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Spatial and Temporal Trends (2004−2016) of Selected Alternative Flame Retardants in Fish of the Laurentian Great Lakes

Yan Wu,[†,‡] Hongli Tan,[†] Chuanlong Zhou,[∥] Bernard S. Crimmins,[∥,§] Thomas M. Holsen,[∥]
James J. Pagano,[⊥] and Da Chen*[,†]

[†] School of Environment and Guangdong Key Laboratory of Environmental Pollution and Health, Jinan University, Guangzhou, 510632, China

[‡] Cooperative Wildlife Research Laboratory and Department of Zoology, Southern Illinois University, Carbondale, Illinois 62901, United States

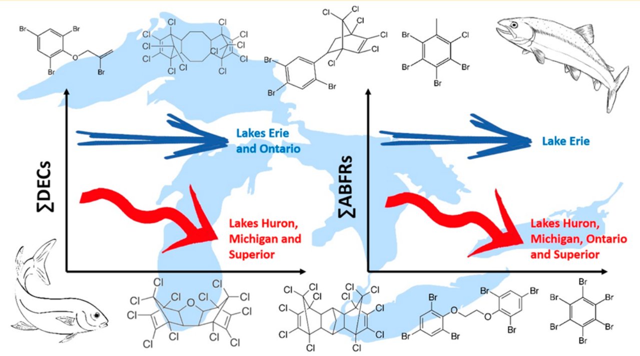
[∥] Department of Civil and Environmental Engineering, Clarkson University, Potsdam, New York 13699, United States

[§] AEACS, LLC, Alliance, Ohio 44601, United States

[⊥] Environmental Research Center, Department of Chemistry, State University of New York at Oswego, Oswego, New York 13126, United States

Supporting Information

ABSTRACT: Following the phase-out of polybrominated diphenyl ether (PBDE) and hexabromocyclododecane (HBCDD) flame retardants (FRs) from North American markets, the use of alternative FRs has increased. In this study the occurrence and spatiotemporal distributions of 18 dechlorane analogues (collectively referred to as DECs) and 20 alternative brominated FRs (referred to as ABFRs, i.e., brominated FRs other than PBDEs and HBCDD) were investigated in top predator fish megacomposites (i.e., a composite of all 50 fish) collected yearly from each of the Great Lakes from 2004 to 2016. Frequently detected substances include dechlorane 602, 603, 604 Component B, *anti-* and *syn*-dechlorane plus, and chlordene plus, as well as several brominated benzene FRs (i.e., hexabromobenzene, pentabromotoluene, and tetrabromo-*o*-chlorotoluene). Concentrations of ΣDECs and ΣABFRs ranged from 0.33−31.9 ng/g lipid weight (lw) (0.01−8.3 ng/g wet weight or ww) and 0.91−54.7 ng/g lw (0.09−7.1 ng/g ww), respectively. Flame retardant contamination exhibited chemical-specific spatial variations across the five lakes. Concentrations of ΣABFRs in Lake Erie fish were generally lower than those from other lakes. By contrast, fish ΣDEC concentrations were highest in Lake Ontario and the composition of dechlorane analogues differed significantly between Lake Ontario and the other lakes, indicating likely point-source influences. Temporal analyses revealed declining trends of ΣDECs and ΣABFRs in most lakes except Lake Erie, with age corrected trend slopes of −13.5% to −8.8% and −20.1% to −7.0% per year, respectively.



INTRODUCTION

Among the myriad of environmental and ecological threats to the Great Lakes of North America, anthropogenic chemical pollution remains a widely recognized issue. A broad range of pollutants has been continuously released to the Great Lakes via point and nonpoint discharges. Of the persistent organic pollutants (POPs) evaluated by federal and state environmental monitoring programs such as the Great Lakes Fish Monitoring and Surveillance Program (GLFMSP), polybrominated diphenyl ether (PBDE) and hexabromocyclododecane (HBCDD) flame retardants have been studied for decades in the Great Lakes basin.[1−6] As a consequence of long-term use in consumer products and because of their substantial persistence and bioaccumulation potential,[7,8] PBDEs and HBCDD have contaminated all compartments of the Great Lakes and have been demonstrated to be global pollutants. The major North American manufacturers have discontinued the production, sale, and importation of PentaBDE and OctaBDE since 2004 and DecaBDE since 2012.[9,10] HBCDD was added to an amendment to Annex A (decision SC-6/13) by the parties to the Stockholm Convention at a May 2013 conference and listed for production and use only in specific circumstances.[11] It is also listed as a binational Chemical of Mutual Concern from the Great Lakes Water Quality Agreement.[12]

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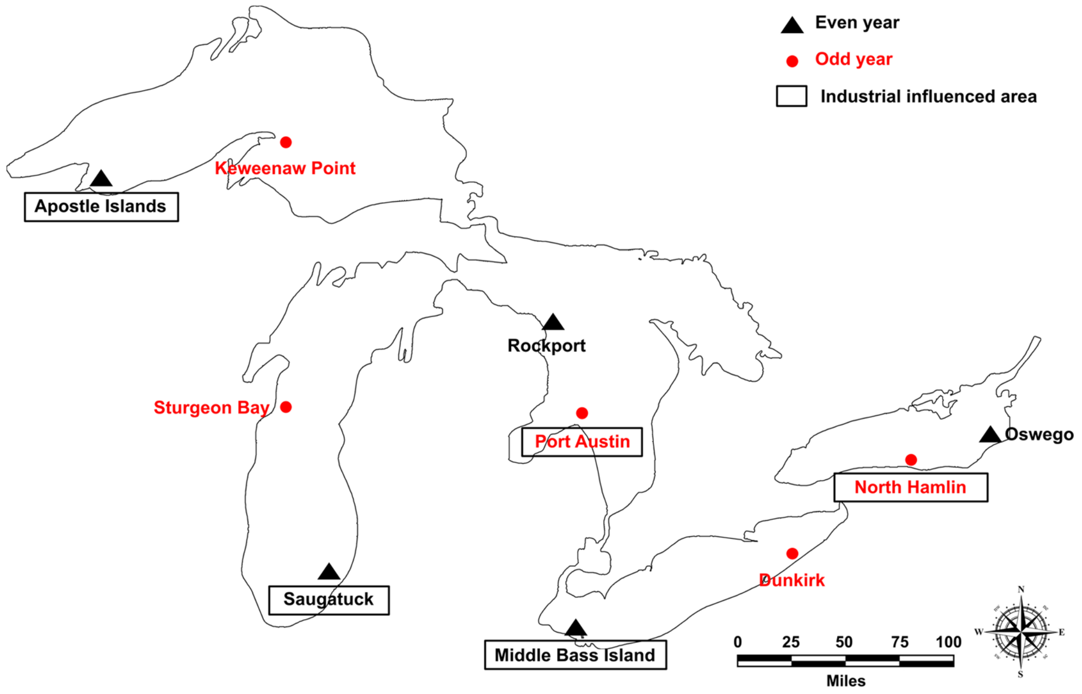

Figure 1. Fish sampling sites in the Great Lakes.

Restrictions on PBDEs and HBCDD use have stimulated the development and application of alternative flame-retardant chemicals to meet flammability standards.[11,13] Covaci et al. estimated that more than 75 brominated flame retardants (BFRs) alternative to PBDEs and HBCDD (collectively referred to as alternative BFRs or ABFRs) have been commercially manufactured.[14] For example, 2-ethylhexyltetrabromobenzoate (EH-TBB) and bis(2-ethylhexyl)-3,4,5,6-tetrabromo-phthalate (BEH-TEBP) were considered as PentaBDE replacements, while 1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE) and decabromodiphenylethane (DBDPE) were suggested as a replacement for OctaBDE and DecaBDE, respectively.[13,14] In addition to brominated substances, chlorinated flame retardants are also broadly used, which include primarily a group of dechlorane-related chemicals (collectively referred to as DECs). The main DEC flame retardants include *syn*- and *anti*-dechlorane plus (DP) isomers, dechlorane (or Dec-) 601, Dec-602, Dec-603, and Dec-604.[15−17] Dechlorane analogues are applied to items including electrical wire coatings, plastic roofing material, and connectors used in computers.[16] Dec-602, Dec-604, or a mixture of them were used as DP alternatives, when the latter substances did not meet the specific voltage leakage and thermal standards in some applications.[15] Some analogues (e.g., chlordene plus or Cplus and Dec-603) also existed as impurities in hexachlorocyclo-pentadiene (HCCP)-based pesticides (e.g., chlordane, chlordene, aldrin, or dieldrin).[18] The DECs and ABFRs have diverse physicochemical properties, environmental behavior, and fate, thus representing different risks to the environment.

Available studies have reported the occurrence of selected flame retardants other than PBDEs and HBCDD in Great Lakes sediment,[4,16,17,19] air,[6,20] fish,[16,21−23] and herring gulls.[24,25] These environmental data highlight the need for a comprehensive evaluation on how the concentrations and composition of alternative flame retardants have changed in Great Lakes biota, following the discontinuation of PBDEs and HBCDD. To address this question, our present work utilized archived GLFMSP fish samples to investigate the contamination status and temporal trends of a wide range of alternative flame retardants. Specific objectives were to (1) investigate the concentrations and composition of DECs and ABFRs in archived Great Lakes lake trout (*Salvelinus namaycush*) and walleye (*Sander vitreus*) composites; and (2) assess chemical-specific spatiotemporal changes in fish concentrations of DECs and ABFRs during the period of 2004−2016. Findings from this retrospective study constitute a baseline for continued surveillance of flame retardants in the Great Lakes basin.

