Environmental Toxicology and Chemistry, Vol. 31, No. 10, pp. 2191–2200, 2012
© 2012 SETAC
Printed in the USA
DOI: 10.1002/etc.1942

Environmental Chemistry

POLYTOPIC VECTOR ANALYSIS OF SOIL, DUST, AND SERUM SAMPLES TO EVALUATE EXPOSURE SOURCES OF PCDD/Fs

TIMOTHY P. TOWEY,*† NOÉMI BARABÁS,† AVERY DEMOND,‡ ALFRED FRANZBLAU,§ DAVID H. GARABRANT,§
BRENDA W. GILLESPIE,|| JAMES LEPKOWSKI,# and PETER ADRIAENS‡
†LimnoTech, Ann Arbor, Michigan, USA
‡Department of Civil and Environmental Engineering, University of Michigan College of Engineering, Ann Arbor, Michigan, USA
§Department of Environmental Health Sciences, University of Michigan School of Public Health, Ann Arbor, Michigan, USA
||Department of Biostatistics, University of Michigan School of Public Health, Ann Arbor, Michigan, USA
#Institute for Social Research, University of Michigan, Ann Arbor, Michigan, USA

(*Submitted* 30 *November* 2011; *Returned for Revision* 23 *January* 2012; *Accepted* 30 *May* 2012)

Abstract—As part of the University of Michigan Dioxin Exposure Study, soil, household dust, and serum samples were collected from more than 750 households in five populations around the city of Midland and in Jackson and Calhoun Counties, Michigan, USA. Polytopic vector analysis, a type of receptor model, was applied to better understand the potential sources of polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans found in these samples and to quantify the contributions of the sources in each matrix across populations. The results indicated that source signatures found in soil are similar to those found in dust, reflecting various combustion profiles, pentachlorophenol, and graphite electrode sludge. The profiles associated with contamination in the Tittabawassee River, likely related to historical discharges from the Dow Chemical Company facility in Midland, exhibited the largest differences among the regional populations sampled. Differences in serum source contributions among the study populations were consistent with some of the regional differences observed in soil samples. However, the age trends of these differences suggested that they are related to past exposures, rather than ongoing sources. Environ. Toxicol. Chem. 2012;31:2191–2200. © 2012 SETAC

Keywords—Dioxins   Environmental chemistry   Multivariate statistics   Organochlorines

INTRODUCTION

Polytopic vector analysis (PVA) is a multivariate statistical technique that uses the relationships among congeners to extract source profiles and their relative contributions, allowing the identification of sources. This tool has been applied to a variety of persistent organic pollutants, including polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and polychlorinated biphenyls (PCBs), predominantly in soil or river sediment, air, and human serum. For example, Ehrlich et al. [1] applied PVA to surface sediment samples from Newark Bay, New Jersey, USA, and showed that the sediments were impacted by multiple sources, including combustion, municipal sewage, and paper mill effluent. Huntley et al. [2] also analyzed sediments from Newark Bay and determined that buried contaminants from historic sources were not impacting surface sediments. Barabás et al. [3,4] used a modified form of PVA to identify PCDD and PCDF sources and a dechlorination signature in sediments in the Passaic River, New Jersey, USA. Applications to PCB measurements in the environment [5–7] and in human serum [8] have also been reported. In a study of serum samples from 753 Akwesasne Mohawk Indians, DeCaprio et al. [8] identified congener profiles of PCBs that were likely linked to lifetime background exposure and a single profile consistent with a localized exposure, volatilized Aroclor 1248. They also found that some of the profiles extracted by PVA were probably linked to differential elimination of various PCBs rather than particular exposure sources, suggesting the need to account for elimination rates in the treatment of the data or in the interpretation of results. The results of this study suggest the value of applying PVA to unmix complex source patterns, even when these patterns have been altered by environmental or biological processes.

The Dow Chemical Company facility in Midland, Michigan, USA, historically emitted PCDDs and PCDFs (PCDD/Fs) to the environment. These emissions can be classified broadly into two categories: (1) discharges to the Tittabawassee River and (2) aerial emissions and deposition resulting from incineration activities. Previous investigations have demonstrated that soils in the floodplain of the Tittabawassee River contain elevated levels of certain PCDD and PCDF congeners, with PCDFs as the primary contributor to the toxic equivalent (TEQ). The aerial depositional area in and around the city of Midland contains elevated levels of PCDDs, PCDFs, and PCBs, with less-chlorinated PCDDs as the primary contributors to the TEQ [9–12].

The University of Michigan Dioxin Exposure Study (UMDES) was undertaken to better understand the relationship between PCDD/Fs in the environment and in serum. More specifically, the study was designed to evaluate the extent to which contamination from the Dow facility has impacted serum levels of the population in the city of Midland and along the Tittabawassee River. We used PVA to identify sources impacting the soil and dust samples and to compare their profiles to those found in the participants' serum samples. The present study was the first investigation in which PVA was applied to soil, household dust, and serum samples from the same populations. Because exposure to PCDD/Fs via incineration is historical, the concentrations in serum have been impacted by differential elimination. Therefore, the present study also

---

All Supplemental Data may be found in the online version of this article.

* To whom correspondence may be addressed (ttowey@limno.com).

explored two methods for pretreating serum data to account for elimination processes.

METHODS

Sample collection and analysis

As part of the UMDES, soil ($n = 2{,}081$ from 766 properties), household dust ($n = 764$), and serum ($n = 946$) samples were collected from participants in four populations in Midland, Saginaw, and part of Bay Counties and from a comparison population in Jackson and Calhoun Counties—all in the state of Michigan. The four Midland and Saginaw study populations were as follows: (1) Tittabawassee River floodplain, properties that included any land in the Federal Emergency Management Administration–defined 100-year floodplain of the Tittabawassee River downstream of the Dow facility in the city of Midland and upstream of the confluence of the Tittabawassee and Shiawassee Rivers; (2) near floodplain, properties from census blocks that contain a portion of the 100-year floodplain of the Tittabawassee but in which the property itself is not in the floodplain; (3) plume, properties from census blocks in the area of deposition from emissions stacks at the Dow facility in Midland, as defined by environmental modeling based on historical emission data and current soil concentrations [13]; and (4) other Midland/Saginaw (MS), properties in Midland, Saginaw, and part of Bay Counties outside the designated areas for the floodplain, near floodplain, and plume populations and outside the floodplain of the Shiawassee and Saginaw Rivers [14]. Supplemental Data, Figure 1 shows a map of the locations of the four study populations. A fifth population, drawn from Jackson and Calhoun Counties, located approximately 180 km to the southwest of the Midland/Saginaw study area, served as a reference. Samples from each of three matrices were analyzed for the PCDD/F and PCB congeners included in the World Health Organization–designated 29 congeners [15] by Vista Analytical in El Dorado Hills, California, USA, using U.S. Environmental Protection Agency (U.S. EPA) methods 8290 and 1668 [16,17]. Details regarding respondent selection, sample collection, and summary statistical results for each matrix can be found in Garabrant et al. [14,18], Hedgeman et al. [19], and Demond et al. [10,20].

Data treatment

Preliminary multivariate analysis using all 29 congeners indicated that the variability of the PCB congeners was greater than the variability in the PCDD/F congeners and that information related to the PCDD/Fs, particularly in serum samples, was obscured. Therefore, the PCDD/F congeners were analyzed separately from the PCBs. The present study presents only the results of the PCDD/F analysis; the results of the PCB analysis are reported in Towey et al. [21].

Concentrations below the limits of detection (LOD) can affect correlations between congeners and, in turn, the results of PVA. To decrease the impact of values below the LOD, congeners with >50% of the values below the LOD were excluded from the analysis. Samples where >50% of the remaining congeners were below the LOD were also excluded.

