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Spatial distribution of Chlordanes and PCB congeners in soil in Cedar Rapids, Iowa, USA

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Abstract

Residential soils from Cedar Rapids, Iowa, USA were collected and analyzed for chlordanes and polychlorinated biphenyls (PCBs). This study is one of the very few urban soil investigations in the USA. The chlordanes concentrations ranged from 0 to 7500 ng g⁻¹ dry weight (d.w.), with a mean and standard deviation of 130 ± 920 ng g⁻¹ d.w., which is about 1000 times larger than background levels. ΣPCB concentrations ranged from 3 to 1200 ng g⁻¹ d.w., with a mean and standard deviation of 56 ± 160 ng g⁻¹ d.w. and are about 10 times higher than world-wide background levels. Both groups exhibit considerable variability in chemical patterns and site-to-site concentrations. Although no measurements of dioxins were carried out, the potential toxicity due to the 12 dioxin-like PCBs found in the soil is in the same order of magnitude of the provisional threshold recommended by USEPA to perform soil remediation.

Keywords

Chlordanes; PCBs; urban soil; TEQ

1. Introduction

Persistent organic pollutants (POPs) like chlordane and polychlorinated biphenyls (PCBs) are anthropogenic pollutants found throughout the globe. Their widespread distribution occurred because of how they were used and their physical-chemical properties. Chlordane was produced as an insecticide and consists of a mixture of chlorinated compounds. PCBs were manufactured throughout the world in large volumes for a wide variety of industrial uses. Production of both groups of compounds was banned in the 1970s and 1980s because of concerns about their tendency to bioaccumulate in animals and humans and because of laboratory tests showing that they may be harmful to human (ATSDR, 1994, 2000). The bans produced marked reduction in environmental exposure. The history of rise and decline in environmental exposure has been reported from sediment cores collected in lakes and remote areas (Eisenreich et al., 1989; VanMetre and Callender, 1997).

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Supporting Data Available

Five figures and 4 tables are available as supplementary data, which can be found free of charge via the Internet at X.

Despite clear evidence of environmental recovery in remote areas, it is unclear if the production ban eliminated or reduced human exposure to these compounds. Of particular interest is the residual levels of these compounds in soils of residential areas and places where children and other vulnerable populations may be exposed on a regular basis. Soil is one of the largest reservoirs of POPs. Meijer et al. (2003) estimated that the global accumulation of PCBs in soils is at least 21,000 t. The global reservoir of chlordanes compounds has not been evaluated, but because these compounds were often applied directly and purposefully to crops and soils, the accumulation of chlordanes may also be large and widely distributed. At these magnitudes, it is surprising how few measurements of soil concentrations have been reported for these compounds in the USA.

Most reports of chlordanes or PCBs in the USA soil concern remote areas (USEPA, 2007). There seems to have been much more interest in global background levels than in populated locations. In fact, we have found only one study that reports urban-residential soil concentrations for chlordanes in USA (Mattina et al., 1999) and only two studies that report USA urban-residential soil concentrations of PCBs, one of which was conducted in the 1970s and the other near a PCB superfund site in Massachusetts (Carey et al., 1979; Vorhees et al., 1999). More studies were conducted in urban areas elsewhere. For example, (Cachada et al., 2009) reported concentrations of PCBs in five European cities and found the concentrations were very heterogeneous. The authors suggested that local sources and the age of contamination greatly affect observed levels. Indeed, the expected variability in urban concentrations may be a major reason why few studies have been conducted. Because these compounds were used in such a variety of ways and with little oversight, it is difficult to plan an urban field campaign that adequately captures the real variability of expected concentrations.

