# 3. LEVELS OF CDD, CDF, AND PCB CONGENERS IN ENVIRONMENTAL MEDIA AND FOOD

## 3.1. INTRODUCTION

Polychlorinated dibenzo-*p*-dioxins (CDDs), polychlorinated dibenzofurans (CDFs), and polychlorinated biphenyls (PCBs) have been found throughout the world in practically all media including air, soil, water, sediment, fish and shellfish, and other food products such as meat and dairy products. Also, not unexpectedly, considering the recalcitrant nature of these compounds and their physical/chemical properties (i.e., low water solubilities, low vapor pressures, and high $K_{ow}$s and $K_{oc}$s), the highest levels of these compounds are found in soils, sediments, and biota (parts-per-trillion (ppt) and higher); very low levels are found in water (parts-per-quadrillion (ppq) and lower) and air (pg/m$^3$). The widespread occurrence observed is not unexpected considering the numerous sources that emit these compounds into the atmosphere (See Volume II), and the overall resistance of these compounds to biotic and abiotic transformation. (See Chapter 2 of this volume.) Part-per-trillion levels of CDDs/CDFs have been found in everyday materials that are contaminated with dust--clothes dryer lint (2.4 to 6.0 ng I-TEQ$_{DF}$/kg); vacuum cleaner dust (8.3 to 12 ng I-TEQ$_{DF}$/kg); room air filters (27 to 29 ng I-TEQ$_{DF}$/kg); and house furnace filter dust (170 ng I-TEQ$_{DF}$/kg) (Berry et al., 1993). Although Berry et al. (1993) only analyzed one or two samples of these materials, the findings suggest that these compounds may be ubiquitous.

This chapter provides an overview of the concentrations at which dioxin-like compounds have been found in the U.S. environment and food based on data presented in the published literature. This literature summary is not all inclusive, but is meant to present the reader with a general overview of values reported in the recent literature. Only data from Government-sponsored monitoring studies and studies reported in the peer-reviewed literature are discussed in this chapter. Data are presented as presented in the original studies/reports. No attempt was made to verify or assess the adequacy of the quality assurance/quality control (QA/QC) measures employed in these studies beyond those described in the published reports. In order to represent current exposure concentrations, data used for the calculation of background media levels were based on studies published in the late 1980s and 1990s, but primarily in the 1990s. The studies used for the estimation of background concentrations were also chosen on the basis of credibility and representativeness.

CDD/CDF profiles for environmental media are also presented in this chapter. CDD/CDF homologue group and 2,3,7,8-substituted congener profiles were calculated for each medium by dividing the mean concentration of individual homologue groups or congeners by the mean total CDD/CDF concentrations for a group of studies or samples. Total CDD/CDF concentration was calculated as the sum of homologue group concentrations. In some cases, however, homologue group concentrations were not available. When this occurred (i.e., for foods), total concentration was redefined as the sum of the concentrations of the 2,3,7,8-substituted congeners rather than the sum of the homologue group concentrations. The fractions of the total for each congener add up to 1.00 in this case rather than some fraction of 1.00, as they would if the total concentration were, more appropriately, the sum of the homologue group concentrations. The text carefully identifies where this occurred. Nondetects were assumed to be zero in the calculation of CDD/CDF profiles. This was done as a matter of consistency - some studies used did not report on the detection limits for some congeners; some had high detection limits such that an assumption of one-half the detection limit would have led to unreasonably high contributions of some congeners to total CDD/CDFs. When available, data on media levels in European countries and in other parts of the world are also presented for comparison to U.S. values. These data are not intended to provide estimates that are representative of CDD/CDF levels in all parts of Europe or the world, but are used to depict similarities or differences between U.S. levels and those observed by researchers in other parts of the world.

Media levels discussed in this chapter that represent background conditions in the United States are used in Chapter 4 to estimate background exposures to dioxin-like compounds. For the purposes of this document, background is defined as the level of dioxin-like compounds in samples of environmental media originating from sites not known to be impacted by point source releases. For soil and air, background concentrations of CDD/CDFs were calculated for both rural and urban background locations. However, urban background concentrations were used in calculating human exposures to CDD/CDFs because a large percentage of the population resides in urban environments. Also, it should be noted that background levels in environmental media are represented by mean concentrations of multiple background samples. Because mean values are used, it is likely that some background sites may have concentrations that are less than the mean

concentration, while others may have higher levels. It should be noted that background concentrations were calculated based on the best available, current (i.e., late 1980s in a few cases, but primarily from the 1990s) studies. For most media, background concentrations used to assess exposure in Chapter 4 were calculated by setting nondetectable concentrations to one-half the detection limit. For some media (i.e., soil, vegetable oil), however, nondetects were set to zero because detection limits were unavailable for some studies or the detection limits were too high. These instances are noted in the text.

Studies used for development of background concentrations were also chosen to be representative of nationwide exposures. In general, data were selected that represent typical exposure conditions, so that these levels could be combined with typical ingestion/contact values to estimate background exposures (see Chapter 4). However, the data and strategy used for estimating background media levels varied somewhat, depending on the media:

- **Air and Soil** - Urban data were used to derive background estimates because most people are exposed to these levels. No data were included that were collected near known uncommon point sources (i.e., large incinerators, cement kilns, smelters, etc.). Vehicles, fireplaces, home heating furnaces, etc. are all recognized potential point sources, but are so ubiquitous as to be considered a normal part of the urban background. Estimates are also presented for rural areas, and these are more relevant for evaluating impacts in rural areas. Also a degree of uncertainty is expected from the air and soil concentration estimates due to a lack of geographic coverage and non-uniform study design.

- **Water and Sediment** - No data were included that were collected near known uncommon point sources (pulp and paper mills, POTWs, etc.). No distinction was made between urban and rural sites. For water, the data for treated drinking water were very limited (i.e., based on octa-chlorinated compounds only). Sediment data were collected from non-impacted lakes.

- **Fish** - No data were included that were collected near known uncommon point sources (pulp and paper mills, POTWs, etc.). Background data for freshwater fish were based on a national sampling strategy, while background data for marine fish and shellfish were based on limited data.

- **Food** - Only samples from national EPA/USDA surveys and grocery stores (e.g., eggs) were used. National EPA/USDA surveys used statistically-based sampling methods to collect samples representative of the national food supply. Grocery store samples represent the most common source of foods.

## 3.2. CONCENTRATIONS IN AIR

Tables B-1 through B-3 (Appendix B) contain summaries of data from studies of ambient air measurements of CDDs, CDFs, and PCBs in the United States and Europe. Environmental levels of PCBs in North American air are based on a single source of information (Hoff et al., 1992). Relatively few studies have been conducted to measure ambient air levels of CDDs/CDFs. This may be, in part, because of the low analytical detection limits required to detect the expected low concentrations of specific CDD/CDF congeners and the relatively large volumes of air (e.g., 350 to 450 cubic meters of ambient air over a 24-hour period) required to obtain subparts-per-trillion levels of analytical detection. These low detection limits in ambient air samples were not achieved until the mid 1980s. The results of several of these more recent ambient air studies are summarized in the following paragraphs. It should be noted, however, that these studies lack geographic coverage and may not be representative of the nation as a whole. Currently, EPA is establishing a network of stations equipped with high-volume air samplers capable of detecting concentrations of dioxin and dioxin-like compounds as low as 0.1 parts per trillion. The network, known as the National Dioxin Air Monitoring Network (NDAMN), will provide information on background concentrations of dioxin-like compounds, as well as data for use in tracking long-range transport of dioxin and calibrating atmospheric models. The sampling sites included in the network were selected with the intent of covering a wide geographic region, with special attention to rural, agricultural areas. Many of the sites are shared with the National Atmospheric Deposition Program (NADP), which is a collaborative effort involving dozens of public and private research and educational institutions. The NDAMN project calls for a total of 29 sites in 24 states. Data from these sites will eventually be used to update the background air concentration data presented in this chapter.

It should also be noted that this chapter focuses on the concentrations of CDD/CDF/PCBs in outdoor air. Data on indoor air concentrations are extremely limited. PCB data for one recent indoor air study in the United Kingdom are presented. No background CDD/CDF data for indoor environments were available.

### 3.2.1. U.S. Data

An extensive ambient air monitoring study of CDD/CDFs was conducted as part of a multiyear monitoring effort at eight sampling locations in the Southern California area by the Research Division of the California Air Resources Board from December 1987 through March 1989 (Hunt et al., 1990). The monitoring network "included a number of sites situated in primarily residential areas (San Bernadino, El Toro, and Reseda), as well as several sites in the vicinity of suspected sources of CDD/CDFs (Cal. Trans, Commerce, North Long Beach, and West Long Beach) (Hunt et al., 1990)." The seven sites mentioned above were classified as urban locations by the definitions used in this document, while the eighth site was classified as an industrial site (i.e., Carson--onsite at manufacturer of gas cooking equipment). Additionally, four of the eight sites were part of the South Coast Air Quality Management District (SCAQMD) monitoring network. All totaled, there were nine sample collection intervals throughout this study. "Typically, five to seven stations were in contemporaneous operation during a particular session" (i.e., samples were not collected from each location at each interval). Total tetra- through octa- chlorinated CDDs and CDFs were screened for in the study as well as various 2,3,7,8-substituted CDD and CDF congeners. A total of 34 analyses were performed throughout the study for all congeners except for OCDD and OCDF, respectively, for which only 31 analyses were performed. Samples were collected over a maximum of seven intervals at each site throughout the study (i.e., Reseda and El Toro--six dates, duplicate samples on one date), while a sample was collected from the Commerce site during only a single collection interval. Sample collection intervals generally averaged 24 hours (Hunt et al., 1990).

Generally, higher substituted CDD and CDF congeners accounted for the majority of positive samples containing quantifiable CDD/CDF residues in this study (i.e., total HxCDD/HxCDF and above). In fact, over 90 percent of the samples collected contained quantifiable levels of 1,2,3,4,6,7,8-HpCDD, total HpCDD, and OCDD. Additionally, approximately 50 to 70 percent of the samples collected contained quantifiable levels of total HxCDD; 2,3,7,8-TCDF; total TCDF; total PeCDF; total HxCDF; 1,2,3,4,6,7,8-HpCDF; total HpCDF; and OCDF. For all other congeners, quantifiable residues were detected in less than 25 percent of the samples collected. All CDD congener concentrations ranged from nonquantifiable levels (low limit of 0.0026 pg/m$^3$) to an upper limit of 18.0 pg/m$^3$.

Additionally, CDF congener levels ranged from nonquantifiable levels (low limit of 0.0040 pg/m$^3$) to an upper limit of 2.70 pg/m$^3$.

According to Hunt et al. (1990), "The highest concentration of CDDs/CDFs congener class sums ($Cl_4$-$Cl_8$) and 2,3,7,8-substituted species were noted during a period predominated by off-shore air flows in December 1987, suggesting a regional air mass and transport phenomena. Concentrations of the CDDs/CDFs were diminished markedly in subsequent sessions where air flow patterns were primarily off-shore or of coastal origin." Hunt et al. (1990) indicated that the "CDD/CDF congener profiles ($Cl_4$-$Cl_8$) and 2,3,7,8-substituted isomeric patterns strongly suggest combustion source influences in the majority" of the samples collected.

Smith et al. (1989) quantified CDD/CDFs in air samples collected from two locations in Niagara Falls, New York, over a 6-month period in 1986/87. One site was located upwind of a large industrial complex (i.e., background), and the other site was located downwind of the complex (i.e., industrial). OCDD concentrations at the downwind location were more variable, but consistently higher, than at the upwind location. The maximum OCDD concentration observed at the downwind site was 8.8 pg/m$^3$. Total I-$TEQ_{DF}$s for the two locations were estimated to be 0.038 pg/m$^3$ ($TEQ_{DF}$-$WHO_{98}$ = 0.041 pg/m$^3$) (n = 3) for the upwind (i.e., background) site and 0.84 pg/m$^3$ ($TEQ_{DF}$-$WHO_{98}$ = 0.92 pg/m$^3$) (n = 3) for the downwind site, using one-half the detection limit to represent nondetects (Table 3-1). In another study, Smith et al. (1990a) analyzed ambient air samples from several other New York cities for CDD/CDFs. Samples were collected in Albany (n = 3), Binghamton (n = 1), and Utica (n = 2). Total CDD/CDF concentrations ranged from 3.02 pg/m$^3$ to 13.1 pg/m$^3$. None of the samples had detectable levels of 2,3,7,8-TCDD, but 2,3,7,8-TCDF was detected at concentrations ranging from 0.18 pg/m$^3$ to 1.24 pg/m$^3$.

Maisel and Hunt (1990) reported on ambient air concentrations of CDD/CDFs in Los Angeles, California, during the winter of 1987. Concentrations were highest for OCDD, and the estimated I-$TEQ_{DF}$ concentration was 0.12 pg/m$^3$ ($TEQ_{DF}$-$WHO_{98}$ = 0.13 pg/m$^3$, assuming that nondetects equal one-half the detection limits. Edgerton et al. (1989) measured CDD/CDFs in air samples from Ohio during 1987 to evaluate the impact of potential CDD/CDF sources on ambient levels in air. Samples were collected from various locations, including those at an industrial site, near a municipal refuse-derived fuel power plant, near a sewage sludge incinerator, downwind of a municipal incinerator, at a high

traffic density site, and at a rural background site in Waldo, Ohio. Total CDD/CDF concentrations were found to be higher in samples collected near incinerators than at the background site. None of the samples had detectable levels of 2,3,7,8-TCDD, and the hepta- and octachlorinated CDD/CDFs were the most abundant. Using congener profiles developed for several source categories, Edgerton et al. (1989) compared the CDD/CDF patterns in ambient air from this study to the profiles for each source and found that the profile for the background sample was "almost identical to the profile constructed for municipal incinerators."

The Ohio Environmental Protection Agency, Division of Air Pollution Control, conducted an ambient air monitoring study in 1994/1995 for CDD and CDF compounds in the vicinity of the Columbus Waste to Energy (WTE) facility. The purpose of the study was to evaluate the impact of the facility on air quality. The Columbus WTE was a major source of dioxins to the Columbus environment. Based on a 1992 stack emission test, the Ohio EPA estimated that annual emissions from the facility exceeded 900 g I-$TEQ_{DF}$/yr (OEPA, 1994a). The sampling in 1994 occurred while the facility was operating. The facility ceased operation in December 1994; therefore, the 1995 sampling did not include impacts from the facility. A total of seven sites were sampled; six were located in the urban area of Columbus, within 1-2 miles of the facility, and mostly in the historically predominant downwind direction, and the seventh was located 28 miles away in the upwind direction in a rural background setting. Five urban samples were taken in both March and April 1994 (one of the six samplers was not operating on each of these sampling dates), and six urban samples were taken in June 1995, for a total of 16 urban samples. One rural sample was taken on each sampling date for a total of three background samples. All samples were collected over a 48-hour sampling period using modified high-volume air samplers. Further details on these studies can be found in OEPA (1994b, 1995).

Table 3-2 presents the mean concentrations of congeners and homologue groups from four groups of air samples:

1. The "impacted air" samples include one sample taken in each of the March and April 1994 sampling periods. Wind rose data in OEPA (1994b) show

that the samplers from which these samples came from were downwind of the Columbus WTE during the 48-hour sample.

2. The 1994 urban samples include the eight other samples taken in 1994 while the Columbus WTE was still operating.
3. The 1995 urban samples include the six samples taken in the urban setting once the incinerator was no longer operating.
4. The three rural samples include those taken at a site 28 miles away in the historical upwind direction from the Columbus WTE.

OEPA (1994b) also notes that the highest air concentrations were found in the downwind samples.

Generally, total CDD/CDF concentrations were higher for the urban sites than for the background sites. Overall, the average total urban background air CDD/CDF concentration was 3.5 pg/m$^3$, and the I-TEQ$_{DF}$ and TEQ$_{DF}$-WHO$_{98}$ concentrations were 0.050 pg/m$^3$ and 0.055 pg/m$^3$, respectively. These values were the average of the 1994 and 1995 urban samples. The rural background total CDD/CDF concentration was 2.2 pg/m$^3$, and the I-TEQ$_{DF}$ and TEQ$_{DF}$-WHO$_{98}$ concentrations were 0.022 pg/m$^3$ and 0.024 pg/m$^3$, respectively. The impact of the Columbus facility can be seen by examining the impacted air samples. As noted above, one sample in each of the two 1994 sampling dates was downwind of the Columbus facility. In fact, the air sampler was the same in both cases and was located about 1.5 miles in the easterly direction. The air concentration in this sampler was the highest in both these sampling dates, equaling 9.2 pg/m$^3$ total and 0.17 pg I-TEQ$_{DF}$/m$^3$ in March and 19.0 pg/m$^3$ total and 0.35 pg I-TEQ$_{DF}$/m$^3$ in April. The average total CDD/CDF concentration for these impacted samples was 14 pg/m$^3$, and the I-TEQ$_{DF}$ and TEQ$_{DF}$-WHO$_{98}$ concentrations were 0.26 pg/m$^3$ and 0.29 pg/m$^3$, respectively. Further analysis of these Columbus air data can be found in Lorber et al. (1998a).

In accordance to the Connecticut Ambient Air Quality standards, the State of Connecticut's Department of Environmental Protection (CDEP) implemented a monitoring program which measured CDD/CDFs in ambient air at six sites in Connecticut (CDEP, 1995). The air monitoring program was conducted from fall 1993 through summer 1994 in the vicinity of five Resource Recovery Facilities (RRFs), located in Bridgeport, Hartford (mid-Connecticut), Bristol, Preston, and Wallingford, Connecticut, as well as one background

rural site located at Mohawk Mountain. The monitoring activity involved four quarterly 1-month sampling periods (CDEP, 1995). Ambient concentrations measured for the four quarterly monitoring sessions at the rural background site (Mohawk Mountain) are presented in Table 3-3. Based on the CDD congeners, OCDD had the highest background concentration (0.451, 0.196, 0.155, and 0.056 pg/m$^3$) for all four sampling periods. The total I-TEQ$_{DF}$s for these rural background samples were 0.015 pg/m$^3$, 0.009 pg/m$^3$, 0.006 pg/m$^3$, and 0.005 pg/m$^3$ for the November 1993, February 1994, May 1994, and August 1994 sampling periods, respectively. The average I-TEQ$_{DF}$ for these sampling periods was 0.0087 pg/m$^3$ (TEQ$_{DF}$-WHO$_{98}$ = 0.010). The average I-TEQ$_{DF}$ and TEQ$_{DF}$-WHO$_{98}$ values for the five urban locations, over the four sampling periods, were 0.026 pg/m$^3$ and 0.029 pg/m$^3$, respectively. Data for all samples collected during the monitoring program show that the I-TEQ$_{DF}$ concentrations were below the 1.0 annualized pg/m$^3$ ambient standard adopted by the State of Connecticut for CDD/CDFs (CDEP, 1995).

In an earlier study of ambient air monitoring in the vicinity of Wallingford, Connecticut, CDEP (1988) reported on CDD/CDF levels in 28 ambient air samples collected in 1988. The mean total I-TEQ$_{DF}$ for these samples was 0.05 pg/m$^3$, when nondetects were set to one-half the detection limit. Hunt and Maisel (1990) conducted pre-operational air monitoring in the vicinity of the site of a municipal solid waste incinerator. I-TEQ$_{DF}$ concentrations, averaged over seven sites and all seasons, were 0.1 pg/m$^3$, when nondetects set to one-half the detection limit. Mean concentrations were highest for the higher-chlorinated dioxin homologue groups and lower furan homologue groups.

In a long-term study of CDD/CDFs in the ambient air around Bloomington, Indiana, methods were developed for measuring individual CDD/CDFs at concentrations as low as 0.001 pg/m$^3$ (Eitzer and Hites, 1989). Total CDD/CDF concentrations were 0.480 pg/m$^3$ and 1.360 pg/m$^3$ for the vapor phase and the particle-bound phase, respectively. For individual congeners, CDFs were found to decrease in concentration with increasing levels of chlorination, and CDD concentrations were found to increase with increasing levels of chlorination (Eitzer and Hites, 1989).

Fiedler et al. (1995a) conducted a sampling and monitoring program in rural Mississippi using pine needles as indicators of the presence of CDDs and CDFs in the atmosphere. Pine needles have been shown to be passive samplers for lipophilic substances present in the air, because their outer waxy surface absorbs these atmospheric

pollutants. Pine needle samples were collected from eight locations in southern Mississippi. CDD and CDF I-$TEQ_{DF}$ concentrations ranged from 0.11 to 0.23 pg/kg dry mass for 1994 shoots and 0.07 to 0.51 pg/kg dry mass for 1993 shoots. The authors concluded that the data suggest that the atmospheric concentrations of CDDs and CDFs in rural Mississippi are relatively low, but higher concentrations were observed at more populated sites.

In a subsequent study, Fiedler et al. (1997a) analyzed CDD/CDF levels in ambient air in a rural area in southern Mississippi using three sampling methodologies: high-volume ambient air sampling to measure direct atmospheric levels of CDD/CDFs; Bergerhoff samplers to collect atmospheric deposition samples; and pine needles as passive samplers. The study was conducted from December 1995 to January 1996 and from June to July 1996 to assess the concentrations of CDD/CDFs during these time periods. The sampling location had no known local sources of CDD/CDFs. In general, winter CDD/CDF concentrations were higher than summer concentrations. CDD/CDF concentrations measured using high-volume air samples averaged 1.126 pg/$m^3$ in the winter and 0.36 pg/$m^3$ in the summer (i.e., three times higher in winter than in summer). The mean I-$TEQ_{DF}$ concentrations were 0.0109 pg I-$TEQ_{DF}$/$m^3$ in the winter and 0.0037 pg I-$TEQ_{DF}$/$m^3$ in the summer, when using one half the limit of quantification for nonquantifiable congeners. The results of the deposition study suggested that deposition is also higher in winter than in summer. The mean CDD/CDF deposition rate was 152 pg/$m^3$-day in winter and 108 pg/$m^3$-day in summer, when nonquantifiable congeners were set to one-half the limit of quantification. Normalized to pg I-$TEQ_{DF}$/$m^3$-day, the deposition rate was 2.6 pg I-$TEQ_{DF}$/$m^3$-day in winter and 0.58 pg I-$TEQ_{DF}$/$m^3$-day in summer. Analysis of the pine needles showed that CDD/CDF concentrations increase with increasing exposure times. Concentrations in pine needles ranged from 10 to 54 pg/g d.m. (0.16 to 0.79 pg I-$TEQ_{DF}$/g d.m.), when nonquantifiable congeners were set to one-half the limit of quantification. The authors concluded that the CDD/CDF concentrations observed in ambient air in this study (0.0023 to 0.017 pg I-$TEQ_{DF}$/$m^3$) were lower than similar remote locations in Germany (0.015 to 0.020 pg I-$TEQ_{DF}$/$m^3$). Deposition rates were also lower in southern Mississippi (0.42 to 3.1 pg I-$TEQ_{DF}$/($m^2$-day)) than in rural areas of Germany (5 to 7 pg I-$TEQ_{DF}$/($m^2$-day)).

Hunt et al. (1997) conducted a study in Phoenix, Arizona, aimed at assessing the influence of motor vehicle emissions on ambient air concentrations of CDD/CDFs. Four sets

of 24-hour integrated samples were collected between December 15 and 20, 1994. The sampling site was located near a heavily traveled roadway in metropolitan Phoenix. The month of December was chosen for sampling, because inversion conditions are expected during the winter months. CDD/CDFs were detected in all four sample sets. Average total I-$TEQ_{DF}$ values varied from 0.092 pg/$m^3$ (December 15) and 0.094 pg/$m^3$ (December 19) to 0.37 pg/$m^3$ (December 16) and a high of 0.45 pg/$m^3$ (December 20). Average congener-specific I-$TEQ_{DF}$ (and $TEQ_{DF}$-$WHO_{98}$) values are shown in Table 3-4. The average total I-$TEQ_{DF}$ value of 0.25 pg/$m^3$ and $TEQ_{DF}$-$WHO_{98}$ values of 0.27 pg/$m^3$ are higher than the TEQ data reported for other U.S. urban locations, such as Los Angeles and Connecticut (Maisel and Hunt, 1990). The first 2 sampling days of this study demonstrated congener class profiles typical of those reported in the literature for urban U.S. settings, showing a predominance of CDDs over CDFs. Hunt et al. (1997) also noted that the "predominance of 1,2,3,4,6,7,8-HpCDD as the most persistent 2,3,7,8-substituted CDD congener is consistent with the observations of others in the open literature, and is prevalent at sites known to be influenced by stationary or mobile combustion source emissions." Data from the last 2 sampling days of this study produced distinctly different congener profiles from the first 2 days. The last 2 sample dates' results showed a predominance of CDFs over CDDs. On December 20, for example, 1,2,3,4,6,7,8-HpCDF was present at a level of 2.16 pg/$m^3$, while the typically most predominant isomer 1,2,3,7,8-HpCDD had a value of 1.30 pg/$m^3$. Hunt et al. (1997) stated that further study "beyond examination of CDDs/CDFs data alone is warranted to provide a more conclusive source determination." Data from this study are not included in the ambient background level determinations in this chapter due to recommendations of the authors that "due to site-specific bias likely introduced by vehicular traffic at the Indian School Road site, the ambient CDDs/CDFs measured should not be construed to be representative of ambient CDDs/CDFs burdens in metropolitan Phoenix, as a whole."

Beginning in September 1996, a Canadian survey of CDD/CDFs in air was conducted at locations across Canada (Belzer et al., 1998). Some of the samples were collected near a coastal pulp mill operation at a location 1 kilometer southeast of a mill area. Data analysis indicated that CDD/CDFs concentrations from the mill area ranged from 0.006 to 0.067 pg I-$TEQ_{DF}$/$m^3$. These values were similar to those observed in other Canadian urban sites (0.01 to 0.08 pg I-$TEQ_{DF}$/$m^3$), but were lower than those measured near industrial

point sources (0.01 - 0.4 pg I-$TEQ_{DF}$/$m^3$). The total CDD and CDF congener values for the mill area ranged from 0.17-3.94 pg/$m^3$, and the values of total homologue groups ranged from 0.77-5.53 pg/$m^3$. The results of this study suggest that production of CDD/CDFs from combustion sources is highly dependent on combustion material, temperature and moisture values (Belzer at al., 1998).

PCBs in ambient Canadian air were evaluated by Hoff et al. (1992). A total of 143 air samples from Egbert, Ontario, Canada, taken in 1988 and 1989 were analyzed for various vapor-phase PCB congeners. The annual mean concentrations for these samples are presented in Table 3-5. These means were calculated by assuming that nondetectable concentrations were zero. Based on the mean concentrations for the limited set of toxic PCB congeners in Hoff et al. (1992) (i.e., PCBs 105, 114, 118, 156, 170, 180, and 189), the total $TEQ_P$-$WHO_{94}$ concentration is estimated to be 0.00094 pg/$m^3$ ($TEQ_P$-$WHO_{98}$ = 0.00088 pg/$m^3$).

### 3.2.2. European Data

Clayton et al. (1993) conducted a study of CDDs and CDFs in the ambient air of three major cities (London, Manchester, and Cardiff) and an industrial town (Stevenage) in the United Kingdom. Annual median I-$TEQ_{DF}$ concentrations of CDDs and CDFs ranged from 0.04 to 0.10 pg/$m^3$. Hepta- and octachlorinated dioxin congeners contributed the most to the total concentration of 2,3,7,8-substituted CDD/CDFs, and a large number of nondetect values were reported for the tetra-, penta-, and hexachlorinated dioxins. Congeners that contributed most to the total I-$TEQ_{DF}$ concentrations were 2,3,7,8-TCDF; 1,2,3,4,7,8-; 1,2,3,6,7,8-; and 2,3,4,6,7,8-HxCDF. The United Kingdom's Department of the Environment, Transport and the Regions has posted air monitoring data from 1991 through 1993 on the Internet (Department of the Environment, Transport and the Regions, 1998). Sampling locations include the same UK cities monitored by Clayton et al. (1993), as well as an urban site in Middlesbrough and a rural site at Hazelrigg. Mean CDD/CDF I-$TEQ_{DF}$ concentrations from the quarterly sampling periods ranged from 0.15 to 0.34 pg/$m^3$ for Cardiff (8 quarters monitored), 0.10 to 0.34 pg/$m^3$ for Stevenage (5 quarters), 0.012 to 0.33 pg/$m^3$ for Middlesbrough (7 quarters), 0.044 to 1.4 pg/$m^3$ for Manchester (12 quarters), 0.016 to 0.28 pg/$m^3$ for London (12 quarters), and 0.004 to 0.21 pg/$m^3$ for Hazelrigg (12 quarters), when nondetects were set at the detection limit. When nondetects

were set to zero, mean I-$TEQ_{DF}$ concentrations ranged from 0.063 to 0.30 pg/$m^3$ for Cardiff, 0.034 to 0.30 pg/$m^3$ for Stevenage, < 0.001 to 0.21 pg/$m^3$ for Middlesbrough, 0.036 to 0.69 pg/$m^3$ for Manchester, 0.008 to 0.24 pg/$m^3$ for London, and 0.003 to 0.17 pg/$m^3$ for Hazelrigg. The sum of the quarterly mean PCB concentrations of PCBs 2, 52, 101, 118, 138, 153, and 180, for these same cities, ranged from 164 to 985 pg/$m^3$ for Cardiff (10 quarters monitored), 189 to 395 pg/$m^3$ for Stevenage (5 quarters), 78 to 359 pg/$m^3$ for Middlesbrough (9 quarters), 181 to 704 pg/$m^3$ for Manchester (14 quarters), 651 to 2,482 pg/$m^3$ for London (14 quarters), and 77 to 198 pg/$m^3$ for Hazelrigg (8 quarters). These CDD/CDF values are relatively consistent with the concentrations in ambient German air observed by Liebl et al. (1993) and König et al. (1993a). Liebl et al. (1993) analyzed ambient air samples collected from 10 sites in Hessen, Germany, from 1990 through 1992. Concentrations ranged from 0.04 to 0.15 pg I-$TEQ_{DF}$/$m^3$. The higher concentrations were presumed to result from direct local industrial sources. König et al. (1993a) collected air samples from six sites in Hessen, Germany. CDD/CDF concentrations ranged from 0.048 pg I-$TEQ_{DF}$/$m^3$ at a rural reference site to 0.146 pg I-$TEQ_{DF}$/$m^3$ at an industrial site. The results of the study also indicated that concentrations of CDDs and CDFs are typically higher in the winter than in the summer. Sugita et al. (1993) also observed higher concentrations of CDDs and CDFs in winter than in summer in an ambient air study in urban Japan. The average concentration of CDDs and CDFs was 0.788 pg I-$TEQ_{DF}$/$m^3$ in the summer and 1.464 pg I-$TEQ_{DF}$/$m^3$ in winter.

Fiedler et al. (1997b) conducted a 3-year air monitoring study that examined the CDD/CDF levels near two municipal solid waste incinerators (MSWI) located in Bavaria, Germany (Augsburg and Burgkirchen). The authors observed that the I-$TEQ_{DF}$ concentrations at these locations were comparable and concentrations were consistently higher during winter months that in summer months. For example, at Augsburg, the lowest concentration (0.009 pg I-$TEQ_{DF}$/$m^3$) was obtained during the summer of 1995, and the highest concentration (0.206 pg I-$TEQ_{DF}$/$m^3$) was observed in the winter of 1994/1995. Background concentrations for these time periods ranged from 0.0076 to 01.29 pg I-$TEQ_{DF}$/$m^3$. For Burgkirchen, the lowest concentration (0.0044 pg I-$TEQ_{DF}$/$m^3$) was observed during the summer of 1994, and the highest concentration (0.078 pg I-$TEQ_{DF}$/$m^3$) was observed during the winter of 1995/1996.

In a Swedish study, air samples were collected from a city center, suburb, remote countryside, and open coastal area (Broman et al., 1991). Analyses of the samples for dioxins and furans indicated that the concentrations of these compounds decreased with increasing distance from the city center. Total CDD/CDF concentrations were 1.40 pg/m$^3$, 1.10 pg/m$^3$, 0.40 pg/m$^3$, and 0.22 pg/m$^3$ for the city center, suburb, countryside, and open coastal areas, respectively. Similar patterns of decreasing concentrations with increasing distances from urban areas were also observed for individual CDD/CDF congeners (Broman et al., 1991). In a study of ambient air concentrations of CDDs and CDFs in Flanders, samples were collected and analyzed at rural, industrial, and urban sites (Wevers et al., 1993). Average ambient air concentrations ranged from 0.0696 pg I-TEQ$_{DF}$/m$^3$ at a rural site to 0.254 pg I-TEQ$_{DF}$/m$^3$ at a site believed to be influenced by a chemical industry and a highway. Naf et al. (1990) analyzed urban air samples from a site near a wastewater treatment plant in Sweden. The samples were collected as part of a study to estimate the flux of CDD/CDFs through the treatment plant. All 2,3,7,8-substituted congeners (except 1,2,3,7,8,9-HxCDD and 1,2,3,4,7,8,9-HpCDF) were detected. The mean I-TEQ$_{DF}$ for urban air was estimated to be 0.02 pg/m$^3$.

Hiester et al. (1995) observed a decrease of CDD/CDF concentrations in Germany's ambient air over a 6-year period. Ambient air samples were collected over 12 sampling intervals from 4 sites in the heavily industrialized Rhine-Ruhr region of Germany during 1987/88 and 1993/94. Total I-TEQ$_{DF}$s for these sites ranged from 0.13 pg/m$^3$ to 0.33 pg/m$^3$ during 1987/88 and from 0.04 pg/m$^3$ to 0.12 pg/m$^3$ for the 1993/94 time period. Reductions in I-TEQ$_{DF}$s ranged from 46 to 69 percent at these sites over the 6-year period (i.e, from 0.22 pg/m$^3$ to 0.13 pg/m$^3$ at Dortmund and from 0.13 pg/m$^3$ to 0.04 pg/m$^3$ at Köln). These reductions were attributed to abatement actions taken since 1989 (Hiester et al. 1995).

Between November 1992 and October 1993, the Austrian Federal Environmental Agency monitored six air stations for ambient air concentrations of CDD/CDFs (Umweltbundesamt, 1994; 1996). One hundred samples were taken from industrial and population sites; three sites in Vienna, one in Steyregg, one in Linz, and one in Graz. The arithmetic annual average value of ambient levels for all samples ranged from 0.03 to 0.12 pg I-TEQ$_{DF}$/Nm$^3$. Average winter levels ranged from 0.05 to 0.24 pg I-TEQ$_{DF}$/Nm$^3$; while the summer levels ranged from 0.02 to 0.04 pg I-TEQ$_{DF}$/Nm$^3$. Winter levels observed in Graz

were two times the winter concentrations found at the other sampling sites. Levels were consistently highest (i.e., two to three times higher) for the measuring period between February 1 and 4 at all the measuring sites. This time period coincided with an extremely stable meteorological condition or inversion. All locations demonstrated a decrease in the proportion of CDFs from the tetra- to octachlorinated congeners, while the opposite was true for CDDs. However, there were differences in the congener profiles for different locations. For example, the Graz location showed a higher proportion of octa- and heptachlorinated dioxins, while tetrachlorofurans predominated at the hospital site in Vienna and also at the Steyregg and Linz sites.

Samples of ambient air were collected from 15 locations throughout Slovakia, including urban, industrial, agricultural and rural sites, between October 1996 and August 1997 (Stenhouse et al., 1998). A total of 113 samples were analyzed for CDD/CDFs. The average ambient I-$TEQ_{DF}$ concentrations for these locations ranged from 0.05 pg/$m^3$ to 0.13 pg/$m^3$ at urban/industrial areas (with an average of 0.1 pg/$m^3$), 0.07 pg/$m^3$ for agricultural areas and 0.04 pg/$m^3$ for rural background. The values of any congeners below the detection limit (0.01 pg/$m^3$) were included in the I-$TEQ_{DF}$ at the detection limit. Higher ambient I-$TEQ_{DF}$ values were observed in the winter than in the summer at the places where the major source was combustion. Stenhouse et al. (1998) suggested that seasonal variation would not be expected if industrial processes and traffic were significant contributors.

PCBs have also been evaluated in European air samples (Halsall and Jones, 1993; König et al., 1993b). Halsall and Jones (1993) monitored urban air at two sites in the United Kingdom. The annual mean total PCB concentrations were 520 and 590 pg/$m^3$. PCBs existed in ambient air predominantly in the vapor phase. This study also indicated that summer PCB concentrations were higher than winter concentrations. These researchers attributed the differences in seasonal patterns to volatilization from soil during summer months. Ambient air concentrations of PCBs in Hessen, Germany, ranged from 350 to 1630 pg/$m^3$ during the period of 1990 to 1992 (König et al., 1993b). Urban areas characterized by industry and/or heavy traffic had the highest PCB concentrations in ambient air. Hiester et al. (1995) also evaluated total PCB concentrations in ambient air samples from several sites in Germany during 1993/94. Table 3-6 presents the annual average dioxin-like PCB concentrations in ambient air of several German cities. Annual

mean total PCB concentrations ranged from 1,000 pg/m$^3$ to 2,000 pg/m$^3$ in urban locations and from 100 pg/m$^3$ to 300 pg/m$^3$ in rural locations.

Recently, Currado and Harrad (1997) measured indoor air concentrations of PCBs from nine different indoor environments, including two laboratories, two offices, and five residential homes in the United Kingdom. The results indicated that the total PCB levels found in indoor air (1.4 to 19.1 ng/m$^3$; mean 7.1 ng/m$^3$) were between 2 and 19 times higher than the levels in outdoor air (0.77 to 0.87 ng/m$^3$; mean 0.82 ng/m$^3$). It should be noted that the study did not focus on dioxin-like PCBs; only concentrations of four dioxin-like PCB congeners were reported in indoor and outdoor areas. Thus, TEQ concentrations in indoor air were not calculated. Studies that examine background CDD/CDF levels in indoor environments were not available.

### 3.2.3. Air Observations and Trends

Some general observations for CDD/CDF levels in air are possible from the various air studies discussed in this chapter:

- Concentrations in urban settings are higher than those in rural settings.
- Concentrations associated with source impacted areas are the highest.
- As the degree of chlorination increases, so does the congener concentration.
- Based on the limited ambient air measurements made in selected cities in the United States and Europe, there appears to be good agreement with respect to the magnitude of specific congeners of CDDs and CDFs in urbanized areas in the United States and Europe.
- Many of the air measurements tend to be very close to the current analytical detection limit. This increases the probability that congeners indicated as not detected (ND) may actually be present.

### 3.2.4. Air CDD/CDF Profiles and Background TEQ Concentrations

CDD/CDF profiles were calculated for rural and urban air. Rural air profiles used data from OEPA (1995) and CDEP (1988). Urban air profiles used data from CDEP (1988, 1995), Smith et al. (1989, 1990a), Maisel and Hunt (1990), Hunt et al. (1990), and OEPA (1995). CDD/CDF homologue group and 2,3,7,8-substituted congener profiles for air are presented in Table 3-7 and Figures 3-1 and 3-2. The CDD/CDF homologue profile was

calculated by dividing individual homologue group concentrations by the total CDD/CDF concentration. This profile indicates that OCDD is the predominant homologue group in rural and urban background air followed by HpCDD. TCDD accounts for the lowest percentage of total CDD/CDFs. Congener group profiles were calculated as the ratio of individual 2,3,7,8-substituted congener concentrations to total CDD/CDF concentration (i.e., the sum of homologue group concentrations). The concentration of 2,3,7,8-substituted congeners accounts for 57 percent of the total CDD/CDF concentration in urban background air. In rural background air, 2,3,7,8-substituted CDD/CDFs account for 58 percent of the total CDD/CDF concentration. Of the 2,3,7,8-substituted CDD/CDF congeners, OCDD accounts for the highest percentage (i.e., 38 percent rural; 34 percent urban) of total CDD/CDFs, followed by the 1,2,3,4,6,7,8-HpCDD in rural and urban background air.

Table 3-8 presents a summary of the $TEQ_{DF}$-$WHO_{98}$ concentrations of CDD/CDFs in the United States. Assuming that nondetects are equal to one-half the detection limit, the mean $TEQ_{DF}$-$WHO_{98}$ concentration was 0.017 pg/m$^3$ for rural background sites (i.e., sites in Connecticut and Ohio) (n = 7; CDEP, 1995; OEPA, 1995), and 0.12 pg/m$^3$ for urban background sites (i.e., from 14 sites in Connecticut, California, Ohio, and New York) (n = 106; CDEP, 1988; CDEP, 1995; Hunt and Maisel, 1990; Maisel and Hunt, 1990; OEPA, 1995; Smith et al., 1989; Smith et al., 1990a). These mean concentrations represent the average of the mean concentrations for the various sites and not the mean of all individual samples. (See weighted mean in Table 3-8.) The mean value is used to ensure that each site is weighted equally (i.e., heavily sampled sites do not have any greater impact on the mean than sites with fewer samples). Samples collected from urban locations not expected to be impacted by industrial point sources were assumed to represent "background" conditions for the majority of the U.S. population. The "typical" urban background $TEQ_{DF}$-$WHO_{98}$ level was estimated to be 0.12 pg/m$^3$ based on the mean of the background samples collected in urban environments. (The I-$TEQ_{DF}$ value for these sites is 0.11 pg/m$^3$). This value was used in Chapter 4 to characterize background exposures.

Based on the results of European studies, ambient air concentrations of CDDs and CDFs appear to be similar to those found in the United States. Based on the midpoints of the European studies for which I-$TEQ_{DF}$ concentrations were reported (Clayton et al. 1993;

Liebl et al. 1993; König et al. 1993a; Wevers et al. 1993), the I-$TEQ_{DF}$ air concentration for Europe is 0.11 pg/$m^3$. Data for these European studies are not included in Tables B-1 and B-2 of Appendix B because individual congener data were not reported.

It is interesting to compare these background air values with the CDD/CDF concentrations in air measured by Lugar (1993) in and around McMurdo Station, Antarctica, a logistics and staging facility with a population of about 1,100. Four locations were sampled: a site upwind of the station, downwind of the station, in the center of the station, and a remote unpopulated island 30 kilometers distant from the station. CDD/CDFs were not detected in the samples from the upwind site (congener detection limits ranged from $<$0.01 to 0.03 pg/$m^3$), and few CDD/CDF congeners were detected at the remote island sites (congener detection limits ranged from 0.001 to 0.008 pg/$m^3$). CDD/CDFs were detected only sporadically at the downwind site (some congeners detected in three of five samples) and in all five samples collected from the station center site (mean I-$TEQ_{DF}$ concentration of 0.0153 pg/$m^3$). Similar results were obtained in a follow-up study during the austral summers 1992/93 and 1993/94. A total of 28 air samples were collected from these four sites (Lugar et al., 1996). CDD/CDFs were not detected at the upwind or remote island sites and trace levels of only a few CDD/CDFs were found in the downwind site. The highest CDD/CDF concentrations were observed at the downtown site, where CDD concentrations ranged from 0.12 to 1.80 pg/$m^3$, and CDFs ranged from 0.02 to 2.77 pg/$m^3$. I-$TEQ_{DF}$ values for this central McMurdo location were 0.074 pg/$m^3$ for 1992/93 and 0.0015 pg/$m^3$ for 1993/94. The most frequently detected congeners were the octa- and hepta-chlorinated CDDs.

## 3.3. CONCENTRATIONS IN SOIL

Tables B-4 and B-5 (Appendix B) contain summaries of data from several of the numerous studies in the published peer-reviewed literature regarding concentrations of CDDs and CDFs in soil. Data on dioxin-like PCB congener soil concentrations were not found in the literature; the PCB soil concentration data found in the literature were reported as either total PCB concentrations or concentrations of Aroclor PCB mixtures. Descriptions of several of the studies summarized in Appendix B are presented below. It should be noted that, the review of soil data presented here is not based on a comprehensive review of the published studies on CDD/CDFs in soil. Instead, it is intended to provide a brief

overview of soil levels of CDD/CDFs from a sampling of representative studies. Because of the lack of geographic coverage and non-uniform study design associated with the soil data presented in this section, there is a degree of uncertainty associated with the estimates of background concentrations of CDD/CDFs in soil.

### 3.3.1. North American Data

Soil sampled in 1987 from the vicinity of a sewage sludge incinerator was compared with soil from rural and urban sites in Ontario, Canada, by Pearson et al. (1990). Soil in the vicinity of the incinerator showed a general increase in CDD concentration with increasing degree of chlorination (Table 3-9). Of the CDFs, only OCDF was detected (mean concentration 43 ppt). Rural woodlot soil samples contained only OCDD (mean concentration of 30 ppt). Soil samples from undisturbed urban parkland settings revealed only HpCDDs and OCDD, but all CDF congener groups ($Cl_4$ to $Cl_8$) were present. Those samples showed an increase in concentration from the HpCDDs to OCDD and PeCDFs to OCDF. TCDFs had the highest mean value (29 ppt) of all the CDF congener groups. Resampling of one urban site in 1988, however, showed high variability in the concentrations of CDDs and CDFs.

Reed et al. (1990) analyzed background soil samples from a semi-rural location in Elk River, Minnesota, as part of a baseline assessment prior to the operation of a refuse-derived fuel-powered electric generation station. Four soil samples (two from an untilled site and two from a tilled site) were collected and analyzed for CDD/CDFs. Of the CDD/CDF congeners, OCDD concentrations were the highest, ranging from 340 ppt to 3,300 ppt. OCDF concentrations ranged from nondetect to 270 ppt. The 2,3,7,8-tetra and penta chlorinated congeners were not detected in any of the samples analyzed (Table 3-10).

Data were collected on CDD and CDF levels in soil samples from industrial, urban, and rural sites in Ontario and some U.S. Midwestern States (Birmingham, 1990). CDD/CDF levels in rural soils were primarily nondetect (ND), although the HpCDDs and OCDD were found in a few samples. In urban soils, the tetra- through octa-homologue groups were measured for both CDDs and CDFs. The HpCDDs and OCDD dominated and were two orders of magnitude greater than in the rural soils. These soils also contained measurable quantities of the TCDDs and PeCDDs. Industrial soils did not contain any TCDDs or PeCDDs, but they contained the highest levels of the TCDFs, HpCDFs, and OCDF. Total

CDD/CDF concentrations averaged 73 ± 50 ppt in rural soils (n = 30), 2,075 ± 3,608 ppt in urban soils (n = 47), and 8,314 ± 9,955 ppt in industrial soils (n = 20) when nondetects were assumed to be zero. I-$TEQ_{DF}$s were also calculated for these three types of sites by Birmingham (1990) by assuming that the 2,3,7,8-substituted CDD/CDF congeners represent specified proportions of the homologue group concentrations and by applying I-$TEF_{DF}$s. Birmingham (1990) estimated the I-$TEQ_{DF}$s to be 0.4 ± 0.6 ppt for rural soil, 11.3 ± 21.8 ppt for urban soils, and 40.8 ± 33.1 for industrial soils.

In another study, soils from industrialized areas of a group of cities from Midwestern and Mid-Atlantic States (Michigan, Illinois, Ohio, Tennessee, Pennsylvania, New York, West Virginia, Virginia) were analyzed for levels of 2,3,7,8-TCDD (Nestrick et al., 1986). Many of the samples were taken within 1 mile of major steel, automotive, or chemical manufacturing facilities, or municipal solid waste incinerators. Concentrations of 2,3,7,8-TCDD measured in this study ranged from ND to 9.4 ppt.

Nine background soil samples were collected from the Yarmouth Pole Yard Site located in Yarmouth, Maine (Tewhey Associates, 1997). One of these samples, collected from soil near the base of a utility pole, yielded an I-$TEQ_{DF}$ concentration of 57,000 pg/g. The I-$TEQ_{DF}$ concentrations for the other eight samples ranged from 0.73 pg/g to 5.9 pg/g when nondetects were assumed to be zero, and 1.46 pg/g to 6.07 pg/g when nondetects were assumed to be one-half the detection limit. These samples are from rural background locations. The mean I-$TEQ_{DF}$ for these eight samples was 3.58 pg/g ($TEQ_{DF}$-$WHO_{98}$ was 2.89 pg/g) when nondetects were set to zero and 3.93 pg/g when nondetects were set to one-half the detection limit. The sample collected near the utility pole was not included in these mean TEQ values, because its results were not considered to be representative of typical rural background concentrations.

In an effort to determine whether incineration of municipal waste influenced CDD/CDF levels in the immediate area of waste incineration facilities, soil samples were collected from cities with, and without operating incinerators throughout Connecticut. Between the years of 1987 and 1990, 34 soil samples were collected from eight different Connecticut cities where no municipal waste incinerators were operating (MRI, 1992). These pre-operational samples were considered to be representative of rural background concentrations. The total I-$TEQ_{DF}$ reported for these samples was 6.07 pg/g, with nondetects assumed to be one-half the detection limit. When the total TEQ was

recalculated in units of $TEQ_{DF}$-$WHO_{98}$, the total TEQ for these samples was 5.74 pg $TEQ_{DF}$-$WHO_{98}$/g. The proportion of nondetects ranged from 3 to 11 percent of samples for each analyte, with the exception of 2,3,7,8-TCDD and 1,2,3,7,8,9-HxCDF, which had 56 and 49 percent nondetects, respectively (MRI, 1992).

The Ministry of Environment in British Columbia conducted a 2-year monitoring study during 1990/91 and 1991/92 to evaluate the levels of CDD/CDF contamination in various types of environmental media (BC Environment, 1995). Soil samples were collected from sites close to a source (primary sites), in the receiving environment adjacent to a suspected source (secondary sites), and in areas not expected to be contaminated (background). Primary and secondary sources were identified as chemical or combustion sources. Chemical sources included sites associated with chlorophenol, herbicide, or PCB contamination; oil refineries; pulpmill landfills; or sewage facilities. Combustion sources included biomedical, industrial, municipal, or sewage sludge incineration; PCB or forest fires; pulp mill boilers; salt-laden wood burning, woodwaste burners, or slash burning; and scrap iron yards or smelters. The highest mean concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF were observed in primary and secondary soils associated with chemical sources (Table 3-11). For the 53 background samples, 2,3,7,8-TCDD was not detected, and 2,3,7,8-TCDF concentrations ranged from nondetected to 3.2 ppt. For the purposes of calculating I-$TEQ_{DF}$ values for this study, nondetects were set to zero. I-$TEQ_{DF}$s were highest among samples associated with primary and secondary chemical sources (Table 3-11). The mean I-$TEQ_{DF}$ for the background soil samples was 5.0 ppt (BC Environment, 1995). When the mean TEQ was recalculated in units of $TEQ_{DF}$-$WHO_{98}$, the total TEQ for these samples was 4.4 pg $TEQ_{DF}$-$WHO_{98}$/g.

Grundy et al. (1995) and Bright et al. (1995) collected soil samples from remote locations in the Canadian Arctic as part of an environmental assessment of abandoned military installations in the Canadian North. Four soil samples from remote pristine areas (i.e., at least 20 km away from any human activity) were analyzed for CDD/CDFs. The total I-TEQ concentrations for these samples ranged from 0.2 to 0.9 ppt (Grundy et al., 1995). Of the CDD/CDF homologue groups, OCDD and TCDF levels were the highest among these remote soil samples, and the HxCDFs made up the smallest portion of the total CDD/CDF concentrations (Bright et al., 1995).

