichment of syn-DP in chick tissues. Because there was no excretion pathway during hatching, the loss can only be attributed to biotransformation. This result provides solid evidence that anti-DP is preferably metabolized in the organisms. The in vivo exposure experiment provided a completely reversed result. In
the uptake period, the \( f_{\text{anti}} \) value in eggs laid by the hens was just slightly higher than that in the food. However, the eggs laid during the depuration phase showed a sharp increase in \( f_{\text{anti}} \), and the \( f_{\text{anti}} \) in the tissue of hens was higher than the original \( f_{\text{anti}} \) in the food [Figure 2]. The increase of \( f_{\text{anti}} \) in the depuration period agreed with the results in Sprague-Dawley rats\(^{[23]}\). Selective accumulation of \textit{anti}-DP was observed in adult chickens, but the isomer preferentially metabolized in developing chicken embryos, indicating that selective excretion for \textit{syn}-DP played a more important role compared to selective metabolism for \textit{anti}-DP.

These exposure experimental results verify the earlier hypothesis that \textit{anti}-DP may be more reactive than \textit{syn}-DP in biotransformation\(^{[3]}\). However, selective biotransformation for \textit{anti}-DP and selective excretion for \textit{syn}-DP coexisted in the organisms. Only at high concentration levels, where the selective biotransformation surpasses the selective excretion, can organisms show a relative enrichment of \textit{syn}-DP. In the case of low exposure concentration, selective excretion plays a leading role in the accumulation of DP. If the intensity of the two processes is equal, no obvious selective enrichment of DP stereoisomers will be observed.

If this assumption is correct, there will be a selective excretion of \textit{anti}-DP for fish but \textit{syn}-DP for birds. The underlying cause of this species difference in excretion of DP isomer is unclear based on the present data, which could be explained by the difference in the interaction between DP and macromolecules in organisms.

\textbf{In vitro studies using liver microsomal}

In addition to \textit{in vivo} exposure experiments, some \textit{in vitro} studies using liver microsomes were conducted to investigate the potential DP biotransformation. Peng \textit{et al.} conducted \textit{in vitro} experiments using microsomal fraction of the liver in Chinese sturgeon but failed to detect any dechlorinated metabolites\(^{[67]}\). Chabot-Giguère \textit{et al.} conducted an \textit{in vitro} assay of DP using liver microsomes in ring-billed seagulls collected from a polluted hotspot area of the St. Laurence River in Canada\(^{[74]}\). The results show no obvious degradation products of dechlorination and hydroxylation. During field monitoring, dechlorination products of DP, \textit{anti}-Cl\(_1\)-DP, and \textit{ayn}-Cl\(_1\)-DP were detected in birds\(^{[40]}\) and human hair samples\(^{[31]}\). However, these dechlorination products were also found in the environmental matrix. The dechlorination products may be directly derived from the environment rather than biological metabolism.

\textbf{Plant absorption of DP}

Zhao \textit{et al.} studied \textit{Ulva pertusa} exposed to DP solution for 21 days before transferring them to seawater without DP for 14 days\(^{[77]}\). It was observed that \textit{syn}-DP was rapidly accumulated in \textit{Ulva pertusa} during the uptake phase, and the uptake rate of \textit{syn}-isomer (0.164 ± 0.056 day\(^{-1}\)) was higher than that of \textit{anti}-isomer (0.083 ± 0.071 day\(^{-1}\)). However, the elimination rate of \textit{syn}-DP (0.337 ± 0.057 day\(^{-1}\)) was higher than that of \textit{anti}-DP (0.236 ± 0.095 day\(^{-1}\)), which resulted in lower \textit{syn}-DP in the depuration phase. Zhang \textit{et al.} investigated absorption and translocation of DP for rice planted in an e-waste contaminated site\(^{[79]}\). It was found that the \( f_{\text{anti}} \) values of the root, stem, and leaf in the rice were significantly lower than those of soil, indicating a selective enrichment of \textit{syn}-DP from the soils to the root. Similarly, Cheng \textit{et al.} found the \( f_{\text{anti}} \) values of root (0.54) and stem (0.65) were significantly lower than that of the corresponding soil (0.75) in the rhizobox\(^{[79]}\). Fan \textit{et al.} studied DP isomer fractions in peanuts planted in the e-waste recycling area and observed that the \( f_{\text{anti}} \) value in different growth stage [seed (0.37-0.50) and growth (0.37-0.50) stages] was lower than that of the soil and air, which verified that the \textit{syn}-DP was preferably accumulated in the plant\(^{[40]}\).

