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Comparing the Results of a Monte Carlo Analysis with EPA's
Reasonable Maximum Exposed Individual (RMEI):
A Case Study of a Former Wood Treatment Site

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In the United States, there are about 250 former sites that treated wood with preservatives that are now in need of some degree of remediation. The soil at many of these sites is contaminated with creosote, polycyclic aromatic hydrocarbons (PAHs), polychlorinated dibenzo-*p*-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs). This paper compares the results of the current USEPA point estimate (deterministic) approach for predicting the health risks associated with exposure to PCDDs/PCDFs in soil with the results of a probabilistic approach which uses a Monte Carlo analysis. At many of these wood treatment sites the hazard posed by the PAHs, and especially pentachlorophenol, can be much greater than that due to PCDD and PCDFs; however, because at this site the health risk associated with PAHs was deemed negligible by ATSDR, only PCDDs/PCDFs were evaluated. Octachlorodibenzo-*p*-dioxin (OCDD) and octachlorodibenzofuran (OCDF) congeners were evaluated independently from the other congeners due to their prevalence in the environment and the availability of congener-specific data. The results of the reevaluation of the rodent bioassay data for 2,3,7,8-TCDD were considered in the probability distribution for the cancer potency factor. The authors' analyses indicate that when assessing exposure to soil via inhalation, ingestion, and dermal contact, the current regulatory approach used to estimate the reasonable maximally exposed individual (RMEI) (USEPA, *Risk Assessment Guidance for Superfund*, Vol. 1, Part A, 1989) can predict risks which are 10- to 100-fold greater than the 95th percentile risk predicted by a Monte Carlo analysis. © 1993 Academic Press, Inc.

INTRODUCTION

Current United States Environmental Protection Agency (USEPA) guidance suggests that it is acceptable to consider only a "reasonable maximum exposure" (RME) when assessing the potential human health risks associated with exposure to environmental contaminants (USEPA 1989, 1992a). An RME, as defined by the USEPA, is the "highest exposure that is reasonably expected to occur" and is estimated by using

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upperbound values (often the upper 95th confidence limit) of the environmental contaminant concentrations and human exposure factors (USEPA, 1989). Although perhaps useful as a screening method to determine if risks fall below a de minimis value, this approach should not be considered an accurate estimate of the potential health risks associated with a specific site or condition (Paustenbach *et al.*, 1990). As noted by Burmaster and Lehr (1991), rather than estimate the risks to the 95th percentile person, the RME approach can often predict the risks for the 99.9th percentile person.

The deficiencies in the RME approach were discussed at a recent symposium held at the National Academy of Science (NAS, 1992) which was sponsored by the USEPA and others. A consensus was reached by the conference attendees that the optimal method for characterizing health risks was to use probability-based techniques, such as a Monte Carlo analysis (Finley and Paustenbach, 1993). One advantage of using this approach is that it allows consideration of a range of plausible values for contaminant concentrations, each having as many as 100 exposure parameters, and a number of cancer potency factors; as opposed to the point estimate approach for which no quantitative information regarding uncertainty is provided (Finkel, 1990; Burmaster and Lehr, 1991; Burmaster and von Stackelberg, 1991; Paustenbach *et al.*, 1991b; McKone and Bogen, 1991, 1992). As noted by those who have used the techniques, a description of uncertainty in the exposure and risk estimates will nearly always improve the quality of risk management decisions since it provides the range and relative frequency of the estimated health risks and the associated probabilities.

The advantages and disadvantages of the point estimate approach and the RMEI scenario have been addressed in a number of papers. In an attempt to remedy these, the EPA Exposure Assessment Group recently proposed using the "central" and "high end" scenario method. The central scenario "corresponds to average or median levels and high scenarios are defined as level above the 90th percentile but within the actual range of exposure levels" (USEPA, 1993).

The Monte Carlo approach has been heralded as one of the most important advances in exposure assessment of the past 20 years. It has been used to assess PCBs in soil (Eschenroeder and Faeder, 1988), chrome in soil (Paustenbach *et al.*, 1991b), TCE in groundwater (McKone and Bogen, 1991), volatiles in ambient air (Burmaster and von Stackelberg, 1991), and TCDD in soil (Paustenbach *et al.*, 1992a).

In this paper, we evaluated a former wood treatment facility. Both the point estimate and probabilistic approaches were used to estimate the uptake of polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) for residents living adjacent to the facility. It operated for over 60 years during which time creosote, pentachlorophenol, CCA (chromated copper arsenate), and ZMA (zinc meta-arsenate) were used as preservatives of wooden utility poles and railroad ties. Contaminated soil and sediment were the only media of concern.

METHODS

The Monte Carlo technique has been described in several papers (Finkel, 1990; Paustenbach *et al.*, 1992a). Generally, one of two or three commercial programs are used. The most popular ones are Crystal Ball (1991) and @Risk (1990).

In contrast to the point estimate or deterministic approach, which uses single exposure values to calculate a single risk estimate (MEI, MLEI, RME, etc.), probabilistic

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risk assessments rely upon a Monte Carlo analysis of population distributions (also known as probability density functions, or "PDFs") to develop a range of risk estimates, each with an associated probability (Finkel, 1990; McKone and Bogen, 1991). Each exposure factor is assigned a range of values (rather than a single value) compiled from some or all of the relevant data pertaining to that parameter.

In the first step of a probabilistic assessment, the available data for each exposure parameter are evaluated with respect to distribution type (e.g., normal, lognormal) and the appropriate descriptors of the distribution (e.g., mean, maximum, and minimum values). Commercially available software programs can take this information and simulate a full distribution frequency for the parameter. If only the range of values is known, a uniform distribution may be assigned to the exposure parameter. If only the range and mode are known, a triangular distribution may be most appropriate. If the data set is not accurately described by a standard distribution (normal, lognormal, etc.), it is usually acceptable to use only the empirical data points themselves ("bootstrapping") rather than attempting to "force fit" the data into a PDF (McKone and Bogen, 1991). The authors have found that one can usually characterize the data by one of these distributions and that the eventual impact of selecting a distribution in a less-than-perfect manner has minimal impact on the final risk estimates (usually less than 10%).

In the next step, the risk calculation algorithm is solved several thousand times using a Monte Carlo program which draws values randomly from each exposure PDF. The Monte Carlo method selects values from each PDF at a frequency that is related to the percentile of the value in the PDF; (i.e., for a normal distribution the 50th percentile is chosen more often than 5th and 95th percentile values). Typically, a minimum of 5000 iterations is needed to ensure that a "point of convergence" is reached, i.e., additional iterations will not significantly alter the results (McKone and Bogen, 1991). The output is a single distribution of risk values and the associated probabilities.

As an initial step in our probabilistic assessment, a sensitivity analysis was conducted to identify the key parameters, (e.g., exposure variables and cancer potency) for which appropriate probability distributions were developed. The distributions were based on data from USEPA guidance documents and the recent scientific literature. Consistent with other probabilistic risk assessments, parameters with an undefined data distribution were assigned a uniform distribution (Salhotra *et al.*, 1991; Paustenbach *et al.*, 1991b; Finley and Paustenbach, 1993). For parameters involving a manageable data set of discrete values, the individual values were "bootstrapped" (e.g., considered with equal probability) in the Monte Carlo analysis.

The importance of dependency relationships among specific parameters (so-called interlinking variables) was considered in our analysis. Examples of interlinking variables include the relationship between inhalation rate and body weight, and the amount of breast milk ingested per day and infant body weight. When interlinking parameters were identified, the value selected for the dependent variable (e.g., breast milk ingestion rate) was restricted based on specific correlations with the independent variable. This technique has been suggested by McKone and Bogen (1992).

HAZARD IDENTIFICATION

The majority of former wood treatment sites have soils that are contaminated with varying concentrations of polycyclic aromatic hydrocarbons (PAHs), copper, and/or

arsenic, tars, pentachlorophenol, PCDDs, and PCDFs. The number of years of operation and the processing methods will usually influence the extent and severity of contamination. Due to the environmental persistence and toxicity of the PAHs and PCDDs/PCDFs, these are typically the chemicals which dictate whether remediation is necessary, and the degree of cleanup.

In an initial health risk assessment of this site, it was concluded by the ATSDR (1988) that the health risks associated with the presence of PAHs and metals were negligible. Therefore, the authors' analysis focused on the health risks associated only with PCDDs and PCDFs in soils and sediment. Octachlorodibenzo-*p*-dioxin (OCDD) and octachlorodibenzofuran (OCDF) congeners were evaluated independently from the other PCDD and PCDF congeners. This approach was used because (1) OCDDs and OCDFs were the predominant congeners at this site, as well as at other sites where pentachlorophenol has been used; (2) the concentration times toxicity of the OCDD/OCDF posed a potentially greater hazard than the PAHs; and (3) recent studies using rodents have suggested that the bioavailability of OCDD is significantly lower than that of 2,3,7,8-TCDD (Birnbaum and Couture, 1988; Couture *et al.*, 1988).

TCDD Toxicity Equivalents

Studies focusing on the structure-activity relationships of the various PCDD/PCDF congeners have clearly shown that not all PCDDs and PCDFs are equally potent carcinogens (Safe, 1987). This complicates many risk assessments since PCDDs and PCDFs are generally found in the environment as mixtures containing many of the possible 210 congeners at varying concentrations. An interim approach has been adopted by the USEPA (Bellin and Barnes, 1989) for evaluating the carcinogenic hazard posed by PCDDs and PCDFs. This approach normalizes the potency of individual congeners relative to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), which has a potency of 1.0.

Potency values, termed toxicity equivalency factors (TEFs), have been assigned only for the 2,3,7,8-substituted congeners since numerous structure-activity and structure-binding studies have established that non-2,3,7,8-substituted congeners are relatively inactive in biological systems (Mason *et al.*, 1985, 1986; Harris *et al.*, 1990). The TEFs for the PCDD/PCDFs are based on their relative potencies which are estimated from the results of various biological endpoints including (1) induction of cytochrome P450-dependent monooxygenases in both human and rodent cell culture as well as in animal models, (2) immunotoxicity in mice, (3) body weight loss in a variety of animal models, (4) reproductive toxicity in mice, and (5) receptor-binding studies (Safe, 1987). It is important to emphasize that, although the TEFs determined from these assays are used in risk assessments, the endpoints measured in these assays (e.g., increased cytochrome P450 activity) have not been shown to be directly related to the carcinogenic process or carcinogenic potency. Indeed, there is no evidence of carcinogenicity for most of the congeners assigned TEFs.

For the purposes of assessing the health risks of mixtures of PCDDs and PCDFs, the USEPA has suggested the 2,3,7,8-TCDD "toxicity equivalent" (TEQ) approach. To obtain a 2,3,7,8-TCDD TEQ value for a given congener in a mixture, the TEF value for that congener is multiplied by the concentration of the individual congener in the mixture. For example, the current international TEF (I-TEF) for OCDD is

0.001; therefore, ingesting 20,000 $\mu\text{g}/\text{kg}$ of OCDD is (for regulatory purposes) theoretically as potent as ingesting 20 $\mu\text{g}/\text{kg}$ of 2,3,7,8-TCDD. This procedure is applied to data for each individual congener, or each class of congeners, and the 2,3,7,8-TCDD TEQs are then summed to obtain a total 2,3,7,8-TCDD TEQ concentration in the contaminated media (Bellin and Barnes, 1989).

The current I-TEFs and those used previously by USEPA are presented in Table 1. The USEPA TEFs were formally adopted in the spring of 1987 and were based on data available through 1985. The 1989 I-TEFs represent modifications to the original 1987 values. For wood treatment sites, the most significant modification is the increase in TEF values for OCDD and OCDF from 0 to 0.001. The modification was based on the results of two studies which reported that (1) the OCDD concentration needed to achieve maximal hepatic cytochrome P450 induction was 650 times that of 2,3,7,8-TCDD and (2) OCDD is shown to accumulate in rodent tissue following repeated administration (Birnbaum and Couture, 1988). Interestingly, approximately 75% of the PCDDs typically found in the environment are OCDD (NRCC, 1981) and as much as 90–95% of the PCDDs at a pentachlorophenol wood treatment facility are OCDD or OCDF. Therefore, the assignment of a TEF potency value to octa-chlorinated congeners significantly increased the regulatory concern about these sites.

Weight of Evidence for Carcinogenicity in Humans

There is a good deal of epidemiological data concerning occupational and environmental exposure to 2,3,7,8-TCDD. Prior to 1992, the majority of these studies reported

TABLE 1
COMPARISON OF THE 1989 INTERNATIONAL TEFs (I-TEFs) AND THE 1987 USEPA TEFs

| Congener | 1989 I-TEFs | 1987 USEPA TEFs |
|-------------------------|-------------|-----------------|
| Mono-, di-, and tri-CDD | 0 | 0 |
| 2,3,7,8-TCDD | 1.0 | 1.0 |
| Other TCDDs | 0 | 0.01 |
| 2,3,7,8-PeCDDs | 0.5 | 0.5 |
| Other PeCDDs | 0 | 0.005 |
| 2,3,7,8-HxCDDs | 0.1 | 0.04 |
| Other HxCDDs | 0 | 0.004 |
| 2,3,7,8-HpCDDs | 0.01 | 0.001 |
| Other HpCDDs | 0 | 0.0001 |
| OCDD | 0.001 | 0 |
| Mono-, di-, and triCDF | 0 | 0 |
| 2,3,7,8-TCDF | 0.1 | 0.1 |
| Other TCDFs | 0 | 0.001 |
| 1,2,3,7,8-PeCDF | 0.05 | 0.1 |
| 2,3,4,7,8-PeCDF | 0.5 | 0.1 |
| Other PeCDFs | 0 | 0.001 |
| 2,3,7,8-HxCDFs | 0.1 | 0.01 |
| Other HxCDFs | 0 | 0.00001 |
| 2,3,7,8-HpCDFs | 0.01 | 0.001 |
| Other HpCDFs | 0 | 0.00001 |
| OCDF | 0.001 | 0 |

no statistically significant increase in the overall incidence of cancers and no consistent organ-specific increased tumor rate. For example, older studies of the largest potentially exposed populations (CDC, 1987; Wiklund and Holm, 1986, 1987; Coggon *et al.*, 1986; Riihimaki *et al.*, 1983; Lynge, 1985; Breslin *et al.*, 1987) and studies of production workers having presumably the highest PCDD exposures (Zack and Gaffey, 1983; Suskind and Hertzberg, 1984; Ott *et al.*, 1987; Bond *et al.*, 1983, 1988, 1989) did not demonstrate significant increases in the total incidence of cancer or a consistent organ-specific excess cancer rate.

The results of the most recent epidemiological studies of dioxin-exposed populations are generally considered more informative than previous ones since better estimates of exposure are available and larger populations were studied. For example, four recent studies used blood measurements of 2,3,7,8-TCDD to estimate past exposure (Zober *et al.*, 1990; Fingerhut *et al.*, 1991; Manz *et al.*, 1991; Roegner *et al.*, 1991). Although there remains disagreement among experts about the meaning of the results, the USEPA and NIOSH believe that people exposed to very high doses are at increased risk of several diseases, including an increase in the overall cancer rate.

Two of these studies involving relatively large (>250 cancer deaths) exposed populations (Saracci *et al.*, 1991; Fingerhut *et al.*, 1991) examined the relationship between 2,3,7,8-TCDD exposure and cancer. Saracci *et al.* (1991) analyzed the largest cohort of production workers potentially exposed to PCDDs and reported no significant excess in overall cancers or common cancer types. However, Fingerhut (1992) believes that the NIOSH study, Zober *et al.* (1990), and Manz *et al.* (1991) found significant excesses in the total cancer rate. The increases in the lung cancer incidence could not be fully evaluated since data about the workers smoking habits were not robust. Although no chemical or agent, heretofore, has been shown to produce an increased incidence in all cancers, some believe that a promoter such as 2,3,7,8-TCDD could increase the risk in several organs. These data suggest that 2,3,7,8-TCDD may be able to cause a slight increased cancer risk in those who had peak body burdens above 5000 ppm. Sweeney *et al.* (1992) reported an increased incidence of diabetes in the workers.