EPA conducted a 2-year nationwide study to investigate the national extent of 2,3,7,8-TCDD contamination (U.S. EPA, 1987). Results of this large study were summarized broadly in the primary reference (i.e., the number and types of samples per site and range of detection). The method used to analyze samples for five of the seven study "tiers" had a detection limit in soil, sediment, and water of 1 part per billion (ppb). [Each "tier" of sites is a grouping of sites with a common past or present use (e.g., industrialized, pristine, etc.)]. Only Tier 5 (sites where pesticides derived from 2,4,5-trichlorophenol (TCP) had been or were being used for commercial purposes) and Tier 7 (ambient sampling for fish and soil) had detection limits of 1 ppt. Consequently, the data from this study are not included in the Appendix B tables; however, some observations from this study with regard to soil contamination are discussed below.

Soil concentrations found in most of the 100 Tier 1 and 2 sites (i.e., sites already on or expected to be on the NPL list) were in the ppb range; although in a few sites where concentrated 2,4,5-TCP production wastes were stored or disposed, concentrations were as high as 2,000 parts per million (ppm). Off-site soil contamination of concern was confirmed in 7 of the 100 Tier 1 and 2 sites, with soil concentrations in the ppb range. Eleven of the 64 Tier 3 sites (facilities and associated disposal sites where 2,4,5-TCP and its derivatives were formulated into pesticide products) were found to have soil concentrations exceeding 1 ppb, and in 7 of 11 sites where contamination was found, only one or two soil samples were above 1 ppb. Fifteen of the 26 Tier 5 sites (areas where 2,4,5-TCP and pesticide derivatives had been or were being used) had concentrations above 1 ppt, and one of those had a single detection of 6 ppb. Two-thirds of all detections at the Tier 5 sites were below 5 ppt. Three of the 18 Tier 6 sites (organic chemical and pesticide manufacturing facilities where improper quality control on production processes could have resulted in 2,3,7,8-TCDD being introduced into the wastestreams) had soil concentrations that exceeded the detection limit of 1 ppb, although these levels were limited to one or two samples per site. Seventeen of the 221 urban soil sites and 1 of the 138 rural sites from Tier 7 (background sites not expected to have contamination) had soil concentrations exceeding 1 ppt. The highest concentration detected (11.2 ppt) was found in an urban sample. Results from Tier 7 are consistent with the other studies discussed in this chapter regarding soil concentrations of 2,3,7,8-TCDD in nonindustrial settings.

Rappe et al. (1995a) and Fiedler et al. (1995a) analyzed soil samples collected from rural sites in southern Mississippi for CDDs and CDFs. Sites not directly impacted by human activities such as heavy traffic or dust were selected. A total of 36 composite soil samples from 8 Mississippi counties were analyzed. The I-$TEQ_{DF}$ concentration of CDD/CDFs in soil ranged from 0.16 to 22.9 ppt dry mass (Fiedler et al., 1995a). The mean I-$TEQ_{DF}$ concentration was 3.1 ppt dry mass, and the median I-$TEQ_{DF}$ concentration was 0.8 ppt dry mass (Fiedler et al., 1995a). CDDs were found at higher concentrations than CDFs, and OCDD was the most dominant congener.

Soil samples were collected from the National Institutes of Health (NIH) campus in Bethesda, Maryland, during 1995 in an effort to determine the effect of 30 years of pathological waste incineration on the campus and its surroundings (NIH, 1995). Thirty-seven samples were collected from the soil at a depth of 6 inches. The total I-$TEQ_{DF}$ for these samples was 7.83 pg/g, when nondetects were assumed to be zero, and 8.49 pg/g, when nondetects were assumed to be one-half the detection limit. OCDD, at a I-$TEQ_{DF}$ concentration of 6.29 pg/g, was the principal contributor to the total I-$TEQ_{DF}$ for these samples, regardless of whether nondetects were assumed to be zero or one-half the detection limit. It should be noted that using the new $TEF_{DF}$-$WHO_{98}$s, the TEQ for OCDD would be 10 times lower (i.e., 0.63 pg/g). This reduction would also result in a significant decrease in the total TEQ. The total $TEQ_{DF}$-$WHO_{98}$ would be 2.21 pg/g, when nondetects were set to zero. Samples were also collected at depths of 12 and 24 inches for comparison to levels found in the shallow (6-inch) samples. While CDD/CDF concentrations found at the surface indicate deposition, strong correlation with I-$TEQ_{DF}$ concentrations at the deeper depths were observed. This seemed to indicate either long-term presence of the source (i.e., greater than 40 years), or soil mixing that has occurred either during or after deposition. An expert panel (comprised of toxicologists, chemists, soil scientists, engineers, risk assessors, and public health professionals) concluded that the levels of I-$TEQ_{DF}$ in the samples are low and not significantly different from background. Thus, these samples are assumed to be representative of urban background concentrations. The spatial pattern of I-$TEQ_{DF}$ concentrations showed no particular trends that could be related to the incinerator. Other anthropogenic activities, such as vehicular traffic, other medical waste incinerators not related to NIH, and fireplaces burning in the vicinity, may have contributed to the deposition (NIH, 1995).

Recently, U.S. EPA (1996) collected soil samples in the vicinity of a municipal waste-to-energy (WTE) facility in Columbus, Ohio, to determine whether surface soils around the incinerator contained higher CDD/CDF levels than soils collected from background sites. The facility is not currently in operation, but CDD/CDF residues may be present in the soil near the facility as a result of past emissions. Samples were collected from (1) on-site, (2) urban background locations near the incinerator, and (3) areas remote from the facility (i.e., rural background sites). The results of the analyses indicate that soil from the rural background sites had the lowest I-$TEQ_{DF}$ concentrations and on-site samples had the highest I-$TEQ_{DF}$ concentrations (Table 3-12). For rural background soil samples, total I-$TEQ_{DF}$s ranged from 0.9 to 1.3 ppt (n = 3) with a mean of 1.1 ppt ($TEQ_{DF}$-$WHO_{98}$ = 0.9 ppt), when nondetects were assumed to be zero, and 1.0 to 2.0 ppt with a mean of 1.4 ppt ($TEQ_{DF}$-$WHO_{98}$ = 1.3 ppt), when nondetects were set to one-half the detection limit. Total I-$TEQ_{DF}$s for urban background soils ranged from approximately 3 to 60 ppt (n = 18) with a mean of 19 ppt ($TEQ_{DF}$-$WHO_{98}$ = 21 ppt), when nondetects were set to either zero or one-half the detection limit. For on-site samples, all 2,3,7,8-CDD/CDF congeners were detected in all samples (n = 4). Total I-$TEQ_{DF}$ concentrations ranged from 50 to 760 ppt with a mean of 356 ppt ($TEQ_{DF}$-$WHO_{98}$ = 444 ppt). Additional detail and analyses of these data are presented in Lorber et al. (1998a).

Brzuzy and Hites (1995) examined soil cores from four U.S. locations to evaluate the accuracy of using measurements of CDD/CDF homologue groups in estimating the atmospheric flux of these compounds into the environment. Soil cores were collected from undisturbed areas near Shingleton, Grayling, and Verona, Michigan, and near Mitchell, Indiana. CDD/CDF concentrations varied according to depth of the soil samples, with deeper samples having lower CDD/CDF concentrations. Approximately 80 percent of the CDD/CDF load were contained in the top 15 cm of the cores, and CDD/CDF concentrations were close to the detection limit in samples collected at a depth of 20-25 cm. Based on the graphs presented in Brzuzy and Hites (1995), total CDD/CDF concentrations in the uppermost 5 cm of the core ranged from approximately 60 pg/g to 200 pg/g for the three Michigan sites. CDD/CDFs in these soil cores were also found to be highly correlated with the organic carbon content of the soil, indicating that organic carbon is an important factor in the sorption of CDD/CDFs to soil (Brzuzy and Hites, 1995). Higher concentrations of CDD/CDFs were observed in two cores taken from the Indiana site. Concentrations in the

uppermost layer (i.e., 9 cm) of these cores ranged from approximately 700 pg/g to nearly 10,000 pg/g. CDD/CDF concentrations in these cores peaked at a depth of approximately 40 to 50 cm with concentrations ranging from approximately 1,000 pg/g to over 20,000 pg/g. Brzuzy and Hites (1995) used the Michigan data to estimate soil-derived CDD/CDF flux rates ranging from 264 ng/m$^2$/yr for upper Michigan to 663 ng/m$^2$/yr for lower Michigan. These soil-derived flux estimates were compared to sediment-derived fluxes from previous studies to determine if soil samples can also be used to accurately predict atmospheric flux. Good agreement for the fluxes to these two media was observed. In addition, the CDD/CDF homologue profiles for soil and sediment were similar.

### 3.3.2. European Data

Soil samples from rural and semi-urban sites in England, Wales, and lowland Scotland showed a general increase in concentration from the TCDDs to OCDD, whereas CDF levels showed very little variation between the congener groups (Creaser et al., 1989). Concentrations of 2,3,7,8-TCDD at those sites ranged from $<0.5$ to 2.1 ppt. The median values for the TCDDs to OCDD were 6.0, 4.6, 31, 55, and 143 ppt, respectively. The median values for the TCDFs to OCDF were 16, 17, 32, 15, and 15 ppt. Evaluation of soil data from urban sites in the same geographical area showed that the mean levels for the CDD and CDF congeners were significantly greater ($p<0.01$) than those for rural and semi-urban background soils (Creaser et al., 1990). Concentrations of 2,3,7,8-TCDD at the urban sites ranged from $<0.5$ to 4.2 ppt. The median values for the TCDDs to OCDD were 40, 63, 141, 256, and 469 ppt, respectively. The median values for the TCDFs to OCDF were 140, 103, 103, 81, and 40 ppt. Significantly elevated levels of the lower congeners, together with higher overall CDD/CDF concentrations, are indicative that local sources and short-range transport mechanisms are major contributors of CDDs and CDFs to urban soils. Cox and Creaser (1995) evaluated soils from urban and rural locations in the United Kingdom before the introduction of Integrated Pollution Control in 1991. I-TEQ$_{DF}$s for 11 rural locations ranged from 0.78 ppt to 17.48 ppt, with a mean of 5.17 ppt, and the I-TEQ$_{DF}$s for 5 urban samples ranged from 4.88 pt to 87.34 ppt with a mean of 28.37 ppt.

Analysis of four sites in Hamburg, Germany, contaminated by an organochlorine pesticide manufacturing company showed patterns of CDD and CDF distribution similar to the urban and industrial sites examined in England, Wales, and Scotland (Sievers and

Friesel, 1989). The study indicated that CDDs and CDFs showed a regular increase in concentration with increasing degree of chlorination (although individual data points were not presented). Maximum concentrations of 2,3,7,8-TCDD ranged from 900 ppt to 874,000 ppt. Very high concentrations of 2,3,7,8-TCDD at the sites were attributed to an admixture of wastes from 2,4,5-T production.

A soil sampling survey in Salzburg, Austria, also showed that the concentrations of CDD/CDFs were higher in urban and industrial sites than in rural sites (Boos et al., 1992). The total CDD content of the soils ranged from 33.7 to 1236.7 ppt for urban sites, 92.2 to 455 ppt for industrial sites, and 7.1 to 183.6 ppt for rural sites. The total CDF content of the soils ranged from 45.6 to 260.8 ppt for urban sites, 53.0 to 355.3 ppt for industrial sites, and 12.0 to 77.7 ppt for rural sites. I-$TEQ_{DF}$s ranged from 0.1 ppt to 3.1 ppt for rural sites, 1.0 ppt to 8.3 ppt for urban sites, and 3.5 ppt to 11.5 ppt for industrial sites, when nondetects were assumed to be zero. When nondetects were set to one-half the limit of detection, I-$TEQ_{DF}$s ranged from 1.3 ppt to 3.8 ppt for rural sites, 2.0 ppt to 8.6 ppt for urban sites, and 4.1 ppt to 12.5 ppt for industrial sites.

Rappe and Kjeller (1987) presented data on CDD/CDFs in soil collected from rural (n = 3) and industrial (n = 2) sites in various parts of Europe. Concentrations were higher among industrial soils than rural soils for all of the CDD/CDF homologue groups, and the hepta-chlorinated compounds made up the largest portion of the total CDD/CDF concentrations in both rural and industrial samples. HpCDDs ranged from nondetected to 17 ppt in rural samples and 370 to 1,600 ppt in industrial samples. HpCDFs ranged from 14 to 22 ppt in rural soils and 260 to 4,500 ppt in industrial soils.

Rotard et al. (1994) measured CDD/CDFs in soil samples collected from forest, grassland, and plowland sites in western Germany. The highest mean concentration of CDD/CDFs were found in the subsoil and topsoil layers of deciduous (38.0 ng I-$TEQ_{DF}$/kg dry matter; n = 9) and coniferous forests (36.9 ng I-$TEQ_{DF}$/kg dry matter; n = 11). Grassland and plowland sites had mean concentrations of 2.3 ng I-$TEQ_{DF}$/kg dry matter (n = 7) and 1.7 ng I-$TEQ_{DF}$/kg dry matter (n = 14), respectively.

Stenhouse and Badsha (1990) collected baseline data for soils around a site proposed for a chemical waste incinerator in Great Britain. All of the 2,3,7,8-substituted CDD/CDF congeners, except PeCDD, were detected in all samples. Concentrations were highest for the octa-chlorinated CDD/CDFs. Background I-$TEQ_{DF}$ concentrations ranged

from 3 to 20 ppt. The mean I-$TEQ_{DF}$ concentration was 8 ppt (n = 12), with a standard deviation of 4 ppt.

Buckland et al. (1998) evaluated soils collected in New Zealand. Dry weight CDD/CDF concentrations ranged from 0.17 to 1.99 pg I-$TEQ_{DF}$/g for pristine soils, 0.17 to 0.90 pg I-$TEQ_{DF}$/g for agricultural soils, and 0.52 to 6.67 pg I-$TEQ_{DF}$/g for urban soils. The PCB concentrations ranged from 0.067 to 2.3 pg $TEQ_P$/g (the TEFs used for PCBs were not identified) for provincial centers and 0.087 to 1.33 pg $TEQ_P$/g for metropolitan centers. The congeners below the detection limit were included in the total TEQ using half their limits of detection.

Masahide et al. (1998a) examined soil samples collected at the depth of 0-10 cm from various sites located in Poland between 1990 and 1994. The mean dry weight total PCB concentration was 8.6 ng/g for agricultural and forest soils, 170 ng/g (n = 31) for urban soils, and 900 ng/g for the soils sampled at the military area. Dry weight PCB concentrations increased from 21 ng/g in Northern Poland to 48-380 ng/g in highly populated and industrialized regions in Southern Poland.

### 3.3.3. Soil Observations and Trends

Some general observations for CDD and CDF levels in soils are possible from the data presented in the various soil studies discussed above:

- As the degree of chlorination increases, the concentrations increase. Concentrations of the hepta- and octa-chlorinated congeners are generally higher than the tetra-, penta-, and hexa-chlorinated congeners.
- Concentrations in settings identified as urban are higher than those in areas identified as rural.
- Concentrations associated with industrial sites clearly are the highest, with concentrations in the hundreds to thousands of parts per trillion.

### 3.3.4. Soil CDD/CDF Profiles and Background TEQ Concentrations

CDD/CDF homologue group profiles for soil were calculated as the mean homologue group concentrations divided by the mean total CDD/CDF concentration. Nondetects were assumed to be zero in these calculations. For rural background soil, homologue group

profiles were calculated based on data from Reed et al. (1990), Birmingham (1990), Pearson et al. (1990), BC Environment (1995), U.S. EPA (1985, 1996), Tewhey Associates (1997), and MRI (1992). They indicate that of the homologue groups, the higher chlorinated compounds dominate. OCDD and HpCDD account for the highest percentages of total CDD/CDFs (Figure 3-3; Table 3-13). CDD/CDF homologue group profiles for urban background samples, based on Birmingham (1990), Pearson et al. (1990), NIH (1995), and U.S. EPA (1996), are similar (Figure 3-4; Table 3-13). The sum of 2,3,7,8-substituted congener concentrations account for 84 percent of the total CDD/CDF concentrations in rural background soil and 93 percent in urban background soil. Profiles based on the ratio of 2,3,7,8-substituted congeners to total CDD/CDFs in rural background soil indicate that OCDD and 1,2,3,4,6,7,8-HpCDD account for the highest percentages of total CDD/CDFs, followed by OCDF and 1,2,3,4,6,7,8-HpCDF (Reed et al., 1990; BC Environment, 1995; U.S. EPA, 1996; Tewhey Assoc., 1997; MRI, 1992) (Figure 3-3). For urban background soils, profiles were similar to those observed in rural background soils, based on data from NIH (1995) and U.S. EPA (1996) (Figure 3-4).

Based on several of the studies described in this chapter, mean $TEQ_{DF}$-$WHO_{98}$ levels were calculated (Table 3-14), based on the available data, to represent "typical" background conditions in the North America. The mean rural background $TEQ_{DF}$-$WHO_{98}$ level was estimated to be 3.6 ppt, and the "typical" urban background $TEQ_{DF}$-$WHO_{98}$ level was estimated to be 11.9 ppt, assuming that nondetects equal zero. The mean rural background concentration is based on data from eight studies in five U.S. States (i.e., Ohio, Minnesota, Illinois, Maine, and Connecticut) and two areas in Canada (i.e., British Columbia and Ontario) (n = 222; BC Environment, 1995; Birmingham, 1990; MRI, 1992; Pearson et al., 1990; Reed et al., 1990; Tewhey Associates, 1997; U.S. EPA, 1985; U.S. EPA, 1996). The mean urban background concentration is based on data from three U.S. States (i.e., Michigan, Maryland, and Ohio) and Ontario, Canada (n = 157; Birmingham, 1990; Nestrick et al., 1986; NIH, 1995; Pearson et al., 1990; U.S. EPA, 1985; U.S. EPA, 1996). In calculating the mean concentrations, each site was weighted equally (i.e., the means are based on the average of the mean concentrations from each location). In contrast, the weighted means in Table 3-14 represent average concentrations for rural and urban sites that have been weighted according to the number of samples at each site; all soil samples are treated individually regardless of location. Thus, more heavily sampled locations have a

greater impact on the weighted mean than those sampled less frequently. Therefore, means, not weighted means, were used to depict typical rural and urban background concentrations in North America. It should be noted, however, that the means and weighted means in Table 3-14 are quite similar.

Background $TEQ_{DF}$-$WHO_{98}$ levels for soils were estimated by setting nondetects to zero instead of one-half the detection limit, because congener-specific detection limits were not available for most studies. If one-half the detection limits had been used to represent nondetected congeners, the estimated background $TEQ_{DF}$-$WHO_{98}$ levels may have been slightly higher. Based on the results of European studies, it appears that background $TEQ_{DF}$-$WHO_{98}$ concentrations in European soil are similar to those of the United States. The $TEQ_{DF}$-$WHO_{98}$ concentration in urban background soil is used in Chapter 4 to calculate background exposures. Urban soils are used because the majority of the population lives in urban areas. Thus, using these concentrations would represent typical exposure levels.

## 3.4. CONCENTRATIONS IN WATER

Tables B-6 and B-7 (Appendix B) contain summaries of data from the limited number of published studies regarding concentrations of CDDs and CDFs in water. Data on dioxin-like PCB congener concentrations in water were not found in the literature. Several of the CDD/CDF studies are discussed below.

### 3.4.1. North American Data

A survey of 49 drinking water supplies in Ontario, including supplies in the vicinity of chemical industries and pulp and paper mills, was initiated in 1983 (Jobb et al., 1990). As of February 1989, 4,347 congener analyses had been performed on 399 raw and treated water samples. OCDD was detected in 36 of 37 positive results and ranged from 9 to 175 ppq in raw samples (33 positive samples) and 19 to 46 ppq in treated samples (4 positive samples). These low concentrations were found primarily in water obtained downstream of industrialized areas in the St. Clair/Detroit River system. No samples contained detectable levels of 2,3,7,8-TCDD. Because CDDs and CDFs are hydrophobic compounds, and consequently, have a tendency to sorb onto particulate matter in water, conventional water treatment processes are expected to be effective in removing the contaminants along with the particulates. This is substantiated by the fact that 33 of the

37 positive results were raw water samples. Because of the relatively low levels of CDDs detected in the samples, it is difficult to ascertain whether the CDDs were particulate-associated or dissolved.

A survey of 20 community water systems throughout New York State was conducted in 1986 (Meyer et al., 1989). The sampling sites were representative of the major surface source waters in New York. They included sources receiving industrial discharges or known to contain dioxin-contaminated fish, as well as waters in more remote areas. TCDFs were detected in the finished water at the Lockport facility (duplicate samples had concentrations of 2.1 and 2.6 ppq). Except for a trace of OCDF detected at one location, no other CDDs/CDFs were detected in finished water at any of the other 19 community water systems surveyed. Raw water sampled at the Lockport facility contained concentrations of TCDDs (1.7 ppq), as well as TCDFs to OCDF (18, 27, 85, 210, and 230 ppq, respectively). As can be seen from the data, the CDF congener group concentrations increased with increasing chlorine number.

### 3.4.2. European and Japanese Data

CDDs in surface water samples collected from the Eman River in southern Sweden generally increased in concentration from the TCDDs to OCDD; whereas, the CDF levels showed very little variation between the congener groups (Rappe et al., 1989a). In general, however, the levels of CDFs were higher than the levels of CDDs. Concentrations of 2,3,7,8-TCDF were 0.022 ppq in Jarnsjon and 0.026 ppq in Fliseryd. The filtered water, before chlorination and distribution as drinking water, had no detectable tetra-, penta-, or hexa-chlorinated congeners of CDDs or CDFs, but the HpCDDs and OCDD were detected at 120 and 170 ppq, respectively.

A survey was conducted at the Venice lagoon in the north of Italy by the Ministry of Justice and the Ministry of Works to assess the CDD/CDFs contamination produced by industrial and municipal waste water discharges (Ramacci et al., 1998). The results showed that four out of seven waste water samples were characterized by a prevalence of OCDF; two had an almost equal distribution between OCDD and OCDF; and only one collected from a septic tank of the city of Venice presented a prevalence of OCDD.

In Japan, water samples were collected from a coastal area, river, and pond in Matsuyama between October 1996 and September 1997 (Seike et al., 1998). The total

concentrations of CDD/CDFs ranged from 15 to 170 pg/L (n = 3) in coastal waster with an average of 51 pg/L, from under the detection limit to 1500 pg/L (n = 22) in river water with an average of 180 pg/L, and from 44 to 530 pg/L (n = 3) in pond water with an average of 260 pg/L. CDD/CDFs in coastal water were relatively lower than from other sources, and were thought to be diluted with seawater and/or deposited to the coastal sediment.

### 3.4.3. Water Observations and Trends

Some general observations for CDD and CDF levels are possible from the limited data available from the various water studies described above:

- CDD/CDFs are seldom detected in drinking water at ppq levels or higher.
- Raw water samples generally have higher concentrations of CDD/CDFs than finished water samples.
- The concentration of CDDs and CDFs in surface water generally increases from the tetra-chlorinated to the octa-chlorinated congener groups.

### 3.4.4. Water CDD/CDF Profiles and Background TEQ Concentrations

CDD/CDF congener profiles could not be generated for water because of the lack of congener-specific data for treated drinking water. For the background studies reviewed here, only data for OCDD and OCDF were available.

Based on the above studies, a total of 236 samples from Ontario, Canada, and Lockport, New York, were selected as representing background conditions in North America. The "typical" $TEQ_{DF}$-$WHO_{98}$ level was computed as 0.00056 ppq, assuming that nondetects equal half the detection limit (Jobb et al., 1990). This value is used in Chapter 4 to estimate background exposures to the U.S. population from drinking water consumption. It should be noted, however, that OCDD and OCDF were the only congeners for which background data were available. Of the 214 samples analyzed for OCDD, only 4 were positive, and only 2 out of 22 samples analyzed for OCDF were positive. No appropriate data on drinking water concentrations could be found for Europe.

## 3.5. CONCENTRATIONS IN SEDIMENT

Tables B-8 through B-10 (Appendix B) contain summaries of data from several of the numerous studies in the published literature regarding concentrations of CDDs, CDFs, and dioxin-like PCB congeners in sediment. Several of these studies are discussed in the following paragraphs. It should be noted that the review of sediment data presented here is not based on a comprehensive review of the published studies on CDD/CDFs in sediment. Instead, it is intended to provide a brief overview of sediment levels of CDD/CDFs from a sampling of representative studies. In addition, because the levels of CDD/CDFs and PCBs in sediment layers may be indicative of the cumulative history of contamination at a site (i.e., reservoir sources), the studies presented here represent only those that analyzed surficial (i.e., recently deposited) sediment samples or the uppermost layers from sediment cores, and not deep sediment core samples.

### 3.5.1. North American Data

In sediment samples collected from estuaries adjacent to an industrial site in Newark, New Jersey, where chlorinated phenols had been produced, the level of OCDD was many times higher than that of 2,3,7,8-TCDD (Bopp et al., 1991). Studies conducted by Wenning et al. (1992; 1993) in Newark Bay also indicated that OCDD was found in higher concentrations than the lower-chlorinated congeners in this water body. Based on 19 sediment samples, OCDD levels ranged from 310 ppt to 17,000 ppt, and 2,3,7,8-TCDD levels ranged from 2.8 pt to 480 ppt. Observed congener patterns for this area were similar to those found in sediments from several other U.S and European water bodies. The authors suggest that these similarities are a result of similar municipal and industrial sources in heavily industrialized and populated areas. Based on the results of principal components analyses, the congener profiles for sediments from Newark Bay are closely related to those of several different potential sources, including chemical manufacturing processes and municipal activities. Hudson River sediment samples contained primarily the higher chlorinated ($Cl_6$ to $Cl_8$) CDD and CDF congeners (Petty et al., 1982). Concentrations of the HpCDD and OCDD homologues ranged from 5 to 15 ppb, and the OCDD homologue in most instances accounted for more than half of the total CDD residue. Likewise, the HpCDFs and OCDF occurred at the highest levels (ca. 1 ppb).

Surface sediment samples were collected from several estuaries in the United States (Norwood et al., 1989). Sampling sites included Black Rock Harbor in Bridgeport, Connecticut, (an industrialized urban estuary); central Long Island Sound (a relatively clean reference site); Narragansett Bay, Rhode Island, (where chemical industries may have contributed to the input); New Bedford Harbor, Massachusetts, (a section of which is a National Superfund Site because of PCB contamination); and Eagle Harbor, Washington, (the site of a creosote wood treatment facility). Sediments in New Bedford Harbor were reported to be more heavily contaminated with CDFs than sediments from the other sites. In particular, HxCDF congeners were greater by a factor of 40 (although individual data points were not presented) at one of the more contaminated New Bedford Harbor sites. In contrast, sediments from Eagle Harbor were practically devoid of CDFs and showed a large increase in the HpCDD and OCDD congeners closer to the treatment facility. Narragansett Bay and Black Rock Harbor were similar in both concentration and distribution of CDDs and CDFs, and Black Rock Harbor contained slightly higher levels of the tetra- to hepta-CDD and CDF congeners. Sediment from Long Island Sound was cleaner and had a distribution of CDFs between that of Narragansett Bay and Black Rock Harbor. Sediment with the least contamination was collected in New Bedford Harbor, up-river from the PCB facilities; the highest OCDD concentration (1,400 ppt) was detected in Eagle Harbor.

Sediment samples from Siskiwit Lake, on Isle Royale, Lake Superior, were examined to evaluate the atmospheric input of CDDs and CDFs to the lake (Czuczwa et al., 1984). The water level in Siskiwit Lake is 17 meters higher than that in Lake Superior, and in addition, there are no anthropogenic inputs in the drainage basin of Siskiwit Lake. Consequently, the atmosphere is the only source of anthropogenic chemicals in that lake. OCDD was most predominant, and the HpCDD and HpCDF congeners also were abundant. The study indicated a considerable decrease in concentration of all CDDs and CDFs between 6 and 8 cm of the sediment core depth (i.e., sediment believed to have been deposited about 1940).

Surficial sediments collected from Jackfish Bay on the north shore of Lake Superior contained moderate concentrations of the TCDF and OCDD congeners, with trace concentrations of other congeners (Sherman et al., 1990). The concentration of OCDD was similar to that found in the Siskiwit Lake sediment samples. The OCDF and OCDD profile for a sediment core collected from Moberly Bay was similar to the surficial sediment

pattern. These congener groups predominated at all depths where detectable concentrations occurred. In addition, low concentrations of the HpCDD and PeCDF and HpCDF congeners were detected. The concentration profile of the HpCDF congener group showed a relatively high value that dropped abruptly to nondetectable (< 60 ppt) below a depth of 10 cm. This abrupt change corresponded to a section date 1973 that reflects an operational change at the pulp mill.

A survey of surficial harbor sediments collected near a wood preserving plant in Thunder Bay, Ontario, Canada, on the north shore of Lake Superior, found the highest concentrations of CDD/CDFs at stations closest to the plant dock, and lower concentrations at locations further from the source (McKee et al., 1990). No CDDs or CDFs were detected below the surficial layer. TCDD and PeCDD congeners were below analytical detection limits in all samples. However, the concentrations of the HxCDDs to OCDD congeners increased with the degree of chlorination. The maximum concentrations of the HxCDDs to OCDD ranged from 5,700 ppt for the HxCDDs to 980,000 ppt for OCDD. As with the CDD distribution profile, the HxCDFs to OCDF increased with the degree of chlorination.

Sediment levels of CDD/CDFs in British Columbia were found to be higher in samples collected from sites in the receiving environment adjacent to a suspected source (secondary sites) than in areas not expected to be contaminated (background) (Table 3-15) (BC Environment, 1995). These observations are based on a 2-year study conducted during 1990/91 and 1991/92 by British Columbia's Ministry of Environment. Secondary sites were identified as being associated with chemical or combustion sources. Background samples did not contain detectable levels of 2,3,7,8-TCDD, and the mean 2,3,7,8-TCDF concentration was 1.4 ppt. For the purposes of calculating I-$TEQ_{DF}$ values for this study, nondetects were set to zero. Mean I-$TEQ_{DF}$ levels were 32.5 ppt for secondary sites (all sources) and 3.9 ppt for background sites (Table 3-15).

Bottom surficial sediments (0-3 cm) were collected from the sedimentation basins of Lake Ontario to assess the levels of the various PCB congeners (Oliver and Niimi, 1988). Concentrations of 2,3',4,4',5-PeCB (PCB 118); 2,3,3',4,4'-PeCB (PCB 105); and 2,3,3',4,4',5-HxCB (PCB 156) in the sediment were 15, 10, and 2.1 ppb, respectively. A baseline assessment of CDDs and CDFs was performed on the Elk River, a semi-rural area located about 25 miles northwest of Minneapolis-St. Paul, Minnesota (Reed et al., 1990).

Sediment samples were collected from Lake Orono, a reservoir on the Elk River, and from an abandoned gravel pit. Although none of the sediment samples contained 2,3,7,8-TCDD, the gravel pit sediments contained measurable concentrations of TCDFs. Only one Lake Orono sample contained measurable concentrations of 2,3,7,8-TCDF (0.31 ppt) and total TCDF (0.54 ppt). Gravel pit samples also contained HpCDDs to OCDD and PeCDFs to OCDF. Lake Orono samples contained HpCDDs, OCDD, and HpCDF congeners. HpCDDs ranged from 7.3 ppt in the lake inlet to 110 ppt in the gravel pit and in the lake near the dam. OCDD concentration ranged from 450 ppt in the gravel pit to 600 ppt in the middle of Lake Orono. The sediment profiles reflected combustion source influences.

The Sheboygan River, a Wisconsin tributary to Lake Michigan, is polluted with PCBs from the mouth to about 14 miles upstream (Sonzogni et al., 1991). That portion of the river is a Superfund site, as well as one of the Great Lakes "Areas of Concern." Sediment cores were collected at Rochester Park, near the original source of the PCBs, about 14 miles upstream from the mouth. The PCB congeners 2,3',4,4',5-PeCB (PCB 118); 2,3,3',4,4'-PeCB (PCB 105); and 3,3',4,4'-TCB (PCB 77) were detected in all samples and ranged from about 5 to 1,500 ppb. Remaining dioxin-like PCB congeners were detected less frequently and ranged from nondetectable to slightly over 100 ppb. The PCB congener 2,3',4,4',5-PeCB (PCB 118) appears to be the most common dioxin-like PCB in environmental samples and was found in the Sheboygan River sediments in the highest weight percent. The eight toxic PCBs detected in this study were present in relatively low concentrations compared to total PCBs or other more abundant congeners.

Sediments collected from Waukegan Harbor in Lake Michigan contained the dioxin-like PCB congeners 3,3',4,4'-TCB (PCB 77) and 2,3,3',4,4'-PeCB (PCB 105) (Huckins et al., 1988). The percentage of 3,3',4,4'-TCB (PCB 77) in the samples averaged 0.16 percent ± 0.15, with concentrations ranging from 13 to 27,500 ppb. The percentage of 2,3,3',4,4'-PeCB (PCB 105) averaged 0.66 percent ± 0.37, with concentrations ranging from 102 to 131,000 ppb. In another Lake Michigan study, sediment samples collected from Green Bay contained concentrations of all 11 dioxin-like PCB congeners (Smith et al., 1990b). The dominant congeners were 2,3,4,4',5-PeCB (PCB 114) and 2,3,3',4,4'-PeCB (PCB 105), with concentrations of 11 and 5.8 ppb, respectively.

Fiedler et al. (1995b) and Rappe et al. (1995b) analyzed sediments samples from a river system in southern Mississippi to determine if the concentrations of CDDs and CDFs in

the sediments were influenced by a local pulp and paper mill. The pulp mill, which is located adjacent to the Leaf River and currently uses chlorine dioxide in its bleaching process (elemental chlorine was used from 1984 through 1989), was suspected of contributing to CDD and CDF contamination in the region (Fitzpatrick, 1995). A total of 61 sediment samples were collected from sites located upstream and downstream from the mill. Study results indicated that most CDD and CDF congeners were present in all sediment samples collected, but the predominant congeners were the HpCDDs and OCDD (Rappe et al., 1995b). Congener profiles were generally similar for samples collected from both populated, potentially polluted areas and pristine areas. In addition, for the majority of sampling sites in the study, the ratio of the sum of CDDs to CDFs ranged from 43 to 1,200. Rappe et al. (1995b) stated that this observation "is notable because, with the exception of sewage sludge, no environmentally significant source has been identified with such a dominance of CDDs." The mean I-$TEQ_{DF}$ concentration observed in this study was 10.6 ng/kg dry weight, and the median I-$TEQ_{DF}$ concentration was 9.90 ng/kg dry weight (Fiedler et al., 1995b). Median values reported in this study were consistent with those of an earlier study conducted by the Mississippi Department of Environmental Quality in 1992 in the same river system.

Rappe et al. (1997a) analyzed sediment core samples from five lakes in southern Mississippi. The sediment cores were collected from five man-made recreational lakes with no known industrial point source of CDD/CDFs and low atmospheric deposition rates. Cores were subdivided into sections to evaluate temporal trends in deposition of CDD/CDFs. I-$TEQ_{DF}$ values for the lake cores ranged from 0.38 to 9.52 ppt (dry weight). 2,3,7,8-TCDD was present at levels below the detection limit in all of the sediment core samples. The higher chlorinated congeners (hexa to octa CDD/CDFs) predominated. OCDD levels ranged from 150 ppt dry weight to 5,500 ppt dry weight, while total CDD levels ranged from 176 to 7,577 ppt dry weight. CDF levels ranged from nondetectable to 14.4 ppt dry weight. As observed in another analysis of sediment cores from the region (Rappe et al., 1997b), the CDD to CDF ratios were very high, ranging from 79.1 to 9,920. The CDD and CDF congener patterns were similar to those observed by Rappe et al. (1995b) in previous sediment studies in the region. No observable trend for levels of CDDs, homologues, or I-$TEQ_{DF}$s correlating to the age of the strata could be identified.

Rappe et al. (1997b) also studied sediment samples from 15 manmade lakes in the same region of southern Mississippi. Lakes were selected from pristine areas, areas not known to be impacted by industrial point sources of CDDs and CDFs. As in previous studies from this region (Rappe et al., 1995b and Fiedler et al., 1995b), HpCDDs and OCDD were the predominant congeners in these deep lake and sedimentation area samples. OCDD levels ranges from 1,400 to 43,000 ppt dry weight (median 7,700 ppt). The concentration of HpCDDs ranged from 63 ppt to 2,000 ppt dry weight (median 430 ppt). 2,3,7,8-TCDD was detected in 20 of the 27 sediment samples at levels not exceeding 1.0 ppt dry weight. CDDs dominated, and the ratio of CDDs to CDFs ranged from 19 to 764 (79 percent had ratios $>100$). The I-$TEQ_{DF}$ concentration in the sediment samples ranged from 2 ppt dry weight to 63.7 ppt dry weight. Rappe et al. (1997b) postulated that the high levels of OCDD found in this region of the United States are due to natural formation. The authors base this theory on the fact that the sediment samples were collected in pristine areas with no known industrial sources of CDD/CDFs and low atmospheric deposition rates. However, they have observed consistently higher than expected OCDD levels and high CDD/CDF ratios in this region, which do not correspond with other levels in the published literature.

In an effort to determine whether incineration of municipal waste influenced CDD/CDF levels in the immediate area of waste incineration facilities, sediment samples were collected from surface water near cities with, and without, operating incinerators throughout Connecticut. A total of 344 sediment samples were collected between 1987 and 1990. The mean total CDD/CDF concentrations for pre-operational and operational status were 3,590 pg/g and 4,523 pg/g, respectively, when nondetects were assumed to be one-half the detection limit (MRI, 1992). Based on the concentration data reported in MRI (1992), mean I-$TEQ_{DF}$ values for pre-operational and operational sediments were 21 pg/g and 24 pg/g, respectively.

Sediment samples collected at the lowermost Tennessee River (Kentucky Dam Tailwater) and Kentucky Lake at the depth of 0-5 cm had total PCB concentrations ranging from the detection limit (1.0 ng/g dry weight) to 26.36 ng/g dry weight (Loganathan et al., 1998). The total PCB concentrations sampled from the lowermost Tennessee River were generally higher that those from Kentucky Lake. The PCB congeners found in the sediment samples included PCB-8, 29, 50, 28, 52, 44, 101, 87, 154, 118, 153, 105, 138, 187,

128, 200, 180, and 170; the dominant congeners were PCB-44 and PCB-101. Of these congeners, only PCBs 118 and 105 are considered to be dioxin-like.

Recently, EPA conducted a time-trend study of dioxin-like compounds in sediment cores (Versar, 1996a; Cleverly et al., 1996). Cores from 11 lakes/reservoirs were collected, sectioned and dated, and analyzed for CDD/CDFs and PCBs 77, 105, 118, 126, 156, 157, and 169. The lakes were located in various geographic locations throughout the United States and were selected to represent background conditions (i.e., no known CDD/CDF sources). Based on the most recently deposited sediments (i.e., the uppermost core sections), total I-$TEQ_{DF}$ concentrations ranged from 0.11 ppt to 15.6 ppt with a mean of 5.3 ($TEQ_{DF}$-$WHO_{98}$ concentrations ranged from 0.12 ppt to 16.3 ppt, with a mean of 5.3 ppt) when nondetects were set to one-half the detection limit, and from 0.11 ppt to 14.3 ppt with a mean of 4.6 ppt ($TEQ_{DF}$-$WHO_{98}$ concentrations ranged from 0.12 ppt to 15.6 ppt, with a mean of 4.7 ppt) when nondetects were set to zero (Table 3-16). Chandler Lake, an Arctic lake located in North Slope, Alaska, had the lowest $TEQ_{DF}$ concentration, and Canandaigua Lake in New York and Santeetlah Reservoir in North Carolina, both eastern lakes, had the highest $TEQ_{DF}$s. In general, the higher chlorinated CDD/CDFs accounted for the largest proportion of total TEQ concentrations. PCB $TEQ_P$-$WHO_{98}$ and $TEQ_P$-$WHO_{94}$ ranged from 0.07 ppt for Chandler Lake to 2.2 ppt for Canandaigua Lake, with a mean $TEQ_P$-$WHO_{98}$ of 0.53 ppt when nondetects were set to either zero or one-half the detection limit. Total CDD/CDF concentrations for these 11 lakes are presented in Table 3-17. Total CDD/CDFs ranged from 9.1 ppt for Chandler Lake, Alaska, to 2,916 ppt for Santeetlah Reservoir, North Carolina, with a mean of 926 ppt. Total PCBs ranged from 34 ppt for Chandler Lake, Alaska, to 2,116 ppt for Canandaigua Lake, new York, with a mean of 489 ppt. Table 3-17 also presents the estimated annual flux of CDD/CDFs to these lakes. Flux was calculated by multiplying sediment concentrations by lake-specific sedimentation rates ($g/cm^2$-yr), and dividing by lake-specific sediment focusing (redistribution) factors. CDD/CDF Flux ranged from 0.05 $pg/cm^2$-yr for Chandler Lake to 190 $pg/cm^2$-yr for Santeetlah Reservoir. PCB flux ranged from 0.19 $pg/cm^2$-yr for Chandler Lake to 103 $pg/cm^2$-yr for Santeetlah Reservoir. These data are considered to be the best available data for characterizing sediment CDD/CDF and dioxin-like PCB background concentrations in the United States because they are representative of sites not expected to be impacted from within several geographic regions.

### 3.5.2. European Data

This section presents a brief overview of some data on European sediments. It is not intended to be a comprehensive review of all available studies on European sediments. No attempt was made to characterize background levels of CDD/CDFs for Europe because of the variability and limited scope of this review.

Sediment samples from the vicinity of a magnesium production plant in Norway were analyzed for CDDs and CDFs (Oehme et al., 1989). The concentration distribution of CDD and CDF congeners was rather homogeneous, except for a slight decrease at a sampling station downstream of the plant and higher levels in deeper sediments (4-6 and 11-13 cm depth) at that site. TCDF congener profiles were the same as those for magnesium production. In addition, the PeCDF congener profiles were very similar to those found in the wastewater.

Trapped sediments from the archipelago of Stockholm, Sweden, displayed CDD and CDF congener distribution patterns that were very similar to those exhibited in total air and air particulates (Rappe and Kjeller, 1987). HpCDDs, OCDD, and HpCDF were the dominant congener groups in the sediment.

Bottom surface sediment samples collected from the Baltic Sea showed interesting CDD and CDF distribution patterns (Rappe et al., 1989b). Background samples, one between the Swedish and Soviet coasts and the other between the Swedish and Finnish coasts, contained similar levels and distribution profiles. The study indicated that the pattern of the TCDF congeners at these sites was typical of the "incineration pattern" (i.e., patterns resulting from MSW incineration, car exhausts, steel mills, etc.), which also had been found in samples of air and air particulates. However, sediment samples collected at a distance of 4 to 30 km from a pulp mill revealed a congener distribution pattern typical of bleaching mills. TCDFs found in the sediment 4 km from the pulp mill contained only two major congeners. The sediment collected 30 km from the mill displayed the same pattern.

Evaluation of sediments in Hamburg Harbor in Germany revealed high concentrations of the TCDDs through OCDD (mean concentrations of 564, 1112, 2744, 4040, and 7560 ppt, respectively) and the TCDFs through OCDF (mean concentrations of 526, 2980, 4106, 2358, and 2712 ppt) (Gotz and Schumacher, 1990). The average concentration of 2,3,7,8-TCDD was 375.3 ppt. High concentrations of 2,3,7,8-TCDD, especially in the Moorfleeter Canal and the Auserer Vering Canal, were attributed to discharges from an

organochlorine pesticide manufacturing plant. Patterns of 2,3,7,8-TCDD and the other HpCDD congeners are characteristic of the patterns resulting from the production of 2,4,5-T and 2,4,6-trichlorophenol. In addition, the pattern of the HpCDF congeners can be linked to emissions from thermal processes employed by chemical industries in the production of chlorinated organic chemicals. High concentrations of hepta- and octa-CDDs/CDFs may also be the result of other industrial combustion processes in the Hamburg area.

The Venice lagoon in the north of Italy, covering a surface of approximately 500 $km^2$ with a depth of $< 2$ m, has a limited water exchange with its surrounding area. The pollution loading sources includes the effluent of water streams and industrial and municipal waste water discharges, agricultural runoff, and an intense traffic of motorboats. Two sediment studies were conducted at this lagoon. Ramacci et al. (1998) used the data collected by the Ministry of Justice and the Ministry of Works to assess the CDD/CDFs contamination. The sediment samples were collected from various depths, including surficial to 60 cm. The dry weight I-$TEQ_{DF}$ concentrations for various depth were 16.0 pg/g for surficial sediment, 14.3 pg/g and 19.8 pg/g for sediment samples from a depth of 0-20 cm, 6.2 pg/g and 0.5 pg/g for sediment samples from a depth of 20-40 cm, and 6.0 pg/g for the sediment at the depth of 40-60 cm. Di Domenico et al. (1998) collected the data from the top 10-30 cm thick sediment layer of the Venice lagoon bottom in 1992 and 1995. For the area under industrial or prevailing industrial exposure, the concentrations of total CDD/CDFs ranged from 840 pg/g to 29,000 pg/g; the I-$TEQ_{DF}$ concentrations ranged from 12 pg/g to 570 pg/g; and the total PCB concentrations ranged from 53 ng/g to 720 ng/g. For the area exposed to municipal waste water discharges, the concentrations of total CDD/CDFs ranged from 210 pg/g to 1,400 pg/g; the I-$TEQ_{DF}$ concentrations ranged from 4.8 pg/g to 23 pg/g; and the total PCB concentrations ranged from 71 ng/g to 610 ng/g. The CDD/CDF congener profile for sediments from the Venice lagoon appeared to be strongly influenced by the waste water discharges. A prevalence of OCDD and OCDF over other congeners was observed (Ramacci et al., 1998).

Masahide et al. (1998b) examined sediment samples collected from a depth of 0-10 cm in the Districts of Gdansk, Szczecin, and Katowice in Poland in 1993-1994. The dry weight total PCB concentrations sampled in Northern Poland ranged from 1.7 ng/g to 630 ng/g with an average of 110 ng/g. The dry weight total PCB concentrations were 1.7-2.2 ng/g at the area without industrial activity, 78-99 ng/g at the places receiving untreated

municipal effluents, and 46-630 ng/g at the sites under the influence of human activities. The dry weight PCB concentrations sampled in Southern Poland ranged from 46 ng/g to 1,300 ng/g with an average of 540 ng/g. This highest concentration was sampled from a pond receiving effluent water from a coal mine in Katowice.

### 3.5.3. Vietnamese and Japanese Data

Schecter et al. (1989a) collected sediment samples from three rivers in Vietnam and analyzed them for CDD/CDF residues. Rivers included the Red River in the nonindustrialized North, the Saigon River in the industrialized South, and the Dong Nai River in an area sprayed with Agent Orange, a TCDD-contaminated herbicide. Results of these analyses are presented in Table 3-18. The average total concentrations of CDD/CDFs were 240 pg/g dry weight for the Red River, 6,800 pg/g dry weight for the Saigon River, and 1,200 pg/g dry weight for the Dong Nai River. Schecter et al. (1989a) suggested that the total CDD/CDF levels in these Vietnamese rivers is comparable to the total CDD/CDF levels observed in the lake sediments in the United States and Europe (Table 3-18).

In Japan, sediment samples were collected from a coastal area, river, and pond in Matsuyama between April 1995 and December 1997 (Seike et al., 1998). The dry weight total concentrations of CDD/CDFs ranged from 2.0 to 16 ng/g ($n = 3$) in the coastal area with an average of 8.1 ng/g, from 0.95 to 4.3 ng/g ($n = 22$) in river water with an average of 2.0 ng/g, and from 0.77 to 3.1 ng/g ($n = 3$) in pond water with an average of 2.3 ng/g. TCDD and OCDD were the main contributors to total CDD concentrations in all sediment samples, while CDF homologue compositions varied with the samples. Sediment samples were also collected in June-July 1993 at eight sites from upper, mid- and downstream of the Tama River, Japan, which was polluted by industrial and domestic wastewater (Onodera et al., 1998). Total CDD/CDF dry weight concentrations ranged from 27.0 to 231.6 pg/g with an average of 90.7 pg/g. The total I-$TEQ_{DF}$ concentrations ranged from 0.05 to 2.8 pg/g with an average of 1.2 pg/g.

### 3.5.4. Sediment Observations and Trends

Some general observations for CDD and CDF levels in the United States are possible from the data presented in the various sediment studies above:

- CDD and CDF congener distribution patterns in sediment generally follow those exhibited by the contaminant source.
- The concentration of CDD and CDF congeners in sediment generally increases with the degree of chlorination, but decreases uniformly with distance from the source.

### 3.5.5. Sediment CDD/CDF Profiles and Background TEQ Concentrations

CDD/CDF homologue group profiles for sediment were calculated as the mean homologue group concentrations divided by the mean total CDD/CDF concentrations. Congener profiles are the ratio of 2,3,7,8-substituted congeners to total CDD/CDFs in sediment. These congener profiles for sediment are presented in Figure 3-5. They are based on data from the recent EPA sediment core study, which evaluated sediment data from 11 non-source-impacted sites throughout the United States (i.e., 1 Alaska site, 3 New York sites, 1 North Carolina site, 1 Georgia site, 3 Utah sites, and 2 Washington sites) (Cleverly et al., 1996; Versar, 1996a). Only the uppermost sediment core samples (i.e., the most recently dated samples) were used in this analysis. The congener that accounts for the highest proportion of total CDD/CDFs is OCDD, with 1,2,3,4,6,7,8-HpCDD and OCDF also accounting for significant portions of total CDD/CDFs (Table 3-19). For the homologue group profile for sediment, OCDD and HpCDD account for the highest proportion of total CDD/CDFs.

Based on data from the uppermost sediment samples from EPA's recent sediment core study (Cleverly et al., 1996; Versar, 1996a) (n = 11), the mean background $TEQ_{DF}$-$WHO_{98}$ level was 5.3 ppt, assuming that nondetects equal half the detection limit (Table 3-20). When nondetects were set to zero, the mean TEQ was 4.7 ppt. These data were considered to be the most appropriate data set for characterizing background CDD/CDF TEQ concentrations, because they are representative of sites not expected to be impacted from various geographic locations in the United States. Thus, the "typical" background concentration in sediment is assumed to be 5.3 ppt $TEQ_{DF}$-$WHO_{98}$.

## 3.6. CONCENTRATIONS IN FISH AND SHELLFISH

Tables B-11 through B-13 (Appendix B) contain summaries of data from the numerous studies in the published literature regarding concentrations of CDDs, CDFs, and

dioxin-like PCB congeners in fish and shellfish. It should be noted that some studies reported fish concentrations on a whole weight basis and others reported concentrations for fish fillets. In the appendix tables and in the data used for calculation of background fish levels, whole weight concentrations were converted to fillet concentrations, assuming that the fillet contained one-half the concentration of the whole fish (U.S. EPA, 1990; Branson et al., 1985). This was necessary for estimating human exposures, because it is assumed that fish fillets, and not whole fish, are ingested by humans. In the following studies, summaries of CDD/CDF and PCB concentrations are presented as reported by the authors.