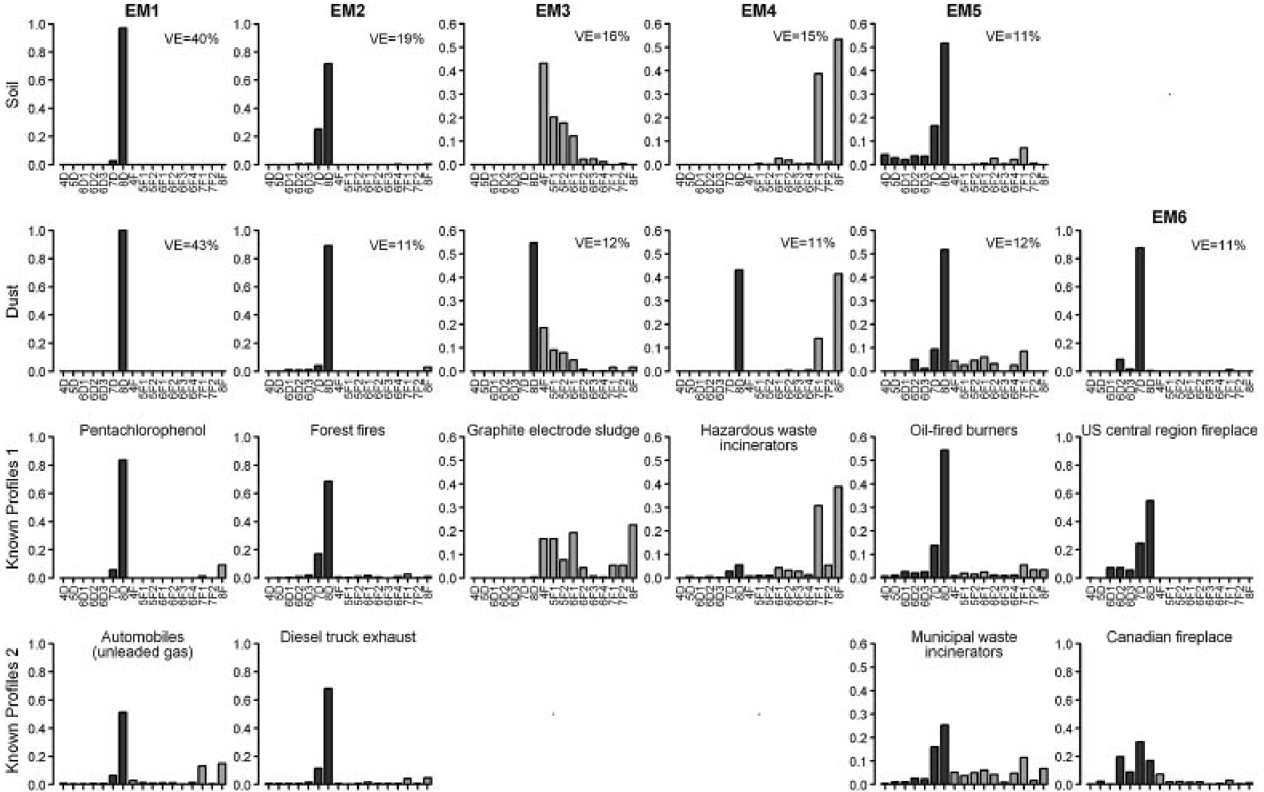

Fig. 1. Congener profiles of end members (EMs) in soil (row 1) and dust (row 2), along with known sources from the U.S. Environmental Protection Agency (U.S. EPA) Source Inventory [29] (rows 3 and 4). Profiles show the fraction of total polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) (PCDD/Fs) contributed by the individual congener. VE = variance explained; 4D = 2,3,7,8-TCDD; 5D = 1,2,3,7,8-PeCDD; 6D1 = 1,2,3,4,7,8-HxCDD; 6D2 = 1,2,3,6,7,8-HxCDD; 6D3 = 1,2,3,7,8,9-HxCDD; 7D = 1,2,3,4,6,7,8-HpCDD; 8D = OCDD; 4F = 2,3,7,8-TCDF; 5F1 = 1,2,3,7,8-PeCDF; 5F2 = 2,3,4,7,8-PeCDF; 6F1 = 1,2,3,4,7,8-HxCDF; 6F2 = 1,2,3,6,7,8-HxCDF; 6F3 = 1,2,3,7,8,9-HxCDF; 6F4 = 2,3,4,6,7,8-HxCDF; 7F2 = 1,2,3,4,6,7,8-HpCDF; 7F3 = 1,2,3,4,7,8,9-HpCDF; 8F = OCDF [Color figure can be seen in the online version of this article, available at wileyonlinelibrary.com.].

For the PCDD/F data sets, these steps resulted in the exclusion of no congeners and nine samples in soil, four congeners (including both TCDD and 1,2,3,7,8-penta-CDD) and one additional sample in dust, and 6 of the 10 furan congeners (but no dioxin congeners) and one additional sample in serum. After eliminating the congeners and samples with large numbers of nondetects, relatively few values below the LOD remained. Of the remaining congener measurements, 5.6, 9.6, and 4.3% were below the LOD in soil, dust, and serum, respectively. These values were imputed as the LOD/$\sqrt{2}$. As a sensitivity analysis, PVA was performed using the dust data set with all of the samples with any value below the LOD removed. The resulting congener profiles were nearly identical to those produced using the data set from which only congeners and samples with >50% of values below the LOD had been removed.

Polytopic vector analysis

We performed PVA using the MATLAB programming language [22]. The PVA algorithm begins with data transformations. Samples are first transformed with a constant row sum, in which each congener concentration is divided by the sum of the total measured PCDD/Fs for that sample. A range transformation is then applied so that the constant row sum–transformed concentrations vary between 0.0 and 1.0 for each congener. The range transformation reduces the influence of the high-concentration congeners. Principal components analysis is then performed on the transformed matrix. The principal component axes are rotated with a varimax rotation followed by an oblique rotation toward extreme values. Finally, the axes are iteratively rotated until a nonnegativity constraint is satisfied. Both the congener patterns (referred to as "end members" [EMs]) and the contribution of each EM to each sample (referred to as "loadings") must satisfy the nonnegativity constraint. The additional rotations and the nonnegativity constraint in PVA differentiate it from principal components analysis and allow the resulting EMs to better represent real-world sources. For each sample matrix, PVA was repeated using a range of different numbers of EMs. Determination of the appropriate number of EMs to retain is based on a number of criteria, including stability and interpretability of EMs, parsimony, percentage of variance explained, coefficient of determination (reproducibility of each congener), and communality (reproducibility of each sample). Further details regarding the PVA algorithm can be found elsewhere [1,2,23,24].

A bootstrapping technique was applied to evaluate the variability of the extracted profiles. For each matrix, PVA was performed on 100 data sets formed by random resampling of the original data set as described by Henry [25]. Based on the 100 outcomes of PVA, a standard deviation for each congener fraction of each EM was calculated.

Cosine θ calculations

The soil and dust congener patterns extracted by PVA were compared to each other and to known source profiles by computing the cosine of the angle between the vectors formed by plotting the congener concentrations in multidimensional space. The cosine θ value between patterns i and j with n congeners was calculated as

$$\cos\theta = \frac{\sum_{k=1}^{n} \chi_{ik}\chi_{jk}}{\sqrt{\sum_{k=1}^{n} \chi_{ik}^2 \sum_{k=1}^{n} \chi_{jk}^2}} \quad (1)$$

where χ is the fraction of congener k of the total measured PCDD/Fs. The cosine θ value is a measure of similarity, like a correlation coefficient. However, because it always yields a positive number, it is a more intuitive metric for pattern comparison [26,27].