Immunotoxicity

It has been suggested that the most sensitive indicator of the adverse effects associated with exposure to 2,3,7,8-TCDD is immunotoxicity. A spectrum of immunotoxicological effects has been reported in animals including atrophy of thymus and spleen (associated with a reduction in the generation of cytolytic T-lymphocytes), suppression of antibody response, effects on serum immunoglobulins, alterations in lymphocyte development and homing, reduced activity of activated polymorphonuclear neutrophils, and effects on host resistance (Holsapple *et al.*, 1991). Unfortunately, the available data are inadequate to quantitatively evaluate the dose-response relationship for immunotoxic effects in animals.

While the most sensitive animal responses are the acquired immune responses to primary antigen challenge, most human studies have measured either the response to various mitogens, the secondary response to recall antigens, or changes in lymphocyte subpopulations (Holsapple *et al.*, 1991). Because the effects studied in human populations have not been studied in animal models, it is difficult to extrapolate findings

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in animals to humans. In an epidemiological study of 18 British workers exposed to 2,4,5-T, it was reported that the levels of B and T cells were unaltered; however, the levels of circulating natural killer (NK) cells were elevated (Jennings *et al.*, 1988). In a study of children exposed to relatively high concentrations of 2,3,7,8-TCDD in Seveso, Italy, no changes in immune function were reported 3 years after exposure; however, 6 years postexposure an increase in complement protein levels was reported in children having overt signs of chloracne (Tognoni and Bonaccorsi, 1982). In a study of individuals exposed in Times Beach, Missouri, a nonstatistically significant decrease in T4⁺/T8⁺ ratios was reported; however, there was a statistically significant increase in the total number of platelets (Knutson, 1984). The results were not reproducible in a follow-up study and no adverse effects of any type were observed (Stehr-Green *et al.*, 1986). Due to the lack of clinical evidence of less than optimal immune function and questions about the administration of the sensitized in the original study, the initial findings have generally been considered an artifact.

Until recently, doses of 2,3,7,8-TCDD which were low enough to produce no adverse effect on the immune system were based on studies of only guinea pigs or mice. Vos *et al.* (1973) reported depressed delayed-type hypersensitivity (DTH) responses to tuberculin in adult guinea pigs fed a diet of 0.04 $\mu\text{g}/\text{kg}\text{-week}$ (6000 $\text{pg}/\text{kg}\text{-day}$) for 8 weeks. In an attempt to identify a regulatory limit which would prevent adverse effects on the immune system of humans, the United Kingdom applied a safety factor of 100 to this minimal effect level and they suggested a 60 $\text{pg}/\text{kg}\text{-day}$ exposure guideline (UK, 1989).

Developmental and Reproductive Toxicity

The developmental toxicity of 2,3,7,8-TCDD has been evaluated on several occasions in animals. Teratogenic effects have been reported in mice at doses less than those resulting in maternal toxicity; however, developmental effects in other species have been noted only at maternally toxic doses (ATSDR, 1988; Couture *et al.*, 1990). In contrast, hydronephrotic kidney and cleft palate have been reported at doses as low as 1 $\mu\text{g}/\text{kg}$ in C57BL/6N mice (Couture *et al.*, 1990).

Birnbaum *et al.* (1987) treated pregnant mice with either 2,3,4,7,8-PeCDF (0–30 $\mu\text{g}/\text{kg}$) or 1,2,3,4,7,8-HxCDF (0–300 $\mu\text{g}/\text{kg}$) on Gestation Days 10–13, followed by necropsy on Day 18. Both chemicals were found to cause hydronephrosis and cleft palate in the fetuses without overt toxicity to the dam (Birnbaum *et al.*, 1987). Weber *et al.* (1984) reported teratogenic effects of 2,3,7,8-TCDF in C57BL/6N mice at doses of 250 $\mu\text{g}/\text{kg}$. Schwetz *et al.* (1973) investigated the effect of OCDD on the maternal and fetal body measurements as well as the incidence of fetal resorption in rats. At doses as high as 500 $\text{mg}/\text{kg}\text{-day}$, no significant difference from control groups was observed with regard to fetal body measurements, maternal weight gain, or fetal resorption.

Developmental effects have been reported in the rabbit at doses associated with chemically induced maternal stress (Khera, 1984, 1987). There is no evidence that PCDDs or PCDFs, including 2,3,7,8-TCDD, act as developmental toxicants in humans (ATSDR, 1988; Couture *et al.*, 1990).

Reproductive effects associated with administration of 2,3,7,8-TCDD have been reported in rats, mice, rabbits, and nonhuman primates (DHS, 1985; ATSDR, 1988).

In a three-generation study of Sprague-Dawley rats fed a diet containing doses of 0, 0.001, 0.01, and 0.1 $\mu\text{g}/\text{kg}\text{-day}$ for 90 days, a significant reduction in fertility and litter size was observed in the f_0 offspring of female rats receiving 0.1 $\mu\text{g}/\text{kg}\text{-day}$. At 0.01 $\mu\text{g}/\text{kg}\text{-day}$, significant reductions in fertility, postnatal survival, and growth were observed in f_1 and f_2 generations. Significant decreases in gestational survival, litter size, and average postnatal body weight were noted among the f_2 and f_3 litters of this same dose group. The authors concluded that 0.001 $\mu\text{g}/\text{kg}\text{-day}$ was a no-observable-effect level (NOEL). NOELs for reproductive toxicity in nonhuman primates are lower than those reported for rodents. In a subchronic fetotoxicity study, McNulty (1984) reported occurrences of abortion in rhesus monkeys treated with 1 $\mu\text{g}/\text{kg}$ of 2,3,7,8-TCDD between Days 20 and 40 of gestation. At 0.2 $\mu\text{g}/\text{kg}$, 1 of 4 animals aborted. A low incidence of palatal abnormalities was the only developmental effect reported. The authors concluded that the number of animals in the 0.2 $\mu\text{g}/\text{kg}$ group was not adequate for statistical evaluation or the assignment of a NOEL.

There is insufficient evidence to indicate that 2,3,7,8-TCDD is a reproductive toxicant in humans. Epidemiological studies have been conducted for human populations exposed to 2,3,7,8-TCDD in herbicides and other industrial chemicals, and no adverse effects have been reported for reproductive performance in exposed males, or developmental effects in offspring (ATSDR, 1988). Because of the limitations in these studies, particularly in terms of exposure, the results of the Ranch Hand investigation of Army personnel working with Agent Orange, likely to be completed in 1993, are of special interest.

SITE-SPECIFIC DATA

Soil and Sediment Data

At this site, a total of 23 off-site surface soil and sediment samples were collected and these were used to estimate exposure to residents (Table 2). Thirteen samples were collected by the USEPA in locations approximately 500 to 3000 feet from the site and 10 additional off-site samples were collected by an independent contractor. Six surface soil samples were collected near the residences adjacent to the facility. Seventeen sediment samples were collected from nearby drainage ditches which may or may not have historically received runoff from the site (Fig. 1).

The soil and sediment samples were analyzed for pentachlorophenol (USEPA Method 8040) and 2,3,7,8-substituted PCDDs and PCDFs (USEPA Method 8280). Pentachlorophenol was not detected in any sample at a limit of detection of 1.7 mg/kg. Soils were also analyzed for metals and polycyclic aromatic hydrocarbons (PAHs); however, the frequency of detection and concentrations were not considered to be significant and were therefore not quantitatively assessed.

Six types of data were considered in our analysis: octa congeners, non-octa congeners, and all congeners combined. A Kolmogorov-Smirnov distribution analysis (Wilkinson, 1989) was conducted on each of the data sets to determine the appropriate distribution. The results indicated that the soil and sediment contaminant concentrations were neither normally nor lognormally distributed (Fig. 2), in part, due to the small sample size. The octa congener data for the six soil samples taken in the residential area appeared to (and would be expected to) fit a lognormal distribution.

The concentrations of PCDDs and PCDFs (reported as 2,3,7,8-TCDD TEQ) are presented in Table 2. Most of the congeners were not detectable in a majority of the samples. In the exposure calculations, one-half the detection limit was used whenever no detectable quantity was measured, as recommended by various guidance documents (USEPA, 1989) and several published papers (Parkin *et al.*, 1988; Haas and Scheff, 1990; Travis *et al.*, 1990). As shown in Table 2, the 2,3,7,8-TCDD TEQ concentrations for off-site soil (including nondetects; where 0.005 $\mu\text{g}/\text{kg}$ was used for those samples below the LOD) ranged from 0.006 to 5.4 $\mu\text{g}/\text{kg}$ (arithmetic mean = 1.5 $\mu\text{g}/\text{kg}$). The range of concentrations reported reflects the variability of the contamination patterns with geographic location. No subsurface samples (below 12 inches) had detectable concentrations of PCDD and PCDF congeners.

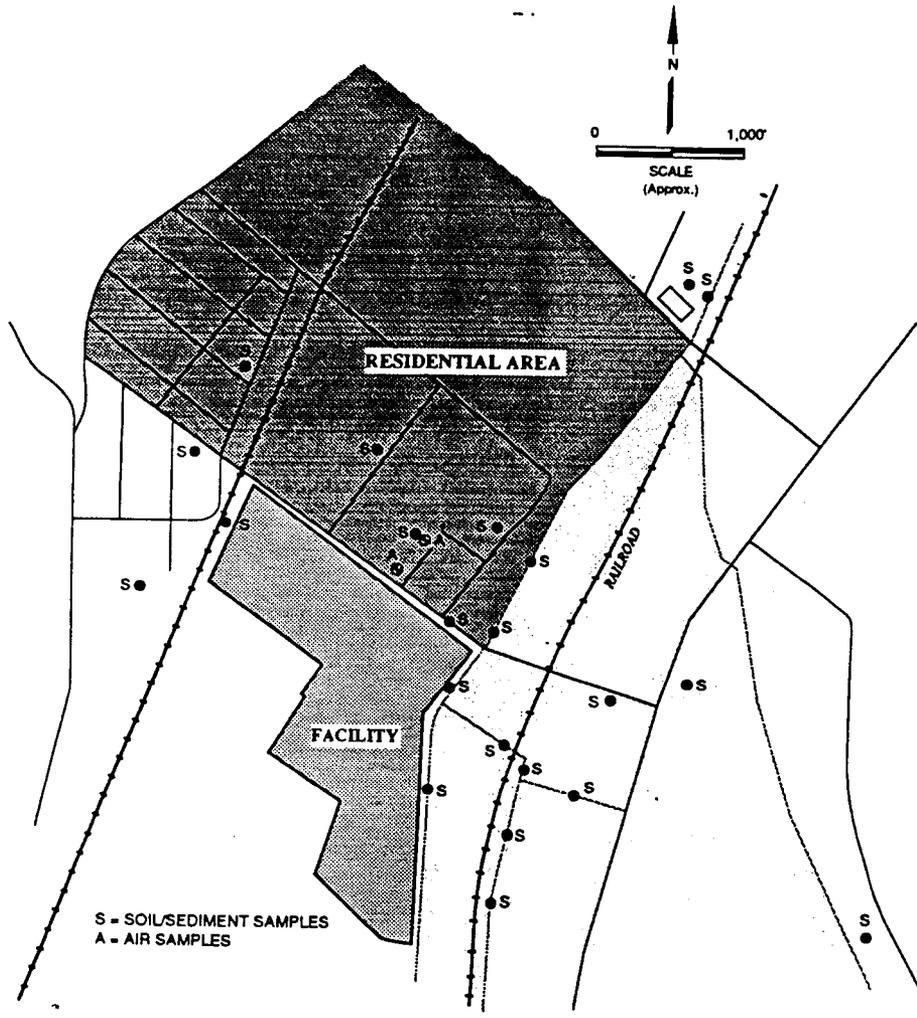


FIG. 1. Plan view of the site evaluated in this assessment. The locations of soil/sediment and air samples are shown.

TABLE 2
CONCENTRATIONS OF PCDDs AND PCDFs IN OFF-SITE SOIL AND SEDIMENT SAMPLES^a

| Sample | Non-octa ^b TEQ (ng/kg) | Octa ^c TEQ (ng/kg) | Total TEQ (ng/kg) | % Contribution of OCDD/OCDF to total 2,3,7,8-TCDD TEQ | Media sampled |
|--------|---|-------------------------------------|-------------------------|---|------------------|
| 01-S | 13 | 8 | 21 | 38 | Soil |
| 02-S | 6 | 1 | 7 | 10 | Sed |
| 03-S | 6 | 1 | 7 | 18 | Soil |
| 04-S | 5 | 1 | 6 | 10 | Soil |
| 05-S | 7 | 3 | 10 | 31 | Soil |
| 06-S | 6 | 1 | 7 | 10 | Sed |
| 07-S | 1330 | 540 | 1870 | 31 | Sed |
| 08-S | 6 | 2 | 8 | 29 | Soil |
| 09-S | 6 | 4 | 10 | 44 | Sed |
| 11-S | 19 | 6 | 25 | 23 | Sed |
| 12-S | 50 | 20 | 70 | 29 | Sed |
| 14-S | 6 | 1 | 7 | 15 | Sed |
| 15-S | 6 | 1 | 7 | 12 | Sed |
| SD-1 | 24 | 52 | 76 | 32 | Sed |
| SD-2 | 2450 | 2942 | 5390 | 45 | Sed |
| SD-3 | 6 | 12 | 18 | 34 | Sed |
| SD-4 | 103 | 352 | 455 | 23 | Sed |
| SD-5 | 1510 | 3071 | 4581 | 33 | Sed |
| SD-6 | 1166 | 749 | 1915 | 61 | Sed |
| SD-7 | 1320 | 1924 | 3244 | 41 | Sed |
| SD-8 | 980 | 3147 | 4127 | 24 | Sed |
| SD-9 | 6 | 9 | 15 | 39 | Sed |
| SB-4 | 13 | 25 | 38 | 35 | Soil |

Note. Soil: soil samples collected near residences (see Fig. 1 for sample location). Sed: sediment samples collected from drainage ditches (see Fig. 1 for sample location).

^a Reported as 2,3,7,8-TCDD TEQ.

^b Sum of mono- through hepta-substituted congeners.

^c Octa-substituted congeners only.

Air-Sampling Data

Sampling for airborne particulates was conducted during the remediation of on-site soils. The data were used to estimate the airborne concentration of PCDD/PCDFs in residential areas. Five air-monitoring stations located on-site ($n = 3$) as well as in adjacent residential areas ($n = 2$) collected particulate samples on three consecutive days (Table 3).