### 3.6.1. North American Data

A large quantity of fish data were collected as part of EPA's National Study of Chemical Residues of Fish (NSCRF), more commonly referred to as the National Bioaccumulation Study, during the period of 1986 to 1989 (U.S. EPA, 1992). Based on these data, several summaries were prepared and are presented here. Tables B-11 and B-12 include the dioxin and furan data collected as part of the National Bioaccumulation Study. Samples were collected from a wide variety of sites across the United States, including 314 sites thought to be influenced by point or nonpoint sources and 34 sites identified as relatively free of influence from point and nonpoint sources. This latter group of sites can be characterized as background per the definition used in this document. Background data are presented in Table 3-21. The mean I-$TEQ_{DF}$ concentration was 0.59 ppt for the background sites, when nondetects were set at zero and 1.2 ppt when non-detects were set to one-half the detection limit. For other sites, I-$TEQ_{DF}$ concentrations ranged from 0.7 ppt (POTWs) to 33.9 ppt (Superfund sites). Table B-13 includes similar data for the various PCB congener groups from 362 National Bioaccumulation Study sites. Because the specific PCB congeners could not be identified, it is not known what percentage of these concentrations represent the PCBs identified as dioxin-like. Of these sites, 20 were identified as background sites. The total PCB (all 209 congeners) mean concentration for these background sites was 46,900 ppt. Because the dioxin-like PCBs consist of only 11 of the 209 possible PCB congeners, it may be that they are a small percentage of the total; however, only congener specific analysis can ultimately confirm

this. As discussed at the end of this section (Section 3.6.4), this study was selected as the best basis for estimating background fish levels in the United States.

Fish muscle and hepatopancreas samples of striped bass, blue crabs, and lobsters collected from Newark Bay and the New York Bight all contained high levels (up to 6,200 ppt) of 2,3,7,8-chlorine substituted tetra- and penta-CDDs and CDFs (Rappe et al., 1991). Levels of 2,3,7,8-TCDD were higher than any other New Jersey samples, and the highest sample in this study (found in crab hepatopancreas) may be the highest level of 2,3,7,8-TCDD ever reported for aquatic animals. Crustaceans resembled one another in congener pattern. Specifically, they all contained both a large number and large amounts of CDD and CDF congeners in addition to the 2,3,7,8-chlorine substituted compounds. The striped bass samples, on the other hand, contained primarily the 2,3,7,8-chlorine substituted congeners.

Carp, catfish, striped bass, large mouth bass, and lake trout were collected from sites in the Hudson River and the Great Lakes Basin that were contaminated with industrial chemicals or contained known or suspected levels of PCBs (Gardner and White, 1990). The congener 2,3,7,8-TCDF was detected in 12 fish fillets at levels that ranged from 3 to 93 ppt. A 2,3,7,8-chlorine substituted PeCDF was detected in 14 fish fillets at levels ranging from 4 to 113 ppt. An interesting observation in this study was that 2,4,6-chlorine substituted CDFs were detected in four fish samples, suggesting that those fish may have been exposed to chlorinated phenols. The study indicated that the 2,4,6-chlorine substituted CDFs occurred in the fish at levels similar to those of the 2,3,7,8-chlorine substituted CDFs, but with less frequency.

Composited whole fish samples of lake trout or walleye collected from each of the Great Lakes and Lake St. Clair were analyzed for CDDs and CDFs (De Vault et al., 1989). CDF and CDD concentrations in lake trout were substantially different for each lake and between sites in Lake Michigan, probably reflecting differences in types and amounts of loadings to the lakes. In all of the sampling sites (except Lake Ontario), 2,3,7,8-TCDF was the dominant CDF congener in lake trout and ranged from 14.8 ppt in Lake Superior to 42.3 ppt in the whole fish samples of Lake Michigan. In Lake Ontario, the dominant congener in lake trout was a 2,3,7,8-chlorine substituted PeCDF. The distribution of CDF congeners in the Lake Erie walleye was very similar to that of the lake trout from Lake Superior. With regard to CDDs, the concentrations of 2,3,7,8-TCDD in the whole fish samples ranged from 1 ppt in Lake Superior to 48.9 ppt in Lake Ontario. With the

exception of Lake Ontario, the dominant CDD congener was a 2,3,7,8-chlorine substituted PeCDD. A 2,3,7,8-chlorine substituted HxCDD also contributed significantly to the total CDD concentrations. As with CDFs, the distribution of CDD congeners in the Lake Erie walleye was very similar to that of the lake trout from Lake Superior. Total I-$TEQ_{DF}$s for these samples ranged from 5.3 ppt to 67.0 ppt on a whole weight basis when nondetected congeners were set to one-half the detection limit.

In another study, CDDs and CDFs were measured in four species of salmonids (lake trout, coho salmon, rainbow trout, and brown trout) collected from Lake Ontario (Niimi and Oliver, 1989a). Levels of 2,3,7,8-TCDD in whole fish ranged from 6 to 20 ppt, and the HxCDD congener group was most dominant in all fish. High levels of OCDD also were detected in lake trout and coho salmon, but not in rainbow trout or brown trout. Although total CDF levels were about 25 percent lower than the total CDD concentrations, the levels of 2,3,7,8-TCDF (which was the dominant component of the TCDF congener group) were the same range as 2,3,7,8-TCDD (6 to 20 ppt). I-$TEQ_{DF}$ concentrations ranged from 7.3 ppt to 22.3 ppt on a whole weight basis, when nondetects were set to zero. The study suggested that, although collection sites can influence chemical levels and congener composition, comparisons of chemical levels and congener frequencies may not be suitable because of differences resulting from localized factors. The study also indicated that the importance of the various CDD and CDF congeners can differ with location (i.e., the same species of fish collected at different locations in a study area may reveal that the most common congener is different at each site).

Travis and Hattemer-Frey (1991) evaluated data generated as part of the National Dioxin Study regarding 2,3,7,8-TCDD concentrations in fish. Fish were collected from 304 urban sites in the vicinity of population centers or areas with known commercial fishing activity, including sites from the Great Lakes Region. Data from that study indicated that concentrations of 2,3,7,8-TCDD in whole fish from urban sites ranged from nondetectable to 85 ppt. In addition, only 29 percent of the fillets from urban sites had detectable levels of 2,3,7,8-TCDD, with a geometric mean concentration of 0.3 ppt. Whole fish samples from the Great Lakes Region had higher 2,3,7,8-TCDD levels than fish from urban areas (e.g., 80 percent vs. 35 percent detectable levels). In the Great Lakes Region, 2,3,7,8-TCDD concentrations in whole fish samples ranged from nondetectable to 24 ppt, with a geometric mean of 3.8 ppt. These levels were 10 times higher than the concentrations in

whole fish from urban areas. Likewise, the mean concentration of 2,3,7,8-TCDD in Great Lakes Region fish fillets (2.3 ppt) was about seven times higher than the levels in the fillets from urban areas (0.3 ppt). As with the whole fish samples, fish fillet samples from the Great Lakes Region had higher 2,3,7,8-TCDD levels than fillets from background urban areas (e.g., 67 percent vs. 29 percent detectable levels). Comparable levels of 2,3,7,8-TCDD were detected in whole bottom feeders and predators from the Great Lakes Region.

In an effort to determine whether incineration of municipal waste influenced CDD/CDF levels in the immediate area of waste incineration facilities, fish samples were collected from surface water near cities with, and without, operating incinerators throughout Connecticut. A total of 550 fish samples were collected between the years of 1987 and 1990. The total CDD/CDF concentrations for pre-operational and operational status were 28.44 pg/g and 58.38 pg/g, respectively, when nondetects were assumed to be one-half the detection limit (MRI, 1992).

Recently, Cooper et al. (1995) and Fiedler et al. (1997c) collected fish samples from grocery stores and local fish markets in southern Mississippi. All samples had detectable concentrations of CDD/CDFs. High I-$TEQ_{DF}$ concentrations were observed in farm-raised catfish nuggets (mean = 2.1 ppt/sample I-$TEQ_{DF}$) and in the parts of the crustacea containing the digestive gland (Cooper et al., 1995). The congener profile for the shellfish samples was similar to that observed for sediments collected in the same area and, reported sewage sludge patterns. For marine fish fillets (i.e., Spanish mackerel and mullet), the mean I-$TEQ_{DF}$ concentration was 0.27 ppt. The meat of marine shellfish (i.e., claw and body of blue crab and whole American oysters) had I-$TEQ_{DF}$ concentrations averaging 0.63 ppt, and freshwater shellfish (i.e., crawfish) had concentrations averaging 1.0 ppt. I-$TEQ_{DF}$ concentrations in fish and shellfish are presented in Table 3-22.

To further examine the high $TEQ_{DF}$ concentrations of the farm-raised catfish from southern Mississippi, Cooper et al. (1997) and Fiedler et al. (1998) performed a follow up to their 1995 study examining catfish feed and pond sediment along with catfish samples from the previously tested facility and other sites in the southeastern United States. The follow-up study also tested for PCB levels. Samples included three catfish fillets and three catfish nugget (i.e., small pieces of fillet) samples from the same store and distribution supplier as sampled in the previous study (Cooper et al., 1995), one catfish fillet from an Alabama supplier, three catfish fillets and one feed and pond sediment sample from a

different catfish farm in Mississippi, and two catfish and one catfish feed samples from a site in Arkansas. A summary of the results is presented in Table 3-23. Three farm-raised catfish fillet samples and three catfish nugget samples from Mississippi had lipid-based 2,3,7,8-TCDD levels ranging from 2.1 to 4.7 ppt and total lipid-based $TEQ_{DFP}$-$WHO_{94}$ ranging from 10.9 to 30.2 ppt. Catfish samples from Arkansas had lipid-based 2,3,7,8-TCDD levels ranging from 27 to 32 ppt and total lipid-based $TEQ_{DFP}$-$WHO_{94}$ ranging from 41.9 to 44.9 ppt. Similar results were observed in the feed samples. The feed from the Mississippi aquaculture facility, which supplied the food for the Mississippi catfish fillet and nugget samples, contained 2.7 ppt lipid 2,3,7,8-TCDD and a total lipid-based $TEQ_{DFP}$-$WHO_{94}$ concentration of 10.5 ppt, compared with feed levels from the Arkansas facility that contained 44 ppt lipid 2,3,7,8-TCDD, and a total lipid-based $TEQ_{DFP}$-$WHO_{94}$ concentration of 61. CDD/CDF congener profiles are also consistent between the catfish and the respective feed suppliers' products with the exception of OCDD and 1,2,3,4,6,7,8-HpCDD. Pond sediment congener profiles were not consistent with the profiles exhibited in the catfish samples, demonstrating significantly lower levels of most 2,3,7,8-substituted CDDs, and higher levels of 2,3,7,8-substituted CDFs. PCB analysis of these same catfish samples demonstrated that for all but one sample, the highest concentrations of PCBs were observed for congeners 153 and 138, respectively. PCB $TEQ_P$-$WHO_{94}$ in the catfish samples (both fillet and nuggets) ranged from 0.45 to 4.9 ppt lipid, with the PCB fraction of the total $TEQ_P$-$WHO_{94}$ ranging from 4 to 16 percent. The feed sample $TEQ_P$-$WHO_{94}$ level from the Mississippi site was 3.31 ppt lipid, while the level from the Arkansas site was 0.19 ppt lipid, and the pond sediment level contained a $TEQ_P$-$WHO_{94}$ level of 0.04 ppt lipid. Rappe et al. (1997c) continued this investigation by evaluating one combined catfish feed sample from Arkansas and its eight ingredients (i.e., soybean meal, meat and bone meal, wheat, corn, fish meal, cottonseed meal, and midds). The soybean meal had the highest I-$TEQ_{DF}$ concentration (i.e., 576 pg/g lipid). The 2,3,7,8-TCDD concentration in this ingredient was 370 pg/g lipid. The combined catfish feed sample had a I-$TEQ_{DF}$ concentration of 101 pg/g lipid and a 2,3,7,8-TCDD concentration of 67 pg/g lipid. Rappe et al. (1997c) suggested the ball clay anticaking agent in the soybean meal as the source of CDD/CDFs in this ingredient.

Schecter et al. (1997) analyzed samples of freshwater (n = 10) and marine fish (n = 13) collected from grocery stores in five U.S. cities (Binghamton, New York; Chicago,

Illinois; Louisville, Kentucky; Atlanta, Georgia; and San Diego, California). Whole weight mean I-$TEQ_{DF}$ values were 0.69 ppt for freshwater fish and 0.25 ppt for ocean fish, when nondetects were set to one-half the detection limit. Total mean CDD/CDF concentrations were 16.2 ppt (whole weight) for freshwater fish and 3.4 ppt (whole weight) for ocean fish, when nondetects were set to one-half the detection limit. Additional detail on the Schecter et al. (1997) study are presented in the section on concentrations in food products (Section 3.7).

Samples from all trophic levels in the Lake Ontario ecosystem were analyzed for PCB congeners (Oliver and Niimi, 1988). Analysis revealed that the PCB concentration increased from water to lower organisms to small fish to salmonids, demonstrating the classical biomagnification process. In addition, the chlorine content of the PCBs increased at the higher trophic levels. PCBs with the highest chlorine content (57 percent) were found in sculpin, small bottom-living fish that feed on benthic invertebrates. TrCBs and TCBs comprised a much higher percentage of the PCBs in the lower trophic levels than in salmonids and small fish. The percentage of PeCBs and OCPB in all samples was fairly uniform, but the HxCBs and HpCBs comprised a much larger fraction of the PCBs in the small fish and salmonids than in the lower trophic levels.

A study regarding the distribution of PCBs in Lake Ontario salmonids (brown trout, lake trout, rainbow trout, and coho salmon) showed that the PeCBs and HxCBs were dominant in all species (Niimi and Oliver, 1989b). The 10 most common PCB congeners represented about 52 percent of the total content and did not appear to be influenced by species or total concentration. Homologues observed averaged approximately 56 percent chlorine by weight in whole fish and muscle. The analysis of the chlorine content suggested that the more persistent congeners tend to behave as a homogeneous mixture instead of as individual congeners.

Petreas (1991) conducted a study to assess the influence of a bleached pulp and paper mill and other industrial facilities on the PCB congener levels in aquatic species and sediment in northern California. Petreas (1991) collected samples of local fish species, bivalves (freshwater clams that had been transplanted 2 months earlier), and sediment samples at sites upstream, downstream, and within the vicinity of a pulp and paper mill plant. Whole body fish samples were composited, or analyzed individually, based on size. These samples were analyzed for PCBs 77, 126, and 169. Results of an analysis of the

raw data from these samples are presented in Table 3-24. Levels of PCBs in fish tissue ranged from 1.2 pg/g for PCB congener 169 to 1,095 pg/g for PCB congener 77. Results of the fish tissue analysis according to sampling location indicated that "no special impact could be attributed to the pulp mill discharge" (Petreas, 1991). Bivalve concentrations ranged from 0.7 pg/g for PCB congener 169 to 102 pg/g for PCB congener 77. Concentrations of PCB congeners 77 and 126 were at least an order of magnitude higher in bivalves than in sediments from the same sampling location, indicating that these congeners bioconcentrate in the aquatic bivalve species evaluated. Bivalve and sediment impacts that could be attributed to facility discharges were not observed in this study.

Krahn et al. (1995) analyzed marine fish and invertebrates collected from several coastal sites of the northeastern United States for dioxin-like PCBs. Samples of winter flounder (muscle tissue), northern lobster (muscle tissue and hepatopancreas), and blue mussel (whole bodies) were analyzed for dioxin-like PCBs. Total mean PCB concentration ranged from 4 ppt to 351 ppt in muscle tissue from flounder and lobster, and blue mussel whole bodies. Total PCB concentrations in lobster hepatopancreas ranged from 764 to 32,800 ppt. Total mean $TEQ_P$-$WHO_{94}$ for these PCBs ranged from 0.1 ppt to 6.9 ppt for muscle tissue samples of winter flounder, 0.1 ppt to 3.7 ppt for whole blue mussels, and 0.1 ppt to 5.4 ppt for lobster muscle. Hepatopancreas tissue from lobster showed considerably higher $TEQ_P$-$WHO_{94}$ concentrations, ranging from 5.2 ppt to 1,820 ppt.

Mes and Weber (1989) analyzed a freshwater fish composite sample and a canned fish composite sample for PCBs 77, 126, and 169. Respective wet weight concentrations of these three PCBs were 36 ppt, 8 ppt, and $<1$ ppt for freshwater fish, and 8 ppt, 3 ppt, and $<1$ ppt for canned fish samples. In a later study, Mes et al. (1991) evaluated numerous PCB congeners including dioxin-like PCBs 105, 114, 118, 156, 157, and 189 in foods. A total of five composite freshwater fish, marine fish, and shellfish samples, each composite taken from major Canadian city, were analyzed for these PCBs. Total PCB congener residues for freshwater fish, marine fish, and shellfish were 31.9 ppt, 4.6 ppt, and 0.9 ng/g, respectively, on a wet weight basis, based only on congeners observed in three out of five composites. Schecter et al. (1997) also analyzed freshwater and marine fish samples for dioxin-like PCBs 77, 105, 114, 118, 126, 169, and 180. Concentrations of these congeners ranged from not detected to 1,800 ppt on a whole weight basis in fresh fish, and from 0.20 ppt to 320 ppt in ocean fish. The total whole weight $TEQ_P$-$WHO_{94}$ was

0.7 ppt for fresh fish and 0.2 for ocean fish, when nondetects were set to one-half the detection limit (Schecter et al., 1997).

### 3.6.2. European Data

Evaluation of fish in the Baltic Sea (Gulf of Bothnia) and northern Atlantic Ocean in the vicinity of Sweden revealed that concentrations of CDDs and CDFs in composited whole fish herring samples from the Atlantic Ocean were lower than those in the Gulf of Bothnia (Rappe et al., 1989b). Detectable levels of 2,3,7,8-TCDD in salmon muscle were found in both wild homing (4.6 to 19 ppt) and hatchery-reared (0.2 to 0.3 ppt) varieties in the Gulf of Bothnia. In addition, concentrations of the same representative congeners of the $Cl_5$ to $Cl_8$ CDD and CDF congener groups found in herring were found in both varieties of salmon. Levels of those congeners in the muscle of wild salmon, however, were five to ten times higher than the herring levels, while the levels in the hatched salmon essentially were the same as in the herring samples. Perch collected at a distance of 1-6 km from a pulp mill in the southern part of the Gulf of Bothnia contained 2,3,7,8-TCDD and 2,3,7,8-TCDF; the levels were higher in the samples collected closer to the pulp mill. These two compounds have been identified in bleaching effluents from pulp mills, as well as in bleached pulp. Arctic char collected from Lake Vattern, a popular fishing lake in southern Sweden, contained levels of 2,3,7,8-TCDD (6.5 to 25 ppt whole weight), 2,3,7,8-TCDF (21 to 75 ppt whole weight), and representative congeners of the PeCDD and PeCDF homologues. There was a good correlation between the weight of the fish and the levels of CDDs and CDFs. The main general pollution sources of the long, deep, narrow lake are two pulp mills.

Fish (cod, haddock, pole flounder, plaice, flounder, and eel), mussels, and edible shrimp from a fjord area contaminated by wastewater from a magnesium factory in Norway were analyzed for CDDs and CDFs (Oehme et al., 1989). Certain magnesium production processes can result in the formation of substantial amounts of CDDs and CDFs as byproducts. The congener pattern of tetra- and penta-chlorinated CDDs and CDFs released in wastewater during the magnesium production process is very characteristic and is dominated by congeners with chlorine in the positions 1,2,3,7 and/or 8. Fish and shellfish differ in their ability to bioconcentrate CDD and CDF congeners. For example, fish generally only concentrate the most toxic 2,3,7,8-substituted congeners; whereas, shellfish

can usually concentrate most of the congeners. Nearly all congeners were present in the shrimp and mussel samples. Although these organisms displayed the very characteristic PeCDF congener pattern of the magnesium production process, some deviations were found in the TCDF congener distribution within those species. For fish, the concentrations of CDDs and CDFs are dependent on the exposure level, fat content, living habit, and the species degree of movement. The highest CDD and CDF levels were found in comparatively high fat-content bottom fish collected close to the source. Cod and haddock (lower fat-content nonstationary fish) had much lower concentrations, even in the vicinity of the magnesium production factory. An interesting note is that the main stream of the fjord follows the west coast; subsequently, cod and eel samples collected along the west coast of the fjord had considerably higher levels of CDDs and CDFs than those collected from the eastern fjord entrance. Similarly, the level of 2,3,7,8-TCDD in mussels decreased by one order of magnitude from the vicinity of the magnesium production factory to the outer region of the fjord system.

Brown trout, grayling, barbel, carp, and chub collected in the Neckar River in southwest Germany contained much higher levels of 2,3,7,8-TCDF than in eels collected from the same river and the Rhine River (Frommberger, 1991). In addition, eels from both rivers showed very similar patterns for CDD and CDF congener distribution; whereas, the patterns of CDD and CDF distribution generally showed some degree of difference among the other fish collected from the Neckar River. Perch and bream collected from various locations in the vicinity of Hamburg Harbor, however, showed similar patterns in the distribution of the $Cl_4$ to $Cl_8$ CDD and CDF congener groups (Gotz and Schumacher, 1990). In general, the levels of CDFs were higher than the level of CDDs in these fish, especially with regard to the TCDFs to HxCDFs. Pooled samples of eels, collected at six different localities in The Netherlands, contained low levels of CDDs and CDFs, the major congeners of which were 2,3,7,8-chlorine substituted (Van den Berg, 1987). Concentrations of the various congeners identified in the eel samples ranged from 0.1 to 9.1 ppt. The sample with the highest concentration of 2,3,7,8-TCDD (9.1 ppt) was collected from Broekervaart in a location that was not far from a chemical waste dump that contained high concentrations of the same congener.

In 1992, Falandysx et al. (1997) examined congener-specific PCB data in fish from the southern part of the Baltic Sea. Eight fish species were collected from the Gulf of

Gdańsk near Gdynia. Total PCB levels were high in whole fish samples from this area, ranging from 1.4-million ppt lipid to 11-million ppt lipid. Measurements were performed for 94 PCB congeners. Predominant homologue groups were PeCBs and HxCBs, which constituted 33 to 46 percent and 36 to 46 percent of the total PCB concentrations, respectively. Of the five dioxin-like PCBs examined, levels were highest for PCB 118, and ranged from 160,000 ppt lipid in whole cod samples to 2-million ppt lipid in whole eelpout specimens. Also detected, but in lesser quantities, were TrCBs (0 to 0.9 percent), TCBs (5.5 to 11 percent), HpCBs (7.4 to 13 percent), OCBs (0.23 to 0.53 percent), nonachlorobiphenyls (0 to 0.5 percent), and aldo decachlorobiphenyl (< 0.01 to 0.01 percent). A large degree of variability in total PCB levels between species of fish was observed. Total PCB levels in the whole fish samples ranged from 1,400 ng/g lipid weight (47.6 ng/g wet weight) in cod to 11,000 ng/g lipid weight (48 ng/g wet weight) in pikeperch; 11,000 ng/g lipid weight (332.2 ng/g wet weight) in eelpout. The authors observed that with the relatively similar lipid content of the fish, which varies from 3.02 to 6.26 percent, the interspecies variability in the PCB congener pattern could possibly be attributed to variabilities in "enzyme activity, feeding behavior and trophic niche" (Falandysx et al., 1997).

Lulek et al. (1997) examined five random freshwater fish samples from five different Swiss and French lakes and rivers for 13 dioxin-like PCBs. The sampling locations included two from Lake Geneva, one each from the upper and lower Maggia River in Switzerland, and one from the Saône River in France. The analyzed fish included one each of three different species (i.e., bream, burbot, and arctic char) and two each of a fourth species (i.e., trout). Total PCB levels in the whole fish samples ranged from 1,137,000 ppt lipid (65,980 ppt wet weight) in the trout from the upper Maggia River to 3,235,000 ppt lipid (237,460 ppt wet weight) in the trout from the bottom of the Maggia River to 1,594,000 ppt lipid (254,430 wet weight) in the arctic char, to 2,179,000 ppt lipid (101,970 wet weight) in the burbot of Lake Geneva to 10,590,000 ppt lipid (402,420 ppt wet weight) in the bream of the Saône River. Congener profiles were similar in all species. Total $TEQ_P$-$WHO_{94}$ concentrations in the whole fish ranged from 104 ppt lipid in the burbot to 523 ppt lipid in the bream (Lulek et al., 1997).

In 1995, the Ministry of Agriculture, Fisheries and Food (MAFF, 1998) sampled trout farms throughout England and Wales. Forty samples, each consisting of several fillets of

muscle, were analyzed. When setting nondetects to the limit of detection, lipid-based CDD/CDF concentrations ranged from 2.1 to 13 ppt I-$TEQ_{DF}$ (mean 5.1). Wet weight concentrations ranged from 0.06 to 0.67 ppg I-$TEQ_{DF}$ (mean 0.24). Lipid based PCB concentrations varied from 8.9 to 51 ppt $TEQ_P$-$WHO_{94}$ (mean 19.0), and wet weight levels ranged from 0.22 to 2.4 ppt $TEQ_P$-$WHO_{94}$ (mean 0.87). Fat content in the samples ranged from 1.8 to 8.6 percent.

### 3.6.3. Fish Observations and Trends

Some general observations for CDD and CDF levels are possible from the data presented in the various fish and shellfish studies above:

- For fish, the concentrations of CDDs and CDFs are dependent on the exposure level, fat content, living habits, and the degree of movement of the species. Comparatively high fat-content bottom fish, collected close to the contaminant source, generally have the highest CDD/CDF levels; whereas, lower fat content, nonstationary fish have much lower concentrations, even in the vicinity of the contaminant source.
- The National Dioxin Study indicated that the levels of 2,3,7,8-TCDD in fish from the Great Lakes Region were higher than those from urban areas. Comparable levels were detected in whole bottom feeders and predators from the Great Lakes Region.
- With regard to PCBs, concentrations increase from water to lower organisms to small fish to salmonids, and the chlorine content of the PCBs increase at the higher trophic levels.

### 3.6.4. Fish CDD/CDF Profiles and Background TEQ Concentrations

CDD/CDF congener profiles for freshwater fish were generated using data for 10 freshwater fish samples collected from five U.S. States (i.e., New York, Illinois, Kentucky, California, and Georgia), as reported by Schecter et al. (1997). These data are different from those used to estimate the background I-$TEQ_{DF}$ concentration in freshwater fish (U.S. EPA, 1992). Limited congener-specific data were reported in U.S. EPA (1992); therefore, a profile could not be generated for the U.S. EPA (1992) data. The CDD/CDF profile for freshwater fish based on Schecter et al. (1997) was generated by setting nondetects to zero and calculating the ratio of individual congener concentrations to total 2,3,7,8-substituted CDD/CDFs. This was done for consistency with other food profiles presented in this chapter. For marine fish and shellfish, congener-specific data from Mississippi, as

reported in Fiedler et al. (1997c), were used. The profile for marine fish was based on a total of three samples, while the profile for shellfish was based on nine samples. Profiles were generated in the same manner as for freshwater fish. Profiles for freshwater fish, marine fish, and shellfish are presented in Table 3-25 and Figure 3-6. In general, CDDs account for a higher percentage of the total 2,3,7,8-substituted CDD/CDFs than CDFs. OCDD is the dominant congener for all three fish groups with 1,2,3,4,6,7,8-HpCDD accounting for the second highest percentage.

The background freshwater fish data collected as part of EPA's National Bioaccumulation Study (U.S. EPA, 1992) were selected as the best basis for identifying background levels in U.S. freshwater fish. A total of 60 fish samples were collected from fresh and estuarine water at 34 sites where no obvious industrial sources were present. The average I-$TEQ_{DF}$s were 0.59 ppt (assuming zero for the nondetects) and 1.2 ppt (assuming half the detection limit for the nondetects). In the original study, some of the samples were analyzed on a whole body basis and others on a fillet basis. However, for purposes of this document, whole body data were converted to a fillet basis. All concentrations were expressed on a wet weight basis. (This information is based on the report and clarifications provided via a personal communication from Annette Huber, Office of Water, to John Schaum, Office of Research and Development, May 17, 1993.) Several points should be considered in using these estimates to assess human exposure:

- The National Bioaccumulation Study data were derived from fresh and estuarine water fish and, therefore, are not representative of open ocean fish. Some types of open ocean fish, such as tuna and sword fish, are commonly eaten. The mullet and Spanish mackerel data from Cooper et al. (1995) are assumed to be representative of marine fish. These data showed a mean I-$TEQ_{DF}$ of 0.23 ppt in the fillet samples. Presumably, the contaminant levels in open ocean fish are lower than fresh water and estuarine fish, because they live in waters farther from dioxin sources.

- Whole fish contaminant levels are normally twice as high as fillets, which is generally considered the edible portion. However, some small fish and shell fish, such as clams, are typically eaten whole.

- These "background" samples may not be representative of what many individuals consume. U.S. EPA (1992) found an average of 11 ppt of I-$TEQ_{DF}$ across all 314 locations sampled. Even though these other locations were near industrial point sources, recreational or subsistence fishermen from local populations may consume fish from these waters.

- Market basket surveys would probably provide the best information on dioxin levels in fish commonly consumed by the general population. Data of this type were provided by Schecter et al. (1993a). This study analyzed five fish collected from a supermarket and found an average of 0.05 ppt of I-$TEQ_{DF}$. These data may mean that fish exposure levels are lower than the value selected here as representative of background levels (i.e., 1.2 ppt). However, only a limited number of samples were analyzed. Also the fish samples represent ocean species, whereas the National Bioaccumulation Study sampled freshwater and estuarine fish.

The background $TEQ_P$-$WHO_{98}$ concentration in freshwater fish was estimated to be 1.2 ppt, when nondetects were set to one-half the detection limit, based on data from Schecter et al. (1997), Mes and Weber (1989), and Mes et al. (1991). For marine fish and shellfish, background $TEQ_{DF}$-$WHO_{98}$ were estimated based on the data from Cooper et al. (1995). The background $TEQ_{DF}$-$WHO_{98}$ concentration was estimated to be 0.36 ppt for marine finfish (i.e., mullet and Spanish mackerel) and 0.79 ppt for marine shellfish (i.e., oyster and crabs), when nondetects were set to either one-half the detection limit or to zero. $TEQ_P$-$WHO_{98}$ for marine fish and shellfish were estimated to be 0.25 ppt and 0.04 ppt, respectively, when nondetects were set to one-half the detection limit, based on Mes and Weber (1989), Mes et al. (1991), and supermarket data from Schecter et al. (1997), as presented in Section 3.7.2.

## 3.7. CONCENTRATIONS IN FOOD PRODUCTS

Dietary intake is generally recognized as the primary source of human exposure to CDD/CDFs (Rappe, 1992). Several studies estimated that over 90 percent of the average daily exposure to CDD/CDFs are derived from foods (Rappe, 1992; Henry et al., 1992; Fürst et al., 1991). CDD/CDFs in fatty foods such as dairy, fish, and meat products are believed to be the major contributors to dietary exposures (Rappe, 1992; Henry et al., 1992). Travis and Hattemer-Frey (1991), using a fugacity model, estimated that the food chain, especially meat and dairy products, accounts for 99 percent of human exposure to 2,3,7,8-TCDD.

Analysis of trace levels of CDD and CDF congeners in food has been hindered in the past by lack of sensitive analytical detection methods, extraction difficulties from the high-lipid content food products in which these chemicals are most often found, and the presence of other potentially interfering organochlorine compounds. However, as the

analytical difficulties associated with detecting CDD and CDF congeners at ppt levels or lower (Firestone, 1991) were overcome, more food data began to be generated. In recent years, EPA, in association with the U.S. Department of Agriculture (USDA), has conducted several studies of dioxin-like compounds in foods. The results of these studies are presented in the following sections.

Tables B-14 and B-15 (Appendix B) contain summaries of data from the recent published literature regarding concentrations of CDDs and CDFs in food products. Most of the selected studies investigated "background" levels of CDDs and CDFs rather than studies targeted at areas of known contamination. Table B-16 contains a summary of PCB congener concentrations in food products.

Studies summarized in Tables B-14 and B-15 primarily examined CDD and CDF levels in products of animal origin (i.e., fish, meat, eggs, and dairy products). Because of their lipophilic nature, CDDs and CDFs are expected to accumulate in these food groups. Data in the tables indicate that CDDs and CDFs are found at levels ranging from the intermediate ppq up to the low ppt range. As expected, the highest levels reported are those measured in foods with high animal fat content. The highest reported congener concentrations are for the HpCDDs and OCDD. In general, for the less-chlorinated congener groups (i.e., $Cl_4$ - $Cl_6$), the CDD and CDF levels are both low and of similar magnitude. However, for the $Cl_7$ and $Cl_8$ congener groups, CDDs are higher than CDFs.

### 3.7.1. Migration of CDD/CDF from Paper Packaging Into Food

In the past, low levels of CDDs and CDFs have been detected in bleached paper. (See discussion in Volume II.) Because bleached paper is sometimes used for food packaging, concern has been expressed that CDD/CDFs may migrate from the paper into the food.

Using refined and highly sensitive analytical methods, LaFleur et al. (1990) observed the migration of 2,3,7,8-TCDD, 2,3,7,8-TCDF, and 1,2,7,8-TCDF from bleached paper milk cartons into whole milk. After 12 days of exposure, 6.7 percent of the 2,3,7,8-TCDD; 18 percent of the 2,3,7,8-TCDF; and 13 percent of the 1,2,7,8-TCDF in the milk carton leached into the milk. The concentrations of the three congeners in milk were 8.5, 110, and 49 pg/kg for 2,3,7,8-TCDD; 2,3,7,8-TCDF; and 1,2,7,8-TCDF, respectively. [Note: These data are not reported in Appendix B; only data for raw milk are reported.] LaFleur et

al. (1990) also analyzed a single background milk sample for 2,3,7,8-TCDD and 2,3,7,8-TCDF. The sample contained 2,3,7,8-TCDD at a concentration of 1.8 pg/kg and nondetectable concentrations of 2,3,7,8-TCDF.

Study results reported by LaFleur et al. (1990) were performed by the National Council of the Paper Industry for Air and Stream Improvement (NCASI) at the request of the U.S. Food and Drug Administration (FDA) as part of a cooperative Federal agency effort to assess the risks posed by dioxin contamination of paper products (i.e., the Federal Interagency Working Group on Dioxin-in-Paper). In addition to assessing the migration of CDDs and CDFs from milk cartons, studies were also conducted to assess the extent of CDD/CDF migration into food from coffee filters, cream cartons, orange juice cartons, paper cups for hot beverages, paper cups for soup, paper plates for hot foods, dual ovenable trays, and microwave popcorn bags. Migration of CDD/CDFs from the paper into food was observed in all studies. Results of these migration studies and an assessment of the risks to the general population posed by migration from paper are addressed in detail in U.S. EPA (1990). CDD/CDF levels currently found in food due to any leaching of dioxin-like compounds from paperboard containers are expected to be significantly lower than those reported in U.S. EPA (1990) because of process changes implemented by the pulp and paper industry to reduce formation of CDDs and CDFs (59 FR 17384).

In 1990, EPA referred the issue of potential CDD/CDF contamination from bleached food-contact paper products to FDA, because the risks were considered to be unreasonable in accordance with Section 9(a) of TSCA (55 FR 53047). In a 1994 Federal Register notice (59 FR 17384), FDA outlined various options being considered to address this issue, including the voluntary industry program to reduce TCDD in food-contact bleached paper products that had been in effect since 1990. As discussed in Volume II, the paper industry has made process changes that they expect have generally reduced dioxin levels in bleached paper pulp to less than 2 ppt of I-$TEQ_{DF}$. Similar or lower levels could be expected in final paper products. NCASI reports that essentially no detectable migration of dioxin to milk occurs from cartons at these levels. According to an industry-wide survey conducted in 1993 by the American Forest and Paper Association, the voluntary specification for 2 ppt or less TCDD has been met by industry (59 FR 17384). This standard was still being met in 1995 (personal communication between G. Schweer, Versar, Inc. and E. Machuga, FDA, October 5, 1995).

### 3.7.2. North American Food

Until recently, data on measured levels of CDDs, CDFs, and dioxin-like compounds in U.S. food products have generally come from studies of a specific food product(s) in a specific location(s) rather than from large survey studies designed to allow estimation of daily intake of the chemicals for a population. For example, CDD/CDFs are not routinely monitored in the U.S. FDA Surveillance Monitoring Program for domestic and imported foods (conversation between Dr. S. Page, FDA, and G. Huse, Versar, Inc., February 8, 1993) nor are they routinely monitored by the U.S. Department of Agriculture (USDA) in the National Meat and Poultry Residue Monitoring Program (conversation between Dr. E. A. Brown, USDA-FSIS, and G. Schweer, Versar, Inc., February 8, 1993).

However, USDA has developed some site-specific, though dated (late 1970s), CDD monitoring data, and recently, EPA and USDA conducted joint statistically-based national studies to evaluate the amount of CDD/CDF residues in animal products. Earlier efforts by USDA to examine CDD/CDFs in animal products were in response to a decline in general health noted by inspectors in several cattle herds in Michigan. Wood products in the local barns and other cattle holding premises, presumed to be treated with pentachlorophenol (PCP), were suspected as the cause of this health decline (Buttrill et al., 1978; Tiernan and Taylor, 1978). PCP was suspected to contain trace CDD and CDF levels as manufacturing contaminants at that time. In response to this incident, USDA performed two national investigations. The first study analyzed peritoneal adipose and liver samples collected from beef cattle in 23 States (Tiernan and Taylor, 1978), while the second study analyzed adipose tissue samples (body region not specified) collected from dairy cattle in 30 States (Buttrill et al., 1978)--neither study specified the cattle breeds for any sample. HxCDD, HpCDD, and OCDD were screened for in the analyses of samples from each study. In the beef cattle study (Tiernan and Taylor, 1978), 220 samples were analyzed: 189 peritoneal adipose samples and 31 liver samples. No residues were detected in any liver samples. A total of 19 (i.e., 10 percent) of the 189 adipose samples were found to contain HxCDD, HpCDD, or OCDD at levels $>0.10$ ppb (assumed to be on a whole weight basis), while 56 (i.e., 30 percent) contained levels $<0.10$ ppb (assumed to be on a whole weight basis) that were detectable based on the signal-to-noise ratio of the analytical instrumentation. OCDD accounted for the majority of the samples that positively contained CDDs (i.e., 17 or 9.0 percent), while only 3 samples contained HxCDD and 2 samples contained HpCDD

residues. In the dairy cattle study, 358 adipose samples were analyzed (Buttrill et al., 1978). Nine samples (i.e., 2.5 percent) positively contained CDD levels >0.19 ppb or the "level of reliable measurement," while another 30 samples (i.e., 8.4 percent) contained identifiable CDD levels that were below the "level of reliable measurement" (i.e., not positively identified due to low concentration levels). As with the beef cattle study results, OCDD accounted for the majority (eight) of positive samples. HpCDD was identified in only a single sample that also contained OCDD. HxCDD was also identified in only a single sample. Data from the USDA studies are not useful for estimating CDD/CDF exposure for two reasons. First, the samples were analyzed for only 3 of the 17 CDD/CDF congeners with dioxin-like toxicity, and these were reported on a homolog basis rather than a congener-specific basis. Second, the limit of detection was at or above 0.1 ppb or 100 ppt. Background levels for individual congeners appear to be much less than 100 ppt. For example, the highest congener levels in beef fat analyzed by Fürst et al. (1990) were 5.4 ppt for OCDD.

FDA has also conducted some limited analyses for the higher-chlorinated dioxins in market basket samples collected under FDA's Total Diet Program (Firestone et al., 1986). Food samples found to contain PCP residues >0.05 µg/g were analyzed for 1,2,3,4,6,7,8-HpCDD and OCDD. Also, selected samples of ground beef, chicken, pork, and eggs from the market basket survey were analyzed for these dioxin congeners, regardless of the results of PCP residue analysis. Between 1979 and 1984, 16 ground beef samples, 18 pork samples, 16 chicken samples, and 17 eggs samples with no PCP contamination were collected at various locations throughout the United States and analyzed for 1,2,3,4,6,7,8-HpCDD and OCDD. No dioxin residues were detected in any of the ground beef or egg samples. OCDD was observed at detectable concentrations in only 2 of the 18 pork samples (27 ppt 53 ppt) and 2 of the 16 chicken samples (29 ppt, 76 ppt). One chicken sample with PCP residues >0.05 µg/g had detectable residues of both 1,2,3,4,6,7,8-HpCDD (28 ppt) and OCDD (252 ppt). Egg samples from Houston, Texas, and Mesa, Arizona, with PCP residues >0.05 $\mu$g/g, had detectable 1,2,3,4,6,7,8-HpCDD levels ranging from 21 ppt to 588 ppt, and OCDD levels ranging 80 ppt to 1,610 ppt. These levels were attributed to local PCP contamination (Firestone et al., 1986). Milk samples, contaminated with PCP at levels ranging from 0.01 µg/g to 0.05 µg/g PCP, contained no detectable dioxins. It should be noted that these food residue data were not used in this

assessment of dioxin exposures in the United States, because the reported limits of detection (10 to 40 ppt) for the FDA analyses were considerably higher than the levels of dioxins observed in foods from more recent studies. Also, the study only analyzed for residues of 2 of the 17 toxic CDD/CDF congeners. Finally, the study focused on samples with PCP contamination and, therefore, was not generally representative of background exposures.

FDA conducted a market basket survey of dairy products and commercial fish and shellfish in 1995/96 (Bolger and Jensen, 1998). Analysis of the foods for CDD/CDFs demonstrated that, other than the catfish samples, few of the food products had quantifiable levels of CDD/CDFs below 1 ppt. Samples containing detectable levels below 1 ppt yielded uncertain results, highly dependent on what value nondetects were set to (i.e., zero, one-half the detection limit, or the detection limit). Consequently, the market basket survey results were not used in calculation of background estimates of CDD/CDFs. Catfish fillet samples did, however, show quantifiable results. Twelve of the 19 catfish samples were suspected of being linked to the use of ball clay as a feed additive. CDD/CDF levels in these fillets ranged from 1.20 to 5.66 ppt I-$TEQ_{DF}$ (mean = 3.11), when nondetects were set to one-half the detection limit and also when nondetects were set to the limit of detection. The seven uncontaminated catfish samples had levels ranging from 0.03 to 0.70 ppt I-$TEQ_{DF}$ (mean = 0.29), when nondetects were set to one-half the detection limit, and 0.05 to 0.71 ppt I-$TEQ_{DF}$ (mean = 0.31), when nondetects were set to the limit of detection.

Cooper et al. (1995) collected 38 samples of various food items from grocery stores and local fish markets in southern Mississippi during the spring of 1994. Food items "were selected based on their suspected high levels of CDD/CDF to the dietary intake." Thus, locally consumed dairy products, meat, egg, and seafood samples were collected, but items such as vegetables, fruit, grain, and cereal products were not sampled. All 38 samples collected had detectable levels of CDD/CDFs. I-$TEQ_{DF}$ concentrations for each sample are reported in Table 3-26. In general, the levels of CDD/CDFs in fish and shellfish were higher than the levels in meat and dairy products, and farm-raised catfish had the highest I-$TEQ_{DF}$ concentrations of all the food types analyzed. I-$TEQ_{DF}$ concentrations in meat and dairy products were slightly lower than those reported in other U.S. and European studies (Cooper et al., 1995).

As an extension of previous studies of food samples collected in southern Mississippi, Fiedler et al. (1997d) examined the CDD/CDF I-$TEQ_{DF}$ concentrations of seven restaurant-prepared food dishes. Samples included: a veal chop, chicken strips and fries, blackened amberjack fish fillet, seafood soup, pasta with cheese and cream sauce, cheese sticks, and cheese cake. I-$TEQ_{DF}$ values of CDD/CDFs ranged from 0.0197 ppt to 0.173 ppt on a fresh weight basis and from 0.128 ppt to 1.67 ppt on a lipid basis. The veal chop contained the highest I-$TEQ_{DF}$ levels with 1.67 ppt I-$TEQ_{DF}$ (lipid) and 0.173 ppt (whole weight), and also had measurable quantities of all the analyzed CDD/CDF congeners with the exception of 2,3,7,8-TCDD. 2,3,7,8-TCDD was not detected in quantifiable amounts in any of the restaurant samples. The major fraction of the total I-$TEQ_{DF}$ came from PeCDDs and HxCDDs in most of the samples. The authors observed similar congener patterns in the dairy-based dishes (cheese sticks, pasta with cream sauce, and cheese cake) and in the veal chop sample. Specifically, the ratios between the HxCDDs (1,2,3,4,7,8-, 1,2,3,6,7,8-, and 1,2,3,7,8,9-) were approximately 1:4:1, and the concentration of 1,2,3,7,8-PeCDD and 2,3,4,7,8-PeCDF was approximately the same in dairy-based and veal chop dishes. The authors speculated that a dairy diet for the source cattle would possibly explain the similar pattern in the veal chop and dairy-based dishes. I-$TEQ_{DF}$ levels in the restaurant-prepared seafood dishes were lower than I-$TEQ_{DF}$ levels of store purchased seafood items and farm-raised catfish items from southern Mississippi studied by Cooper et al. (1995). An I-$TEQ_{DF}$ level of 0.5 ppt (lipid) was found by Fiedler et al. (1997d) in restaurant-prepared fish fillet samples, while a mean I-$TEQ_{DF}$ level of 20.5 ppt (lipid) was observed in commercially bought fresh farm-raised catfish nuggets by Cooper et al. (1995) in the same area of southern Mississippi. The restaurant-prepared seafood soup had an I-$TEQ_{DF}$ of 0.646 ppt (lipid), while the I-$TEQ_{DF}$ levels of store purchased fresh crab and crawfish ranged from 5.23 to 40.1 ppt (lipid). Fiedler et al. (1997d) calculated the contribution of the seven restaurant food items to the percentage of daily intake. Based on an assumed average daily dietary CDD/CDF intake of 100 pg I-$TEQ_{DF}$/person, the veal chop would contribute 46 percent of the daily intake, the chicken strips 6.7 percent, the fish fillet 7.7 percent, the seafood soup 18 percent, the pasta with cream sauce 51 percent, the cheese sticks 13 percent, and the cheese cake 15 percent.

The California Air Resources Board (CARB) collected multiple samples of seven types of foods from commercial food sources in two urban areas of California (Stanley and Bauer,

1989). Foods were collected randomly, but an emphasis was placed on food stuffs of California origin (Stanley and Bauer, 1989). The types of food stuffs included saltwater fish, freshwater fish, beef, chicken, pork, milk, and eggs. A total of 210 samples were collected in Los Angeles (30 individual samples of each of the 7 types of foods), and 140 samples were collected in San Francisco (20 individual samples of each of the 7 types of foods). Food items were composited before chemical analysis to obtain a sample that was representative of average levels of CDDs and CDFs in the food stuffs, increase the probability of detection, and reduce the cost of chemical analysis. Samples were composited separately for each type of food stuff within each geographical area. Each composite sample contained 6 to 10 individual food samples, and 5 to 8 composite samples were analyzed for each food type. CARB data are summarized in Table 3-27. Beef (ground beef), pork (bacon), chicken, fish, and milk samples were analyzed on a lipid weight basis; however, for the purposes of this report, they were subsequently converted to a wet weight basis by multiplying the lipid weight concentration of CDD/CDFs by the fraction of fat contained in the food product of interest. Assumed lipid contents of 19, 15, and 4 percent for beef, pork and chicken, and milk were used. When nondetects were set to one-half the detection limit, the mean I-$TEQ_{DF}$s were 0.29 ppt, 0.24 ppt, 0.21 ppt, and 0.06 ppt for beef, pork, chicken, and milk, respectively. When nondetects were set to zero, I-$TEQ_{DF}$s were 0.03, 0.05, 0.08, and 0.02, respectively. Egg samples were analyzed for CDD/CDFs on a wet weight basis. I-$TEQ_{DF}$s for eggs were 0.14 ppt, when nondetects were set to one-half the detection limit and 0.004 when nondetects were set to zero.

The NCASI study (as described by LaFleur et al., 1990; and Henry et al., 1992) collected random food samples directly from the shelves of grocery stores located in the southern, midwestern, and northwestern regions of the United States. The samples were analyzed for 2,3,7,8-TCDD and 2,3,7,8-TCDF. These data are summarized in Table 3-28.

Schecter et al. (1993a) conducted complete congener analyses of 18 food samples collected from a supermarket in Binghamton, New York, in early 1990. Samples included five fish, three types of beef (ground beef, beef sirloin tip, and beef rib steak), one chicken drumstick, one porkchop, one lamb, one ham, one bologna, one heavy cream, and four types of cheese. The following ranges of I-$TEQ_{DF}$ levels on a whole weight basis were found: fish: 0.02 - 0.24 ppt; meat: 0.03 - 1.5 ppt; and dairy products: 0.04 - 0.7 ppt. These data are summarized in Table 3-29.

In a more recent study, Schecter et al. (1997) analyzed food samples collected directly from supermarkets in five U.S. cities: Binghamton, New York; Chicago, Illinois; Louisville, Kentucky; Atlanta, Georgia; and San Diego, California. The types of foods collected included: beef, chicken, ocean fish, fresh fish, and pork. Samples of each food type from the five geographic regions were pooled and analyzed for CDD/CDFs and selected PCBs. Ranges of CDD/CDF I-$TEQ_{DF}$ values were calculated for each food group by assigning either zero or the detection limit to undetected congeners. The following ranges of CDD/CDF I-$TEQ_{DF}$ levels on a lipid weight basis were found: beef: 0.89 - 2.86 ppt; chicken: 0.10 - 5.17 ppt; ocean fish: 2.45 - 21.14 ppt; fresh fish: 12.51 - 16.07 ppt; and pork: 0.64 - 3.97 ppt. These data are summarized in Table 3-30. Using the reported lipid content of these pooled samples to calculate the whole weight concentrations, the whole weight equivalents for these ranges are estimated to be: beef: 0.12 - 0.38 ppt; chicken: 0.005 - 0.28 ppt; ocean fish: 0.035 - 0.30 ppt; fresh fish: 0.60 - 0.78 ppt; and pork: 0.06 -0.36 ppt. PCB concentrations are also presented in Table 3-30.

Schecter and Li (1997) also analyzed four kinds of U.S. fast foods (i.e., hamburger, pizza, fried chicken, and ice cream) for CDD/CDFs and dioxin-like PCBs (105, 108, 156, 180). The I-$TEQ_{DF}$ concentrations were similar for hamburger, pizza, and chicken, ranging from approximately 0.01 ppt wet weight, when a value of zero was used for nondetected congeners, to 0.3 ppt wet weight, when the detection limit was used for nondetected congeners to calculate the total I-$TEQ_{DF}$ value. These values are similar to those observed for other raw food groups and may indicate that cooking does not have a significant effect on the levels of CDD/CDFs in foods. (Further discussion on the effects of cooking can be found in Section 3.7.5.) The I-$TEQ_{DF}$ concentrations for ice cream ranged from 0.03 to 0.49 ppt. Only PCB 180 was detected in hamburger at a concentration of 126 ppt. In pizza, PCB 118 (189 ppt) and PCB 180 (152 ppt) were detected, and in fried chicken, only PCB 118 (250 ppt) was detected. None of the PCB congeners evaluated were quantifiable in ice cream.