Age adjustment for serum

Age may influence how serum congener profiles are extracted because of both differential elimination and differential exposure over time. These factors can influence the correlations among congeners and therefore impact the profiles extracted by PVA. Two methods of adjusting for age were evaluated. The first method was age stratification. The data set was stratified into four age groups (18–29 years, $n = 55$; 30–44 years, $n = 188$; 45–59 years, $n = 369$; and ≥ 60 years, $n = 309$), and PVA was applied to each group. This method allowed for evaluating differences in congener profiles between the different age groups, including how exposures may have changed over time. A similar age-stratification method was applied in a study using PVA on the serum PCB concentrations of Akwesasne Indians [8]. The second method was to control for age, sex, body mass index, breast-feeding, and smoking, as these factors affect congener elimination rates. The methods presented in Milbrath et al. [28] were used to predict elimination rates for each study respondent. A serum concentration was calculated for each congener for each respondent using the predicted elimination rates and a constant intake rate. Intake rates were determined so that the median serum concentration for each congener was reproduced. The constant intake rate was not intended to be a realistic assumption; rather, it was applied so that changes in the intakes of various source patterns over time could be evaluated by examining age trends. Residuals were calculated as the difference between the actual and predicted values for each congener, and the residuals were then shifted so that all values were greater than or equal to zero by adding the minimum residual value. The PVA algorithm was then applied to the positively shifted residuals. This method was intended to adjust only for differential elimination so that temporal exposure differences could still be evaluated.

RESULTS AND DISCUSSION

Soil and household dust results

Five- and six-EM models were selected for the soil and dust PCDD/F PVA, respectively. Figure 1 shows the PCDD/F EMs produced by these models. Figure 1 also includes known PCDD/F source profiles from the U.S. EPA source inventory [29] that are possible matches to the extracted EMs. Table 1 shows the cosine θ values between the soil EMs and dust EMs as well as between the EMs and known profiles from the source inventory.

The soil EMs are ordered by their contribution to explained variance, and the dust EMs are ordered to best match the soil profiles. The five-EM soil model explains 98.8% of the variability, and the six-EM dust model explains 99.4%. Bootstrapping analysis indicates that the EMs are stable: the largest standard deviation for any congener was 0.001 in soil and 0.004 in dust.

Many of the soil and dust PCDD/F EMs are likely related to combustion processes; combustion profiles typically contain variable but large fractions of more highly chlorinated PCDD/Fs, particularly octa-CDD (OCDD) [29]. Photochemical synthesis of OCDD from pentachlorophenol has also been suggested as an important source of OCDD in the environment [30]. Of the five soil EMs, four appear to be related to combustion or

Table 1. Cosine θ values for the congener profiles shown in Figure 1

Comparison	EM1	EM2	EM3	EM4	EM5	EM6
Soil to dust	1.00	0.96	0.38[a]	0.68	0.97	—
Soil to known profile 1[b]	0.99	0.99	0.69	0.98	0.99	—
Soil to known profile 2[c]	0.93	0.98	—	—	0.88	—
Dust to known profile 1	0.99	0.98	0.30[d]	0.73	0.98	0.42
Dust to known profile 2	0.92	0.99	—	—	0.88	0.77

[a] Soil to dust cosine θ value for EM3 is 0.99 if octa-chlorinated dibenzo-*p*-dioxin (OCDD) is excluded.

[b] Known profile 1 corresponds to row 3 in Figure 1.

[c] Known profile 2 corresponds to row 4 in Figure 1.

[d] Dust to known profile 1 is 0.74 if octa-CDD is excluded.

EM = end member.

pentachlorophenol: EM1, which consists entirely of OCDD; EM2, which is also high in OCDD but contains a higher relative contribution from 1,2,3,4,6,7,8-hepta-CDD (HpCDD); EM4, which is dominated by two more highly chlorinated PCDFs (consistent with higher-temperature incineration sources such as hazardous waste incineration); and EM5, which again contains a large fraction of OCDD but includes contributions from all of the PCDD congeners.

The EM that is present in both soil and dust samples but does not appear to be associated with combustion or atmospheric pentachlorophenol is EM3. This EM is composed only of PCDF congeners in soil and PCDF congeners plus OCDD in dust. A potential match for the soil and dust EM3 is graphite electrode sludge [29]. This correspondence in profiles supports the hypothesis that residues from chlor-alkali processes using graphite electrodes at Dow in the early 1900s were a source of the contamination in the Tittabawassee River [11,12]. The cosine θ value between soil EM3 and the known graphite electrode sludge pattern is relatively low (0.68, Table 1). The relatively low cosine θ value is because EM3 has a much lower contribution of octa-CDF than is present in the known graphite electrode sludge pattern. This could be related to dechlorination in the environment, which would tend to decrease the contribution of more highly chlorinated congeners and increase the contribution of less-chlorinated congeners.

Dust EM6, the EM that is present in the dust model but does not have a matching soil EM, is dominated by HpCDD. A possible explanation for the presence of this EM in household dust samples is the influence of wood burning in fireplaces and stoves. The ratio of HpCDD to OCDD varies across wood-burning processes [28]. The HpCDD congener may vary independently in dust because individual fireplaces burn different woods and at different temperatures, creating a range of HpCDD to OCDD ratios. Figure 1 shows the congener profiles of fireplaces from two geographic regions. Neither is a good match with EM6 (Table 1); however, they illustrate the range of HpCDD to OCDD ratios that may be present in wood-burning processes.

Figure 2 shows the distributions of EM loadings for soil and dust in each study population. A comparison of the distributions of loadings shows that differences among study populations are apparent for EM3, particularly in soil. This is the PCDF profile that appears to be linked to graphite electrode sludge, the likely major source of PCDD/F contamination in the Tittabawassee River. The loadings of soil EM3 show that this profile is present in relatively high proportions in both the floodplain and near floodplain soils. The 95th percentile of the loadings of dust EM3 is also elevated in the floodplain and near floodplain.

Additional regional differences are observable in the loadings of PCDD/F soil EMs but not in dust. Specifically, the plume had a higher median loading of PCDD/F EM1 (the OCDD profile), Jackson/Calhoun had a higher distribution of loadings of EM2 (OCDD and HpCDD), and all of the Midland/Saginaw populations had higher loadings of EM5 (all PCDD congeners) compared to Jackson/Calhoun.

