

# **Polybrominated Diphenyl Ethers (PBDEs) in American Eels from the Delaware River, USA**

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## **Introduction**

During the past decade, polybrominated diphenyl ethers (PBDEs) have fallen under appreciable scientific scrutiny. In addition to two other classes of brominated compounds, PBDEs are used as very effective flame retardants in products ranging from computers to textiles (Raloff, 2003). Production and use of these chemicals has been largely concentrated in industrialized countries however, due to their persistence and volatility, PBDEs are globally ubiquitous (e.g., Betts, 2002; Ikonomou et al., 2002). Moreover, they are bioaccumulative, allowing them to concentrate in the fatty tissue of organisms, including humans, to a higher degree than would be found in the organisms' environment (e.g., Boon et al, 2002). Finally, these compounds may elicit developmental impairment in organisms (e.g., Ericksson et al, 2002). These traits, and indeed their chemical structure to an extent, remind environmental chemists of the notorious class of chlorinated compounds called polychlorinated biphenyls (PCBs), listed as suspected carcinogens by the U.S. EPA. But unlike PCBs, whose production and further use was banned in the mid-1970s resulting in decreased concentrations in target organisms worldwide, some formulations of PBDEs are still used. As of December 31, 2004, the production of tetra and octa-BDE has been voluntarily ceased in the U.S though the production and subsequent use of deca-BDE continues.

The Delaware River, like other rivers that bisect urbanized and industrialized lands, receives substantial loadings of contaminants through numerous point and non-point inputs such as urban and agricultural run-off, industrial discharges, and atmospheric deposition. The river also has significant 'in-place' repositories of contamination within its sediments. Although many research studies and monitoring efforts have characterized the magnitude and extent of contaminants such as heavy metals, PCBs and organochlorine pesticides (e.g., Sheldon and Hites, 1978; Kennish and Ruppel, 1996; Ashley et al, 2003a), less information regarding other contaminant classes exist. With the exception of one published study centering on osprey eggs (Toschik et al., 2005) and recent data acquired through New Jersey Department of Environmental Protection

(NJDEP)'s monitoring program (<http://www.state.nj.us/dep/dsr/njmainfish.htm>), there is a paucity of data regarding PBDE concentrations within the living resources of the Delaware River and its tributaries. The objective of this research was to assess the levels of PBDE contamination in American eels from Delaware River locations stretching from the mouth of the Delaware Bay to Delaware Water Gap. American eels are thought to have a limited home range and consequently have been shown to be reliable bio-indicators of PCB pollution within water bodies such as the Delaware and Hudson Rivers (Ashley et al., 2003b). Because PBDEs have similar structures and lipophilic natures compared to PCB congeners, it was hypothesized that eels may also be reliable indicators of localized PBDE contamination within the Delaware River and its tributaries.

## **Experimental Methods**

### **Sample Collection**

In 1998, as part of an Academy of Natural Sciences' project funded by the New Jersey Department of Environmental Protection (NJDEP), American eels were captured (using eel pots or electroshocking) from various locations within the Delaware River and subsequently analyzed for mercury, PCBs and other organochlorine pesticides (NJDEP Report; Ashley et al., 2004). The unused homogenized filets (skin-off) from these eels were subsequently archived in individual sealed jars at -20°C. In late 2005, seventeen of these homogenates were thawed, re-extracted, and subsequently re-analyzed for PCBs and analyzed for the first time for PBDEs. Re-analysis for PCBs allowed comparison of the concentrations obtained in this study to the original concentrations garnered as part of the NJDEP study. American eel samples were chosen to span a range of collection sites within the Delaware River and its tributaries (Figure 1). In addition to these archived samples, three additional eels were collected using eel pots deployed adjacent to Ft. Mifflin, PA (Figure 1). Length and weight measurements of collected eels were made upon collection while lipid content was determined gravimetrically prior to PCB and PBDE analysis in 2005. Identification of sex was not consistently performed and is not reported.

In addition to American eels, four sediment samples, previously extracted for PCB analysis in 2002, were analyzed for PBDEs (Figure 2). These four samples were collected in spring of 2002 as part of an Academy of Natural Sciences' (ANS) project funded by the Delaware River Basin Commission (DRBC) which evaluated the trophic transfer of PCBs within four "zones" within the Delaware River Estuary (DRBC Report). These samples represent composite sediments collected from a minimum of four petit Ponar grabs. Percent moisture and organic carbon content were determined in 2002.

### **Extraction and Instrumental Analyses**

Subsamples (several grams wet weight) of homogenized eel tissue were extracted using previously published methods (Ashley et al, 2003b; Stapleton et al., 2004). Briefly, thawed archived samples were mixed with Na<sub>2</sub>SO<sub>4</sub> to eliminate water and were extracted with dichloromethane (DCM) using a Soxhlet extractor for 18 hours. The extracts were

sub-sampled for gravimetric lipid determination. For this procedure, a known volume of extract (1.0 mL) was transferred to a pre-weighed aluminum pan. The samples were placed into a fume hood and allowed to evaporate for at least 12 hours. The residue remaining (lipid) was weighed and percent lipid was calculated. Lipids were removed from sample extracts by gel permeation chromatography (GPC) using DCM as the mobile phase. The collected fraction containing analytes was concentrated by roto-evaporation and an N<sub>2</sub> stream. Solid-liquid chromatography using florisil (petroleum ether as the eluant) was performed as an additional clean-up step.

Internal standards were added to all the samples and calibration standards prior to PCB and PBDE analysis. For PCBs, 2,3,6-trichlorobiphenyl (PCB#30) and 2,2',3,4,4',5,6,6'-octachlorobiphenyl (PCB#204) were used while for PBDE quantification, brominated diphenyl ether (PBE) 166 was used. One hundred and eight PCB congeners (Table 1), either singly or co-eluting, were analyzed using a Hewlett Packard 5890 gas chromatograph equipped with a <sup>63</sup>Ni electron capture detector and a 5% phenyl methylpolysiloxane column at the Academy of Natural Sciences. The identification and quantification of PCB congeners followed a previously published method (Swackhamer, 1985) in which the identities and concentrations of each congener in a mixed Aroclor standard (25:18:18 mixture of Aroclors 1232, 1248 and 1262) were determined by calibration with individual PCB congener standards.