These four uptake experiments showed selective enrichment of \textit{syn}-DP in plants, but the mechanism remains unknown. It was speculated that the difference in solubility of the two isomers of DP might be the reason for the phenomenon. The solubility of the two isomers is 207 and 572 ng/L as reported by Oxychem, but no specific information is provided on the isomers’ solubility\(^{[40]}\). \textit{Syn}-DP could be prone to transfer from
the soil into pore water and be absorbed by plant root systems because of its relatively high solubility. The relative content of anti-DP increasing during the depuration stage for Ulva pertusa\textsuperscript{[77]} may be due to the high solubility of syn-DP, which leads to preferential distribution into the pure aqueous solution during the depuration phase. In addition to solubility, biotransformation of DP isomers in plants remains unclear, and further studies are needed.

CONCLUSION AND PROSPECTIVE

Stereoisomers of DP exhibited species-specific enrichment in organisms. Based on available data, selective enrichment of syn-DP was mainly observed in fish, while selective enrichment of anti-DP mainly appeared in birds. It is very likely that this different DP isomer fractionation during bioaccumulation exists between ectotherms and endotherms. Most studies reported that syn-DP enrichment was observed in ectotherms, while anti-DP enrichment was observed in endotherms. However, this hypothesis needs to be confirmed in future research.

A direct comparison of $f_{anti}$ values between organisms and technical DP mixture was conducted to determine the occurrence of DP isomer fractionation. This will cause great uncertainty when considering the diversity of industrial DP products in $f_{anti}$ and the alternation of DP composition after releasing into environment. A comparison of $f_{anti}$ values between exposure source and organisms will provide more reliable results. For a given food chain/web, when the prey and predator selectively enrich different isomers, such as fish and fish-eating birds, the $f_{anti}$ value in the predator will be close to that of the technical DP mixture. In that case, the predator was expected to have no clear stereoisomer selective enrichment. Thus, stereoselective enrichment of DP in organisms may be underestimated.

The hypothesis that anti-DP is more readily eliminated through biotransformation than syn-DP has been evidenced by some studies. Unfortunately, potential metabolites have not been identified yet. Screening the phase II type transformation, such as uridine diphosphoglucuronosyltransferase- or sulfotransferase-mediated conjugation, could be a possibility. Selective excretion of one DP isomer was found to be the main
reason for DP isomer fractionation in bioaccumulation. Both selective excretion of anti-DP and selective biotransformation of anti-DP facilitate syn-DP enrichment in fish. However, selective excretion of syn-DP and selective biotransformation of anti-DP seem to coincide in birds. This could be the reason for inconsistent results observed among different studies.

The influences of concentration and trophic levels on the DP composition were evident but did not provide consistent results in past studies and their mechanism remains unknown. Tissue- or organic-specific uptake and elimination of DP isomer were also observed. The preferential enrichment of DP in the liver was observed in both field and laboratory exposure experiments, and the $f_{anti}$ was found to be tissue-specific. However, the key drivers of this process remain unknown. Future studies should address these knowledge gaps.

DECLARATIONS

Authors’ contribution
Conceptualization, formal analysis, investigation, data curation, writing, original draft preparation, Writing, review and editing: Luo XJ
Investigation, writing and original draft preparation: Guan KL
Investigation, writing, review and editing: Liu HY
All authors have read and agreed to the published version of the manuscript.

