1. BACKGROUND AND SUMMARY

1.1. BACKGROUND

This reassessment is comprised of three reports:

Part 1. *Estimating Exposure to Dioxin-Like Compounds* (EPA, 2000a) (which expanded upon a 1988 draft exposure report titled, *Estimating Exposure to 2,3,7,8-TCDD* [EPA, 1988]);
Part 2. *Health Assessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds* (EPA, 1994; EPA, 2000b); and
Part 3. *Dioxin: Integrated Summary and Risk Characterization for 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds* (EPA, 2000c).

Throughout the remainder of this document, these three parts as a whole will be abbreviated as the Reassessment Documents, and the individual parts will be referred to as the Exposure Reassessment Document, the Health Reassessment Document, and the Risk Characterization. The Exposure Reassessment Document has expanded to three volumes, as discussed below. Volumes 1 and 2 of the Exposure Reassessment Document are summarized in Section 4 of the Risk Characterization.

The process for developing the Reassessment Documents has been open and participatory. Each of the documents has been developed in collaboration with scientists from inside and outside the Federal Government. Each document has undergone extensive internal and external review, including review by EPA's Science Advisory Board (SAB). In September 1994, drafts of each document were made available for public review and comment. This included a 150-day comment period and 11 public meetings around the country to receive oral and written comments. These comments, along with those of the SAB (EPA, 1995a), have been considered in the drafting of this final document. The Dose-Response Chapter of the Health Document underwent peer review in 1997 (EPA, 1997a); an earlier version of the Integrated Summary and Risk Characterization underwent development and review in 1997 and 1998, and comments have been incorporated. In 1998, EPA released a workshop review version of the sources inventory (EPA, 1998), one of the three volumes of the Exposure Reassessment Document. In addition, as requested by the SAB, a chapter on Toxic Equivalency has been developed and underwent external peer review in parallel with the Integrated Summary and Risk Characterization in July, 2000. The November, 2000, review by the SAB of the Dose-Response Chapter, the Toxic Equivalency Chapter and the Integrated Summary and Risk Characterization was the final step in this open and participatory process of reassessment. The full set of background documents and the integrative summary and risk characterization replace the previous dioxin assessments as the scientific basis for EPA decision-making.

The final Exposure Reassessment Document reflects changes made as a result of both review comments and analyses of a variety of other types of information that has come available. These include relevant information obtained from published peer-reviewed literature, EPA program offices, and other Federal agencies. This version of the Exposure Reassessment Document is current in this regard through 2000.

The purpose of the Exposure Reassessment Document is threefold: 1) to inventory the known sources of release of dioxins into the environment, 2) to develop an understanding of dioxins in the environment, including fate and transport properties, environmental and exposure media concentrations, background as well as elevated exposures, and temporal trends in exposure, and 3) provide site-specific procedures for evaluating the incremental exposures due to specific sources of dioxin-like compounds. Following this structure, the Exposure Reassessment Document is presented in three volumes:

Volume I - Sources of Dioxin-Like Compounds in the United States

This volume presents a comprehensive review of known sources of environmental releases of dioxin-like compounds in the United States. It includes an inventory of known source activity in terms of estimates of annual releases of dioxin-like compounds into the U.S. environment (i.e., air, water and land). This inventory is specific for two reference years, 1987 and 1995. From these data, it is possible to compare and contrast releases of dioxin-like compounds among the sources and between the reference years.

Volume II - Properties, Environmental Levels, and Background Exposures

This volume presents and evaluates information on the physical-chemical properties, environmental fate, environmental and exposure media levels, background and elevated human exposures, and temporal trends of dioxin-like compounds in the U.S. environment during the 20th century.

Volume III - Site-Specific Assessment Procedures

This volume presents procedures for evaluating the incremental impact from sources of dioxin release into the environment. The sources covered include contaminated soils, stack emissions, and point discharges into surface water. This volume includes sections on: exposure parameters and exposure scenario development; stack emissions and atmospheric transport modeling; aquatic and terrestrial fate, and food chain modeling; demonstration of methodologies; and uncertainty evaluations including exercises on sensitivity analysis and model validation, review of Monte Carlo assessments conducted for dioxin-like compounds, and other discussions.