Twenty-six singly or coeluting PBDE congeners (Table 1) were quantified at the Nicholas School of the Environment at Duke University using gas chromatography (GC) with a mass spectrometer (MS) operated in negative chemical ionization mode (NCI). On-column injection was employed. A 0.25 mm x 15 m fused silica capillary column coated with a 5% phenyl methylpolysiloxane column (DB-5MS; 0.25 μm film thickness) was installed in the GC and connected to the MS. The oven temperature program was held at 80°C for 2 min followed by a temperature ramp of 12 °C/min to 140°C, followed by a second temperature ramp of 5°C/min to a final temperature of 280°C, which was held for an additional 20 min. The auxiliary temperature was maintained at 280°C. For all PBDE congeners, ion masses 79 and 81 (bromide ions) were monitored as quantitative and qualitative ions.

### **Analytical Quality Assurance**

Brominated and chlorinated analyte loss through analytical manipulations was assessed by the addition of surrogate PCB congeners 14, 65 and 166 prior to extraction by Soxhlet apparatus. These surrogates were not industrially prepared and therefore are not present in the environment. Average recoveries of these three congeners were 103%, 109%, and 109%, respectively.

Matrix blanks were generated to monitor possible laboratory contamination and to calculate the detection limits for PCBs and PBDEs. Each matrix blank, consisting of approximately 30 g of clean Na<sub>2</sub>SO<sub>4</sub>, was analyzed using the same procedures as the samples. Chromatograms of most blanks were void of significant peaks suggesting that little contamination through laboratory exposure occurred. The detection limits for PCBs

and PBDEs were calculated as the mass plus three times the standard deviation of the mass. The matrix blank-based detection limits total PCBs (t-PCBs), the sum of all quantified PCB congeners, and total PBDEs (t-PBDEs) were 21 ng/g and 2.0 ng/g.

Additions of known volumes of calibration standards to matrix blanks, or ‘spiked samples’, were used to further evaluate quality assurance of the analytical procedure. Analytes were quantified and resulting masses were compared to the masses initially spiked into the matrix prior to extraction. With an average recovery of 85%, most PCB congener recoveries ranged from 50 % (congeners 189 and 209) to 113 % (congener 41/47). PBDE congener recoveries ranged from 41% (congener 190) to 80% (congener 209). Average PBDE recovery was 61%. It should be noted that reported concentrations in this paper represent values that were not corrected for surrogate loss.

National Institute for Standards and Technology (NIST) standard reference material (SRM 1946, Lake Superior Fish Tissue) was used to evaluate extraction efficiency and analytical accuracy. For PCBs, the majority (70%) of congeners fell within 10% of published values (after correction for surrogate loss). For PBDEs congeners, determined concentrations were 60 to 130% of the NIST certified values (Figure 3).

## **Results and Discussion**

### **Use of Archived Samples for Contaminant Analysis**

This study relied upon American eel samples collected in 1998. Suggested holding times for samples slated for PCB analyses are on the order of months to one year (e.g., EPA method 1668A for PCB analysis). To validate that PCBs, and presumably PBDEs, could be reliably measured in archived samples that were close to eight years old, the PCB concentrations originally determined in 1999 were compared to those from this study’s re-extractions and re-analyses of the same samples. The recent quantification of PCB levels compared very well to the values obtained years prior (Figure 4). For the most part, the recently analyzed samples had total PCB (t-PCB) values that were less than the original concentrations. Surrogate recoveries for these two data sets were similar so the differences could not be attributed to analyte loss through analytical manipulations. Moreover, the differences observed for each sample are on par with the differences we often observe for replicate PCB analysis (relative percent differences from 2 to 30%). From these comparative data, it is reasonable to suggest that the process of long-term archiving at low temperatures did not have an affect on the samples’ PCB levels. Because PCB and PBDE congeners have similar structures and similar physicochemical properties, PBDE loss through long term archiving is hypothesized to be minimal as well.

### **Summary of Contaminant Concentrations**

Values for total PBDEs (t-PBDE) and total PCBs (t-PCBs) are defined as the sum of all individual or coeluting PBDE and PCB congeners, respectively, that were analyzed. These ‘totals’ may be expressed both on a wet weight (ng/g wet) and lipid normalized (ng/g lipid) basis (Table 2). The majority of PCB congeners in American eels (>95%)

were well above detection limits. However, for PBDEs, approximately 60% of the congeners analyzed fell below the limits of quantification or were undetected altogether.

Total PCB concentrations (ng/g wet), as determined in this study from analyses of archived samples, ranged from 70 (Trenton) to 4090 (Deepwater). Two of the seventeen eels exceeded the U.S. Food and Drug Administrations “do not eat” guideline of 2000 ng/g wet weight. Total PBDE concentrations ranged from 1 to 408 ng/g wet weight. Values for t-PBDEs were consistently an order of magnitude less than their counterpart PCB levels. Lipid contents varied widely from 1% to 19% (Table 2).

Based on location caught, the American eels were grouped into five regions (DE Water Gap, Trenton, Ft. Mifflin, Deepwater, and the DE Bay tributaries). Total PCBs and t-PBDEs (on a ng/g wet weight basis) varied widely within and between these groups (top plot of Figures 5 and 6). For PCBs, wet weight concentrations were fairly consistent between the four ‘northern’ regions of the Delaware River system but the Bay tributaries harbored eels with much lower concentrations (Figure 5). The same trend was observed for t-PBDEs however two of the four eels captured at Trenton had about four times higher concentrations than eels from other regions (Figure 6). Differences in wet weight concentrations may often be attributed to differences in individual lipid contents. Upon lipid normalization, some of the spatial trends observed with wet weights levels changed but did not disappear (bottom plots of Figures 5 and 6). For PCBs, Trenton and Bay tributary eels had the lowest concentrations, while the Water Gap, Ft. Mifflin and Deepwater eels were similar (with one notably high eel collected from Ft. Mifflin). High lipid-normalized concentrations arise in those eels that had moderate wet weight concentrations but whose lipid contents were relatively low. For t-PBDEs, a DE Water Gap eel contained over 5500 ng/g lipid (Figure 6). Again, this eel had relatively low lipid content which translated to a large normalized concentration.