Available data and materials
The data presented in this review are available from the corresponding author.

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Conflicts of interest
All authors declared that there are no conflicts of interest.

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REFERENCES
1. Luo X J, Mai B X. Bioaccumulation of emerging persistent organic pollutants. Science Press, Beijing; 2017. p.254-255 (In Chinese).
2. Hoh E, Zhu L, Hites RA. Dechlorane plus, a chlorinated flame retardant, in the Great Lakes. Environ Sci Technol 2006;40:1184-9. DOI PubMed
3. Sverko E, Tomy GT, Reiner EJ, et al. Dechlorane plus and related compounds in the environment: a review. Environ Sci Technol 2011;45:5088-98. DOI PubMed
4. Xian Q, Siddique S, Li T, Feng YL, Takser L, Zhu J. Sources and environmental behavior of dechlorane plus-a review. Environ Int 2011;37:1273-84. DOI PubMed
5. Wu B, Liu S, Guo X, et al. Responses of mouse liver to dechlorane plus exposure by integrative transcriptomic and metabolomic studies. Environ Sci Technol 2012;46:10758-64. DOI PubMed
6. Zhang L. Comprehensive toxicity study of dechlorane plus on the earthworm eisenia fetida. Nanjing, Nanjing University; 2014. p.17-22 (in Chinese).
7. ECHA (European Chemicals Agencies. Dechlorane). Plus and its syn- and anti-isomers. Draft risk profile. Available from: https://echa.europa.eu/documents/10162/28991553/draft_risk_profile_dechlorane-plus_en.pdf/d683e4a-06d5-6768180-160bac98bfdf4 [Last accessed on 28 Jun 2022].
8. Feo ML, Barón E, Eljarrat E, Barceló D. Dechlorane Plus and related compounds in aquatic and terrestrial biota: a review. Anal Bioanal Chem 2012;404:2625-37. DOI PubMed
9. Li YL, Meng YZ, Zhao LF, et al. Progress on dechlorane plus, an emerging organic pollutant in environment. Environ Sci Technol 2013;35:79-83 (in Chinese).
10. Luo XJ, Wu JP, Chen SJ, et al. Species specific bioaccumulation of polybrominated diphenyl ethers, hexabromocyclododecan and dechlorane plus in biota: a review. Sci Total Environ 2016;582:17-22.
11. Wang P, Zhang Q, Zhang H, et al. Sources and environmental behavior of Dechlorane Plus and related compounds - a review. Environ Int 2016;88:206-20. DOI PubMed
12. Zafar MI, Kali S, Ali M, et al. Dechlorane Plus as an emerging environmental pollutant in Asia: a review. Environ Sci Pollut Res Int 2020;27:42369-89. DOI PubMed
13. Qiu X, Marvin CH, Hites RA. Dechlorane plus and other flame retardants in a sediment core from Lake Ontario. Environ Sci Technol 2007;41:6014-9. DOI PubMed
14. Tomy GT, Pleskach K, Ismail N, et al. Isomers of dechlorane plus in Lake Winnipeg and Lake Ontario food webs. Environ Sci Technol 2007;41:2249-54. DOI PubMed
15. Wang DQ, Yang M, Qi H, et al. An Asia-specific source of dechlorane plus: concentration, isomer profiles, and other related compounds. Environ Sci Technol 2010;44:6608-13. DOI PubMed
16. Wu JP, Zhang Y, Luo XJ, et al. Isomer-specific bioaccumulation and trophic transfer of Dechlorane Plus in the freshwater food web from a highly contaminated site, South China. Environ Sci Technol 2010;44:696-11. DOI PubMed
17. Zhu J, Feng YL, Shoeb M. Detection of dechlorane plus in residential indoor dust in the city of Ottawa, Canada. Environ Sci Technol 2007;41:7694-8. DOI PubMed