DOSE-RESPONSE ASSESSMENT

2,3,7,8-TCDD has not been shown to be an initiator (Shu *et al.*, 1987); however, it may act as a promoter of carcinogenic processes. Promoters require a minimum threshold dose to elicit the necessary cellular responses leading to tumor growth (Wasom, 1977; Shu *et al.*, 1987; Hebert *et al.*, 1990). Due to the lack of genotoxicity of the PCDD and PCDFs, several agencies in the United States, Canada, and Europe

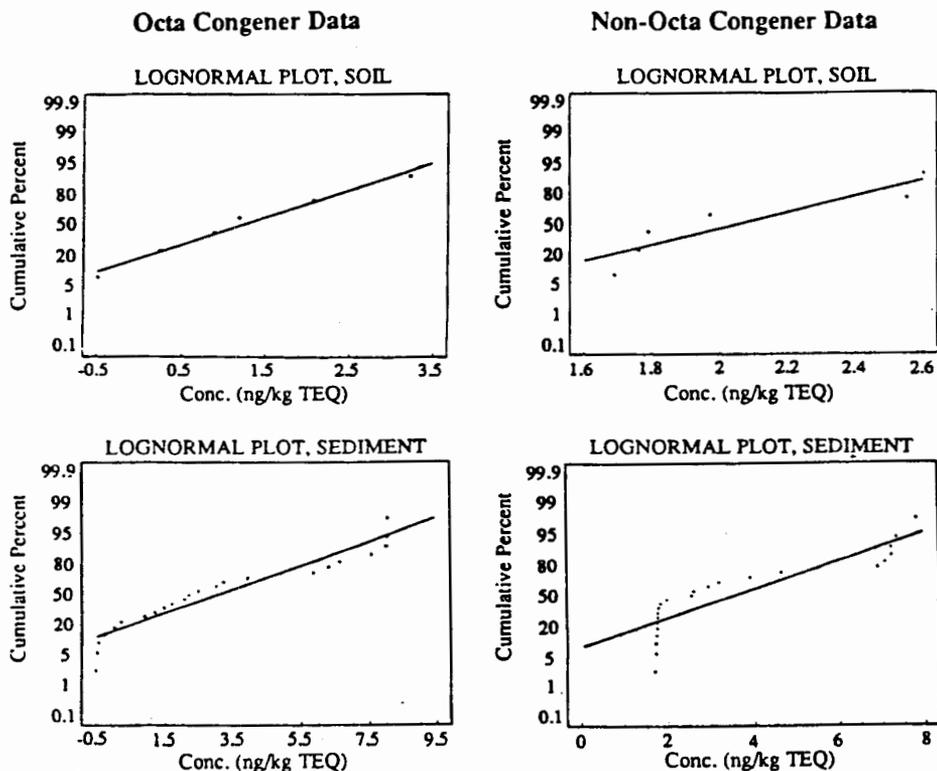


FIG. 2. Results of Kolmogorov-Smirnov data distribution analysis (Wilkinson, 1989) for soil and sediment data. This analysis determines the shape and location of a sample distribution based on the maximum difference between a continuous and a discrete cumulative distribution function. The graphs are generated as cumulative probability plots for each data set. Only one data set (soil data for the octa congeners) appeared to fit a lognormal distribution; however, the number of samples was not adequate to define the distribution.

have previously used a threshold approach for evaluating 2,3,7,8-TCDD cancer risks, i.e., establishing a no-observable-adverse-effect level (NOAEL) for estimating "safe" or acceptable levels of exposure (Fries and Paustenbach, 1990). For 2,3,7,8-TCDD, a NOAEL of 1 ng/kg-day in rats has been reported in both a three-generation reproduction study (Murray *et al.*, 1979) and a 2-year carcinogenicity study (Kociba *et al.*, 1978). On the basis of these studies, some regulatory agencies in Canada, The Netherlands, Germany, and the United States have developed threshold-based exposure guidelines of 1-10 pg/kg-day by applying safety factors ranging from 100 to 1000 to the NOAEL (Paustenbach *et al.*, 1992b).

In assigning a cancer potency factor (CPF) for 2,3,7,8-TCDD, some federal agencies such as the USEPA have relied on the use of nonthreshold dose-response models to develop a cancer potency value based on the Kociba *et al.* (1978) bioassay data. The use of these models is based on the assumption that there is no threshold for carcinogenesis, i.e., any dose poses some level of risk (Krewski *et al.*, 1989). Cancer potency estimates for 2,3,7,8-TCDD have been calculated by the USEPA (1985a), the CDC (Kimbrough *et al.*, 1984), the FDA (1983), and selected states. The differences in estimated potency are dependent on how one interprets the rodent bioassay data, the

and 9700 (mg/kg-day)¹ when hepatocellular carcinomas and adenomas were combined. When the data were not adjusted for survival rates, the estimates ranged from 1500 to 8,200 (mg/kg-day)⁻¹. These ranges are approximately 16- to 58-fold less than the USEPA's CPF of 156,000 (mg/kg-day)⁻¹ and equate to an acceptable daily intake (at a 1×10^{-6} risk level) of 100 to 370 fg/kg-day.

We developed a probability distribution for the CPF of 2,3,7,8-TCDD using a version of the LMS model which provides the necessary data as a subprogram (Crouch, 1992). The probability distribution is shown in Fig. 3. The 95th percentile value of 9700 (mg/kg-day)⁻¹ was used in point estimate (deterministic) approach.

EXPOSURE ASSESSMENT

In this assessment, the authors focused solely on the possible risks associated with residential exposure to soil or sediment. Residents were assumed to have no direct access to the former site since it is fenced. However, because samples from drainage ditches leading away from the site contained higher concentrations of PCDDs and PCDFs than those of soil samples collected in residential areas, a recreational scenario in which children played in the drainage ditch 5 hr per week was evaluated. The recreational scenario was considered extremely conservative due to the fact that one ditch is isolated by fencing, both ditches are currently undergoing remediation, and because children would not be expected to play in the ditch frequently.

The uptake of PCDD/PCDF due to soil ingestion and dermal contact by children 1.5 to 12 years old was estimated by summing the estimated uptake due to residential exposure (97% of the lifetime) with uptake due to the recreational exposure (3% of the lifetime). The estimate of uptake by residents was based on the results of soil sampling from residential yards and recreational exposure was based on the results of sediment sampling (Table 2). For adults, uptake due to soil ingestion and dermal contact was estimated based on the concentrations found in residential soil because it was assumed that adults did not have routine contact with the soil in the drainage

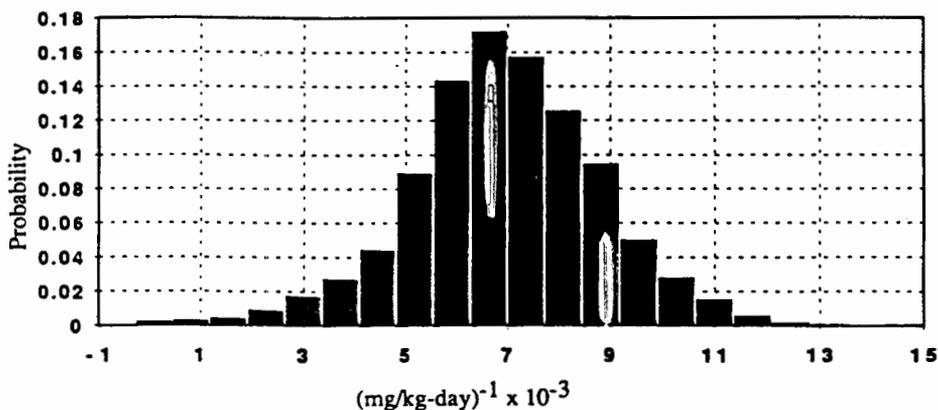


FIG. 3. Probability distribution for the cancer potency factor for 2,3,7,8-TCDD using the linearized multistage model, M-stage (Crouch, 1992) and histopathological reevaluation of cancer bioassay data (Keenan *et al.*, 1991).

TABLE 5
EXPOSURE PARAMETERS USED IN THE DETERMINISTIC AND PROBABILISTIC EVALUATIONS

| Exposure parameter | Point estimate value | Distribution value | Reference |
|--|----------------------|---------------------------|---------------------------|
| Body weight (kg) | | | |
| 0-1.5 years | 11.6 (normal) | Mean = 10 SD = 0.12 | Snyder, 1975 ^a |
| 1.5-5 years | 14.5 (normal) | Mean = 14 SD = 0.13 | USEPA, 1990 |
| 5-12 years | 30.5 (normal) | Mean = 26 SD = 0.75 | |
| 12-70 years | 70 (normal) | Mean = 62 SD = 0.30 | |
| Exposure duration (other than mother's milk) (years) | | | |
| 0-1.5 years | 1.5 | | Site-specific |
| 1.5-5 years | 3.5 | | |
| 5-12 years | 7 | | |
| 12-70 years | 58 | | |
| Height (cm) | | | |
| 1.5-5 years | N/A | 58.5-111.5 (uniform) | USEPA, 1985b |
| 5-12 years | N/A | 98.6-135 (uniform) | |
| 12-70 years | N/A | Mean = 162.4 SD = 9.76 | |

^a Reference for point estimate value(s), if different from reference for distribution value. N/A, not applicable. (The equation provided by Hawley (1985) was not used in the point estimate to calculate surface area.)

ditches. Uptake due to the inhalation of particulate pathways and ingestion of inhaled particulates was calculated for all age groups using residential exposure assumptions and air sampling data. Exposure due to the ingestion of mother's milk by nursing children was estimated for the 0- to 1-year age group.

In the Monte Carlo analysis, the simulations began at the time of the sampling, which is assumed to be the time of birth for the resident. The exposure factors used to estimate the lifetime average daily dose (LADD), for both the deterministic and probabilistic assessments, are presented in Tables 5-10.

Soil Ingestion

The incidental ingestion of soil by both adults and children was considered to be predominantly due to incidental hand-to-mouth contact as a result of activities such as gardening and normal outdoor play. The lifetime uptake due to ingestion of soil is a function of the chemical concentration in surface soil, the oral bioavailability of the contaminant in the soil matrix, the half-life of the contaminant in soil, the age-specific soil ingestion rate, and the frequency of contact with soil (USEPA, 1989; Paustenbach *et al.*, 1991b, 1992a).

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PCDD/PCDF Concentrations in Surface Soil

The PCDD and PCDF concentrations in soil and sediment are presented in Table 2 (Geraghty & Miller, 1990a; USEPA, 1991b). For the deterministic assessment, the concentration at the beginning of the exposure period was assumed to be represented by the 95th upper confidence limit (UCL) of the arithmetic mean of the soil (or sediment) data. Representative lifetime concentrations were estimated by assuming a chemical half-life in soil of 35 years, age-specific exposure periods, and first order degradation/loss kinetics.

For the probabilistic analysis, the individual soil or sediment sample results were considered as discrete data points (each had an equal probability of representing the exposure concentration) due to the relatively small sample size. This approach is consistent with that recommended by McKone and Bogen (1991). Ranges for environmental half-life were considered in the probabilistic assessment.

Oral Bioavailability

It has been suggested that the oral bioavailability of 2,3,7,8-TCDD is dependent upon the dose (Lucier *et al.*, 1986), as well as, soil-specific characteristics such as the organic content (Umbreit *et al.*, 1986; Shu *et al.*, 1988a). The results of numerous studies indicate that the oral bioavailability of 2,3,7,8-TCDD in animals ranges from 0.5 to 50% depending on these and other factors (Lucier *et al.*, 1986; Umbreit *et al.*, 1986; Shu *et al.*, 1988a; Birnbaum and Couture, 1988). In the deterministic assessment, a mean value of 43% was used to characterize the oral bioavailability of non-octa and octa-PCDDs and PCDFs (USEPA, 1993). For the probabilistic assessment, the range reported by Shu *et al.* (1988a) of 39 to 49% was used to describe the oral bioavailability of non-octa congeners. A uniform distribution was assigned since there were inadequate data to accurately assign a distribution.

Little information exists on the oral bioavailability of PCDDs and PCDFs other than 2,3,7,8-TCDD; however, results of a study conducted by Couture *et al.* (1988) suggest that the oral bioavailability of OCDD is less than that of 2,3,7,8-TCDD. Specifically, 1.5 to 10% of the OCDD dose was absorbed when rats were orally administered 500 or 5000 mg/kg OCDD in corn oil, whereas about 80% of 2,3,7,8-TCDD in corn oil was absorbed. By analogy to the data for 2,3,7,8-TCDD, we assumed that soil-bound OCDD was approximately 10-fold less bioavailable than 2,3,7,8-TCDD in soil. Since most assessments have used values of 25–50% for the oral bioavailability of 2,3,7,8-TCDD in soil, we chose a uniform distribution of 2.5 to 5% to represent the oral bioavailability of OCDD/OCDF in soil in humans.

Environmental Degradation of PCDDs and PCDFs in Soil

Because the source of contamination is no longer present at this site, soil concentrations throughout the total exposure period were corrected to account for the known degradation/loss of PCDD/PCDFs in surface soil over time. Degradation generally follows an exponential decay process described by the following equation:

$$N = N_0 e^{-kT} \quad (1)$$

This equation provides an estimate of the amount of chemical (N) remaining at time T given the initial concentration at time N_0 , and the first order decay constant, k . To calculate the *average* concentration C_{av} over the interval time T_1 to T_2 , the concentration must be summed over the time period of interest and divided by the total time, $T_2 - T_1$. Subsequent to integrating equation (1) and substituting C for N from $T_2 - T_1$, the following equation was used to estimate a representative chemical concentration for any time interval of interest:

$$C_{av} = -\frac{C_0}{(T_2 - T_1)k} (e^{-kT_2} - e^{-kT_1}). \quad (2)$$

Although a range of values has been reported for the environmental half-life of 2,3,7,8-TCDD in soil, it is clear that 2,3,7,8-TCDD is very stable. Variability in soil half-life has been attributed to differences in soil type, climatic factors, the depth of soil contamination, and the presence of cocontaminants (Cerlesi *et al.*, 1989; Yanders *et al.*, 1989, 1990; Paustenbach *et al.*, 1991a). The factors contributing to loss include photodegradation, volatilization, and leaching and surface runoff (Podoll *et al.*, 1986; Freeman and Schroy, 1989; Arthur and Frea, 1989; Yanders *et al.*, 1989, 1990). There is no good evidence for biologic degradation of 2,3,7,8-TCDD in soil. These factors have been considered by investigators who generally agree on a range of 9–35 years for the half-life of 2,3,7,8-TCDD in surface soils ($\frac{1}{4}$ to 3 inches) (di Domenico, 1990; Gough, 1991; Paustenbach *et al.*, 1992a). A half-life of 30–100 years is plausible for subsurface 2,3,7,8-TCDD (Paustenbach *et al.*, 1992a). Due to a paucity of environmental half-life data on congeners other than 2,3,7,8-TCDD, a uniform distribution of 9 to 35 years was used in the probabilistic assessment for all congeners. A half-life of 35 years was used for all congeners in the deterministic assessment.

Soil Ingestion Rate

Numerous investigators have attempted to estimate the soil ingestion rates for both children and adults (Kimbrough *et al.*, 1984; Hawley, 1985; Paustenbach, 1987; Davis *et al.*, 1990; Calabrese *et al.*, 1989, 1990; Van Wijnen *et al.*, 1990; USEPA, 1990; Paustenbach *et al.*, 1992a; Calabrese and Stanek, 1992). The more sophisticated studies use tracer elements to estimate soil ingestion rates. The results of the most thorough and most accurate investigation suggest that soil ingestion rates range from 9 to 40 mg/day for young children (Calabrese *et al.*, 1990). Later work by the same group, which used zirconium as a tracer, confirmed this range and concluded that the data were lognormally distributed with a geometric mean of 20.5 mg/day and a standard deviation of 87 mg/day (Calabrese and Stanek, 1991a,b). Accordingly, this range of values was used to characterize soil ingestion for ages 1.5 to 12 years. Estimates of soil ingestion rates for older children and adults range from 1 to 10 mg/day (Paustenbach *et al.*, 1992a; 1992b) and these were used as the soil ingestion rates for the 12- to 70-year age group. Consistent with the approach used in other assessments (Paustenbach *et al.*, 1991b), a uniform distribution was assigned to this age group. The soil ingestion rates of 200 and 50 mg/day estimated by the USEPA (1992b) for children and adults, respectively, were used in the deterministic assessment.

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Exposure Frequency

For the ingestion of soil it was assumed that children were exposed to contaminated sediment from the drainage ditch for 5 hr per week (3% of the total exposure period) and were exposed to soils in the residential area for 97% of the exposure duration.

Exposure Duration

The USEPA has estimated ranges for residence time based on data collected by the Bureau of the Census (USEPA, 1985a). Ranges for residential tenure were based on the number of years each person in the survey lived in their current household. For the probabilistic assessment, these ranges were normalized for age group and each range was assumed to have a uniform distribution. The deterministic assessment used standard regulatory-suggested default values (Table 5) for exposure duration (USEPA, 1989).