The lack of urban measurements in typical USA cities is a major concern that this study addresses. We examined residential soils in a small Midwestern American city. Cedar Rapids, Iowa is a typical small medium-sized American city, with a population of ca. 130 000, located in a large agricultural region of the USA. The grain processing industry is the most important industrial sector. No records of production, heavy use, or concentrated disposal (i.e. Superfund Sites) of these compounds are known in Cedar Rapids. However, there is clear evidence of the presence of chlordanes and PCBs in the only urban lake in the city. The concentrations of these two compound groups in Cedar Lake sediments is up to 2000 ng g⁻¹ d.w. (DNR, 2001; Lubben, 1994). Runoff of urban water/sediment into the lake was determined to be the main source of both chemicals (Lubben, 1994). In June 2008, a catastrophic flood occurred in Cedar Rapids. Flooding of the Cedar River exceeded the historical record of flood discharge in Cedar Rapids and affected a large portion of residential, commercial and industrial land in the city (ca. 23 km²), including Cedar Lake (Mutel, 2010). It is not known if this flooding caused any redistribution of these pollutants in the city.

Given the absence of urban soil studies in the USA, especially in locations where there is no record of production or intense use, the aim of this investigation was to measure chlordanes and PCBs from surficial soil. Although they were used in different periods and for different purposes, we hypothesized that both compound groups are at levels similar to those reported in other locations around the world. Ratios between the different chlordanes and PCB congeners profile in the samples, technical commercial mixtures and background signals were used to evaluate the variability of the samples. We also investigated the potential toxicity of the soil (TEQs) and how it compares with other locations, as well as the USEPA preliminary soil toxicity guideline. The findings presented here allow us to better understand the actual levels of these chemicals in urban locations of the USA and elsewhere.

2. Methods

2.1 Sampling Method

Sixty-six soil samples were collected in Cedar Rapids on August 25th, 2008 - approximately 70 days after a major flood (Mutel, 2010). We avoided sampling on industrial property and focused on residential land use; the public right-of-way in front of homes and near the street. Most of the sampling sites were inside the estimated flood area (94%) (Figure 1). We concentrated sampling sites south of Cedar Lake (downstream) and west of the Cedar River. The total area of sampling covered almost 10 km². Approximately 1 kg (ca. 12 cm deep) of soil was collected using a trowel. The soil was placed in a labeled plastic Ziplock freezer bag. The trowel and any other instruments used for collection were washed with deionized water and dried with a clean paper towel. The samples were brought to The University of Iowa and kept refrigerated at 4° C until extraction and analysis.

2.2 Analytical Method

The analytical method employed here is described elsewhere (Martinez et al., 2010). Briefly, soil samples (no grass or roots) were manually homogenized, weighed (ca. 5 g) and mixed with a known amount of combusted diatomaceous earth and spiked with 50 ng of surrogate standard, PCB14 (3,5-dichlorobiphenyl), PCB65 (2,3,5,6-tetrachlorobiphenyl) or d-PCB65 (2,3,5,6-tetrachlorobiphenyl-d5) and PCB166 (2,3,4,4',5,6-hexachlorobiphenyl). The samples were extracted by pressurized solvents (Accelerated Solvent Extractor, Dionex ASE-300) with equal parts acetone and hexane. The soil water content was determined gravimetrically for each sample from a separate aliquot by drying for 12 hrs at 104° C. Polar interferences and other compounds were removed from the solvent extracts with sulfuric acid. The final hexane extracts were passed through a Pasteur pipette filled with 0.1 g of combusted silica gel and 1 g of acidified silica gel (2:1 silica gel:acid weight/weight) and eluted with hexane. The final solutions were concentrated to 0.5 mL and 100 ng internal standard PCB204 was added to each sample. (2,2',3,4,4',5,6,6'-octachlorobiphenyl).

Chlordanes (TC, CC and TN) were analyzed using a Gas Chromatography/Mass Selective Detector Mode (GC/MSD, Hewlett-Packard 5973) in selected ion monitoring mode. The gas chromatograph (GC) was equipped with a Supelco SLB-5ms capillary column (30 m × 0.25 mm ID, 0.25 µm film thicknesses) with helium as carrier gas. PCB quantification was carried out employing a modification of EPA method 1668B (USEPA, 2008). Tandem Mass Spectrometry GC/MS/MS (Quattro Micro™ GC, Micromass MS Technologies) in multiple reaction monitoring (MRM) mode was used to quantify all 209 congeners in 164 individual or coeluting congener peaks. The GC was equipped with a Supelco SBP-Octyl capillary column (30 m × 0.25 mm ID, 0.25 µm film thicknesses) with helium as carrier gas at a constant flow rate of 0.8 mL min⁻¹.