18. Gauthier LT, Letcher RJ. Isomers of Dechlorane Plus flame retardant in the eggs of herring gulls (Larus argentatus) from the Laurentian Great Lakes of North America: temporal changes and spatial distribution. Chemosphere 2009;75:115-20. DOI PubMed
19. Kang JH, Kim JC, Jin GZ, Park H, Baek SY, Chang YS. Detection of Dechlorane Plus in fish from urban-industrial rivers. Chemosphere 2010;79:850-5. DOI PubMed
20. Möller A, Xie Z, Sturm R, Ebinghaus R. Large-scale distribution of dechlorane plus in air and seawater from the Arctic to Antarctica. Environ Sci Technol 2010;44:8977-82. DOI PubMed
21. Zheng J, Wang J, Luo XJ, et al. Dechlorane Plus in human hair from an e-waste recycling area in South China: geographical pattern and implications for anthropogenic exposure. Environ Sci Technol 2010;44:9298-303. DOI PubMed
22. Shen L, Reiner EJ, Helm PA, et al. Historic trends of dechloranes 602, 603, 604, dechlorane plus and other norbornene derivatives and their bioaccumulation potential in lake ontario. Environ Sci Technol 2011;45:3333-40. DOI PubMed
23. Guo J, Venier M, Salamova A, Hites RA. Bioaccumulation of Dechloranes, organophosphate esters, and other flame retardants in Great Lakes fish. Sci Total Environ 2017;583:1-9. DOI PubMed
24. Santin G, Barón E, Eljarrat E, Barceló D. Emerging and historical halogenated flame retardants in fish samples from Iberian rivers. J Hazard Mater 2013;263 Pt 1:116-21. DOI PubMed
25. Abdel Malak I, Cariou R, Vénisseau A, et al. Occurrence of Dechlorane Plus and related compounds in catfish (Silurus spp.) from rivers in France. Chemosphere 2018;207:413-20. DOI PubMed
26. Sühring R, Möller A, Freese M, et al. Brominated flame retardants and dechloranes in eels from German Rivers. Chemosphere 2013;90:118-24. DOI PubMed
27. He MJ, Luo XJ, Wu JP, Chen SJ, Wei SQ, Mai BX. Isomers of Dechlorane Plus in an aquatic environment in a highly industrialized area in Southern China: spatial and vertical distribution, phase partition, and bioaccumulation. Sci Total Environ 2014;481:1-6. DOI PubMed
28. Sun R, Luo X, Tang B, et al. Persistent halogenated compounds in fish from rivers in the Pearl River Delta, South China: geographical pattern and implications for anthropogenic effects on the environment. Environ Res 2016;146:371-8. DOI PubMed
29. Zhang Z, Tong X, Xing Y, et al. Polybrominated diphenyl ethers, decabromodiphenyl ethane and dechlorane plus in aquatic products from the Yellow River Delta, China. Mar Pollut Bull 2020;161:111733. DOI PubMed
30. Wang L, Jia H, Liu X, et al. Dechloranes in a river in northeastern China: spatial trends in multi-matrices and bioaccumulation in fish (Enchelyopus elongatus). Ecotoxicol Environ Saf 2012;84:262-7. DOI PubMed
31. Tao W, Zhou Z, Shen L, Zhao B. Determination of dechlorane flame retardants in soil and fish at Guiyu, an electronic waste recycling site in south China. Environ Pollut 2015;206:361-8. DOI PubMed
32. Kakimoto K, Nagayoshi H, Yoshida J, et al. Detection of Dechlorane Plus and brominated flame retardants in marketed fish in Japan. Chemosphere 2012;89:416-9. DOI PubMed
33. Barón E, Rudolph I, Chang G, Barra R, Eljarrat E, Barceló D. Occurrence and behavior of natural and anthropogenic (emerging and
historical) halogenated compounds in marine biota from the Coast of Concepcion (Chile). Sci Total Environ 2013;461:2-258-64. DOI PubMed