The primary technical resource supporting the development of the inventory of sources of dioxin-like compounds discussed in Volume I (above) is the Database of Sources of Environmental Releases of Dioxin-Like Compounds in the United States (EPA/600/C-01/012. March, 2001). This database includes congener-specific CDD and CDF emissions data extracted from original engineering test reports. It has been published independently from the Reassessment and is available on Compact Disk -Read only Memory (CD-ROM), without cost, from EPA's National Service Center for Environmental Publications (NSCEP) in Cincinnati, Ohio (telephone: 1-800-490-9198, or 513-489-8190; fax: 513-489-8695). In addition, it can be downloaded from the web page of the National Center for Environmental Assessment, www.epa.gov/ncea/dioxin.htm.

1.2. DEFINITION OF DIOXIN-LIKE COMPOUNDS

This assessment addresses specific compounds in the following chemical classes: polychlorinated dibenzo-*p*-dioxins (PCDDs or CDDs), polychlorinated dibenzofurans (PCDFs or CDFs), polybrominated dibenzo-*p*-dioxins (PBDDs or BDDs), polybrominated dibenzofurans (PBDFs or BDFs), and polychlorinated biphenyls (PCBs), and describes this subset of chemicals as "dioxin-like." Dioxin-like refers to the fact that these compounds have similar chemical structure, similar physical-chemical properties, and invoke a common battery of toxic responses. Because of their hydrophobic nature and resistance towards metabolism, these chemicals persist and bioaccumulate in fatty tissues of animals and humans. The CDDs include 75 individual compounds; CDFs include 135 different compounds. These individual compounds are referred to technically as congeners. Likewise, the BDDs include 75 different congeners and the BDFs include an additional 135 congeners. Only 7 of the 75 congeners of CDDs, or of BDDs, are thought to have dioxin-like toxicity; these are ones with chlorine/bromine substitutions in, at a minimum, the 2, 3, 7, and 8 positions. Only 10 of the 135 possible congeners of CDFs or of BDFs are thought to have dioxin-like toxicity; these also are ones with substitutions in the 2, 3, 7, and 8 positions. This suggests that 17 individual CDDs/CDFs, and an additional 17 BDDs/BDFs, exhibit dioxin-like toxicity. The database on many of the brominated compounds regarding dioxin-like activity has been less extensively evaluated, and these compounds have not been explicitly considered in this assessment.

There are 209 PCB congeners. Only 12 of the 209 congeners are thought to have dioxin-like toxicity; these are PCBs with 4 or more lateral chlorines with 1 or no substitution in the ortho position. These compounds are sometimes referred to as coplanar, meaning that they can assume a flat configuration with rings in the same plane. Similarly configured polybrominated biphenyls (PBBs) are likely to have similar properties. However, the database on these compounds with regard to dioxin-like activity has been less extensively evaluated, and these compounds have not been explicitly considered in this assessment. Mixed chlorinated and brominated congeners of dioxins, furans, and biphenyls also exist, increasing the number of compounds potentially considered dioxin-like within the definitions of this assessment. The physical/chemical properties of each congener vary according to the degree and position of chlorine and/or bromine substitution. Very little is known about occurrence and toxicity of the mixed (chlorinated and brominated) dioxin, furan, and biphenyl congeners. Again, these compounds have not been explicitly considered in this assessment. Generally speaking, this assessment focuses on the 17 CDDs/CDFs and a few of the coplanar PCBs that are frequently encountered in source characterization or environmental samples. While recognizing that other “dioxin-like” compounds exist in the chemical classes discussed above (e.g., brominated or chlorinated/brominated congeners) or in other chemical classes (e.g., halogenated naphthalenes or benzenes, azo- or azoxybenzenes), the evaluation of less than two dozen chlorinated congeners is generally considered sufficient to characterize environmental “dioxin.”

The chlorinated dibenzodioxins and dibenzofurans are tricyclic aromatic compounds with similar physical and chemical properties. Certain of the PCBs (the so-called coplanar or mono-ortho coplanar congeners) are also structurally and conformationally similar. The most widely studied of this general class of compounds is 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). This compound, often called simply “dioxin,” represents the reference compound for this class of compounds. The structure of TCDD and several related compounds is shown in Figure 1-1 . Although sometimes confusing, the term “dioxin” is often also used to refer to the complex mixtures of TCDD and related compounds emitted from sources, or found in the environment or in biological samples. It can also be used to refer to the total TCDD “equivalents” found in a sample. This concept of toxic equivalency is discussed below.