To assess the extent by which lipid content determines the body burden of PCBs and PBDEs, contaminant classes were correlated to corresponding lipid contents (Figure 7). For both t-PCBs and t-PBDEs, correlations with lipid content were very low ( $r^2=0.28$  and  $0.11$ , respectively) and not significant (at  $p=0.05$ ) suggesting that lipid variation was not a large determinant of contaminant burden. Linear regressions between each contaminant class and lengths of eels and weights of eels were equally low ( $r^2$  values all less than  $0.1$ ) suggesting these characteristics were not large factors in determining contaminant loads in these particular eels either. Correlating t-PCBs with t-PBDEs did produce a significant positive correlation when two eels from Trenton were not included in the regression analysis (Figure 8). With the exception of these two eels, this trend suggests that the two contaminants are likely delivered to the aquatic system via similar processes. The two anomalous eels had high t-PBDE concentrations yet relatively low t-PCB levels. This may suggest that although the two contaminant classes have similar sources to most of the portions of the Delaware River (at least those studied), t-PBDE sources may be elevated at Trenton. Moreover, looking at the lipid normalized concentrations for t-PBDEs, there seems to be a slight but steady increase in concentration with northward collection sites, a trend that was mirrored in t-PBDE levels in osprey eggs (Toschik et al., 2005). However, given the limited number of eels caught

at each location (n=3 or 4), additional studies including a greater number of eels would have to be conducted before definitively drawing conclusions from the trends observed in this limited study.

### **Congeneric Patterns**

The most abundant brominated diphenyl ether (BDE) congeners detected in American eels, in order of largest contributor to smallest, were BDE47 < BDE100 < BDE154 < BDE119 and BDE49 (Table 3). Coeluting PDE congeners 28+33, BDE66, BDE75, PBE99, PBE153, PBE154, and PBE155 were often detected but at very low concentrations. BDE209, the fully brominated conformation of the diphenyl ether, was not detected in any of the eel samples.

The coeluting PCB congener groups 153+132+105 and 163+138 were the dominant PCB congeners in all eel samples. These were the dominant congeners in the industrial mixtures of Aroclor 1254 and 1260. Their appearance in these samples and in those from other studies (e.g., Ashley et al., 2003b) reconfirms the persistent nature of these congeners.

The paucity of BDE209 in biota from previously published studies initially was thought to be explained by the congener's very strong affinity for sedimentary particles and/or the fact that its conformation sterically hindered its uptake and accumulation in biota. There is substantial evidence that supports that metabolic debromination may be occurring in some fish, transforming BDE 209 to lesser brominated products which are consequently observed in biological tissue (e.g., Stapleton et al., 2004).

Both PCB and PBDE congeneric patterns remained relatively invariant over the spatial scales studied. That is, congener profiles of DE Bay tributary eels looked similar to those in eels captured in the northern reaches of the Delaware River at DE Water Gap, and sites between these two end members.

### **Comparison to Other Data Sets**

Despite the fact that analyses of biota for t-PBDEs levels has increased over the past five years, this relatively recently studied class of compounds has yet to be fully characterized in many fish species. Not surprisingly, this data set represents one of the few which highlights the levels of t-PBDEs in biota within the Delaware River system. American eels from the heavily urbanized and industrialized Passaic River (NJ) have recently been analyzed as part of a NJDEP survey (unpublished data) with t-PBDEs comparable to the ranges observed in this study (Table 4). More data has been garnered on European eels. Concentrations on a lipid normalized basis were comparable to this study (Table 4). For example, a study in Holland conducted from 1983-1989 revealed t-PBDE concentrations ranging from <50 to 1,700 ng/g lipid (de Boer, 1990). The highest reported literature values comes from a report by Goemans and Belpaire (2004) who found up to 32,000 ng/g lipid in eels collected from Belgium. The numbers of eels analyzed in this study were comparable to other American and European studies but the need to more fully

evaluate these contaminant levels is warranted to more fully and statistically evaluate the factors determining their levels. Moreover, it is prudent to recognize that reported biotic concentrations of total PBDEs will vary simply due to the definition of 'total' PBDEs in each study. However, in these studies, the predominant congeners found were consistently similar to those found in this study reflecting the widespread usage of the technical formulations used as flame retardants. However, with the U.S. now following western European bans on the tetra and octa formulations and the fact that fishes may selectively debrominate some congeners, differences in world-wide congeneric patterns may begin to be discerned.

### **Comparison to Sediment**

Like American eels, sedimentary concentrations of t-PCBs were almost an order of magnitude higher than t-PBDE levels Table 5. In the four broad 'zones' within the Delaware River, t-PBDEs concentrations ranged from very low levels around Delaware City, DE to only slightly higher levels around Little Tincum Island (Figure 9). Delaware City sediments contained the lowest values of t-PBDEs evaluated. Unlike PBDEs, PCBs were distinctly elevated in the sediments collected near Little Tincum Island and Delaware City and are likely reflective of increased loadings from urban and industrial portions to the river. Differences in sedimentary levels of hydrophobic organic contaminants that have high affinities for organic matter may often be dampened when dry weight concentrations are normalized to the fraction of organic carbon of the sediment particles. Organic carbon normalized t-PCB and t-PBDE concentrations did not produce any dampening or leveling effect as compared to the dry weight concentrations suggesting proximity to source of these contaminants, and not sediment characteristics like grain size or percent organic carbon, are determining the levels within these benthic repositories.

Because of their primarily benthic diets and their relatively limited home ranges, eels have been shown to be effective biomonitoring tools for regional PCB contamination (Ashley et al., 2003b). PCB sedimentary patterns may be reflected in the accumulated patterns observed in eels living in or near those contaminated sediments (Ashley et al., 2003b). In this study, sedimentary patterns of t-PBDEs were not reflected in the bioaccumulated patterns observed in these eels. As stated, the most abundant t-PBDE congener in eels was BDE 47 (56% relative contribution). However, the most abundant congener was most of the sediment samples analyzed was BDE 209 (49% relative contribution), the fully brominated congener and currently the most widely used congener in flame retardant formulations (Table 3). Researchers have found that fishes may debrominate certain congeners such as BDE 209 (Stapleton et al., 2004). The American eel may be able to metabolize and/or debrominate certain congeners. This would explain the discrepancy between observed eel patterns and those seen in sediments. Another explanation may be that eels may be reflective of their prey items ability to debrominate certain PBDE congeners. Further studies, including a more extensive data set of PBDE levels in eels and complementary sediment analyses should be conducted. Moreover, PBDE analyses of prey items of this species would be helpful in elucidating the factors determining the bioaccumulation of this contaminant class in eels. From this limited data

set though, the use of the American eel as a biomonitoring tool of localized PBDE contamination needs to be assessed further.