34. Klosterhaus SL, Stapleton HM, La Guardia MJ, Greig DJ. Brominated and chlorinated flame retardants in San Francisco Bay sediments and wildlife. Environ Int 2012;47:56-65. DOI PubMed
35. Barón E, Giménez J, Verborgh P, et al. Bioaccumulation and biomagnification of classical flame retardants, related halogenated natural compounds and alternative flame retardants in three delphinids from Southern European waters. Environ Pollut 2015;203:107-15. DOI PubMed
36. Zhang XL, Luo XJ, Liu HY, Yu LH, Chen SJ, Mai BX. Bioaccumulation of several brominated flame retardants and dechlorane plus in waterbirds from an e-waste recycling region in South China: associated with trophic level and diet sources. Environ Sci Technol 2011;45:400-5. DOI PubMed
37. Zhang H, Wang P, Li Y, et al. Assessment on the occupational exposure of manufacturing workers to Dechlorane Plus through blood and hair analysis. Environ Sci Technol 2013;47:10567-73. DOI PubMed
38. Chen W, Li J, Dong Z, et al. Correlations between dechlorane plus concentrations in paired hair and indoor dust samples and differences between dechlorane plus isomer concentrations in hair from males and females. Chemosphere 2019;231:378-84. DOI PubMed
39. Sühring R, Freese M, Schneider M, et al. Maternal transfer of emerging brominated and chlorinated flame retardants in European eels. Sci Total Environ 2015;530-531:209-18. DOI PubMed
40. Jia H, Sun Y, Liu X, et al. Concentration and bioaccumulation of dechlorane compounds in coastal environment of northern China. Environ Sci Technol 2011;45:2613-8. DOI PubMed
41. Na G, Yao Y, Gao H, et al. Trophic magnification of Dechlorane Plus in the marine food webs of Fildes Peninsula in Antarctica. Mar Pollut Bull 2017;117:456-61. DOI PubMed
42. Wu JP, Chen XY, Si-Kang W, et al. Dechlorane Plus flame retardant in a contaminated frog species: biomagnification and isomer-specific transfer from females to their eggs. Chemosphere 2018;211:218-25. DOI PubMed
43. Venier M, Hites RA. Flame retardants in the serum of pet dogs and in their food. Environ Sci Technol 2011;45:4602-8. DOI PubMed
44. Chen W, Bao I, Bu T, et al. Dechlorane Plus biomagnification and transmission through prairie food webs in Inner Mongolia, China. Environ Toxicol Chem 2021;40:413-21. DOI PubMed
45. Zheng XB, Luo XJ, Zeng YH, Wu JP, Mai BX. Sources, gastrointestinal absorption and stereo-selective and tissue-specific accumulation of Dechlorane Plus iso (DP) in chicken. Chemosphere 2014;114:241-6. DOI PubMed
46. Sun Y, Luo X, Wu J, et al. Species- and tissue-specific accumulation of Dechlorane Plus in three terrestrial passerine bird species from the Pearl River Delta, South China. Chemosphere 2012;89:445-51. DOI PubMed
47. Sun YX, Xu XR, Hao Q, et al. Species-specific accumulation of halogenated flame retardants in eggs of terrestrial birds from an ecological station in the Pearl River Delta, South China. Chemosphere 2014;95:442-7. DOI PubMed
48. Muñoz-Arnanz J, Sáez M, Hiraldo F, et al. Dechlorane plus and possible degradation products in white stork eggs from Spain. Environ Int 2011;37:1164-8. DOI PubMed
49. Kim JT, Son MH, Kang JH, Kim JH, Jung JW, Chang YS. Occurrence of legacy and new persistent organic pollutants in Avian Tissues from King George Island, Antarctica. Environ Sci Technol 2015;49:13628-38. DOI PubMed