1.3. TOXIC EQUIVALENCY FACTORS

CDDs, CDFs, and PCBs are commonly found as complex mixtures when detected in environmental media and biological tissues, or when measured as environmental releases from specific sources. Humans are likely to be exposed to variable distributions of CDDs, CDFs, and dioxin-like PCB congeners that vary by source and pathway of exposures. This complicates the human health risk assessment that may be associated with exposures to variable mixtures of dioxin-like compounds. In order to address this problem, the concept of toxic equivalency has been considered and discussed by the scientific community, and TEFs have been developed and introduced to facilitate risk assessment of exposure to these chemical mixtures.

On the most basic level, TEFs compare the potential toxicity of each dioxin-like compound comprising the mixture to the well-studied and understood toxicity of TCDD, the most toxic member of the group. The background and historical perspective regarding this procedure is described in detail in Part II, Chapter 9, Section 9.1, 9.2, and in Agency documents (EPA 1987, 1989a,b, 1991). This procedure involves assigning individual TEFs to the 2,3,7,8-substituted CDD/CDF congeners and "dioxin-like" PCBs. To accomplish this, scientists have reviewed the toxicological databases along with considerations of chemical structure, persistence, and resistance to metabolism, and have agreed to ascribe specific, "order of magnitude" TEFs for each dioxin-like congener relative to TCDD, which is assigned a TEF of 1.0. The other congeners have TEF values ranging from 1.0 to 0.00001. Thus, these TEFs are the result of scientific judgment of a panel of experts using all of the available data and are selected to account for uncertainties in the available data and to avoid underestimating risk. In this sense, they can be described as "public health conservative" values. To apply this TEF concept, the TEF of each congener present in a mixture is multiplied by the respective mass concentration and the products are summed to represent the 2,3,7,8-TCDD Toxic Equivalence (TEQ) of the mixture, as determined by Equation (1-1):

The TEF values for PCDDs and PCDFs were originally adopted by international convention (EPA, 1989a). Subsequent to the development of the first international TEFs for CDD/CDFs, these values were further reviewed and/or revised and TEFs were also developed for PCBs (Ahlborg et al., 1994; van den Berg et al., 1998). A problem arises in that past and present quantitative exposure and risk assessments may not have clearly identified which of three TEF schemes was used to estimate the TEQ. This reassessment introduces a new uniform TEQ

$$TEQ \cong \sum_{i-n} \left(Congener_i \times TEF_i \right) + \left(Congener_j \times TEF_j \right) + ...\left(Congener_n \times TEF_n \right) \quad (1\text{-}1)$$

nomenclature that clearly distinguishes between the different TEF schemes and identifies the congener groups included in specific TEQ calculations. The nomenclature uses the following abbreviations to designate which TEF scheme was used in the TEQ calculation:

1. I-TEQ refers to the International TEF scheme adopted by EPA in 1989 (EPA, 1989a). See Table 1-1.
2. TEQ-WHO$_{94}$ refers to the 1994 WHO extension of the I-TEF scheme to include 13 dioxin-like PCBs (Ahlborg et al., 1994). See Table 1-2.

3. TEQ-WHO_{98} refers to the 1998 WHO update to the previously established TEFs for dioxins, furans, and dioxin-like PCBs (van den Berg et al., 1998). See Table 1-3.

The nomenclature also uses subscripts to indicate which family of compounds is included in any specific TEQ calculation. Under this convention, the subscript D is used to designate dioxins, the subscript F to designate furans and the subscript P to designate PCBs. As an example, "TEQ_{DF}-WHO_{98}" would be used to describe a mixture for which only dioxin and furan congeners were determined and where the TEQ was calculated using the WHO_{98} scheme. If PCBs had also been determined, the nomenclature would be "TEQ_{DFP}-WHO_{98}." Note that the designations TEQ_{DF}-WHO_{94} and I-TEQ_{DF} are interchangeable, as the TEFs for dioxins and furans are the same in each scheme. Note also that in this document, I-TEQ sometimes appears without the D and F subscripts. This indicates that the TEQ calculation includes both dioxins and furans.