Serum results

Serum PVA with no adjustment for age yielded five EMs that explained 98.1% of the variance (Fig. 3). The largest standard deviation for any congener in the bootstrapping analysis was

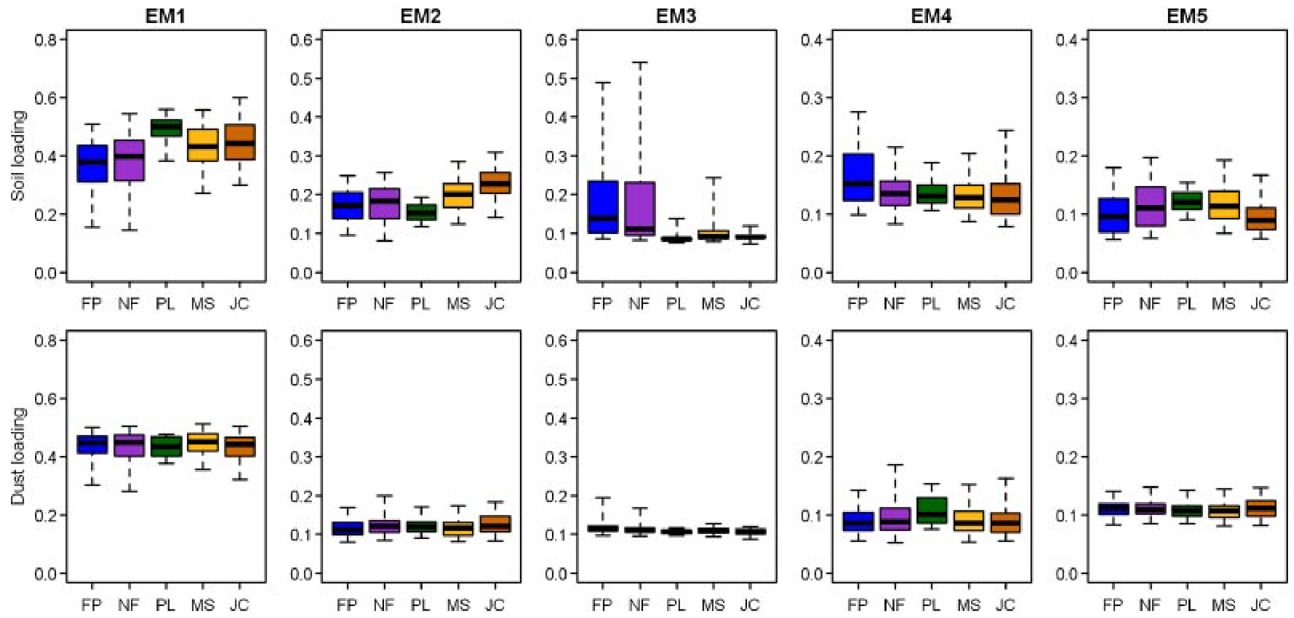

Fig. 2. End member (EM) loadings in soil (row 1) and dust (row 2) by study population. The line in the middle of the box represents the median, the box edges show the 25th and 75th percentiles, and the whiskers extend to the 5th and 95th percentiles. FP = floodplain; NF = near floodplain; PL = plume; MS = Midland and Saginaw counties general population; JC = Jackson and Calhoun counties [Color figure can be seen in the online version of this article, available at wileyonlinelibrary.com.].

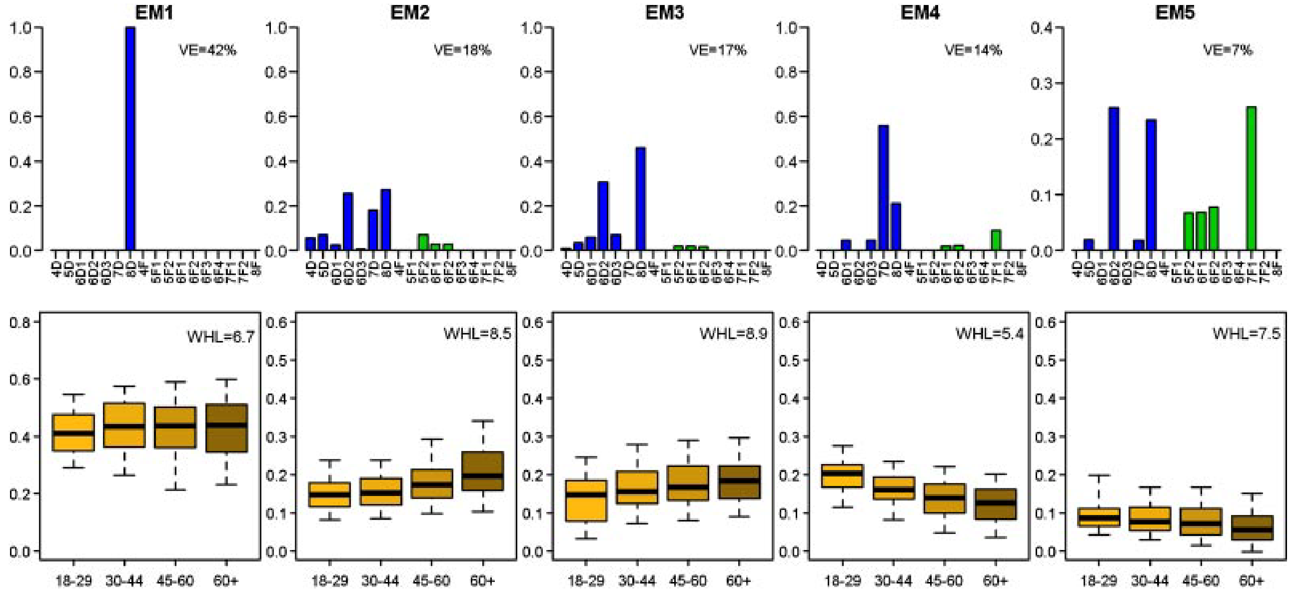

Fig. 3. End members (EMs) extracted from unadjusted serum analysis, the distributions of the EM loadings for four age groups, and the half-life of the congeners weighted by their contribution to each EM. The line in the middle of the box represents the median, the box edges show the 25th and 75th percentiles, and the whiskers extend to the 5th and 95th percentiles. VE = variance explained; WHL = weighted half-life in years. See Figure 1 for additional abbreviations [Color figure can be seen in the online version of this article, available at wileyonlinelibrary.com.].

0.0007. The distribution of the EMs was compared among age groups. Several of the EMs showed noticeable trends with age. To determine if age differences were linked to congener elimination rate, a weighted half-life for each EM was calculated by multiplying the adult reference half-life [28] for each congener by its contribution to the EM. Figure 3 shows that loadings of the EMs with the longest half-lives (EM2 and EM3) increased with age, while the EM with the shortest weighted half-life (EM4) decreased with age. This suggests that differential elimination, rather than differential exposure, was a major determinant in the congener correlations and, therefore, in the extraction of profiles.

The age-stratified serum PVA yielded three EM models for the 18 to 29 and 30 to 44 year age groups and four EM models for the 45 to 60 and over-60 groups (Fig. 4). These models explained 92.3, 95.2, 96.9, and 97.0% of the variability, respectively. The EMs are ordered by the variance explained in the 18 to 29 year-old age group, and the EMs in the other age groups are ordered so that the profiles best match that group. The variability in the bootstrapping EMs is greater in the age-stratified analysis, likely due to the smaller data sets. The maximum standard deviations are 0.01, 0.006, 0.006, and 0.04 for the 18 to 29, 30 to 44, 45 to 60, and >60 year age groups, respectively. The following three general profiles occur in all of the age groups: a profile consisting entirely, or nearly entirely, of OCDD (EM1); a profile that includes a large fraction of OCDD and HpCDD (EM2); and a profile that includes all of the PCDD and PCDF congeners, with a large contribution from 1,2,3,6,7,8-hexa-CDD and a fraction of OCDD that varies across age groups (EM3). In the 45 to 60 and >60 year age groups, an additional EM that resembles EM3 but includes larger contributions from TCDD and 1,2,3,7,8-penta-CDD was also extracted. The consistency in EM composition observed across the age strata suggests that the congener profiles of some of the primary sources of human exposure to PCDD/Fs have remained consistent over time. However, the magnitude of those exposure sources has decreased [31–33].

The elimination rate–adjusted serum PVA yields five EMs that explain 99.5% of the variance (Fig. 5). These EMs were also stable as 0.008 was the largest standard deviation calculated for any congener in the bootstrapping analysis. Two of the EMs derived from the elimination rate–adjusted analysis are similar to those from the age-stratified analysis: an EM dominated by OCDD (EM1) and an EM with a large contribution from 1,2,3,6,7,8-hexa-CDD (EM3).