50. Kim JT, Choi YJ, Barghi M, et al. Occurrence, distribution, and bioaccumulation of new and legacy persistent organic pollutants in an ecosystem on King George Island, maritime Antarctica. J Hazard Mater 2021;405:124141. DOI PubMed
51. Chen K, Zheng J, Yan X, et al. Dechlorane Plus in paired hair and serum samples from e-waste workers: correlation and differences. Chemosphere 2015;49:123-43. DOI PubMed
52. Yan X, Zheng J, Chen KH, et al. Dechlorane Plus in serum from e-waste recycling workers: influence of gender and potential isomer-specific metabolism. Environ Int 2019;122:49-31. DOI PubMed
53. Rjabova J, Bartkevics V, Zacs D. The occurrence of Dechlorane Plus and related norbornene-based flame retardants in Baltic wild salmon (Salmo salar). Chemosphere 2016;147:210-7. DOI PubMed
54. Muñoz-Arnanz J, Roscales JL, Vicente A, Aguirre JL, Jiménez B. Dechlorane Plus in eggs of two gull species (Larus michahellis and Larus audouinii) from the southwestern Mediterranean Sea. Anal Bioanal Chem 2012;404:2765-73. DOI PubMed
55. Gentes ML, Letcher RJ, Caron-Beaudoin E, Verreault J. Novel flame retardants in urban-feeding ring-billed gulls from the St. Lawrence River, Canada. Environ Sci Technol 2012;46:9735-44. DOI PubMed
56. Zheng XB, Wu JP, Luo XJ, Zeng YH, She YZ, Mai BX. Halogenated flame retardants in home-produced eggs from an electronic waste recycling region in South China: levels, composition profiles, and human dietary exposure assessment. Environ Int 2012;45:122-8. DOI PubMed
57. Mo L, Wu JP, Luo XJ, et al. Dechlorane Plus flame retardant in kingfishers (Alcedo atthis) from an electronic waste recycling site and a reference site, South China: influence of residue levels on the isomeric composition. Environ Pollut 2013;174:57-62. DOI PubMed
58. Peng Y, Wu JP, Tao L, et al. Accumulation of Dechlorane Plus flame retardant in terrestrial passerines from a nature reserve in South China: the influences of biological and chemical variables. Sci Total Environ 2015;514:77-82. DOI PubMed
59. Chen D, Wang Y, Yu L, Luo X, Mai B, Li S. Dechlorane Plus flame retardant in terrestrial raptors from northern China. Environ Pollut 2013;176:80-6. DOI PubMed
60. Yu L, Luo X, Zheng X, et al. Occurrence and biomagnification of organohalogen pollutants in two terrestrial predatory food chains. Chemosphere 2013;93:506-11. DOI PubMed
61. Zeng YH, Luo XJ, Tang B, Mai BX. Habitat- and species-dependent accumulation of organohalogen pollutants in home-produced eggs from an electronic waste recycling site in South China: Levels, profiles, and human dietary exposure. Environ Pollut 2016;216:64-70. DOI PubMed
62. de la Torre A, Alonso MB, Martínez MA, et al. Dechlorane-related compounds in franciscana dolphin (Pontoporia blainvilliei) from southeastern and southern coast of Brazil. Environ Sci Technol 2012;46:12364-72. DOI PubMed
63. Ren G, Yu Z, Ma S, et al. Determination of Dechlorane Plus in serum from electronics dismantling workers in South China. Environ Sci Technol 2009;43:9453-7. DOI PubMed
64. Siddique S, Xian Q, Abdelouahab N, et al. Levels of dechlorane plus and polybrominated diphenylethers in human milk in two Canadian cities. Environ Int 2012;39:50-5. DOI PubMed
65. Brasseur C, Pirard C, Scholl G, et al. Levels of dechloranes and polybrominated diphenyl ethers (PBDEs) in human serum from France. Environ Int 2014;65:33-40. DOI PubMed