This reassessment recommends that the WHO_{98} TEF scheme be used to assign toxic equivalency to complex environmental mixtures for assessment and regulatory purposes. Sections in the Health Reassessment Document, and summarized in the Risk Characterization, describe the mode(s) of action by which dioxin-like chemicals mediate biochemical and toxicological actions. These data provide the scientific basis for the TEF/TEQ methodology. In its 20-year history, the approach has evolved, and decision criteria supporting the scientific judgment and expert opinion used in assigning TEFs has become more transparent. Numerous states, countries, and several international organizations have evaluated and adopted this approach to evaluating complex mixtures of dioxin and related compounds. It has become the accepted methodology, although the need for research to explore alternative approaches is widely endorsed. Clearly, basing risk on TCDD alone or assuming all chemicals are equally potent to TCDD is inappropriate on the basis of available data. Although uncertainties in the use of the TEF methodology have been identified (which are described in detail in the Health Reassessment Document, Chapter 9, Section 9.5), one must examine the use of this method in the broader context of the need to evaluate the potential public health impact of complex mixtures of persistent, bioaccumulative chemicals. It can be generally concluded that the use of TEF methodology for evaluating complex mixtures of dioxin-like compounds decreases the overall uncertainties in the risk assessment process as compared to alternative approaches. Use of the latest consensus values for TEFs assures that the most recent scientific information informs this "useful, interim approach" (EPA, 1989a; Kutz et al., 1990) to dealing with complex environmental mixtures of dioxin-like compounds. As stated by the U.S. EPA Science Advisory Board (EPA, 1995a), "The use of the TEFs as a basis for developing an overall index of public health risk is clearly justifiable, but its practical application depends on the reliability of the

TEFs and the availability of representative and reliable exposure data." EPA will continue to work with the international scientific community to update these TEF values to assure that the most up-to-date and reliable data are used in their derivation and to evaluate their use on a periodic basis.

A chemical is assigned a TEF value based on all the available data comparing the chemical to either TCDD or PCB 126. In addition, there are weighting criteria that place more emphasis on chronic and subchronic studies examining toxic endpoints (van den Berg et al., 1998). There is a broad range in the quantity and quality of the data available for individual congeners. For example, the TEF for PCB 126 is based on over 60 in vivo endpoints examining responses as diverse as enzyme induction, developmental toxicity, immunotoxicity, hepatic toxicity, alterations in hormones and tumor promotion, while the TEF for 3,4,4',5-tetrachlorobiphenyl (PCB 81) is based on in vitro CYP1A induction and QSAR calculations. Fortunately, PCB 81 does not significantly contribute to human TEQ exposures. There are 5 congeners that contribute approximately 80% of the total TEQ in humans: 2,3,7,8-TCDD, 1,2,3,7,8-PCDD, 1,2,3,6,7,8-HxCDD, 2,3,4,7,8-PCDF, and PCB 126 (See Part I, Volume 3 and Section 4.4.3 of this document). With the exception of 1,2,3,6,7,8-HxCDD, the TEFs for these chemicals are based on a number of different endpoints from multiple studies performed in different laboratories. The TEF for 1,2,3,6,7,8-HxCDD is based on a two-year bioassay in which rats were exposed to a mixture of 1,2,3,6,7,8-HxCDD and 1,2,3,7,8,9-HxCDD. The TEFs for 2,3,4,7,8-PCDF and PCB 126 are similar to the mean REP value for all in vivo endpoints and are similar to their REPs for tumor promotion. The TEF for 12378-PCDD is based largely on its REP for tumor promotion in rats. From these data, it is clear that the chemicals that contribute approximately 80% to the total human TEQ are well studied and the assigned TEFs provide reasonable estimates of the relative potency of these chemicals. In contrast, while there are some chemicals in the TEF methodology which have minimal data sets to reliably assess their relative potency, these chemicals do not contribute substantially to the human blood TEQ.

The ability of the TEF methodology to predict the biological effects of mixtures containing dioxin-like chemicals has been evaluated in a number of experimental systems. These studies generally demonstrate that the assumption of additivity provides a reasonable estimate of the dioxin-like potential of a mixture (described in the Health Reassessment Document, Chapter 9, Section 9.4). In addition, there are examples of non-additive interactions between dioxins and non-dioxins. Both greater than additive and less than additive interactions have been observed in these studies. In general the non-additive interactions between the dioxins and non-dioxins have been observed at doses that are considerably higher than present background human exposures.