Schecter et al. (1989b) compared the levels of CDD/CDFs in cow's milk and infant formulas from Thailand and the United States. Samples of cow's milk and infant formulas were obtained from grocery stores in the Binghamton, New York, area in 1987. Thai formulas were purchased in Bangkok, Thailand, in 1986. In general, the I-$TEQ_{DF}$ levels, on a lipid basis, were lower in infant formula than in cow's milk (Table 3-31). In addition, the

formulas that were purchased in the United States had lower I-$TEQ_{DF}$ levels than those purchased in Thailand. On a sample weight basis, the I-$TEQ_{DF}$ level for whole cow's milk was 0.04 ppt (i.e., 1.2 lipid based ppt x 3.4 percent fat). This is similar to I-$TEQ_{DF}$ levels observed for cow's milk in other parts of the United States and in Europe. Using a Nordic model for calculating TEQs, the data for cow's milk and infant formulas were compared to the Nordic-TEQ levels found in human milk samples from various countries. Human milk from industrialized areas (i.e., United States, Germany, and South Vietnam) had higher CDD/CDF Nordic-TEQ levels than either cow's milk or soy-based infant formulas.

U.S. EPA (1991) collected milk samples from several sites in the vicinity of a municipal waste incinerator in Rutland, Vermont, and two background samples from a dairy farm 123 kilometers from the incinerator, where no obvious industrial sources of CDD/CDF were present. All samples were taken from bulk storage tanks at the farms. The report indicated that facility emissions could not be correlated with the levels of CDD/CDF and other contaminants measured in various environmental media. For all milk samples, the majority of the congeners were not detected. It was reported that only OCDD was consistently detected at levels from 0.2 to 2.4 pg/g in the farms near the facility. The levels in milk from the three farms near the facility ranged from about 0.2 to 0.4 pg of I-$TEQ_{DF}$/g whole milk, and the I-$TEQ_{DF}$ for the background samples collected from the distant farm was 0.12 pg/g. I-$TEQ_{DF}$s were calculated by U.S. EPA (1991) by setting the nondetects equal to the detection limit. The 0.12 ppt I-$TEQ_{DF}$ background value estimated by EPA is nearly 2 orders of magnitude higher than the I-$TEQ_{DF}$ for milk based on the NCASI data. (This is probably due largely to the incomplete congener analysis conducted by LaFleur et al., 1990.) Examination of the raw data supporting this study indicated that all of the CDD/CDF congeners in the background sample were nondetectable. Consequently, if nondetects are set to zero, the total background I-$TEQ_{DF}$ for milk would be zero. If half the detection limits are used to calculate the total I-$TEQ_{DF}$ level, the estimated value is 0.07.

Birmingham et al. (1989) analyzed CDD/CDF residues in food collected in Ontario, Canada. Most of the food was grown in Canada, although some was from the United States. They reported analyzing 25 composite samples from 10 food groups. The precise number of samples in each food group was not reported. No TCDD, PeCDD, HxCDD, TCDF, or PeCDF were found at detection limits of 0.1 to 7 ppt. Low ppt levels of some of

the higher chlorinated CDD/CDFs were detected in some foods. I-$TEQ_{DF}$ levels were also estimated for the major food groups. However, as shown in Table 3-32, these data were reported on a homolog basis. It is unclear what procedure was used to convert the homolog data to I-$TEQ_{DF}$. The text implies that nondetects were treated as zero for purposes of estimating I-$TEQ_{DF}$. In addition to the animal food data shown in Table 3-32, measurements were also made in potatoes, apples, tomatoes, peaches, and wheat. Only OCDD was detected at levels ranging from 0.6 to 8 pg/g fresh weight. The I-$TEQ_{DF}$ totals for vegetables were reported as 0.004 ppt for fruit, 0.002 ppt for vegetables, and 0.0007 ppt for wheat- based products. The procedure used to develop these I-$TEQ_{DF}$ estimates was not clear.

Canada has a food safety program that analyzes total diet samples (i.e., representative food samples of the general population) for chemical substances, The analyses are run on commercially bought and prepared foods. As a part of that program, Ryan et al. (1997) analyzed CDD/CDFs and non-ortho PCBs in 44 food composites from Toronto and also in 44 composite samples from Montreal during the summers of 1992 and 1993, respectively. To optimize specific congener detection, samples of primarily animal origin were analyzed. Food groups included beef (ground beef, beef steak, beef roast), cured pork, organ meat, and poultry, dairy products of varying lipid content (whole milk, 1 percent milk, cream, cheddar cheese, and butter), fish (fresh water and marine), and cooking fats and salad oils. Congener-specific data were not presented in this report. I-$TEQ_{DF}$ levels and $TEQ_P$-$WHO_{94}$ levels were calculated by setting nondetects to one-half the detection limit. These data were reported on a whole weight basis. TEQ values for each city are summarized in Table 3-33. The highest concentrations of total $TEQ_{DFP}$-$WHO_{94}$ were found in butter (0.93 ppt in Toronto and 0.62 ppt in Montreal), fresh water fish (0.62 ppt in Toronto and 0.48 ppt in Montreal), oils (0.44 ppt in Toronto and 0.31 ppt in Montreal), and in ground beef (0.39 ppt in Toronto and 0.37 ppt in Montreal). TEQs for dairy products increased with increasing lipid content. Butter (lipid content approximately 70 percent) had a total $TEQ_{DFP}$-$WHO_{94}$ of 0.93 ppt in Toronto and 0.62 in Montreal, while 1 percent milk (lipid content approximately 0.48 percent) had a $TEQ_{DFP}$-$WHO_{94}$ of 0.036 ppt in Toronto and 0.025 ppt in Montreal. Fresh water fish, with a total $TEQ_{DFP}$-$WHO_{94}$ of 0.62 ppt (Toronto) and 0.48 ppt (Montreal), were found to have higher total $TEQ_{DFP}$-$WHO_{94}$ levels than marine fish (0.28 ppt in Toronto, and 0.12 ppt in Montreal). PCBs constituted 58 percent of the

total $TEQ_{DFP}$-$WHO_{94}$ value in Toronto (0.36 ppt) and 67 percent in Montreal (0.32 ppt). In the meat food group, more of the total $TEQ_{DFP}$-$WHO_{94}$ was attributable to the CDD/CDF portion than the non-ortho PCB portion. The proportion of total $TEQ_{DFP}$-$WHO_{94}$ attributable to non-ortho PCBs was greater in the dairy products and fish than in the meat samples.

Mes and Weber (1989) analyzed one composite sample each of beef, butter, cheese, eggs, organ meats, and poultry for PCBs 77, 126, and 169. Whole weight concentrations of total PCBs (i.e., the sum of PCB 77, 126, and 169) were as follows: beef - 2 ppt, butter - 24 ppt, cheese - 11 ppt, eggs - 3 ppt, organ meats - 4 ppt, and poultry - 2 ppt. In a later study, Mes et al. (1991) evaluated numerous PCB congeners including dioxin-like PCBs 105, 114, 118, 156, 157, and 189 in foods. Five composite samples of the following food types were collected: milk, various dairy products, various cuts of beef and pork, lamb, organ meats, fish, eggs, luncheon meats, cooking fats, and soup. Each composite sample was taken from one of five major Canadian cities. Among these foods, total whole weight PCB concentrations were highest for freshwater (31.9 ng/g), canned (9.9 ng/g), and marine (4.6 ng/g) fish, followed by butter (3.0 ng/g) and cheese (1.7 ng/g). Concentrations were lowest (i.e., $<0.1$ to 0.1 ng/g) in canned meat soup, margarine, milk, yogurt, and lamb.

Recently, EPA has worked in cooperation with USDA to estimate the levels of CDD/CDFs and dioxin-like PCBs in U.S. food products. Analyses have been conducted for beef, pork, poultry, and milk products. A study of CDD/CDFs in vegetable oils has also been conducted. Data from the completed EPA/USDA studies are used in this Chapter to estimate background levels of CDD/CDFs and dioxin-like PCBs in foods. These studies were designed to be representative of the animal products used in the United States, and are believed to be suitable for calculating national averages.

***Beef***

EPA conducted a joint study with USDA to evaluate the amount of CDD/CDF residues in beef animals from federally inspected establishments (Winters et al., 1996a). Using a statistically-based sampling plan, 63 back fat samples were collected. Back fat was selected for sampling because: (1) it was assumed to be representative of fat people consume, because it is an extension of the fat reservoir, which, at another point, is the fat that is on rib cuts; (2) it was obtainable with little disruption by the USDA Federal

inspectors who collected the samples; and (3) it has high fat content, which would optimize the analytical capability of measuring dioxins in the matrix. The average fat content of the samples was 80 percent. The sampling plan was designed to provide samples representative of the slaughtering establishments, cattle class (i.e., bulls, steer, heifers, beef cows, and dairy cows) and region of the United States in order to provide a national estimate of CDD/CDFs in beef. Tissue samples were analyzed for the residues of the 17 toxic CDD/CDF congeners and for percent lipid content. Limits of detection for the study were 0.05 ppt for the tetra-CDD/CDFs, 0.5 ppt for the penta-, hexa-, and hepta-CDD/CDFs, and 3.0 ppt for OCDD/CDF, on a whole weight basis. Because the samples were 80 percent fat, these whole weight detection limits translate to average lipid-based detection limits of 0.0625 ppt for the tetra-CDD/CDFs, 0.625 ppt for the penta-, hexa-, and hepta-CDD/CDFs, and 3.75 ppt for OCDD/OCDF.

Based on the analytical results, the most frequently detected CDD/CDF congener was 1,2,3,4,6,7,8-HpCDD. This congener was detected in over 70 percent of the samples and most frequently had the highest concentration of all the CDD/CDF congeners. The second most frequently detected congener was 1,2,3,6,7,8-HxCDD, which was detected in approximately 32 percent of the samples. The most toxic congener, 2,3,7,8-TCDD, occurred in 11 of the 63 samples. Congeners not detected in any of the 63 beef samples included 2,3,7,8-TCDF, 1,2,3,7,8-PeCDF, 1,2,3,7,8,9-HxCDF, 1,2,3,4,7,8,9,-HpCDF, and OCDF. Of the 17 congeners, 1,2,3,4,6,7,8-HpCDD and OCDD had the highest mean concentrations for the 63 beef samples. Overall, total CDD concentrations were higher than CDF concentrations in 44 of the 45 samples that had detectable CDD/CDF concentrations. Of the 63 samples, 18 (i.e., 28.6 percent) had no detectable CDD/CDFs. When nondetects were set to zero, the mean total CDD concentration accounted for approximately 88 percent of the mean total CDD/CDF concentration, while total CDFs accounted for only 12 percent. When nondetects were set to one-half the detection limit, the mean total CDD concentration accounted for 70 percent of total CDD/CDF concentrations, and CDFs accounted for 30 percent. Based on the cattle classes, both I-$TEQ_{DF}$ and $TEQ_{DF}$-$WHO_{98}$ concentrations were highest in bulls and lowest in dairy cows.

The mean lipid-based $TEQ_{DF}$-$WHO_{98}$ value for the 63 beef samples was estimated to be 0.36 ppt (I-$TEQ_{DF}$ = 0.35 ppt), when nondetects were set to zero, and 1.06 ppt (I-$TEQ_{DF}$ = 0.89 ppt), when nondetects were set to one-half the detection limit. These mean values

were calculated by statistically extrapolating the sample size for each cattle class to the percentage of the U.S. food supply that they represent. CDDs made up almost 76 percent of the total $TEQ_{DF}$-$WHO_{98}$, while CDFs accounted for only about 24 percent, when nondetects were set to zero. When nondetects were set to one-half the detection limit, CDD concentrations accounted for 65 percent of the total $TEQ_{DF}$-$WHO_{98}$ and CDFs accounted for 35 percent of the $TEQ_{DF}$-$WHO_{98}$.

Assuming that the $TEQ_{DF}$-$WHO_{98}$ concentration in the lipid portion of the back fat samples is equivalent to the $TEQ_{DF}$-$WHO_{98}$ concentration in the lipid portion of edible cuts of beef, the lipid-based results from this study may be used to estimate the $TEQ_{DF}$-$WHO_{98}$ concentration of CDD/CDFs in beef that is consumed by the general population. For example, if it is assumed that the average fat content of edible cuts of beef is 19 percent, then the $TEQ_{DF}$-$WHO_{98}$ concentration in beef consumed by the general population is 0.20 ppt (i.e., 0.19 times 1.06 ppt), when nondetects are set to one-half the detection limit, and 0.07 ppt (i.e., 0.19 times 0.36), when nondetects are set to zero.

U.S. EPA, in cooperation with USDA, also recently analyzed beef samples for dioxin-like PCBs (Winters et al., 1996b; Saunders, 1997). The same samples that were analyzed for CDD/CDFs, as described above, were analyzed for PCBs 77, 118, 105, 126, 156, 157, and 169. Results of these analyses are presented in Table 3-34. Dioxin-like PCB congeners were found to be present in beef fat at an occurrence of greater than 85 percent; however, PCB 77 was found in only 19 percent of the samples. Using a statistical extrapolation of the data to account for the percentage of the different cattle classes in the U.S. food supply, the mean lipid-based total $TEQ_P$-$WHO_{98}$ concentration for these dioxin-like PCBs was estimated to be 0.49 ppt ($TEQ_P$-$WHO_{98}$ is also 0.49 ppt), when nondetects were set to either zero or the detection limit. PCB 126 contributed the most to the total $TEQ_P$-$WHO_{98}$. Assuming that the average fat content of edible cuts of beef is 19 percent, the PCB concentration in beef consumed by the general population was estimated to be 0.094 ppt $TEQ_P$-$WHO_{98}$(i.e., 0.49 times 0.19).

To ensure the relationship between lipid concentrations of dioxin-like compounds in the back fat of cattle is comparable to the level in edible meat products, EPA and USDA collaborated on an additional beef study examining the CDD/CDF/PCB concentrations in various cattle fat reservoirs (Lorber et al., 1997a). Fat matrices under examination included back fat (60 to 90 percent lipid), perirenal (i.e., kidney) fat (70 to 90 percent lipid), muscle

tissue (less than 5 percent lipid), and liver (less than 5 percent lipid). Three data sets, cited in Lorber et al. (1997a), were analyzed in the study. The first data set came from a 1995 study in which the 17 dioxin and furan congeners and 6 coplanar PCBs were measured in 5 tissue samples (i.e., back fat, muscle, liver, serum) from animals at 3 research facilities around the United States (Feil et al., 1995). The five selected animals came from research facilities at Pennsylvania State University (PSU), North Dakota State University (NDSU), and Oregon State University (OSU). These animals were raised under the same conditions as cattle raised in routine U.S. feedlot operations, and were slaughtered after about 1-1/2 years. The second data set came from a 1996 dosing study in which four animals were fed high amounts of several, but not all, of the dioxin and furan congeners (Feil et al., 1996). Dosed animals experienced unanticipated exposure to some higher chlorinated congeners that exceeded the dosing levels. The likely source of this exposure comes from the PCP-treated wood in the feeding facilities. Feil et al. (1996) reported the homologue group concentrations in back fat, perirenal fat, muscle tissue, serum, and liver. Lorber et al. (1997a) analyzed the unpublished congener-specific data from this study. The third data set came from a 1995 depletion study of CDD/CDFs in five animals from a herd in Bolsover, Derbyshire, England (Startin et al., 1994). The animals in this herd had very high CDD/CDF concentrations in milk, which was traced to locally contaminated feed.

Results reported below are based solely on analyses of the five animals in the first data set, because they were believed to be representative of a typical food source (i.e., raised under routine feedlot conditions). The animals in the other two studies were not used in the analysis because they experienced high levels of exposure. Examination of total and $TEQ_{DF}$-$WHO_{98}$ lipid-based concentrations in back fat compared with intramuscular fat concentrations, demonstrated that sampling for CDD/CDFs in back fat can be assumed to be representative of the levels found in edible fat. Assumptions used in this document to estimate CDD/CDF and PCB concentrations in edible beef from back fat are, therefore, believed valid. This finding is based on examination of the ratios derived by dividing TEQ and total CDD/CDF/PCB concentrations in back fat by the same levels in intramuscular fat as shown in Table 3-35. A ratio of 1.0 indicates that muscle and back fat concentrations are equal. The ratio of total CDD/CDF concentrations in intramuscular fat (ppt lipid) to the same level in back fat (ppt lipid) mostly ranged from 0.5 to 1.5 for the individual CDD/CDF congeners, and the $TEQ_{DF}$-$WHO_{98}$ ratios ranged from 0.6 to 1.7 (average of 0.9) (Lorber et

al., 1997a). PCB comparisons are less straightforward. PCBs 77, 118, and 106 contain total and $TEQ_P$-$WHO_{98}$ concentrations up to 16 times higher in muscle than in back fat. The ratios for PCBs 126, 156, 157, and 169, which ranged from 0.3 to 1.5, however, indicate that back fat levels are comparable to edible fat concentrations.

***Pork***

In addition to a national survey of CDD/CDFs and dioxin-like PCB residues in beef animals, EPA and USDA recently reported on the completion of a second survey of these compounds in pork (Lorber et al., 1997b). Using a statistically-based sampling plan, 78 belly fat samples were collected from 46 slaughtering establishments. The same justification for collection of back fat samples from beef animals applies to the collection of belly fat samples from the pork animals. These samples averaged 60 percent lipid (standard deviation of 12 percent), similar to the 80 percent lipid of the beef back fat samples. Tissue samples were analyzed for the 17 toxic CDD/CDF congeners and 7 coplanar PCBs, including PCBs 77, 118, 105, 126, 156, 157, and 169. Procedures for analysis were similar to the procedures for beef fat reported in Ferrario et al. (1996a) for CDD/CDFs and for PCBs in Ferrario et al. (1996b). Limits of detection for CDD/CDFs for the study were: 0.1 ppt for the tetras; 0.5 for the pentas, hexas, and hepta; and 1.0 ppt for the octas. Detection limits for the PCBs were: 1.5 ppt for PCB 77, 50.0 ppt for PCB 118, 26.0 for PCB 126, 10.0 for PCB 156, 2.5 ppt for PCB 157, and 0.1 ppt for PCB 169. These were detection limits for the sample matrix, and because the pork samples were about 60 percent lipid, the lipid-based detection limits can be estimated by dividing these detection limits by 0.60.

The sampling plan was designed to be representative of the pork class as a whole, and its three major subclasses: barrows/gilts, sows, and boars/stags. These classes are referred to by their common names: market hogs, sows, and boars, respectively. One major difference in the pork survey design as compared to the beef survey design was that the two minor classes of pork animal (i.e., the sows and boars) were oversampled in relation to their prevalence in the national slaughter of pork animals as a whole. In the beef survey, the number of animals sampled from each cattle class (which included bulls, steers, heifers, beef cows, and dairy cows) were proportional to their prevalence in the national slaughter, with one exception. The exception was the sampling of two bulls; whereas,

sampling in accordance to their prevalence in the class would have required only one sample. Results of the beef survey showed that the concentrations of CDD/CDFs in the bull were 3 to 10 times higher than the other four cattle classes (Winters et al., 1996b). However, it was difficult to draw conclusions and determine the variability in this class because of the small sample size. Based on this experience, an alternate strategy of oversampling the minor swine classes was adopted. This oversampling optimized the ability to distinguish concentration patterns among the three classes, and allowed for an estimate of the variability of the slaughter class estimates. Nationally, market hogs comprise about 95 percent of the total slaughter, with sows comprising about 4 percent and boars 1 percent of the slaughter. In the final sample of 78 animals, 56 were market hogs, 11 were sows, and 11 were boars. These classes represent 71.8 percent, 14.1 percent, 14.1 percent of the total sample size, respectively.

The most toxic congener, 2,3,7,8-TCDD, occurred in only 3 of the 78 samples. Congeners not detected in any of the 78 pork samples included 1,2,3,7,8-PeCDF; 1,2,3,7,8,9-HxCDF; and 1,2,3,4,7,8,9-HpCDF. Overall, CDD concentrations were higher than CDF concentrations. The four most frequently detected congeners were 1,2,3,4,6,7,8-HpCDD; OCDD; 1,2,3,4,6,7,8-HpCDF; and OCDF, all at a frequency of 50 to 60 percent detected. Results also indicated important differences among the swine classes. The $TEQ_{DF}$-$WHO_{98}$ concentration in sows appeared higher than market hogs: 1.85 ppt (I-$TEQ_{DF}$ = 1.72 ppt) for sows versus 1.44 ppt (I-$TEQ_{DF}$ = 1.26 ppt) for market hogs. This may be due to a longer life span for sows (i.e., >2 years) than for market hogs (i.e., <1 yr). With a longer life, sows accumulate more dioxins and have greater body burdens than market hogs, despite also having the dissipation route of milk excretion. Perhaps the most striking result, however, was for the boar class. While a very small class in terms of exposure (only 1 percent of the pork food supply), older boars were significantly different from all other classes, while younger boars were similar to the other pork classes. The older boars' lipid concentrations of 6.32 ppt $TEQ_{DF}$-$WHO_{98}$ (I-$TEQ_{DF}$ = 6.48 ppt) for CDD/CDFs and 0.54 ppt $TEQ_{DF}$-$WHO_{98}$ ($TEQ_{P}$-$WHO_{94}$ = 0.54 ppt) for coplanar PCBs were about 5 and 10 times higher than the overall averages for CDD/CDFs and coplanar PCBs, respectively. Like the sows, age is the principal factor that likely explains the higher concentrations. The average age of slaughter for market hogs is less than 1 year, while the old boars live longer than 2 years.

As shown in Table 3-36, the mean lipid-based $TEQ_{DF}$-$WHO_{98}$ value for the 78 pork fat samples was 1.48 ppt (I-$TEQ_{DF}$ = 1.30 ppt), when nondetects were set equal to one-half the detection limit, and 0.42 ppt (I-$TEQ_{DF}$ = 0.46 ppt), when nondetects were set to zero (Lorber et al., 1997b). Assuming the $TEQ_{DF}$-$WHO_{98}$ concentration in the lipid portion of belly fat samples is equivalent to the $TEQ_{DF}$-$WHO_{98}$ concentration in the lipid portion of edible cuts of pork, the lipid-based results of this study can be used to estimate the $TEQ_{DF}$-$WHO_{98}$ concentrations people are exposed to by eating pork. The average fat content of edible cuts of pork is assumed to be 15 percent. Thus, the average $TEQ_{DF}$-$WHO_{98}$ content of pork consumed by the general population would be 0.22 ppt (i.e., 1.48 ppt times 0.15), when nondetects are set to one-half the detection limit, and 0.06, when nondetects are set to zero. For PCBs, the mean lipid-based $TEQ_P$-$WHO_{98}$ was 0.06 ppt, when nondetects were set to one-half the detection limit, and 0.04 ppt, when nondetects were set to zero (the $TEQ_P$-$WHO_{94}$ are the same as the $TEQ_P$-$WHO_{98}$ for pork) (Table 3-36) (Lorber et al., 1997b). Assuming the average fat content of edible pork cuts is 15 percent, the $TEQ_P$-$WHO_{98}$ concentration in pork consumed by the general population is estimated to be 0.009 ppt (i.e., 0.06 ppt times 0.15), when nondetects are set to one-half the detection limit, and 0.006 ppt (i.e., 0.04 ppt times 0.15), when nondetects are set to zero.

*Poultry and Eggs*

EPA and USDA jointly participated in a study of dioxin-like compounds in the U.S. poultry supply (Ferrario et al., 1997). The study is a companion report to the cooperative studies on beef and pork (Winters et al., 1996a; Winters et al., 1996b; and Lorber et al., 1997b), and is the basis for the background TEQ concentrations for poultry. Using a statistically based sampling plan, 80 abdominal samples were collected from 70 U.S. slaughtering establishments. Abdominal fat was selected for sampling because it has a very high lipid content, thereby optimizing the analytic capability of detecting and quantifying dioxins in the samples. The average fat content of the samples was 80 percent. The sampling plan was designed to be representative of the four poultry classes: young chickens, light fowl, heavy fowl, and young turkeys. Nationally, young chickens account for about 95 percent of total poultry slaughter. In the final sample of 80 animals, 41 (51 percent) were young chickens, 12 (15 percent) were light fowl, 12 (15 percent) were heavy fowl, and 15 (19 percent) were young turkeys. Samples were analyzed for

percentage lipid, the same 17 toxic CDD/CDF congeners, and the same coplanar PCBs as the beef and pork samples, as discussed previously. Procedures for analysis of CDD/CDFs are described in Ferrario et al. (1996a) and for analysis of coplanar PCBs in Ferrario et al. (1996b). The detection limits for the study were 0.05 ppt for the tetra-CDD/CDFs; 0.25 ppt for the penta-, hexa-, and hepta-CDD/CDFs; and 0.5 ppt for the OCDD/CDF, on a whole weight basis. The detection limits for PCBs were: 0.80 ppt for PCB 77, 30.0 ppt for PCB 118, 10.0 ppt for PCB 105, 0.10 ppt for PCB 126, 4.0 ppt for PCB 156, 1.0 ppt for PCB 157, and 0.08 ppt for PCB 169. These were detection limits for the sample matrix, and because the poultry samples were about 80 percent lipid, the lipid-based detection limits can be estimated by dividing these detection limits by 0.80.

Laboratory analyses of the samples revealed that two young chicken samples could be classified as outliers by the Dixon and Grubs outlier tests. These two samples demonstrated significantly higher concentration levels of all the dioxin congeners, but CDF and PCB levels were comparable to the results from the other young chickens and other poultry classes. The results are reported below and also used to calculate national background levels; they do not include the data from these two young chicken samples.

The most toxic congener, 2,3,7,8-TCDD, occurred in 67 percent of the young chickens (mean 0.16 ppt lipid-based), 25 percent of the light fowl (mean 0.05 ppt lipid-based), 92 percent of the heavy fowl (mean 0.43 ppt lipid-based), and 73 percent of the young turkeys (mean 0.24 ppt lipid-based). No samples had detectable concentrations of 1,2,3,7,8,9-HxCDF. Ten percent of the young chicken samples had detectable levels of 1,2,3,4,7,8,9-HpCDF; none was detected in the other poultry classes. Overall, CDD concentrations were higher than CDF concentrations, and a larger percentage of the samples had detectable levels of CDDs than CDFs. The most frequently detected congeners were 1,2,3,4,6,7,8-HpCDD; OCDD; and 2,3,7,8-TCDF, at a detection frequency of 94, 73, and 84 percent, respectively.

The mean lipid-based $TEQ_{DF}$-$WHO_{98}$ value for the 78 poultry fat samples was 0.77 ppt (I-$TEQ_{DF}$ = 0.65 ppt), when nondetects were set to one-half the detection limit, and 0.48 ppt (I-$TEQ_{DF}$ = 0.42 ppt), when nondetects were set to zero. For PCBs, the mean lipid-based $TEQ_P$-$WHO_{98}$ (and $TEQ_P$-$WHO_{94}$) value for the same samples was 0.29 ppt, when nondetects were set to either one-half the detection limit or when nondetects were set to

zero. Table 3-37 presents a summary of the lipid-based $TEQ_{DF}$-$WHO_{98}$ results on the basis of poultry class.

Assuming the $TEQ_{DFP}$-$WHO_{98}$ concentration in the lipid portion of the abdominal fat samples is equivalent to the $TEQ_{DFP}$-$WHO_{98}$ concentration in the lipid portion of generally consumed poultry, the lipid-based results from this study may be used to estimate the $TEQ_{DFP}$-$WHO_{98}$ concentration in poultry consumed by the general population. For example, if it is assumed that the average fat content of poultry is 15 percent, then the $TEQ_{DF}$-$WHO_{98}$ concentration in poultry consumed by the general population would be 0.12 ppt (i.e., 0.15 times 0.77 ppt), when nondetects are set to one-half the detection limit, and 0.072 ppt (i.e., 0.15 times 0.48 ppt), when nondetects are set to zero. Using the same assumptions, the $TEQ_P$-$WHO_{98}$ concentration in poultry consumed by the general population is estimated to be 0.044 ppt (i.e., 0.15 times 0.29 ppt), when nondetects are set to zero and to one-half the detection limit.

Background $TEQ_{DF}$-$WHO_{98}$s for eggs are based on whole weight data from Cooper et al. (1995) and Fiedler et al. (1997c). Three egg samples were analyzed for CDD/CDFs. The estimated total $TEQ_{DF}$-$WHO_{98}$ for eggs is 0.032 ppt (I-$TEQ_{DF}$ = 0.026 ppt), using one-half the detection limit for nondetectable concentrations, and 0.023 ppt (I-$TEQ_{DF}$ = 0.017 ppt), using zero to represent nondetectable concentrations. Schecter et al. (1997) also analyzed eggs. However, many of the congeners were not detected at higher detection limits than those in the Cooper et al. (1995) and Fiedler et al. (1997c) studies. The whole weight I-$TEQ_{DF}$ concentration reported by Schecter et al. (1997) was 0.31 ppt, ten times higher than that reported in the other studies. Thus, the Schecter et al. (1997) data were not used in calculating the background estimate for CDD/CDFs. Background $TEQ_P$-$WHO_{98}$ concentrations were estimated to be 0.1 ppt in eggs ($TEQ_P$-$WHO_{98}$ was also 0.1 ppt), based on U.S. data from Schecter et al. (1997) and Canadian data from Mes and Weber (1989) and Mes et al. (1991). Schecter et al. (1997) analyzed egg samples (n = 18) for PCBs 118, 126, and 169. Mes and Weber (1989) analyzed one composite egg sample for PCBs 77, 126, and 169, and Mes et al. (1991) analyzed five composite egg samples for PCBs 105, 114, 118, 156, and 157.

*Milk and Milk Products*

Background $TEQ_{DF}$-$WHO_{98}$ concentrations in milk were based on data from a recent study conducted by EPA that utilized the Environmental Radiation Ambient Monitoring System (ERAMS) for collecting milk samples (Lorber et al., 1998b). ERAMS has 51 sampling stations in 41 of the 50 U.S. States, and Panama and Puerto Rico. Milk samples from these ERAMS stations collected during four time periods (i.e., April, July, and October 1996, and January 1997) were composited into duplicate composites samples (n = 8) and analyzed for the 2,3,7,8-CDD/CDF congeners and dioxin-like coplanar PCBs 77, 105, 118, 126, 156, 157, and 169 to generate national estimates of the CDD/CDF content of milk. In addition, samples from individual ERAMS stations, collected over the same time period, were analyzed to evaluate geographic or temporal variability in CDD/CDF/PCB concentrations. Composite samples had a mean $TEQ_{DF}$-$WHO_{98}$ content of 0.98 ppt (I-$TEQ_{DF}$ = 0.82 ppt), when nondetects were set to one-half the detection limit, and 0.97 ppt (I-$TEQ_{DF}$ = 0.81 ppt), when nondetects were set to zero. For PCBs, the lipid-based $TEQ_P$-$WHO_{98}$ concentration was 0.49 ppt ($TEQ_P$-$WHO_{94}$ was also 0.49 ppt), when nondetects were set to either one-half the detection limit or zero. These samples had a mean lipid content of 3.19 percent. Using this lipid content, the whole-weight $TEQ_{DF}$-$WHO_{98}$ concentration in milk, as consumed, would be 0.031 ppt (i.e., 0.0319 times 0.98 ppt) and the $TEQ_P$-$WHO_{98}$ concentration in milk, as consumed, would be 0.016 ppt (i.e., 0.0319 times 0.49 ppt). Congener-specific data for the eight composite samples are presented in Table 3-38. Little evidence of a temporal trend in $TEQ_{DF}$-$WHO_{98}$ concentrations was observed, based on the results of individual station samples. Results did, however, suggest a geographic trend with CDD/CDF concentrations in milk being highest in the southeastern United States and lowest in the southwestern United States. The $TEQ_{DF}$-$WHO_{98}$ estimates for U.S. milk obtained by this study are consistent with the levels observed in previous, more limited milk studies (Schecter et al., 1989b; U.S. EPA, 1991).

Additionally, some idea of the total $TEQ_{DF}$ level in milk samples can be gained by assuming that levels in beef fat are similar to levels in milk fat. This assumption implies that the differences in feeding/raising practices of dairy cattle vs. beef cattle do not cause substantial differences in CDD/CDF exposure. Beef contains approximately 20 percent fat, and whole milk is about 4 percent fat. Thus, on a whole food basis, CDD/CDF levels in beef should be about five times higher than in milk. Support for this concept can be seen

in the German data presented by Fürst et al. (1990, 1991). These data show that the I-$TEQ_{DF}$ level is 1.35 ppt in milk fat and 1.08 ppt in beef fat. On this basis, the North American data for beef (0.20 ppt of $TEQ_{DF}$-$WHO_{98}$) suggest that milk would be about 0.04 ppt of $TEQ_{DF}$-$WHO_{98}$. This value is consistent with the $TEQ_{DF}$-$WHO_{98}$ value obtained from the recent EPA study that utilized the ERAMS for sampling U.S. milk.

Whole weight $TEQ_{DFP}$-$WHO_{98}$ concentrations for dairy products (other than milk) were derived from the $TEQ_{DFP}$-$WHO_{98}$ concentrations in milk fat by assuming that the concentration of CDD/CDFs is the same in fat of dairy products as in milk fat. Whole weight $TEQ_{DFP}$-$WHO_{98}$ concentrations were calculated by multiplying the milk fat $TEQ_{DFP}$-$WHO_{98}$ concentrations by the fractional fat content of dairy products. However, because the dairy products category included a variety of food types (i.e., cheese, yogurt, milk-based desserts, etc.), it was first necessary to calculate a fractional fat content value that is representative of the percentage of fat in the diet, on average, that originates from dairy products other than milk. This composite fractional dairy fat value was based on dietary intake data from USDA's 1989-1991 Continuing Survey of Food Intake Among Individuals (CSFII) (USDA, 1995), and fat content data from USDA's Agricultural Handbook Number 8 (USDA, 1979-1984), as reported in EPA's Exposure Factors Handbook (U.S. EPA, 1997), as shown in Table 3-39. The composite percent of fat in dairy products was estimated to be approximately 12 percent (i.e., 11.84 percent). Thus, the whole weight $TEQ_{DF}$-$WHO_{98}$ concentration in dairy products was estimated to be 0.12 ppt (i.e., 0.98 ppt [milk fat concentration] times 0.12). The whole weight $TEQ_{P}$-$WHO_{98}$ concentration was estimated to be 0.058 ppt (i.e., 0.49 ppt [milk fat concentration] times 0.12).

These values are similar to lipid-based I-TEQ concentration estimates for dairy products based on data from Schecter et al. (1992a), Cooper et al. (1995), and Fiedler et al. (1997c). Schecter et al. (1992a) reported on the analysis of 2,3,7,8-substituted CDD/CDFs in U.S. dairy products. Cottage cheese, soft cream cheese, and American cheese samples were selected randomly from New York supermarkets and analyzed on a wet-weight basis. All dairy products sampled had at least 13 detectable congeners out of the 17 evaluated, and only one congener (1,2,3,7,8,9-HxCDF) was not detectable in any of the five dairy products. Whole weight I-$TEQ_{DF}$s ranged from 0.04 to 0.72 ppt, when nondetects were set to one-half the detection limit. Assuming a fat content of 25 percent for these cheeses, the lipid weight I-$TEQ_{DF}$ content would be 0.16 to 2.9 ppt. This lipid-

based concentration range brackets the mean milk fat concentration observed by Lorber et al. (1998b). Cooper et al. (1995) and Fiedler et al. (1997c) also reported on CDD/CDFs in three samples of cheddar cheese and three samples of butter. Lipid-based I-$TEQ_{DF}$ concentrations ranged from 0.70 to 0.97 ppt in butter and 0.74 to 0.86 ppt in cheddar cheese. In view of the similarities between the milk fat I-$TEQ_{DF}$ concentrations observed by Lorber et al. (1998b) and the lipid-based I-$TEQ_{DF}$ concentrations observed in dairy products, the use of the national milk fat data to estimate CDD/CDF concentrations in dairy products is a reasonable approach for estimating background concentrations of dioxin-like compounds for this food group.

*Fruits and Vegetables*

Data on CDDs and CDFs in U.S. fruit and vegetable products are extremely limited. The Ministry of the Environment, Ontario, conducted a study of CDDs and CDFs in locally produced and imported fruits and vegetables, some of which originated in the United States (Ministry of the Environment, 1988; Birmingham et al., 1989). Samples of fresh apples, peaches, potatoes, tomatoes, and wheat products were analyzed. In general, the minimum detection limits for these analyses were less than 1 ppt. The report indicated that "fruit and vegetable samples were substantially free of PCDD and PCDF residues, especially the more toxic tetra, penta, and hexachlorinated forms" (Ministry of the Environment, 1988). OCDD was the only congener detected in any of the samples. One apple and one peach sample contained detectable OCDD concentrations (8 ppt and 0.6 ppt, respectively). Detectable OCDD concentrations were found at concentrations ranging from 1 to 3 ppt in potatoes and 0.6 to 0.7 ppt in wheat samples. None of the tomato samples contained detectable levels of any CDD or CDF congeners. Based on these results, Birmingham et al. (1989) estimated the I-$TEQ_{DF}$s for fruits, vegetables, and wheat products to be 0.004 ppt, 0.002 ppt, and 0.0007 ppt, respectively.

As discussed in Volume IV, dioxin contamination of fruits and vegetables is thought to occur primarily via particle deposition or vapor adsorption onto outer layers with little penetration to inner portions. Plant uptake from the soil via the roots is generally considered negligible. However, the work of Hülster and Marschner (1993) indicates that zucchini and pumpkins were exceptions. For these plant species, it appears that root uptake occurs and leads to a uniform concentration within the fruit. The concentration of

CDDs and CDFs in zucchini squash grown on "uncontaminated" soil (0.4 ppt I-$TEQ_{DF}$ soil concentration) ranged from 0.5 to 0.7 ppt I-$TEQ_{DF}$ dry weight. These reported values may be converted to whole weight I-$TEQ_{DF}$ concentrations by using an assumed moisture content of 93.7 percent (USDA, 1979-1984). The resulting range of whole weight concentrations for zucchini is 0.03 to 0.04 ppt I-$TEQ_{DF}$. Müller et al. (1993) also evaluated CDDs and CDFs in vegetables (carrots, lettuce, and peas) grown at both contaminated plots and control plots. For the control plots, the highest levels of CDDs and CDFs were observed in carrot peels: 0.55 ppt I-$TEQ_{DF}$ dry weight, or 0.07 ppt I-$TEQ_{DF}$ whole weight, assuming a moisture content for carrots of 87.8 percent (USDA, 1979-1984). Lower concentrations were observed in samples from the cortex of the carrots, indicating that the "contamination source for the peel of carrots is the soil" (Müller et al., 1993). Lettuce concentrations ranged from 0.1 to 0.4 ppt I-$TEQ_{DF}$ dry weight. This is equivalent to a whole weight concentration range of 0.005 to 0.018 ppt I-$TEQ_{DF}$, assuming a moisture content of 95.4 percent for lettuce (USDA, 1979-1984). Concentrations in peas from contaminated plots ranged from 0.04 to 0.12 ppt I-$TEQ_{DF}$ dry weight (0.004 to 0.013 ppt I-$TEQ_{DF}$ whole weight, assuming a moisture content of 88.9 percent). Lower concentrations in peas (i.e., close to the detection limit; exact value not given) were reported for control plots. Similar data for vegetables grown in the United States were not available.

***Vegetable Oil***

High fat levels in vegetable oil suggest that it may be important to consider as a source of human exposure. Vegetable oils can be made from a variety of plants, including soybeans, corn, olives, peanuts, sunflower seeds, safflower seeds, linseed, and cotton seed. Many of these items are protected from atmospheric deposition, which implies that their CDD/CDF levels would be low. However, Theelen (1991) estimated that vegetable oil could contribute about 10 percent of a person's total daily intake in The Netherlands (14 of 120 pg I-TEQ/d). This estimate was based on the Fürst et al. (1990) study that found nondetects for most congeners, except some of the higher chlorinated congeners of CDD and CDF (detection limit = 0.5 ppt). Half the detection limit was used for the nondetects, and most of the congeners were not detected. Consequently, the actual value could be much lower.

Recently, EPA conducted a study to evaluate the levels of CDD/CDFs in vegetable fats and oils using an adaptation of EPA Method 8290 (Versar, 1996b; Schrock et al., 1996). A total of 30 oil samples collected from various geographical regions of the United States were analyzed for CDD/CDFs. Samples included soybean, corn, peanut, canola, olive, safflower, and sunflower oils, in addition to margarine, solid shortening, and canola oil spray. OCDD was the only analyte detected in all 30 oil samples above background levels found in method blanks. Concentrations of OCDD detected in the oil samples ranged from 3.6 to 33.1 pg/g compared to OCDD method blank levels of 2.8 to 4.4 pg/g. When subtracting out the appropriate method blank concentrations of OCDD from the vegetable oil samples, the range of concentrations was 0.2 to 30.3 ppt, with a mean of 5.6 ppt. Detection limits were generally near 1 pg/g for all analytes and ranged from 0.1 to >2 pg/g. None of the oil samples or blanks with detection limits ranging from 0.2 to 1.8 ppt showed 2,3,7,8-TCDD. Other than OCDD, all detections were at or near the detection limits. Because the occurrences of the CDD/CDFs in vegetable oil were near detection limits and there was only a small percentage of occurrences overall (not including OCDD), an average $TEQ_{DF}$-$WHO_{98}$ concentration calculated for the case where nondetects were set equal to 0 is evaluated as more meaningful than a $TEQ_{DF}$-$WHO_{98}$ concentration calculated for nondetects set equal to one-half detection limit. The mean $TEQ_{DF}$-$WHO_{98}$ calculated at ND = 0 was 0.056 ppt, and this value was used in calculating background exposures. By way of comparison, the mean $TEQ_{DF}$-$WHO_{98}$ calculated at ND = ½ detection limit was 1.5 ppt. The difference between the mean concentration calculated both ways is much larger than the difference seen for other food products. This suggests that the detection limits for the vegetable oil were too high to render a calculation of a mean at one-half the detection limit meaningful; for this reason, the mean calculated this way is not used in Chapter 4 for calculating background exposures.

$TEQ_P$-$WHO_{98}$ concentrations for PCBs were based on data from Mes et al. (1991). A total of five composite samples of cooking fats and salad oils were analyzed for PCBs. The total $TEQ_P$-$WHO_{98}$ concentration for these samples was 0.037 ppt, whole weight, based on the geometric mean of positive samples.

### 3.7.3. European Food

One of the most extensive investigations reported to date that involve testing of a variety of randomly selected food samples collected within the framework of official food control have been performed in the Federal Republic of Germany (Beck et al. 1989; Fürst et al., 1990; Malisch, 1998). Detailed results of these studies are included in Appendix B. Fürst et al. (1990) analyzed 107 food samples collected in Germany. The results of this study are presented in Table 3-40. All samples, except some of the milk, were randomly collected during official food monitoring programs. The authors speculated that a source may have been near the areas where the milk samples were collected, because they appeared higher than other milk tested in Germany, which showed levels around 1 ppt I-$TEQ_{DF}$. In a later report, Fürst et al. (1991) reported that a much larger survey of dairies in Germany had been completed. This survey analyzed 168 samples of milk and milk products collected at dairies prior to bottling in 1990. They found an arithmetic mean of 1.35 pg of I-$TEQ_{DF}$/g of fat. I-$TEQ_{DF}$s in these studies were estimated by assuming that nondetects equaled half the detection limits. Except for 1,2,3,7,8,9-HxCDF and 1,2,3,4,7,8,9-HpCDF, the 2,3,7,8-substituted congeners were detected at a frequency of greater than 99 percent in these samples (Fürst, 1995). In a more recent study, Fürst and Wilmers (1995) compared the levels of CDD/CDFs found in dairy products from 1990 to the levels in 120 dairy samples collected in 1994. Over the 4-year period, mean I-$TEQ_{DF}$ concentration in milk fat decreased by almost 25 percent from 1.35 ppt to 1.02 ppt. Similar reductions were noted in human milk fat (Fürst and Wilmers, 1995). Fürst et al. (1991) also provided a summary of the results of several European studies. The data summaries relevant to background levels in meat and dairy products from Fürst et al. (1991) are presented in Table 3-41. Fürst et al. (1991) report that information on CDD and CDF levels in vegetables and fruits is scarce and that the available data indicate a background of below 1 ppt.

Beck et al. (1989) analyzed 12 food samples collected randomly from food markets in West Berlin, Germany. Chicken, eggs, butter, pork, ocean perch, cod, herring, vegetable oil, cauliflower, lettuce, cherries, and apples were analyzed for CDD/CDFs. CDD/CDFs were detected in samples of animal origin in the ppq to ppt range (fat weight basis). No CDD/CDF congeners were detected at a detection limit of 0.01 ppt (whole weight basis) in samples of plant origin. Mayer (1995) analyzed 27 bulk milk samples (i.e., background)

collected from large dairies in Bavaria, Germany, and 160 milk samples from farms in the vicinity of suspected dioxin sources between 1989 and 1993. Background I-$TEQ_{DF}$ concentrations ranged from 0.69 ppt to 1.12 ppt, with a mean of 0.9 ppt on a lipid basis. Nondetected congeners were assumed to equal one-half the detection limit. Few of the potentially impacted samples had I-$TEQ_{DF}$ concentrations exceeding 5 ppt. Malisch et al. (1994) analyzed one background egg sample and one egg sample from a contaminated site in Germany. The mean I-$TEQ_{DF}$ concentrations were 1.2 pg/g fat for the background site and 12.7 pg/g fat for the contaminated site. These results are based on analyses using four different analytical methods, which showed similar results.

Malisch (1998) followed up with a more recent study of intake of food in Germany. In this study, CDD/CDF levels in food from the southwestern part of Germany were measured between 1993 and 1996. Malisch (1998) analyzed 1,414 food samples for CDD/CDF concentrations. The results indicated that the more recent I-$TEQ_{DF}$ concentrations are lower than those previously observed by Fürst et al. (1991) and Fürst et al. (1990) (Table 4-42).

Schmid and Schlatter (1992) found low background levels of CDD/CDFs in milk samples from Switzerland. A total of 28 cow's milk samples and 1 goat's milk sample were collected during 1990 and 1991 from industrial dairies and from both rural alpine sites and potentially impacted sites (i.e., highly industrialized areas, and areas with waste incineration and metal recycling). Due to insufficient analytical sensitivity, 2,3,7,8-TCDD was not detected; thus, an assumed concentration was used for 2,3,7,8-TCDD in calculating I-$TEQ_{DF}$s. The lowest I-$TEQ_{DF}$ concentrations were observed in samples from rural and alpine areas. The mean lipid-based I-$TEQ_{DF}$ for these seven milk samples ranged from 0.70 ppt to 3.28 ppt. The goat milk sample had a I-$TEQ_{DF}$ concentration of 0.88 ppt. Based on pooled milk samples from nine industrial dairies in Switzerland, the average lipid-based I-$TEQ_{DF}$ was 1.31 ppt. This is approximately equivalent to a whole weight I-$TEQ_{DF}$ concentration of 0.05 ppt, assuming a lipid content of 4 percent for these samples. No significant differences were observed between samples stored in cardboard containers and those stored in glass bottles. Higher I-$TEQ_{DF}$ concentrations were observed in samples collected from potentially impacted sites (2.02 ppt to 4.85 ppt on a lipid basis).

In 1996, the French Ministry of Agriculture, Fisheries and Food Products (FMAFFP) (1997) undertook a study to investigate "background" levels of CDD/CDFs in the French

Republic (FMAFFP, 1997; Defour et al., 1997). This followed a 1993/94 FMAFFP study examining dioxin levels found in French milk samples collected from areas near known polluting industries. In this 1996 study, 40 dairy products, including cheese, butter, milk-based desserts, and cream, were sampled from 34 regions of France. Also, 12 cow's milk samples were collected from different locations in two regions (i.e., Seine-Maritime and Pas-de-Calais). These two regions had shown elevated levels over other regions of the country in the previous study, which targeted areas near pollution sources. The seven milk samples from the Seine-Maritime region had a mean lipid-based I-$TEQ_{DF}$ of 1.77 ppt. The five milk samples from the Pas-de-Calais region had a mean I-$TEQ_{DF}$ of 2.13 ppt. Each of these regions had one sample with a I-$TEQ_{DF}$ greater than 3.0 ppt; whereas, the remaining samples clustered together at levels around 1.5 ppt and 1.9 ppt, respectively. The congener profiles of the milk samples showed a predominance of OCDD, and the furans typically were higher than the dioxins. Table 3-43 presents the mean I-$TEQ_{DF}$ levels of the dairy products tested by the FMAFFP. The lipid-based I-$TEQ_{DF}$ values of the eight butter samples ranged from 0.51 ppt to 2.10 ppt (mean of 1.01 ppt). The lipid-based I-$TEQ_{DF}$ values of the 20 cheese samples ranged from 0.54 ppt to 1.44 ppt (mean of 1.11 ppt). The 12 fresh cream and milk-based desserts had lipid-based I-$TEQ_{DF}$ values ranging from 0.78 ppt to 3.15 ppt (mean of 1.34 ppt). The sample from the Nord region had an I-$TEQ_{DF}$ value of 3.15 ppt, which was significantly higher than the other cream or milk-based dessert samples, possibly reflecting industrial sources in the region. The mean value of the remaining 11 samples was 1.18 ppt I-$TEQ_{DF}$.

Theelen et al. (1993) collected food products from various locations in The Netherlands and analyzed them for 2,3,7,8-chlorine substituted dioxins, furans, and planar PCBs. Meat samples were collected from slaughter houses throughout The Netherlands. Fish, mixed meats, and cheeses were gathered at various grocery stores. Mixtures of foods in these categories were prepared based on the proportion of the average annual consumption rate that different food items in these categories represented. The food industry provided purified oils and fats. Mixtures of these items were also prepared in proportion to their annual use in The Netherlands. The concentrations of CDD/CDFs in these food products are presented in Table 3-44.

Food samples were collected in 1996 from both local markets and supermarkets from Catalonia, Spain (Domingo et al., 1999). A total of 35 food samples were collected and

analyzed for CDD/CDF concentrations. The food samples included various types of beef, pork, chicken, lamb, fish, seafood, canned fish, milk and dairy products, vegetables, cereals, fruits, fats and oils, and eggs. The lipid-based and wet-weight I-$TEQ_{DF}$ concentrations in these foods are presented in Table 4-45. As shown in Table 4-45, both the lipid-weight and wet-weight I-$TEQ_{DF}$ concentrations were highest in fish and seafood. It is interesting to note that reported concentrations of CDD/CDFs in fruits, vegetables, and cereals were similar to those observed for meat and dairy products. However, the number of samples collected from each food group was not reported, and the method used for treating non-detects in calculating total I-$TEQ_{DF}$s was not reported.

CDDs and CDFs have been studied in dairy products in Spain. Ramos et al. (1999) analyzed butter samples for CDD/CDFs/PCBs and estimated $TEQ_{DFP}$-$WHO_{94}$s. Eight of the best known brands of butter were purchased from Spanish supermarkets. A total of 21 samples were analyzed. The results of the study indicated that the I-TEQ averages were 0.41 ppt for CDDs, 0.70 ppt for CDFs, and the $TEQ_P$-$WHO_{94}$ average was 0.09 ppt for PCBs. The most toxic CDD congener, 2,3,7,8-TCDD, was found at detectable levels in 15 of the 21 samples analyzed and the most toxic CDF congener, 2,3,7,8-TCDF, was found in all samples. Ramos et al. (1999) did not report whether non-detects were set to zero or one-half the detection limit in calculating TEQs.