Body Weight

The range of body weights in the population is known to be normally distributed (Snyder, 1975). Percentile distributions for body weight for men and women, and male and female children, were developed by the USEPA (1985a) using data obtained in the National Health and Nutrition Examination Survey (NHANES) II. The age-specific body weight ranges presented in the study were used in the probabilistic assessment. The mean body weight for each age group was used in the deterministic assessment.

Estimation of Uptake Due to Ingestion of Soil

The equation used to estimate PCDD/PCDF uptake due to incidental ingestion of soil was

$$\text{LADD} = (C_{\text{soil}} \times \text{IR} \times \text{B} \times \text{EF} \times \text{ED}) / (\text{BW} \times \text{AT}), \quad (3)$$

where

LADD = Lifetime average daily dose (mg/kg-day)

C_{soil} = Concentration of PCDD/PCDF in surface soil (mg/kg 2,3,7,8-TCDD TEQ)

IR = Soil ingestion rate (kg soil/day)

B = Bioavailability due to soil matrix binding (fraction)

EF = Exposure frequency (fraction of year)

ED = Exposure duration (years)

BW = Body weight (kg)

AT = Averaging time (years).

The values used to estimate soil ingestion are presented in Table 6. The probability distribution for uptake via soil ingestion is presented in Fig. 4. The 50th and 95th percentile LADDs for soil ingestion are 6.6×10^{-14} and 7.6×10^{-13} mg/kg-day (TEQ), respectively. The estimate for uptake via soil ingestion was 3.6×10^{-10} mg/kg-day

using the point estimate approach, which is 473-fold greater than the 95th percentile dose predicted by the probabilistic assessment. Octa congeners contributed 6% (50th percentile) and 10% (95th percentile) to the total soil ingestion dose.

Uptake of Inhaled Particulates

To estimate the uptake of PCDD/PCDFs via inhalation, the following parameters must be accounted for: the contaminant concentration in air (as particulate), inhalation rate, fraction of inspired particles retained in lung, and particulate bioavailability in the lung.

Contaminant Concentration in Air

Average values for three consecutive days of sampling, at the five air monitoring stations, were used to estimate the airborne concentrations of PCDD/PCDF (Table 3). For the 180-day period of site remediation, the average concentrations of airborne particulates at each monitoring station were within local background ranges and therefore were not attributed to dispersion of site soils. The concentrations of PCDD/PCDF in particulates were considered representative for both the residential and recreational scenarios and were given equal probability (e.g., treated as discrete values) in the probabilistic analysis. The highest of the five values was used in the deterministic assessment.

Respirable Particle Fraction

The deposition of inhaled particulates in the lung is dependent on the particle size. Because site-specific data on particle size distribution were not available, the mass mean aerodynamic diameter (MMAD) was conservatively assumed to be 2.4 μm . This is regarded as a conservative assumption as generally 70% of airborne particulates (by weight) are larger than 2.4 μm in diameter (Trijonis *et al.*, 1980). The ICRP (1966) derived a model relating the particulate size distribution and breathing rates to the deposition pattern. Based on results from this model, a value of 20% of inspired particulates was assumed to be available for pulmonary absorption (ICRP, 1966). An upperbound value of 33% was used in the deterministic assessment.

Inhalation Rate

Total ventilation or minute volume, V_E (liters/min), has been correlated with the three-fourths power of body weight (kg) for purposes of pharmacokinetics modeling (USEPA, 1988). For humans, the relationship is

$$V_E = 0.302 \times (\text{BW})^{0.75}. \quad (4)$$

This equation was applied to the age-specific body weight distributions described above to characterize age-specific ventilation rates in the probabilistic assessment. For the deterministic assessment, standard inhalation rates were used (Snyder, 1975). The values for the probabilistic and deterministic assessments are presented in Table 7.

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Inhalation Bioavailability

Some information regarding the pulmonary absorption of 2,3,7,8-TCDD and other PCDD/PCDFs is available (Nessel *et al.*, 1990). Based on these data and the characteristics of soil particles, an inhalation bioavailability of 100% was used (Paustenbach *et al.*, 1992b) in both analyses.

Exposure Frequency

It was assumed that both children and adults were exposed to airborne dust for 365 days per year (fraction of year exposed = 1). This is an unreasonably conservative approach since most eastern and southeastern states have about 200 days per year of either frozen soil or precipitation, which effectively eliminates dust resuspension from soil.

The values for exposure frequency, exposure duration, and body weight used to calculate the dose via soil ingestion were also used to estimate uptake via inhalation.

Estimating Uptake via Inhalation

The equation used to estimate uptake via inhalation of particulates was:

$$\text{LADD} = (C_{\text{air}} \times \text{IR} \times \text{FR} \times \text{EF} \times \text{ED}) / (\text{BW} \times \text{AT}), \quad (5)$$

where

LADD = Lifetime average daily dose of PCDD/PCDF (mg/kg-day)

C_{air} = Concentration of PCDD/PCDF in air (mg/m³)

IR = Inhalation rate (m³/day)

FR = Fraction of inspired particulate absorbed in lower respiratory tract

EF = Exposure frequency (fraction of year)

ED = Exposure duration (years)

BW = Body weight (kg).

The values used to estimate uptake via inhalation are presented in Table 7. The probability distribution for the uptake via inhalation is presented in Fig. 4. The 50th and 95th percentile LADDs are 3.8×10^{-13} and 1.1×10^{-12} mg/kg-day, respectively. The octa congeners represented 25% of the total dose at the 50th and 95th percentiles. The uptake of PCDD/PCDF using the deterministic approach was 1.3×10^{-11} mg/kg-day (TEQ), about 13-fold greater than the 95th percentile dose.

Ingestion of Inhaled Particulates

The fraction of respirable particles that is swallowed should be considered when assessing total uptake of PCDDs/PCDFs (Leung *et al.*, 1990; Sheehan *et al.*, 1991). The dose due to ingestion of inhaled particulates results from the deposition of inhaled particulates in upper regions of the respiratory tract. Approximately one-half of the 8% of particulates deposited into the tracheobronchial region are rapidly cleared by ciliary mucous transport, leaving 4% available for systemic absorption via ingestion

(Paustenbach *et al.*, 1986; Sheehan *et al.*, 1991). Furthermore, approximately 42 to 67% of inhaled particulates are deposited in the nasopharynx and consequently swallowed. Thus, on the average, 58% of inhaled particulate matter reaches the gastrointestinal tract (ICRP, 1966). Accordingly, this value was used in the probabilistic assessment. An upperbound value of 67% was used in the deterministic assessment.

The dose from ingestion of inhaled particulates was calculated using the equation

$$\text{LADD} = (C_{\text{air}} \times \text{IR} \times \text{FI} \times \text{B} \times \text{EF} \times \text{ED}) / (\text{BW} \times \text{AT}), \quad (6)$$

where

LADD = Lifetime average daily dose (mg/kg-day)

C_{air} = Concentration of PCDD/PCDF in air (mg/m³)

IR = Inhalation rate (m³/day)

FI = Fraction of inhaled particles ultimately ingested

B = Oral bioavailability due to soil matrix binding (fraction)

EF = Exposure frequency (fraction of year)

ED = Exposure duration (years)

BW = Body weight (kg)

AT = Averaging time (years).

PDF data and point estimate values are presented in Table 8. The probability distribution for dose via ingestion of inhaled particles is presented in Fig. 4. The 50th and 95th percentile LADDs due to ingestion of large airborne particles are 3.3×10^{-13} and 1.3×10^{-12} mg/kg/day (TEQ), respectively. Octa congeners represented 3% of the total dose at both the 50th and 95th percentiles. The LADD predicted by the deterministic assessment was 1.3×10^{-11} mg/kg-day, approximately 9-fold greater than the estimated 95th percentile dose.

Dermal Uptake

Factors influencing the amount of soil-bound chemical that may be dermally absorbed include chemical concentration in surface soil, skin surface area available for contact, skin adherence properties of soil, and dermal bioavailability (USEPA, 1992b; Paustenbach and Leung, 1992).

Chemical Concentration in Surface Soil

The chemical concentrations in soil are discussed in the section on soil ingestion and are presented in Table 2.

Skin Surface Area

Age-specific ranges for exposed skin surface area were developed based on the relationship of body surface area to height and weight (Hawley, 1985):

$$\text{SA} = 3.73 \times \text{height}^{0.417} \times \text{weight}^{0.517}. \quad (7)$$

Age-specific ranges for exposed body surface area in the probabilistic assessment were based on Eq. (4) (Hawley, 1985). In developing the distributions for exposed body

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surface area, body weight was described by a normal distribution and height was described by a uniform distribution (USEPA, 1985a). For the deterministic assessment, values recommended by USEPA (1991a, 1992a) were used (Table 9).

Skin Adherence Factor

Several studies suggest that the amount of soil which can adhere to the skin is between 0.2 and 1.0 mg/cm² (Lepow *et al.*, 1975; Roels *et al.*, 1980; Que Hee *et al.*, 1985; Driver *et al.*, 1989; Paustenbach and Leung, 1992). The range of 0.2 and 1.0 mg/cm², suggested by USEPA (1992b), was used to estimate the degree of soil adherence. This factor was assumed to have a uniform distribution (Paustenbach *et al.*, 1991c). The upperbound value of 1 mg/cm² value suggested by the USEPA (1992b) was used in the deterministic assessment.

Dermal Bioavailability

A range of 0.1 to 2% for the dermal bioavailability of PCDD/PCDF in soil was used in the probabilistic assessment, based on data from rat studies (Poiger and Schlatter, 1980; Shu *et al.*, 1988b) and information regarding the relative permeabilities of rat and human skin (Bartek *et al.*, 1972; Bartek and La Budde, 1975; Shu *et al.*, 1988b; Banks and Birnbaum, 1991). Because data regarding the distribution frequency for this parameter were not available, a uniform distribution was used in the Monte Carlo analysis. The upperbound value of 2% was used in the deterministic assessment.

Recently, a fugacity model was offered to estimate the dermal uptake of organic chemical contaminants from a soil matrix (McKone, 1990). The model considers a number of factors not considered previously in estimating dermal uptake of a chemical from a soil matrix. These factors include the mass transfer of chemical from soil to air, soil to skin, and skin to systemic circulation. McKone tested the model by predicting an uptake of soil-bound 2,3,7,8-TCDD from rat skin, using the laboratory conditions of experiments conducted by Poiger and Schlatter (1980) and Shu *et al.* (1988b). The results of McKone's model predicted an uptake of 0.5%, which is comparable to the value of 1% suggested in the two laboratory studies.

Estimating Dermal Uptake

The values for exposure duration, exposure frequency, body weight, and averaging time used to estimate uptake via soil ingestion (USEPA, 1985a, 1990a) were used to estimate dermal uptake. The equation used to estimate dermal uptake was

$$\text{LADD} = (C_{\text{soil}} \times \text{SAF} \times \text{SA} \times \text{B} \times \text{EF} \times \text{ED}) / (\text{BW} \times \text{AT}), \quad (8)$$

where

LADD = Lifetime average daily dose (mg/kg-day)

C_{soil} = Concentration of PCDD/PCDFs in soil (mg/kg)

SAF = Skin adherence factor (kg/cm²/day)

SA = Surface area exposed (cm²)

B = Bioavailability due to soil matrix binding (fraction)

TABLE 6

EXPOSURE PARAMETERS USED IN THE DETERMINISTIC AND PROBABILISTIC ASSESSMENTS TO ESTIMATE THE UPTAKE OF PCDD/PCDF DUE TO SOIL INGESTION

| | Point estimate | Distribution (mg/kg TEQ) | Reference |
|---|------------------------|---|--|
| Concentration of PCDD/PCDF in soil (sediment) (ng/kg TEQ) | | | |
| 1.5-5 years | 648 | Discrete, $n = 6$ | Geraghty & Miller, 1990a; USEPA, 1990a |
| 5-12 years | 581 (868) ^a | Discrete, $n = 6$ ($n = 17$) ^a | |
| 12-70 years | 6.41 | Discrete, $n = 6$ | |
| Soil ingestion rate (mg/day) | | | |
| 1.5-12 years | 200 | Mean $\bar{x} = 20.5$, SD = 87 (truncated lognormal) | USEPA, 1989 ^b ; Calabrese and Stanek, 1991a,b |
| 12-70 years | 100 | 1-10 (uniform) | Paustenbach 1991b, 1992a,b; Calabrese and Stanek, 1991a,b |
| Bioavailability (percentage) | | | |
| Non-octa congeners | 43 | 39-49 (uniform) | Shu <i>et al.</i> , 1988a |
| Octa congeners | 43 | 2.5-5 (uniform) | Birnbaum and Couture, 1988 |
| Soil half-life | 35 years | 9-35 years | Gough, 1991; di Domenico, 1990; Paustenbach <i>et al.</i> , 1991a,b |
| Exposure frequency (fraction of time exposed) | | | |
| 1.5-5 years | 1 | 1 | Site-specific |
| 5-12 years | 0.97 (0.03) | 0.97 (0.03) | |
| 12-70 years | 1 | 1 | |

^a Point estimate value for representative concentration calculated using a soil half-life of 35 years, the upper 95th UCL of the arithmetic mean concentration of sample results, and first order decay. Distribution based on soil half-life range of 9-35 years. Values in parentheses represent values for sediment.

^b Reference for point estimate value(s), if different from reference for distribution value.

EF = Exposure frequency (fraction of year)

ED = Exposure duration (years)

BW = Body weight (kg)

AT = Averaging time (years).

The parameter values used in both assessments to estimate dermal uptake are presented in Table 9. The probability distribution for the LADD via dermal uptake is presented in Fig. 4. The 50th and 95th percentile LADDs are 1.4×10^{-12} and 1.6×10^{-11} mg/kg-day (TEQ), respectively. Octa congeners represented 49 and 60% of the risk, at the 50th and 95th percentiles, respectively. The deterministic assessment predicted a LADD of 3.2×10^{-10} mg/kg-day, a value approximately 20-fold greater than the 95th percentile dose.

TABLE 7

EXPOSURE PARAMETERS USED IN THE DETERMINISTIC AND PROBABILISTIC ASSESSMENTS TO ESTIMATE THE UPTAKE OF PCDD/PCDF DUE TO INHALATION OF PARTICULATES

| Parameter | Point estimate value | Distribution value | Reference |
|--|--------------------------|---|--|
| Concentration of PCDD/PCDF in air | 119.91 fg/m ³ | Discrete ($n = 5$) (see Table 3) | Geraghty & Miller, 1990b |
| Inhalation rate (m ³ /day) | | | |
| 0-1.5 years | 2.3 | Dependent upon body weight ^b | Snyder, 1975 ^a |
| 1.5-5 years | 6.6 | Dependent upon body weight ^b | USEPA, 1988 |
| 5-12 years | 12.2 | Dependent upon body weight ^b | |
| 12-70 years | 23 | Dependent upon body weight ^b | |
| Fraction of inspired particulates retained in lung | 0.33 | 0.20 (point estimate) | ICRP, 1966 Trijonis <i>et al.</i> , 1980 ^a |
| Inhalation bioavailability (percentage) | 100 | 100 (point estimate) | ATSDR, 1988 |
| Exposure frequency (fraction of time exposed) | 1.0 | 1.0 | Site-specific |

^a Reference for point estimate value(s), if different from reference for distribution value.

^b I_E (liters/min) = $0.302 \times BW$ (kg)^{0.75}.