Figure 6 shows the distributions of the loadings for the age-stratified analysis of the serum. No notable differences in EM loading across study populations were observed in the 18 to 29 year group. End member 1, which primarily consists of OCDD, is slightly elevated in the plume population in the 30 to 44 year age group. End member 1 is still slightly elevated in the plume in the 45 to 60 year age group; however, the largest difference among study populations is found in EM4. This EM is elevated in all of the Midland/Saginaw populations compared to Jackson/Calhoun. In the >60 group, these two differences among study populations are still evident.

The distributions of loadings by study population of the PCDD/F serum EMs from the elimination rate–adjusted PVA are shown in Figure 7. As in the age-stratified analysis, the first row shows that the loadings of some of the PCDD/F EMs vary by study population. End members 1 and 5 vary most notably, but the distribution of loadings in the plume differs from that of the other populations for each of the EMs. End member 1 is elevated only in the plume population, while EM5 is elevated in all of the Midland/Saginaw populations compared to the Jackson/Calhoun population. All of the other EMs have relatively lower loadings in the plume population. Among the older age groups, the plume population has higher OCDD and approximately average concentrations for many of the other congeners (UMDES, "Dioxin Measurements in Blood, House Dust, and Soil"; http://www.sph.umich.edu/dioxin/BDSmeasure.html). Because the loading values for each sample sum to unity, the lower loadings in the plume for EM2, EM3, and EM4 are likely a result of the higher loadings for the OCDD profile (EM1).

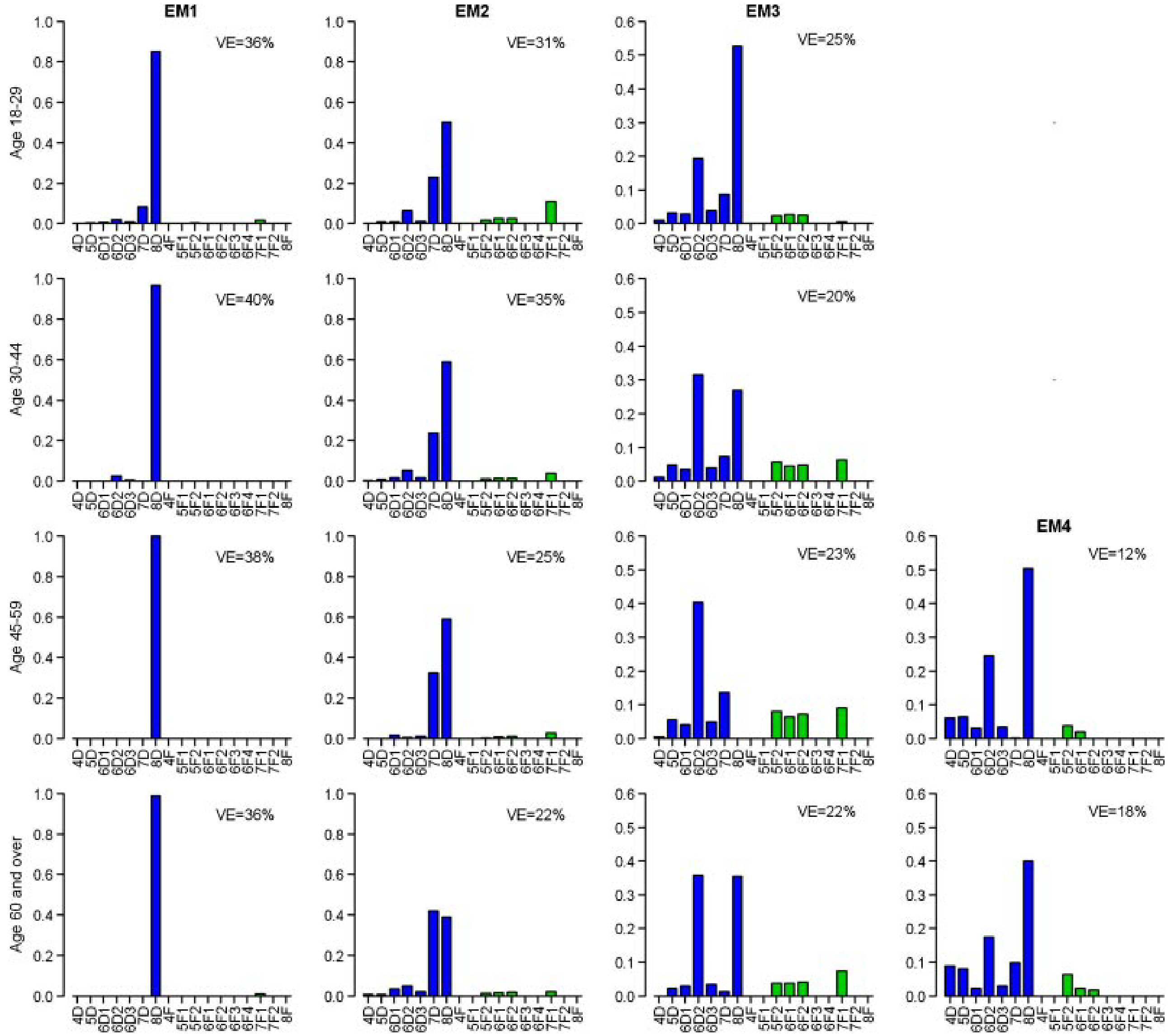

Fig. 4. Congener profiles of end members (EMs) in serum from age-stratified analysis: row 1, 18 to 29 years old; row 2, 30 to 44 years old; row 3, 45 to 50 years old; and row 4, greater than 60 years old. Profiles show the fraction of total polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) (PCDD/Fs) contributed by the individual congener. See Figure 1 for abbreviations [Color figure can be seen in the online version of this article, available at wileyonlinelibrary.com.].

The profiles that exhibit differences in EM loading across the study populations are largely consistent between the age stratification– and elimination rate–adjustment methods. Using both methods, the OCDD profile (EM1) is elevated in the plume population. The congener composition of the other EM that varies across study populations (EM4 in the age-stratified analysis, EM5 in the elimination rate–adjusted analysis) is somewhat different between the two methods. However, in both methods the EM contains TCDD, 1,2,3,7,8-penta-CDD, 2,3,4,7,8-penta-CDF, and 1,2,3,4,7,8-hexa-CDF. Also, in both methods it is the EM that contains the highest fraction of TCDD and 1,2,3,7,8-penta-CDD.

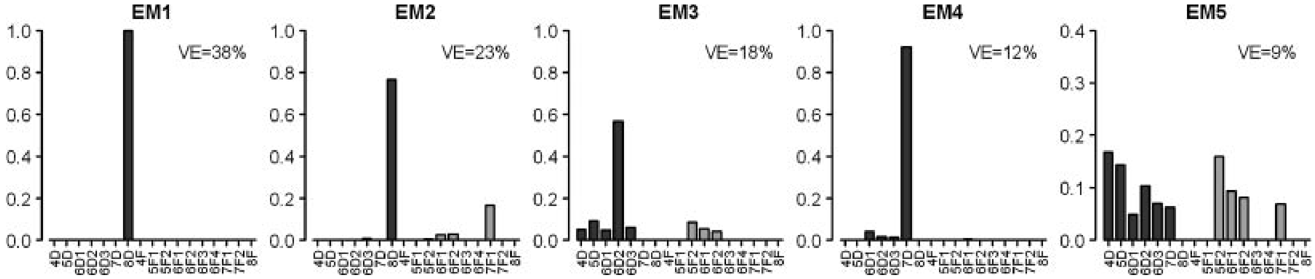

Fig. 5. Congener profiles of end members (EMs) in serum from elimination-rate adjusted analysis. Profiles show the fraction of total polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) (PCDD/Fs) contributed by the individual congener. See Figure 1 for abbreviations [Color figure can be seen in the online version of this article, available at wileyonlinelibrary.com.].