2.3 Quality Assurance and Control (QA/QC)

QA/QC was rigorously assessed using surrogate PCB standards, blanks, replicates and standard reference material. In the case of PCBs, congener profiles were also visually inspected for each sample. PCB166 was employed as surrogate standard for chlordane compounds analysis. The mean and standard deviation of the recovery of spiked PCB166 was 96 ± 12%, with a relative standard deviation (RSD) of 12%. Samples that yielded percentage recovery below 40% were reanalyzed. Chlordane masses were corrected using the percentage recovery of PCB166. No laboratory contamination was found in any of the blanks, thus masses were not corrected for laboratory blanks. Samples from 6 sites were extracted and analyzed 3 times for chlordanes. The triplicates yielded < 25% RSD. In the case of PCBs, percentage recovery of surrogate standard PCB14, PCB65, d-PCB65 and PCB166 yielded 79 ± 12%, 89 ± 13%, 81 ± 9% and 96 ± 10%, respectively. RSD for each of

the surrogate standards were below 16%. Samples that yielded percentage recoveries of any of the surrogate standards below 40% were reanalyzed. PCB congener masses were corrected using the percentage recovery of congeners PCB14 (congeners 1 to 39), PCB65 or d-PCB65 (congeners 40 to 128) and PCB166 (congeners 129 to 209), except for sample F1, where PCB65 was excluded due to an unknown interference. One sample extract was quantified 3 times (same collected sample), yielding a 7% RSD. Mass and congener in each sample were compared to their respective laboratory blanks (1 blank for every 9 samples), and if the laboratory blanks contained > 20% of total mass of PCBs detected in the sample or the same congener was detected in both sample and laboratory blank, the sample was not further considered and reanalyzed. Congener masses were not corrected for laboratory blanks. Two samples could not be reanalyzed because of lost sample, hence 64 sites are reported here for PCBs. Nondetects were set to 0.0. Sediment from New York, New Jersey Waterway (SRM 1944, National Institutes of Standards and Testing) was extracted and analyzed to test the accuracy of our methods. The analysis of SRM 1944 resulted in identification of all congeners, with an acceptable result with respect to the certified values (Figure S1). The mean percentage different between the measured and certified values was $9 \pm 6\%$ for 27 congeners reported.

3. Results and Discussion

3.1. Chlordane Compounds in Soil

Technical chlordane together with heptachlor were widely used as a residential termiticide, and a general insecticide on agricultural crops (corn and citrus), home garden and lawn use, and turf and ornamentals (US National Library of Medicine, 1993). Technical chlordane is a mixture of ca. 140 chemicals, where *trans*-chlordane (CT), *cis*-chlordane (CC) and *trans*-nonachlor (TN) are approximately 25% in weight of the total technical mixture (Dearth and Hites, 1990). In 1988, the USEPA banned all uses of chlordane (USEPA, 1990).

Chlordane concentrations in Cedar Rapids soils (sum of TC, CC and TN) ranged from 0 to 7500 ng g⁻¹ d.w. (Table 1, Table S1). The chlordane concentration distribution is highly skewed (median = 4 ng g⁻¹ d.w. << mean = 130 ng g⁻¹ d.w.) (Figure S2). Individual TC, CC and TN also yielded a highly skewed distribution (Table 1). Generally, TN (76%) was found in higher concentration than CC (12%) and TC (11%). Figure 1 shows the spatial distribution of chlordanes and PCBs found in Cedar Rapids. No spatial correlation between the samples was found, using the sum of chlordanes, as well as the individual chemicals (Moran's I Test and variograms were performed). No significant difference was found between samples inside and outside the flooded area.

The chlordane compound values measured in Cedar Rapids are in the top extreme of samples reported for urban, rural/agricultural and background locations around the world, from ca. 1 to 4 orders of magnitude higher (Figure 2, Table S4). Values reported by (Carey et al., 1979) were not included here because there is no mention of the type of chlordane chemicals analyzed. The world-wide data suggest that there is no relationship between the type of soil and the concentration of chlordanes reported, although the lowest value reported is soil from the Andes, Peru (Tremolada et al., 2008).