Ingestion of Mother's Milk

Environmentally persistent lipophilic chemicals such as PCDDs and PCDFs are often detectable in mother's milk. Fuerst *et al.* (1986) identified PCDDs and PCDFs in milk from European nursing mothers. Only 2,3,7,8-substituted congeners were detected in the milk which is consistent with data obtained when other tissue compartments were analyzed (Rappe *et al.*, 1983; Ryan *et al.*, 1985; Schecter *et al.*, 1985). OCDD (143 ppt in fat) and 1,2,3,4,6,7,8-HpCDD (49 ppt in fat) were the predominant congeners detected in the study by Fuerst *et al.* (1986). Chemical uptake via ingestion of mother's milk is dependent upon the mother's average daily dose, whole body half-life of the chemical, milk ingestion rate, and exposure duration.

Mother's Average Daily Dose

The probability distribution for the mother's average daily dose developed from the Monte Carlo simulation was based on the sum of the doses estimated for all other exposure pathways.

TABLE 8

EXPOSURE PARAMETERS USED IN THE DETERMINISTIC AND PROBABILISTIC ASSESSMENTS TO ESTIMATE THE UPTAKE OF PCDD/PCDF DUE TO INGESTION OF INHALED PARTICULATES

| Parameter | Point estimate value | Distribution value | Reference |
|---|----------------------|---|---|
| Concentration of PCDD/PCDF in air (fg/m ³ TEQ) | 119.91 | Discrete (n = 5) (see Table 3) | Geraghty & Miller, 1990b |
| Inhalation rate (m ³ /day) | | | |
| 0-1.5 years | 2.3 | Dependent upon body weight ^b | Snyder, 1975 ^a ; USEPA, 1988 |
| 1.5-5 years | 6.6 | Dependent upon body weight | |
| 5-12 years | 12.2 | Dependent upon body weight | |
| 12-70 years | 23 | Dependent upon body weight | |
| Fraction of inspired particulates ingested | 0.67 | 0.58 (point estimate) | Trijonis <i>et al.</i> , 1990 ^a ; ICRP, 1966 |
| Oral bioavailability (percentage) | | | |
| Non-octa congeners | 43 | 39-49 | Shu <i>et al.</i> , 1988a |
| Octa congeners | 43 | 2.5-5 | Birnbaum and Couture, 1988 |
| Exposure frequency (fraction of time exposed) | 1.0 | 1.0 (point estimate) | Site-specific |

^a Reference for point estimate value(s), if different from reference for distribution value.

^b $I_E = 0.302 \times (BW)^{0.75}$.

Whole Body Half-Life

The half-life in the human body of 2,3,7,8-TCDD has been extensively studied. Poiger and Schlatter (1986) investigated the pharmacokinetics of radiolabeled 2,3,7,8-TCDD ingested by a 92-kg male. The estimated half-life of 2,3,7,8-TCDD, for this individual, assuming first-order kinetics, was 5.8 years. Pirkle *et al.* (1989) estimated the half-life of 2,3,7,8-TCDD in the human body based on serum data from 36 Air Force Vietnam veterans exposed to the herbicide Agent Orange during spraying operations. First-order kinetics and serum concentrations for 1982 and 1987 were used to estimate the half-life for each veteran. A frequency plot of the results is shown in Fig. 5. The median half-life for this population was found to be 7.1 years (95% confidence interval about the median of 5.8 to 9.6 years). A later study of the same population found a marginally significant dependence of half-life on percentage body fat (Michalek *et al.*, 1992); however, the specific mathematical correlation has not yet been characterized.

Because the Pirkle *et al.* (1989) half-life estimates are based on the largest chronically exposed population and have been used by other investigators (Mocarelli *et al.*, 1991; Michalek *et al.*, 1992), their results were used in our probabilistic assessment. The

TABLE 9

EXPOSURE PARAMETERS USED IN THE DETERMINISTIC AND PROBABILISTIC ASSESSMENTS TO ESTIMATE THE UPTAKE OF PCDD/PCDF TO DERMAL ABSORPTION

| | Point estimate value soil (sediment) | Distribution value soil (sediment) | Reference |
|---|--------------------------------------|--|---|
| Concentration of PCDD/PCDF in soil (sediment) (ng/kg TEQ) | (See Table 6) | Discrete $n = 6$ ($n = 17$) (see Table 2) | Geraghty & Miller, 1990; USEPA, 1991b |
| Skin surface area (cm ²) | | | |
| 1.5-5 years | 3.601 | Dependent upon body weight and height ^b | USEPA, 1985b; Hawley, 1985 |
| 5-12 years | 4.258 | Dependent upon body weight and height ^b | USEPA, 1991a; 1992a ^a |
| 12-70 years | 5.800 | Dependent upon body weight and height ^b | |
| Soil to skin adherence (mg/cm ²) | 1.0 | 0.2-1 mg/cm ² (uniform) | USEPA, 1992b |
| Dermal bioavailability (percentage) | 2 | 0.1-2 (uniform) | Shu et al., 1988b Banks and Birnbaum, 1991 |
| Exposure frequency (fraction of time exposed) | | | |
| 1.5-5 years | 0.285 | 0.285 | Site-specific |
| 5-12 years | 0.285 | 0.285 | Site-specific |
| 12-70 years | 0.285 | 0.285 | Site-specific |

^a Reference for point estimate values, if different from reference for distribution value.

^b SA (cm²) = 3.73 × height^{0.417} (cm) × height^{0.517} (kg).

data were entered as a continuous distribution with probabilities equal to the frequencies presented in Fig. 4. We assumed that the biologic half-lives (and the distributions) for all PCDDs and PCDFs are adequately represented by the values determined for 2,3,7,8-TCDD. A value of 7.1 years was used as the point estimate. Recent data suggest that the half-life in humans is closer to 12 years (Michalek *et al.*, 1992). This could easily be incorporated in our analysis but would not appreciably alter our results.

Mother's Milk Ingestion Rate

An infant breast milk ingestion rate of 600 to 900 gm/day (Butte *et al.*, 1984, 1991) has been accepted for use in risk assessment (CAPCOA, 1992). This range is in agreement with observations by others (Lonnerdal *et al.*, 1976; Whitehead and Paul, 1981). Recently, Dewey *et al.* (1991) reported breast milk intake rates that are correlated with infant body weight. An analysis of these data indicates that the total grams of

TABLE 10
EXPOSURE PARAMETERS USED IN THE DETERMINISTIC AND PROBABILISTIC ASSESSMENTS TO ESTIMATE UPTAKE OF PCDD/PCDF VIA MOTHER'S MILK

| Exposure parameter | Point estimate value | Distribution value | Reference |
|--|----------------------|---|---|
| Percent of mother's weight which is fat | 33 | 33 (point estimate used) | Butte <i>et al.</i> , 1984, 1991 |
| Percent of fat in mother's milk | 4 | 4 (point estimate used) | Butte <i>et al.</i> , 1984, 1991 |
| Percent of 2,3,7,8-TCDD that partitions to mother's milk | 9 | 9 (point estimate used) | Smith, 1987 |
| Whole body half-life of 2,4,7,8-TCDD in mother (days) | 7.1 | Discrete, see Fig. 5 | Pirkle <i>et al.</i> , 1989 |
| Mother's milk ingestion rate (g/day) | 900 | Dependent upon body weight (point estimate used) ^b | Butte <i>et al.</i> , 1991 ^a Dewey <i>et al.</i> 1991 |
| Exposure frequency | 365 | 365 (uniform) | Butte <i>et al.</i> , 1984 |
| Exposure duration (years) | 1 | 1 (point estimate value) | La Leche, 1991 |
| Infant body weight (kg) | 6.5 | 6.5 (point estimate used) | USEPA, 1990 |

^a Reference for point estimate value(s), if different from reference for distribution value.

^b Specific relationships taken from the DARLING study (Dewey *et al.*, 1991).

breast milk ingested per kilogram body weight per day are significantly different at 3, 6, 9, and 12 months of age. Accordingly, we divided the breast-feeding period of 1 year into four age-specific dose groups, for which we utilized the age-specific relationship between body weight and milk ingestion rate. The national percentage of breast-fed infants ranges from 21 to 25% (MacGowan *et al.*, 1991; Maxwell and Burmaster, 1992). Consistent with Maxwell and Burmaster (1993), we used a value of 22% in the analysis. For the deterministic assessment, a daily ingestion rate of 900 g/day was used.

Exposure Duration (Infant)

The length of time that infants breast-feed varies significantly from family to family. Although breast-feeding periods range from zero to well over 1 year, the most common total time period is estimated from 6 months to 1 year (La Leche, 1991). Accordingly, a range of 6 months to 1 year, with a uniform distribution, was assumed for the length of time an infant breast-feeds. The upperbound value of 1 year was used in the deterministic assessment.

Estimating Intake via Mother's Milk Ingestion

The equation for estimating the dose received by an infant is presented below. Uptake via ingestion of mother's milk is a function of the average chemical concen-

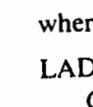
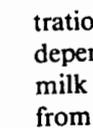
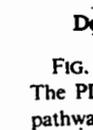


FIG. 4
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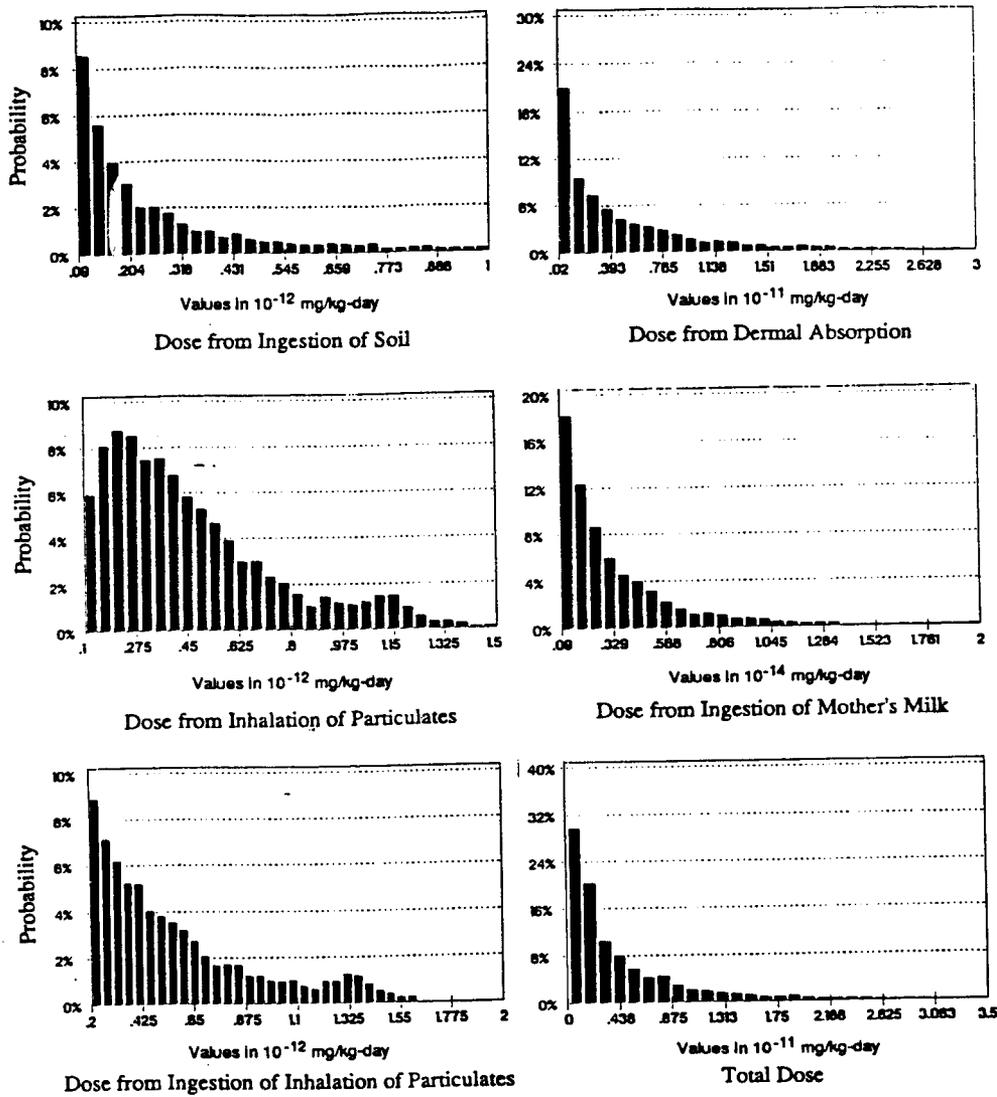


FIG. 4. Probability distributions for the predicted uptake (mg/kg-day) of PCDD/PCDF (2,3,7,8-TEQ). The PDF for total dose was used to characterize incremental lifetime cancer risk (ILCR) for exposure pathways evaluated in the Monte Carlo analysis.

tration in the milk and the amount of milk ingested. The concentration in milk is dependent upon the mother's average daily dose (ADD), percentage body fat, and milk fat. The equation used to determine the LADD via mother's milk is adapted from CAPCOA (1992) and Smith (1987)

$$LADD_{\text{infant}} \text{ (mg/kg-day)} = (C_m \times MI \times EF \times ED) / BW \times AT. \quad (9)$$

where

LADD = Lifetime average daily dose (mg/kg-day)

C_m = Concentration of contaminant in mother's milk (mg/kg)

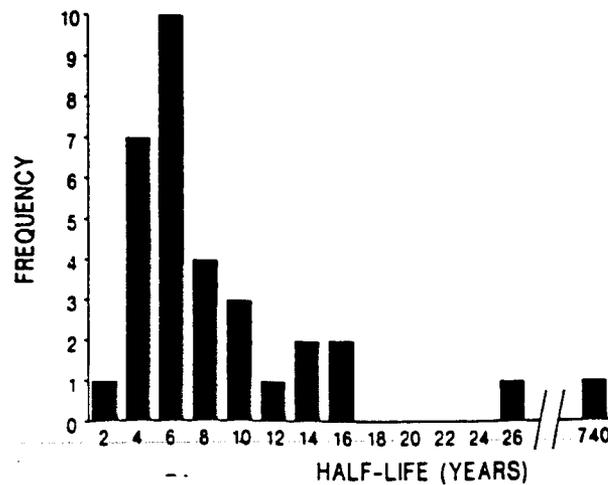


FIG. 5. Half-life of 2,3,7,8-TCDD based on serum concentrations in 36 Air Force veterans exposed to Agent Orange (Pirkle *et al.*, 1989).

- MI = Daily mother's milk ingestion rate (kg/day)
- EF = Exposure frequency (days per year)
- ED = Exposure duration (years)
- BW = Average body weight of child (kg)
- AT = Averaging time (days).

The concentration in mother's milk is determined as

$$C_m \text{ (mg/kg)} = (\text{ADD}_m \times t_{1/2} \times f_1 \times f_3) / (f_2 \times 0.693), \quad (10)$$

where

ADD_m = Mother's average daily dose (mg/kg-day)

$t_{1/2}$ = Whole body half-life of contaminant in mother (days)

f_1 = Fraction of contaminant that partitions to mother's fat

f_2 = Percentage mother's weight that is fat

f_3 = Percentage fat of mother's milk

0.693 = Natural log of 2 (converts $t_{1/2}$ to decay rate constant k , assuming first-order kinetics).

The values used to estimate the LADD via mother's milk ingestion are presented in Table 10. The 50th and 95th percentile LADDs are 1.8×10^{-15} and 1.6×10^{-14} mg/kg-day (TEQ), respectively. The contribution of octa congeners to the dose via mother's milk was 50% and 52% at the 50th and 95th percentiles, respectively. The probability distribution for uptake via mother's milk is presented in Fig. 4. The RME value predicted in the deterministic assessment was 3.7×10^{-11} mg/kg-day, over three orders of magnitude greater than the 95th percentile dose.