■ MATERIALS AND METHODS

Chemicals and Reagents. This study measured 18 dechlorane-related chemicals, including *syn*- and *anti*-DP, monodechlorinated DP (Cl_{11}-DP), didechlorinated DP (Cl_{10}-DP), Cplus, DP mono adduct (DPMA), dibromochlordene (DBCD), hexachloro(phenyl)norbornene (HCPN), HCCP, Dec-601, Dec-602, Dec-603, Dec-604, Dec-604 Component B (Dec-604CB), Br-Dec604, Br_2-Dec604, Br_2Cl_2-Dec604, and Cl_4-DEC604. We also measured 20 ABFRs, including 2,4,6-tribromophenyl allyl ether (ATE), BEH-TEBP, BTBPE, decabromodiphenylethane (DBDPE), EH-TBB, hexabromobenzene (HBBZ), hexachlorocyclopentadienyl-dibromocyclooctane (HCDBCO), pentabromobenzyl acrylate (PBBA), pentabromobenzyl bromide (PBBB), pentabromobenzene (PBBZ), pentabromoethylbenzene (PBEB), pentabromotoluene (PBT), 1,3,5-tribromobenzene (TBB), tetrabromo-*o*-chlorotoluene (TBCT), *α*- and *β*-1,2,5,6-tetrabromocyclooctane (TBCO), *α*-, *β*-, and *γ*-1,2-dibromo-4-(1,2-dibromoethyl)cyclohexane (TBECH), and 2,3,5,6-tetrabromo-*p*-xylene (TBX). These alternative FRs were chosen for the present work because some of them have been detected in the Great Lakes or other aquatic systems from around the world, while the others have been reported with emerging commercial applications.[14] Surrogate standards included 4′-fluoro-2,3′,4,6-tetrabromodiphenyl ether (F-BDE69), 4′-fluoro-2,3,3′,4,5,6-hexabromodiphenyl ether (F-BDE160), and 2,2′,3,3′,4,5,5′,6,6′-nonabromo-4′-chlorodiphenyl ether (4PC-BDE208), while 3′-fluoro-2,2′,4,4′,5,6′-hexabromodiphenyl ether (F-BDE154) was used as an internal standard. Reference standards were purchased from AccuStandard (New Haven, CT) or Wellington Laboratories (Guelph, ON, Canada), except for Br-Dec604, Br_2-Dec604, Br_2Cl_2-Dec604, and Cl_4-Dec604 (synthesized by Toronto Research Chemicals, Toronto, Canada).

Samples. Lake trout and walleye samples were collected through the GLFMSP (2004−2016). The detailed sampling protocol has been described elsewhere.[26,27] In brief, lake trout were collected at one of the two sites in alternative years from Lakes Huron (LH), Michigan (LM), Ontario (LO) and Superior (LS), as well as in 2011, 2013, and 2015 from Lake Erie (LE) (Figure 1). Walleye were only collected from LE yearly from 2004 to 2010, as well as in 2012, 2014, and 2016 (Table S2). Each year 50 individual fish with similar sizes (600−700 mm in length for lake trout or 400−500 mm for walleye) were collected from each site. These sites have a distance to the closest hatchery site of approximately 2−91 km.[28] Therefore, both hatchery fish (identified by fin clip or wire tags) and wild fish were collected, but there is no systematic trend in the percentage of hatchery fish collected at any site (Figure S1).[29] Hatchery and wild lake trout have similar diet thus their mix did not produce a significant influence on measured POPs concentrations.[30] Fish sex may affect chemical bioaccumulation through metabolic difference and maternal transfer.[31,32] The GLFMSP compositing design generates fish composites primarily based on age. While fish of the same age are grouped into male/female only composites whenever possible, the number of composites consisting of single sex fish is too low to assess whether fish sex is a driver in contaminant concentrations.

The GLFMSP has determined the age of individual fish via multiple techniques, including otoliths, fin clips, coded wire tags (CWT), scales, and maxillae. Since 2012, the traditional otolith age approach was replaced by a maxillae aging method, as the latter approach demonstrates higher accuracy, precision, and smaller bias and allows samples to be composited according to a known age versus an assumed age based on length.[33,34] As measured maxillary (new method) and otolith ages (old method) did not exhibit a significant difference (ANOVA; $p = 0.12$), the determined ages before and after this change were combined for age normalization of measured FR concentrations.[29]

Fifty individual fish were homogenized to produce 10 composite samples per site. Since 2004, GLFMSP also combined the 10 composites to create a mega-composite for each site, resulting in a single pooled composite of 50 lake trout/walleye from each lake every year. This allows for a more efficient investigation of emerging contaminants in Great Lakes ecosystems. Only mega-composite samples were used in the present study.

Sample Analysis. The measurement of FR residues followed the previously reported analytical procedures with minor modifications.[29,35] Briefly, approximately 0.5−1 g of fish mega-composite was spiked with surrogate standards and then subjected to accelerated solvent extraction (Dionex ASE 350, Sunnyvale, CA), employing two 5 min extraction cycles with dichloromethane (DCM) at 100 °C and 1500 psi. After gravimetric determination of lipid content using 10% of the extract, the remaining extract was cleaned through a gel permeation chromatography column (diameter, 1.5 cm; length, 40 cm) packed with 6 g of styrene divinylbenzene beads (3% cross-linkage, 40−80 μm bead size, ≤2000 MW limit; purchased from Bio-Rad, Hercules, CA) in a mixture of hexane and DCM (HEX/DCM, 1:1, v/v).[29] The resulting extract was further purified through a 2 g Isolute silica cartridge (Biotage Inc., Charlotte, NC). The cartridge was preconditioned with 10 mL of HEX, and the concentrated extract was loaded and washed with 3 mL of HEX. Flame retardants were then eluted out with 11 mL of HEX/DCM mixture (6:4, v/v). The final extract was spiked with FBDE-154.

Instrumental analysis of FRs was performed on an Agilent 7890B gas chromatograph (GC) coupled with a 5977A single quadrupole mass analyzer (Agilent Technologies, Palo Alto, CA) operated in electron-capture negative ionization (ECNI) mode. The GC was equipped with a 15 m DB-5HT column (0.25 mm i.d., 0.1 μm, J&W Scientific, Agilent Technologies) and the injector was operated in pulsed-splitless mode (held at 260 °C). The initial oven temperature was set at 50 °C (held for 3 min) and then ramped to 300 °C at 8°C/min (held for 10 min). Quantification of the FRs was conducted based on each congener's characteristic ions under selected ion monitoring (SIM) mode (Table S3).

Quality Assurance and Control. Target analytes and surrogate standards were spiked into a composite of tilapia (*Oreochromis niloticus*) fillets purchased from a local supermarket. Pretest of the composite demonstrated that it contains no quantifiable FRs. The spiked FRs had absolute recoveries (mean ± standard deviation) from sample treatment ranging between 81.1 ± 8.4% and 106.9 ± 7.7% in six replicates and relative recoveries of 90.4 ± 5.2% to 109.0 ± 6.3% after adjustment with the recoveries of surrogate standards (i.e., FBDE-69 for analytes with a retention time earlier than BDE-85, 4PC-BDE208 for DBDPE only, and FBDE-160 for all other analytes). The absolute recoveries of three surrogate standards in the Great Lakes fish megacomposites were 88.4 ± 13.7%, 81.5 ± 11.6%, and 101.6 ± 15.4%, respectively. The procedural blanks processed along with megacomposites contained none of the alternative FRs. Replicate analyses of six randomly chosen megacomposites resulted in a relative standard deviation of analyte concentrations ranging from 7.5% to 13.5%. Owing to the lack of certified values for alternative FRs in the National Institute of Standard Technology Standard Reference Material 1947 LM Fish Tissue, PBDEs were determined as an alternative to assess analytical accuracy. The recoveries of PBDEs from triplicate analyses ranged from 91.1 ± 3.8% to 107.7 ± 5.6% of the reference values after adjustment with surrogate recoveries. The GC−MS instrumental detection limit (IDL) for a FR was estimated by running eight replicate standard solutions of the lowest possible concentration resulting in a signal-to-noise ratio of 5−20, and was then calculated by multiplying the standard deviation of the replicate measurements by the Student's t value appropriate for a 99% confidence level.[36] The IDLs of target FRs ranged from 0.02 to 0.26 ng/mL (injection volume = 2 μL). The method limit of quantification (MLOQ) for each analyte was determined by multiplying a Student's t-value designated for a 99% confidence level with standard deviations in replicate analyses ($n = 8$) of tilapia fillets spiked with a low amount of analyte (2−20 ng). The MLOQs ranged between 0.05 and 2.2 ng/g lipid weight (Table S2).