3.2. Chlordane Ratio Distributions Analysis

Differences in ratios between the 3 chemicals (i.e. CC:TC, TN:TC and CC:TN) within the samples and also in the technical mixture allow us to analyze possible chlordane weathering processes occurring in the soil, as well as sources. The mean and standard deviation of 5 different measurements of technical chlordane exhibit ratios of CC:TC = 0.89 ± 0.09 , TN:TC = 0.54 ± 0.19 and CC:TN = 1.79 ± 0.54 (Cochrane and Greenhalgh, 1976; Dearth and Hites, 1990; Jantunen et al., 2000; Mattina et al., 1999; Sovocool et al., 1977). The

mean ratios for our soil samples are CC:TC = 1.40 ± 0.48 , TN:TC = 2.40 ± 1.50 and CC:TN = 0.96 ± 1.50 . The sample mean CC:TC is significantly higher than the technical mixture ratio ($p < 0.001$) and the site with the highest chlordane concentration ($7500 \text{ ng g}^{-1} \text{ d.w.}$) presents a ratio of 1.60, suggesting significant weathering of the technical mixture, if direct application was the source. TC is slightly more volatile than CC, and the mean ratio CC:TC > 1 in soil is consistent with relative loss of TC via volatilization (Bidleman et al., 2002). Aerobic biodegradation in soil is also possible, because it has been reported that TC is more labile than CC (Beeman and Matsumura, 1981). It seems that the CC:TC ratio cannot explain if chlordanes were used in crops or as a termiticide, or the location of the source of chlordane (local vs. long atmospheric transport). There is no relation between CC:TC reported world-wide and the type of soil sample (agricultural, urban, etc.) and location (e.g. background) (Figure 3). For example, high altitude/background soils yielded a very similar CC:TC ratio (1.30) (Tremolada et al., 2008) to the one obtained here.

3.3. PCBs in Soil

PCBs consist of 209 different chemical compounds whose biological and chemical properties are very similar to chlordanes. They persist in the environment and are semivolatile but were manufactured and employed for various industrial processes. PCBs were banned from production and new use in the late 1970s (ATSDR, 2000).

The summed concentration of the 164 congener peaks (ΣPCB) in Cedar Rapids soils ranged from 3 to $1200 \text{ ng g}^{-1} \text{ d.w.}$ (Table 1, Table S2). The ΣPCB concentration distribution is highly skewed (median = $20 \text{ ng g}^{-1} \text{ d.w.}$ < mean = $56 \text{ ng g}^{-1} \text{ d.w.}$) (Table 1), and follows a lognormal distribution (Figure S2). Figure 1 shows the spatial distribution of ΣPCBs found in Cedar Rapids. As for chlordanes, no spatial correlation was found and no statistical difference between samples inside and outside the flooded area was observed (different quartiles, untransformed and log transformed data, total and individual congeners were used). Conducting a comparison of PCBs reported elsewhere is not straightforward due to the difference in PCB congeners, as well as the quantity of them that each study reports. The diversity of analytical techniques used also result in coelution issues. Nevertheless, Cedar Rapids values are in the same range or higher than values reported around the world for urban locations (Figure 4). As expected, sites near known PCB contamination sites present much higher concentrations than Cedar Rapids, such as New Bedford Harbor, MA (Vorhees et al., 1999), the Superfund site in Kalamazoo, MI (Blankenship et al., 2005), and Wenling, China (e-waste recycling city) (Tang et al., 2010). In Gadsden, AL, PCBs are about 5 times higher than Cedar Rapids- perhaps because the samples were collected in 1971 when PCB production was near its peak (Carey et al., 1979).

Rural USA background concentrations are statistically different ($p < 0.01$) than Cedar Rapids' values, in particular with samples collected in McNay Farm, Iowa, $0.35 \text{ ng g}^{-1} \text{ d.w.}$ (USEPA, 2007). Global mean background concentrations of the sum of 29 individual or coeluting congeners are one order of magnitude lower than Cedar Rapids' values, $5.4 \text{ ng g}^{-1} \text{ d.w.}$ (Meijer et al., 2003), and are statistically different ($p < 0.01$) (Figure 4).