## **Conclusions**

The data set presented here represents one of the few to characterize the magnitude and extent of PBDE contamination within the Delaware River. Data garnered from this study will provide a baseline to which future PBDE concentrations within the biotic and abiotic components of the Delaware River may be gauged. Evaluating the sources, transport mechanisms, and ultimate fate of PBDEs within the Delaware River will require more extensive data encompassing greater spatial coverage and resolution in sedimentary concentrations as well as levels in other biotic compartments.

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Table 1. List of PCB and PBDE congeners analyzed in this study. Reported t-PCB and t-PBDE values are comprised of the sum of the individual congeners on this list.

	Polychlorinated Biphenyl Congeners		Polybrominated Diphenyl Ether Congeners
1	100	158	17
3	63	129,178	25
4,10	74	187,182	28,33
7,9	70,76	183	30
6	66,95	128	47
8,5	91	185	49
19	56,60/92,84	174	66
12,13	89	177	71
18	101	202,171,156	75
17	99	157,200	85,155
24	119	172,197	99
16,32	83	180	100
29	97	193	116
26	81,87	191	119
25	85	199	138
31, 28	136	170,190	153
33,21,53	77,110	198	154
51	82, 151	201	156
22	135,144	203,196	181
45	107	189	183
46	123,149	208,195	190
52	118	207	191
49	134	194	203
47,48	146	205	205
44	132,105,153	206	206
37,42	141	209	209
41,64,71	137,130,176		
40	163,138		

Table 2. Summary of t-PCB and t-PBDE concentrations on a wet weight and lipid normalized basis for American eels collected from various regions of the Delaware River and its tributaries (Cohansey and Maurice Rivers).

Location Caught	Lipid Content, %	t-PBDE, ng/g wet	t-PCBs, ng/g wet	t-PBDE, ng/g lipid	t-PCBs, ng/g lipid
DE Water Gap	2.0	25.6	894.6	1312	45764
DE Water Gap	9.5	124.4	2073.2	1303	21717
DE Water Gap	1.2	68.8	376.5	5652	30910
Trenton	16.5	373.7	1170.9	2271	7117
Trenton	12.9	1.3	69.6	10	540
Trenton	13.6	3.1	229.3	23	1692
Trenton	15.1	407.9	746.9	2709	4961
Ft. Mifflin	7.8	67.3	1261.4	862	16140
Ft. Mifflin	1.3	21.9	942.1	1735	74482
Ft. Mifflin	9.3	46.2	1405.9	495	15061
Deepwater	4.8	71.2	1862.6	1498	39193
Deepwater	18.6	157.2	4091.7	845	21994
Deepwater	8.6	77.7	1318.2	905	15351
Cohansey River	12.9	3.6	223.3	28	1733
Cohansey River	6.6	3.3	169.1	51	2579
Cohansey River	2.3	1.2	147.2	51	6374
Maurice River	9.1	11.0	426.7	121	4689

Table 3. Mean contributions of each PBDE congener in American eels and sediment samples.

PBDE Congener	Mean Percent Contribution in American Eels	PBDE Congener	Mean Percent Contribution in Sediment
BDE 47	55.5	BDE 209	48.7
BDE 100	29.6	BDE 99	15.0
BDE 154	3.2	BDE 47	13.8
BDE 119	2.9	BDE 49	5.2
BDE 49	2.3	BDE 100	3.5
BDE 153	1.7	BDE 28,33	2.4
BDE 99	1.6	BDE 71	2.2
BDE 71	0.9	BDE 154	2.1
BDE 85,155	0.8	BDE 206	1.8
BDE 116	0.5	BDE 116	1.4
BDE 66	0.4	BDE 153	1.3
BDE 28,33	0.4	BDE 119	0.9
BDE 75	0.2	BDE 85,155	0.8
BDE 156	0.1	BDE 66	0.7
BDE 30	0.002	BDE 203	0.1
BDE 17	ND	BDE 183	0.1
BDE 25	ND	BDE 138	0.0
BDE 138	ND	BDE 190	0.0
BDE 181	ND	BDE 17	ND
BDE 183	ND	BDE 25	ND
BDE 190	ND	BDE 30	ND
BDE 191	ND	BDE 75	ND
BDE 203	ND	BDE 156	ND
BDE 205	ND	BDE 181	ND
BDE 206	ND	BDE 191	ND
BDE 209	ND	BDE 205	ND

Table 4. Comparison of ranges of t-PBDE concentrations in American and European eels.

<b>Species</b>	<b>Location</b>	<b>Range of t-PBDEs Concs</b>	<b>Units</b>	<b>Number Analyzed</b>	<b>Reference</b>
European Eel	The Netherlands	<50 to 1,700	ng/g lipid	34	de Boer, 1990
European Eel	Belgium	2 to 14	ng/g wet	4	Covaci et al., 2005
European Eel	Germany	3.6 to 21.4	ng/g lipid	5	Lepom et al., 2002
European Eel	Belgium	~50 to 32,000	ng/g lipid	18	Geomans and Belpaire, 2004
American Eel	Passaic River	1,604 to 6,439	ng/g lipid	11	Ashley et al., unpublished data
American Eel	Delaware River	10 to 5,652	ng/g lipid	17	This Study

Table 5. Summary of t-PCB (2002 data) and t-PBDE (this study) concentrations for sediment samples collected from four regions of the Delaware River.