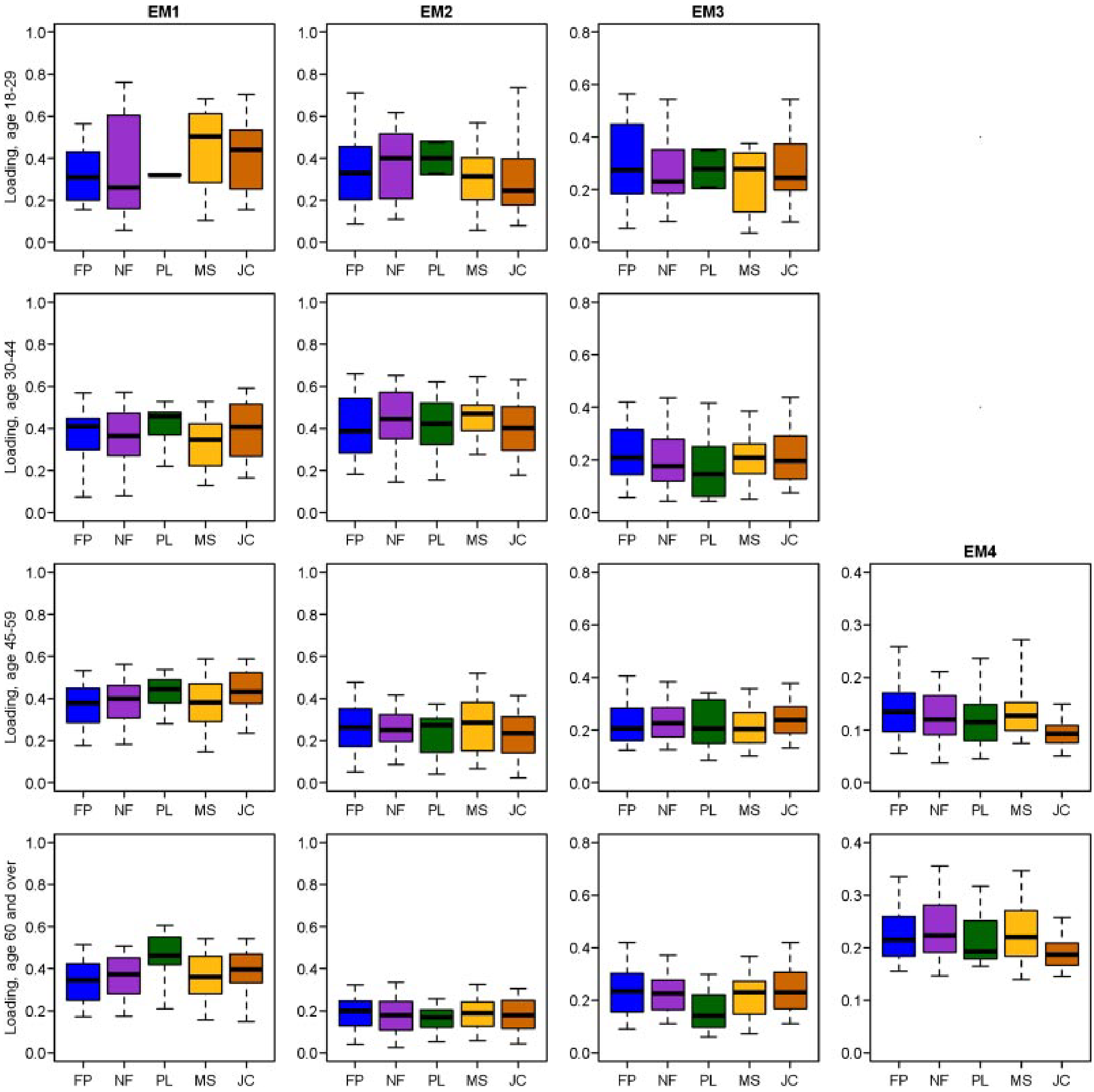

Fig. 6. Distribution of end member (EM) loading by study population for each age stratum. The line in the middle of the box represents the median, the box edges show the 25th and 75th percentiles, and the whiskers extend to the 5th and 95th percentiles. See Figure 2 for abbreviations [Color figure can be seen in the online version of this article, available at wileyonlinelibrary.com.].

Using age stratification, the differences in EM distribution by study population are primarily present in the older age groups (Fig. 6). To determine if age was also a factor in the elimination rate–adjusted PVA, the EM loading for each study population was plotted as a function of age, and a linear trend line was fit (Fig. 7, second row). The age trend lines for EM1 and EM5 diverge with increasing age. Because this method adjusts for both the elimination rate and the change in elimination rate with age, the divergence in trend lines suggests that historic exposures are responsible for the differences in study populations. If the exposure sources were current, the linear age trends of EM loading for each population would be expected to be closer to parallel, and some difference among the study populations would be evident for the youngest study participants. In fact, because background PCDD/F sources have decreased dramatically since 1970 [31–33], older individuals still have serum PCDD/F contributions from the peak emission period, and EM loadings must sum to one for every individual, the loadings resulting from a relatively constant, ongoing source would be expected to be higher for young people.

Only two of the EMs exhibited regional variability; however, Figure 7 shows that several EMs varied with age. In the case of EM2 and EM3, the age trends were consistent with the weighted half-lives of the EMs in that the loadings of EMs with longer weighted half-lives increased with age. Although the congeners were adjusted for elimination rate prior to the PVA application, differential elimination may still be a factor in determining which congeners occur in stable patterns. This is because the congeners with the longest half-lives would be the most likely to have serum levels that continue to reflect exposures from the peak emission period. However, EM1 and EM3 increase more, or decrease less, with age than would be

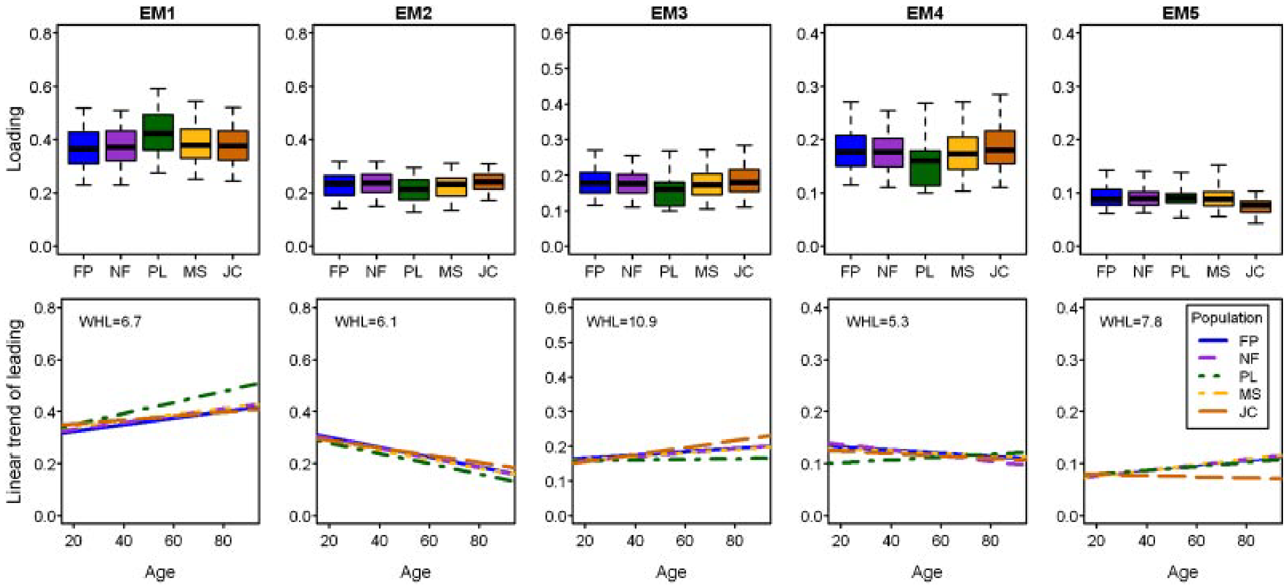

Fig. 7. Distribution and linear trends of end member (EM) loading by study population for elimination-rate adjusted analysis. In the boxplots in the top row, the line in the middle of the box represents the median, the box edges show the 25th and 75th percentiles, and the whiskers extend to the 5th and 95th percentiles. See Figure 2 for abbreviations [Color figure can be seen in the online version of this article, available at wileyonlinelibrary.com.].