In the early 1990s, the United Kingdom's Ministry of Agriculture, Fisheries, and Food (MAFF) conducted a survey of CDD/CDFs in foods collected as part of their Total Diet Study (MAFF, 1992). Food samples were collected in 1988 from two UK locations: Port Talbot and Stonehaven, selected to represent an urban/industrial site and a rural site, respectively. Additional samples were collected in Norwich. Selected food items included: meat and meat products, milk products, fish, fats and oils, eggs, and fruits and vegetables. I-$TEQ_{DF}$s were calculated by setting nondetects to either zero or to the limit of detection to provide a range of possible I-$TEQ_{DF}$ values for each food item. Results of these analyses are presented in Table 3-46 in terms of total I-$TEQ_{DF}$s. Higher I-$TEQ_{DF}$ concentrations were found in fatty food products, such as fish, meats, and fats and oils, than in food items with lower fat contents, such as fruits and vegetables. For some food items, higher I-$TEQ_{DF}$ concentrations were observed at the urban/industrial area than at the rural site. Milk samples were collected from farms in rural/remote areas that were not expected to be impacted and from farms closer to urban/industrialized areas. In addition, retail milk samples were collected in

1990 from several UK locations during both winter and summer months. The results of these analyses are presented in Table 3-47. After laboratory procedures were developed for the fractionation of ortho and nonortho substituted chlorobiphenyls, Kroskos et al. (1996) analyzed frozen aliquots of the same 1990 milk samples for PCB congeners. The mean concentration for seven congeners (PCBs 28, 52, 101, 118, 138, 153, and 180) was 0.26 $\mu$g/kg whole milk or 6.7 $\mu$g/kg on a lipid basis assuming a typical fat content of 3.9 percent for cow's milk. The mean $TEQ_P$-$WHO_{94}$ concentration of PCBs 77, 118, 126, 169, and 180 in the milk samples was 0.06 ppt whole milk. PCB 118 and 126 accounted for 98 percent of the total PCB TEQ (Kroskos et al., 1996). Additional MAFF analyses of Total Diet Study samples examining temporal trends in dietary intake are summarized in Chapter 6.

Foxall et al. (1995) analyzed samples of fruits and vegetables from urban and rural areas in Wales and England for CDD/CDF and PCB residues. The study was initiated as a result of concerns over elevated CDD/CDF and PCB concentrations in the air and soil in the vicinity of a chemical waste incinerator. Samples were collected from gardens at five sites within a 1.5-mile radius of the incinerator and from five similar sites in three rural areas for comparison. The produce evaluated included apples, courgettes, lettuce, and potatoes. Median I-$TEQ_{DF}$ concentrations ranged from 0.3 ppt to 0.4 ppt for the four fruit and vegetable products, when nondetects were set to the limit of detection. In addition, no significant differences were observed between CDD/CDF and PCB concentrations in produce taken from urban and rural sites. The authors noted that CDD/CDF concentrations in produce directly exposed to atmospheric deposition (i.e., apples and lettuces) are not significantly different from root vegetables (Foxall et al., 1995).

The Ministry of Agriculture, Fisheries, and Food (MAFF) also analyzed commercially available cow's milk for dioxins and PCBs in samples collected in 1995 from 12 locations in England (MAFF, 1997a). The locations were chosen to be representative of the different regions. Full fat milk purchased in glass bottles was tested. Lipid-based CDD/CDF concentrations ranged from 0.67 to 1.4 ppt I-$TEQ_{DF}$. Lipid-based PCB levels ranged from 0.75 to 2.3 ppt $TEQ_P$-$WHO_{94}$. Concentration levels of dioxins found in the 1995 samples were lower than those found in a comparable MAFF survey conducted on milk collected during 1990 (MAFF, 1992). (See Section 6.5, Temporal Trends in Food Products, for details.)

In a Lancaster University study, Stewart and Jones (1996) also examined PCBs in cow's milk from rural and urban dairy farms in the northwest of England. Sites were chosen to be representative of farms providing milk for human consumption. Stewart and Jones (1996) sampled pooled milk taken from 10 herds between 1993 and 1994. The sum of the lipid based levels of PCBs 77, 105, 118, 126, 156, 169, 170, and 180 ranged from 1.2 to 2.1 ppt $TEQ_P$-$WHO_{94}$.

The Ministry (MAFF) has also studied dioxin levels in samples of cow's milk from individual farms located around known emission sources in the United Kingdom annually since 1993. Additionally, beginning in 1994, the samples were analyzed for PCBs. Between 1993 and 1995, MAFF collected samples from 93 farms in the vicinity of 29 industrial sites (MAFF, 1997b). The concentration of dioxins in the cow's milk (for sample years 1993-1995) ranged from 0.87 to 11 ppt TEQ milk fat. In all but two of the samples, dioxin levels were within or below the normal range of 1.1 to 7.1 ppt I-TEQ milk fat previously described for the United Kingdom (MAFF, 1992). Two samples with slightly elevated results were obtained from farms in the vicinity of a municipal waste incinerator, which closed in 1996 due to noncompliance of plant emission standards. PCB levels (sample years 1994-1995) ranged from 1.1 to 9.3 ppt $TEQ_P$-$WHO_{94}$ lipid. In 1996, the Ministry sampled 26 farms in the vicinity of 7 industrial sites and found dioxin levels ranging from 0.81 to 8.6 ppt $TEQ_P$-$WHO_{94}$ milk fat. PCB concentrations ranged from 1.2 to 8.0 ppt $TEQ_P$-$WHO_{94}$ milk fat (MAFF, 1997c).

The Ministry (MAFF) also analyzed 40 samples of cow's milk obtained from 20 dairies and farms in Northern Ireland during 1993 and 1994 (MAFF, 1997d). Sites were chosen to be representative of all the regions of Northern Ireland, and the samples were collected in polypropylene containers. Lipid-based dioxin concentrations in the retail samples from dairies ranged from 0.74 to 2.7 ppt I-$TEQ_{DF}$ lipid (mean = 1.2 ppt I-$TEQ_{DF}$), and the concentrations in individual farm samples ranged from 0.84 to 3.0 ppt I-$TEQ_{DF}$ lipid (mean = 1.2 ppt I-$TEQ_{DF}$). I-$TEQ_{DF}$s were calculated by setting nondetects to the limit of detection.

Vartiainen and Hallikainen (1994) conducted a survey of CDD/CDFs in cow's milk from Finland's largest dairies, eggs from major Finnish producers, and meat (pork and bovine) from major Finnish slaughter houses. Twenty samples in each food category were analyzed for CDD/CDFs. Low levels of CDD/CDFs were observed in cow's milk. Based on

Nordic $TEF_{DF}$s (N-$TEF_{DF}$s), the mean lipid-based Nordic $TEQ_{DF}$ (N-$TEQ_{DF}$) concentrations were 0.83 ppt for milk stored in glass bottles, 1.17 ppt for milk stored in paper milk cartons, and $<$0.5 ppt for meats. Whole-weight N-$TEQ_{DF}$ concentrations in eggs averaged 0.12 ppt. The method used for treating nondetects in calculating these mean N-$TEQ_{DF}$s was not described in the paper. Also, use of N-$TEQ_{DF}$s instead of I-$TEQ_{DF}$s adds uncertainty to the interpretation of these data.

Himberg (1993) analyzed Finnish food samples for dioxin-like PCBs 77, 105, 126, and 169 to estimate the average daily intake of these PCBs. A total of 34 food samples were collected from food stores in the city of Helsinki. Concentrations of these PCBs in foods are presented in Table 3-48. Concentrations were higher in fish than meat by approximately one order of magnitude. Van Rhijr et al. (1993) analyzed 39 cow's milk samples from various agricultural, industrial, and impacted sites in The Netherlands for dioxin-like PCBs 77, 126, and 169 and CDD/CDFs. Mean lipid-based I-$TEQ_{DF}$s ranged from 0.8 to 1.8 for agricultural sites, 2.7 ppt for an industrial site, and 3.6 ppt to 7.7 ppt for sites near municipal waste incinerators. Lipid-based average $TEQ_P$-$WHO_{94}$ ranged from 1.1 ppt to 1.9 ppt for agricultural sites, 2.1 ppt for the industrial site, and 1.8 pt to 4.5 ppt for sites near municipal waste incinerators.

### 3.7.4. Eastern European and Asian Food

Schecter et al. (1990, 1992b) analyzed foods collected from sites within the former Soviet Union between 1988 and 1990 for CDD/CDF residues. A total of 31 samples were collected from markets and restaurants in four cities (i.e., Moscow, Irkutsk, Novosibirsk, and Baikalsk). Fish samples were collected from local rivers in these areas. The study compared CDD/CDF levels from Moscow, which represented an area that had been industrialized for a long period of time, and Siberian cities, where industrialization had occurred more recently. Fathead minnow samples were collected from Lake Baikal in Baikalsk, because it was believed that they may have been impacted by a nearby pulp and papermill. Results of these analyses are presented in Table 3-49. I-$TEQ_{DF}$s ranged from 0.02 to 0.7 ppt wet weight for samples from Moscow, 0.04 to 0.8 ppt wet weight for samples from Irkutsk, 0.005 to 0.8 ppt wet weight for samples from Novosibirsk, and 0.9 to 1.4 ppt wet weight for samples from Baikalsk. Schecter et al. (1993b) analyzed foods from the same cities within the former Soviet Union for PCBs and organochlorine pesticides. PCB 180 was the

only PCB congener analyzed for which a $TEF_P$-$WHO_{94}$ had been developed. Foods analyzed included pork, poultry, fish, beef, lamb, and cheese. PCB 180 in these samples ranged from less than 0.001 to 0.007 ppm on a lipid basis.

Amirova et al. (1997) reported on the results of a 1996 examination of I-$TEQ_{DF}$ levels in 17 foods purchased from food stores in Ufa, a town in the agricultural region of Bashkortostan in Russia. Foods were selected that contained high fat contents, such as dairy products, fish, beef, port, poultry, and vegetable oil (Amirova et al., 1997). The highest I-$TEQ_{DF}$ concentrations, on a lipid basis, were found in freshwater fish (9.2 ppt), cream (5.45 ppt), and milk (3.32 ppt). Using data on the food consumption patterns of both rural and urban regions of the Republic of Bashkortostan, the dietary intake of I-$TEQ_{DF}$s for the urban population of Bashkortostan was calculated to be 2.31 pg/kg/day, while the rural population was estimated to have a lower level of 1.15 pg/kg/day.

Olie et al. (1989) analyzed food and wildlife samples from North and South Vietnam for CDD/CDF residues. Samples were collected between 1985 and 1987 from markets, fishermen, and women in the fields. The collection protocol used non-random sampling and did not provide a statistically representative sample of foods in these regions of the country. However, based on the limited number of samples collected, the study results suggest that food samples collected in the South contain higher levels of CDD/CDFs than samples collected in the North. The authors suggest that these differences may be due, in part, to differences in the level of industrialization in these regions (i.e., the South is more industrialized than the North), and the spraying of South Vietnam with Agent Orange during the Vietnamese War (Olie et al., 1989). Concentrations of CDD/CDF in North Vietnamese food and wildlife samples ranged from 0.26 ppt I-$TEQ_{DF}$ wet weight (catfish) to 3.30 ppt I-$TEQ_{DF}$ wet weight (chicken fat) and 3.51 ppt I-$TEQ_{DF}$ wet weight (cow fat) (Olie et al., 1989). For these same food products (i.e., catfish and chicken fat; cow fat was not analyzed), I-$TEQ_{DF}$ concentrations in South Vietnamese food samples were higher (i.e., 5.68 and 31.54 ppt I-$TEQ_{DF}$ wet weight, respectively) than those observed in samples collected from North Vietnam. The highest I-$TEQ_{DF}$ concentrations in South Vietnamese wildlife were observed in turtle ovaries (85.71 ppt I-$TEQ_{DF}$ wet weight). Schecter et al. (1990) reported the levels of PCBs in samples of pork and chicken collected in Vietnam. PCB 180 was the only PCB congener analyzed for which a $TEF_P$-$WHO_{94}$ had been developed. PCB 180 ranged

from less than 2.0 to 2.0 ppb on a lipid basis in pork and 3.0 to 4.0 ppb on a lipid basis in chicken.

### 3.7.5. Effects of Cooking and Trimming, or Processing on Residue Levels in Foods

Data on the effects of cooking on the levels of dioxin-like compounds in food products are limited, and the available data on this subject are somewhat contradictory. Cooking losses of dioxin-like compounds are reported in the literature in two ways. One method calculates losses by comparing total residues in a sample before cooking to total residues after cooking (i.e., by comparing total micrograms of dioxin-like compounds in raw and cooked foods). The other method calculates losses on the basis of the sample weight (i.e., by comparing the concentrations of residues in raw and cooked food). Losses of total residues are often accompanied by similar losses of water and/or fats. Thus, although total residues are reduced by cooking, concentrations based on the uncooked and cooked sample weights show little difference. Because dietary doses of dioxin-like compounds are calculated on the basis of dioxin residues in uncooked foods and intake rates of as-eaten (i.e., cooked) foods, changes in the concentrations of dioxin-like compounds from cooking would be relevant to these calculations. In contrast, losses of total residues, although an interesting phenomena, would have less effect on the results of these estimates. This section summarizes some of the data on the effects of cooking on dioxin-like residues in foods, as reported in the scientific literature.

Stachiw et al. (1988) evaluated changes in the residue levels of 2,3,7,8-TCDD in fish from cooking. Restructured carp fillets (i.e., fabricated fish products that use mechanically deboned fish) from Saginaw Bay, Michigan, measuring either 7.5-cm or 10-cm diameters, with a uniform thickness of 1 cm, were roasted (covered or uncovered) or charbroiled to internal end temperatures of 60°C, 70°C, or 80°C. Both spiked samples (i.e., spiked to levels approximating 100 ppt) and control samples (i.e., unspiked, containing levels ranging from 37 to 45 ppt) were tested for residues of 2,3,7,8-TCDD before and after cooking. Samples were also tested for total cooking losses (i.e., losses of fat and moisture). Cooking losses ranged from approximately 5 to 20 percent, depending on the cooking method. Results of the TCDD residue analyses indicated that both cooking methods resulted in significant reductions of total 2,3,7,8-TCDD residues (Table 3-50). Reductions of total

TCDD residues by cooking ranged from 34.2 percent to 67.5 percent for control samples and 44.2 percent to 70.6 percent for spiked samples (Stachiw et al., 1988). These percentage reductions were not significantly different for control vs. spiked samples. Thus, the concentration of TCDD in the raw samples did not appear to have a significant impact on the percent reduction by cooking. However, increasing the end point temperature and surface area of the sample resulted in significantly increased losses. Also, TCDD losses were two to eight times greater than total cooking losses (i.e., fat and moisture losses) (Stachiw et al., 1988). When TCDD levels were evaluated on a ppt whole weight concentration basis, reductions ranged from 24 to 60 percent for control samples and 38 to 65 percent for spiked samples.

In a similar study, Zabik et al. (1979) compared the reductions in PCBs in lake trout achieved by various cooking methods. Samples of fat trout from Lake Superior, Michigan, were analyzed before and after broiling, roasting, or microwave cooking. Significant total residue losses of xenobiotics were observed for all cooking methods (i.e., 26 to 53 percent for PCBs). However, when PCB losses were calculated on a wet weight concentration (ppm) basis, losses ranged from 5 to 16 percent, depending on the cooking method. On a fat weight concentration basis (ppm), PCB losses ranged from 16 to 40 percent, depending on the cooking method. In a later study, Zabik et al. (1982) found that cooking was not an effective means of reducing the total residues or concentrations of xenobiotics in the edible tissue of carp from Saginaw Bay.

Poston et al. (1994) and Moya et al. (1997) found that total PCB levels in winter flounder from New Bedford Bay, Massachusetts, were reduced by one cooking method, but not others. Significant reductions in total PCB residues was observed in deep fried fish, but not in fillets pan fried or broiled. Similar results were obtained on an individual congener basis. Deep frying reduced total PCB levels 47 percent. However, deep-fried fillets also showed a weight loss of approximately 40 percent. Pan fried fillets showed a much lower percent reduction in total PCB residues. Water losses were also significantly lower for these two cooking methods (i.e., 7 percent and 15 percent, respectively) (Moya et al., 1997).

Smith et al. (1973) reported that the total residue levels of PCBs (Aroclor 1248 and 1254) in chinook and coho salmon steaks from Lake Michigan were reduced only slightly by baking and poaching; however, Cichy et al. (1979) observed significant losses (38 to 43 percent) of PCBs in Michigan lake trout that were irradiated and broiled. Losses were

calculated by comparing the total micrograms of PCBs in raw and cooked fish. Pan-frying of white croaker fillets from Santa Monica Bay and Orange County, California, resulted in total PCB (Aroclor 1242 and 1254) losses of 65 percent and 28 percent, respectively (Puffer and Gossett, 1983). Trotter et al. (1989) found that, on the basis of concentration, PCB levels in cooked and uncooked bluefish were similar. However, when the results of cooked samples were corrected for moisture losses and compared to raw samples based on total residue levels, PCBs were found to be reduced by 27 percent.

Based on a study of Atlantic bluefish collected near Long Island, New York, Armbruster et al. (1989) concluded that trimming resulted in the largest reductions in PCB residues in fish. Armbruster et al. (1989) reported that trimming bluefish fillets resulted in an average total PCB residue reduction of 59.4 percent and that baking, broiling, frying, or poaching resulted in further losses averaging only 7.5 percent. The magnitude of reduction observed for the various cooking methods (combined with trimming) did not differ significantly. In addition, Armbruster et al. (1989) analyzed oil drippings released during cooking, and found that the total PCB residues in the oil did not account for the total losses of PCBs that occurred during cooking. Based on these results, Armbruster et al. (1989) concluded that "PCB losses by vaporization during the various cooking procedures may have constituted the major portion of the mean total (7.5 percent) loss from cooking." Skea et al. (1979) also reported that trimming resulted in significantly greater reductions in PCB concentrations in fish than cooking. Skea et al. (1979) evaluated the effects of trimming and various cooking methods on the residue levels of PCB, Mirex, and DDT in brown trout and smallmouth bass collected from Lake Ontario. PCB results from this study are summarized in Table 3-51 according to the percent reduction in concentration ($\mu$g/g) and total reduction of PCB residues (i.e., calculated by comparing the total micrograms of PCBs in raw and cooked fish). Trimming alone resulted in total percent reductions of approximately 80 percent. Of the cooking methods, deep fat frying resulted in the greatest additional reductions in PCB residues Skea et al. (1979).

Zabik (1974) found that cooked chicken pieces had significantly lower concentrations of PCBs than raw pieces. Ten hens that had been fed PCB Aroclor 1254 were slaughtered, split in half, and cut into several pieces (i.e., drumsticks, breasts, thigh, etc.). Pieces obtained from one-half of the chicken were analyzed raw, and pieces from the other half were analyzed after stewing or pressure cooking. The two cooking methods

resulted in similar losses of PCBs (Table 3-52). Cooking resulted in greater losses from abdominal adipose tissue and thigh skin. These pieces also had the highest fat content. Recovery of PCBs was calculated by comparing the levels of Aroclor in cooked chicken and broth to the levels in raw chicken. Percentage recoveries ranged from 60 to 95 percent.

Schecter et al. (1996) studied the effects of cooking on CDD/CDF levels in hamburger. Ground beef was purchased from a supermarket in Binghamton, New York, and divided into eight samples. Four samples were analyzed for CDD/CDFs uncooked, and the other four samples were broiled and then analyzed for CDD/CDFs. Cooking produced a 42 to 49 percent decrease in I-$TEQ_{DF}$s per hamburger. However, this decrease was identical to the decrease in weight due to cooking. Thus, reduction in CDD/CDFs in hamburger are due to loss of fat and water during cooking. The I-$TEQ_{DF}$s calculated on a whole weight concentration basis were 0.128 to 0.134 ppt for uncooked samples and 0.116 to 0.145 ppt for cooked samples. These results indicated little change in total I-$TEQ_{DF}$ concentrations from cooking, despite the significant losses observed on the basis of total residue level.

Schecter et al. (1999) recently extended their studies to examine the effect of broiling on CDD, CDF, and co-planar PCB levels in hamburger, bacon, and catfish. The samples were again purchased from a supermarket in Binghamton, New York, and each food type was divided into nine samples. Five samples of each food type were broiled in an electric oven (one of each cooked food type sample was consumed to guarantee edibility), and the other four samples were analyzed uncooked. Reductions of approximately 50 percent in the total CDD/CDF/PCB TEQ levels (using I-TEFs for CDD/CDFs and $TEF_P$-$WHO_{94}$ for PCBs) were observed in the broiled samples. However, after adjusting for moisture losses due to broiling (i.e., looking at concentration on a weight basis), the following TEQ concentrations of CDD/CDF/PCBs were observed: (1) TEQ concentrations in the hamburger remained the same (about 0.155 ppt); (2) TEQ concentrations in the bacon increased 84 percent (from 0.079 ppt uncooked to 0.145 ppt cooked); and (3) TEQ concentrations in the catfish decreased by 34 percent (from 0.577 ppt uncooked to 0.378 ppt cooked).

Petroske et al. (1997) studied the effect of pan frying on CDD/CDF concentrations in ground beef. Samples were collected from four control animals and four dosed (16 congeners) cattle in a dioxin/furan feeding experiment. Muscle tissue (rib eye) and back fat from the cattle were blended into patties containing approximately 20 percent fat. The patties were then cooked in a stainless steel frying pan to an internal temperature of 74°C.

During the cooking process, the fats, juices, and volatiles that formed were collected on an inverted funnel placed over the frying pan, draining back into the pan. These volatiles were analyzed along with the cooked patties. Analysis of the samples showed significant reductions in total congener residue levels (pg/patty) after pan frying (assuming the fats and juices are not consumed). Decreases after cooking ranged from approximately 21 to 50 percent for the control samples and approximately 31 to 50 percent for the dosed animals. The majority of reductions after pan frying, for both types of sample, were in the 40 to 50 percent range, which is similar to the reductions observed by Schecter et al. (1996; 1999) after broiling. The findings were not analyzed on a concentration (i.e., ppt per unit weight) basis, however. Consistent with the findings of Stachiw et al. (1988), the concentration of CDDs and CDFs in the raw samples did not appear to have a significant impact on the percent reduction by pan frying. Taking into consideration the CDD/CDFs found in the volatiles and juices released during the cooking process, between 6 and 16 percent of the dosed CDD/CDF congeners were unaccounted for after cooking. Unaccounted losses were greater for the lower chlorinated dioxin congeners than for the more highly chlorinated congeners, while the opposite was observed with the furans. Petroske et al. (1997) hypothesized that "pan frying of ground beef patties, and likely non-patty ground beef significantly reduces the quantity of dioxin and furan congeners consumed if the fat and juices are discarded, while congeners releases as volatiles may pose a secondary mode of human exposure."

To evaluate the potential for contamination of foods with CDD/CDFs during the curing process, Mayer (1998) examined CDD/CDFs levels in 41 smoked ham samples produced in southern Germany, and compared the results to CDD/CDFs in 21 untreated pork samples. Only the results from samples of the outer layer of the smoked ham, about 1 cm thickness, were included in the comparison because it was assumed that CDD/CDFs derived from the curing smoke would be concentrated in this surface. I-$TEQ_{DF}$ concentrations ranged from 0.08 pg/g fat to 85 pg/g fat, with an average of 6.2 pg/g fat in the smoked ham samples. In comparison, the I-$TEQ_{DF}$ concentrations in the untreated pork samples were between 0.09 pg/g fat and 1.2 pg/g fat, with an average of 0.31 pg/g fat. The median I-$TEQ_{DF}$ levels for these two groups were approximately the same; 0.33 pg/g fat for the outer parts of smoked ham samples and 0.31 pg/g fat for the untreated pork samples. Most pork samples had lipid-based I-$TEQ_{DF}$ concentrations below 0.5 pg/g. A total of 61

percent of the smoked ham samples, compared to 90 percent of the untreated pork samples, had I-$TEQ_{DF}$ concentrations lower than 0.5 pg/g fat. Mayer (1998) also presented I-$TEQ_{DF}$ levels based on all edible parts of the smoked ham samples. The results indicated that 30 samples had I-$TEQ_{DF}$ concentrations lower than 0.04 pg/g; 8 samples showed levels between 0.06 and 0.35 pg/g; and 3 samples were highly contaminated with I-$TEQ_{DF}$s ranging from 1.7 to 3.7 pg/g. CDD/CDF homologue group profiles illustrated that smoked ham with low CDD/CDF contamination had a similar profile to that of untreated pork. Smoked ham with elevated levels of CDD/CDFs had a similar profile to that of highly contaminated ham, but a quite different profile to that of untreated pork. Mayer (1998) indicated that the curing process may be a source of CDD/CDFs in smoked ham.

Results of the preceding studies suggest that processes such as smoke curing may increase CDD/CDFs in foods. Cooking may significantly reduce total residues of dioxin-like compounds in foods. However, the data reported on the basis of concentration are somewhat contradictory. Schecter et al. (1996, 1999) observed that concentrations of dioxin-like compounds in beef were not significantly affected by cooking, but the effects of cooking were more significant in bacon and fish. Skea et al. (1979) also observed significant concentration changes of PCBs in fish. In contrast, the Zabik et al. (1979, 1982) studies did not show significant losses of PCB concentrations in fish from cooking. Therefore, based on the existing data, it is not possible to draw conclusions with regard to reductions in food concentrations of dioxin-like compounds from cooking. As a result, potential reductions in concentrations from cooking are not accounted for in Chapter 4 for the purpose of estimating dietary intake of dioxin-like compounds.

### 3.7.6. Food Observations and Trends

Some general observations for CDD/CDF levels are possible from the data presented in the various food product studies above:

- TEQ concentrations of CDD/CDFs and PCBs are similar in the studies from the United States and Europe.
- CDD/CDF levels in the higher chlorinated congeners (i.e., HpCDDs and OCDD) are present in higher concentrations than the lower chlorinated congeners. In

the higher chlorinated congeners, CDD levels are present in greater concentrations than the CDFs.

- PeCDD is frequently the highest contributors to total TEQs in foods.
- Food products of animal origin (i.e., fish, meat, eggs, and dairy products), which have a high fat content, have a higher concentration of CDD/CDFs than those food products that have lower lipid contents.
- Generally, of all the food products, fish and shellfish contain the highest levels of CDD/CDFs.

### 3.7.7. Food CDD/CDF Congener Profiles and Background TEQ Concentrations

The 2,3,7,8-substituted congener profile for various foods are presented in Table 3-53 and Figures 3-7 through 3-10. These profiles are calculated as the ratio of individual congener concentrations to the sum of concentrations for all of the 2,3,7,8-substituted congeners and are based on data from the studies discussed previously and footnoted on Table 3-53. Profiles for beef, milk, and dairy products are similar, with 1,2,3,4,6,7,8-HpCDD and OCDD dominating the profiles in nearly equal proportions. In contrast, OCDD is the single dominant congener in both chicken, eggs, and pork.

U.S. food data on CDD/CDFs are summarized in Table 3-54. Background $TEQ_{DF}$-$WHO_{98}$ estimates are presented first assuming that nondetects equal half the detection limits and second assuming that nondetects equal zero. Large national surveys conducted by EPA/USDA (i.e., beef, pork, poultry, and milk) provide an adequate basis for estimating the concentrations of dioxin-like compounds believed to be representative of background levels in U.S. foods. For some food groups (i.e., eggs, fish, and shellfish), however, the small sample size and high number of nondetects provide an uncertain basis for estimating national background levels. Overall, the general agreement between the national U.S. estimates and the food level estimates for Canada and Europe provides some reassurance that the U.S. values are reasonable. For the purposes of calculating background exposures to CDD/CDFs via dietary intake, the upper-range background $TEQ_{DF}$-$WHO_{98}$ (i.e., those calculated using one-half the detection limit for the nondetects) were used, except for vegetable oil. (See Chapter 4.)

North American food data on PCBs are summarized in Table 3-55. These data are used in Chapter 4 to estimate background dietary exposures to PCBs.

## 3.8. SUMMARY OF CDD/CDF AND PCB LEVELS IN ENVIRONMENTAL MEDIA AND FOOD

This chapter summarizes data on CDD/CDF and PCB levels in environmental media and food with emphasis on "background levels." Data representative of background conditions in environmental media are considered to be those collected in rural, pristine, and urban (soil and air only) areas not believed to be impacted by any local sources (e.g., incinerators).

The mean background levels for the various environmental media and foods presented in this chapter are summarized in Table 3-56. These total background level $TEQ_{DFP}$-$WHO_{98}$ are used in Chapter 4 to estimate typical exposure levels in the United States. Standard deviations of the total mean $TEQ_{DF}$-$WHO_{98}$ for each media were also calculated to depict the "range" of probable CDD/CDF levels in various media. For media for which complete congener-specific data for multiple samples from the same study (i.e., beef, pork, poultry, milk, dairy, and sediments) were available, means and standard deviations were calculated by conventional methods. However, for media for which mean total $TEQ_{DF}$-$WHO_{98}$ were calculated by summing congener-specific $TEQ_{DF}$-$WHO_{98}$ from multiple studies (e.g., soil, freshwater fish, air), the use of typical methods for calculating standard deviations was not possible. Therefore, standard deviations were based on the standard deviation of the congener that contributed most to the total $TEQ_{DF}$-$WHO_{98}$. The percentage deviation from the mean for that congener was applied to the total mean $TEQ_{DF}$-$WHO_{98}$ for all congeners combined. The congeners selected for use in the standard deviation estimates are presented in Table 3-57. The data in this table indicate that the pentachlorinated dioxins and furans were frequently the highest contributors to total $TEQ_{DF}$-$WHO_{98}$ in foods and other environmental media in the United States. Media levels presented in Table 3-56 are shown graphically in Figure 3-11. Table 3-56 illustrates that of all the food products, levels (whole weight basis) of CDD/CDF/PCBs are highest in freshwater fish.

*Estimates for background levels of dioxin-like compounds in environmental media are based on data from a variety of studies conducted at different locations in North America*. Of the studies available for this compilation, only those conducted in locations described as “background” were selected. The amount and representativeness of the data varies, but in general these data lack the statistical basis to be considered true national means. The environmental media concentrations found in the United States were consistent across the

various studies, and were consistent with similar studies in Europe. These data were the best available for comparing site-specific values to National background levels. The limited data on dioxin-like PCBs in environmental media are summarized in the document, but were not deemed adequate for estimating background levels. Because of the limited number of locations examined, however, it is not known if these ranges adequately capture the full national variability, if significant regional variability exists making national means of limited utility, or if elevated levels above this range could still be the result of background contamination processes. As new data are collected these ranges are likely to be expanded and refined.

*Estimates in food are based on data from a variety of studies conducted in North America*. Beef, pork, and poultry data were derived from statistically based national surveys. Milk estimates were derived from a nationwide sampling network. Dairy estimates were derived from the milk fat concentrations, coupled with appropriate assumptions for the amount of fat in dairy products. Egg data were grab samples from retail stores. Fish data were collected from a combination of field and grocery store studies. As with environmental media, food levels found in the United States are similar to levels found in Europe.

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Table 3-1. Mean CDD/CDF Ambient Air Concentrations from Sites Located Upwind and Downwind of an Industrial Site

| | Upwind (Background) | | | Downwind | | |
|---|---|---|---|---|---|---|
| | No. of Positive Samples | Mean Concentration nd = ½ LOD[a] ($pg/m^3$) | Mean Concentration nd = 0[b] ($pg/m^3$) | No. of Positive Samples | Mean Concentration nd = ½ LOD[a] ($pg/m^3$) | Mean Concentration nd = 0[b] ($pg/m^3$) |
| 2,3,7,8-TCDD | 0/3 | 0.0070 | 0 | 0/3 | 0.0070 | 0 |
| 1,2,3,7,8-PeCDD | 0/3 | 0.0070 | 0 | 1/3 | 0.17 | 0.16 |
| 1,2,3,4,7,8-HxCDD | 0/3 | 0.0070 | 0 | 3/3 | 0.24 | 0.24 |
| 1,2,3,6,7,8-HxCDD | 1/3 | 0.015 | 0.010 | 3/3 | 0.39 | 0.39 |
| 1,2,3,7,8,9-HxCDD | 1/3 | 0.015 | 0.010 | 2/3 | 0.062 | 0.060 |
| 1,2,3,4,6,7,8-HpCDD | 3/3 | 0.41 | 0.41 | 2/3 | 2.0 | 2.0 |
| OCDD | 15/16 | 1.1 | 1.1 | 14/14 | 3.0 | 3.0 |
| 2,3,7,8-TCDF | 0/3 | 0.090 | 0.090 | 3/3 | 1.5 | 1.5 |
| 1,2,3,7,8-PeCDF | 0/3 | 0.0050 | 0 | 3/3 | 0.25 | 0.25 |
| 2,3,4,7,8-PeCDF | 0/3 | 0.0070 | 0 | 2/3 | 0.69 | 0.68 |
| 1,2,3,4,7,8-HxCDF | 1/3 | 0.023 | 0 | 2/3 | 0.11 | 0.11 |
| 1,2,3,6,7,8-HxCDF | 1/3 | 0.010 | 0.010 | 3/3 | 0.45 | 0.45 |
| 2,3,4,6,7,8-HxCDF | 1/3 | 0.018 | 0.010 | 1/3 | 0.76 | 0.76 |
| 1,2,3,7,8,9-HxCDF | 0/3 | 0.0070 | 0 | 1/3 | 0.038 | 0.038 |
| 1,2,3,4,6,7,8-HpCDF | 1/3 | 0.053 | 0.050 | 3/3 | 2.1 | 2.1 |
| OCDF | 12/16 | 0.089 | 0.071 | 8/15 | 0.62 | 0.59 |
| TOTAL I-$TEQ_{DF}$ | -- | 0.038 | 0.019 | -- | 0.84 | 0.83 |
| TOTAL $TEQ_{DF}$-$WHO_{98}$ | -- | 0.041 | 0.020 | -- | 0.92 | 0.91 |

[a] Nondetects assumed to be one-half the detection limit in calculating the mean.
[b] Nondetects assumed to be zero in calculating the mean.

Source: Smith et al. (1989).

Table 3-2. Congener-Specific, Homologue, Total, and TEQ Concentrations for the Four Clusters of Air Samples (pg/m$^3$)

| Congener | 1994 Impacted Air (n = 2) | 1994 Urban (n = 8) | 1995 Urban (n = 6) | Rural (n = 3) |
|---|---|---|---|---|
| 2,3,7,8-TCDD | 0.019 | 0.003 | 0.007 | 0.003 |
| 1,2,3,7,8-PCDD | 0.062 | 0.012 | 0.008 | 0.005 |
| 1,2,3,4,7,8-HxCDD | 0.081 | 0.017 | 0.011 | 0.008 |
| 1,2,3,6,7,8-HxCDD | 0.095 | 0.028 | 0.024 | 0.009 |
| 1,2,3,7,8,9-HxCDD | 0.086 | 0.029 | 0.020 | 0.013 |
| 1,2,3,4,6,7,8-HpCDD | 0.633 | 0.248 | 0.205 | 0.227 |
| OCDD | 1.765 | 1.062 | 0.807 | 0.904 |
| 2,3,7,8-TCDF | 0.051 | 0.012 | 0.017 | 0.003 |
| 1,2,3,7,8-PCDF | 0.121 | 0.024 | 0.022 | 0.007 |
| 2,3,4,7,8-PCDF | 0.169 | 0.028 | 0.020 | 0.010 |
| 1,2,3,4,7,8-HxCDF | 0.205 | 0.038 | 0.063 | 0.014 |
| 1,2,3,6,7,8-HxCDF | 0.302 | 0.056 | 0.058 | 0.016 |
| 1,2,3,7,8,9-HxCDF | 0.006 | 0.003 | 0.003 | 0.003 |
| 2,3,4,6,7,8-HxCDF | 0.189 | 0.033 | 0.027 | 0.009 |
| 1,2,3,4,6,7,8-HpCDF | 0.939 | 0.165 | 0.165 | 0.061 |
| 1,2,3,4,7,8,9-HpCDF | 0.131 | 0.027 | 0.038 | 0.014 |
| OCDF | 0.411 | 0.124 | 0.159 | 0.067 |
| TCDD | 0.761 | 0.097 | 0.110 | 0.015 |
| PCDD | 0.939 | 0.158 | 0.082 | 0.027 |
| HxCDD | 1.193 | 0.331 | 0.252 | 0.188 |
| HpCDD | 1.290 | 0.533 | 0.416 | 0.494 |
| TCDF | 1.793 | 0.374 | 0.378 | 0.083 |
| PCDF | 2.373 | 0.420 | 0.294 | 0.122 |
| HxCDF | 2.044 | 0.363 | 0.361 | 0.134 |
| HpCDF | 1.542 | 0.287 | 0.325 | 0.144 |
| TOTAL | 14.11 | 3.75 | 3.18 | 2.18 |
| I-$TEQ_{DF}$ | 0.26 | 0.050 | | 0.022 |
| $TEQ_{DF}$-$WHO_{98}$ | 0.29 | 0.055 | | 0.024 |

Note: Non-detects assumed to be one-half the detection limit.
Source: OEPA, 1995.

Table 3-3. Background Air Concentrations of CDD/CDFs at Mohawk Mountain, Connecticut

| Parameter | Sample Period | | | |
|---|---|---|---|---|
| | Oct. 29-Nov. 29, 1993 ($pg/m^3$) | Jan. 20-Feb. 18, 1994 ($pg/m^3$) | Apr. 26-May 26, 1994 ($pg/m^3$) | Jul. 26-Aug. 25, 1994 ($pg/m^3$) |
| 2,3,7,8-TCDD | 0.001 | (0.000) | (0.001) | (0.000) |
| Total TCDD | 0.032 | 0.017 | 0.012 | 0.007 |
| 1,2,3,7,8-PeCDD | 0.004 | 0.002 | 0.002 | (0.001) |
| Total PeCDD | 0.043 | 0.027 | 0.022 | 0.009 |
| 1,2,3,4,7,8-HxCDD | 0.007 | 0.004 | 0.002 | (0.001) |
| 1,2,3,6,7,8-HxCDD | 0.010 | 0.005 | 0.003 | 0.002 |
| 1,2,3,7,8,9-HxCDD | 0.011 | 0.005 | 0.002 | (0.001) |
| Total HxCDD | 0.122 | 0.072 | 0.032 | 0.020 |
| 1,2,3,4,6,7,8-HpCDD | 0.159 | 0.065 | 0.029 | 0.016 |
| Total HpCDD | 0.317 | 0.133 | 0.061 | 0.033 |
| OCDD | 0.451 | 0.196 | 0.155 | 0.056 |
| 2,3,7,8-TCDF | 0.004 | 0.003 | 0.002 | 0.004 |
| Total TCDF | 0.134 | 0.090 | 0.095 | 0.112 |
| 1,2,3,7,8-PeCDF | 0.004 | 0.003 | 0.003 | 0.003 |
| 2,3,4,7,8-PeCDF | 0.006 | 0.005 | 0.004 | 0.004 |
| Total PeCDF | 0.078 | 0.057 | 0.057 | 0.073 |
| 1,2,3,4,7,8-HxCDF | 0.008 | 0.008 | 0.004 | 0.004 |
| 1,2,3,6,7,8-HxCDF | 0.007 | 0.006 | 0.003 | 0.004 |
| 2,3,4,6,7,8-HxCDF | 0.009 | 0.008 | 0.004 | 0.004 |
| 1,2,3,7,8,9-HxCDF | 0.004 | 0.003 | 0.001 | (0.001) |
| Total HxCDF | 0.078 | 0.060 | 0.041 | 0.041 |
| 1,2,3,4,6,7,8-HpCDF | 0.035 | 0.028 | 0.012 | 0.016 |
| 1,2,3,4,7,8,9-HpCDF | 0.006 | 0.005 | 0.001 | 0.002 |
| Total HpCDF | 0.070 | 0.049 | 0.020 | 0.029 |
| OCDF | 0.028 | 0.027 | 0.011 | 0.011 |

( ) = Parameter not detected at the indicated detection limit.

Source: CDEP (1995)

Table 3-4. Ambient Air Concentrations Near a Roadway in Phoenix, Arizona

| Parameter | Average I-$TEQ_{DF}$ ($pg/m^3$) | Average $TEQ_{DF}$-$WHO_{98}$ |
|---|---|---|
| 2,3,7,8-TCDD | 0.0077 | 0.0077 |
| 1,2,3,7,8-PeCDD | 0.0254 | 0.0508 |
| 1,2,3,4,7,8-HxCDD | 0.0096 | 0.0096 |
| 1,2,3,6,7,8-HxCDD | 0.0220 | 0.0220 |
| 1,2,3,7,8,9-HxCDD | 0.0182 | 0.0182 |
| 1,2,3,4,6,7,8-HpCDD | 0.0295 | 0.0295 |
| OCDD | 0.0096 | 0.00096 |
| 2,3,7,8-TCDF | 0.0033 | 0.0033 |
| 1,2,3,7,8-PeCDF | 0.0026 | 0.0026 |
| 2,3,4,7,8-PeCDF | 0.0562 | 0.0562 |
| 1,2,3,4,7,8-HxCDF | 0.0147 | 0.0147 |
| 1,2,3,6,7,8-HxCDF | 0.0127 | 0.0127 |
| 2,3,4,6,7,8-HxCDF | 0.0215 | 0.0215 |
| 1,2,3,7,8,9-HxCDF | 0.0078 | 0.0078 |
| 1,2,3,4,6,7,8-HpCDF | 0.0076 | 0.0076 |
| 1,2,3,4,7,8,9-HpCDF | 0.0010 | 0.0010 |
| OCDF | 0.0003 | 0.00003 |
| TOTAL TEQ | 0.2499 | 0.2664 |

Source: Hunt et al. (1997)

Table 3-5. Annual Mean PCB Concentrations in Ambient Air, Ontario, Canada ($pg/m^3$)

| PCB Congener | No. of Positive Samples | Annual Mean Concentration ($pg/m^3$) |
|---|---|---|
| 105 | 63/143 | 0.16 |
| 114 | 79/143 | 1.2 |
| 118 | 122/142 | 2.3 |
| 156 | 13/143 | 0.07 |
| 170 | 53/143 | 0.48 |
| 180 | 111/143 | 1.1 |
| 189 | 3/143 | 0.01 |
| TOTAL $TEQ_{DF}$-$WHO_{94}$ | -- | 0.00094 |
| TOTAL $TEQ_{DF}$-$WHO_{98}$ | -- | 0.00088 |

Source: Hoff et al. (1992).

Table 3-6. Annual Average Dioxin-Like PCB Concentrations in Ambient Air in Germany (pg/m$^3$)

| | Köln | Duesburg[a] | Essen | Dortmund |
|---|---|---|---|---|
| PCB 77 | 3.0 | 4.5-4.8 | 2.6 | 4.5 |
| PCB 126 | 0.27 | 0.50-0.62 | 0.24 | 0.48 |
| PCB 169 | 0.02 | 0.05 | 0.06 | 0.07 |
| TOTAL (77 + 126 + 169) | 3.3 | 5.2-5.4 | 2.9 | 5.1 |

[a] Two sites were sampled at this location.

Source: Hiester et al. (1995).

Table 3-7. Mean Background CDD/CDF Profiles for Air

| 2,3,7,8-Substituted CDD/CDFs | Rural Background[a] | | Urban Background[b] | | CDD/CDF Homologue Group | Rural Background[a] | | Urban Background[c] | |
|---|---|---|---|---|---|---|---|---|---|
| | Concentration (pg/m$^3$) | Fraction of Total CDD/CDFs | Concentration (pg/m$^3$) | Fraction of Total CDD/CDFs | | Concentration (pg/m$^3$) | Fraction of Total CDD/CDFs | Concentration (pg/m$^3$) | Fraction of Total CDD/CDFs |
| 2,3,7,8-TCDD | 0.0001 | 0.0001 | 0.0007 | 0.0001 | TCDD | 0.015 | 0.0105 | 0.0165 | 0.0030 |
| 1,2,3,7,8-PeCDD | 0.0029 | 0.0020 | 0.0039 | 0.0007 | PeCDD | 0.026 | 0.0176 | 0.0406 | 0.0075 |
| 1,2,3,4,7,8-HxCDD | 0.0042 | 0.0029 | 0.0110 | 0.0020 | HxCDD | 0.125 | 0.0854 | 0.2278 | 0.0418 |
| 1,2,3,6,7,8-HxCDD | 0.0057 | 0.0039 | 0.0248 | 0.0046 | HpCDD | 0.315 | 0.2157 | 0.8886 | 0.1632 |
| 1,2,3,7,8,9-HxCDD | 0.0061 | 0.0042 | 0.0280 | 0.0051 | OCDD | 0.559 | 0.3830 | 1.8547 | 0.3406 |
| 1,2,3,4,6,7,8-HpCDD | 0.1473 | 0.1008 | 0.5561 | 0.1021 | TCDF | 0.096 | 0.0654 | 1.1784 | 0.2164 |
| OCDD | 0.5591 | 0.3829 | 1.8547 | 0.3406 | PeCDF | 0.094 | 0.0645 | 0.6905 | 0.1268 |
| 2,3,7,8-TCDF | 0.0016 | 0.0011 | 0.2158 | 0.0396 | HxCDF | 0.095 | 0.0647 | 0.2804 | 0.0515 |
| 1,2,3,7,8-PeCDF | 0.0039 | 0.0027 | 0.0567 | 0.0104 | HpCDF | 0.093 | 0.0637 | 0.1764 | 0.0324 |
| 2,3,4,7,8-PeCDF | 0.0065 | 0.0044 | 0.0213 | 0.0039 | OCDF | 0.043 | 0.0295 | 0.0911 | 0.0167 |
| 1,2,3,4,7,8-HxCDF | 0.0074 | 0.0050 | 0.0436 | 0.0080 | | | | | |
| 1,2,3,6,7,8-HxCDF | 0.0066 | 0.0045 | 0.0671 | 0.0123 | | | | | |
| 1,2,3,7,8,9-HxCDF | 0.0010 | 0.0007 | 0.0008 | 0.0001 | | | | | |
| 2,3,4,6,7,8-HxCDF | 0.0060 | 0.0041 | 0.0204 | 0.0037 | | | | | |
| 1,2,3,4,6,7,8-HpCDF | 0.0379 | 0.0260 | 0.1264 | 0.0232 | | | | | |
| 1,2,3,4,7,8,9-HpCDF | 0.0079 | 0.0054 | 0.0070 | 0.0013 | | | | | |
| OCDF | 0.0430 | 0.0295 | 0.0919 | 0.0169 | | | | | |
| TOTAL | 0.847 | 0.580 | 3.130 | 0.5749 | TOTAL | 1.460 | 1.000 | 5.445 | 1.00 |

NOTE: Non-detects are assumed to be zero.

[a] Based on data from OEPA (1995) and CDEP (1988).
[b] Based on data from CDEP (1988, 1995); Smith et al. (1989); Maisel and Hunt (1990); Hunt et al. (1990); and OEPA (1995).
[c] Based on data from CDEP (1988, 1995); Smith et al. (1989, 1990a); Maisel and Hunt (1990); Hunt et al. (1990); and OEPA (1995).

See Table 3-8 for sampling locations and numbers of samples from these studies.

Table 3-8. $TEQ_{DF}$-$WHO_{98}$ Concentrations of CDD/CDFs in Air in the United States (pg/m$^3$)
(ND = 1/2 LOD)

| Reference | 2,3,7,8-TCDD | 1,2,3,7,8-PeCDD | 2,3,7,8-HxCDDs | 1,2,3,4,6,7,8-HpCDD | OCDD | 2,3,7,8-TCDF | 1,2,3,7,8-PeCDF | 2,3,4,7,8-PeCDF | 2,3,7,8-HxDCFs | 2,3,7,8-HpCDFs | OCDF | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| RURAL BACKGROUND | | | | | | | | | | | | |
| CDEP, 1995 Mohawk Mt., CT (n = 4) | 0.00038 | 0.0021 | 0.0013 | 0.00067 | 0.000021 | 0.00033 | 0.00016 | 0.0024 | 0.0019 | 0.00026 | 0.0000019 | 0.010 |
| Ohio EPA, 1995 Rural Ohio (n = 3) | 0.0029 | 0.0052 | 0.0031 | 0.0023 | 0.000090 | 0.00028 | 0.00033 | 0.0048 | 0.0041 | 0.00076 | 0.0000067 | 0.024 |
| MEAN | 0.0016 | 0.0037 | 0.0022 | 0.0015 | 0.000056 | 0.00030 | 0.00024 | 0.0036 | 0.0030 | 0.00051 | 0.0000043 | 0.0167 |
| SD | 0.0013 | 0.0015 | 0.00089 | 0.00080 | 0.000034 | 0.000023 | 0.000081 | 0.0012 | 0.0011 | 0.00025 | 0.0000024 | |
| WEIGHTED MEAN | 0.0015 | 0.0034 | 0.0021 | 0.0014 | 0.000051 | 0.00031 | 0.00023 | 0.0034 | 0.0029 | 0.00047 | 0.0000040 | 0.0156 |
| URBAN BACKGROUND | | | | | | | | | | | | |
| CDEP, 1988 Wallingford, CT (n = 28) | 0.0019 | 0.0063 | 0.0086 | 0.0029 | 0.00055 | 0.0072 | 0.00039 | 0.010 | 0.012 | 0.0029 | 0.000022 | 0.053 |
| CDEP, 1995 Connecticut (n = 20) | 0.00012 | 0.0063 | 0.0036 | 0.0014 | 0.000044 | 0.00093 | 0.00046 | 0.0080 | 0.0065 | 0.00081 | 0.0000055 | 0.029 |
| Hunt and Maisel, 1990 Bridgeport, CT (n = 7) | 0.012 | 0.024 | 0.015 | 0.0048 | 0.00021 | 0.0078 | 0.0016 | 0.024 | 0.024 | 0.0025 | 0.000021 | 0.115 |
| Hunt et al., 1990 W. Long Beach, CA (n = 2) | 0.0075 | 0.032 | 0.0046 | 0.0034 | 0.00029 | 0.025 | 0.00078 | 0.00275 | 0.012 | 0.00081 | 0.000037 | 0.088 |
| Hunt et al., 1990 Reseda, CA (n = 7) | 0.0080 | 0.034 | 0.026 | 0.024 | 0.00054 | 0.0028 | 0.0016 | 0.015 | 0.026 | 0.0024 | 0.000012 | 0.140 |
| Hunt et al., 1990 San Bernadino, CA (n = 5) | 0.013 | 0.12 | 0.015 | 0.0059 | 0.00031 | 0.0038 | 0.020 | 0.019 | 0.028 | 0.0029 | 0.000016 | 0.232 |
| Hunt et al., 1990 El Toro, CA (n = 7) | 0.010 | 0.018 | 0.0075 | 0.0014 | 0.00011 | 0.0015 | 0.0017 | 0.018 | 0.016 | 0.00098 | 0.0000076 | 0.076 |
| Hunt et al., 1990 N. Long Beach, CA (n = 6) | 0.013 | 0.017 | 0.020 | 0.0079 | 0.00014 | 0.0018 | 0.0019 | 0.020 | 0.023 | 0.0031 | 0.000015 | 0.110 |
| Maisel and Hunt, 1990 Los Angeles, CA (n = 1) | 0.0048 | 0.020 | 0.012 | 0.0025 | 0.00019 | 0.0021 | 0.0039 | 0.034 | 0.048 | 0.0010 | 0.0000056 | 0.132 |
| Ohio EPA, 1995 Franklin Co., OH (n = 14) | 0.0048 | 0.010 | 0.0066 | 0.0023 | 0.00010 | 0.0014 | 0.0012 | 0.012 | 0.014 | 0.0020 | 0.000014 | 0.055 |
| Smith et al., 1989 Niagra Falls, NY (n = 3) | 0.0070 | 0.0070 | 0.0037 | 0.0041 | 0.00011 | 0.0090 | 0.00025 | 0.0035 | 0.0058 | 0.00053 | 0.0000089 | 0.041 |
| Smith et al., 1990a Albany, NY (n = 3) | 0.048 | | | | 0.000057 | 0.094 | | | | | 0.000028 | 0.142 |
| Smith et al., 1990a Binghampton, NY (n = 1) | 0.030 | | | | 0.00014 | 0.018 | | | | | 0.000015 | 0.048 |
| Smith et al., 1990a Utica, NY (n = 2) | 0.058 | | | | 0.00012 | 0.12 | | | | | 0.000031 | 0.173 |
| MEAN | 0.016 | 0.027 | 0.011 | 0.0055 | 0.00021 | 0.021 | 0.0031 | 0.015 | 0.020 | 0.0018 | 0.000017 | 0.120 |
| SD | 0.017 | 0.032 | 0.007 | 0.0063 | 0.00016 | 0.035 | 0.0054 | 0.010 | 0.011 | 0.00094 | 0.0000093 | |
| WEIGHTED MEAN | 0.008 | 0.018 | 0.010 | 0.0045 | 0.00026 | 0.009 | 0.0019 | 0.013 | 0.015 | 0.0020 | 0.000015 | 0.081 |

Table 3-9. Mean PCDD and PCDF Concentrations in Canadian Soil from 1987 (ppt)[a]

| Homologue Group | Soil Near Sludge Incinerator (n = 12) | Urban Background (n = 11) | Rural Background (n = 26) |
|---|---|---|---|
| TCDDs | 69 (ND-430) | ND | ND |
| PeCDDs | 81 (ND-540) | ND | ND |
| HxCDDs | 9 (ND-70) | ND | ND |
| HpCDDs | 43 (ND-300) | 31 (ND-140) | ND |
| OCDDs | 570 (ND-1,500) | 1,461 (ND-11,000) | 30 (ND-100) |
| Total CDDs | 772 (ND-2,770) | 1,492 (ND-11,140) | 30 (ND-100) |
| TCDFs | ND | 29 (ND-120) | ND |
| PeCDFs | ND | 1 (ND-10) | ND |
| HxCDFs | ND | 7 (ND-35) | ND |
| HpCDFs | ND | 9 (ND-60) | ND |
| OCDFs | 43 (ND-230) | 16 (ND-160) | ND |
| Total CDFs | 43 (ND-230) | 65 (ND-262) | ND |

[a] Data collected in 1987 in Ontario Canada; range presented in parentheses.