RISK CHARACTERIZATION

Quantitative risk assessments are often limited by inadequate characterization of the uncertainty in the risk estimates. As recently discussed by USEPA, risk assessments

based on only a point estimate of risk "do not fully convey the range of information considered and used in developing the assessment" (USEPA, 1991b; Graham *et al.*, 1992). The probabilistic approach to the characterization of health risk provides the risk manager with a more complete perspective on the potential variability in the risk estimate and can also identify factors contributing most significantly to variance in the risk results (Finkel, 1990; McKone and Bogen, 1991). Furthermore, the characterization of risk as a probability distribution provides the risk manager with a tool for more appropriately evaluating (e.g., based on mathematical probability) the risk for the most highly exposed as well as the typical individual.

Carcinogenic Risk

The plausible incremental lifetime cancer risk (ILCR) was estimated as the product of lifetime average daily dose (LADD) (mg/kg-day) and cancer potency (mg/kg-day)⁻¹. Results of the Monte Carlo analysis are presented in Fig. 6. Figure 6a presents the log₁₀ ILCR versus the relative probability for the final range of risks associated with residential exposure to PCDDs and PCDFs. Figure 6b presents the log₁₀ ILCR versus the cumulative probability for the predicted risk range. The cumulative probability is the sum of probabilities of all risk values less than and equal to the risk value being read. Presentation of risk as a cumulative function facilitates the interpretation of specific percentile risks.

The results of the probabilistic assessment predicted a 95th percentile ILCR of 1.2×10^{-7} . This value is considered a conservative estimate for a reasonable maximum exposure. The predicted risk at the 50th percentile, representative of the most likely exposure, was 1.8×10^{-8} . Comparatively, the risk calculated using the RME approach, based on the upperbound value for the revised CPF for 2,3,7,8-TCDD of 9700 (mg/kg-day)⁻¹ and standard upperbound values for the exposure parameters, was 8.3×10^{-6} . In this case, the predicted RME cancer risk exceeds the 95th percentile value predicted by the probabilistic assessment by approximately 70-fold.

One of the goals of this assessment was to understand the contribution of octa congeners to the total PCDD/PCDF risk. Interestingly, octa congeners contributed

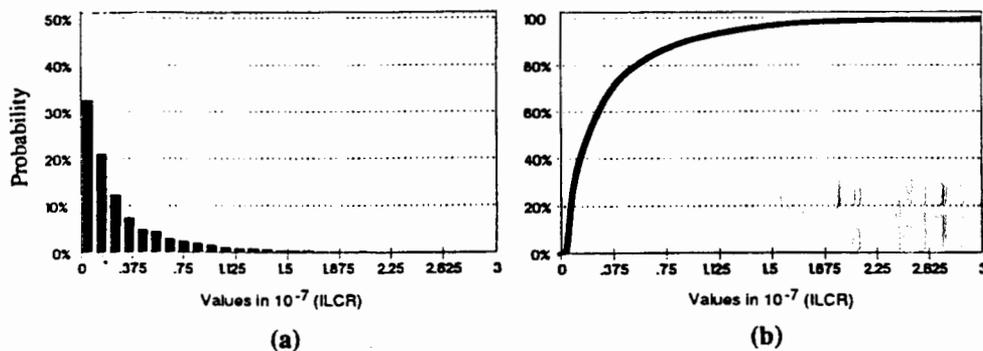


FIG. 6. The probability distribution (a) and cumulative distribution (b) for incremental lifetime cancer risk (ILCR) from exposure to PCDDs and PCDFs based on uncertainty in the cancer potency and key exposure factors.

significantly to the total risk, with a 23% contribution at the 50th percentile risk and a 53% contribution at the 95th percentile risk. These contributions reflect the comparatively high soil and sediment concentrations of OCDD and OCDF, relative to other 2,3,7,8-congeners detected.

As shown in Fig. 7, the relative contribution of the pathway-specific risks to total risk is dependent on the subset of congeners assessed. When all congeners were considered, dermal contact represented 65% of the total at the 50th risk percentile and 84% of the total at the 95th risk percentile. Inhalation of particulates represented 17% of the risk at the 50th risk percentile and 6% of the risk at the 95th risk percentile. Ingestion of inhaled particulates represented 15 and 6% of the risk at 50th and 95th percentiles, respectively.

The variability of the relative contribution to risk is a consequence of the unique shapes of the individual input parameter distributions. This phenomenon needs to be recognized by risk managers when identifying those chemicals and exposure pathways which warrant the bulk of remediation and cost.

Comparison to Background Exposure

Background exposure to PCDD/PCDFs has been estimated to be approximately 0.03 to 2.86 pg/kg day based on mean human adipose tissue concentrations of 6 to 12 ppt (Kang *et al.*, 1991; Leung *et al.*, 1990). Travis and Hester (1990) have estimated the uptake due to ingestion of background levels of PCDDs and PCDFs in meat and dairy products to be 1.3 pg/kg-day (TEQ). This value falls within the ranges predicted by Kang *et al.* (1991) and Leung *et al.* (1990). Using a cancer potency of $9700 \text{ (mg/kg-day)}^{-1}$, this background dose is associated with a plausible cancer risk of 1.26×10^{-5} . These data indicate that the typical uptake of PCDD/PCDF associated with ingestion of food products is over two orders of magnitude greater than the highest plausible uptake of PCDD/PCDF, as predicted by the probabilistic assessment.

CONCLUSION

In summary, our assessment indicates the following:

- At least 95% of the persons potentially exposed to the PCDD/PCDF at this site do not have an incremental cancer risk greater than 1.2×10^{-7} . The most likely risk estimate, represented by the 50th percentile risk, should be no greater than 1.8×10^{-8} .
- Octa congeners contributed significantly to the total ILCR. At the 50th and 95th percentiles, octa congeners represented 23 and 53% of the risk, respectively. In the deterministic assessment, octa congeners represented approximately 57% of the plausible cancer risk.
- The results of the deterministic assessment predicted a risk of 8.3×10^{-6} . This value is approximately 70-fold greater than the 95th percentile risk predicted in the probabilistic assessment and exceeds even the 99.9th percentile risk estimate.
- The 95th percentile risk for persons who live near the site is two orders of magnitude lower than the U.S. background cancer risk from the dietary intake of PCDD/PCDFs.

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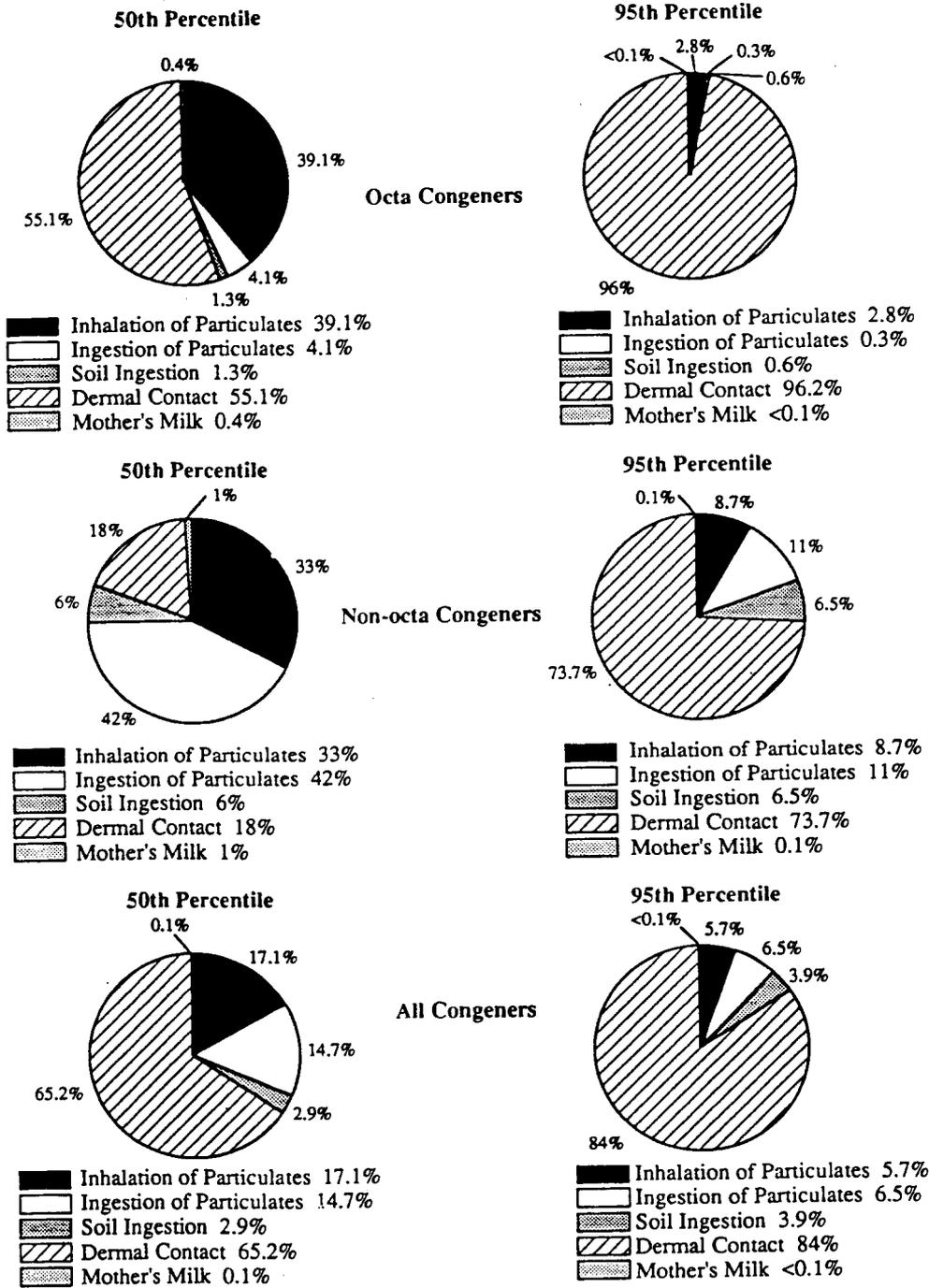


FIG. 7. Comparisons of the relative contributions of individual exposure pathways to incremental lifetime cancer risk (ICLR).

REFERENCES

- Agency for Toxic Substances and Disease Registry (ATSDR) (1988). *Toxicological Profile for 2,3,7,8-Tetrachlorodibenzo-p-dioxin* (M. W. NEAL AND D. K. BASU, eds.) Contract No. 68-03-3228, Task 53. Center for Chemical Hazard Assessment, Syracuse Research Corporation, Syracuse, New York.
- ARTHUR, M. A., AND FREA, J. I. (1989). 2,3,7,8-Tetrachlorodibenzo-p-dioxin: Aspects of its important properties and its potential biodegradation in soils. *J. Environ. Qual.* **18**, 1-11.
- @Risk (1990). *A Computer Program for Conducting Monte Carlo Analyses*. Cambridge, MA.
- BANKS, Y. B., AND BIRNBAUM, L. S. (1991). Absorption of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) after low dose dermal exposure. *Toxicol. Appl. Pharmacol.* **107**, 302-310.
- BARTEK, M. J., AND LA BUDDE, J. A. (1975). Percutaneous absorption in vivo. In *Animal Models in Dermatology*, pp. 103-120. Churchill Livingstone, New York.
- BARTEK, M. J., LA BUDDE, J., AND MAIBACH, H. I. (1972). Skin permeability in vivo: Comparison in rat, rabbit, pig and man. *J. Dermatol. Invest.* **58**, 114-124.
- BELLIN, J. S., AND BARNES, D. G. (1989). 1989 Update to the Interim Procedures for Estimating Risks Associated with Exposures to Mixtures of Chlorinated Dibenzo-p-dioxins and Dibenzofurans (CDDs and CDFs). EPA/625/3-89/016. U.S. Environmental Protection Agency, Washington, DC.
- BIRNBAUM, L. S., AND COUTURE, L. A. (1988). Disposition of octachlorodibenzo-p-dioxin (OCDD) in male rats. *Toxicol. Appl. Pharmacol.* **93**, 22-30.
- BIRNBAUM, L. S., HARRIS, M. W., CRAWFORD, D. D., AND MORRISSEY, R. E. (1987). Teratogenic effects of polychlorinated dibenzofurans in combination in C57BL/6N mice. *Toxicol. Appl. Pharmacol.* **91**, 246-255.
- BOND, G. G., BODNER, K. M., AND COOK, R. R. (1989). Phenoxy herbicides and cancer: Insufficient epidemiological evidence for a causal relationship. *Fundam. Appl. Toxicol.* **12**, 172-188.
- BOND, G. G., OTT, M. G., BRENNER, F. E., AND COOK, R. R. (1983). Medical and morbidity surveillance findings among employees potentially exposed to TCDD. *Br. J. Ind. Med.* **40**, 318-324.
- BOND, G. G., WETTERSTROM, N. H., AND ROUSE, G. J. (1988). Cause-specific mortality among employees engaged in the manufacture, formulating or packaging of 2,4-dichlorophenoxyacetic acid and related salts. *Br. J. Ind. Med.* **45**, 98-105.
- BRESLIN, P., KANG, H. K., LEE, Y., BURT, V., AND SHEPARD, B. M. (1987). *Proportionate Mortality of Army and Marine Corps Veterans of the Vietnam War*. Office of Environmental Epidemiology, Veterans Administration, Washington, DC.
- BURMASTER, D. E., AND VON STACKELBERG, K. (1991). Using Monte Carlo simulations in public health risk assessments: Estimating and presenting full distributions of risk. *J. Expos. Anal. Environ. Epidemiol.* **1**, 491-521.
- BURMASTER, D. E., AND LEHR, J. H. (1991). "It's time to make risk assessment a science." *Ground Water Monitor. Rev.: Editorial*. **June**, 1-3.
- BUTTE, N. F., GARZA, C., STUFF, J. E., SMITH, E. O., AND NICHOLS, B. L. (1984). Effects of maternal diet and body composition on lactational performance. *Am. J. Clin. Nutr.* **39**, 296-306.
- BUTTE, N. F., WONG, W. W., KLEIN, P. D., AND GARZA, C. (1991). Measurement of milk intake: Tracer-to-infant deuterium dilution method. *Br. J. Nutr.* **65**, 3-14.
- CALABRESE, E. J., BARNES, R., STANEK, E. J., PASTIDES, H., GILBERT, C. E., VENEMAN, P., WANG, X., LASZTITY, A., AND KOSTECKI, P. T. (1989). How much soil do young children ingest: An epidemiologic study. *Regul. Toxicol. Pharmacol.* **10**, 123-137.
- CALABRESE, E. J., AND STANEK, E. J. (1991a). A guide to interpreting soil ingestion studies. I. Development of a model to estimate the soil ingestion detection level of soil ingestion studies. *Regul. Toxicol. Pharmacol.* **13**, 263-277.
- CALABRESE, E. J., AND STANEK, E. J. (1991b). A guide to interpreting soil ingestion studies. II. Qualitative and quantitative evidence of soil ingestion. *Regul. Toxicol. Pharmacol.* **13**, 278-292.
- CALABRESE, E. J., AND STANEK, E. J. (1992). What proportion of household dust is derived from outdoor soil? *J. Soil Contamination* **1**(3), 253-263.
- CALABRESE, E. J., STANEK, E. J., GILBERT, C. E., AND BARNES, R. M. (1990). Preliminary adult soil ingestion estimates: Results of a pilot study. *Regul. Toxicol. Pharmacol.* **12**, 88-95.
- California Air Pollution Officer's Association (CAPCOA) (1992). *Air Toxics "Hot Spots" Program Risk Assessment Guidelines*. Air Toxicology Unit, Office of Environmental Health Hazard Assessment, Sacramento, CA.