Age Normalization. GLFMSP fish are collected using size as a metric for age. Fish ages in the designated size ranges have

Table 1. Median Concentrations (ng/g Lipid Weight) of Main Alternative Flame Retardants Detected in Great Lakes Fish Megacomposites. The Concentration Ranges Are Given in Parentheses

	Erie	Huron	Michigan	Ontario	Superior
N	13	13	13	13	13
lipid%	11.1 (8.7−29.6)	16.3 (10.0−30.8)	20.5 (17.5−36.1)	24.0 (20.4−32.9)	14.2 (2.4−18.8)
ATE	<MLOQ[a] (<MLOQ-0.2)	<MLOQ (<MLOQ-0.1)	<MLOQ (<MLOQ-0.2)	<MLOQ (<MLOQ-0.1)	<MLOQ (<MLOQ-0.4)
BEH-TEBP	<MLOQ (<MLOQ-0.6)	<MLOQ (<MLOQ-3.8)	<MLOQ (<MLOQ-3.8)	<MLOQ	<MLOQ
BTPBE	<MLOQ (<MLOQ-0.6)	0.2 (<MLOQ-0.4)	<MLOQ (<MLOQ-0.4)	<MLOQ (<MLOQ-0.4)	<MLOQ (<MLOQ-0.3)
HBBZ	0.4 (<MLOQ-1.9)	1.3 (0.5−2.6)	0.8 (<MLOQ-1.5)	1.2 (0.5−1.7)	0.8 (<MLOQ-3.1)
PBEB	<MLOQ	<MLOQ (<MLOQ-0.2)	<MLOQ (<MLOQ-0.3)	<MLOQ	<MLOQ (<MLOQ-0.3)
PBT	<MLOQ	0.3 (<MLOQ-0.6)	0.4 (<MLOQ-0.8)	0.1 (<MLOQ-0.3)	0.2 (<MLOQ-0.7)
TBCT	3.2 (1.5−5.9)	15.2 (6.2−33.3)	16.1 (9.5−35.1)	6.6 (3.8−11.5)	16.3 (3.9−50.6)
β-TBECH	<MLOQ	<MLOQ	<MLOQ (<MLOQ-1.0)	<MLOQ (<MLOQ-1.2)	<MLOQ
∑ABFRs[b]	3.8 (0.9−7.3)	19.9 (6.9−36.9)	18.8 (10.1−36.9)	7.8 (5.0−14.0)	17.4 (3.9−54.7)
Cplus	0.1 (<MLOQ-0.5)	<MLOQ (<MLOQ-0.4)	<MLOQ (<MLOQ-0.5)	<MLOQ (<MLOQ-0.2)	<MLOQ (<MLOQ-0.5)
DEC-602	0.9 (0.3−1.2)	2.3 (1.7−4.6)	0.9 (0.5−1.7)	8.5 (4.3−16.7)	0.6 (0.3−1.8)
DEC-603	0.2 (<MLOQ-0.4)	0.1 (<MLOQ-0.2)	0.1 (<MLOQ-0.3)	0.1 (0.1−0.2)	<MLOQ (<MLOQ-0.3)
DEC-604CB	<MLOQ (<MLOQ-0.2)	0.1 (<MLOQ-0.3)	<MLOQ (<MLOQ-0.3)	8.0 (4.8−10.0)	<MLOQ (<MLOQ-0.1)
Br-DEC604	<MLOQ	<MLOQ	<MLOQ	1.4 (0.8−4.0)	<MLOQ
Br2-DEC604	<MLOQ	<MLOQ	<MLOQ	0.4 (0.1−1.1)	<MLOQ
HCPN	<MLOQ	<MLOQ	<MLOQ	1.0 (0.6−4.0)	<MLOQ
Cl_{11}-DP	<MLOQ (<MLOQ-0.2)	<MLOQ (<MLOQ-0.1)	<MLOQ (<MLOQ-0.1)	0.1 (<MLOQ-0.2)	<MLOQ
Syn-DP	<MLOQ (<MLOQ-0.2)	<MLOQ (<MLOQ-0.3)	<MLOQ (<MLOQ-0.1)	0.1 (<MLOQ-0.2)	<MLOQ (<MLOQ-0.3)
Anti-DP	0.1 (<MLOQ-0.4)	0.1 (<MLOQ-0.2)	0.1 (<MLOQ-0.2)	0.3 (0.1−0.4)	<MLOQ (<MLOQ-0.3)
∑DECs[c]	1.2 (0.6−2.9)	3.2 (1.9−5.2)	1.3 (0.5−2.4)	21.0 (12.1−31.9)	1.0 (0.3−2.3)

[a] <MLOQ = less than method limit of quantification. [b] Summed concentration of alternative brominated flame retardants. [c] Summed concentration of dechlorane analogues.

significantly increased in LH, LM, and LS over the past decade (Figure S2).[37] Given that the bioaccumulation rate of some chemicals vary with fish size and age (exposure period),[38−40] fish age variations may bias spatiotemporal trend analysis of a contaminant. Thus, an age normalization approach was developed to normalize measured FR concentrations based on the method described in recent Great Lakes studies (see Supporting Information for details).[29,37] In brief, a lake- and year-specific normalization factor was determined for a POPs chemical based on the relationship between concentration and the measured age (average of the five individual fish) in each of the 10 composites per site.[29,37] However, the age normalization factors for alternative FRs (measured only in megacomposites) cannot be directly determined in the same way since there was only one sample per year. Instead, the age normalization factor of a POPs chemical exhibiting the best concentration correlation with an alternative FR in megacomposites was used to normalize the FR concentration.[29] This is based on an assumption that the chosen POPs chemical has a similar bioaccumulation pattern with an alternative FR in Great Lakes fish. This approach results in a lake- and year-specific age factor for each alternative FR (Table S4).

Data Analysis. Concentrations of FRs were adjusted with surrogate standard recoveries and reported on a lipid weight (lw) basis. Any measurement below MLOQ is treated as <MLOQ. For an analyte with detection frequency greater than 60%, a half MLOQ was assigned to a measurement below the MLOQ for statistical analyses if its geometric standard deviation is greater than three; otherwise, a $\text{LOQ}/\sqrt{2}$ was used.[41] The analysis of variance (ANOVA) with Tukey's posthoc test and the principal component analysis (PCA) were conducted using the OriginPro 9.0 (OriginLab Corporation). Non-normally distributed data were logarithmically transformed prior to statistical analyses. For an analyte with a detection frequency greater than 60% in megacomposites from the same site, trend analyses were conducted via the nonparametric Kendall's tau test (OriginPro 9.0, OriginLab Corporation) and Kendall−Theil robust line (KTRLine software, United States Geological Survey), respectively.[42] A significance level of $\alpha = 0.05$ was applied. The Kendall−Theil trend percentage was determined via the following equation:

$$\text{Kendall−Theil trend\%} = \text{slope}/\text{average} \qquad (1)$$

where slope represents the generated Kendall−Theil slope and average is the mean concentration value used for the Kendall−Theil slope estimation.

The halving time ($t_{1/2}$) of a FR in Great Lakes fish was determined using

$$t_{1/2} = \ln(0.5)/\ln(1 + a) \qquad (2)$$

where a is the slope of the Kendall−Theil trendline (percentage/year).

■ RESULTS AND DISCUSSION

Age Normalization. Even though the GLFMSP selected fish with similar body sizes, the ages of sampled fish have varied significantly over the past decade in some lakes. Annualized fish age trends (%/yr) were determined to be 7.2% in even years and 7.4% in odd years from LH, 8.3% in even years and 6.9% in odd years from LS, and 4.8% in even years from LM (see Figure S2).[37] The age variations were partially attributed to the changes in lake trophic structure caused by invasive species such as round goby (*Neogobius melanostomus*) and dreissenid mussels (*Dreissena rostriformis bugensis* and *Dreissena bugensis*)[43−45] as well as eutrophication,[43−46] which has differed among the lakes.[34,37] These factors may lead to lake-specific changes of trophic structure. Bioaccumulation rates of contaminants (e.g., polychlorinated

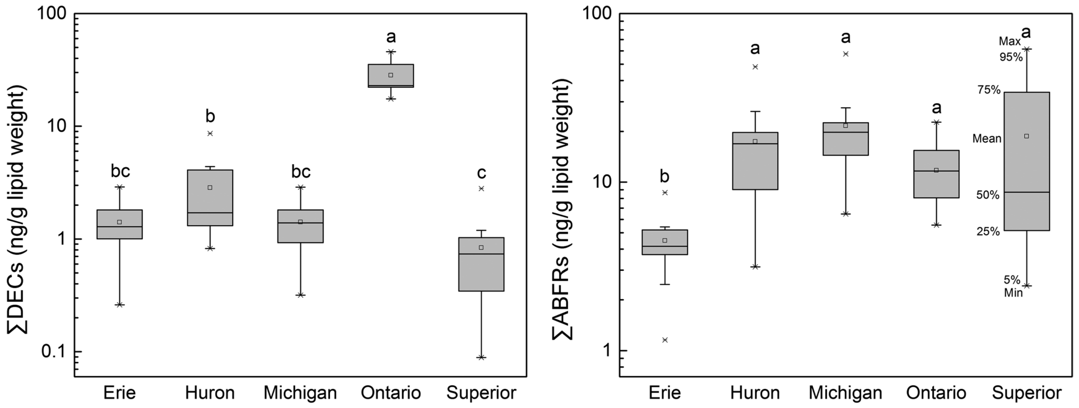

Figure 2. Comparison of the total concentrations of dechloranes (ΣDECs) and alternative brominated flame retardants (ΣABFRs) (age normalized) in fish megacomposites from five lakes over time (2004−2016). Boxes without a common letter indicate a statistically significant difference in concentrations.

biphenyls or PCBs, PBDEs, organochlorine pesticides, and mercury) have been found to vary with fish size and age.[37−40] Age-dependent bioaccumulation of PBDEs has been reported in Great Lakes lake trout.[38] Therefore, age must be taken into consideration for valid spatiotemporal FR analyses in Great Lakes fish. The normalized data were used for the interpretation of interlake differences and temporal trends. Both age normalized and non-normalized data are summarized in the Supporting Information (Tables S2, S5, and S6).