Collection Region	Bristol PA	Bridesburg, PA	Little Tinicum Isl., PA	Delaware City, DE
<b>Zone(DRBC designation)</b>	2	3	4	5
<b>Sediment % Solid</b>	63	74	44	61
<b>Sediment % Water</b>	37	26	56	39
<b>Sediment % Org Carbon</b>	2.315	1.400	1.965	1.864
<b>dry mass extracted</b>	4.196	9.389	4.981	3.130
<b>t-PCBs (ng/g dry weight)</b>	<b>30</b>	<b>12</b>	<b>216</b>	<b>115</b>
<b>t-PCBs (ng/g OC)</b>	1278	861	11009	6151
<b>PCB Surrogate Recovery</b>				
14	Int	69%	Int	Int
65	109%	64%	109%	103%
166	113%	65%	137%	135%
<b>BDE 17</b>	ND	ND	ND	ND
<b>BDE 25</b>	ND	ND	ND	ND
<b>BDE 28,33</b>	0.09	0.01	1.51	ND
<b>BDE 30</b>	ND	ND	ND	ND
<b>BDE 47</b>	2.90	0.57	1.21	0.15
<b>BDE 49</b>	0.24	0.06	0.16	0.04
<b>BDE 66</b>	0.12	0.02	0.03	ND
<b>BDE 71</b>	0.04	0.01	0.02	0.01
<b>BDE 75</b>	ND	ND	ND	ND
<b>BDE 85,155</b>	ND	0.02	0.08	0.03
<b>BDE 99</b>	4.00	ND	ND	0.25
<b>BDE 100</b>	0.71	0.19	0.37	0.04
<b>BDE 116</b>	ND	ND	1.78	ND
<b>BDE 119</b>	ND	ND	ND	0.02
<b>BDE 138</b>	0.03	ND	0.01	ND
<b>BDE 153</b>	0.40	0.05	0.17	0.02
<b>BDE 154</b>	0.29	0.06	0.17	0.02
<b>BDE 156</b>	ND	ND	ND	ND
<b>BDE 181</b>	ND	ND	ND	ND
<b>BDE 183</b>	ND	0.02	0.05	ND
<b>BDE 190</b>	ND	0.01	ND	ND
<b>BDE 191</b>	ND	ND	ND	ND
<b>BDE 203</b>	ND	0.02	0.06	ND
<b>BDE 205</b>	ND	ND	ND	ND
<b>BDE 206</b>	ND	0.41	1.29	ND
<b>BDE 209</b>	7.27	7.00	14.79	0.16
<b>t-PBDEs (ng/g dry weight)</b>	<b>16.09</b>	<b>8.46</b>	<b>21.69</b>	<b>0.73</b>
<b>t-PBDEs (ng/g dry weight)</b>	<b>695.16</b>	<b>604.37</b>	<b>1103.91</b>	<b>38.95</b>



Figure 1. Map (Source: DRBC) showing sites of American eel collection in 1998 (all sites except Ft. Mifflin) and 2005 (Ft. Mifflin).

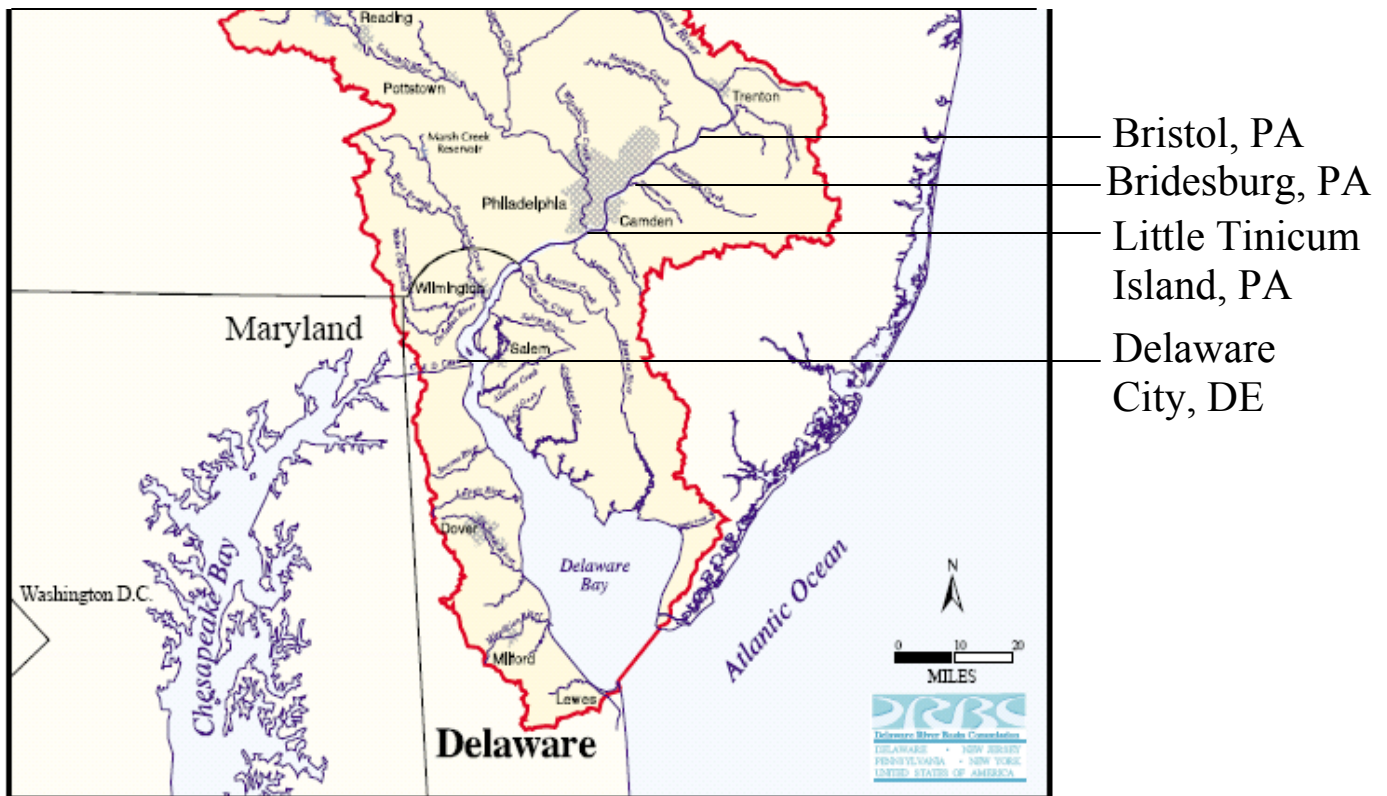


Figure 2. Map (Source: DRBC) showing sites of collection of sediment in spring 2002 (DRBC).