There are a number of natural chemicals that bind and activate the AhR and induce some dioxin-like effects. It has been proposed by some scientists that these chemicals contribute significantly to the total TEQ exposures and that these exposures far out weigh those from PCDDs, PCDFs and PCBs (Safe, 1995). While this hypothesis is intriguing, there are several limitations to these analyses. The in vivo data on the natural AhR ligands is limited to enzyme induction and a single developmental study. Few, if any, toxicology studies demonstrating clear dioxin-like toxicities have been published. The natural AhR ligands are rapidly metabolized and result in both transient tissue concentrations and transient effects. The natural ligands also have significant biological effects that are independent of the AhR and it is not clear as to the role of the AhR in the biological effects of these chemicals. Clearly this issue requires further research in order to better understand the relative potential health effect of dioxin and related chemicals as compared to natural AhR ligands.

One of the limitations of the use of the TEF methodology in risk assessment of complex environmental mixtures is that the risk from non-dioxin-like chemicals is not evaluated in concert with that of dioxin-like chemicals. Another limition of the TEF methodology is their application to non-biological samples. The fate and distribution of PCDDs, PCDFs and PCBs are not necessarily related to their TEF. Thus, the use of the TEF for non-biological media must be done cautiously. Future approaches to the assessment of environmental mixtures should focus on the development of methods that will allow risks to be predicted when multiple mechanisms are present from a variety of contaminants.

1.4. OVERALL COMMENTS ON THE USE OF VOLUME III OF THE DIOXIN EXPOSURE DOCUMENT

Users of the dioxin exposure document should recognize the following:

1. This document does not present detailed procedures for evaluating multiple sources of release. However, it can be used in two ways to address this issue. Incremental impacts estimated with procedures in Volume III can be compared to background exposure estimates which are presented in Volume II. This would be a way of comparing the incremental impact of a specific source to an individual's total exposure otherwise. Assuming the releases from multiple sources behave independently, it is possible to model them individually and then add the impacts. For example, if several stack emission sources are identified and their emissions quantified, and it is desired to evaluate the impact of all sources simultaneously, then it is possible with ISCST3 to model each stack emission source individually and then sum the concentrations and depositions at points of interest in the surrounding area.

2. The demonstration of the site-specific procedures presented in this exposure document best serve as general examples for evaluating exposures to dioxin-like compounds, rather than specific assessments. This demonstration scenarios in Chapter 5 of this document were not generated for purposes of supporting any specific regulation. Rather, they were only intended to demonstrate the procedure described earlier in Chapters 2 through 4. Certainly, the goal of developing "high end" and "central" is consistent with Agency policy, and even assignment of many of the exposure and fate parameters can be adopted for other assessments. Therefore, assessors may find even the specifics of the demonstration scenarios useful for other purposes.

3. The understanding of the exposure to dioxin-like compounds continues to expand. Despite being one of the most studied groups of organic enivronmental contaminants, new information is generated almost daily about dioxin-like compounds. This document is considered to be current through 2000.

Numerous parameter values are used in this document and it is important to understand their degree of "endorsement" by EPA. The parameters can be divided into the following four classes for purposes of addressing this issue:

1) **First Order Defaults:** As defaults, these parameters are independent of site specific characteristics and can be used for any assessment. Also, as first order defaults, it is felt that the values selected for the demonstration scenarios carry a sufficient weight of evidence from current literature such that these values are recommended for other assessments. Several of the chemical specific parameters, such as the Henry's Constant, H, and the organic carbon partition coefficient, Koc, fall into this category. The qualifier above, "current literature", indicates that new information could lead to changes in these values.

2) **Second Order Defaults:** Like the above category, these parameters are judged to be independent of site specific characteristics. However, unlike the above category, the current scientific weight of evidence is judged insufficient to describe values selected for demonstration purposes as first order defaults. Parameters of principal note in this category are the bioconcentration parameters specific to the chemicals, such the Biota Sediment Accumulation Factor, or BSAF. This parameter translates a bottom sediment concentration to a fish tissue concentration. The science is evolving for this parameter, including thought on the extent to which BSAFs generated for one species at one site can be generalized to other sites and/or species, the differences in BSAF between column and bottom feeders, the differences between past and ongoing contamination, and so on. Users should carefully review the justification for the SOD values selected for the demonstration scenarios before using the same values.