3.4. PCB Congener Profile Distributions Analysis

The PCB congener profiles differ from sample to sample (see high standard deviations in mean congener profile, Figure 5). But in general, the soil samples are enriched in penta-, hexa- and heptachlorobiphenyls (85% in mass). The less chlorinated congeners (mono- to trichlorobiphenyls) are very low (4% in mass) in the samples (Figure S3). Comparison with rural soils in the USA (USEPA, 2007) shows that Cedar Rapids mean homolog group profile is significantly enriched in penta- and hexachlorobiphenyls, but significantly depleted from mono- to tetra-, and hepta- to octachlorobiphenyls ($p < 0.05$) (Figure S3).

To compare the similarity between congener profiles in each sample with commercial Aroclor mixtures, as well as with Iowa background samples, the cosine theta metric ($\cos \theta$) was calculated. This metric uses the cosine of the angle between two multivariable vectors (the profiles) where a value of 0.0 describes two completely different vectors and 1.0 describes two identical (DeCaprio et al., 2005). For our sample set, only 13% of the samples exhibited a similarity with another sample (i.e. $\cos \theta > 0.9$). No spatial correlation was found between those paired samples. Sample F1 (highest concentration, 1200 ng g⁻¹ d.w.) showed the lowest $\cos \theta$ with the rest of the samples (< 0.3). Commercial mixtures Aroclors 1016, 1221, 1242, 1248 and 1254 were analyzed and quantified using our same analytical method. Comparison between samples and Aroclors showed little similarity ($\cos \theta$ ca. 0.3), except for samples E35 (Aroclor 1248, $\cos \theta > 0.8$) and F1 (Aroclor 1016, $\cos \theta > 0.8$) (Figure S4). Although the background concentration in Iowa is at least 1 order of magnitude lower than Cedar Rapids' values, the congener profiles are similar. Indeed, the $\cos \theta$ between our samples and the Iowa background sample yielded a mean of 0.77, except for samples E35 and F1 ($\cos \theta < 0.4$). Hence, most of the samples resemble the background signal. By visual inspection, it was also possible to find a similarity between our Cedar Rapids samples, the background Iowa signal and 2 soil samples from New Bedford Harbor, MA, (Vorhees et al., 1999).

3.5. TEQs

PCBs cause effects in humans and laboratory animals related to cancer, endocrine disruption, neurological damage, and developmental disturbance. Historically, the toxicity of PCB congeners has been compared to 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD). Although this measure does not capture all potential avenues of PCB toxicity, toxic equivalents (TEQs) are commonly used to quantitatively compare potential toxicity in samples. Potential toxicity to biota and humans through dioxin-like PCBs in surficial soil have been estimated for background locations (USEPA, 2007), urban sites (Franzblau et al., 2009; Wang et al., 2008), as well as impacted urban areas (e.g. incinerator) and contaminated sites (Blankenship et al., 2005; Franzblau et al., 2009; USEPA, 2002).

We calculated TEQs using the 1998 and 2005 WHO-human toxic equivalent factors (TEFs) for the 12 dioxin-like PCBs (Van den Berg et al., 1998; Van den Berg et al., 2006). Notice that our analytical method allows us to individually quantify 11 of the 12 dioxin-like PCBs, where PCBs156+157 coelute (see mean congener profile in Figure 5). Soil samples ranged from 0 to 63 pg TCDD TEQ g⁻¹ d.w. (mean = 2.7 pg TCDD TEQ g⁻¹ d.w.) (1998 version), and 0 to 71 pg TCDD TEQ g⁻¹ d.w. (mean = 2.4 pg TCDD TEQ g⁻¹ d.w.) (2005 version). None of Cedar Rapids human TEQ values were above the USEPA temporary recommended remediation goals for dioxin TEQ in residential soil, 72 pg TCDD TEQ g⁻¹ d.w. (USEPA, 2009). Moreover, they are in the same range of rural soil values in the USA (USEPA, 2007), and urban soils from Jackson and Calhoun counties, MI (Franzblau et al., 2009). Note that no measurements of dioxins (Polychlorinated Dibenzo-p-dioxins and Dibenzofurans) were conducted on Cedar Rapids soils, which could significantly increase the values in ca. 70% (USEPA, 2002).