expected based on a comparison of the weighted half-lives. This suggests that the congeners in these EMs may have represented a larger share of PCDD/Fs in historic exposures.

The two methods of adjusting for age, age stratification and elimination rate adjustment, yielded partially consistent results; and both proved valuable in the evaluation of PCDD/F profiles in serum. Stratifying by age yielded EMs that are more intuitively interpretable because they are based on the original concentrations, not residuals. However, dividing the data into four age groups within each population resulted in relatively few samples per population. On the other hand, the analysis of residuals after adjusting for elimination rates allows for the use of the entire data set, leading to less variable EM compositions and more straightforward comparisons among populations. Additionally, the method using elimination-rate adjustment clearly demonstrated that the relative EM loading differences among populations were age dependent and, therefore, likely related to historical exposures.

Serum congener profiles may be affected by a number of factors, particularly differences in diet and congener-specific elimination rates. The observed regional differences in EM loading could be related to these factors. However, the differences observed in the two serum EMs that vary by population appear to be consistent with some of the differences in the soil EMs that vary by population, which suggests localized exposure sources. The difference in the OCDD-dominated EM1 loading occurs only in the plume population in both soil (Fig. 2) and serum (Figs. 6 and 7), suggesting that this impact may be related to Dow emissions. The other serum EM that exhibits differences in loading across populations (Fig. 3, EM4, and Fig. 4, EM5) varies in composition, by both age strata and age-adjustment method, but always contains the highest fraction of TCDD and 1,2,3,7,8-penta-CDD of any of the extracted EMs. Soil EM5 contains the highest fraction of these congeners among the soil EMs (Fig. 1, first row), and the soil EM5 loadings are elevated in the Midland/Saginaw study populations (Fig. 2). The loadings of soil EM5 decrease with distance from the Dow facility in the samples collected from 0 to 1 inch, indicating that this profile, along with that of EM1, may also be related to past incinerator emissions. This conclusion is supported by a hierarchical cluster analysis of the soil concentrations, which indicated that Dow incinerator emissions are characterized by a relatively higher fraction of TCDD compared to other combustion sources in the study area [14].

The age trend of the loadings of the serum EM containing TCDD and 1,2,3,7,8-penta-CDD (Fig. 7, EM5) suggests that the source is historic. This hypothesis is consistent with Dow incineration activities as a potential source: Dow made numerous pollution-control improvements to its incinerator operations during the period 1975 to 1988, including the addition of natural gas to its incineration stream to increase combustion temperatures and the installation of a wet electrostatic precipitator [11]. The elevated levels of this serum EM in the Midland/Saginaw study populations and the indication that it may be related to historic releases are also consistent with the linear regression results reported by Garabrant et al. [14]. The regression found an association between having lived in Midland and Saginaw Counties during the period 1960 to 1979 and higher serum levels of TCDD and 1,2,3,7,8-penta-CDD. Both the PVA and the linear regression results suggest that historic aerial emissions from the Dow facility likely impacted the serum PCDD/F levels of the population of Midland and Saginaw counties.

CONCLUSIONS

The present study evaluated PCDD/F patterns in soil, dust, and serum samples. The soil EMs are similar to the dust EMs and can be linked to known source patterns. The loadings of several EMs varied by study population in the soil samples, including an EM dominated by OCDD, which is elevated in the plume population; an EM dominated by PCDFs, which is elevated in the soils of the populations in the floodplain and near floodplain of the Tittabawassee River; and a combustion EM with higher-than-typical fractions of TCDD and 1,2,3,7,8-penta-CDD, which was elevated in all of the populations in

Midland and Saginaw Counties. In household dust, the 95th percentile loadings of the PCDF-dominated EM are elevated in the floodplain and near floodplain populations. However, the other differences among populations observed in the soil samples are not found in the dust samples.

Loadings of two-serum EMs varied among study populations, and both signatures may be related to historic emissions from the incinerator complex at the Dow facility. The age trends of the loading differences indicated that the exposures were likely historic. This investigation did not find evidence that contamination in the Tittabawassee River floodplain or in the depositional area of the incinerator complex is currently impacting serum PCDD/F profiles. However, because the number of EMs is limited and large individual variability exists in terms of current exposures, past exposures, and elimination rates, detection of relatively small differences in serum PCDD/F profiles may not be possible using PVA.

SUPPLEMENTAL DATA

Figure S1. (418 KB PDF).

Acknowledgement—Financial support for the present study comes from the Dow Chemical Company through an unrestricted grant to the University of Michigan. The authors acknowledge the contributions of the entire UMDES study team, Vista Analytical, S.Vantine for her assistance with UMDES administration, and L. Birnbaum, R. Hites, P. Boffetta, and M.H. Sweeney for their guidance as members of the UMDES Scientific Advisory Board.