Alternative Flame Retardants Detected. Among the variety of alternative FRs screened, dechlorane analogues were frequently observed in fish megacomposites, demonstrating broad exposure. Dec-602, Dec-603, Dec-604CB, Cplus, and *syn*- and *anti*-DP had a detection rate of 53.9−100%, whereas other analogues (i.e., Cl_{10}-DP, Cl_{11}-DP, Dec-601, Br-Dec604, Br_2-Dec604, and HCPN) were only detected in 7.7−20.0% of the samples. The median ΣDEC concentration (including all detectable dechlorane analogues) ranged from 1.0 ng/g lw (0.16 ng/g ww) in LS fish to 21.0 ng/g lw (5.1 ng/g ww) in LO fish (Table 1). These values are generally 1 to 2 orders of magnitude lower than PBDE concentrations detected in 2016 Great Lakes fish composites which ranged from 9.7 to 47.2 ng/g ww.[47]

Several dechlorane analogues, primarily Dec-602, Dec-603, Cplus, and DP isomers, have been reported in fishes from the St. Lawrence River (Canada) and Illinois (U.S.), Franciscana dolphin (*Pontoporia blainvillei*) from Brazilian coasts, peregrine falcon (*Falco peregrine*) eggs from Canada and Spain, and bobcats (*Lynx rufus*) from midwestern U.S., indicating their bioavailability in both aquatic and terrestrial systems.[35,48−51] Given that dechloranes are subject to no known regulations, their broad exposure and bioaccumulation potency merit particular concerns and further environmental investigation.

The median concentration of ΣABFRs (total concentrations including all detectable alternative BFRs) ranged from 3.8 ng/g lw (0.5 ng/g ww) in LE fish to 19.9 ng/g lw (3.1 ng/g ww) in LH fish (Table 1), also 1 to 2 orders of magnitude lower than PBDE concentrations determined in the same fish. Among the variety of ABFRs determined in fish megacomposites, TBCT, HBBZ, and PBT had relatively higher detection frequencies (i.e., 100%, 89.2%, and 61.5%, respectively) and greater levels (i.e., <MLOQ −50.6; nd −3.1; an nd −0.8 ng/g lw; respectively) than any other analyte. These three chemicals belong to a family of brominated benzene flame retardants. The remaining species in this class were detected infrequently: (compounds, detection frequency) ATE (21.5%), PBBA (0%), PBEB (12.3%), PBBZ (0%), TBB (9.2%), and TBX (3.1%). Less is known about this group of flame retardants compared to PBDEs and HBCDD, and their exact production and usage remains unclear.[5] Some of the brominated benzenes, such as ATE, HBBZ, PBBZ, PBEB, and TBX, have been observed in Great Lakes air or sediment with high detection frequencies (usually >40%).[4−6,52] Measurable levels of ATE, HBBZ, PBEB, PBT, and TBX were also reported in herring gull (*Larus argentatus*) eggs and prey and predatory fish from the Great Lakes basin.[23,25,38] Beyond the Great Lakes, ATE, HBBZ, PBEB, and TBCT have been detected at high rates (i.e., 33−100%) in air based on the Global Atmospheric Passive Sampling (GAPS) Network,[53] demonstrating their global distributions.

Among the other ABFRs (Tables 1 and S2), BTBPE, BEH-TEBP, and TBECH were detected at a rate of 10.8−41.5% and generally present at concentrations less than 2 ng/g lw in Great Lakes fish megacomposites, whereas none of the other ABFRs (i.e., DBDPE, EH-TBB, HCDBCO, and TBCO) was detectable or quantifiable. BTBPE was also detected in Great Lakes herring gull eggs from multiple colonies, but the levels were generally less than 1.4 ng/g lw.[25] BTBPE has been reported in sediment, air, and additional abiotic and biological compartments,[54−57] indicating broad applications of BTBPE-containing products and subsequent releases to the environment. However, fish and wildlife studies generally demonstrated very limited bioavailability of this OctaBDE replacement. *In vivo* exposure to juvenile rainbow trout (*Oncorhynchus mykiss*) suggests that BTBPE has a high uptake potential from aquatic food webs, followed by rapid degradation or depuration.[58] Similarly, BEH-TEBP, EH-TBB, and TBECH had broad distributions in Great Lakes air, water, or sediment,[4,6,59] but exhibited very low concentrations and/or low detection frequencies in fish and birds.[23,25,38,60] Exposure studies have demonstrated rapid metabolism of TBECH in fish, bird, and rat.[61−63] Likewise, significant biotransformation was suggested for BEH-TBEP, EH-TBB, and DBDPE by *in vitro* or *in vivo* studies.[64−66]

Overall, despite broad occurrence of several DEC and ABFR substances in Great Lakes fish, their contamination was still much lower than that of PBDEs in the Great Lakes. Data from other ecosystems also revealed a dominance of PBDEs over alternative FRs in most aquatic and terrestrial organisms.[50,51,55,56,67,68] This pattern may be determined by chemical-dependent production volume and bioaccumulation potency.

Spatial Distribution of FR Concentrations and Compositions. Flame retardant contamination exhibited spatial variations across the five lakes and the variations were chemical-specific (Figure 2). As statistical analyses did not reveal significant concentration differences between odd year and even year collection sites (Student's *t* test, $p > 0.05$ for ΣDECs or ΣABFRs), the two sites from each lake were combined for spatial analysis.

Statistically significant differences in the concentrations of ΣDECs and ΣABFRs were observed among the lakes (ANOVA, $p < 0.001$ in both cases). Post hoc tests indicated that the contamination of ΣABFRs or its main constituent TBCT in LE fish was significantly lower than that in any of the other four lakes, whereas the other four lakes did not differ significantly in fish ΣABFR concentrations (Figure 2). This pattern also resembles the spatial distribution pattern of PBDEs reported in Great Lakes fish.[3] It should be noted that FR concentrations did not differ significantly between lake trout and walleye from LE. Therefore, the pattern may simply reflect lake-specific contamination input and food web structure. Hydrological factors (e.g., water retention time) may also influence spatial distribution. Lake Erie has a relatively shorter water residence times than the other lakes.[3,69] Hydraulic retention time affects the turnover and bioaccumulation rates of chemicals in aquatic systems, thus contributing to spatial variation in contaminant distribution.[3]

Dechloranes exhibited a different spatial distribution pattern compared with other FRs (Figure 2). Concentrations of ΣDECs in LO fish were 1 to 2 orders of magnitude greater than those from other four lakes (Table 1). This was likely due to the proximity of LO sites to a Dechlorane Plus manufacturing facility located in Niagara Falls, New York.[15] When plotting the mean concentration of ΣDECs in Great Lakes fish megacomposites from each fish collection site against the distance of a collection site to the manufacturing facility's location, a significant, negative correlation was observed, indicating a point source influence (Figure 3). The large log K_{ow} (octanol–water partition coefficient) and log K_{oa} (octanol–air partition coefficient) values of dechlorane analogues suggest they have a great affinity for sediment and possess lower long-range transport potential than compounds containing fewer halogens (e.g., lower brominated PBDEs). Indeed, greater concentrations of dechlorane analogues were observed in sediment from LO than those from other lakes.[16,17,22]

Composition of dechlorane analogues in fish also differed between LO and other lakes (Figures 4 and 5). Dec-602 appeared to be the dominant analogue in fish from LE, LM, LH, and LS, constituting an average of 61.1–87.0% of ΣDECs by lakes. In contrast, Dec-602 and Dec-604CB reached similar concentrations in LO fish, constituting an average of 41.6% and 37.7% of ΣDECs, respectively. Dec-604 was only detectable in LO fish, but its levels were generally below MLOQ. These data may indicate limited bioavailability or greater vulnerability of Dec-604 to environmental or biological transformation compared with other major dechlorane analogues. The analyses of LO fish and sediment data suggest that Dec-604 has a much lower biota-sediment accumulation factor (BSAF) than other dechlorane analogues (0.0045 vs 0.013–7.9), with the exception for *syn*-DP (0.0008).[17]

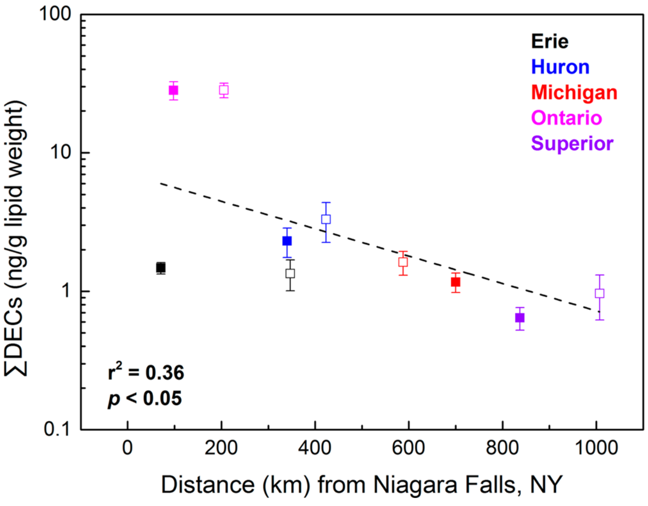

Figure 3. Average (over 2004–2016) of total dechloranes concentrations (ΣDECs) in fish megacomposites from each collection site as a function of its distance from a manufacturing plant in Niagara Falls, NY. Solid and empty squares represent samples from odd and even years, respectively. Error bars represent standard deviations of the concentrations during the period of 2004–2016. The dash line represents the linear regression line of best fit.