4. PCB Soil Concentrations in the USA

Although the USA was the biggest producer and user of PCBs in the world (USEPA, 1976), there is little published data of PCB concentrations from urban soils in the USA. Indeed, Li et al. recently summarized 23 studies of urban soil PCB concentrations, and there is no mention of any American study. On the other hand, there is a good estimate of background/rural soil PCB concentrations in the USA (USEPA, 2007). This is surprising since PCBs were used in cities and it is clear that urban concentrations are higher than background concentrations (Figure 4). (Carey et al., 1979) investigated 5 different sized cities in the

USA, finding that chemicals were higher in urban locations than suburban and agricultural soils. Vorhees et al. (1999) also examined an urban location near New Bedford Harbor, but as a control site to be compared with impacted/contaminated PCBs areas. It is remarkable that the Vorhees et al. report Σ PCB concentrations and congener profiles that are very similar to what we found in Cedar Rapids. This could suggest that Cedar Rapids is representative of American urban-residential soil contamination. This lack of studies also affects TEQs calculations in urban soil. USEPA is attempting to implement a cleanup soil guidance (USEPA, 2009), but without knowing PCB concentrations in urban soil, it seems unrealistic to establish a threshold. The TEQ of Cedar Rapids soils, calculated only from dioxin-like PCBs, is within an order of magnitude that the preliminary threshold currently in consideration.

5. Conclusions

This is one of the few studies of chlordanes and PCBs in urban soils in the USA, and perhaps the only one in the last 2 decades. Although these two chemicals were used in different time periods and for different purposes (chlordanes-agriculture/household vs. PCBs-industrial) but we did not observe any spatial relationship with specific applications or uses. The spatial distributions found for both chlordanes and PCBs do not reflect any type of particular spatial trend, including distance from particular industries or industrial areas or distance from the river or Cedar Lake. No statistical significance was found between samples inside and outside the flood areas. Most of the samples were collected in the 500-year floodplain, although no flood of this size has ever been recorded for this city before 2008. The spatial distribution of chlordanes could reflect residues from historical pesticide application on lawns. Although the use of chlordanes as a termiticide was banned in 1988 (USEPA, 1990), chlordanes are highly persistent in the environment and have a half-life in soil of approximately 40 years (Mattina et al., 1999). The presence of dioxin-like PCBs in residential soils is a concern because of the potential for human exposure. Vulnerable populations, such as children are especially susceptible to the negative effects of PCBs and may ingest or otherwise are exposed to residential soils.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Capsule Abstract

Chlordane compounds (*trans*-, *cis*- and *trans*-nonachlor) and PCBs (164 peaks for 209 congeners) were measured in the soils of a small medium-sized American city.

Figure 1.

Spatial location and measured concentrations of Σ PCB (left, 64 sites) and chlordanes (right, 66 sites) (ng g^{-1} d.w.) in soil from Cedar Rapids, Iowa. Samples were collected in August 2008. Estimated flood area was obtained from the Linn County Auditor's Office. For more details see tables S1 and S2.

Figure 2.

Comparison of chlordane concentrations of different locations and type of soils, including Cedar Rapids. Values are the mean, bars represent the standard error. Study (1) reports on urban, agricultural and rural soils; study (2), (5), (6ii), (7) and (8) report only agricultural; study (3) reports background; study (4) reports background and industrial; study (6i) report residential soils, respectively. Studies (3), (4) and (5) only report TC + CC. Refs. (1) (Wong et al., 2010), (2) (Jiang et al., 2009), (3) (Tremolada et al., 2008), (4) (Shegunova et al., 2007), (5) (Cavanagh et al., 1999), (6) (Mattina et al., 1999), (7) (Harner et al., 1999) and (8) (Aigner et al., 1998).

Figure 3.