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- Centers for Disease Control (CDC) (1987). Postservice mortality among Vietnam Veterans. *J. Am. Med. Assoc.* **257**, 2708-2713.
- CERLESI, S., DOMENICO, A. D., AND RATTI, S. (1989). 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD) persistence in the Serveso (Milan, Italy) soil. *Ecotoxicol. Environ. Saf.* **18**, 149-164.
- COGGON, D., PANNETT, B., AND WINTER, P. D. (1986). Mortality of workers exposed to 2-methyl-4-chlorophenoxyacetic acid. *Scand. J. Work Environ. Health.* **1**, 448-454.
- COUTURE, L. A., ABBOTT, B. D., AND BIRNBAUM, L. S. (1990). A critical review of the developmental toxicity and teratogenicity of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin: Recent advances toward understanding the mechanism. *Teratology* **42**(6), 619-627.
- COUTURE, L. A., ELWELL, M. R., AND BIRNBAUM, L. S. (1988). Dioxin-like effects observed in male rats following exposure to octachlorodibenzo-*p*-dioxin (OCDD) during a 13 week study. *Toxicol. Appl. Pharmacol.* **93**, 31-46.
- CROUCH, E. (1992). *MSTAGE Version 2.0*. Copyright, Edmund Crouch, Cambridge, MA.
- Crystal Ball (1990). Decisioning Corporation, Suite 200, 1727 Conestoga Street, Boulder, CO 80202.
- DAVIS, S., WALLER, P., BUSCHBOM, R., BALLOU, J., AND WHITE, P. (1990). Quantitative estimates of soil ingestion in normal children between the ages 2 and 7 years: Population-based estimates using aluminum, silicon, and titanium as soil tracer elements. *Arch. Environ. Health* **45**(2), 112-122.
- Department of Health Services (DHS-State of California) (1985). *Health Effects of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin and Related Compounds*. The Epidemiological Studies Section, Department of Health Services, 2151 Berkeley Way, Room 515, Berkeley, California 94704-9980. [Draft]
- DEWEY, K. G., HEINIG, M. J., NOMMSEN, L. A., AND LONNERDAHL, B. (1991). Adequacy of energy intake among breast-fed infants in the DARLING study: Relationships to growth velocity, morbidity, and activity levels. *J. Pediatr.* **119**, 538-547.
- DI DOMENICO, A. (1990). Guidelines for the definition of environmental action alert thresholds for polychlorodibenzodioxins and polychlorodibenzofurans. *Regul. Toxicol. Pharmacol.* **11**, 118-123.
- DRIVER, J. H., KONZ, J. J., AND WHITMYER, G. K. (1989). Soil adherence to human skin. *Bull. Environ. Contam. Toxicol.* **43**, 814-820.
- ESCHENROEDER, A. Q., AND E. J. FAEDER (1988). A Monte-Carlo analysis of health risks from PCB-contaminated mineral oil transformer fires. *Risk Anal.* **8**, 291-299.
- FDA (1983). *Statement by Sanford A. Miller, Ph.D., Bureau of Foods, Food and Drug Administration, before the Subcommittee on Natural Resources, Agriculture, Research and Environment, U.S. House of Representatives*. [As cited in Keenan et al., 1991]
- FINGERHUT, M. A. (1992). Presentation at the International Dioxin Meeting in Tampere, Finland (August 18).
- FINGERHUT, M. A., HALPERIN, W. E., MARLOW, D. A., PIACITELLI, L. A., HONCHAR, P. A., SWEENEY, M. H., GREIFE, A. L., DILL, P. A., STEENLAND, K., AND SURUDA, A. J. (1991). Cancer mortality in workers exposed to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin. *New England J. Med.* **324**(4), 212-218.
- FINKEL, A. M. (1990). *Confronting Uncertainty in Risk Management. A Guide for Decision-Makers*. Center for Risk Management Resources for the Future, Washington, DC.
- FINLEY, B. E., LAU, V., AND PAUSTENBACH, D. J. (1992). Using an uncertainty analysis of direct and indirect exposure to contaminated groundwater to evaluate EPA's MCLs and health-based cleanup goals. *J. Hazard. Mat.* **32**, 263-274.
- FINLEY, B. E., PROCTOR, D., SCOTT, P., AND PAUSTENBACH, D. J. (1993). Standard probability density functions for routine use in probabilistic health risk assessments. *Risk Anal.*, in press.
- FREEMAN, R. A., AND SCHROY, J. M. (1989). Comparison of the rate of TCDD transport at Times Beach and at Eglin AFB. *Chemosphere* **18**, 1305-1312.
- FRIES, G. F., AND PAUSTENBACH, D. J. (1990). Evaluation of potential transmissions of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin-contaminated incinerator emissions to humans via foods. *J. Toxicol. Environ. Health* **29**, 1-43.
- FUERST, P., MEEMKEN, H. A., AND GOEBEL, W. (1986). Determination of polychlorinated dibenzodioxins and dibenzofurans in human milk. *Chemosphere* **15**(9-12), 1977-1980.
- Geraghty & Miller, Inc. (1990a). *Soil and Ditch Sediment Investigation Report*. Atlanta, Georgia.
- Geraghty & Miller, Inc., Air Quality Group (1990b). *Air Quality Investigation during Soil Excavation/Remediation*. Atlanta, Georgia.
- GOUGH, M. (1991). Human exposures from dioxin in soil meeting report. *J. Toxicol. Environ. Health* **32**, 205-245.

- GRAHAM, J., BERRY, M., BRYAN, E. F., CALLAHAN, M. A., FAN, A., FINLEY, B., LYNCH, J., MCKONE, T., OZKAYNAK, H., SEXTON, K., AND WALKER, K. (1992). Role of exposure databases in risk assessment. *Arch. Environ. Health* **47**, 408-420.
- GRAHAM, M., HILEMAN, F., KIRK, D., WENDLING, J., AND WILSON, J. (1985). Background human exposure to 2,3,7,8-TCDD. *Chemosphere* **14**(6/7), 925-928.
- HAAS, C. N., AND SCHEFF, P. A. (1990). Estimation of averages in truncated samples. *Environ. Sci. Technol.* **24**, 912.
- HARRIS, M., ZACHAREWSKI, T., PISKORSKA-PLISZCZYNSKA, J., ROSENGREN, R., AND SAFE, S. (1990). Structure-dependent induction of aryl hydrocarbon hydroxylase activity in C57BL/6 Mice by 2,3,7,8-tetrachlorodibenzo-*p*-dioxin and related congeners: Mechanistic studies. *Toxicol. Appl. Pharmacol.* **105**, 243-253.
- HAWLEY, J. (1985). Assessment of health risk from exposure to contaminated soil. *Risk Anal.* **5**, 289-302.
- HEBERT, C. D., CAO, Q. L., AND BIRNBAUM, L. S. (1990). Inhibition of high-density growth arrest in human squamous carcinoma cells by 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). *Carcinogenesis* **11**(8), 1335-1342.
- HOLSAPPLE, M. P., SNYDER, N. K., WOOD, S. C., AND MORRIS, D. L. (1991). A review of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin-induced changes on immunocompetence: 1991 update. *Toxicology* **69**, 219-255.
- International Commission on Radiological Protection (ICRP). (1966). Deposition and retention models for internal dosimetry of the human respiratory tract. International Commission on Radiological Protection (ICRP), Task Group on Lung Dynamics. *Health Phys.* **12**, 173-207.
- JENNINGS, A. M., WILD, G., WARD, J. D., AND MILFORDWARD, A. (1988). Immunological abnormalities 17 years after accidental exposure to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin. *Br. J. Ind. Med.* **45**, 701-708.
- KANG, H. K., WANTABE, K. K., BREEN, J., REMMERS, J., CONOMOS, M. G., STANLEY, J., AND FLICKER, M. (1991). Dioxin and dibenzofurans in adipose tissue of U.S. Vietnam veterans and controls. *Am. J. Public Health* **81**, 344-349.
- KEENAN, R. E., PAUSTENBACH, D. J., WENNING, R. J., AND PARSONS, A. H. (1991). Pathology reevaluation of the Kociba *et al.* (1978) bioassay of 2,3,7,8-TCDD: Implications for risk assessment. *J. Toxicol. Environ. Health* **34**, 279-296.
- KHERA, K. S. (1984). Maternal toxicity—A possible factor in malformations in mice. *Teratology* **29**, 411-416.
- KHERA, K. S. (1987). Maternal toxicity in humans and animals: Effects on fetal development and criteria for detection. *Teratogen. Carcinogen. Mutagen.* **7**, 287-295.
- KIMBROUGH, R. D. (1990). How toxic is 2,3,7,8-tetrachlorodibenzodioxin to humans? *J. Toxicol. Environ. Health* **30**, 261-271.
- KIMBROUGH, R. D., FALK, H., STEHR, P., AND FRIES, G. (1984). Health implications of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) contamination of residential soil. *J. Toxicol. Environ. Health* **14**, 47-93.
- KNUTSON, A. P. (1984). Immunologic effects of TCDD exposure in humans. *Bull. Environ. Contam. Toxicol.* **33**, 673.
- KOCIBA, R. J., KEYES, D. G., BEYER, J. E., CARREON, R. M., WADE, C. E., DITTENBER, D. A., KALNINS, R. P., FRAUSON, L. E., PARK, C. N., BARNARD, S. D., HUMMEL, R. A., AND HUMISTON, C. G. (1978). Results of a two-year chronic toxicity and oncogenicity study of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in rats. *Toxicol. Appl. Pharmacol.* **46**, 279-303.
- KREWSKI, D., THORSLIND, T., AND WITHEY, J. (1989). Carcinogenic risk assessment of complex mixtures. *Toxicol. Environ. Health* **5**(5), 851-867.
- Le Leche (1991). La Leche League of Orange County. personal communication. Irvine, California.
- LEPOW, M. L., BRUCKMAN, L., GILLETTE, M., MARKOWITZ, S., ROBINO, R., AND KAPISH, J. (1975). Investigations into sources of lead in the environment of urban children. *Environ. Res.* **10**, 415-426.
- LEUNG, H., WENDLING, J. M., ORTH, R., HILEMAN, F., AND PAUSTENBACH, D. J. (1990). Relative distribution of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in human hepatic and adipose tissues. *Toxicol. Lett.* **50**, 275-282.
- LONNERDAL, B., FORSUM, E., AND HAMBRAEUS, L. (1976). A longitudinal study of the protein, nitrogen, and lactose contents of human milk from Swedish well-nourished mothers. *Am. J. Clin. Nutr.* **29**, 1127-1133.
- LUCIER, G. W., RUMBAUGH, R. C., MCCOY, Z., HASS, R., HARVAN, D., AND ALBRO, P. (1986). Ingestion of soil contaminated with 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) alters hepatic enzyme activities in rats. *Fundam. Appl. Toxicol.* **6**, 364-371.
- LYNGE, E. (1985). A follow-up study of cancer incidence among workers in manufacture of phenoxy herbicides in Denmark. *Br. J. Cancer* **52**, 259-270.

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- T., ent. MACGOWAN, R. J., MACGOWAN, C. A., SERDULA, M. K., LANE, J. M., JOESOE, R. M., AND COOK, F. H. (1991). Breast-feeding among women attending women, infants, and children clinics in Georgia. 1987. *J. Pediatr.* **87**(3), 361-366.
- sure. MANZ, A., BERGER, J., DWYER, J. H., FLESCH-JANYS, D., NAGEL, S., AND WALTSGOTT, H. (1991). Cancer mortality among workers in chemical plant contaminated with dioxin. *Lancet* **338**, 959-964.
- rol. MARONPOT, R. R., MONTGOMERY, C. A., BOORMAN, G. A., AND MCCONNELL, E. E. (1986). National Toxicology Program nomenclature for hepatoproliferative lesions of rat. *Toxicol. Pathol.* **14**(2), 263-273.
- 90). MASON, G., FARRELL, K., KEYS, B., PISKORSKA-PLISZCZYNSKA, J., SAFE, L., AND SAFE, S. (1986). Polychlorinated dibenzo-*p*-dioxins: Quantitative in vitro and in vivo structure-activity relationships. *Toxicology* **41**, 21-31.
- 7.8-105. MASON, G., SAWYER, T., KEYS, B., BANDIERA, S., ROMKES, M., PISKORSKA-PLISZCZYNSKA, J., ZMUDZKA, B., AND SAFE, S. (1985). Polychlorinated dibenzofurans (PCDFs): Correlation between in vivo and in vitro structure-activity relationships. *Toxicology* **37**, 1-12.
102. MAXWELL, S. I., AND BURMASTER, D. E. (1993). A distribution of lipid intake from breastmilk during the first year of life. Submitted for publication.
- nan 35- MCCONNELL, E. E., LUCIER, G. W., RUMBAUGH, R. C., ALBRO, P. W., HARVAN, D. J., HASS, J. R., AND HARRIS, M. W. (1984). Dioxin in soil: Bioavailability after ingestion by rats and guinea pigs. *Science* **223**, 1077-1079.
- tra-155. MCKONE, T. E. (1990). Dermal uptake of organic chemicals from a soil matrix. *Risk Anal.* **10**, 407-419.
- for tion MCKONE, T. E., AND BOGEN, K. T. (1991). Predicting the uncertainty in risk assessment: A California groundwater case study. *Environ. Sci. Technol.* **25**(10), 1674-1682.
- ities 08. MCKONE, T. E., AND BOGEN, K. T. (1992). Uncertainties in health risk assessment and integrated case study based on tetrachloroethylene in California groundwater. *Regul. Toxicol. Pharmacol.* **15**, 86-103.
- ER. 1. J. McNULTY, W. (1984). Fetotoxicity of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) for Rhesus Macaques (*Macaca mulatta*). *Environ. Health Perspect.* **60**, 77-88.
- tion ron. MICHALEK, J., *et al.*, (1992). Estimating the half-life of TCDD in humans. *12th Annu. Int. Dioxin Meeting*. August, Tampere, Finland.
- 11- MOCARELLI, P., NEEDHAN, L. I., MOROCCHI, A., PATTERSON, D. G., BRAMBILLA, P., GERTHOUX, P. M., MEAZZA, L., AND CARRERI, V. (1991). Serum concentrations of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin and test results from selected residents of Serveto, Italy. *J. Toxicol. Environ. Health* **32**, 357-366.
- eria ron. MURRAY, F. J., SMITH, F. A., NITSCHKE, K. D., HUMISTON, C. G., KOCIBA, R. J., AND SCHWETZ, B. A. (1979). Three-generation reproduction study of rat livers exposed to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). *Toxicol. Appl. Pharmacol.* **50**, 241-252.
- hlo-1. National Academy of Sciences (NAS) (1983). *Risk Assessment in the Federal Government: Managing the Process*. National Academy Press, Washington, DC.
- icol. National Academy of Science (NAS). (1992). *A Symposium on State-of-the-Art Exposure Assessment Methods*. Sponsored by the National Academy of Science, February 10-11, 1992. Washington DC.
- INS. 78). National Research Council Commission (NRCC) (1981). *Polychlorinated Dibenzo-*p*-dioxins: Criteria for Their Effects on Man and His Environment*. Committee on Scientific Criteria for Environmental Quality National Resource Council. Washington, DC.
- n in 1res. NESSEL, C. S., AMORUSO, M. A., UMBREIT, T. H., AND GALLO, M. A. (1990). Hepatic aryl hydrocarbon hydroxylase and cytochrome P450 induction following the transpulmonary absorption of TCDD from intratracheally instilled particles. *Fundam. Appl. Toxicol.* **15**, 500-509.
- 75). NISBET, I. C. T., AND PAXTON, M. D. (1982). Statistical aspects of three-generation studies of the reproductive toxicity of TCDD and 2,4,5-T. *Am. Stat.* **36**(3), 290-298.
- tion 282. OTT, E. G., OLSON, R. A., COOK, R. R., AND BOND, G. G. (1987). Cohort mortality study of chemical workers with potential exposure to the higher chlorinated dioxins. *J. Occup. Med.* **29**(5), 422-429.
- gen. 27- PARKIN, T. B., MELSINGER, J. J., CHESTER, S. T., STARR, J. L., AND ROBINSON, D. (1988). Evaluation of statistical estimation methods for lognormally distributed variables. *Soil Sci. J.* **52**, 323.
- tion ities PAUSTENBACH, D. J. (1987). Assessing the potential environment and human health risks of contaminated soil. *Comments Toxicol.* **1**, 185-220.
- sides PAUSTENBACH, D., SHU, H., AND MURRAY, F. (1986). A critical examination of assumptions used in risk assessment of dioxin contaminated soil. *Regul. Toxicol. Pharmacol.* **6**, 284-307.
- PAUSTENBACH, D. J., JERNIGAN, J. D., FINLEY, B. L., RIPPLE, S. R., AND KEENAN, R. E. (1990). The current practice of health risk assessment: Potential impact on standards for toxic air contaminants. *J. Air Waste Manage. Assoc.* **40**(12), 1620-1630.