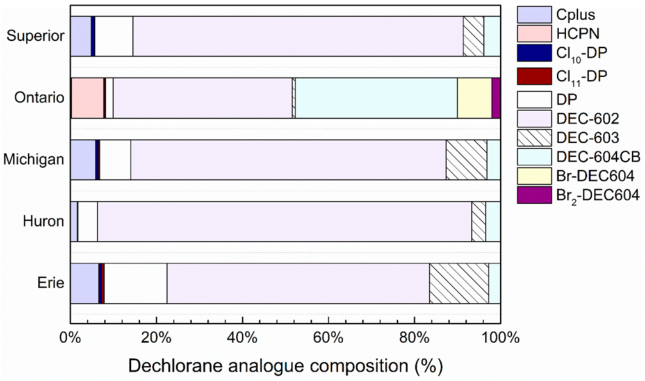

Figure 4. Composition of dechlorane analogues in Great Lakes fish. The data represent the mean composition of each analogue in fish megacomposites from each lake during the study period (2004–2016).

The PCA analysis results in two distinctly different clusters, one including HCPN, Dec-604CB, Br-Dec604, and Br_2-Dec604 and exclusively corresponding to LO fish (Figure 5). Indeed, these analogues were only detectable in LO fish among the five lakes. Br-Dec604 and Br_2-Dec604 have not been reported in any ecosystem beyond the Great Lakes basin. Dec-604CB (Br_3-Dec604), Br_2-Dec604, Br-Dec604, and HCPN are all structurally similar to Dec-604 and were suspected to be impurities in commercial Dec-604 products (or other dechlorane mixtures) or environmental degradation products of Dec-604.[17] It is also possible that Dec-604CB was developed as a component of a flame retardant mixture, because a Br_3-Dec604 product has been listed on the Non-Domestic Substances List by Environment Canada.[22] The existence of these Dec-604 analogues in LO fish points to a significant point-source influence.

Temporal Trends. In response to the decreased use of PBDEs and HBCDD it is likely that alternative FR use has increased. However, it is interesting to see that concentrations

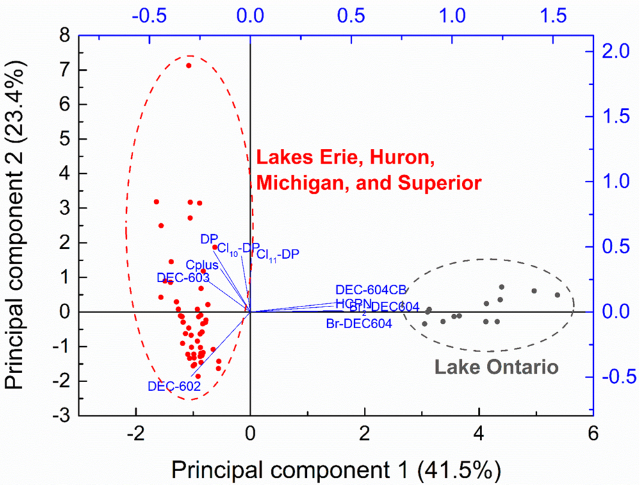

Figure 5. Biplot from the principal component analysis of dechlorane analogue compositions in fish megacomposites from five lakes. The plot indicates a clear difference in dechlorane compositions in fish between Lake Ontario and other lakes.

of ΣABFRs declined significantly in all lakes except LE (no significant change), with age corrected trend slopes ranging from −20.1% to −7.0% per year (Figure 6). The halving time of ΣABFRs ranged between 3.1 and 9.6 years in the lakes where significant trends were determined (Table S6). This was likely driven by the trends of TBCT concentrations which declined in fish from all lakes except LE. Other main brominated benzene FRs (i.e., HBBZ and PBT) and BTBPE also revealed significant declines in most lakes. For example, concentrations of HBBZ declined significantly in fish from LH, LM, and LO (age corrected trend slopes: −11.9% to −7.8%), while PBT declined in LH, LM, and LS (trend slopes: −20.2% to −14.4%). BTBPE also exhibited significant declines in LH and LS fish; however, the trend analysis was affected by limited detection frequencies at other lakes. Recent Great Lakes air studies (2005−2014) revealed declining atmospheric PBEB concentrations (vapor plus particle phases) at various sites in Chicago, Cleveland, and Sturgeon Point with a halving time of 5−7 years, although its concentrations were increasing with a doubling time of 11 years at the most remote site, Eagle Harbor.[6] Atmospheric concentrations of HBBZ and BTBPE also decreased at various Great Lakes monitoring sites with a halving time, if applicable, of 4−13 years and approximately 5 years, respectively.[6] These environmental data suggest decreasing environmental release and exposure of brominated benzene FRs and BTBPE in the Great Lakes basin. Other ABFRs generally had no measurable trends due to very low detection rates (Table S6).

Fish ΣDEC concentrations also significantly declined in LH (Kendall's tau = −0.72, $p < 0.001$), LM (Kendall's tau = −0.49, $p = 0.02$), and LS (Kendall's tau = −0.61, $p = 0.006$), while revealing a marginally significant trend in LO (Kendall's tau = −0.36, $p = 0.09$), with age corrected trend slopes ranging from −13.5% to −8.8% per year (Figure 6). The halving time of ΣDECs ranged between 4.8 and 7.5 years in these lakes (Table S6). No significant trend was observed in LE fish (Kendall's tau = −0.08, $p = 0.71$). The trends generally agreed with LO sediment core studies which indicated that concentrations of *syn*-DP, *anti*-DP, Dec602, Dec-603, and Cplus declined from 1980 to mid-2000s after peaks around 1980, while concentrations of Dec-604 and its less halogenated analogues increased from 1950 to 1990 and then leveled off or declined

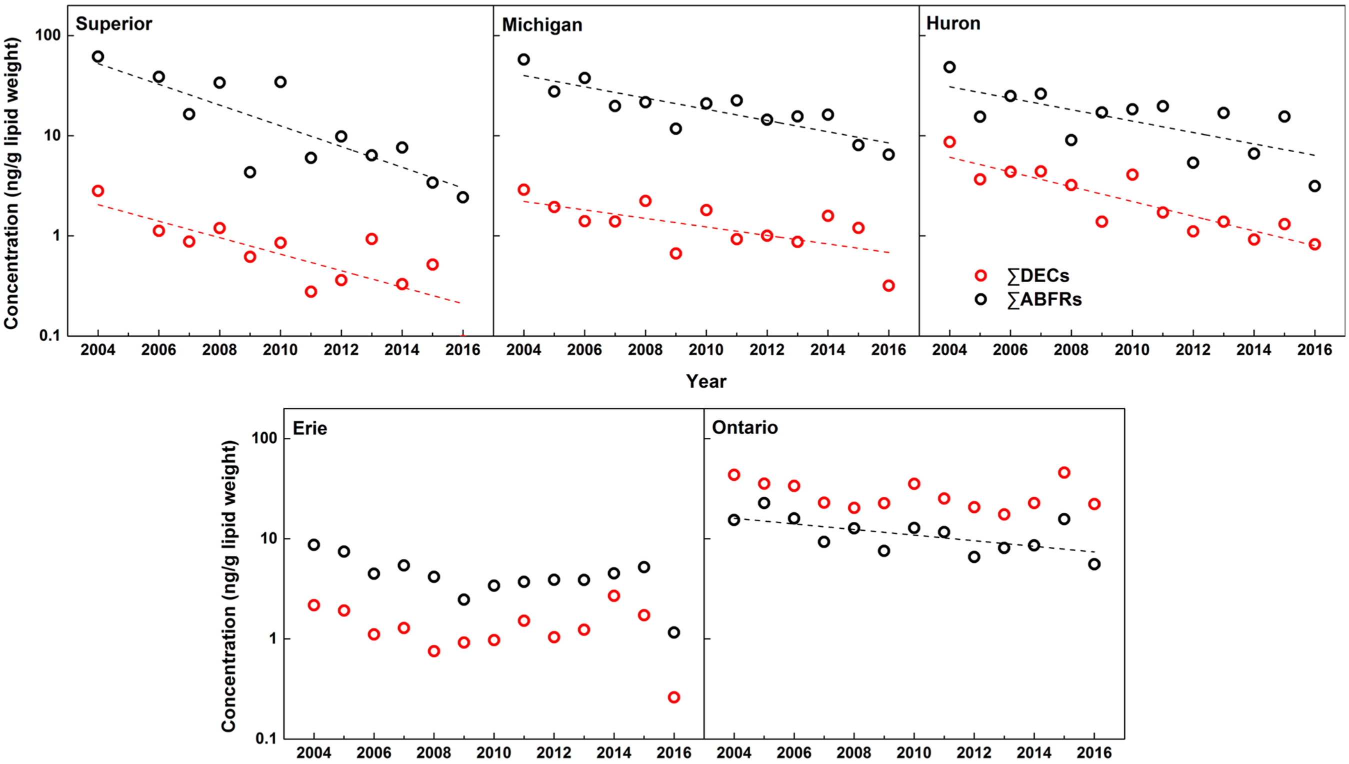

Figure 6. Temporal trends of age normalized concentrations of dechloranes (ΣDECs) and alternative brominated flame retardants (ΣABFRs) in Great Lakes fish megacomposites. Dash lines represent statistically significant correlations between fish concentrations and sampling years. No significant trend exists if no line is given.

since then.[16,22] However, opposite trends were reported in Great Lakes air studies in which ΣDP concentrations (including *anti*- and *syn*-DP isomers) were increasing with doubling times of 4–6 years at Chicago, Eagle Harbor, and Sleeping Bear sites and remained unchanged at Sturgeon Point and Cleveland sites.[6] A recent herring gull study also indicated greater concentrations of ΣDPs in 2012 composites compared with 2008 samples from different colonies.[60] These contradictory results between air/gull and fish/sediment studies require additional investigation.