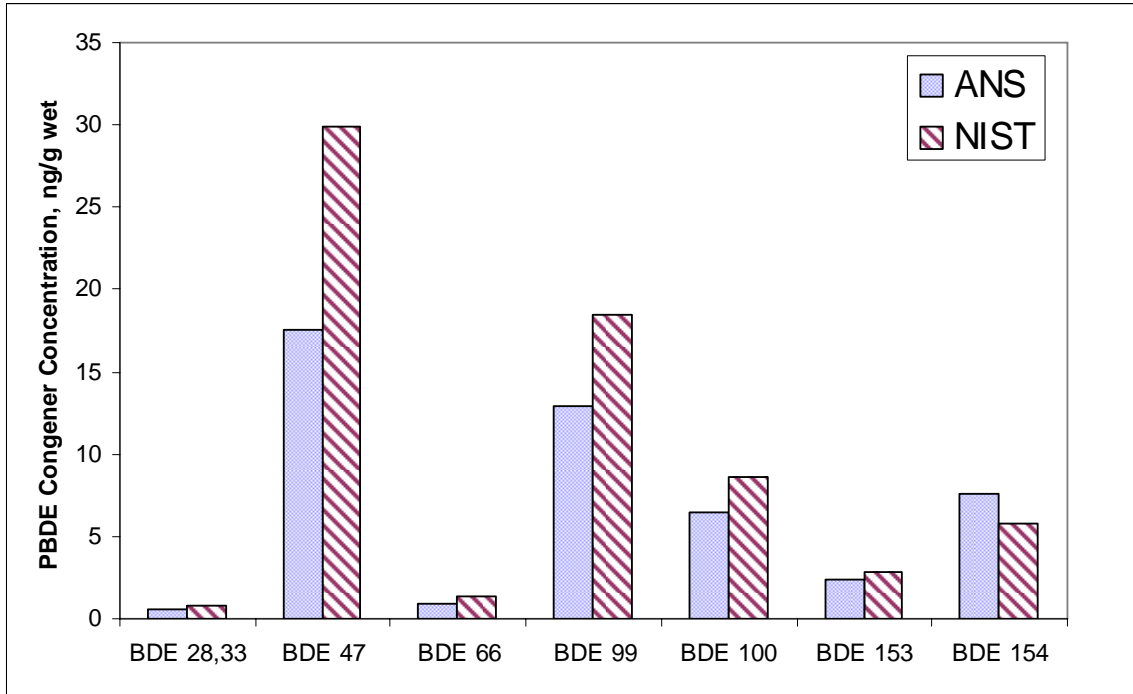


Figure 3. Values obtained for selected PBDE congeners for SRM1946 (Lake Superior fish) from this study compared to published values reported by NIST.

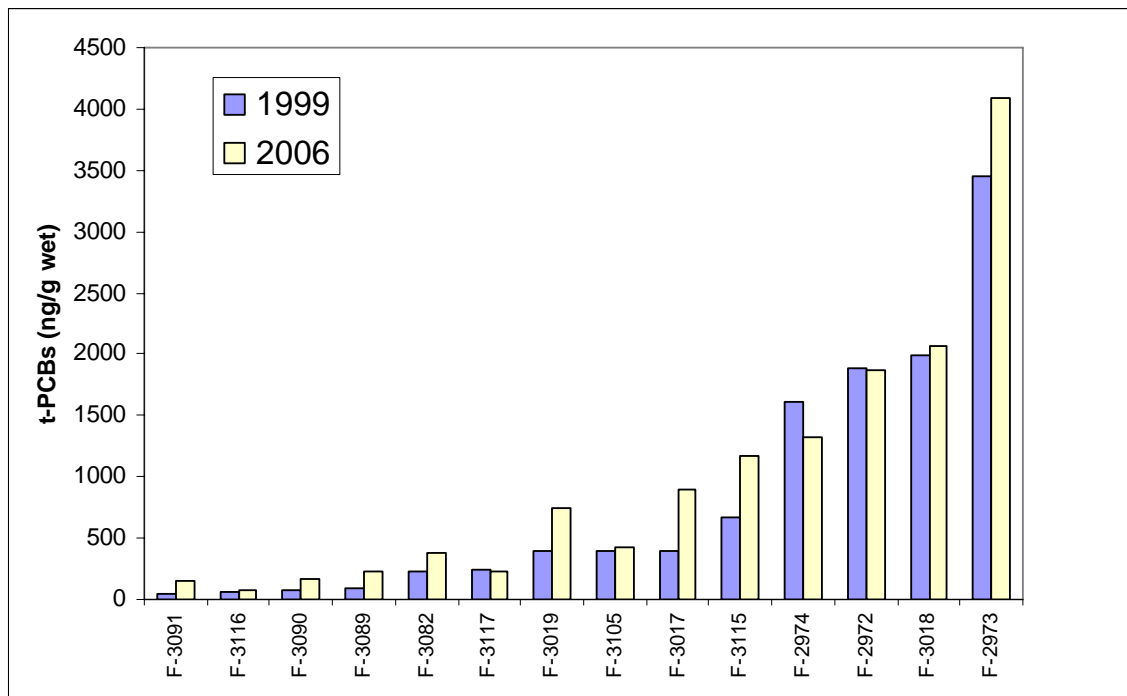


Figure 4. Comparison of t-PCB concentrations obtained for each American eel in 1999 and values from the re-extracted and re-analyzed archived samples in 2006.