Source: Pearson et al. (1990).

Table 3-10. Dioxin/Furan Levels in Four Background Soil Samples from Elk River, Minnesota (ppt)[a]

| Congener | Tilled (n = 2) | Untilled (n = 2) |
|---|---|---|
| 2,3,7,8-TCDD | ND | ND |
| Total TCDD | ND | ND |
| 1,2,3,7,8-PeCDD | ND | ND |
| Total PeCDD | ND-38 | ND |
| 1,2,3,4,7,8-HxCDD | ND | ND |
| 1,2,3,6,7,8-HxCDD | ND | ND-14 |
| 1,2,3,7,8,9-HxCDD | ND-8.7 | ND-9.9 |
| Total HxCDD | 12-99 | 29-53 |
| 1,2,3,4,6,7,8-HpCDD | 37-360 | 78-300 |
| Total HpCDD | 62-640 | 150-530 |
| OCDD | 340-3300 | 680-2300 |
| 2,3,7,8-TCDF | ND | ND |
| Total TCDF | ND-1.2 | ND |
| 1,2,3,7,8-PeCDF | ND | ND |
| 2,3,4,7,8-PeCDF | ND | ND |
| Total PeCDF | ND-41 | 18-45 |
| 1,2,3,4,7,8-HxCDF | ND | ND |
| 1,2,3,6,7,8-HxCDF | ND | ND |
| 2,3,4,6,7,8-HxCDF | ND | ND-7.1 |
| 1,2,3,7,8,9-HxCDF | ND | ND |
| Total HxCDF | 6.7-86 | 20-150 |
| 1,2,3,4,6,7,8-HpCDF | 11-80 | 26-72 |
| 1,2,3,4,7,8,9-HpCDF | ND | ND |
| Total HpCDF | 30-260 | 30-82 |
| OCDF | ND-270 | 60-120 |

[a] ND = Nondetected. Detection limits vary from 0.75 ppt to 2.9 ppt on a congener-specific basis.

Source: Reed et al. (1990).

Table 3-11. Dioxin/Furan Levels in British Columbia Soils

| Sample Category[a] | Dioxin and Furan Concentrations (pg/g)[b] | | I-$TEQ_{DF}$ (pg/g)[b,c] | |
|---|---|---|---|---|
| | Range | Mean[d] | Range | Mean[d] |
| Background Soil<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND[e]<br>ND - 32.0 | ND (53)<br>3.2 (53) | 0.0 - 57.0 | 5.0 (53)[f] |
| Primary Soil (all sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 85.0<br>ND - 520.0 | 5.2 (31)<br>47.9 (31) | 0.0 - 2580.0 | 252.3 (31) |
| Primary Soil (chemical sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 85.0<br>ND - 520.0 | 8.4 (18)<br>60.3 (18) | 0.0 - 2580.0 | 418.5 (18) |
| Primary Soil (combustion sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 3.5<br>ND - 160.0 | 0.8 (13)<br>30.7 (13) | 0.0 - 125.7 | 22.3 (13) |
| Secondary Soil (all sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 550.0<br>ND - 550.0 | 5.4 (137)<br>25.1 (137) | 0.0 - 18721.8 | 241.7 (137) |
| Secondary Soil (chemical sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 550.0<br>ND - 550.0 | 15.4 (47)<br>60.7 (47) | 0.0 - 18721.8 | 668.6 (47) |
| Secondary Soil (combustion sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 5.6<br>ND - 180.0 | 0.09 (90)<br>6.5 (90) | 0.0 - 472.6 | 18.7 (90) |

[a] Background samples were believed to be indicative of ambient levels of dioxins and furans in the environment. Primary samples were collected immediately at a potential source of contamination. Secondary samples were collected from areas directly impacted by the primary source and could be used to indicate movement of contaminants.

[b] Concentrations in picograms/gram (pg/g) dry weight.

[c] I-$TEQ_{DF}$s are the sum of seventeen 2,3,7,8-substituted dioxins and furans after the concentration of each individual dioxin or furan is multiplied by its International Toxicity Equivalency Factor (I-$TEF_{DF}$). For samples with nondetected levels of a dioxin or furan, zero was used as the concentration for the I-$TEQ_{DF}$ calculation.

[d] Numbers in parentheses indicate the number of samples (n) used to calculate mean.

[e] ND = Not Detected.

[f] When the total TEQ was recalculated using $TEF_{DF}$-$WHO_{98}$s, the $TEQ_{DF}$-$WHO_{98}$ was 4.4 pg/g.

Source: BC Environment (1995).

Table 3-12 Number of Positive Soil Samples and CDD/CDF Concentrations in Background, Urban, and Impacted Sites Near a Waste-to-Energy Facility in Ohio

| | Background | | Urban | | Impacted | |
|---|---|---|---|---|---|---|
| | No. of Positive Samples | Mean Conc. (ppt) (nondetects = ½ LOD) | No. of Positive Samples | Mean Conc. (ppt) (nondetects = ½ LOD) | No. of Positive Samples | Mean Conc. (ppt) (nondetects = ½ LOD) |
| 2,3,7,8-TCDD | 2/3 | 0.39 | 15/18 | 2.27 | 3/3 | 28.5 |
| 1,2,3,7,8-PeCDD | 0/3 | 0.14 | 18/18 | 6.58 | 3/3 | 180.0 |
| 1,2,3,4,7,8-HxCDD | 1/3 | 0.35 | 18/18 | 6.14 | 3/3 | 142.3 |
| 1,2,3,6,7,8-HxCDD | 3/3 | 0.82 | 18/18 | 10.9 | 3/3 | 137.8 |
| 1,2,3,7,8,9-HxCDD | 3/3 | 1.23 | 18/18 | 10.8 | 3/3 | 201.6 |
| 1,2,3,4,6,7,8-HpCDD | 3/3 | 17.7 | 18/18 | 190.1 | 3/3 | 765.2 |
| OCDD | 3/3 | 160.9 | 18/18 | 1560.2 | 3/3 | 1495.4 |
| 2,3,7,8-TCDF | 0/3 | 0.45 | 18/18 | 4.12 | 3/3 | 85.9 |
| 1,2,3,7,8-PeCDF | 0/3 | 0.17 | 17/18 | 5.50 | 3/3 | 139.6 |
| 2,3,4,7,8-PeCDF | 1/3 | 0.21 | 17/18 | 7.56 | 3/3 | 199.9 |
| 1,2,3,4,7,8-HxCDF | 1/3 | 0.19 | 15/18 | 8.06 | 3/3 | 196.8 |
| 1,2,3,6,7,8-HxCDF | 3/3 | 0.52 | 17/18 | 8.12 | 3/3 | 209.1 |
| 1,2,3,7,8,9-HxCDF | 0/3 | 0.15 | 6/18 | 0.51 | 3/3 | 11.6 |
| 2,3,4,6,7,8-HxCDF | 3/3 | 0.64 | 18/18 | 6.99 | 3/3 | 156.7 |
| 1,2,3,4,6,7,8-HpCDF | 3/3 | 4.06 | 18/18 | 41.7 | 3/3 | 641.0 |
| 1,2,3,4,7,8,9-HpCDF | 1/3 | 0.27 | 16/18 | 3.82 | 3/3 | 57.9 |
| OCDF | 3/3 | 10.72 | 18/18 | 44.3 | 3/3 | 184.5 |
| Mean Total I-$TEQ_{DF}$, ppt (nondetects = ½ LOD) | -- | 1.4 | -- | 19.2 | -- | 356.0 |
| Mean Total I-$TEQ_{DF}$, ppt (nondetects = 0) | -- | 1.1 | -- | 19.2 | -- | 356.0 |
| Mean Total $TEQ_{DF}$-$WHO_{98}$, ppt (nondetects = ½ LOD) | -- | 1.3 | -- | 21.0 | -- | 444.5 |
| Mean Total $TEQ_{DF}$-$WHO_{98}$, ppt (nondetects = 0) | -- | 0.92 | -- | 21.0 | -- | 444.5 |

Source: U.S. EPA (1996).

Table 3-13. Mean Background CDD/CDF Profiles for Soil

| 2,3,7,8-Substituted CDD/CDFs | Rural Background[a] | | Urban Background[b] | | CDD/CDF Homologue Groups | Rural Background[c] | | Urban Background[d] | |
|---|---|---|---|---|---|---|---|---|---|
| | Concentra-tion (ppt) | Fraction of Total CDD/CDFs | Concentra-tion (ppt) | Fraction of Total CDD/CDFs | | Concentra-tion (ppt) | Fraction of Total CDD/CDFs | Concentra-tion (ppt) | Fraction of Total CDD/CDFs |
| 2,3,7,8-TCDD | 0.24 | 0.0002 | 1.13 | 0.0003 | TCDD | 2.3 | 0.003 | 36.1 | 0.012 |
| 1,2,3,7,8-PeCDD | 0.29 | 0.0002 | 3.3 | 0.0007 | PeCDD | 4.1 | 0.005 | 18.8 | 0.006 |
| 1,2,3,4,7,8-HxCDD | 0.59 | 0.0005 | 3.4 | 0.0008 | HxCDD | 22.7 | 0.028 | 31.7 | 0.010 |
| 1,2,3,6,7,8-HxCDD | 4.59 | 0.0037 | 6.1 | 0.0014 | HpCDD | 114.7 | 0.140 | 194.4 | 0.063 |
| 1,2,3,7,8,9-HxCDD | 2.88 | 0.0023 | 6.2 | 0.0014 | OCDD | 565.1 | 0.688 | 2,596 | 0.846 |
| 1,2,3,4,6,7,8-HpCDD | 95.6 | 0.076 | 125.1 | 0.0280 | TCDF | 6.8 | 0.008 | 39.1 | 0.013 |
| OCDD | 855.2 | 0.683 | 3,926 | 0.8776 | PeCDF | 12.7 | 0.015 | 40.8 | 0.013 |
| 2,3,7,8-TCDF | 1.67 | 0.0013 | 1.98 | 0.0004 | HxxCDF | 21.9 | 0.027 | 23.5 | 0.008 |
| 1,2,3,7,8-PeCDF | 0.46 | 0.0004 | 2.75 | 0.0006 | HpCDF | 37.3 | 0.045 | 46.4 | 0.015 |
| 2,3,4,7,8-PeCDF | 0.62 | 0.0005 | 3.91 | 0.0009 | OCDF | 33.5 | 0.041 | 40.2 | 0.013 |
| 1,2,3,4,7,8-HxCDF | 1.35 | 0.0011 | 5.42 | 0.0012 | | | | | |
| 1,2,3,6,7,8-HxCDF | 0.81 | 0.0006 | 4.78 | 0.0011 | | | | | |
| 1,2,3,7,8,9-HxCDF | 0.55 | 0.0004 | 0.15 | 0.00003 | | | | | |
| 2,3,4,6,7,8-HxCDF | 0.96 | 0.0008 | 3.74 | 0.0008 | | | | | |
| 1,2,3,4,6,7,8-HpCDF | 24.9 | 0.0199 | 22.7 | 0.0051 | | | | | |
| 1,2,3,4,7,8,9-HpCDF | 0.81 | 0.0006 | 1.98 | 0.0004 | | | | | |
| OCDF | 53.6 | 0.0428 | 25.5 | 0.0057 | | | | | |
| TOTAL | 1045.0 | 0.835 | 4144.5 | 0.926 | TOTAL | 821.3 | 1.00 | 3067.1 | 1.00 |

[a] Based on data from Reed et al. (1990), BC Environment (1995), U.S. EPA (1996), MRI (1992), and Tewhey Associates (1997).
[b] Based on data from U.S. EPA (1996) and NIH (1995).
[c] Based on data from Reed et al. (1990), Birmingham (1990), Pearson et al. (1990), BC Environment (1995), U.S. EPA (1985), U.S. EPA (1996), MRI (1992), and Tewhey Associates (1997).
[d] Based on data from Birmingham (1990), NIH (1995), Pearson et al. (1990), and U.S. EPA (1996).

Note: See Table 3-14 for information on the sampling locations and numbers of samples from these studies.

Table 3-14. $TEQ_{DF}$-$WHO_{98}$ Concentrations of CDD/CDFs in North American Soil (ppt)
(nondetects = 0)

| Reference | 2,3,7,8-TCDD | 1,2,3,7,8-PeCDD | 2,3,7,8-HxCDDs | 1,2,3,4,6,7,8-HpCDD | OCDD | 2,3,7,8-TCDF | 1,2,3,7,8-PeCDF | 2,3,4,7,8-PeCDF | 2,3,7,8-HxDCFs | 2,3,7,8-HpCDFs | OCDF | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| RURAL BACKGROUND | | | | | | | | | | | | |
| BC Environment, 1995<br>British Columbia (n = 53)<br>background | 0 | 0.16 | 1.7 | 1.4 | 0.068 | 0.32 | 0.016 | 0.090 | 0.22 | 0.44 | 0.0077 | 4.41 |
| Birmingham, 1990<br>Ontario (n = 30)<br>rural, background | -- | -- | -- | -- | 0.070 | -- | -- | -- | -- | -- | 0 | -- |
| MRI, 1992*<br>Connecticut (n = 34)<br>background | 0.61 | 0.87 | 0.67 | 0.55 | 0.081 | 0.48 | 0.090 | 1.18 | 1.0 | 0.18 | 0.0026 | 5.74 |
| Pearson et al., 1990<br>Ontario (n = 43)<br>rural, background | -- | -- | -- | -- | 0.0038 | -- | -- | -- | -- | -- | 0 | -- |
| Reed et al., 1990<br>Minnesota (n = 4)<br>semi-rural, background | 0 | 0 | 0.82 | 1.9 | 0.17 | 0 | 0 | 0 | 0.18 | 0.47 | 0.011 | 3.58 |
| Tewhey Associates, 1997<br>Maine (n = 8)<br>background | 0.28 | 0.43 | 0.65 | 0.70 | 0.097 | 0.040 | 0.010 | 0.25 | 0.28 | 0.15 | 0.0040 | 2.89 |
| U.S. EPA, 1985<br>Illinois (n = 13)<br>residential, background | 0.15 | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- |
| U.S. EPA, 1985<br>Minnesota (n = 4)<br>natural, background | 0 | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- |
| U.S. EPA, 1985<br>Ohio (n = 22)<br>residential, background | 1.1 | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- |
| U.S. EPA, 1985<br>Ohio (n = 5)<br>residential, background | -- | -- | -- | -- | 0.24 | -- | -- | -- | -- | -- | 0 | -- |
| U.S. EPA, 1985<br>Minnesota (n = 3)<br>natural, background | -- | -- | -- | -- | 0.014 | -- | -- | -- | -- | -- | 0 | -- |

Table 3-14. $TEQ_{DF}$-$WHO_{98}$ Concentrations of CDD/CDFs in North American Soil (ppt) (continued)
(nondetects = 0)

| Reference | 2,3,7,8-TCDD | 1,2,3,7,8-PeCDD | 2,3,7,8-HxCDDs | 1,2,3,4,6,7,8-HpCDD | OCDD | 2,3,7,8-TCDF | 1,2,3,7,8-PeCDF | 2,3,4,7,8-PeCDF | 2,3,7,8-HxDCF | 2,3,7,8-HpCDFs | OCDF | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| U.S. EPA, 1996 Ohio (n = 3) background | 0.30 | 0 | 0.23 | 0.18 | 0.016 | 0 | 0 | 0.028 | 0.12 | 0.042 | 0.0011 | 0.92 |
| MEAN | 0.30 | 0.29 | 0.81 | 0.96 | 0.077 | 0.17 | 0.023 | 0.31 | 0.37 | 0.26 | 0.0030 | 3.56 |
| S.D. | 0.36 | 0.33 | 0.48 | 0.64 | 0.077 | 0.20 | 0.034 | 0.44 | 0.33 | 0.17 | 0.0038 | -- |
| WEIGHTED MEAN | 0.35 | 0.41 | 1.18 | 1.05 | 0.058 | 0.33 | 0.039 | 0.46 | 0.49 | 0.32 | 0.0032 | 4.70 |
| URBAN BACKGROUND | | | | | | | | | | | | |
| Birmingham, 1990 Ontario (n = 47) urban, background | -- | -- | -- | -- | 0.16 | -- | -- | -- | -- | -- | 0.0059 | -- |
| Nestrick et al., 1986 Michigan (n = 20) urban/indus. bckgrd. | 2.1 | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- | -- |
| NIH, 1995 Maryland (n = 37) urban | 0.056 | 0.098 | 0.34 | 0.60 | 0.63 | 0.028 | 0.0070 | 0.14 | 0.27 | 0.039 | 0.0006 7 | 2.21 |
| Pearson et al., 1990 Ontario (n = 29) urban, background | -- | -- | -- | -- | 0.25 | -- | -- | -- | -- | -- | 0.0050 | -- |
| U.S. EPA, 1985 Michigan (n = 6) public areas | -- | -- | -- | -- | 0.51 | -- | -- | -- | -- | -- | 0.032 | -- |
| U.S. EPA, 1996 Ohio (n = 18) urban | 2.2 | 6.5 | 2.8 | 1.90 | 0.16 | 0.37 | 0.27 | 3.8 | 2.6 | 0.45 | 0.0044 | 21.0 |
| MEAN | 1.5 | 3.3 | 1.6 | 1.3 | 0.34 | 0.20 | 0.14 | 2.0 | 1.4 | 0.25 | 0.0096 | 11.9 |
| S.D. | 0.99 | 3.2 | 1.2 | 0.65 | 0.19 | 0.17 | 0.13 | 1.8 | 1.1 | 0.21 | 0.011 | -- |
| WEIGHTED MEAN | 1.1 | 2.2 | 1.1 | 1.0 | 0.32 | 0.14 | 0.09 | 1.3 | 1.0 | 0.18 | 0.0054 | 8.6 |

* Calculated using nondetects = ½ LOD. Proportion of nondetects ranged from 3 to 11 percent of samples per each analyte, with the exception of 2,3,7,8-TCDD and 1,2,3,7,8-HxCDF, which had 56 and 49 percent nondetects, respectively.

Table 3-15. CDD/CDF Levels in British Columbia Sediments

| Sample Category[a] | Dioxin and Furan Concentrations (pg/g)[b] | | I-$TEQ_{DF}$s (pg/g)[b,c] | |
|---|---|---|---|---|
| | Range | Mean[d] | Range | Mean[d] |
| Background Sediment<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND[e]<br>ND - 17.0 | ND (12)<br>1.4 (12) | 0.0 - 24.4 | 3.9 (12) |
| Secondary Sediment (all sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 2.7<br>ND - 33.0 | 0.2 (21)<br>3.5 (21) | 0.0 - 172.0 | 32.5 (21) |
| Secondary Sediment (chemical sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 2.7<br>ND - 33.0 | 0.2 (14)<br>3.8 (14) | 0.0 - 172.0 | 42.1 (14) |
| Secondary Sediment (combustion sources)<br>• 2,3,7,8-TCDD<br>• 2,3,7,8-TCDF | ND - 1.1<br>ND - 12.0 | 0.2 (7)<br>3.0 (7) | 0.0 - 63.6 | 13.2 (7) |

[a] Background samples were believed to be indicative of ambient levels of dioxins and furans in the environment. Secondary samples were collected from areas directly impacted by the primary source, and could be used to indicate movement of contaminants.

[b] Concentrations in picograms/gram (pg/g) dry weight.

[c] I-$TEQ_{DF}$s are the sum of 17 2,3,7,8-substituted dioxins and furans after the concentration of each individual dioxin or furan is multiplied by its International Toxicity Equivalency Factor (I-$TEF_{DF}$). For samples with nondetected levels of a dioxin or furan, zero was used as the concentration for the I-$TEQ_{DF}$ calculation.

[d] Numbers in parentheses indicate the number of samples (n) used to calculate mean.

[e] ND = Not Detected.

Source: BC Environment (1995).

Table 3-16. $TEQ_{DF}$ Concentrations (ppt) and Ratios of 2,3,7,8-Substituted CDD/CDF Concentrations to Total CDD/CDF Concentrations for the Most Recent Sediment Core Sampling Periods for 11 U.S. Lakes

| Lake | I-$TEQ_{DF}$ (ppt) | | $TEQ_{DF}$-$WHO_{98}$ (ppt) | | Ratio of 2,3,7,8-CDD/CDFs to Total CDD/CDFs | | Range of Dates Represented by Uppermost Core Section |
|---|---|---|---|---|---|---|---|
| | n = ½ LOD | nd = 0 | n = ½ LOD | nd = 0 | n = ½ LOD | nd = 0 | |
| Chandler Lake, AK | 0.11 | 0.11 | 0.12 | 0.12 | 0.47 | 0.47 | 1956-1993 |
| Canandaigua Lake, NY | 15.0 | 14.3 | 16.3 | 15.5 | 0.66 | 0.63 | 1981-1991 |
| Skaneateles Lake, NY | 10.1 | 9.1 | 10.8 | 9.7 | 0.73 | 0.71 | 1984-1991 |
| Great Sacandaga Reservoir, NY | 6.4 | 4.9 | 6.1 | 4.8 | 0.75 | 0.49 | 1974-1983 |
| Santeetlah Reservoir, NC | 15.6 | 13.1 | 15.4 | 13.0 | 0.70 | 0.64 | 1974-1983 |
| Blue Ridge Reservoir, GA | 5.6 | 4.8 | 4.9 | 4.2 | 0.75 | 0.71 | 1974-1983 |
| Deer Creek Reservoir, UT | 1.2 | 1.0 | 1.0 | 0.83 | 0.86 | 0.85 | 1973-1982 |
| Echo Lake, UT | 0.82 | 0.68 | 0.67 | 0.50 | 0.91 | 0.90 | 1973-1982 |
| Panguitch Lake, UT | 0.91 | 0.76 | 0.82 | 0.60 | 0.74 | 0.73 | 1976-1985 |
| Ozette Lake, WA | 1.2 | 1.1 | 1.3 | 1.2 | 0.56 | 0.56 | 1977-1985 |
| Beaver Lake, WA | 0.98 | 0.86 | 0.97 | 0.80 | 0.60 | 0.59 | 1974-1985 |
| Mean | 5.3 | 4.6 | 5.3 | 4.7 | 0.70 | 0.66 | -- |

Source: Cleverly et al. (1996); Versar (1996a)

Table 3-17. CDD/CDF and PCB Concentrations and Flux for 11 U.S. Lakes/Reservoirs

| | Total Concentration (pg/g, dry weight)[a] | | Flux (pg/cm$^2$-yr)[a] | |
|---|---|---|---|---|
| | CDD/CDFs | Dioxin-Like PCBs | CDD/CDFs | Dioxin-Like PCBs |
| Chandler Lake, AK | 9.1 | 34.0 | 0.051 | 0.19 |
| Canandaigua Lake, NY | 1790.6 | 2115.5 | 86.9 | 102.7 |
| Skaneateles Lake, NY | 1338.4 | 974.6 | 58.6 | 42.7 |
| Great Sacandaga Reservoir, NY | 1257.4 | 865.7 | 82.0 | 56.5 |
| Santeetlah Reservoir, NC | 2916.2 | 522.0 | 190.0 | 34.0 |
| Blue Ridge Reservoir, GA | 1785.8 | 218.0 | 158.6 | 19.4 |
| Deer Creek Reservoir, UT | 255.3 | 303.6 | 46.4 | 55.2 |
| Echo Lake, UT | 236.9 | 125.9 | 39.5 | 21.0 |
| Panguitch Lake, UT | 265.4 | 76.0 | 15.5 | 4.5 |
| Ozette Lake, WA | 203.4 | 103.0 | 7.9 | 4.0 |
| Beaver Lake, WA | 132.0 | 39.0 | 31.6 | 9.3 |
| Mean | 926.4 | 488.8 | 65.2 | 31.8 |

[a] Nondetects set to one-half the detection limit.

Source: Cleverly et al. (1996); Versar (1996a).

Table 3-18. Average Total Concentrations of CDD/CDFs for Sediments (pg/g)

| Vietnamese River Sediments | | Lake Sediments | |
|---|---|---|---|
| Site | Concentration | Site | Concentration |
| Saigon River | 6,800 | Lake Huron | 1,240 |
| Dong Nai River | 1,200 | Lake Michigan | 1,600 |
| Red River | 240 | Lake Erie | 2,150 |
| | | Lake Ontario | 11,000 |
| | | Siskiwit Lake | 730 |
| | | Lake Zurich | 1,500 |
| | | Lake Balderg | 1,500 |
| | | Lake Lugano | 2,000 |

Source: Schecter et al. (1989a)

Table 3-19. Mean Background Profiles for Sediment[a]

| 2,3,7,8-Substituted CDD/CDFs | Concentration (ppt) | Fraction of Total CDD/CDFs | CDD/CDF Homologue Groups | Concentration (ppt) | Fraction of Total CDD/CDFs |
|---|---|---|---|---|---|
| 2,3,7,8-TCDD | 0.26 | 0.0003 | TCDD | 6.5 | 0.007 |
| 1,2,3,7,8-PeCDD | 0.95 | 0.0010 | PeCDD | 9.1 | 0.0100 |
| 1,2,3,4,7,8-HxCDD | 1.8 | 0.0020 | HxCDD | 43.3 | 0.0468 |
| 1,2,3,6,7,8-HxCDD | 4.7 | 0.0051 | HpCDD | 199.1 | 0.2149 |
| 1,2,3,7,8,9-HxCDD | 4.1 | 0.0044 | OCDD | 400.5 | 0.4323 |
| 1,2,3,4,6,7,8-HpCDD | 100.6 | 0.1086 | TCDF | 25.7 | 0.0278 |
| OCDD | 400.5 | 0.4323 | PeCDF | 17.9 | 0.0193 |
| 2,3,7,8-TCDF | 1.6 | 0.0017 | HxCDF | 38.0 | 0.0411 |
| 1,2,3,7,8-PeCDF | 0.91 | 0.0010 | HpCDF | 82.8 | 0.0893 |
| 2,3,4,7,8-PeCDF | 1.5 | 0.0016 | OCDF | 103.6 | 0.1118 |
| 1,2,3,4,7,8-HxCDF | 1.9 | 0.0020 | | | |
| 1,2,3,6,7,8-HxCDF | 0.003 | 0.0000 | | | |
| 1,2,3,7,8,9-HxCDF | 0.02 | 0.0000 | | | |
| 2,3,4,6,7,8-HxCDF | 1.7 | 0.0018 | | | |
| 12,3,4,6,7,8-HpCDF | 0.01 | 0.0000 | | | |
| 1,2,3,4,7,8,9-HpCDF | 2.1 | 0.0023 | | | |
| OCDF | 83.0 | 0.0896 | | | |
| TOTAL | 605.4 | 0.6536 | | 926.40 | 1.0 |

[a] Based on Cleverly et al. (1996); Versar (1996a).

Table 3-20. $TEQ_{DF}$-$WHO_{98}$ Concentrations of CDD/CDFs in North American Sediment (ppt)
(nondetects = ½ LOD)

| Location | 2,3,7,8-TCDD | 1,2,3,7,8-PeCDD | 2,3,7,8-HxCDDs | 1,2,3,4,6,7,8-HpCDD | OCDD | 2,3,7,8-TCDF | 1,2,3,7,8-PeCDF | 2,3,4,7,8-PeCDF | 2,3,7,8-HxDCFs | 2,3,7,8-HpCDFs | OCDF | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Chandler Lake, AK<br>Cs date = 1952 | 0.016 | 0.025 | 0.010 | 0.002 | 0.0003 | 0.018 | 0.003 | 0.022 | 0.021 | 0.002 | 0.00002 | 0.012 |
| Canadaigua Lake,NY<br>Cs date = 1981 | 0.782 | 3.91 | 3.512 | 2.720 | 0.069 | 0.844 | 0.153 | 2.635 | 1.274 | 0.369 | 0.011 | 16.3 |
| Skaneateles Lake, NY<br>Cs date = 1984 | 0.687 | 2.610 | 1.672 | 1.320 | 0.072 | 0.404 | 0.147 | 2.385 | 1.189 | 0.262 | 0.005 | 10.8 |
| Great Sacandaga Reservoir, NY<br>Cs date = 1974 | 0.098 | 0.726 | 0.613 | 0.487 | 0.054 | 0.382 | 0.098 | 1.610 | 0.983 | 0.989 | 0.023 | 6.1 |
| Santeetlah Reservoir, NC<br>Cs date = 1974 | 0.390 | 2.160 | 4.054 | 4.310 | 0.099 | 0.210 | 0.043 | 0.685 | 1.861 | 1.554 | 0.039 | 15.4 |
| Blue Ridge Reservoir, GA<br>Cs date = 1974 | 0.279 | 0.682 | 0.854 | 1.470 | 0.087 | 0.090 | 0.023 | 0.327 | 0.517 | 0.596 | 0.025 | 4.9 |
| Deer Creek Reservoir, UT<br>Cs date = 1973 | 0.106 | 0.124 | 0.161 | 0.173 | 0.018 | 0.073 | 0.017 | 0.182 | 0.154 | 0.037 | 0.001 | 1.0 |
| Echo Lake, UT<br>Cs = 1973 | 0.042 | 0.055 | 0.127 | 0.143 | 0.010 | 0.012 | 0.005 | 0.100 | 0.138 | 0.029 | 0.010 | 0.67 |
| Panguitch Lake, UT<br>Cs = 1976 | 0.025 | 0.124 | 0.169 | 0.195 | 0.016 | 0.027 | 0.014 | 0.137 | 0.090 | 0.025 | 0.0007 | 0.82 |
| Ozette Lake, WA<br>Cs = 1977 | 0.246 | 0.345 | 0.270 | 0.143 | 0.009 | 0.066 | 0.007 | 0.111 | 0.100 | 0.015 | 0.0003 | 1.3 |
| Beaver Lake, WA<br>Cs = 1974 | 0.299 | 0.125 | 0.163 | 0.104 | 0.006 | 0.021 | 0.007 | 0.163 | 0.094 | 0.008 | 0.0002 | 0.99 |
| Mean | 0.270 | 0.990 | 1.055 | 1.006 | 0.040 | 0.195 | 0.047 | 0.760 | 0.584 | 0.353 | 0.010 | 5.31 |
| S.D. | 0.250 | 1.249 | 1.368 | 1.318 | 0.035 | 0.246 | 0.055 | 0.932 | 0.607 | 0.485 | 0.012 | 5.83 |

Source: Cleverly et al. (1996); Versar (1996a).

Table 3-21. Background Data for Fish from the National Bioaccumulation Study

| Congener | No. of Background Sites | Concentration Range[a] (pg/g) | Mean Conc.[a] (pg/g) | Standard Deviation[a] (pg/g) | Median Conc.[a] (pg/g) |
|---|---|---|---|---|---|
| 2,3,7,8-TCDD | 34 | 0.06 - 2.26 | 0.56 | 0.38 | 0.50 |
| 2,3,7,8-TCDF | 34 | 0.10 - 13.73 | 1.61 | 2.51 | 0.90 |
| 1,2,3,7,8-PeCDD | 33 | 0.15 - 2.67 | 0.77 | 0.54 | 0.54 |
| 1,2,3,7,8-PeCDF | 34 | 0.10 - 1.90 | 0.43 | 0.31 | 0.39 |
| 2,3,4,7,8-PeCDF | 34 | 0.10 - 1.39 | 0.50 | 0.36 | 0.42 |
| Total HxCDDs | 30 | ND - 3.57 | 0.39 | 0.8 | ND |
| Total HxCDFs | 29 | ND - 2.59 | 0.22 | 0.66 | ND |

Source: U.S. EPA (1992)

[a] Concentrations are picograms per gram (pg/g) wet weight. The mean, median, and standard deviation were calculated using one-half the detection limit for samples that were below the detection limit. In cases where multiple samples were analyzed per site, the value used represents the highest concentration.

Note: ND = nondetect.

Table 3-22. Levels of CDD and CDF I-$TEQ_{DF}$s in Fish From the Southern Mississippi Region

| Food Sample | Number of Observations | I-$TEQ_{DF}$ pg/g sample[a] |
|---|---|---|
| Catfish (farm-raised) Nuggets[b] | 3 | 1.19, 2.64, 2.57 |
| Mullet Fillet | 2 | 0.089, 0.027 |
| Spanish Mackerel Fillet | 1 | 0.72 |
| American Oyster Meat | 3 | 0.62, 0.53, 0.60 |
| Blue Crab | | |
| Claw Meat | 3 | 0.06, 0.10, 0.09 |
| Body (soft-shell) | 3 | 1.09, 1.14, 1.44 |
| Crawfish | | |
| Tail Muscle | 2 | 0.033, 0.087 |
| Head and Digestive Gland | 2 | 2.34, 1.55 |

[a] One-half the detection limit was used in calculating the I-$TEQ_{DF}$s.
[b] Nuggets are small pieces of fillet.

Source: Cooper et al. (1995)

Table 3-23. Summary of CDD, CDF, and PCB Analyses in Farm Raised Catfish from the Southeastern United States[a]

| Sample Type | Number of Observations | I-$TEQ_{DF}$ (mean) pg/g lipid | $TEQ_P$-$WHO_{98}$ (mean) pg/g lipid [b] | $TEQ_{DFP}$-$WHO_{98}$ (mean) pg/g lipid |
|---|---|---|---|---|
| Catfish Nuggets from Mississippi[c] | 3 | 9.7 | 1.43 | 11.13 |
| Catfish Fillet from Mississippi[c] | 3 | 22.67 | 2.66 | 25.33 |
| Catfish Fillet from Alabama | 1 | 13 | 0.92 | 13.92 |
| Catfish Fillet from Mississippi Agriculture Facility | 3 | 7.93 | 0.86 | 8.79 |
| Catfish Fillet from Arkansas Agriculture Facility (8% fish meal) | 2 | 40 | 3.42 | 43.42 |
| Feed - Mississippi (4% fish meal) | 1 | 7.2 | 3.31 | 10.51 |
| Feed - Arkansas (8% fish meal) | 1 | 61 | 0.19 | 61.19 |
| Sediment from Mississippi Agriculture Facility | 1 | 3.5 | 0.04 | 3.54 |

[a] One-half the detection limit was used in calculating the TEQs.
[b] Includes PCB 28, 52, 77, 101, 105, 118, 126, 138, 153, 156, 169, and 180.
[c] Purchased from same store and distributed by same supplier as those collected and analyzed in Cooper et al. (1995).

Source: Cooper et al. (1997) and Fiedler et al. (1998).

Table 3-24. Levels of PCBs in Fish Tissue, Bivalves, and Sediment at a Site Near a Pulp and Paper Mill

| Congener | | Upstream | Mill Vicinity | Downstream |
|---|---|---|---|---|
| ***Fish Tissue (Whole Body)*** | | | | |
| PCB77 | range (pg/g, wet wt.) | 26.0-120.6 | 14.2-1095.3 | 30.2-80.6 |
| | mean (pg/g, wet wt.) | 56.0 | 555.7 | 51.3 |
| | mean TEQ (pg/g, wet wt.) | 0.028 | 0.277 | 0.026 |
| PCB126 | range (pg/g, wet wt.) | 12.3-30.5 | 17.7 | 13.8-17.4 |
| | mean (pg/g, wet wt.) | 20.7 | 17.7 | 15.3 |
| | mean TEQ (pg/g, wet wt.) | 2.07 | 1.77 | 1.53 |
| PCB169 | range (pg/g, wet wt.) | 1.2-2.8 | 1.4 | 1.6-2.0 |
| | mean (pg/g, wet wt.) | 2.0 | 1.4 | 1.7 |
| | mean TEQ (pg/g, wet wt.) | 0.020 | 0.014 | 0.017 |
| Total $TEQ_P$-$WHO_{94}$ | | 2.12 | 2.06 | 1.57 |
| ***Bivalves*** | | | | |
| PCB77 | range (pg/g, wet wt.) | 101.7 | - | - |
| | mean (pg/g, wet wt.) | 101.7 | - | - |
| | mean TEQ (pg/g, wet wt.) | 0.05 | - | - |
| PCB126 | range (pg/g, wet wt.) | 19.4 | 11.5 | - |
| | mean (pg/g, wet wt.) | 19.4 | 11.5 | - |
| | mean TEQ (pg/g, wet wt.) | 1.94 | 1.15 | - |
| PCB169 | range (pg/g, wet wt.) | - | 0.7 | 3.3 |
| | mean (pg/g, wet wt.) | - | 0.7 | 3.3 |
| | mean TEQ (pg/g, wet wt.) | - | 0.007 | 0.033 |
| Total $TEQ_P$-$WHO_{94}$ | | 1.99 | 1.16 | 0.03 |
| ***Sediment*** | | | | |
| PCB77 | range (pg/g, wet wt.) | 9.5 | - | 27.8 |
| | mean (pg/g, wet wt.) | 9.5 | - | 27.8 |
| | mean TEQ (pg/g, wet wt.) | 0.005 | - | 0.009 |
| PCB126 | range (pg/g, wet wt.) | 0.9-1.1 | - | 1.38 |
| | mean (pg/g, wet wt.) | 1.0 | - | 1.38 |
| | mean TEQ (pg/g, wet wt.) | 0.10 | - | 0.14 |
| PCB169 | range (pg/g, wet wt.) | 1.5 | - | 0.5 |
| | mean (pg/g, wet wt.) | 1.5 | - | 0.5 |
| | mean TEQ (pg/g, wet wt.) | 0.015 | - | 0.005 |
| Total $TEQ_P$-$WHO_{94}$ | | 0.12 | - | 0.16 |

NOTE: Results of sample analyses that showed interference or had recoveries below 40 percent or above 120 percent (acceptability range specified by this study was 40 to 120 percent recovery) were not included in the data set used here. Half the detection (or quantification) limit was used for samples below the detection (or quantification) limit. $TEF_P$-$WHO_{94}$s used in calculating the $TEQ_P$-$WHO_{94}$s.

Source: Derived from Petreas, 1991.

Table 3-25. Mean CDD/CDF Profiles for Fish

| 2,3,7,8-Substituted CDD/CDFs | Freshwater Fish[a] | | Marine Fish[b] | | Shellfish[b] | |
|---|---|---|---|---|---|---|
| | Concentration Whole Weight (ppt) | Fraction of Total CDD/CDFs | Concentration Whole Weight (ppt) | Fraction of Total CDD/CDFs | Concentration Whole Weight (ppt) | Fraction of Total CDD/CDFs |
| 2,3,7,8-TCDD | 0.149 | 0.0164 | 0.079 | 0.0321 | 0.138 | 0.0149 |
| 1,2,3,7,8-PeCDD | 0.251 | 0.0276 | 0.163 | 0.0664 | 0.335 | 0.0362 |
| 1,2,3,4,7,8-HxCDD | 0.145 | 0.0160 | 0.079 | 0.0321 | 0.324 | 0.0350 |
| 1,2,3,6,7,8-HxCDD | 0.256 | 0.0282 | 0.412 | 0.1677 | 0.544 | 0.0588 |
| 1,2,3,7,8,9-HxCDD | 0.199 | 0.0218 | 0.114 | 0.0464 | 0.516 | 0.0558 |
| 1,2,3,4,6,7,8-HpCDD | 1.135 | 0.1249 | 0.477 | 0.1941 | 1.91 | 0.2062 |
| OCDD | 5.893 | 0.6483 | 0.785 | 0.3195 | 8.09 | 0.8745 |
| 2,3,7,8-TCDF | 0.696 | 0.0765 | 0.118 | 0.0479 | 0.679 | 0.0734 |
| 1,2,3,7,8-PeCDF | 0 | 0 | 0.037 | 0.0150 | 0.036 | 0.0039 |
| 2,3,4,7,8-PeCDF | 0.365 | 0.0402 | 0.057 | 0.0232 | 0.143 | 0.0155 |
| 1,2,3,4,7,8-HxCDF | 0 | 0 | 0.035 | 0.0141 | 0.091 | 0.0099 |
| 1,2,3,6,7,8-HxCDF | 0 | 0 | 0.018 | 0.0075 | 0.042 | 0.0045 |
| 1,2,3,7,8,9-HxCDF | 0 | 0 | 0 | 0 | 0 | 0 |
| 2,3,4,6,7,8-HxCDF | 0 | 0 | 0.011 | 0.0043 | 0.032 | 0.0035 |
| 1,2,3,4,6,7,8-HpCDF | 0 | 0 | 0.016 | 0.0142 | 0.098 | 0.0105 |
| 1,2,3,4,7,8,9-HpCDF | 0 | 0 | 0 | 0 | 0.003 | 0.0003 |
| OCDF | 0 | 0 | 0.038 | 0.0155 | 0.038 | 0.0041 |
| TOTAL | 9.09 | 1.0 | 2.46 | 1.0 | 9.25 | 1.0 |

[a] Based on data from Schecter et al. (1997).

[b] Based on data from Fiedler et al. (1997c).

Table 3-26. I-$TEQ_{DF}$s in Foods From Southern Mississippi

| Food Sample | Number of Observations | I-$TEQ_{DF}$ pg/g sample[a] |
|---|---|---|
| Catfish (farm-raised) Nuggets | 3 | 1.19, 2.64, 2.57 |
| Mullet Fillet | 2 | 0.089, 0.027 |
| Spanish Mackerel Fillet | 1 | 0.72 |
| American Oyster Meat | 3 | 0.62, 0.53, 0.60 |
| Blue Crab | | |
| Claw Meat | 3 | 0.06, 0.10, 0.09 |
| Body (soft-shell) | 3 | 1.09, 1.14, 1.44 |
| Crawfish | | |
| Tail Muscle | 2 | 0.033, 0.087 |
| Head and Digestive Gland | 2 | 2.34, 1.55 |
| Butter | 3 | 0.683, 0.770, 0.552 |
| Milk | 3 | 0.025, 0.026, 0.012 |
| Cheddar Cheese | 3 | 0.300, 0.247, 0.254 |
| Eggs | 3 | 0.038, 0.020, 0.019 |
| Ground Beef | 3 | 0.196, 0.254, 0.152 |
| Chicken | 3 | 0.043, 0.085, 0.053 |
| Chicken Liver | 3 | 0.031, 0.064, 0.070 |
| Sausage | 3 | 0.178, 0.221, 0.282 |

[a] One-half the detection limit was used in calculating the I-$TEQ_{DF}$s.

Source: Cooper et al. (1995)

Table 3-27. Summary of Dioxin/Furan Food Data Collected in the California State Air Resources Board Study

| Congener | Beef | | Pork | | Chicken | |
|---|---|---|---|---|---|---|
| | Fraction of composite samples that were positive | Concentration range of positive samples (pg/g)[a,b] | Fraction of composite samples that were positive | Concentration range of positive samples (pg/g)[a,b] | Fraction of composite samples that were positive | Concentration range of positive samples (pg/g)[a,b] |
| 2,3,7,8-TCDD | 0/8 | - | 0/8 | - | 3/8 | 0.31-1.67 |
| 1,2,3,7,8-PeCDD | 0/8 | - | 0/8 | - | 0/8 | - |
| 1,2,3,4,7,8/1,2,3,6,7,8-HxCDD | 3/8 | 0.72-3.96 | 2/8 | 2.83-3.50 | 1/8 | 2.29 |
| 1,2,3,7,8,9-HxCDD | 0/8 | - | 0/8 | - | 2/8 | 2.14-4.30 |
| 1,2,3,4,6,7,8-HpCDD | 7/8 | 3.53-8.95 | 8/8 | 3.04-45.50 | 7/8 | 1.10-35.20 |
| OCDD | 7/8 | 7.75-11.90 | 8/8 | 13.70-254.0 | 7/8 | 2.61-96.20 |
| 2,3,7,8-TCDF | 3/8 | 0.63-1.56 | 0/8 | - | 1/8 | 0.67 |
| 1,2,3,7,8-PeCDF | 0/8 | - | 0/8 | - | 0/8 | - |
| 2,3,4,7,8-PeCDF | 0/8 | - | 0/8 | - | 0/8 | - |
| 1,2,3,4,7,8-HxCDF | 0/8 | - | 0/8 | - | 0/8 | - |
| 1,2,3,6,7,8-HxCDF | 0/8 | - | 0/8 | - | 0/8 | - |
| 1,2,3,7,8,9-HxCDF | 0/8 | - | 0/8 | - | 0/8 | - |
| 2,3,4,6,7,8-HxCDF | 0/8 | - | 0/8 | - | 0/8 | - |
| 1,2,3,4,6,7,8-HpCDF | 4/8 | 0.48-1.15 | 7/8 | 1.57-10.60 | 6/8 | 1.01-24.60 |
| 1,2,3,4,7,8,9-HpCDF | 0/8 | - | 0/8 | - | 0/8 | - |
| OCDF | 0/8 | - | 5/8 | 1.24-9.36 | 2/8 | 3.79-26.00 |

Table 3-27. Summary of Dioxin/Furan Data Collected in the California State Air Resources Board Study (continued)

| Congener | Eggs | | Milk | | Fish (freshwater & saltwater) | |
|---|---|---|---|---|---|---|
| | Fraction of composite samples that were positive | Concentration range of positive samples (pg/g)[c] | Fraction of composite samples that were positive | Concentration range of positive samples (pg/g)[a,b] | Fraction of composite samples that were positive | Concentration range of positive samples (pg/g)[a,b] |
| 2,3,7,8-TCDD | 0/8 | - | 1/8 | 1.46 | 8/10 | 0.73-9.78 |
| 1,2,3,7,8-PeCDD | 0/8 | - | 0/8 | - | 6/10 | 1.67-23.6 |
| 1,2,3,4,7,8/1,2,3,6,7,8-HxCDD | 0/8 | - | 1/8 | 0.59 | 7/10 | 1.19-84.1 |
| 1,2,3,7,8,9-HxCDD | 0/8 | - | 0/8 | - | 4/10 | 3.91-38.9 |
| 1,2,3,4,6,7,8-HpCDD | 0/8 | - | 7/8 | 2.08-4.25 | 5/10 | 3.15-201 |
| OCDD | 1/8 | 1.30 | 6/8 | 2.23-6.12 | 10/10 | 4.37-1490 |
| 2,3,7,8-TCDF | 1/8 | 0.011 | 8/8 | 1.30-6.11 | 10/10 | 0.83-28.2 |
| 1,2,3,7,8-PeCDF | 0/8 | - | 0/8 | - | 0/10 | - |
| 2,3,4,7,8-PeCDF | 0/8 | - | 0/8 | - | 0/10 | - |
| 1,2,3,4,7,8-HxCDF | 0/8 | - | 0/8 | - | 0/10 | - |
| 1,2,3,6,7,8-HxCDF | 0/8 | - | 0/8 | - | 0/10 | - |
| 1,2,3,7,8,9-HxCDF | 0/8 | - | 0/8 | - | 0/10 | - |
| 2,3,4,6,7,8-HxCDF | 0/8 | - | 0/8 | - | 0/10 | - |
| 1,2,3,4,6,7,8-HpCDF | 1/8 | 0.065 | 1/8 | 0.70 | 2/110 | 2.21-92.9 |
| 1,2,3,4,7,8,9-HpCDF | 0/8 | - | 0/8 | - | 1/10 | 13.3 |
| OCDF | 0/8 | - | 0/8 | - | 0/10 | 0 |

[a] Concentration reported on a lipid weight basis.
[b] For some of the concentrations reported, the ratio of characteristic ions were outside the qualitative identification data quality objectives.
[c] Concentration reported on a whole weight basis.

Source: Stanley and Bauer (1989).

Table 3-28. Summary of U.S. Food Data from NCASI Study

| Food | Number of Samples | 2,3,7,8-TCDD Level[a] (Food basis, pg/kg) | 2,3,7,8-TCDD Level[a] (Lipid basis, pg/kg) | 2,3,7,8-TCDF Level[a] (Food basis, pg/kg) | 2,3,7,8-TCDF Level[a] (Lipid basis, pg/kg) |
|---|---|---|---|---|---|
| Milk | 1 | 1.8 | 48 | ND | ND |
| Half & Half | 2 | 7.2; 8.7 | 55; 67 | NR | NR |
| Ground beef | 3 | 17; 18; 62 | 71; 141; 352 | ND(3.8); ND(4.8); 5.2 | ND(16); ND(27); 41 |
| Corned beef hash | 14 | 7.2-20 | 54-144 | ND(5.9); ND(17); 4.7-12 | ND(39); ND(120); 33-103 |
| Beef hot dogs | 3 | 12; 15; 37 | 44; 56; 128 | ND(7.7); 11; 11 | ND(28); 38; 41 |
| Ground pork | 3 | ND(5.8); ND(6.5); ND(6.5) | ND(18); ND(22); ND(27) | 13; 13; 20 | 45; 53; 62 |
| Chicken broth | 3 | 1.1; 1.3; 1.5 | (lipid content unknown) | NR | NR |
| Coffee | 2 | ND(0.2); 0.08 | NR | NR | NR |
| Orange juice | 3 | ND(0.3); ND(0.3); ND(0.4) | NR | NR | NR |

NOTE: ND = Not detected; NR = Not reported

[a] Values in parentheses are detection limits.