66. Zhang Y, Wu JP, Luo XJ, Wang J, Chen SJ, Mai BX. Tissue distribution of Dechlorane Plus and its dechlorinated analogs in contaminated fish: high affinity to the brain for anti-DP. Environ Pollut 2011;159:3647-52. DOI PubMed
67. Peng H, Zhang K, Wan Y, Hu J. Tissue distribution, maternal transfer, and age-related accumulation of dechloranes in Chinese sturgeon. Environ Sci Technol 2012;46:9907-13. DOI PubMed
68. Liu Y, Luo XJ, Huang LQ, Tao L, Zeng YH, Mai BX. Halogenated organic pollutants in aquatic, amphibious, and terrestrial organisms from an e-waste site: habitat-dependent accumulation and maternal transfer in watersnake. Environ Pollut 2018;241:1063-70. DOI PubMed
69. Wang DG, Guo MX, Pei W, Byer ID, Wang Z. Trophic magnification of chlorinated flame retardants and their dechlorinated analogs in a fresh water food web. Chemosphere 2015;118:293-300. DOI PubMed
70. Tomy GT, Thomas CR, Zidane TM, et al. Examination of isomer specific bioaccumulation parameters and potential in vivo hepatic metabolites of syn- and anti-Dechlorane Plus isomers in juvenile rainbow trout (Oncorhynchus mykiss). Environ Sci Technol 2008;42:5562-7. DOI PubMed
71. Zeng YH, Luo XJ, Tang B, Zheng XR, Mai BX. Gastrointestinal absorption, dynamic tissue-specific accumulation, and isomer composition of dechlorane plus and related analogs in common carp by dietary exposure. Ecotoxicol Environ Saf 2014;100:32-8. DOI PubMed
72. Tang B, Luo XJ, Huang CC, et al. Stereoselective bioaccumulation of syn- and anti-Dechlorane plus isomers in different tissues of common carp (Cyprinus carpio). Sci Total Environ 2018;616-617:1339-46. DOI PubMed
73. Li Y, Yu L, Wang J, Wu J, Mai B, Dai J. Accumulation pattern of Dechlorane Plus and associated biological effects on rats after 90 d of exposure. Chemosphere 2013;90:2149-56. DOI PubMed
74. Li Y, Yu L, Zhu Z, et al. Accumulation and effects of 90-day oral exposure to Dechlorane Plus in quail (Coturnix coturnix). Environ Toxicol Chem 2013;32:1649-54. DOI PubMed
75. Li ZR, Luo XJ, Luo YL, Zeng YH, Mai BX. Comparative study of dechlorane plus (DP) in adult chickens and developing embryos: Stereoselective bioaccumulation of DP in chickens. Environ Pollut 2019;247:550-5. DOI PubMed
76. Chabot-Giguère B, Letcher RJ, Verreault J. In vitro biotransformation of decabromodiphenyl ether (BDE-209) and Dechlorane Plus flame retardants: a case study of ring-billed gull breeding in a pollution hotspot in the St. Lawrence River, Canada. Environ Int 2013;55:101-8. DOI PubMed
77. Zhao L, Gong N, Mi D, et al. Kinetics of stereoselective enrichment of Dechlorane Plus in Ulva Pertusa. Chemosphere 2014;111:580-6. DOI PubMed
78. Zhang Y, Luo XJ, Mo L, Wu JP, Mai BX, Peng YH. Bioaccumulation and translocation of polyhalogenated compounds in rice (Oryza sativa L.) planted in paddy soil collected from an electronic waste recycling site, South China. Chemosphere 2015;137:25-32. DOI PubMed
79. Cheng Y, Ding J, Liang X, et al. Fractions Transformation and dissipation mechanism of dechlorane plus in the rhizosphere of the soil-plant system. Environ Sci Technol 2020;54:6610-20. DOI PubMed
80. Fan Y, Chen SJ, Li QQ, Zeng Y, Yan X, Mai BX. Uptake of halogenated organic compounds (HOCs) into peanut and corn during the whole life cycle grown in an agricultural field. Environ Pollut 2020;263:114400. DOI PubMed