The findings to date suggest that other possible dechlorane-related chemicals in addition to the main analogues discussed here may exist in the environment. Two monodydroDec602 diastereomers were identified in Great Lakes fish with concentrations of 1.6–50 and 0.7–20 ng/g lw, respectively, while one of them was also detected in Arctic beluga whale (*Delphinapterus leucas*) (0.03–0.21 ng/g lw).[70] Suspected dechlorane-related metabolites were also detected in German peregrine falcon eggs, although their structures and sources were not elucidated.[68] Relevant knowledge remains largely lacking of the sources, environmental behavior, and fate of these additional dechlorane-related chemicals.

We would like to point out the limitations of our study design that may affect temporal analyses. First, our analysis is limited by the use of megacomposites which produce a robust mean value, but provide no information on individual fish concentration variability. Second, potential bias may be included in temporal trend analyses by the age normalization approach used. Since there was only one data point per lake for each year, the age correction used was based on the bioaccumulative pattern of the most similar POPs substance that was measured in individual composites. However, it should be noted that the temporal trends of ΣDECs or ΣABFRs were similar when using data with or without age normalization (Tables S5 and S6), implying that our age normalization approach had a limited effect on the observed trends.

Research Perspectives. Findings from the present study demonstrated limited bioavailability of many alternative FRs (e.g., DBDPE, EH-TBB, and BEH-TEBP) in Great Lakes fish and declining trends of fish ΣDEC and ΣABFR concentrations in most lakes. However, potential risks of these alternative FRs to fish and wildlife should not be discounted. Some of them undergo photochemical or microbial degradation.[71] For example, DBDPE, EH-TBB, and BEH-TEBP were reported to undergo photolysis which produces a series of debrominated products.[72,73] Anaerobic microbial degradation was also reported for TBECH, TBCO, and HBBZ.[74,75] These less halogenated degradation products are usually more bioavailable than parent chemicals; thus elevating the overall risks of organismal exposure to alternative FRs. However, very little information is available on the occurrence of transformation products and derivatives of alternative FRs in the Great Lakes basin. These will be the subjects of future investigations.

Moreover, additional FR chemicals not included in the present study are on the market and possibly subject to an increasing demand. The trend in the flame retardant industry is to produce more highly brominated chemicals to replace DecaBDE or HBCDD, partially based on the assumption that heavily brominated chemicals have exceedingly low bioavailability and are more resistant to environmental or metabolic transformation.[71] These include 1,2-bis(2,3,4,5,6-pentabromophenoxy)ethane (CAS 61262-53-1), ethylene bis-(tetrabromophthalimide) (CAS 32588-76-4), tetrabromobisphenol A-bis(2,3-dibromopropyl ether) (CAS 21850-44-2), and 2,4,6-tris(2,4,6-tribromophennoxy)-1,3,5-triazine (CAS 25713-60-4) which have an estimated log K_{ow} greater than 9.8.[71] Although direct studies on these emerging FRs remain limited, previous knowledge on BDE-209 and DBDPE implies that these highly brominated FRs may be subject to photodegradation through stepwise sequential debromination.[71,73] Therefore, although these new FRs themselves are expected to have limited bioavailability, potential degradation would produce less halogenated but more bioavailable substances. In addition to the highly brominated FRs, nonhalogenated chemicals such as organophosphate triesters (OPEs) have also been increasingly used as flame retardants.[76] Most of these OPEs are different from halogenated FRs in their physicochemical properties, bioavailability, environmental behavior, and fate.

Investigations on some of these alternative FRs remain limited for Great Lakes and other large ecosystems worldwide. Knowledge gaps exist in many aspects of current understanding of alternative FRs, including but not limited to (1) production history and volumes; (2) release sources and pathways; (3) environmental distribution and transformation; (4) organismal exposure and metabolism; and (5) toxicity and modes of action. In addition to addressing these knowledge gaps, we also recommend that not only the alternative FRs themselves, but also their main transformation products, be included in future Great Lakes monitoring programs initiated by both the U.S. and Canada.

■ ASSOCIATED CONTENT

Ⓢ Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.8b05300.

Table S1–S6 and Figures S1–S5, as well as a detailed description of age normalization method (PDF)

■ AUTHOR INFORMATION

Corresponding Author

*Phone: +011-86-20-85220949; fax: +011-86-20-85226615; e-mail: dachen@jnu.edu.cn.

ORCID

Yan Wu: 0000-0001-7876-261X
Bernard S. Crimmins: 0000-0002-6786-7229
Thomas M. Holsen: 0000-0001-9599-6733
James J. Pagano: 0000-0001-8278-6905
Da Chen: 0000-0001-5563-0091

Notes

The authors declare no competing financial interest.

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■ REFERENCES

(1) Li, A.; Rockne, K. J.; Sturchio, N.; Song, W.; Ford, J. C.; Buckley, D. R.; Mills, W. J. Polybrominated diphenyl ethers in the sediments of the Great Lakes. 4. Influencing factors, trends, and implications. *Environ. Sci. Technol.* **2006**, *40*, 7528−7534.

(2) Gauthier, L. T.; Hebert, C. E.; Weseloh, D. V. C.; Letcher, R. J. Dramatic changes in the temporal trends of polybrominated diphenyl ethers (PBDEs) in herring gull eggs from the Laurentian Great Lakes: 1982−2006. *Environ. Sci. Technol.* **2008**, *42*, 1524−1530.

(3) Crimmins, B. S.; Pagano, J. J.; Xia, X.; Hopke, P. K.; Milligan, M. S.; Holsen, T. M. Polybrominated diphenyl ethers (PBDEs): turning the corner in Great Lakes trout 1980−2009. *Environ. Sci. Technol.* **2012**, *46*, 9890−9897.

(4) Yang, R.; Wei, H.; Guo, J.; Li, A. Emerging brominated flame retardants in the sediment of the Great Lakes. *Environ. Sci. Technol.* **2012**, *46*, 3119−3126.

(5) Venier, M.; Salamova, A.; Hites, R. A. Halogenated flame retardants in the Great Lakes environment. *Acc. Chem. Res.* **2015**, *48*, 1853−1861.

(6) Liu, L. Y.; Salamova, A.; Venier, M.; Hites, R. A. Trends in the levels of halogenated flame retardants in the Great Lakes atmosphere over the period 2005−2013. *Environ. Int.* **2016**, *92*−*93*, 442−449.

(7) Alaee, M. An overview of commercially used brominated flame retardants, their applications, their use patterns in different countries/regions and possible modes of release. *Environ. Int.* **2003**, *29*, 683−689.

(8) de Wit, C. A.; Herzke, D.; Vorkamp, K. Brominated flame retardants in the Arctic environment − trends and new candidates. *Sci. Total Environ.* **2010**, *408*, 2885−2918.

(9) Stockholm Convention. All POPs listed in the Stockholm Convention. http://chm.pops.int/TheConvention/ThePOPs/tabid/2509/Default.aspx (accessed June 30, 2018).

(10) United States Environmental Protection Agency. DecaBDE phase-out initiative. https://hero.epa.gov/hero/index.cfm/reference/download/reference_id/1003362 (accessed June 30, 2018).

(11) United States Environmental Protection Agency. Flame retardant alternatives for hexabromocyclododecane (HBCD). https://www.epa.gov/sites/production/files/2014-06/documents/hbcd_report.pdf (accessed June 30, 2018).

(12) Government of the United States and Canada. Binational strategy for hexabromocyclododecane (HBCD) risk assessment. https://binational.net/wp-content/uploads/2018/03/HBCD_Strategy-December-19-2017-FINAL.pdf (accessed June 30, 2018).

(13) United States Environmental Protection Agency. An alternatives assessment for the flame retardant decabromodiphenyl ether (DecaBDE). http://www.epa.gov/dfe/pubs/projects/decaBDE/deca-report-complete.pdf (accessed August 1, 2016).

(14) Covaci, A.; Harrad, S.; Abdallah, M. A.; Ali, N.; Law, R. J.; Herzke, D.; de Wit, C. A. Novel brominated flame retardants: a review of their analysis, environmental fate and behaviour. *Environ. Int.* **2011**, 37, 532−556.

(15) Sverko, E.; Tomy, G. T.; Reiner, E. J.; Li, Y. F.; McCarry, B. E.; Arnot, J. A.; Law, R. J.; Hites, R. A. Dechlorane plus and related compounds in the environment: a review. *Environ. Sci. Technol.* **2011**, *45*, 5088−5098.