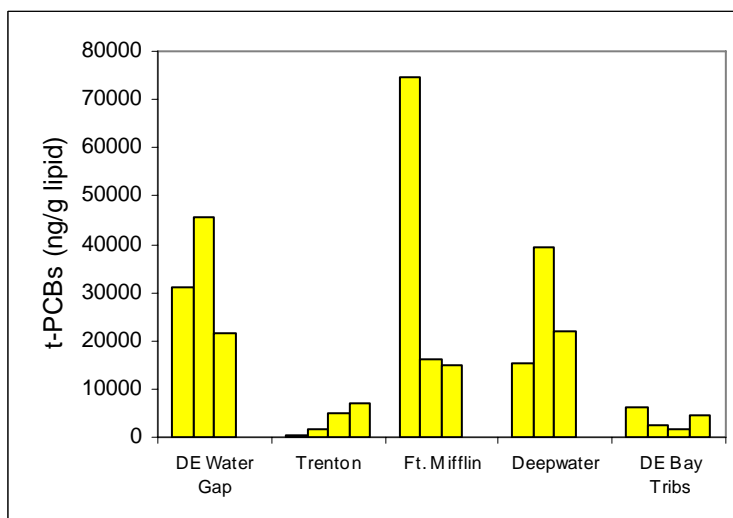
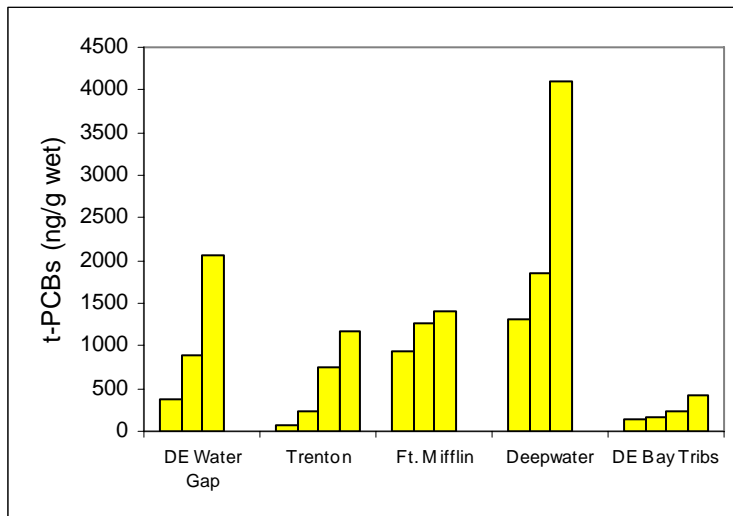


Figure 5. Total PCB concentrations reported on a wet weight and lipid normalized basis for American eels grouped according to collection site.

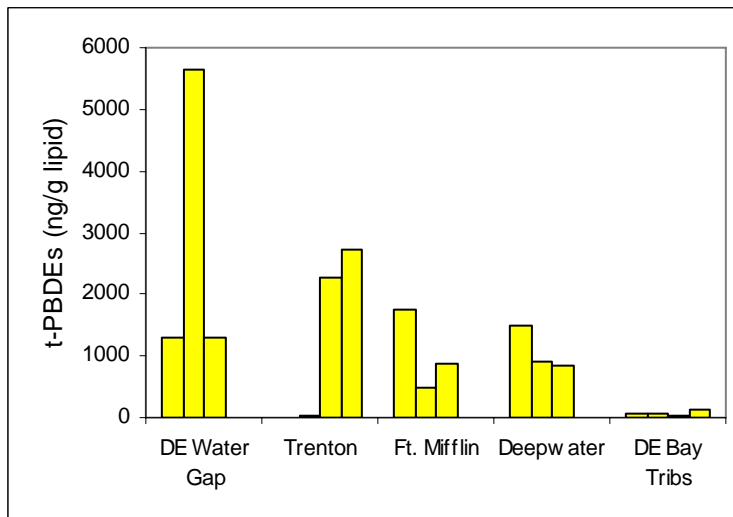
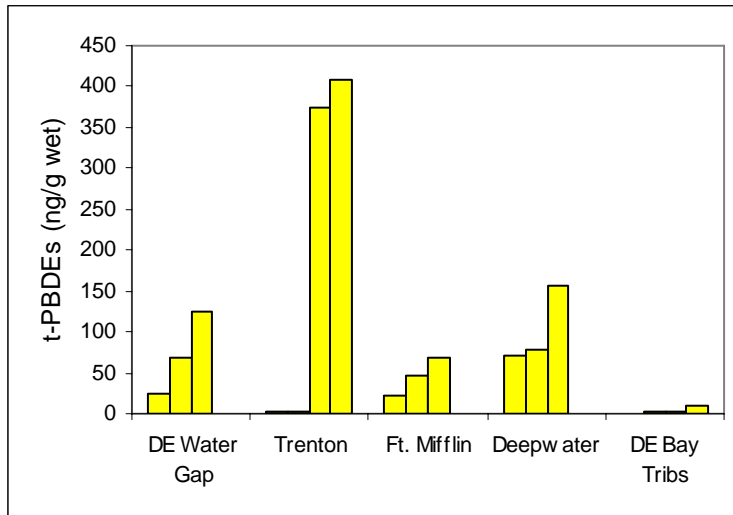


Figure 6. Total PBDE concentrations reported on a wet weight and lipid normalized basis for American eels grouped according to collection site.

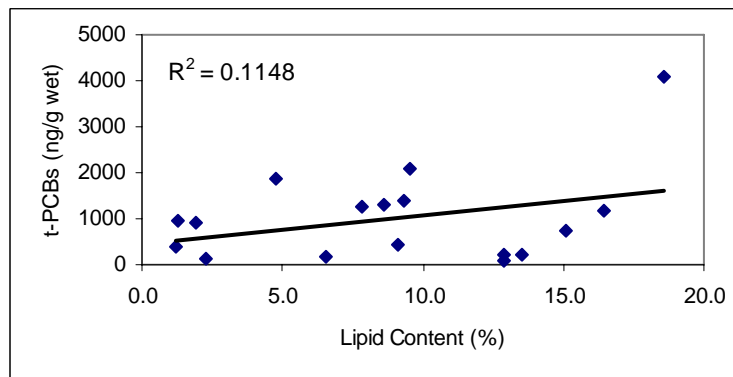
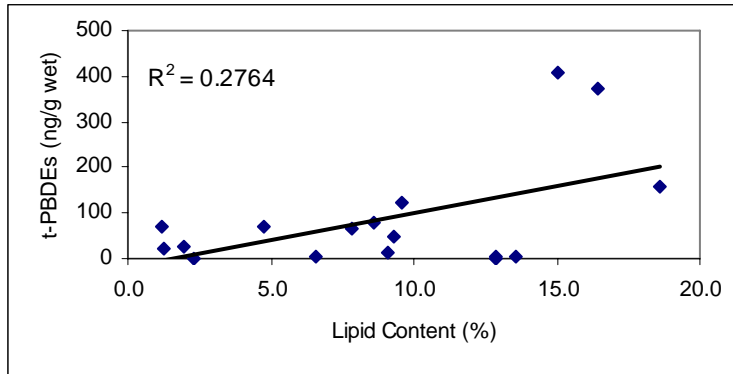


Figure 7. Correlation between t-PBDE and t-PCB concentrations and lipid content of American eels (all data used).