- PAUSTENBACH, D. J., SARLOS, T. T., LAU, V., FINLEY, B. L., JEFFREY, D. A., AND UNGS, M. J. (1991a). An evaluation of the inhalation hazard posed by dioxin-contaminated soil. *J. Air Waste Manage. Assoc.* **41**, 1334-1340.
- PAUSTENBACH, D. J., MEYER, D. M., SHEEHAN, P. J., AND LAU, V. (1991b). An assessment and quantitative uncertainty analysis of the health risks to workers exposed to chromium contaminated soils. *Toxicol. Ind. Health* **7**(3), 159-19.
- PAUSTENBACH, D. J., LAYARD, MAXWELL W., WENNING, RICHARD J., AND KEENAN, RUSSELL E. (1991c). Risk assessment of 2,3,7,8-TCDD using a biologically based cancer model: A reevaluation of the Kociba *et al.* bioassay using 1978 and 1990 histopathology criteria. *J. Toxicol. Environ. Health* **34**, 11-26.
- PAUSTENBACH, D. J., AND LEUNG, H. W. (1992). Techniques for assessing the health risks of dermal contact with chemicals in the environment. In *The Risk Assessment of Skin Exposed to Chemicals* (Maibach, Wong, and Knack, eds.), Chap. 7. CRC Press, Cleveland, OH.
- PAUSTENBACH, D. J., WENNING, R. J., LAU, V., HARRINGTON, N. W., RENNI, D. K., AND PARSONS, A. H. (1992a). Recent developments on the hazards posed by 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in soil: Implications for setting risk-based cleanup levels at residential and industrial sites. *Toxicol. Environ. Health* **34**, 103-148.
- PAUSTENBACH, D. J., JERNIGAN, J.-D., BASS, R., KALMES, R., AND SCOTT, P. (1992b). A proposed approach to regulating contaminated soil: Identify safe concentrations for seven of the most frequently encountered exposure scenarios. *Regul. Toxicol. Pharmacol.* **16**, 21-56.
- PIRKLE, J. L., WOLFE, W. M., PATTERSON, D. G., JR., NEEDHAM, L. L., MICHALEK, J. E., MINER, J. C., PETERSON, M. R., AND PHILLIPS, D. L. (1989). Estimates of the half-life of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in Vietnam veterans of Operation Ranch Hand. *J. Toxicol. Environ. Health* **27**, 165-171.
- PODOLL, R. T., JABER, H. M., AND MILL, T. (1986). Tetrachlorodibenzodioxin: Rates of volatilization and photolysis in the environment. *Environ. Sci. Technol.* **20**, 490-492. [as cited in Paustenbach *et al.*, 1991b]
- POIGER, H., AND SCHLATTER, C. (1980). Influence of solvents and adsorbents on dermal and intestinal absorption of TCDD. *Food Cosmet. Toxicol.* **18**, 477-481.
- POIGER, H., AND SCHLATTER, C. (1986). Pharmacokinetics of 2,3,7,8-TCDD in man. *Chemosphere* **16**, 1489-1494.
- POLAND, A., AND KNUTSON, J. C. (1982). 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin and related halogenated aromatic hydrocarbons: Examination of the mechanism of toxicity. *Annu. Rev. Pharmacol. Toxicol.* **22**, 517-554.
- QUE HEE, S. S., PEACE, B., SCOTT, C. S., BOYLE, J. R., BORNSCHEIN, R. L., AND HAMMOND, P. B. (1985). Evolution of efficient methods to sample lead sources, such as house dust and hand dust, in the homes of children. *Environ. Res.* **38**, 77-95.
- RAPPE, C., NYGREN, M., BUSER, H., MASUDA, Y., KUROKI, H., AND CHEN, P. H. (1983). Identification of polychlorinated dioxins (PCDDs) and dibenzofurans (PCDFs) in human samples, occupational exposure and yusho patients. In *Human and Environmental Risks of Chlorinated Dioxins and Related Compounds* (R. E. Tucker, A. L. Young, and A. P. Gray, eds.), pp. 241-253. Plenum Press, New York.
- RIIHIMAKI, V., ASP, S., AND HERNBERG, S. (1983). Mortality and cancer morbidity among phenoxy acid applicators in Finland. *Chemosphere* **12**(4/5), 779-784.
- ROEGNER, R. H., *et al.* (1991). *Air Force Health Study: An Epidemiologic Investigation of Health Effects in Air Force Personnel Following Exposure to Herbicides*. AL-TR-91-0009. Science Applications International Corporation (SAIC), McLean, VA, and Human Systems Division (AFSC), Brooks Air Force Base, TX.
- ROELS, H. A., BUCHET, J., LAUWERYS, R. R., BRAUAUX, P., CLAEYS-THOREAU, F., LAFONTAINE, A., AND VERDUYN, G. (1980). Exposure to lead by the oral and the pulmonary routes of children living in the vicinity of a primary lead smelter. *Environ. Res.* **22**, 81-94.
- RYAN, J. J., LIZOTTE, R., AND LAU, B. P. Y. (1985). Chlorinated dibenzo-*p*-dioxins and chlorinated dibenzofurans in Canadian human adipose tissue. *Chemosphere* **14**(6/7), 697-706.
- SAFE, S. (1987). Determination of 2,3,7,8-TCDD toxic equivalent factors (TEFs): Support for the use of the in vitro AHH induction assay. *Chemosphere* **14**(4), 791-802.
- SALHOTRA, *et al.* (1991). *Application of Monte Carlo Simulation to Estimate Probabilities of Exposure and Human Health Risks*. Woodward-Clyde Report, Oakland, California.
- SARACCI, R., KOGEVINAS, M., BERTAZZI, P. A., BUENO DE MESQUITA, B. H., COOGON, D., GREEN, L. M., KAUPPINEN, T., L'ABBE, K. A., LITTORIN, M., LYNGE, E., MATHEWS, J. D., NEUBERGER, M., OSMAN, J., PEARCE, N., AND WINKELMANN, R. (1991). Cancer mortality in workers exposed to chlorophenoxy herbicides and chlorophenols. *Lancet* **338**, 1027-1032.

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- SCHecter, A., RYAN, J. J., LIZOTTE, R., SUN, W. F., MILLER, L., GITILITZ, G., AND BOGDASARIAN, M. (1985). Chlorinated dibenzodioxins and dibenzofurans in human adipose tissue from exposed and control New York state patients. *Chemosphere* 14(6/7), 933-937.
- SCHWETZ, B. A., NORRIS, J. M., SPARSCHU, G. L., ROWE, V. K., GEHRING, P. J., EMERSON, J. L., AND GERBIG, C. G. (1973). Toxicology of chlorinated dibenzo-p-dioxins. *Environ. Health Perspect.* 5, 87-98.
- SHEEHAN, P. J., MEYER, D. M., SAUER, M. M., AND PAUSTENBACH, D. J. (1991). Assessment of the human health risks posed by exposure to chromium-contaminated soils. *J. Toxicol. Environ. Health* 34, 161-201.
- SHU, H. P., PAUSTENBACH, D. J., AND MURRAY, F. J. (1987). A critical evaluation of the use of mutagenesis, carcinogenesis, and tumor promotion data in cancer risk assessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin. *Regul. Toxicol. Pharmacol.* 7, 57-88.
- SHU, H., PAUSTENBACH, D., MURRAY, F. J., MARPLE, L., BRUNCK, B., DEI ROSSI, D., AND TEITELBAUM, P. (1988a). Bioavailability of soil-bound TCDD: Oral bioavailability in the rat. *Fundam. Appl. Toxicol.* 10, 648-654.
- SHU, H., TEITELBAUM, P., WEBB, A. S., MARPLE, L., BRUNCK, B., DEI ROSSI, D., MURRAY, F. J., AND PAUSTENBACH, D. (1988b). Bioavailability of soil-bound TCDD: Dermal bioavailability in the rat. *Fundam. Appl. Toxicol.* 10, 335-343.
- SMITH, A. H. (1987). Infant exposure assessment for breast milk dioxins and furans derived from incinerator emissions. *Risk Anal.* 7(3), 347-353.
- SNYDER, W. S. (1975). *Report of the Task Group on Reference Man*. International Commission of Radiological Protection (ICRP), No. 23. Pergamon Press, New York.
- STEHR-GREEN, P. A., STEIN, G., FALK, H., SAMPSON, E., SMITH, E. G., STEINBERG, K., WEBB, L., AYRES, S., SCHRAMM, W., DONNELL, H. D., AND GEDNEG, W. B. (1986). A pilot epidemiologic study of possible health effects associated with 2,3,7,8-tetrachlorodibenzo-p-dioxin contamination in Missouri. *Arch. Environ. Health* 41(1), 16-27.
- SUSKIND, R., AND HERTZBERG, V. (1984). Human health effects of 2,4,5-T and its toxic contaminants. *J. Am. Med. Assoc.* 251, 2372-2380.
- SWEENEY, M., et al. (1992). *Evidence that Exposure to TCDD Produces an Increase in Diabetes*. Presented at the International Dioxin Meeting, August 17, Tampere, Finland.
- TOGNONI, G., AND BONACCORSI, A. (1982). Epidemiological problems with TCDD: A critical review. *Drug Metab. Rev.* 13(3), 447-451.
- TRAVIS, C. C., AND HESTER, S. T. (1990). Background exposure to chemicals: What is the risk? *Risk Anal.* 10(4), 463-466.
- TRAVIS, C. C., LAND, M. L., AND HATTEMER-FREY, H. (1990). Estimating the mean of data sets with nondetectable values. *Environ. Sci. Technol.* 24, 981.
- TRIJONIS, J., ELDON, J., GINS, J., AND BERGLUND, G. (1980). *Analysis of the St. Louis RAMS Ambient Particulate Data*. EPA 450/4-80-006a. Office of Air, Noise, and Radiation, U. S. Environmental Protection Agency, Cincinnati, Ohio.
- UMBREIT, T. H., HESSE, E. J., AND GALLO, M. A. (1986). Acute toxicity of TCDD contaminated soil from an industrial site. *Science* 232, 497-499.
- United Kingdom (UK) (1989). *Dioxins in the Environment*. Report of an Interdepartmental Working Group on polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Pollution Paper No. 27. Central Directorate of Environmental Protection, London.
- United States Environmental Protection Agency (USEPA) (1985a). *Health Assessment Document for Polychlorinated Dibenz-o-p-dioxins* (Final Report) (600/8-84-014F). Office of Environmental Assessment, Cincinnati, OH.
- United States Environmental Protection Agency (USEPA) (1985b). *Development of Statistical Distributions or Ranges of Standard Factors Used in Exposure Assessments*. By E. Anderson, N. Browne, S. Duletsky, J. Ramig, and T. Warn. For Office of Health and Environmental Assessment. EPA/600/8-85/010. United States Environmental Protection Agency, Washington, DC.
- United States Environmental Protection Agency (USEPA) (1988). *Reference Physiological Parameters in Pharmacokinetic Modeling (A. Arms and C. Travis)*. EPA/600/021. Office of Health and Environmental Assessment, Washington DC.
- United States Environmental Protection Agency (USEPA) (1989). *Risk assessment guidance for Superfund. Volume 1. Human Health Evaluation Manual*. Part A. EPA/540/1-89/002. Office of Emergency and Remedial Response, Washington DC.

- United States Environmental Protection Agency (USEPA) (1990). *Exposure Factors Handbook*. EPA/600/8-89/043. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment.
- United States Environmental Protection Agency (USEPA) (1991a). *Preliminary Results from September 26, 1990, Dioxin Investigation*. Washington, DC.
- United States Environmental Protection Agency (USEPA) (1991b). *Guidance on Risk Characterization for Risk Managers and Risk Assessors*. Memo sent to Assistant Administrators and Regional Administrators.
- United States Environmental Protection Agency (USEPA) (1992a). *Guidelines for Exposure Assessment (Draft document)*. Office of Health and Environmental Assessment (RD-689). Washington, DC.
- United States Environmental Protection Agency (USEPA) (1992b). *Dermal Exposure Assessment: Principles and Applications*. EPA/600/8-91/011B. Office of Research and Development.
- United States Environmental Protection Agency (USEPA) (1993). *Assessing Exposure to Dioxin Like Compounds*. Washington, DC, in press.
- VAN WIJNEN, D. E., CLAUSING, P., AND BRUNEKREFF, B. (1990). Estimated soil ingestion by children. *Environ. Res.* **51**, 147-162.
- VOS, J. G., MOORE, J. A., AND ZUNKLE, J. G. (1973). Effect of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin on the immune system of laboratory animals. *Environ. Health Perspect.* **5**, 149-158.
- WASSOM, J. S., HUFF, J. E., AND LOPRIENO, N. (1977). A review of genetic toxicology of chlorinated dibenzo-*p*-dioxins. *Mutat. Res.* **47**, 141-160.
- WEBER, et al. (1984). Tetrogenicity of 2,3,7,8-tetrachlorodibenzofurans (TCDF) in mice. *Toxicol. Lett.* **20**, 183-188.
- WENNING, R. J., HARRIS, M. A., UNGS, M. J., FINLEY, B. L., PAUSTENBACH, D. J., AND BEDBURY, H. (1992). Chemometric comparisons of polychlorinated dibenzo-*p*-dioxin and dibenzofuran residues in surficial sediments from Newark Bay, New Jersey and other industrialized waterways. *Arch. Environ. Contam. Toxicol.* **22**, 397-413.
- WHITEHEAD, R. G., AND PAUL, A. A. (1981). Infant growth and human milk requirements. *Lancet* **2**, 161-163.
- WIKLUND, K., AND HOLM, L. E. (1986). Soft tissue sarcoma risk in Swedish agricultural and forestry workers. *J. Natl. Cancer Inst.* **76**, 229-234.
- WIKLUND, K., AND HOLM, L. E. (1987). Trends in cancer risks among Swedish agricultural workers. *J. Natl. Cancer Inst.* **77**, 657.
- WILKINSON, L. (1989). *SYSTAT: The System for Statistics*. SYSTAT, Inc., Evanston, IL.
- YANDERS, A. F., KAPILA, S., LO, Y-H., PURI, R., AND CERLESI, S. (1990). Persistence of tetrachlorodibenzo-*p*-dioxin in soil: Times Beach case. *Proc. Int. Dioxin 90 Conf.*, 339-342.
- YANDERS, A. F., ORAZIO, C. E., PURI, R. K., AND KAPILA, S. (1989). On translocation of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin: Time dependent analysis at the Times Beach experimental site. *Chemosphere* **19**(1-6), 429-432.
- ZACK, J. A., AND GAFFEY, W. R. (1983). A mortality study of workers employed at the Monsanto Company plant in Nitro, West Virginia. *Environ. Sci. Res.* **26**, 575-591.
- ZOBER, A., MESSERER, P., AND HUBER, P. (1990). Thirty-four-year mortality follow-up of BASF employees exposed to 2,3,7,8-TCDD after the 1953 accident. *Int. Arch. Environ. Health* **62**, 139-157.
- ZWIACHER, W. E., AND DENNISON, W. J. (1988). *Multi-pathway Health Risk Assessment Input Parameters Guidance Document*. South Coast Air Quality Management District.

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