3) **Site Specific:** These parameters should or can be assigned values based on site-specific information. The information provided on their assignment for the demonstration of methodologies in this document can be useful where site specific information is unavailable. A key class of site specific are the source strength terms - the soil concentrations, effluent discharge rates, and stack emission rates. Others include physical properties (organic carbon contents of soil and sediment, climate variables, areas, distances, and volumes) and parameters for bioconcentration algorithms (yields of vegetation, cattle raising practices, fish lipid contents).
4) **Exposure Parameters:** The exposure parameters have not been categorized as have the contaminant fate and transport/transfer parameters. Assignment of these values are critical as Lifetime Average Daily Dose (LADD) estimates are linearly related to parameter assignments - doubling exposure duration assumptions double LADDs, and so on. Some exposure parameters are appropriately described as first order defaults. These include: lifetime, body weights, water ingestion rates, inhalation rates, and an exposure duration for a childhood pattern of soil ingestion. All of the other exposure parameters are better described as either second order defaults or site specific parameters. All exposure parameters were developed based on information and recommendations in EPA's *Exposure Factors Handbook* (EPA, 1997b) and *Dermal Exposure Assessment: Principles and Applications* (EPA, 1992a).

The end products of the exposure assessment procedures presented in this document are estimates of potential dose expressed in mass (pg, ng, etc.) of dioxin-like compound/body weight (usually kg)-day. The procedures for converting these dose estimates to risk estimates, both cancer and non-cancer, are described in Chapter 2 and demonstrated in Chapter 5.

The scope of each chapter in Volume III is summarized below.

Chapter 2, Estimating Exposure and Risks, presents overall framework for conducting exposure assessments. It provides procedures for identifying exposure pathways, estimating contact rates and resulting exposure levels. Approaches for defining exposure scenarios are presented. Procedures for converting exposure dose to lifetime cancer risk estimates are provided, and procedures for evaluating non-cancer risk are also discussed.

Chapter 3, Evaluating Atmospheric Releases of Dioxin-Like Compounds from Combustion Sources, provides procedures to estimate the emission rates of dioxin-like compounds from combustion processes and further atmospheric transport modeling procedures from the stack to the surrounding land surface. This chapter describes and demonstrates the use of the ISCST3 model on a hypothetical incinerator and lists the associated atmospheric dispersion and deposition estimates from that model exercise.

Chapter 4, Estimating Exposure Media Concentrations, provides procedures for estimating concentrations of the dioxin-like compounds in exposure media (soil, air, water, biota) resulting from soil contamination, effluent discharges, and stack emissions.

Chapter 5, Demonstration of Methodology, develops hypothetical scenarios and generates exposure and risk estimates to demonstrate the methodologies of this document.

Chapter 6, User Considerations, discusses key issues for users of the methodologies. All model parameters are listed and categorized according to the scheme noted above. Sensitivity analysis is conducted on the algorithms estimating exposure media concentrations. An exercise on estimating the releases from a bounded area of soil contamination is presented. The purpose of this exercise is to determine whether a reservoir of soil contamination would be depleted prior to an assumed duration of exposure.

Chapter 7, Model Comparisons and Validations, presents extensive information aimed at gaining confidence and establishing credibility for the use of the fate models of this assessment to predict the fate, movement, and resulting exposure media concentrations near sources of dioxin release. One section of this chapter presents alternate fate models, and where possible, generates results from these models to compare with results from the models selected for this assessment. The second major section presents several model validation exercises, where the models are paramterized to predict exposure media concentrations, and the results are compared with appropriate real world observations.

Chapter 8, Uncertainty, discusses the sources and possible magnitude of uncertainty in the exposure assessment procedures. Uncertainty and variability of fate and transport, and exposure parameters, are discussed. Monte Carlo and similar numerical methods to quantify variability and uncertainty are discussed, and several literature examples of these types of exercises conducted for dioxin-like compounds are summarized.

1.5. EXECUTIVE SUMMARY OF VOLUME III

Volume III describes procedures for conducting site specific exposure and risk assessments to estimate potential dose, cancer, and non-cancer risks from exposure to dioxin-like compounds from a nearby source of release. Sections below summarize the key issues and results from each chapter of this volume.