Sources: Henry et al. (1992); LaFleur et al. (1990)

Table 3-29. Summary of Schecter et al. (1993a) Data on U.S. Foods

| | CDD/CDF I-TEQ$_{DF}$ (pg/g)[a] Assuming ND = 0.5 DL | CDD/CDF I-TEQ$_{DF}$ (pg/g)[a] Assuming ND = 0 | Number of Samples[b] |
|---|---|---|---|
| Haddock Fillet | 0.03 | 0.02 | 2 |
| Crunchy Haddock | 0.13 | 0.13 | 1 |
| Perch | 0.24 | 0.24 | 1 |
| Cod | 0.023 | 0.012 | 1 |
| Ground Beef | 1.5 | 1.5 | 1 |
| Beef Rib Sirloin Tip | 0.04 | 0.04 | 1 |
| Beef Rib Steak | 0.65 | 0.65 | 1 |
| Pork Chop | 0.26 | 0.26 | 1 |
| Cooked Ham | 0.029 | 0.024 | 1 |
| Lamb Sirloin | 0.4 | 0.4 | 1 |
| Lebanon Bologna | 0.12 | 0.11 | 1 |
| Chicken | 0.03 | 0.03 | 1 |
| Cottage Cheese | 0.04 | 0.04 | 1 |
| Blue Cheese | 0.73 | 0.70 | 1 |
| Cream Cheese | 0.38 | 0.38 | 1 |
| American Cheese | 0.31 | 0.31 | 1 |
| Heavy Cream | 0.35 | 0.33 | 1 |

ND = nondetect; DL = detection limit

[a] Concentrations reported on whole food, wet weight basis.
[b] Samples collected from a supermarket in New York.

Source: Schecter et al. (1993a)

Table 3-30. CDD/CDFs and PCBs in Foods from Five Regions of the United States

| Congener | Beef N = 9 | Chicken N = 7 | Ocean Fish N = 13 | Fresh Fish N = 10 | Pork N = 7 |
|---|---|---|---|---|---|
| % Lipid | 13.13 | 5.33 | 1.43 | 4.83 | 9.18 |
| CDD/Fs (pg/g, lipid-based) | | | | | |
| 2,3,7,8-TCDF | 0.488 | ND (1.88 EMPC)[a] | ND (11.6 EMPC)[a] | 14.4 | 1.97 |
| 2,3,7,8-TCDD | ND (.19)[b] | ND (.467)[b] | 2.3 | 3.09 | ND (.349 EMPC)[a] |
| 1,2,3,7,8-PeCDF | ND (.95)[b] | ND (2.34)[b] | ND (8.74)[b] | ND (7.59 EMPC)[a] | ND (1.36)[b] |
| 2,3,4,7,8-PeCDF | ND (.95)[b] | 2.6 | ND (8.74)[b] | 7.56 | ND (1.36)[b] |
| 1,2,3,7,8-PeCDD | ND (.95)[b] | ND (2.34)[b] | ND (8.74)[b] | 5.2 | ND (1.36)[b] |
| 1,2,3,4,7,8-HxCDF | ND (1.22 EMPC)[a] | ND (2.34)[b] | ND (10.8)[b] | ND (3.36 EMPC)[a] | ND (1.47 EMPC)[a] |
| 1,2,3,6,7,8-HxCDF | ND (2.15 EMPC)[a] | ND (2.84 EMPC)[a] | ND (16.6 EMPC)[a] | ND (19.9 EMPC)[a] | ND (6.46 EMPC)[a] |
| 2,3,4,6,7,8-HxCDF | ND (.95)[b] | ND (2.34)[b] | ND (9.51)[b] | ND (2.58)[b] | ND (1.36)[b] |
| 1,2,3,7,8,9-HxCDF | ND (.95)[b] | ND (2.91)[b] | ND (14.5)[b] | ND (2.58)[b] | ND (1.36)[b] |
| 1,2,3,4,7,8-HxCDD | ND (1.39 EMPC)[a] | ND (2.34)[b] | ND (8.74)[b] | 3.01 | ND (1.36)[b] |
| 1,2,3,6,7,8-HxCDD | 4.92 | ND (2.34)[b] | ND (8.74)[b] | 5.31 | 1.81 |
| 1,2,3,7,8,9-HxCDD | 1.04 | ND (2.34)[b] | ND (8.74)[b] | 4.11 | ND (1.36)[b] |
| 1,2,3,4,6,7,8-HpCDF | ND (10.8 EMPC)[a] | ND (5.49 EMPC)[a] | ND (48.7 EMPC)[a] | ND (31 EMPC)[a] | ND (20.2 EMPC)[a] |
| 1,2,3,4,7,8,9-HpCDF | ND (.95)[b] | ND (2.34)[b] | ND (8.74)[b] | ND (2.72)[b] | ND (1.36)[b] |
| 1,2,3,4,6,7,8-HpCDD | 20.9 | 8.09 | 11.7 | 23.5 | 17.1 |
| 1,2,3,4,6,7,8,9-OCDF | ND (3 EMPC)[a] | ND (4.67)[b] | ND (17.5)[b] | ND (5.17)[b] | ND (3.26 EMPC)[a] |
| 1,2,3,4,6,7,8,9-OCDD | 32.7 | 20.2 | 31.6 | 122 | 87.1 |
| I-$TEQ_{DF}$s (pg/g) | | | | | |
| lipid-based min. | 0.89 | 0.10 | 2.45 | 12.51 | 0.64 |
| lipid-based max. | 2.86 | 5.17 | 21.14 | 16.07 | 3.97 |
| whole-weight min. | 0.12 | 0.005 | 0.035 | 0.60 | 0.06 |
| whole-weight max. | 0.38 | 0.28 | 0.30 | 0.78 | 0.36 |
| PCBs (ng/g, lipid-based) | | | | | |
| # 118 | 0.717 | 3.70 | 22.3 | 36.3 | -- |
| # 114 | -- | -- | -- | -- | -- |
| # 153 | 0.629 | 2.08 | 27.4 | 39.3 | 0.785 |
| # 105 | -- | 1.47 | 8.34 | 12.4 | -- |
| # 138 | -- | 0.747 | 30.2 | 37.4 | 1.07 |
| # 128 | -- | -- | -- | 5.59 | -- |
| # 156 | -- | -- | -- | 7.27 | -- |
| # 180 | -- | 4.32 | 11.8 | 12.6 | -- |

[a] Nondetected due to an interference. An estimated maximum possible concentration (EMPC) is given as the detection limit.
[b] Nondetected, with the curve based detection limit in pg/g.
[c] Calculated using a value of zero for nondetected congeners.
[d] Calculated using the detection limit value for nondetected congeners.

Source: Schecter et al. (1997).

Table 3-31. I-$TEQ_{DF}$ Levels in Cow's Milk and Infant Formula from the United States and Thailand

| Product/Origin | Lipid-based I-$TEQ_{DF}$ (ppt) | Fat Content (%) |
|---|---|---|
| Cow's Milk/NY | 1.2 | 3.4 |
| 2% Cow's Milk/NY | 0.1 | 1.9 |
| Ultra Pasteurized Heavy Cream/NY | 0.38 | 39.1 |
| Similac Formula/NY | 0.004 | -[b] |
| Isomil Formula/NY | 0.003 | 3.2 |
| Prosoybee Formula/NY | 0.005[a] | 2.5 |
| Laclasoy UHT Soy Milk/Thailand | 0.23 | 2.9 |
| Thai-Danish UHT/Thailand | 0.32 | 3.5 |

[a] Contamination from cap liner.
[b] Not available.

Source: Schecter et al. (1989b).

Table 3-32. Maximum CDD/CDF Levels in Foods Collected in Canada (pg/g fresh weight) as Reported by Birmingham et al. (1989)

| | Eggs | Beef | Milk Products | Pork | Chicken |
|---|---|---|---|---|---|
| TCDD | ND | ND | NR | ND | ND |
| PeCDD | ND | ND | NR | ND | ND |
| HxCDD | ND | ND | NR | ND | ND |
| HpCDD | ND | ND | NR | ND | 15 |
| OCDD | 8[a] | 24[a] | NR | ND | 210 |
| | | | | | |
| TCDF | ND | ND | NR | ND | ND |
| PeCDF | ND | ND | NR | ND | ND |
| HxCDF | 5[a] | ND | NR | ND | ND |
| HpCDF | 7[a] | ND | NR | ND | ND |
| OCDF | 12[a] | ND | NR | ND | ND |
| I-$TEQ_{DF}$ | 0.59 | 0.29 | 0.11 | 0.03 | 0.39 |

NR = Not Reported.
ND = Not Detected; detection limits ranged from 0.1 to 0.7 pg/g.
[a] Data for foods of U.S. origin collected in Canada.

Source: Birmingham et al. (1989).

Table 3-33. Summary of TEQ Levels in Toronto (1992) and Montreal (1993)*

| Food Category | Composite Sample | CDD/CDF (ppt) I-$TEQ_{DF}$ | | Non-Ortho PCB (ppt) $TEQ_P$-$WHO_{94}$ | | Σ CDD/CDF/PCB TEQ (ppt) | |
|---|---|---|---|---|---|---|---|
| | | Toronto | Montreal | Toronto | Montreal | Toronto | Montreal |
| Meat | Beef Ground | 0.32 | 0.32 | 0.067 | 0.045 | 0.39 | 0.37 |
| | Beef Steak | 0.18 | 0.17 | 0.017 | 0.016 | 0.19 | 0.18 |
| | Beef Roast | 0.087 | 0.14 | 0.014 | 0.018 | 0.10 | 0.16 |
| | Pork Cured | 0.045 | 0.044 | 0.007 | 0.004 | 0.053 | 0.049 |
| | Organ Meat | 0.29 | 0.37 | 0.034 | 0.052 | 0.32 | 0.42 |
| | Poultry | 0.066 | 0.043 | 0.010 | 0.019 | 0.076 | 0.062 |
| Dairy | Whole Milk | 0.038 | 0.031 | 0.033 | 0.096 | 0.072 | 0.041 |
| | 1% Milk | 0.024 | 0.021 | 0.012 | 0.004 | 0.036 | 0.025 |
| | Cream | 0.079 | 0.076 | 0.066 | 0.062 | 0.145 | 0.138 |
| | Cheese Cheddar | 0.24 | 0.20 | .015 | 0.16 | 0.39 | 0.36 |
| | Butter | 0.50 | 0.33 | 0.42 | 0.29 | 0.93 | 0.62 |
| Fish | Fresh Water | 0.26 | 0.16 | 0.36 | 0.32 | 0.62 | 0.48 |
| | Marine | 0.033 | 0.013 | 0.24 | 0.105 | 0.28 | 0.12 |
| Oils | Cooking Fats and Salad Oil | 0.42 | 0.28 | 0.019 | 0.029 | 0.44 | 0.31 |

* N = 44 composites for each city.

Source: Ryan et al. (1997).

Table 3-34. Summary of Coplanar PCBs in a Statistical Sample of Beef Fat in the United States

| Description | PCB 77 | PCB 118 | PCB 105 | PCB 126 | PCB 156 | PCB 157 | PCB 169 |
|---|---|---|---|---|---|---|---|
| Number of Samples | 63 | 63 | 63 | 63 | 63 | 63 | 63 |
| Limits of Detection, ppt | 1.00 | 30.0 | 14.0 | 0.3 | 4.0 | 1.0 | 0.2 |
| Percent Positive | 20 | 100 | 88 | 100 | 100 | 99 | 94 |
| Mean, ppt<br>ND = ½ DL<br>ND = 0.00 | <br>1.00<br>0.60 | <br>440.5<br>440.5 | <br>91.5<br>90.6 | <br>4.0<br>4.0 | <br>58.7<br>58.7 | <br>13.4<br>13.4 | <br>0.69<br>0.69 |
| Range, ppt | ND - 7.97 | 61 - 2295 | ND - 438 | 0.74 - 23.2 | 4.87 - 426 | ND - 91.7 | ND - 2.4 |
| $TEF_P$-$WHO_{94}$ | 0.0005 | 0.0001 | 0.0001 | 0.1 | 0.0005 | 0.0005 | 0.01 |
| $TEQ_P$-$WHO_{94}$ Concentration, ppt (ND = ½ LOD) | $5.0 \times 10^{-4}$ | $4.4 \times 10^{-2}$ | $9.2 \times 10^{-3}$ | $4.0 \times 10^{-1}$ | $2.9 \times 10^{-2}$ | $6.7 \times 10^{-3}$ | $6.9 \times 10^{-3}$ |
| $TEF_P$-$WHO_{94}$ | 0.0001 | 0.0001 | 0.0001 | 0.1 | 0.0005 | 0.0005 | 0.01 |
| $TEQ_P$-$WHO_{94}$ Concentration, ppt (ND = ½ LOD) | $1 \times 10^{-4}$ | $4.4 \times 10^{-2}$ | $9.2 \times 10^{-3}$ | $4.0 \times 10^{-1}$ | $2.9 \times 10^{-2}$ | $6.7 \times 10^{-3}$ | $6.9 \times 10^{-3}$ |

Source: Winters et al. (1996b).

Table 3-35. Concentration Levels of CDD/CDF Congeners in Back Fat and Ratios of Muscle Fat/Back Fat in Cattle

| Congener | Oregon State University | | North Dakota State University | | Pennsylvania State University | | Pennsylvania State University | | Pennsylvania State University | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Back Fat Concentration (ppt lipid) | Ratio Muscle/ Back Fat | Back Fat Concentration (ppt lipid) | Ratio Muscle/ Back Fat | Back Fat Concentration (ppt lipid) | Ratio Muscle/ Back Fat | Back Fat Concentratio n (ppt lipid) | Ratio Muscle/ Back Fat | Back Fat Concentration (ppt lipid) | Ratio Muscle/ Back Fat |
| 2,3,7,8-TCDD | 0.3 | 1.2 | 0.3 | 0.9 | 0.2 | 1.2 | 0.6 | 1.4 | 0.9 | 1.1 |
| 1,2,3,7,8-PeCDD | 2.3 | 0.5 | 1.6 | 0.6 | 5.1 | 0.9 | 9.9 | 1.7 | 31.5 | 0.6 |
| 1,2,3,4,7,8-HxCDD | 2.2 | 0.6 | 2.4 | 0.5 | 6.4 | 0.9 | 7.3 | 2.4 | 18.9 | 0.8 |
| 1,2,3,6,7,8-HxCDD | 7.7 | 0.8 | 9.3 | 0.7 | 24.6 | 1.0 | 36.1 | 1.8 | 60.4 | 0.9 |
| 1,2,3,7,8,9-HxCDD | 2.3 | 0.7 | 2.3 | 0.3 | 7.3 | 0.8 | 11.6 | 2.7 | 25.0 | 0.8 |
| 1,2,3,4,6,7,8-HpCDD | 8.0 | 0.8 | 24.8 | 0.7 | 41.7 | 0.9 | 56.7 | 1.7 | 65.7 | 1.1 |
| OCDD | 6.3 | 1.7 | 33.0 | 0.8 | 12.8 | 1.6 | 33.7 | 2.4 | 19.6 | 1.5 |
| 2,3,7,8-TCDF | 0 | NA | 0 | NA | 0 | NA | 0 | NA | 0 | NA |
| 1,2,3,7,8-PeCDF | 0 | NA | 0 | NA | 0 | NA | 0 | NA | 0 | NA |
| 2,3,4,7,8-PeCDF | 0.6 | NA | 1.2 | 0.7 | 1.4 | 0.6 | 2.9 | 1.0 | 6.3 | 0.6 |
| 1,2,3,4,7,8-HxCDD | 0.8 | NA | 1.6 | 0.7 | 1.7 | 0.7 | 2.7 | 0.5 | 5.2 | 0.3 |
| 1,2,3,6,7,8-HxCDF | 0.7 | NA | 1.3 | 0.3 | 1.7 | 0.5 | 6.5 | 1.7 | 14.0 | 0.7 |
| 1,2,3,7,8,9-HxCDF | 0 | NA | 0 | NA | 0 | NA | 0 | NA | 0 | NA |
| 2,3,4,6,7,8-HxCDF | 0.9 | NA | 1.2 | 0.2 | 1.2 | 0.3 | 4.3 | 1.3 | 16.9 | 0.4 |
| 1,2,3,4,6,7,8-HpCDF | 1.3 | NA | 4.3 | 0.4 | 2.8 | 0.6 | 10.6 | 3.4 | 24.2 | 1.0 |
| 1,2,3,4,7,8,9-HpCDF | 0 | NA | 0 | NA | 0 | NA | 0 | NA | 0.4 | NA |
| OCDF | 0 | NA | 0 | NA | 0 | NA | 0.1 | NA | 0.6 | NA |
| PCB 77 | 0.7 | 12.6 | 2.0 | 2.8 | 1.7 | 7.3 | 16.5 | 16.7 | 19.5 | 4.7 |
| PCB 118 | 859 | 1.7 | 1087 | 1.2 | 1332 | 1.8 | 3551 | 2.9 | 3649 | 2.5 |
| PCB 105 | 145 | 2.8 | 237 | 1.2 | 233 | 2.8 | 612 | 5.7 | 486 | 5.3 |
| PCB 126 | 8.4 | 0.9 | 11.0 | 1.3 | 8.8 | 1.1 | 27.8 | 0.8 | 18.1 | 1.2 |
| PCB 156 | 88.4 | 0.9 | 105 | 1.0 | 102 | 1.5 | 390 | 1.5 | 281 | 1.3 |
| PCB 157 | 20.7 | 1.0 | 26.3 | 1.0 | 23.1 | 1.4 | 83.4 | 1.4 | 69.7 | 1.1 |
| PCB 169 | 1.3 | 1.2 | 4.7 | 1.1 | 1.6 | 1.1 | 4.5 | 0.7 | 2.7 | NA |
| I-$TEQ_{DF}$ | 3.3 | 0.60 | 3.8 | 0.64 | 8.3 | 0.86 | 14.6 | 1.7 | 34.8 | 0.71 |
| $TEQ_{DF}$-$WHO_{98}$ | 4.4 | 0.58 | 4.6 | 0.64 | 10.8 | 0.86 | 19.5 | 1.7 | 50.5 | 0.69 |
| $TEQ_{P}$-$WHO_{94}$ or $TEQ_{P}$-$WHO_{98}$ | 1.0 | 1.0 | 1.3 | 1.2 | 1.1 | 1.3 | 3.5 | 1.1 | 2.4 | 1.5 |

NA = Either (or both) intramuscular and back fat samples had nondetect concentration, such that a ratio could not be derived.

Source: Lorber et al. (1997a)

Table 3-36. $TEQ_{DFP}$-$WHO_{98}$ Summary of Nationally Extrapolated Pork Results on a Lipid Basis Assuming Nondetects (ND) Equal ½ Detection Limit (results are in ppt, or pg/g; ND = 0 results are in parenthesis).

| Description | Dioxins and Furans | | | Coplanar PCBs | | |
|---|---|---|---|---|---|---|
| | Mean | Standard Deviation | Min/Max | Mean | Standard Deviation | Min/Max |
| Overall | 1.48 (0.46) | 1.47 (1.50) | 0.76/22.47 (0/22.76) | 0.06 (0.04) | 0.08 (0.09) | 0.02/1.66 (0/1.66) |
| Market Hogs | 1.44 (0.42) | 1.17 (1.34) | 0.78/9.30 (0/9.58) | 0.06 (0.04) | 0.07 (0.08) | 0.02/0.40 (0/0.11) |
| Sows | 1.85 (0.94) | 1.46 (1.76) | 0.80/5.63 (0/5.41) | 0.06 (0.04) | 0.03 (0.04) | 0.02/0.11 (0/0.11) |
| Boars | 3.60 (3.03) | 6.31 (6.63) | 0.76/22.47 (0/22.76) | 0.27 (0.26) | 0.48 (0.48) | 0.02/1.66 (0/1.66) |

Source: Lorber et al. (1997b)

Table 3-37. $TEQ_{DFP}$-$WHO_{98}$ Summary of Nationally Exptrapolated Results From U.S. Poultry Fat on a Lipid Basis Assuming Nondetects (ND) Equal ½ Detection Limit (Results are in ppt, or pg/g; ND = 0 results are in parenthesis)

| Description | Dioxins and Furans | | Coplanar PCBs | |
|---|---|---|---|---|
| | Mean | Max | Mean | Max |
| Overall | 0.77 (0.48) | 4.72 | 0.29 (0.29) | 1.68 |
| Young Chickens | 0.76 (0.47) | 4.08 | 0.28 (0.28) | 1.68 |
| Light Fowl | 0.47 (0.16) | 0.92 | 0.27 (0.27) | 0.75 |
| Heavy Fowl | 1.14 (0.90) | 2.60 | 0.34 (0.34) | 1.12 |
| Young Turkeys | 1.09 (0.88) | 4.72 | 0.65 (0.65) | 1.28 |

Source: Ferrario, et al. (1997)

Table 3-38. Average Congener Concentrations of 8 Composite Milk Samples
(pg/g lipid; ND = 0 in parenthesis)

| CDDs | Concentration | CDFs | Concentration | PCBs | Concentration |
|---|---|---|---|---|---|
| 2,3,7,8-TCDD | 0.07 (0.07) | 2,3,7,8-TCDF | 0.08 (0.08) | PCB 77 | 10.6 (10.6) |
| 1,2,3,7,8-PCDD | 0.32 (0.32) | 1,2,3,7,8-PCDF | 0.05 (0) | PCB 118 | 685.3 (685.3) |
| 1,2,3,4,7,8-HxCDD | 0.39 (0.39) | 2,3,4,7,8-PCDF | 0.28 (0.28) | PCB 105 | 170.3 (170.3) |
| 1,2,3,6,7,8-HxCDD | 1.87 (1.87) | 1,2,3,4,7,8-HxCDF | 0.39 (0.39) | PCB 126 | 3.6 (3.6) |
| 1,2,3,7,8,9-HxCDD | 0.55 (0.55) | 1,2,3,6,7,8-HxCDF | 0.25 (0.25) | PCB 156 | 60.1 (60.1) |
| 1,2,3,4,6,7,8-HpCDD | 5.03 (5.03) | 1,2,3,7,8,9-HxCDF | 0.05 (0) | PCB 157 | 13.8 (13.8) |
| OCDD | 4.89 (4.89) | 2,3,4,6,7,8-HxCDF | 0.28 (0.28) | PCB 169 | 0.5 (0.5) |
| | | 1,2,3,4,6,7,8-HpCDF | 0.83 (0.83) | $TEQ_P$-$WHO_{98}$ | 0.49 (0.49) |
| | | 1,2,3,4,7,8,9-HpCDF | 0.05 (0) | | |
| | | OCDF | 0.05 (0) | | |
| | | $TEQ_{DF}$-$WHO_{98}$ | 0.98 (0.97) | | |

Source: Lorber et al. (1998b)

Table 3-39. Calculation of Fractional Fat Content of Dairy Products Category

| Milk or Dairy Product[a] | Mean Whole Weight Dietary Dairy Intake (g/day)[b] | Mean Fat Content (%) | Total Dairy Fat Intake (g/day) | Mean Whole Weight Dietary Intake of Dairy Other Than Milk (g/day) | Percent Dietary Intake of Dairy Other Than Milk (%) | Dairy Product Fat (%) |
|---|---|---|---|---|---|---|
| Whole milk | 56 | 3.16 | 1.77 | -- | -- | -- |
| Lowfat milk | 83 | 1.33 | 1.10 | -- | -- | -- |
| Skim milk | 33 | 0.17 | 0.06 | -- | -- | -- |
| Yogurt | 4.5 | 1.47 | 0.07 | 4.5 | 7.76 | 0.11 |
| Cheese | 14 | 25.3 | 3.54 | 14 | 24.14 | 6.11 |
| Milk desserts and other[c] | 39.5 | 8.25(d) | 3.26(d) | 39.5 | 68.10 | 5.62 |
| TOTAL | 230 | | 9.80 | 58 | 100 | 11.84(e) |

a CSFII food categories taken from USDA (1995).

b USDA (1995).

c Milk desserts = 21 g/day. Because total dairy = 230 g/day and total dairy minus milk desserts and other = 190.5; other is assumed to = 18.5 g/day (i.e., 230 minus 190.5 minus 21 = 18.5).

d No fat content data available for milk desserts and other dairy products. Total dietary fat intake for milk desserts and other calculated as the total dietary fat intake (i.e., 9.8 g/day, based on adult total fat intake of 81.39 g/day [CDC, 1994; as cited in U.S. EPA, 1997]), times the fraction of fat that comes from dairy products (i.e., 0.12, based on NLMB, 1993; as cited in U.S. EPA, 1997), minus the dairy fat intake of the other milk and dairy foods [i.e., (81.39 g/day * 0.12) - (6.54 g/day) = 3.26 g/day]. The mean fat content of milk desserts and other was calculated as the dairy fat intake (3.26 g fat/day) divided by the dietary intake (39.5 g/day).

e The overall fractional fat content of dairy products other than milk (11.84%) is calculated as the sum of the products of the fractions of dairy intake other than milk times the fractional fat contents, on the various non-milk dairy products [i.e., (0.0776 * 0.0147) + (0.2414 * 0.253) + (0.6810 * 0.0825) = 0.1184].

Table 3-40. CDD/CDF Levels in German Food

| | Mean I-$TEQ_{DF}$[a] (pg/g fat) | Number of Samples |
|---|---|---|
| Cow's Milk | 1.35 | 168 |
| Cheese | 0.98 | 10 |
| Butter | 0.66 | 5 |
| Beef | 1.69 | 3 |
| Veal | 3.22 | 4 |
| Pork | < 0.4 | 3 |
| Sheep | 1.23 | 2 |
| Chicken | 1.41 | 2 |
| Canned Meat | 1.29 | 2 |
| Lard | 0.47 | 4 |
| Fresh Water Fish | 13.25 | 18 |
| Salt Water Fish | 16.82 | 15 |
| Fish Oil | 2.64 | 4 |
| Cod Liver Oil | 13.31 | 4 |
| Salad Oil | < 0.4 | 4 |
| Margarine | < 0.4 | 6 |
| Infant Formula | 0.5 | 10 |

[a] I-$TEQ_{DF}$ computed using one-half the detection limit for nondetects.

Sources: Milk data based on Fürst et al. (1991); other data from Fürst et al. (1990).

Table 3-41. CDD/CDF Background Levels in Some European, Canadian, and New Zealand Food

| Country | Food | Source | pg I-$TEQ_{DF}$/g fat |
|---|---|---|---|
| Germany | Cow's milk<br>Cow's milk | Background contamination<br>Consumer's milk | 1.0 - 2.8<br>0.8 - 2.6 |
| United Kingdom | Cow's milk | Rural area | 1.3 |
| Netherlands | Cow's milk | Background contamination | 0.7 - 2.5 |
| New Zealand | Cow's milk | Background contamination | 0.18 - 0.22 |
| Germany | Pork<br>Beef<br>Veal<br>Sheep<br>Poultry<br>Canned Meat<br>Lard | | 0.5<br>3.5<br>7.4<br>2.0<br>2.3<br>1.7<br>0.8 |
| Canada | Beef<br>Pork<br>Poultry | | 2.9<br>0.2<br>2.6 |

Source: Fürst et al. (1991)

Table 3-42. CDD/CDF Levels in German Food (1993-1996)

| Sample | Mean I-$TEQ_{DF}$ (pg/g fat) | Number of Samples |
|---|---|---|
| Cow's Milk | 0.71 | 538 |
| Cheese | 0.66 | 99 |
| Butter | 0.64 | 222 |
| Beef | 0.71 | 14 |
| Veal | 0.95 | 11 |
| Lamb, Sheep, Mutton | 0.52 | 13 |
| Poultry | 0.62 | 19 |
| Eggs | 2.10 | 218 |
| Ham | 0.39 | 8 |
| Venison | 1.41 | 6 |
| Sausage | 0.28 | 1 |
| Kidney (sheep) | 1.11 | 1 |
| Horsemeat | 3.76 | 1 |
| Salt-water Fish | 14.7 | 42 |
| Trout | 7.44 | 61 |
| Rhine River Fish | 39 | 19 |
| Artificial Lake Fish | 104.1 | 14 |
| Vegetables growing underground (e.g., potatoes, carrots) | 16.9 | 13 |
| Vegetables growing on the ground (e.g., zucchini, Beetroot, kohlrabi, celery, and onions) | 12.2 | 18 |
| Leak vegetables (e.g., lettuce, sugarloaf, endive, savoy cabbage, leek, white cabbage) | 12.9 | 22 |
| Fruit growing above ground (e.g., apples and tomatoes) | 4.3 | 4 |

Source: Malisch (1998).

Table 3-43. I-$TEQ_{DF}$ Levels in Dairy Products in France

| Type of Sample | Mean I-$TEQ_{DF}$ Total pg/g Fat | Number of Samples |
|---|---|---|
| Cow's Milk | 1.91 | 12 |
| Cheese | 1.11 | 20 |
| Butter | 1.01 | 8 |
| Milk Dessert and Cream | 1.34 | 12 |

Source: Defour et al. (1997).

Table 3-44. I-$TEQ_{DF}$ Concentrations in Food from the Netherlands

| Food Category | CDD/CDFs (pg I-$TEQ_{DF}$/g fat) |
|---|---|
| Beef | 1.75 |
| Cow's Liver | 5.7 |
| Pork | 0.43 |
| Pig's Liver | 15.3 |
| Poultry | 1.65 |
| Chicken's Liver | 3.25 |
| Mutton | 1.85 |
| Horse Meat | 13.85 |
| Game [a] | 16.8 |
| Butter | 1.8 |
| Cheese [a] | 1.4 |
| Nuts [a] | 0.2 |
| Cereals [a] | 0.34 |
| Eggs | 2.0 |
| Fatty Sea Fish [a] | 6.65 |
| Lean Fish [a] | 48.65 |
| Eel | 28.0 |
| Fresh Water Fish [a] | 2.4 |
| Mixed Meat Product [a] | 0.67 |
| Dairy Products | 1.58 |
| Soy Bean Oil | 0.025 |
| Rape-Seed Oil | 0.006 |
| Palm Oil | 0.030 |
| Sunflower Oil | 0.006 |
| Coconut Fat | 0.024 |
| Palm Fat | 0.010 |
| Fish Oil | 2.2 |
| Items with Vegetable Oil [a] | 0.02 |
| Items from Food Industry [a] | 0.41 |

[a] A proportional mixture of food items was analyzed.

Source: Theelen et al. (1993)

Table 3-45. I-$TEQ_{DF}$ Concentrations in Food from Spain

| Food Category | CDD/CDFs (pg I-$TEQ_{DF}$/g fat) | CDD/CDFs (pg I-$TEQ_{DF}$/g wet weight) |
|---|---|---|
| Vegetables | -- | 0.14 |
| Pulses | -- | 0.19 |
| Cerales | -- | 0.25 |
| Fruits | -- | 0.09 |
| White Fish | 5.39 | 0.27 |
| Seafood | 10.59 | 0.42 |
| Tinned Fish | 2.57 | 0.24 |
| Blue Fish | 7.90 | 0.76 |
| Pork and Pork Products | 0.90 | 0.11 |
| Chicken and Chicken Products | 1.15 | 0.11 |
| Beef and Beef Products | 1.76 | 0.13 |
| Lamb | 1.76 | 0.13 |
| Eggs | 1.22 | 0.12 |
| Dairy Products | 1.25 | 0.04 |
| Whole Milk | 1.02 | 0.18 |
| Semi-Skimmed Milk | 1.20 | 0.06 |
| Oil | 0.64 | -- |
| Margarine | 0.49 | -- |

Source: Domingo et al. (1999).

Table 3-46. I-TEQ$_{DF}$ Concentrations in UK Foods in 1988

| Food Product | Mean I-TEQ$_{DF}$ (ppt) | | | | | |
|---|---|---|---|---|---|---|
| | Norwich | | Port Talbot | | Stonehaven | |
| | nd = 0 | nd = LOD | nd = 0 | nd = LOD | nd = 0 | nd = LOD |
| Fish[a,b] | 0.69 | 0.73 | 0.63 | 0.63 | 0.07 | 0.07 |
| Carcass Meat[a] | -- | -- | 1.1 | 1.1 | 0.18 | 0.26 |
| Offals (internal organs)[a] | -- | -- | 0.20 | 0.22 | 0.62 | 0.69 |
| Poultry[a] | -- | -- | 0.37 | 0.37 | 0.28 | 0.29 |
| Meat Products[a] | -- | -- | 0.20 | 0.20 | 0.17 | 0.21 |
| Milk Products[a] | -- | -- | 0.08 | 0.33 | 0.09 | 0.09 |
| Butter[b] | 1.2 | 1.2 | -- | -- | -- | -- |
| Cheddar Cheese[c] | 0.16 | 0.16 | -- | -- | -- | -- |
| Reduced Fat Cheese[c] | 0.09 | 0.12 | -- | -- | -- | -- |
| Fats and Oils[a] | -- | -- | 0.84 | 0.88 | 0.11 | 0.41 |
| Eggs[a] | -- | -- | 0.22 | 0.22 | 0.16 | 0.16 |
| Green Vegetables[a] | -- | -- | 0.01 | 0.02 | < 0.01 | 0.01 |
| Other Vegetables[a] | -- | -- | 0.05 | 0.12 | 0.06 | 0.06 |
| Potatoes[a] | -- | -- | 0.04 | 0.04 | 0.02 | 0.03 |
| Fresh Fruit[a] | -- | -- | < 0.01 | 0.04 | 0.04 | 0.06 |

[a] One sample analyzed from Port Talbot, and one sample analyzed from Stonehaven.
[b] Eight samples analyzed.
[c] Two samples analyzed.

Source: MAFF (1992).

Table 3-47. I-$TEQ_{DF}$ Concentrations in Bottled Cow's Milk from the United Kingdom

| | Mean (ppt) nd = 0 | Range (ppt) nd = 0 | Mean (ppt) nd = LOD | Range (ppt) nd = LOD |
|---|---|---|---|---|
| Winter 1990 (n = 8) | 0.08 | 0.05-0.13 | 0.09 | 0.05-0.13 |
| Summer 1990 (n = 7) | 0.06 | 0.04-0.07 | 0.06 | 0.05-0.07 |
| Rural 1989 (n = 9) | 0.04 | 0.03-0.06 | 0.05 | 0.04-0.06 |
| Urban 1989 (n = 9) | 0.19 | 0.12-0.27 | 0.20 | 0.12-0.27 |

Source: MAFF (1992).

Table 3-48. Concentrations and Concentration Ranges (pg/g fresh weight) of Four Dioxin-Like PCBs in Foods from Finland

| | | PCB 77 | PCB 105 | PCB 126 | PCB 169 |
|---|---|---|---|---|---|
| Baltic Herring (n = 6) | Mean<br>Range | 97<br>33 - 136 | 1700<br>960 - 2700 | 17<br>7.4 - 26 | 4.5<br>nd - 12 |
| Rainbow Trout (n = 4) | Mean<br>Range | 100<br>8.0 - 150 | 1200<br>410 - 2100 | 17<br>5.2 - 35 | 3.9<br>nd - 7.4 |
| Other Fish (n = 4) | Mean<br>Range | 53<br>5.6 - 153 | 400<br>113 - 1100 | 11<br>2.3 - 28 | 1.9<br>0.6 - 6.5 |
| Beef (n = 6) | Mean<br>Range | 13<br>0.6 - 38 | 22<br>5.3 - 38 | 3.2<br>0.3 - 7.3 | 0.5<br>nd - 1.0 |
| Pork (n = 3) | Mean<br>Range | 13<br>1.0 - 24 | 24<br>11 - 47 | 1.5<br>0.5 - 3.7 | 0.8<br>nd - 2.2 |
| Poultry (n = 2) | Mean | 8.2 | 68 | 1.2 | nd |
| Inner Organs (n = 5) | Mean<br>Range | 3.2<br>nd - 7.9 | 45<br>8.1 - 111 | 2.6<br>0.4 - 6.0 | 0.3<br>nd - 0.6 |
| Eggs (n = 2) | Mean | 4.1 | 98 | 2.9 | 0.1 |
| Fish Liver Oil (n = 2) | Mean | 2700 | 30000 | 620 | 130 |

Note: nd = not detected.

Source: Himberg (1993).

Table 3-49. I-$TEQ_{DF}$s (ppt wet weight) in Foods From the Former Soviet Union

| Site | Food Product | Lipid Content (%) | I-$TEQ_{DF}$ Concentration ppt wet weight [1] |
|---|---|---|---|
| Moscow | Lamb | 23.8 | 0.30 |
| | Swiss Cheese | 3.0 | 0.04 |
| | Sausage | 57.0 | 0.60 |
| | Hot Sausage | 9.4 | 0.15 |
| | Turkey Fat & Meat | 29.9 | 0.30 |
| | Hamburger (cooked) | 6.7 | 0.02 |
| | Beef | 24.0 | 0.13 |
| | Ground Beef (cooked) | 8.5 | 0.24 |
| | Fish (Motba) | 2.0 | 0.70 |
| Irkutsk | Pork Fat | 54.8 | 0.15 |
| | Pork Fat | 50.8 | 0.20 |
| | Pork Meat | 9.8 | 0.05 |
| | Lamb Fat | 43.5 | 0.40 |
| | Soft Cheese | 4.9 | 0.04 |
| | Duck Livers | 3.0 | 0.08 |
| | Cow Cream | 15.0 | 0.80 |
| | Uncooked Beef | 3.0 | 0.13 |
| Novosibirsk | Pork Fat | 34.1 | 0.06 |
| | Cheese | 27.6 | 0.17 |
| | Vanilla Ice Cream | 1.0 | 0.005 |
| | Fish | 9.2 | 0.07 |
| | Smoked Fish | 16.3 | 0.80 |
| | Meatball | 4.7 | 0.02 |
| | Chicken | 24.6 | 0.05 |
| | Mintai Fish (cooked) | 5.8 | 0.14 |
| | Cheese with Butter | 36.0 | 0.40 |
| | Beef Fat | 34.0 | 0.30 |
| Baikalsk | Fat Head Minnow | 2.7 | 1.3 |
| | Pork | 72.0 | 0.9 |
| | Butter | 53.0 | 1.4 |

[1] One-half the detection limit was used in calculating the I-$TEQ_{DF}$s.

Source: Schecter et al. (1992b).

Table 3-50. Percentage Reduction of Total 2,3,7,8-TCDD Residues in Restructured Carp Fillets from Cooking

| Cooking Method | Fillet Diameter | End Temperature °C | TCDD Concentration Reduction | |
|---|---|---|---|---|
| | | | control[a] | spiked[b] |
| Roasted covered | 7.5 | 60 | 41.4 ± 4.0 | 44.2 ± 3.9 |
| | | 70 | 50.5 ± 7.9 | 47.7 ± 2.1 |
| | | 80 | 63.4 ± 4.3 | 55.0 ± 3.9 |
| Roasted uncovered | 7.5 | 60 | 34.2 ± 8.7 | 47.0 ± 4.4 |
| | | 70 | 49.2 ± 6.3 | 51.3 ± 2.4 |
| | | 80 | 56.6 ± 5.9 | 57.5 ± 2.0 |
| Roasted uncovered | 10.0 | 80 | 65.9 ± 2.6 | 59.2 ± 2.4 |
| Charbroiled | 7.5 | 60 | 55.3 ± 6.5 | 59.3 ± 5.2 |
| | | 80 | 62.0 ± 4.3 | 63.6 ± 3.7 |
| Charbroiled | 10.0 | 80 | 67.5 ± 7.8 | 70.6 ± 7.7 |

[a] N = 4
[b] Samples were spiked to approximately 100 ppt 2,3,7,8-TCDD.

Source: Stachiw et al. (1988)

Table 3-51. Effects of Cooking and Trimming on PCB Levels in Lake Ontario Fish

| Method | Species | Reduction in Concentration (%) | Reduction in Total Amount (%) |
|---|---|---|---|
| Trimming off fat and skin | Smallmouth Bass | 64.3 | 80.0 |
| Trimming off fat and skin | Brown Trout | 43.2 | 77.8 |
| Deep frying trimmed fillets | Smallmouth Bass | 45.9 | 74.0 |
| Smoking untrimmed fillets | Brown Trout | 12.0 | 26.7 |
| Baking untrimmed fillets | Smallmouth Bass | 0 | 16.4 |
| Broiling trimmed fillets | Brown Trout | 0 | 0 |

Source: Skea et al. (1979)

Table 3-52. Means and Standard Deviations of PCB Cooking Losses (%) of Stewed and Pressure Cooked Chicken Pieces

| Chicken Piece | PCB Cooking Losses (%) | |
|---|---|---|
| | Stewed | Pressure Cooked |
| Breast | 31.73 ± 2.36 | 30.24 ± 1.28 |
| Drumstick | 31.21 ± 2.38 | 32.86 ± 1.14 |
| Thigh Meat | 36.82 ± 1.50 | 38.10 ± 0.63 |
| Thigh Skin | 39.31 ± 8.18 | 46.59 ± 4.69 |
| Abdominal Adipose Tissue | 85.94 ± 1.98 | 88.88 ± 2.17 |

Source: Zabik (1974)

Table 3-53. Weighted Mean CDD/CDF Profiles for Foods

| 2,3,7,8-Substituted CDD/CDFs | Beef[a] | | Pork[b] | | Poultry[c] | | Eggs[d] | | Milk[e] | | Dairy[f] | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Whole Weight Concentra-tion (ppt) | Fraction of Total 2,3,7,8-Substituted CDD/CDFs | Whole Weight Concentra-tion (ppt) | Fraction of Total 2,3,7,8-Substituted CDD/CDFs | Whole Weight Concentra-tion (ppt) | Fraction of Total 2,3,7,8-Substituted CDD/CDFs | Whole Weight Concentra-tion (ppt) | Fraction of Total 2,3,7,8-Substituted CDD/CDFs | Whole Weight Concentra-tion (ppt) | Fraction of Total 2,3,7,8-Substituted CDD/CDFs | Whole Weight Concentra-tion (ppt) | Fraction of Total 2,3,7,8-Substituted CDD/CDFs |
| 2,3,7,8-TCDD | 0.006 | 0.0026 | 0.0015 | 0.0001 | 0.023 | 0.018 | 0 | 0 | 0.002 | 0.0046 | 0.029 | 0.0035 |
| 1,2,3,7,8-PeCDD | 0.008 | 0.0037 | 0.0015 | 0.0001 | 0.018 | 0.014 | 0.015 | 0.0080 | 0.010 | 0.021 | 0.169 | 0.0202 |
| 1,2,3,4,7,8-HxCDD | 0.034 | 0.0169 | 0.015 | 0.0014 | 0.007 | 0.0057 | 0.007 | 0.0038 | 0.012 | 0.026 | 0.165 | 0.0198 |
| 1,2,3,6,7,8-HxCDD | 0.229 | 0.1134 | 0.12 | 0.0114 | 0.051 | 0.040 | 0.029 | 0.0157 | 0.060 | 0.123 | 0.877 | 0.1050 |
| 1,2,3,7,8,9-HxCDD | 0.049 | 0.0244 | 0.006 | 0.0006 | 0.041 | 0.033 | 0.006 | 0.0035 | 0.018 | 0.036 | 0.205 | 0.0246 |
| 1,2,3,4,6,7,8-HpCDD | 0.834 | 0.4114 | 1.49 | 0.1416 | 0.22 | 0.173 | 0.153 | 0.0829 | 0.160 | 0.330 | 2.565 | 0.3072 |
| OCDD | 0.619 | 0.3055 | 7.86 | 0.7470 | 0.76 | 0.595 | 1.557 | 0.8415 | 0.156 | 0.321 | 2.561 | 0.3067 |
| 2,3,7,8-TCDF | 0 | 0 | 0.0006 | 0.0001 | 0.043 | 0.034 | 0.011 | 0.0059 | 0.003 | 0.0053 | 0.055 | 0.0066 |
| 1,2,3,7,8-PeCDF | 0 | 0 | 0 | 0 | 0.013 | 0.010 | 0 | 0 | 0 | 0 | 0.010 | 0.0012 |
| 2,3,4,7,8-PeCDF | 0.0114 | 0.0056 | 0.021 | 0.0020 | 0.020 | 0.015 | 0 | 0 | 0.009 | 0.018 | 0.142 | 0.0170 |
| 1,2,3,4,7,8-HxCDF | 0.0513 | 0.0253 | 0.09 | 0.0086 | 0.015 | 0.012 | 0.005 | 0.0029 | 0.012 | 0.026 | 0.351 | 0.0421 |
| 1,2,3,6,7,8-HxCDF | 0.0228 | 0.0112 | 0.087 | 0.0083 | 0.010 | 0.0079 | 0.003 | 0.0017 | 0.008 | 0.016 | 0.174 | 0.0208 |
| 1,2,3,7,8,9-HxCDF | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2,3,4,6,7,8-HxCDF | 0.019 | 0.0094 | 0.024 | 0.0023 | 0.011 | 0.0090 | 0.002 | 0.0013 | 0.009 | 0.018 | 0.111 | 0.0133 |
| 1,2,3,4,6,7,8-HpCDF | 0.1425 | 0.0703 | 0.503 | 0.0478 | 0.028 | 0.022 | 0.027 | 0.0148 | 0.026 | 0.055 | 0.605 | 0.0725 |
| 1,2,3,4,7,8,9-HpCDF | 0 | 0 | 0.0255 | 0.0024 | 0.006 | 0.0045 | 0 | 0 | 0 | 0 | 0.023 | 0.0028 |
| OCDF | 0 | 0 | 0.2775 | 0.0264 | 0.010 | 0.0078 | 0.034 | 0.0182 | 0 | 0 | 0.307 | 0.0368 |
| TOTAL | 2.027 | 1.0 | 10.52 | 1.0 | 1.271 | 1.0 | 1.85 | 1.0 | 0.486 | 1.0 | 8.35 | 1.0 |

NOTE: Non-detects are assumed to be zero.

[a] Based on data from Winters et al. (1996a).
[b] Based on data from Lorber et al. (1997b).
[c] Based on data from Ferrario et al. (1997).
[d] Based on data from Fiedler et al. (1997c).
[e] Based on data from Lorber et al. (1998b).
[f] Based on data from Lorber et al. (1998b).

Table 3-54. Summary of CDD/CDF Levels in U.S. Food (pg/g fresh weight)

| | Mean $TEQ_{DF}$-$WHO_{98}$ Assuming ND = 0.5 DL | Mean $TEQ_{DF}$-$WHO_{98}$ Assuming ND = zero | Number of Samples | Reference |
|---|---|---|---|---|
| Beef | 0.20 | 0.07 | 63 | Winters et al. (1996a) |
| Pork | 0.22 | 0.06 | 78 | Lorber et al. (1997b) |
| Poultry | 0.12 | 0.072 | 78 | Ferrario et al. (1997) |
| Eggs | 0.032 | 0.023 | 3 | Fiedler et al. (1997c); Cooper et al. (1995) |
| Dairy Products | 0.12 | 0.12 | 8 composites | Based on data from Lorber et al. (1998b) |
| Milk | 0.031 | 0.031 | 8 composites | Lorber et al. (1998b) |
| Freshwater Fish | 1.2[a] | 0.59[a] | 60 | U.S. EPA (1992) |
| Marine Fish | 0.36 | 0.36 | 3 | Fiedler et al. (1997a) |
| Marine Shellfish | 0.79 | 0.79 | 9 | Fiedler et al. (1997a) |
| Vegetable Fat | NA[a] | 0.056 | 30 | Versar (1996b) |

[a] Value represents I-$TEQ_{DF}$ and not $TEQ_{DF}$-$WHO_{98}$; the congener-specific data required to calculate the $TEQ_{DF}$-$WHO_{98}$ were not available.

[b] High detection limits led to a calculation of the mean with ND = 0.5 DL that was judged to be misleading. See text for more detail.

ND = Nondetect; DL - Detection Limit
NA = Not available

Table 3-55. Summary of $TEQ_P$-$WHO_{98}$ Levels in North American Food (pg/g fresh weight)

| | Mean $TEQ_P$-$WHO_{98}$ Assuming ND = 0.5DL | Mean $TEQ_P$-$WHO_{98}$ Assuming ND = Zero | Number of Samples | Reference |
|---|---|---|---|---|
| Beef | 0.094 | 0.094 | 63 | Winters et al. (1996b) |
| Pork | 0.009 | 0.006 | 78 | Lorber et al. (1997b) |
| Poultry | 0.044 | 0.044 | 78 | Ferrario et al. (1997) |
| Eggs | 0.1 | NR | 18<br>1 composite<br>5 composites | Schecter et al. (1997)<br>Mes and Weber (1989)<br>Mes et al. (1991) |
| Dairy Products | 0.058 | 0.058 | 8 composites | Based on data from Lorber et al. (1998b) |
| Milk | 0.016 | 0.016 | 8 composites | Lorber et al. (1998b) |
| Freshwater Fish | 1.2 | NR | 10<br>1 composite<br>5 composites | Schecter et al. (1997)<br>Mes and Weber (1989)<br>Mes et al. (1991) |
| Marine Fish | 0.25 | NR | 13<br>5 composites | Schecter et al. (1997)<br>Mes et al. (1991) |
| Marine Shellfish | 0.04 | NR | 5 composites | Mes et al. (1991) |
| Vegetable Fat | 0.037 | NR | 5 composites | Mes et al. (1991) |

NOTE: Schecter et al. (1997) and Mes and Weber (1989) values based on one-half the limit of detection for nondetects and Mes et al. (1991) data based on positive composite samples only.