(16) Shen, L.; Reiner, E. J.; MacPherson, K. A.; Kolic, T. M.; Sverko, E.; Helm, P. A.; Bhavsar, S. P.; Brindle, I. D.; Marvin, C. H. Identification and screening analysis of halogenated norbornene flame retardants in the Laurentian Great Lakes: Dechloranes 602, 603, and 604. *Environ. Sci. Technol.* **2010**, *44*, 760−766.

(17) Shen, L.; Jobst, K. J.; Reiner, E. J.; Helm, P. A.; McCrindle, R.; Taguchi, V. Y.; Marvin, C. H.; Backus, S.; MacPherson, K. A.; Brindle, I. D. Identification and occurrence of analogues of dechlorane 604 in Lake Ontario sediment and their accumulation in fish. *Environ. Sci. Technol.* **2014**, *48*, 11170−11177.

(18) Shen, L.; Reiner, E. J.; MacPherson, K. A.; Kolic, T. M.; Helm, P. A.; Richman, L. A.; Marvin, C. H.; Burniston, D. A.; Hill, B.; Brindle, I. D.; McCrindle, R.; Chittim, B. G. Dechloranes 602, 603, 604, dechlorane plus, and chlordene plus, a newly detected analogue, in tributary sediments of the Laurentian Great Lakes. *Environ. Sci. Technol.* **2011**, *45*, 693−699.

(19) Yang, R.; Wei, H.; Guo, J.; McLeod, C.; Li, A.; Sturchio, N. C. Historically and currently used Dechloranes in the sediments of the Great Lakes. *Environ. Sci. Technol.* **2011**, *45*, 5156−5163.

(20) Liu, L. Y.; Venier, M.; Salamova, A.; Hites, R. A. A novel flame retardant in the Great Lakes atmosphere: 3,3′,5,5′-Tetrabromobisphenol A bis(2,3-dibromopropyl) ether. *Environ. Sci. Technol. Lett.* **2016**, *3*, 194−199.

(21) Ismail, N.; Gewurtz, S. B.; Pleskach, K.; Whittle, D. M.; Helm, P. A.; Marvin, C. H.; Tomy, G. T. Brominated and chlorinated flame retardants in Lake Ontario, Canada, lake trout (*Salvelinus namaycush*) between 1979 and 2004 and possible influences of food-web changes. *Environ. Toxicol. Chem.* **2009**, *28*, 910−920.

(22) Shen, L.; Reiner, E. J.; Helm, P. A.; Marvin, C. H.; Hill, B.; Zhang, X.; Macpherson, K. A.; Kolic, T. M.; Tomy, G. T.; Brindle, I. D. 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*, 3333−3340.

(23) Guo, J.; Venier, M.; Salamova, A.; Hites, R. A. Bioaccumulation of dechloranes, organophosphate esters, and other flame retardants in Great Lakes fish. *Sci. Total Environ.* **2017**, *583*, 1−9.

(24) Gauthier, L. T.; Letcher, R. J. 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*, 115−120.

(25) Gauthier, L. T.; Potter, D.; Hebert, C. E.; Letcher, R. J. Temporal trends and spatial distribution of non-polybrominated diphenyl ether flame retardants in the eggs of colonial populations of Great Lakes herring gulls. *Environ. Sci. Technol.* **2009**, *43*, 312−317.

(26) Zananski, T. J.; Holsen, T. M.; Hopke, P. K.; Crimmins, B. S. Mercury temporal trends in top predator fish of the Laurentian Great Lakes. *Ecotoxicology* **2011**, *20*, 1568−1576.

(27) Xia, X.; Hopke, P. K.; Crimmins, B. S.; Pagano, J. J.; Milligan, M. S.; Holsen, T. M. Toxaphene trends in the Great Lakes fish. *J. Great Lakes Res.* **2012**, *38*, 31−38.

(28) Great Lakes Fish Stocking Database. http://www.glfc.org/fishstocking/exactsearch.htm (accessed June 30, 2018).

(29) Wu, Y.; Tan, H.; Zhou, C.; Crimmins, B. S.; Holsen, T. M.; Chen, D. Bioaccumulation and spatiotemporal trends of polyhalogenated carbazoles in Great Lakes fish from 2004 to 2016. *Environ. Sci. Technol.* **2018**, *52*, 4536−4545.

(30) United States Environmental Protection Agency. Great Lakes Fish Monitoring and Surveillance Program: Quality Assurance Project Plan for Sample Collection Activities. https://www.epa.gov/sites/production/files/2016-02/documents/glfmsp_qapp_version_2_111312_merged_508_0.pdf (accessed June 30, 2018).

(31) Swackhamer, D. L.; Hites, R. A. Occurrence and bioaccumulation of organochlorine compounds in fishes from Siskiwit Lake, Isle Royale, Lake Superior. *Environ. Sci. Technol.* **1988**, *22*, 543−548.

(32) Gewurtz, S. B.; Bhavsar, S. P.; Fletcher, R. Influence of fish size and sex on mercury/PCB concentration: importance for fish consumption advisories. *Environ. Int.* **2011**, 37, 425−434.

(33) Wellenkamp, W.; He, J. X.; Vercnocke, D. Using maxillae to estimate ages of lake trout. *N. Am. J. Fish. Manag.* **2015**, *35*, 296−301.

(34) Murphy, E. W.; Smith, M. L.; He, J. X.; Wellenkamp, W.; Barr, E.; Downey, P. C.; Miller, K. M.; Meyer, K. A. Revised fish aging techniques improve fish contaminant trend analyses in the face of changing Great Lakes food webs. *J. Great Lakes Res.* **2018**, *44*, 725−734.

(35) Widelka, M.; Lydy, M. J.; Wu, Y.; Chen, D. Statewide surveillance of halogenated flame retardants in fish in Illinois, USA. *Environ. Pollut.* **2016**, *214*, 627−634.

DOI: 10.1021/acs.est.8b05300
Environ. Sci. Technol. 2019, 53, 1786−1796

(36) Vanderford, B. J.; Pearson, R. A.; Rexing, D. J.; Snyder, S. A. Analysis of endocrine disruptors, pharmaceuticals, and personal care products in water using liquid chromatography/tandem mass spectrometry. *Anal. Chem.* **2003**, *75*, 6265−6274.

(37) Zhou, C.; Cohen, M. D.; Crimmins, B. A.; Zhou, H.; Johnson, T. A.; Hopke, P. K.; Holsen, T. M. Mercury temporal trends in top predator fish of the Laurentian Great Lakes from 2004 to 2015: Are concentrations still decreasing? *Environ. Sci. Technol.* **2017**, *51*, 7386−7394.

(38) Su, G.; Letcher, R. J.; McGoldrick, D. J.; Backus, S. M. Halogenated flame retardants in predator and prey fish from the Laurentian Great Lakes: Age-dependent accumulation and trophic transfer. *Environ. Sci. Technol.* **2017**, *51*, 8432−8441.

(39) McIntyre, J. K.; Beauchamp, D. A. Age and trophic position dominate bioaccumulation of mercury and organochlorines in the food web of Lake Washington. *Sci. Total Environ.* **2007**, *372*, 571−584.

(40) Volta, P.; Tremolada, P.; Neri, M. C.; Giussani, G.; Galassi, S. Age-dependent bioaccumulation of organochlorine compounds in fish and their selective biotransformation in top predators from Lake Maggiore (Italy). *Water, Air, Soil Pollut.* **2009**, *197*, 193−209.

(41) Liu, X.; Cao, Z.; Yu, G.; Wu, M.; Li, X.; Zhang, Y.; Wang, B.; Huang, J. Estimation of exposure to organic flame retardants vai hand wipe, surface wipe, and dust: comparability of different assessment strategies. *Environ. Sci. Technol.* **2018**, *52*, 9946−9953.

(42) Granato, G. E. *Kendall-Theil Robust Line (KTRLine version 1.0): A visual basic program for calculation and graphing robust nonparametric estimates of linear regression coefficient between two continueous variables*; USGS: Reston, VA, 2006; p 31.

(43) Lepak, R. F.; Krabbenhoft, D. P.; Ogorek, J. M.; Tate, M. T.; Bootsma, H. A.; Hurley, J. P. Influence of cladophora-quagga mussel assemblages on nearshore methylmercury production in Lake Michigan. *Environ. Sci. Technol.* **2015**, *49*, 7606−7613.

(44) Warner, D. M.; Lesht, B. M. Relative importance of phosphorus, invasive mussels and climate for patterns in chlorophyll a and primary production in Lakes Michigan and Huron. *Freshwater Biol.* **2015**, *60*, 1029−1043.

(45) Crane, D. P.; Einhouse, D. W. Changes in growth and diet of smallmouth bass following invasion of Lake Erie by the round goby. *J. Great Lakes Res.* **2016**, *42*, 405−412.

(46) Cornwell, E. R.; Goyette, J.-O.; Sorichetti, R. J.; Allan, D. J.; Kashian, D. R.; Sibley, P. K.; Taylor, W. D.; Trick, C. G. Biological and chemical contaminants as drivers of change in the Great Lakes − St. Lawrence river basin. *J. Great Lakes Res.* **2015**, *41*, 119−130.