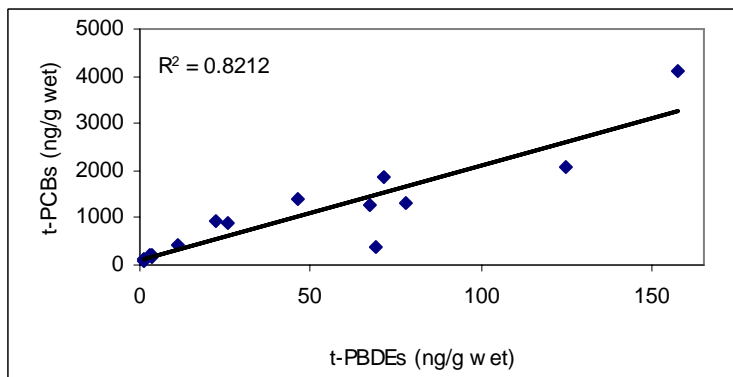
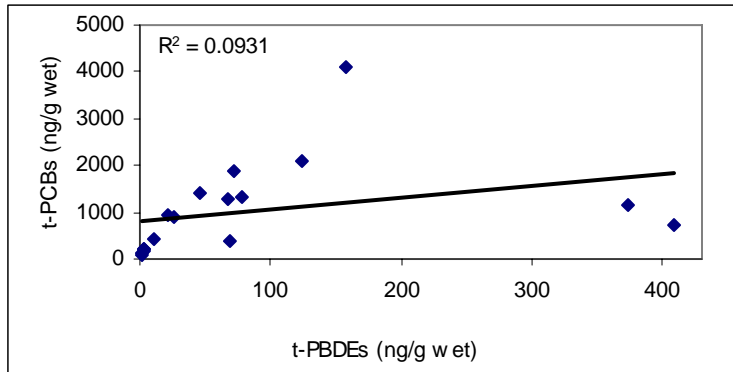


Figure 8. Correlation between t-PBDE and t-PCB concentrations for all data (upper plot) and all data except for two eels caught at Trenton (bottom plot).

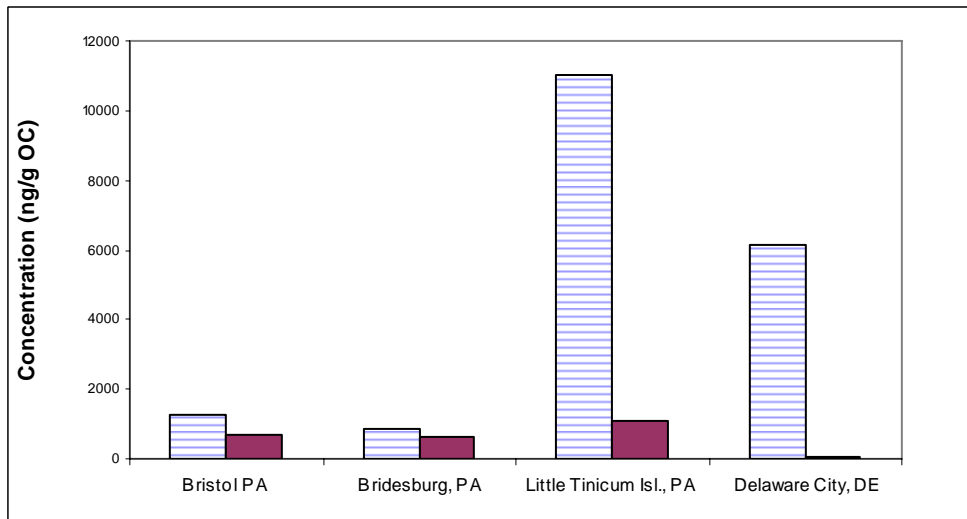
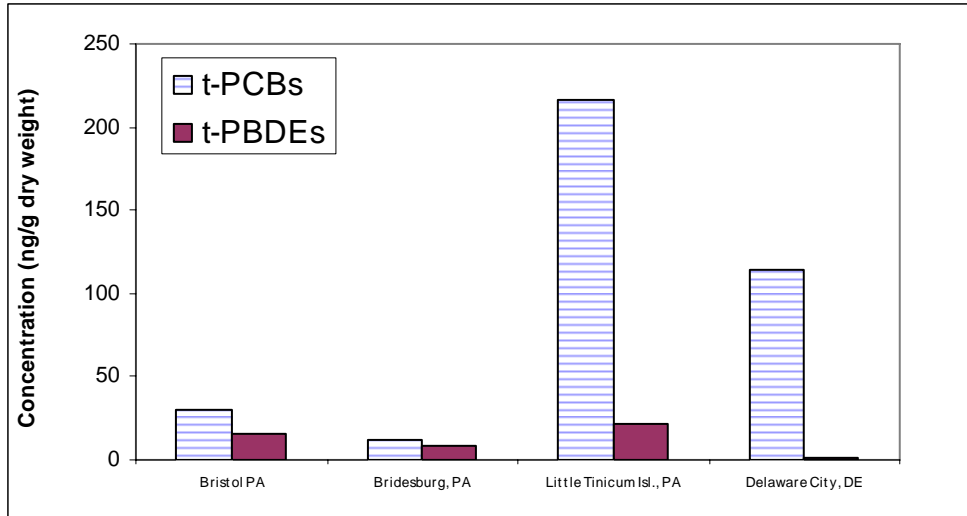


Figure 9. Concentrations of t-PCBs and t-PBDEs (on a dry weight and organic carbon normalized basis) for sediment samples collected in 2002 from four locations within the Delaware River.