Table 3-56. Summary of North American CDD/CDF and PCB TEQ-$WHO_{98}$ Levels in Environmental Media and Food (whole weight basis)

| Media | CDD/CDFs[a] | References | PCBs[a] | References | Mean Total CDD/CDF/PCBs |
|---|---|---|---|---|---|
| Urban Soil, ppt | n = 157<br>11.9 ± 11.5[b]<br>Range = 2 - 21 | Birmingham (1990), Nestrick et al. (1986), NIH (1995), Pearson et al. (1990), U.S. EPA (1985, 1996) | --[c] | -- | 11.9 |
| Rural Soil, ppt | n = 222<br>3.6[b]<br>Range = 1 - 6 | BC Environment (1995), Birmingham (1990), MRI (1992), Pearson et al. (1990), Reed et al. (1990), Tewhey Assoc (1997), U.S. EPA (1985, 1996) | --[c] | -- | 3.6 |
| Sediment, ppt | n = 11<br>5.3 ± 5.8[b]<br>Range = < 1 - 20 | Cleverly et al. (1996) | n = 11<br>0.53 ± 0.69[b] | Cleverly et al. (1996) | 5.8 |
| Urban Air, pg/$m^3$ | n = 106<br>0.12 ± 0.094[b]<br>Range = 0.03 - 0.2 | CDEP (1988, 1995), Hunt and Maisel (1990), Hunt et al. (1990), Maisel and Hunt (1990), OEPA (1995), Smith et al. (1989, 1990) | 0.0009[h] | Hoff et al. (1992) | 0.12 |
| Rural Air, pg/$m^3$ | n = 7<br>0.017[b]<br>Range = 0.01 - 0.02 | CDEP (1995), OEPA (1995) | --[c] | -- | 0.017 |
| Freshwater Fish, ppt | n = 60<br>1.2 ± 1.2[d]<br>(I-$TEQ_{DF}$) | U.S. EPA (1992) | n = 1 plus 6 composites<br>1.2[d,e] | Schecter et al. (1997), Mes and Weber (1989), Mes et al. (1991) | 2.4 |
| Marine Fish, ppt | n = 3<br>0.36 ± 0.42[d]<br>Range = 0.034 - 0.10 | Fiedler et al. (1997a) | n = 13 plus 5 composites<br>0.25[d,f] | Schecter et al. (1997), Mes et al. (1991) | 0.61 |
| Marine Shellfish, ppt | n = 9<br>0.79 ± 0.62[d]<br>Range = 0.08 - 1.9 | Fiedler et al. (1997a) | n = 5 composites<br>0.043[d,g] | Mes et al. (1991) | 0.83 |
| Water, ppq | n = 236<br>0.00056 ± 0.00079 | Jobb et al. (1990), Meyer et al. (1989) | --[c] | -- | 0.00056 |
| Milk, ppt | n = 8 composites<br>0.031 ± 0.0022 | Lorber et al. (1998b) | n = 8 composites<br>0.016 | Lorber et al. (1998b) | 0.047 |
| Dairy, ppt | n = 8 composites<br>0.12 ± 0.22 | Based on data from Lorber et al. (1998b) | n = 8 composites<br>0.058 | Based on data from Lorber et al. (1998b) | 0.18 |
| Eggs, ppt | n = 3<br>0.032 ± 0.0086[d]<br>Range = 0.025 - 0.044 | Cooper et al. (1995), Fiedler et al. (1997c) | n = 18 plus 6 composites<br>0.10[d,e] | Schecter et al. (1997), Mes and Weber (1989), Mes et al. (1991) | 0.13 |
| Beef ppt | n = 63<br>0.20 ± 0.12<br>Range = 0.2 - 1.1 | Winters et al. (1996a) | n = 63<br>0.094 | Winters et al. (1996b) | 0.29 |

Table 3-56. Summary of North American CDD/CDF and PCB TEQ-$WHO_{98}$ Levels in Environmental Media and Food (whole weight basis)

| Media | CDD/CDFs[a] | References | PCBs[a] | References | Mean Total CDD/CDF/PCBs |
|---|---|---|---|---|---|
| Pork, ppt | n = 78<br>0.22 ± 0.22<br>Range = 0.12 - 1.4 | Lorber et al. (1997b) | n = 78<br>0.0093 | Lorber et al. (1997b) | 0.23 |
| Poultry, ppt | n = 78<br>0.12 ± 0.12<br>Range = 0.05 - 0.72 | Ferrario et al. (1997) | n = 78<br>0.044 | Ferrario et al. (1997) | 0.16 |
| Vegetable Fats, ppt | n = 30<br>0.056 ± 0.24[i] | Versar (1996b) | n = 5 composites<br>0.037[g] | Mes et al. (1991) | 0.093 |

[a] Values are the arithmetic mean TEQs, in ppt, and standard deviations. Nondetects were set to one-half the limit of detection, except for soil and CDD/CDFs in vegetable fats for which nondetects were set to zero.
[b] The values for environmental media are means of the data, but lack the spatial representativeness to be considered true national means.
[c] Congener-specific PCB data are limited.
[d] The values for fish and eggs lack the statistical significance to be considered true means; the values for the other food groups were derived from statistically-based surveys and can be considered true national means.
[e] Standard deviations could not be calculated due to limitations associated with the data (i.e., 3 studies with composite analyses).
[f] Standard deviations could not be calculated due to limitations associated with the data (i.e., 2 studies using composites).
[g] Standard deviations could not be calculated due to limitations associated with the data (i.e., 1 study with composite analyses).
[h] Based on data from Canadian air, as reported by Hoff et al. (1992).
[i] TEQ calculated from Versar (1996b) by setting nondetects to zero.

Table 3-57. CDD/CDF Congeners that Contribute the Highest Percentage of $TEQ_{DF}$-$WHO_{98}$ to the Total $TEQ_{DF}$-$WHO_{98}$ for All Congeners Combined

| Media | North America | Percentage of Total $TEQ_{DF}$-$WHO_{98}$ |
|---|---|---|
| Urban Soil | 1,2,3,7.8-PeCDD | 25.6 |
| Sediment[a] | 1,2,3,4,6,7,8-HpCDD | 18.9 |
| Freshwater Fish | 2,3,7,8-TCDD | 29.9[d] |
| Marine Fish[a] | 1,2,3,7,8-PeCDD | 45.2 |
| Marine Shellfish[a] | 1,2,3,7,8-PeCDD | 42.3 |
| Urban Air | 1,2,3,7,8-PeCDD | 22.5 |
| Water[b] | OCDD | 56.3 |
| Milk[a] | 1,2,3,7,8-PeCDD | 32.7 |
| Dairy[a] | 1,2,3,7,8-PeCDD | 32.7 |
| Eggs[a] | 1,2,3,7,8-PeCDD | 46.7 |
| Beef[a] | 1,2,3,7,8-PeCDD | 32.8 |
| Pork[a] | 1,2,3,7,8-PeCDD | 30.5 |
| Poultry[a] | 1,2,3,7,8-PeCDD | 31.5 |
| Vegetable Fats[a,c] | 1,2,3,7,8-PeCDD | 44.0 |

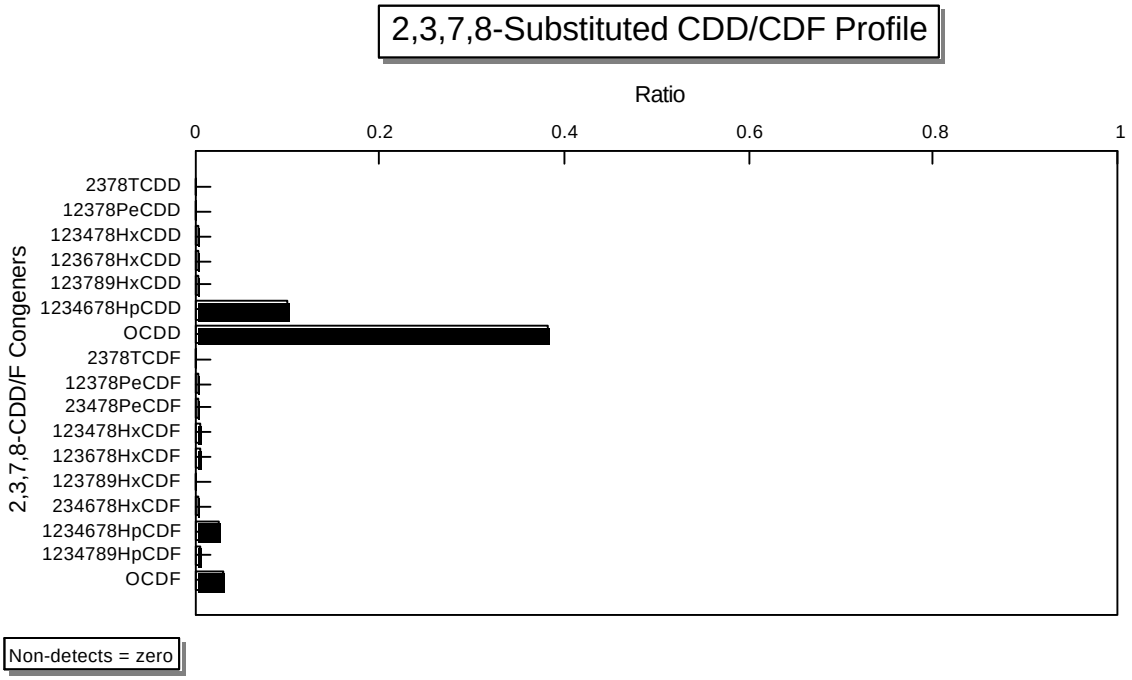
NOTE: Data were not available for all congeners in all media.

[a] Not used in calculation of the standard deviation because adequate congener-specific data were available for all samples allowing for the calculation of means and standard deviations by traditional methods.

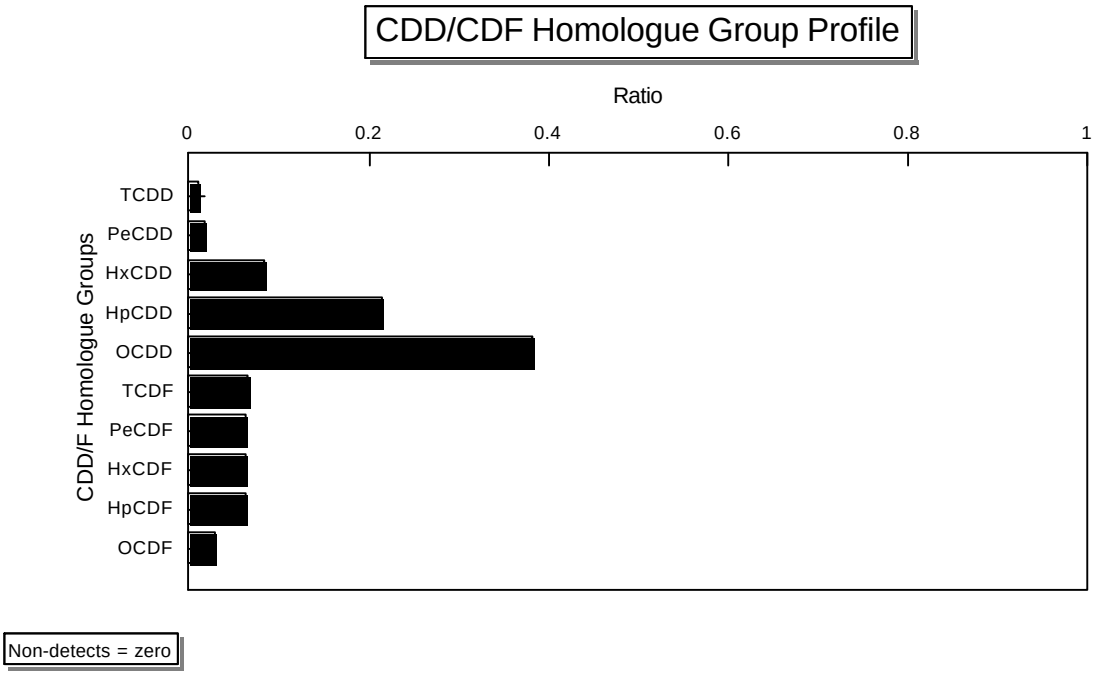
[b] Data available for OCDD and OCDF only.

[c] Due to the large number of nondetects, zero was used to represent nondetects in determining the congener that contributed the most to the total $TEQ_{DF}$-$WHO_{98}$.

[d] Based on I-$TEQ_{DF}$.

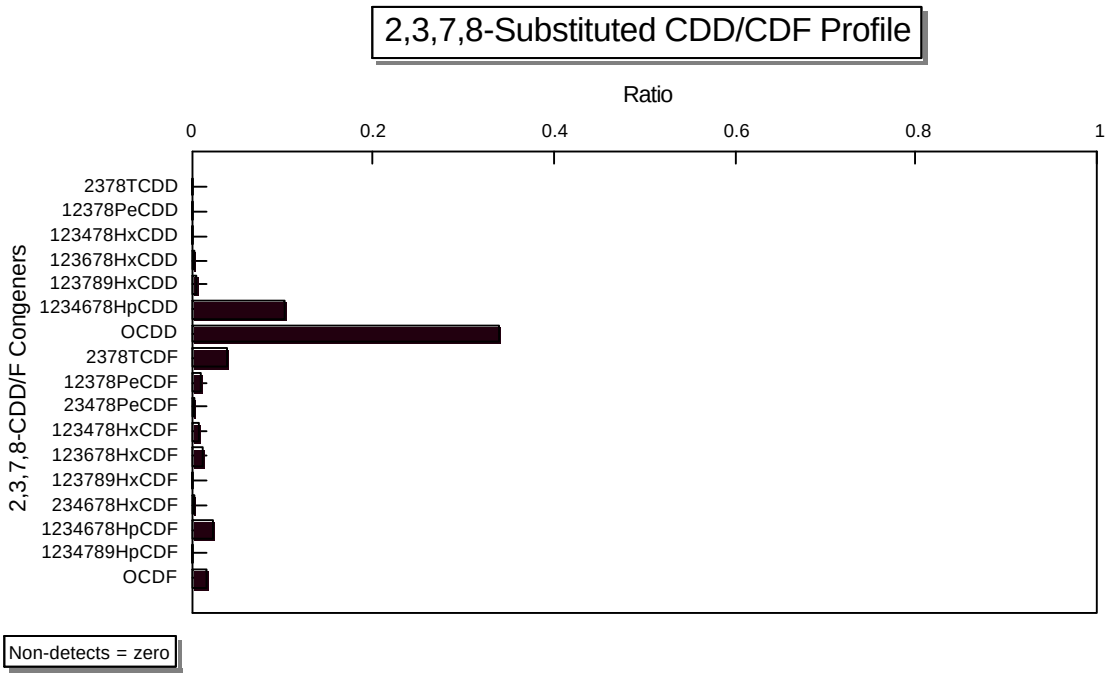



Note: Based on data from OEPA (1995) and CDEP (1998).

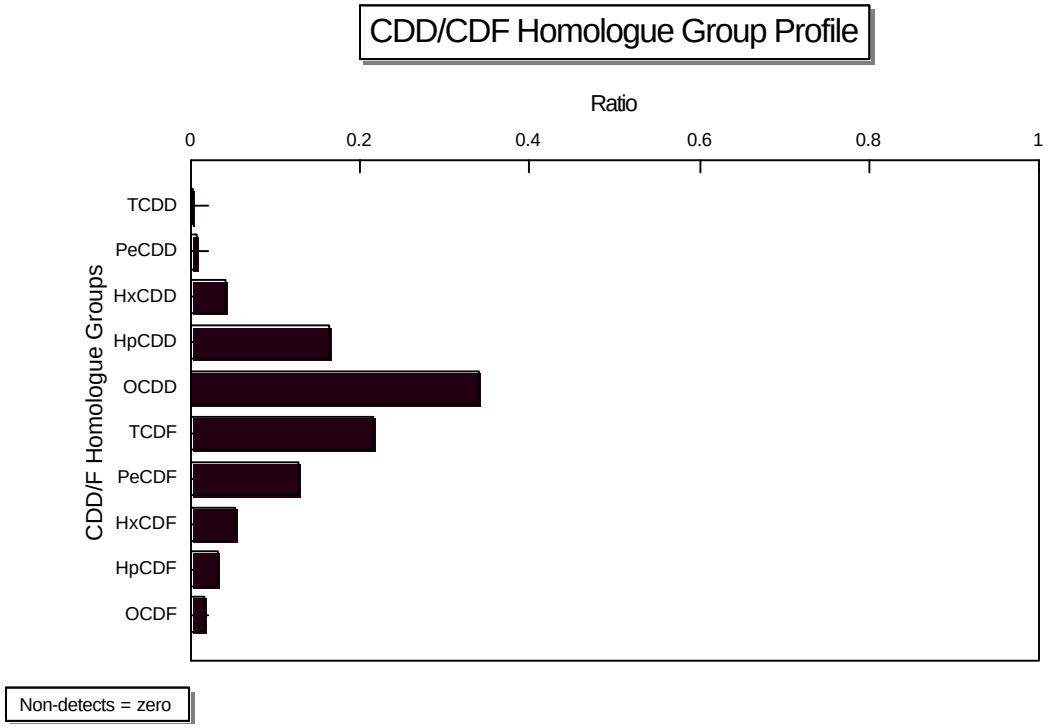



Note: Based on data from OEPA (1995) and CDEP (1998).

Figure 3-1. CDD/CDF Profiles for Rural Background Air

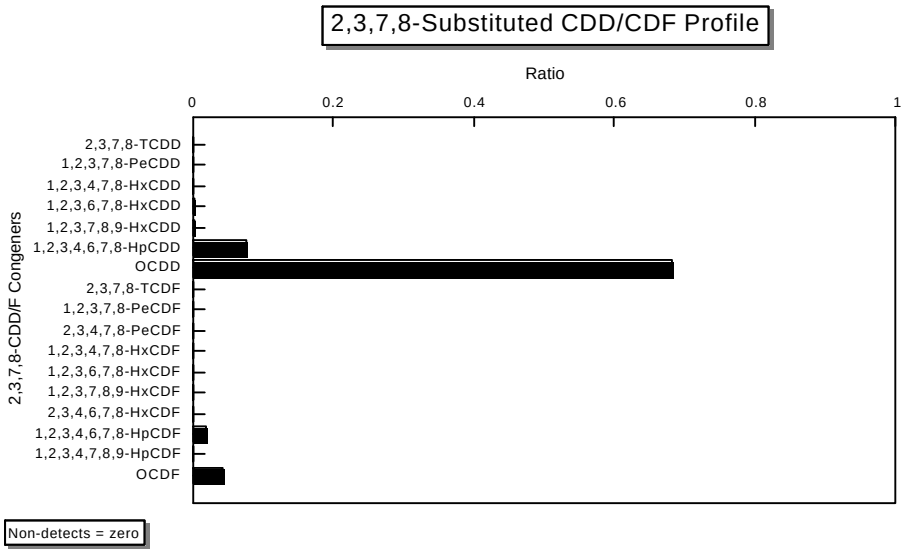



Note:Based on data from CDEP (1988, 1995), Smith et al. (1989), Maisel and Hunt (1990), Hunt et al. (1990, and OEPA (1995).

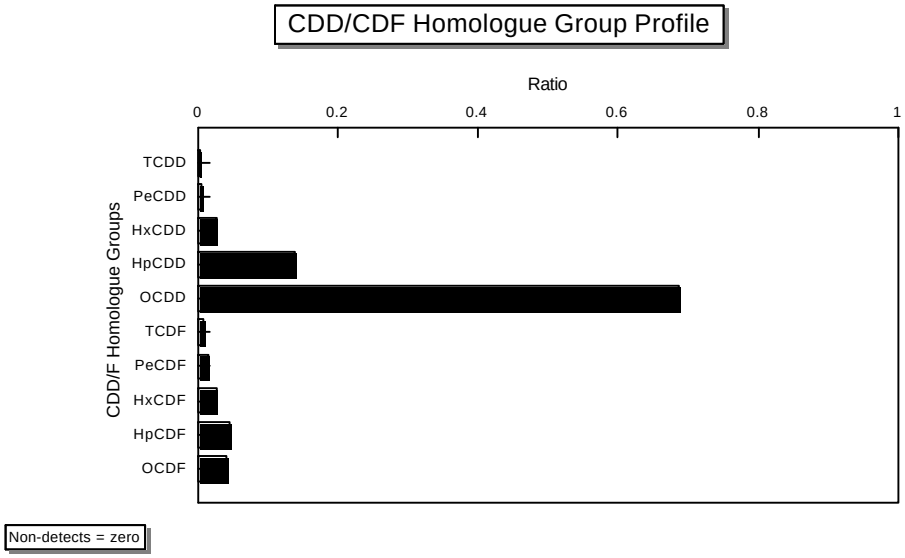



Note:Based on data from CDEP (1988, 1995), Smith et al. (1989), Maisel and Hunt (1990), Hunt et al. (1990, and OEPA (1995).

Figure 3-2. CDD/CDF Profiles for Urban Background Air

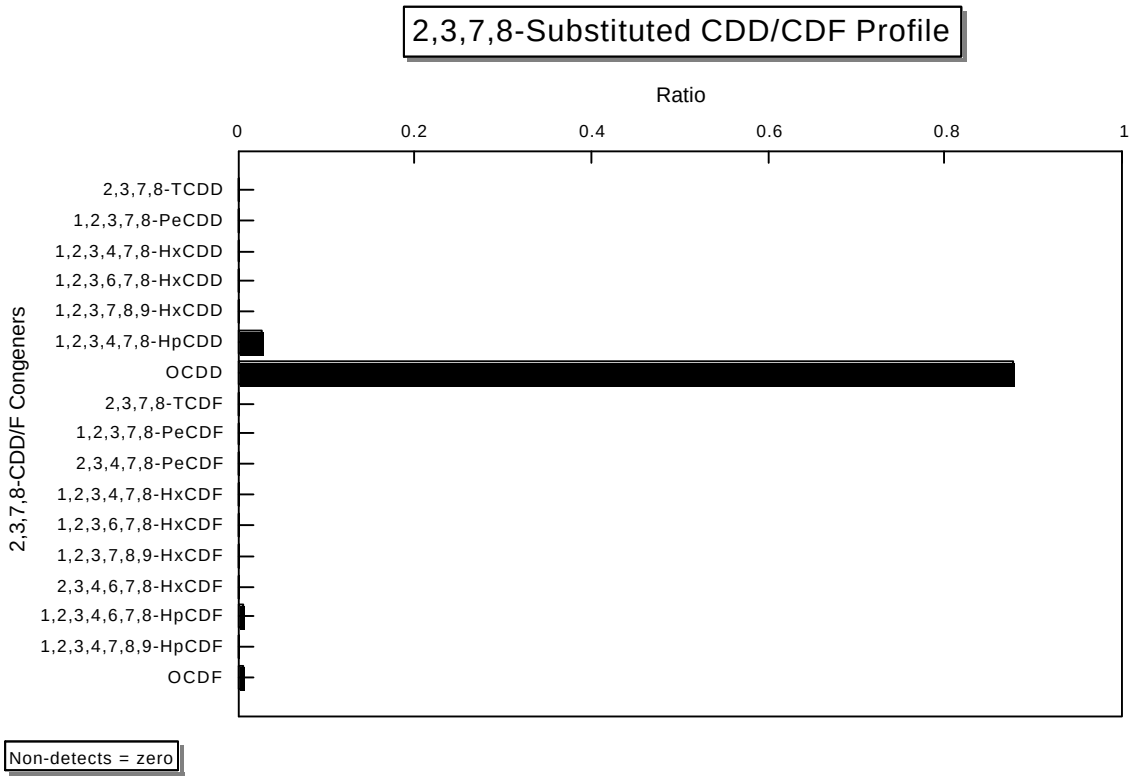



Note: Based on data from Reed et al. (1990), BC Environment (1995), U.S. EPA (1996), MRI (1992), and Tewhey Associates (1997).

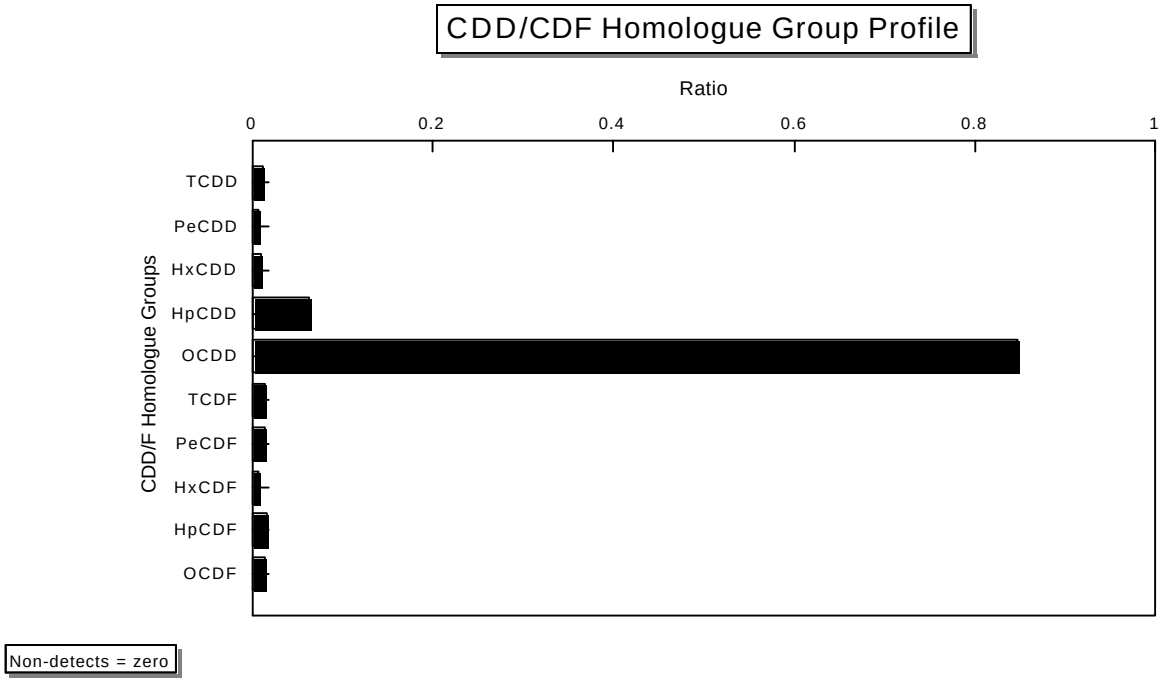



Note: Based on data from Reed et al. (1990), BC Environment (1995), U.S. EPA (1996), MRI (1992), and Tewhey Associates (1997).

Figure 3-3. CDD/CDF Profiles for Rural Soils

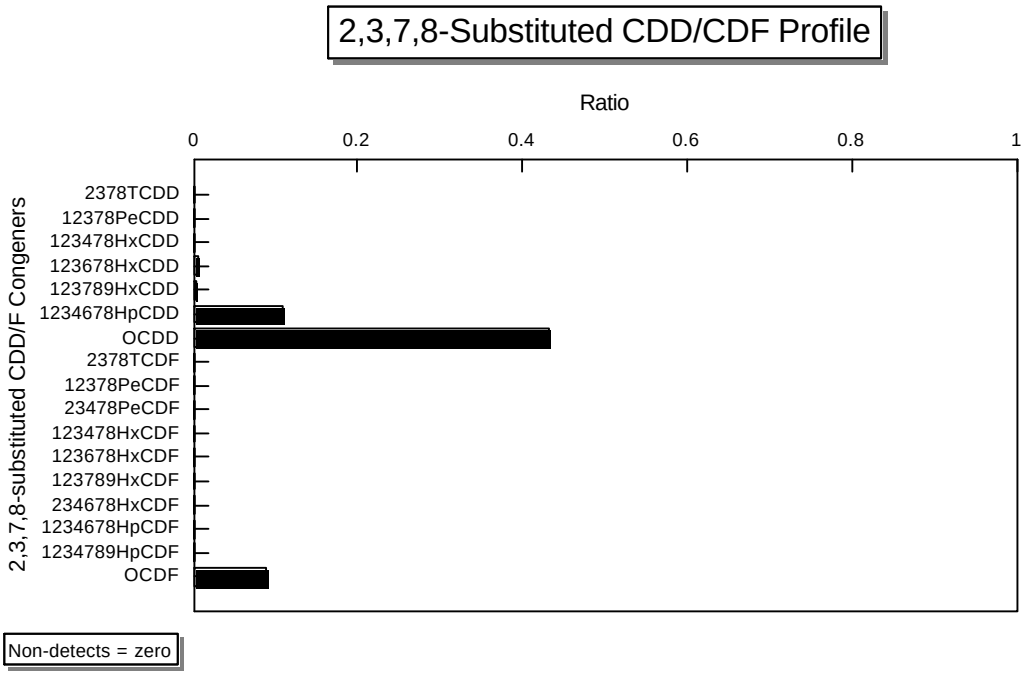



Note: Based on data from U.S EPA (1996), and NIH (1995).

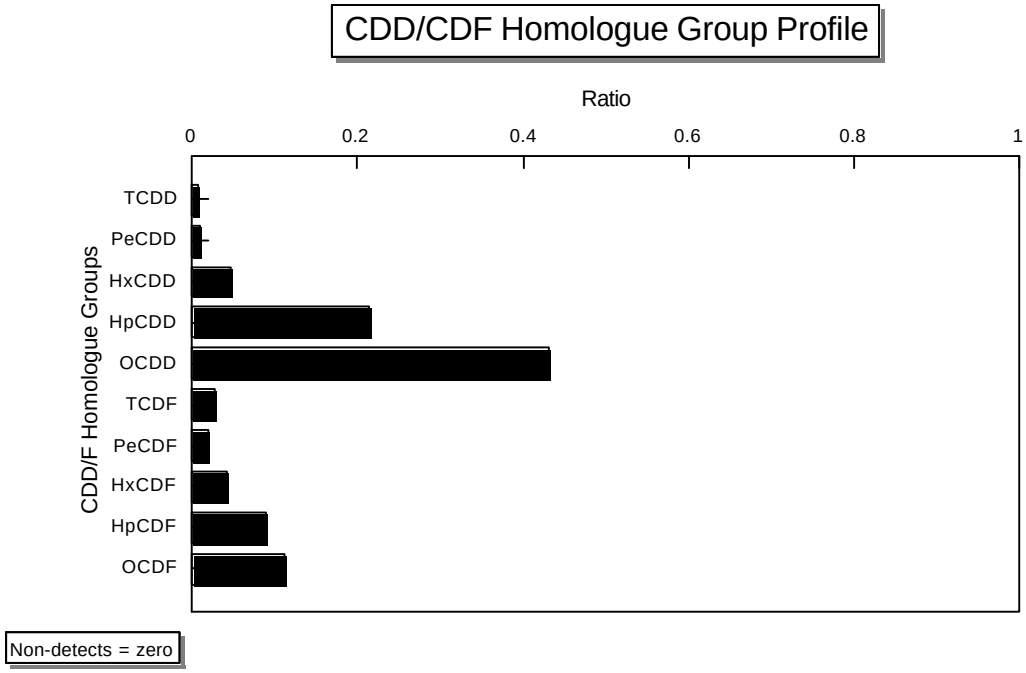



Note: Based on data from Birmingham (1990), Pearson et al. (1990), NIH (1995), and U.S. EPA (1996).

Figure 3-4. CDD/CDF Profiles for Urban Soils

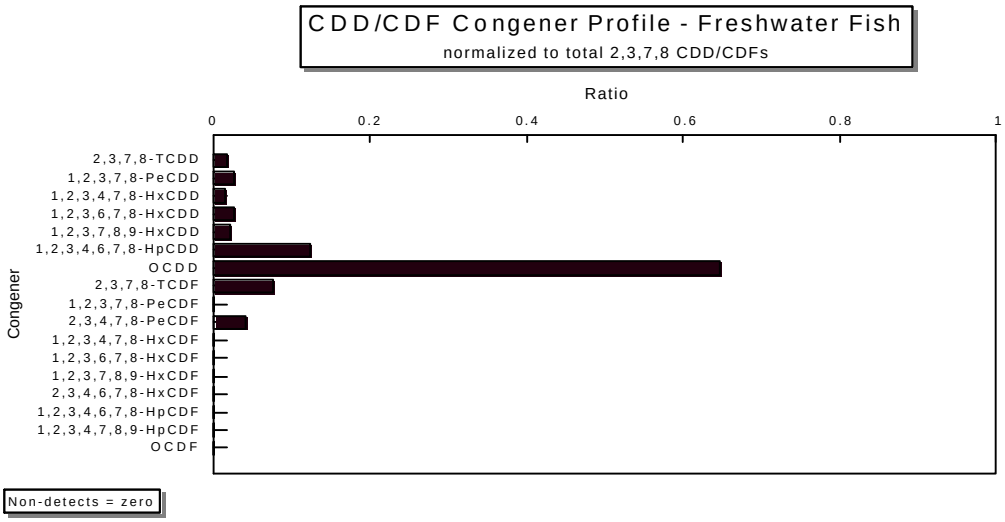



Note: Based on data from Cleverly et al. (1996) and Versar (1996a).

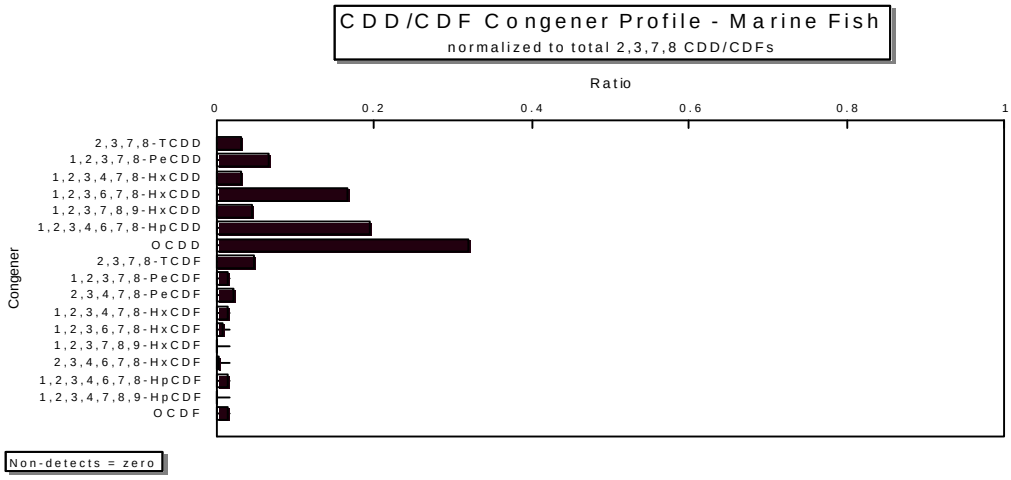



Note: Based on data from Cleverly et al. (1996) and Versar (1996a).

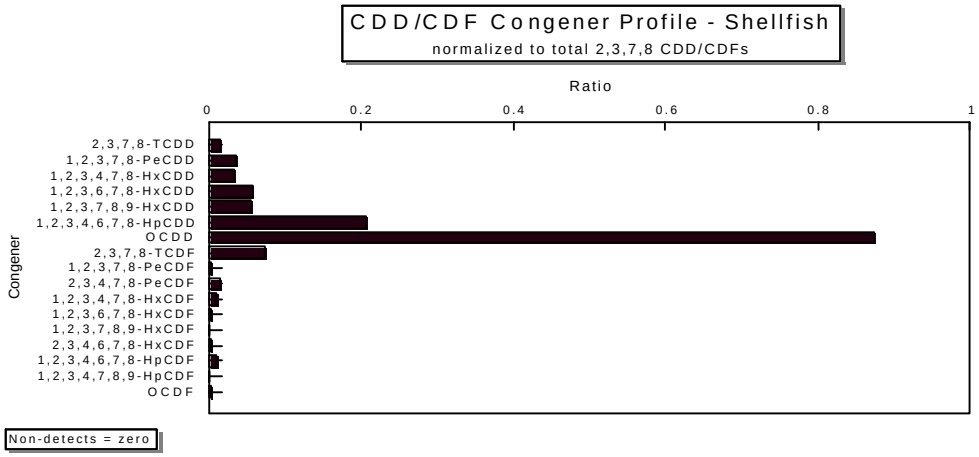
Figure 3-5. CDD/CDF Profiles for Sediment




Note: Based on data from Schecter et al. (1997).

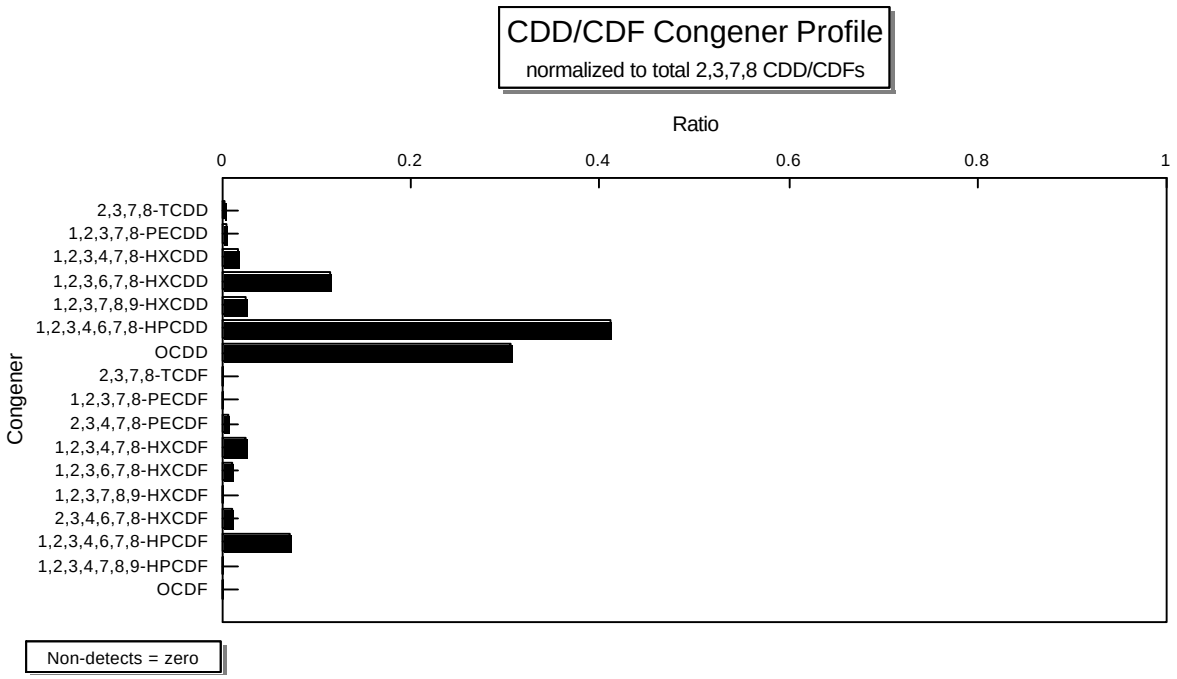



Note: Based on data from Fiedler et al. (1997c).



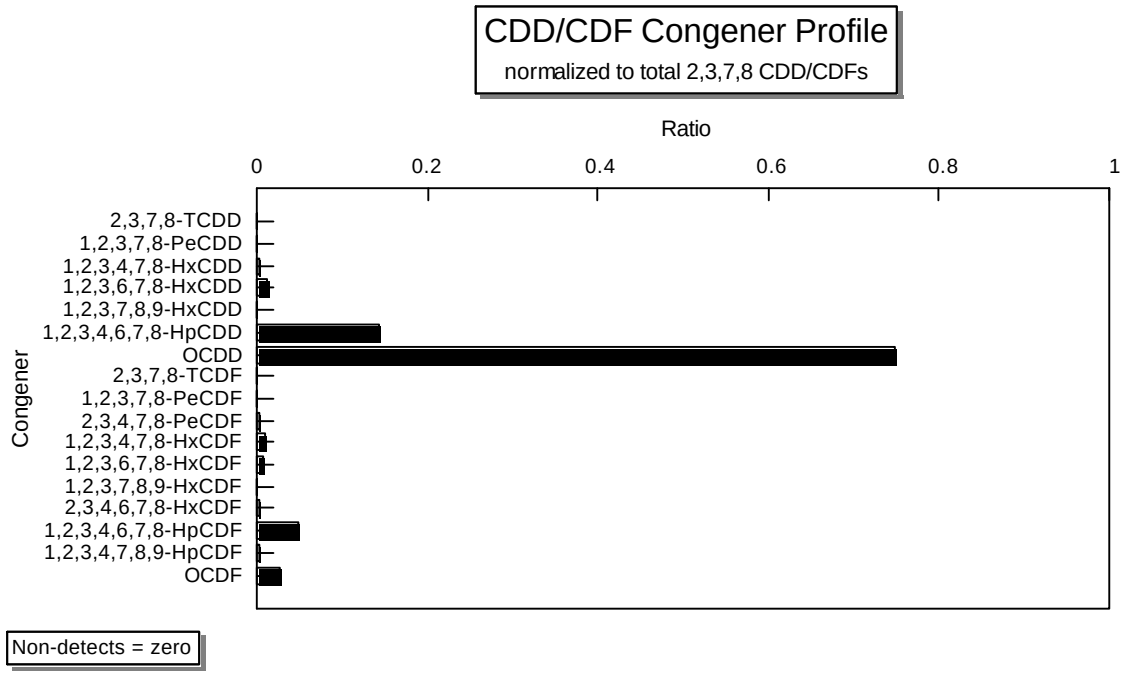

Note: Based on data from Fiedler et. al. (1997c).

Figure 3-6. CDD/CDF Congener Profiles for Fish and Shellfish



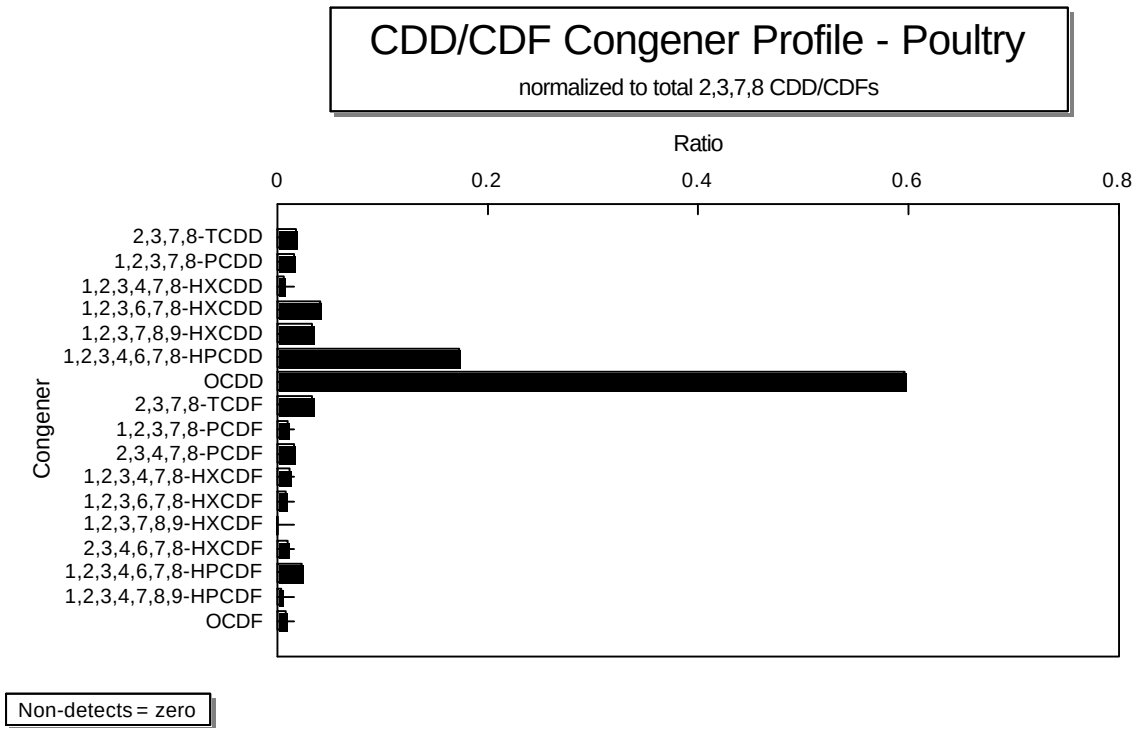

Note: Based on data from Winters et al. (1996a)

Figure 3-7. CDD/CDF Congener Profile for Beef

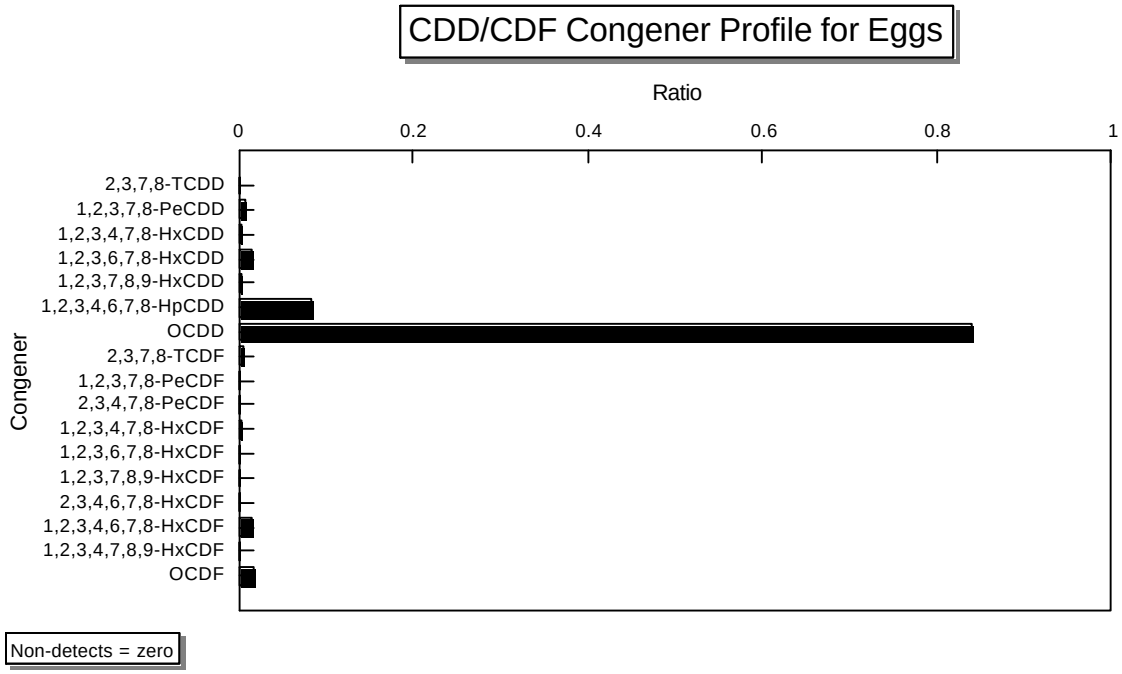



Note: Based on data from Lorber et al. (1997b).

Figure 3-8. CDD/CDF Congener Profile for Pork

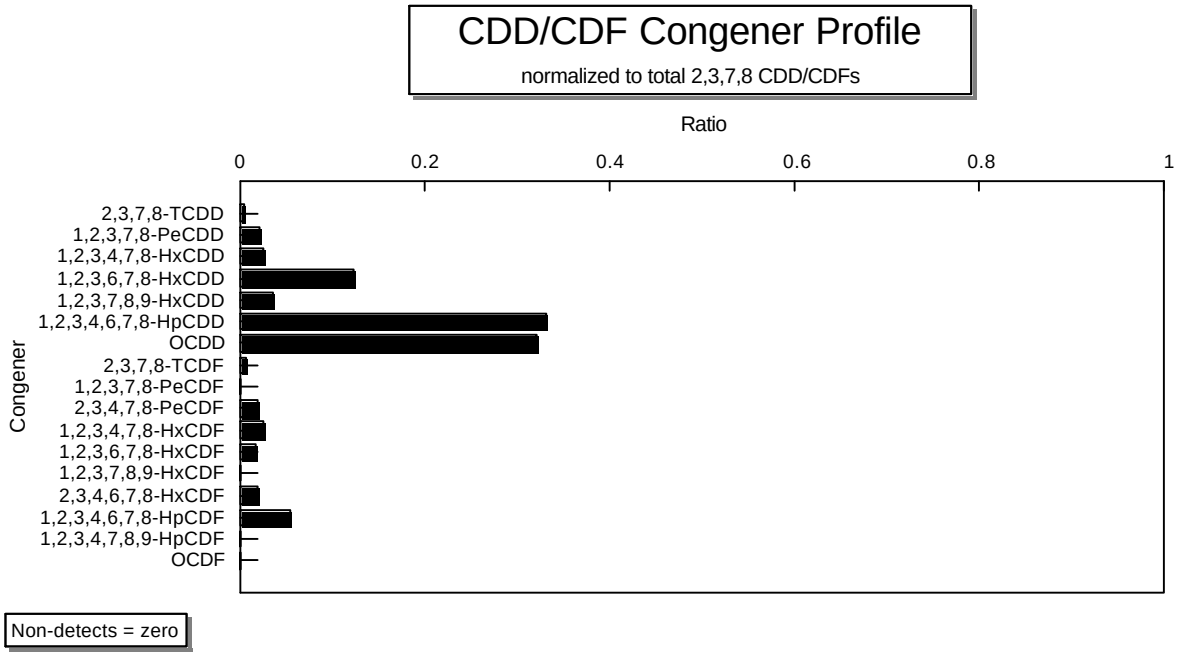



Note: Based on Ferrario et al. (1997)




Note: Based on Fiedler et al. (1997c).

Figure 3-9. CDD/CDF Congener Profiles for Poultry and Eggs




Note: Based on data from Lorber et al. (1998b).

Figure 3-10. CDD/CDF Congener Profiles for Milk and Dairy Products

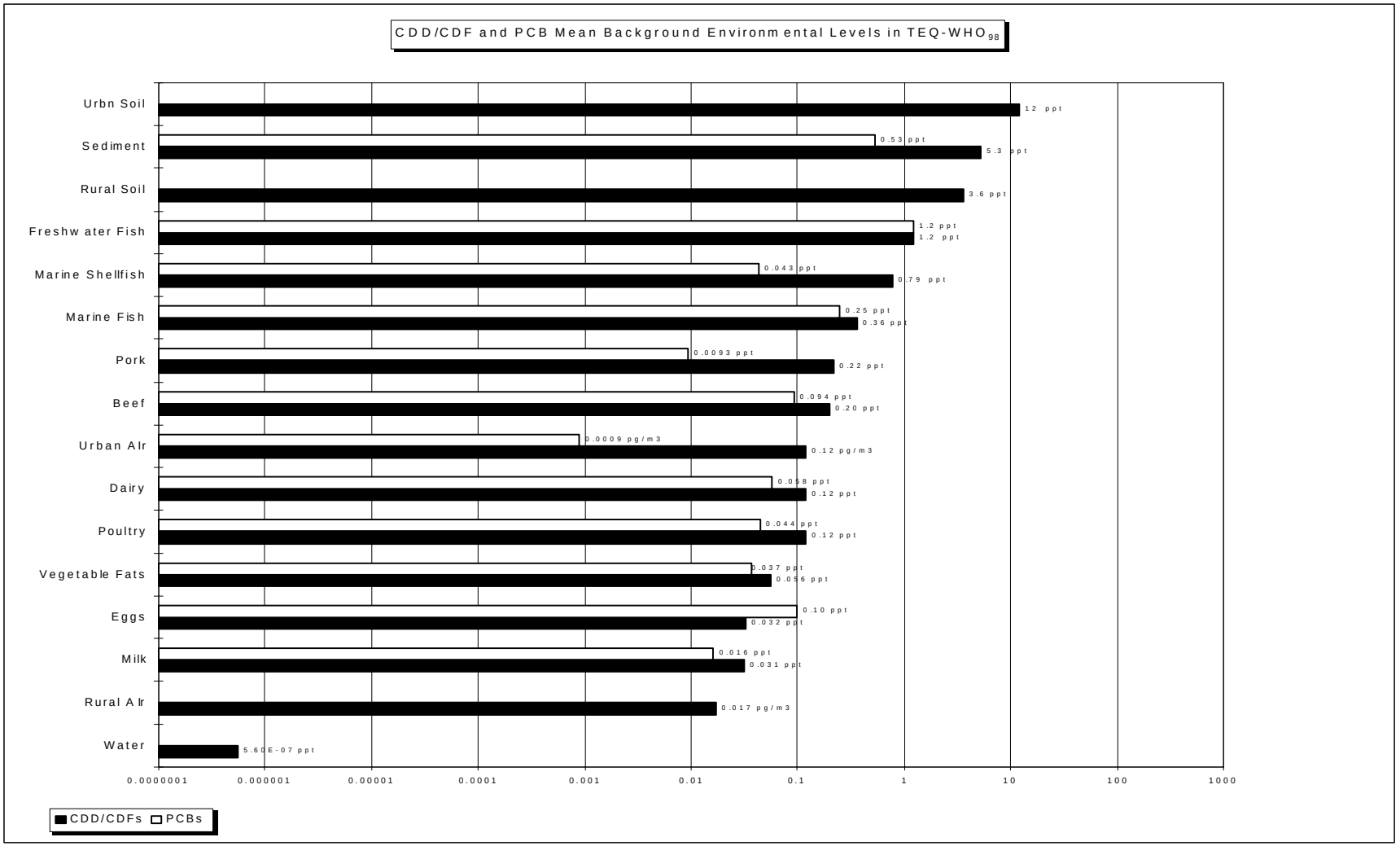



Figure 3-11. CDD/CDF and PCB Mean Background Environmental Levels in TEQ-$WHO_{98}$