Comparison of CC:TC ratios for 10 locations, including Cedar Rapids and the mean of technical chlordane mixtures ($n = 5$). See data details in Table S4. Refs. (1) Wong et al. (2010), (2) Jiang et al. (2009), (3) Tremolada et al. (2008), (4) Shegunova et al. (2007), (5) Cavanagh et al. (1999), (6) Harner et al. (1999) and (7) and Finizio et al. (1998).

Figure 4.

Comparison of Cedar Rapids, IA, vs. background (left panel) and urban (right panel) Σ PCB concentrations. In right panel, Cedar Rapids, (4), (5), (7) and (10) are described by the mean, and the rest by the median, due to the available published data. The error bars in right panel is the standard error. The asterisk (*) indicates a statistical difference ($p < 0.01$). New Bedford Harbor (6i) and (6ii) represent the neighborhoods near the Superfund site and the comparison neighborhoods, respectively. The Wenling data (8) are from recycling sites, but do not include the highest value. Kalamazoo, MI, is a Superfund site (10). Σ PCB > 60 congeners, except (3) with 19 and (7) reported total PCB. Refs. (1) (USEPA, 2007), (2) (Meijer et al., 2003), (3) Cachada et al. (2009), (4) Salihoglu et al. (2011), (5) Wang et al. (2008), (6) (Vorhees et al., 1999), (7) (Carey et al., 1979), (8) (Tang et al., 2010), (9) (Li et al.), and (10) Blankenship et al. (2005).

Figure 5.

Mean individual (top panel), mean homolog groups congener profile distributions of soil samples from Cedar Rapids (bottom panel). Each congener was normalized to the total concentration of PCBs in the sample. The error bars represent one standard deviation above the mean. The insert in top plot shows the 12 dioxin-like PCBs average normalized concentrations (see different y-axis scale). Congeners are ordered by “IUPAC” nomenclature (Ballschmiter and Zell, 1980).

Table 1

Summary statistics of soil concentrations of chlordanes and PCBs (ng g⁻¹ d.w.) from Cedar Rapids, Iowa, USA.

Compound	Minimum	Lower quartile	Median	Upper quartile	Maximum	Mean	Standard deviation	Skewness	Coefficient of variance (%)	Geometric mean	Geometric standard deviation
Chlordanes (n = 66 ^a)											
<i>trans</i> -chlordane [CT]	0.00	0.38	0.86	2.50	2400	40.0	300	8.10	740	1.20	5.60
<i>cis</i> -chlordane (CC)	0.00	0.50	1.00	2.60	3900	64.0	480	8.10	750	1.60	5.50
<i>trans</i> - nonachlor (TN)	0.00	0.82	1.80	4.30	1200	24.0	150	8.00	640	2.20	4.10
Σ (TC+CC+TN)	0.00	1.80	3.90	8.90	7500	130	920	8.10	730	5.20	4.90
PCBs (n = 64 ^b)											
mono-	0.00	0.00	0.00	0.00	0.12	0.00	0.02	4.90	480	0.09	1.50
di-	0.00	0.00	0.00	0.00	55.0	0.93	6.80	8.00	730	1.00	5.90
tri-	0.00	0.00	0.03	1.00	640	11.0	80.0	8.00	730	1.10	5.40
tetra-	0.00	0.00	0.74	2.50	490	11.0	61.0	7.90	580	1.90	4.70
penta-	1.30	4.60	7.20	14.0	160	15.0	24.0	4.30	160	8.30	2.60
hexa-	0.85	4.50	6.90	14.0	97.0	13.0	17.0	3.70	140	7.80	2.60
hepta-	0.00	0.91	3.10	5.20	29.0	4.40	5.40	2.50	120	3.00	2.80
octa-	0.00	0.00	0.00	0.00	11.0	0.33	1.40	6.90	410	1.00	2.80
nona-	0.00	0.00	0.21	0.71	4.90	0.52	0.82	2.90	160	0.78	2.00
deca-	0.00	0.00	0.36	0.69	4.60	0.52	0.75	3.00	140	0.61	2.20
ΣPCB	3.00	14.0	20.0	45.0	1200	56.0	160	6.90	280	24.0	2.90

^a 60 samples + 6 triplicate samples.

^b 63 samples + 1 triplicate sample.