REFERENCES

1. Ehrlich R, Wenning RJ, Johnson GW, Su SH, Paustenbach DJ. 1994. A mixing model for polychlorinated dibenzo-*p*-dioxins and dibenzofurans in surface sediments from Newark Bay, New Jersey, using polytopic vector analysis. *Arch Environ Contam Toxicol* 27:486–500.
2. Huntley SL, Carlson-Lynch H, Johnson GW, Paustenbach DJ, Finley BL. 1998. Identification of historical PCDD/F sources in Newark Bay estuary subsurface sediments using polytopic vector analysis and radioisotope dating techniques. *Chemosphere* 36:1167–1185.
3. Barabás N, Adriaens P, Goovaerts P. 2004. Modified polytopic vector analysis to identify and quantify a dioxin dechlorination signature in sediments. 1. Theory. *Environ Sci Technol* 38:1813–1820.
4. Barabás N, Adriaens P, Goovaerts P. 2004. Modified polytopic vector analysis to identify and quantify a dioxin dechlorination signature in sediments. 2. Application to the Passaic River. *Environ Sci Technol* 38:1821–1827.
5. Johnson G. 1997. *Application of polytopic vector analysis to environmental geochemistry investigations* PhD thesis. University of South Carolina, Columbia, SC, USA.
6. Johnson G, Ehrlich R, Full W. 2002. Principal components analysis and receptor models in environmental forensics. In Murphy B, Morrison R, eds, *Introduction to Environmental Forensics*. Academic, New York, NY, USA, pp 461–515.
7. Magar VS, Johnson GW, Brenner RC, Quensen JF, Foote EA, Durell G, Ickes JA, Peven-McCarthy C. 2005. Long-term recovery of PCB-contaminated sediments at the Lake Hartwell Superfund site PCB dechlorination. 1. End-member characterization. *Environ Sci Technol* 39:3538–3547.
8. DeCaprio AP, Johnson GW, Tarbell AM, Carpenter DO, Jeffrey R, Chiarenzelli JR, Morse GS, Santiago-Rivera AL, Schymura MJ. Akwesasne Task Force on the Environment. 2005. Polychlorinated biphenyl (PCB) exposure assessment by multivariate statistical analysis of serum congener profiles in an adult Native American population. *Environ Res* 98:284–302.
9. Hilscherova K, Kannan K, Haruhiko N, Nobuyasu H, Nobuyoshi Y, Bradley PW, McCabe JM, Taylor AB, Giesy JP. 2003. Polychlorinated dibenzo-*p*-dioxin and dibenzofuran concentration profiles in sediments and flood-plain soils of the Tittabawassee River, Michigan. *Environ Sci Technol* 37:468–474.
10. Demond A, Adriaens P, Towey T, Chang S-C, Hong B, Chen Q, Chang C-W, Franzblau A, Garabrant DH, Gillespie B, Hedgeman E, Knutson K, Lee S-Y, Lepkowski J, Olson K, Ward B, Zwica L, Luksemburg W, Maier M. 2008. Statistical comparison of residential soil concentrations of PCDDs, PCDFs, and PCBs from two communities in Michigan. *Environ Sci Technol* 42:5441–5448.
11. Ann Arbor Technical Services. 2006. *Remedial investigation work plan: Tittabawassee River and Upper Saginaw River*. Ann Arbor Technical Services, MI, USA.
12. Towey T, Chang S-C, Demond A, Wright DW, Barabás N, Franzblau A, Garabrant DH, Gillespie B, Lepkowski J, Luksemburg W, Adriaens P. 2010. Hierarchical cluster analysis of polychlorinated dioxins and furans in Michigan, USA, soils: Evaluation of industrial and background congener profiles. *Environ Toxicol Chem* 29:64–72.
13. Goovaerts P, Trinh H, Demond A, Franzblau A, Garabrant D, Gillespie B, Lepkowski J, Adriaens P. 2008. Geostatistical modeling of the spatial distribution of soil dioxins in the vicinity of an incinerator. 1. Theory and application to Midland, Michigan. *Environ Sci Technol* 42:3648–3654.
14. Garabrant DH, Franzblau A, Lepkowski J, Gillespie B, Adriaens P, Demond A, Ward B, LaDronka K, Hedgeman E, Knutson K, Zwica L, Olson K, Towey T, Chen Q, Hong B. 2009. The University of Michigan Dioxin Exposure Study: Methods for an environmental exposure study of polychlorinated dioxins, furans and biphenyls. *Environ Health Perspect* 117:803–810.
15. Van den Berg M, Birnbaum L, Denison M, De Vito M, Farland W, Feeley M, Fiedler H, Hakansson H, Hanberg A, Haws L, Rose M, Safe S, Schrenk D, Tohyama C, Tritscher A, Tuomisto J, Tysklind M, Walker N, Peterson RE. 2006. The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. *Toxicol Sci* 93:223–241.
16. U.S. Environmental Protection Agency. Method 8290 Polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) by high-resolution gas chromatography/high-resolution mass spectrometry (HRGC/HRMS). Office of Solid Waste and Emergency Response, Washington, DC.
17. U.S. Environmental Protection Agency. Method 1668 revision A: Chlorinated biphenyl congeners in water, soil, sediment, and tissue by high-resolution gas chromatography/high-resolution mass spectrometry (HRGC/HRMS). Office of Water, Washington, DC.
18. Garabrant DH, Franzblau A, Lepkowski J, Gillespie B, Adriaens P, Demond A, Hedgeman E, Knutson K, Zwica L, Olson K, Towey T, Chen Q, Hong B, Chang C-W, Lee S-Y, Ward B, LaDronka K, Luksemburg W, Maier M. 2009. The University of Michigan Dioxin Exposure Study: Predictors of human serum dioxin concentrations in Midland and Saginaw, Michigan. *Environ Health Perspect* 117:818–824.
19. Hedgeman E, Chen Q, Hong B, Chang C-W, Olson K, LaDronka K, Ward B, Adriaens P, Demond A, Gillespie B, Lepkowski J, Franzblau A, Garabrant D. 2009. The University of Michigan Dioxin Exposure Study: Population survey results and serum concentrations for polychlorinated dioxins, furans, and biphenyls. *Environ Health Perspect* 117:811–817.
20. Demond A, Franzblau A, Garabrant D, Jiang X, Adriaens P, Chen Q, Gillespie B, Hao W, Hong B, Jolliet O, Lepkowski J. 2012. Human exposure from dioxins in soil. *Environ Sci Technol* 46:1296–1302.
21. Towey T, Barabás N, Demond A, Zwica L, Knutson K, Franzblau A, Garabrant D, Adriaens P. 2008. Statistical fingerprinting of PCBs using the subset with dioxin-like activity. *Organohalogen Compounds* 70: 1002350.
22. The MathWorks. 2010. *MATLAB* 2010a. Natick, MA, USA.
23. Miesch AT. *Q-mode factor analysis of geochemical and petrologic data matrices with constant row sums*. Geological Survey Professional Paper 574-G. U.S. Geological Survey, Denver, CO, USA.
24. Evans JC, Ehrlich R, Krantz D, Full W. 1992. A comparison between polytopic vector analysis and empirical orthogonal function analysis for analyzing quasi-geostrophic potential vorticity. *J Geophys Res* 97: 2365–2378.
25. Henry RC. 2000. *UNMIX Software*, Version 1.8 manual. West Hills, CA, USA.
26. Johnson G, Quensen J, Chiarenzelli J, Hamilton M. 2006. Polychlorinated biphenyls. In Murphy B, Morrison R, eds, *Environmental Forensics: Contaminant Specific Guide*. Academic, New York, NY, USA, pp 189–292.
27. Demond A, Towey T, Adriaens P, Zhong X, Knutson K, Chen Q, Hong B, Gillespie B, Franzblau A, Garabrant D, Lepkowski J, Luksemburg W, Maier M. 2010. Relationship between polychlorinated dibenzo-*p*-dioxin polychlorinated dibenzofuran and dioxin-like polychlorinated biphenyl concentrations in vegetation and soil on residential properties. *Environ Toxicol Chem* 29:2660–2668.
28. Milbrath MO, Wenger Y, Chang C-W, Emond C, Garabrant D, Gillespie B, Jolliet O. 2009. Apparent half-lives of dioxins, furans, and polychlorinated biphenyls as a function of age, body fat, smoking status, and breast-feeding. *Environ Health Perspect* 117:417–425.

29. U.S. Environmental Protection Agency. An inventory of sources and environmental releases of dioxin-like compounds in the United States for the years 1987, 1995, and 2000. EPA/600/P-3/002F. National Center for Environmental Assessment Office of Research and Development, Washington, DC.
30. Baker JI, Hites RA. 2000. Is combustion the major source of polychlorinated dibenzo-*p*-dioxins and dibenzofurans to the environment? A mass balance investigation. *Environ Sci Technol* 34:2879–2891.
31. Pinsky P, Lorber MN. 1998. A model to evaluate past exposure to 2,3,7,8-TCDD. *J Expo Anal Environ Epidemiol* 8:187–206.
32. Aylward LL, Hays SM. 2002. Temporal trends in human TCDD body burden: Decreases over three decades and implications for exposure levels. *J Expo Anal Environ Epidemiol* 12:319–328.
33. Lorber M. 2002. A pharmacokinetic model for estimating exposure of Americans to dioxin-like compounds in the past, present, and future. *Sci Total Environ* 288:81–95.