(47) Zhou, C.; Pagano, J. J.; McGoldrick, D. J.; Chen, D.; Crimmins, B. A.; Hopke, P. K.; Milligan, M. S.; Murphy, E. W.; Holsen, T. M. Legacy polybrominated diphenyl ethers (PBDEs) trends in top predator fish of the Laurentian Great Lakes (GL) from 1979 to 2016: Will concentrations continue to decrease? *In preparation.*

(48) de la Torre, A.; Alonso, M. B.; Martinez, M. A.; Sanz, P.; Shen, L.; Reiner, E. J.; Lailson-Brito, J.; Torres, J. P.; Bertozzi, C.; Marigo, J.; Barbosa, L.; Cremer, M.; Secchi, E.; Malm, O.; Eljarrat, E.; Barcelo, D. Dechlorane-related compounds in franciscana dolphin (*Pontoporia blainvillei*) from southeastern and southern coast of Brazil. *Environ. Sci. Technol.* **2012**, *46*, 12364−12372.

(49) Houde, M.; Berryman, D.; de Lafontaine, Y.; Verreault, J. Novel brominated flame retardants and dechloranes in three fish species from the St. Lawrence River, Canada. *Sci. Total Environ.* **2014**, *479−480*, 48−56.

(50) Guerra, P.; Fernie, K.; Jimenez, B.; Pacepavicius, G.; Shen, L.; Reiner, E.; Eljarrat, E.; Barcelo, D.; Alaee, M. Dechlorane plus and related compounds in peregrine falcon (*Falco peregrinus*) eggs from Canada and Spain. *Environ. Sci. Technol.* **2011**, *45*, 1284−1290.

(51) Boyles, E.; Tan, H.; Wu, Y.; Nielsen, C. K.; Shen, L.; Reiner, E. J.; Chen, D. Halogenated flame retardants in bobcats from the midwestern United States. *Environ. Pollut.* **2017**, *221*, 191−198.

(52) Venier, M.; Ma, Y.; Hites, R. A. Bromobenzene flame retardants in the Great Lakes atmosphere. *Environ. Sci. Technol.* **2012**, *46*, 8653−8660.

(53) Lee, S. C.; Sverko, E.; Harner, T.; Pozo, K.; Barresi, E.; Schachtschneider, J.; Zaruk, D.; DeJong, M.; Narayan, J. Retrospective analysis of "new" flame retardants in the global atmosphere under the GAPS Network. *Environ. Pollut.* **2016**, *217*, 62−69.

(54) Moller, A.; Xie, Z.; Cai, M.; Zhong, G.; Huang, P.; Cai, M.; Sturm, R.; He, J.; Ebinghaus, R. Polybrominated diphenyl ethers vs alternate brominated flame retardants and dechloranes from East Asia to the Arctic. *Environ. Sci. Technol.* **2011**, *45*, 6793−6799.

(55) Isobe, T.; Ogawa, S. P.; Ramu, K.; Sudaryanto, A.; Tanabe, S. Geographical distribution of non-PBDE-brominated flame retardants in mussels from Asian coastal waters. *Environ. Sci. Pollut. Res.* **2012**, *19*, 3107−3117.

(56) Klosterhaus, S. L.; Stapleton, H. M.; La Guardia, M. J.; Greig, D. J. Brominated and chlorinated flame retardants in San Francisco Bay sediments and wildlife. *Environ. Int.* **2012**, *47*, 56−65.

(57) Mo, L.; Wu, J. P.; Luo, X. J.; Zou, F. S.; Mai, B. X. Bioaccumulation of polybrominated diphenyl ethers, decabromodiphenyl ethane, and 1,2-bis(2,4,6-tribromophenoxy) ethane flame retardants in kingfishers (*Alcedo atthis*) from an electronic waste-recycling site in South China. *Environ. Toxicol. Chem.* **2012**, *31*, 2153−2158.

(58) Tomy, G. T.; Palace, V. P.; Pleskach, K.; Ismail, N.; Oswald, T.; Danell, R.; Wautier, K.; Evans, B. Dietary exposure of juvenile rainbow trout (*Oncorhynchus mykiss*) to 1,2-bis(2,4,6-tribromophenoxy)ethane: Bioaccumulation parameters, biochemical effects, and metabolism. *Environ. Sci. Technol.* **2007**, *41*, 4913−4918.

(59) Guo, J.; Romanak, K.; Westenbroek, S.; Hites, R. A.; Venier, M. Current-use flame retardants in the water of Lake Michigan tributaries. *Environ. Sci. Technol.* **2017**, *51*, 9960−9969.

(60) Su, G.; Letcher, R. J.; Moore, J. N.; Williams, L. L.; Martin, P. A.; de Solla, S. R.; Bowerman, W. W. Spatial and temporal comparisons of legacy and emerging flame retardants in herring gull eggs from colonies spanning the Laurentian Great Lakes of Canada and United States. *Environ. Res.* **2015**, *142*, 720−730.

(61) Nyholm, J. R.; Norman, A.; Norrgren, L.; Haglund, P.; Andersson, P. L. Uptake and biotransformation of structurally diverse brominated flame retardants in zebrafish (*Danio rerio*) after dietary exposure. *Environ. Toxicol. Chem.* **2009**, *28*, 1035−1042.

(62) Chu, S.; Gauthier, L. T.; Letcher, R. J. Alpha and beta isomers of tetrabromoethylcyclohexane (TBECH) flame retardant: Depletion and metabolite formation *in vitro* using a model rat microsomal assay. *Environ. Sci. Technol.* **2012**, *46*, 10263−10270.

(63) Currier, H. A.; Letcher, R. J.; Williams, T. D.; Elliott, J. E. An assessment of *in ovo* toxicity of the flame retardant 1,2-dibromo-4-(1,2-dibromoethyl) cyclohexane (TBECH) in the zebra finch. *Bull. Environ. Contam. Toxicol.* **2013**, *91*, 455−459.

(64) Bearr, J. S.; Mitchelmore, C. L.; Roberts, S. C.; Stapleton, H. M. Species specific differences in the *in vitro* metabolism of the flame retardant mixture, Firemaster(R) BZ-54. *Aquat. Toxicol.* **2012**, *124−125*, 41−47.

(65) Roberts, S. C.; Macaulay, L. J.; Stapleton, H. M. *In vitro* metabolism of the brominated flame retardants 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (TBB) and bis(2-ethylhexyl) 2,3,4,5-tetrabromophthalate (TBPH) in human and rat tissues. *Chem. Res. Toxicol.* **2012**, *25*, 1435−1441.

(66) de Jourdan, B. P.; Hanson, M. L.; Muir, D. C.; Solomon, K. R. Fathead minnow (*Pimephales promelas Rafinesque*) exposure to three novel brominated flame retardants in outdoor mesocosms: Bioaccumulation and biotransformation. *Environ. Toxicol. Chem.* **2014**, *33*, 1148−1155.

(67) Morris, A. D.; Muir, D.; Solomon, K.; Teixeira, C.; Duric, M.; Wang, X. Bioaccumulation of polybrominated diphenyl ethers (PBDEs) and alternative halogenated flame retardants in a vegetation-caribou-wolf food chain of the Canadian Arctic. *Environ. Sci. Technol.* **2018**, *52*, 3136.

(68) Vetter, W.; Gallistl, C.; Schlienz, A.; Preston, T.; Muller, J.; von der Trenck, K. T. Brominated flame retardants (BFRs) in eggs from birds of prey from southern Germany, 2014. *Environ. Pollut.* **2017**, *231*, 569−577.

(69) United States Environmental Protection Agency. Physical features of the Great Lakes. https://www.epa.gov/greatlakes/physical-features-great-lakes (accessed June 30, 2018).

(70) Shen, L.; Jobst, K. J.; Helm, P. A.; Reiner, E. J.; McCrindle, R.; Tomy, G. T.; Backus, S.; Brindle, I. D.; Marvin, C. H. Identification and determination of the dechlorination products of dechlorane 602 in Great Lakes fish and Arctic beluga whales by gas chromatography-high resolution mass spectrometry. *Anal. Bioanal. Chem.* **2012**, *404*, 2737−2748.

(71) Chen, D.; Hale, R. C.; Letcher, R. J. Photochemical and microbial transformation of emerging flame retardants: Cause for concern? *Environ. Toxicol. Chem.* **2015**, *34*, 687−699.

(72) Davis, E. F.; Stapleton, H. M. Photodegradation pathways of nonabrominated diphenyl ethers, 2-ethylhexyltetrabromobenzoate and di(2-ethylhexyl)tetrabromophthalate: Identifying potential markers of photodegradation. *Environ. Sci. Technol.* **2009**, *43*, 5739−5746.

(73) Wang, J.; Chen, S.; Nie, X.; Tian, M.; Luo, X.; An, T.; Mai, B. Photolytic degradation of decabromodiphenyl ethane (DBDPE). *Chemosphere* **2012**, *89*, 844−849.

(74) Nyholm, J. R.; Lundberg, C.; Andersson, P. L. Biodegradation kinetics of selected brominated flame retardants in aerobic and anaerobic soil. *Environ. Pollut.* **2010**, *158*, 2235−2240.

(75) Wong, F.; Kurt-Karakus, P.; Bidleman, T. F. Fate of brominated flame retardants and organochlorine pesticides in urban soil: Volatility and degradation. *Environ. Sci. Technol.* **2012**, *46*, 2668−2674.

(76) van der Veen, I.; de Boer, J. Phosphorus flame retardants: properties, production, environmental occurrence, toxicity and analysis. *Chemosphere* **2012**, *88*, 1119−1153.