1.5.1. Exposure Equation

A potential dose is defined as a daily amount of contaminant inhaled, ingested, or otherwise coming in contact with outer surfaces of the body, averaged over an individual's body weight and lifetime. The general equation used to estimate potential dose normalized over body weight and lifetime is as follows:

$$\text{Lifetime Average Daily Dose (LADD)} = (\text{exposure media concentration} \times \text{contact rate} \times \text{contact fraction} \times \text{exposure duration}) / (\text{body weight} \times \text{lifetime}) \qquad (1\text{-}2)$$

This procedure is used to estimate dose in the form needed to assess cancer risks. For non-cancer risks, an ADD term is instead derived. ADD is calculated as above except that exposure duration and lifetime are taken out of the equation above. Each of the terms in this exposure equation is discussed briefly below:

- **Exposure media concentrations:** These include the average concentrations in the media to which individuals are exposed. Media considered in this assessment include soil, air, water, vegetables/fruits, fish, beef, milk, and poultry.
- **Contact rate:** These include the ingestion rates, inhalation rates, and soil contact rates for the exposure pathways.
- **Contact fraction:** This term describes the distribution of total contact between contaminated and uncontaminated media. This assessment describes exposures which occur at homes, so the contact rate translates to time spent at home for air, soil, and water exposures, and fraction of total food product produced at home (vegetables/fruits, beef, milk, and poultry) or obtained recreationally (fish) from an impacted water body. This assessment assumes time at home fractions of 0.70 and 0.90 for central and high end scenarios, respectively, and home food production factors, or food contact fractions, of about 0.50 and less.
- **Exposure duration:** This is the overall time period of exposure, mostly pertinent to adult exposures. Central and high end durations of 9 and 30 years, respectively, are assumed in this assessment. Another exposure duration considered in this methodology is one associated with a childhood pattern of soil ingestion. The exposure duration in this case is 5 years.
- **Body weight:** For all the pathways, the human adult body weight of 70 kg is assumed. Although the United States population average is closer to 60 kg (EPA, 1997b), the value of 70 kg has been more traditionally used. The body weight for child soil ingestion is 17 kg.
- **Lifetime:** Following convention, and because cancer slope factors are derived based on a 70-year human lifetime, the average adult lifetime assumed throughout this document is 70 years.

1.5.2. Procedures for Evaluating Cancer and Non-Cancer Risk

Although the focus of the site-specific methodology is to estimate exposures to dioxins, procedures are also presented and demonstrated for estimating cancer and non-cancer risk. The usual procedure used to calculate an upper-limit incremental cancer risk is as follows:

$$R \;=\; 1 - e^{-q_1{}^{*}\ LADD} \;\approx\; q_1{}^{*}\ LADD \qquad \text{(1-3)}$$

when $q_1{}^{*}$ LADD < 10^{-3} and where $q_1{}^{*}$ is the 95% upper confidence limit of the linearized cancer slope factor of the dose-response function (expressed in inverse units of the dose quantity, such as kg-day/pg, or equivalently, $(\text{pg/kg-day})^{-1}$) and LADD is the dose (which needs to be in units appropriate to cancel those of $q_1{}^{*}$, pg/kg-day). This assessment uses the simplified $q_1{}^{*}$ LADD since the exposures and risks demonstrated are generally less than 10^{-3}. The slope factor, $q_1{}^{*}$, for 2,3,7,8-TCDD has been previously estimated by EPA as 0.000156 $(\text{pg/kg-d})^{-1}$ (EPA, 1984; 1981), but has been reevaluated as 0.001 $(\text{pg/kg-d})^{-1}$ in this Reassessment. Also, it is being applied to a TEQ dose in this Assessment.

This selected cancer slope factor was based primarily on the meta analysis of the human epidemiology studies where exposure was estimated from dioxin concentrations in blood in occupationally exposed cohorts. The dose estimates used to derive the slope factor were obtained by using a PK model to convert the blood concentrations to an administered, or potential, dose. An administered dose is defined as the dose which contacts the body boundary surfaces, such as the skin as in dermal exposure or the dose ingested prior to absorption. This administered dose was derived by first calculating an absorbed dose and then dividing by 0.8 - i.e., an absorbed dose was assumed to be 80% of an administered dose. Because the potency factor was derived based on an administered dose, the new slope factor can be applied to an administered dose without any adjustment for absorption as long as the absorption is approximately 80%. Although the data are limited, this is probably a reasonable assumption for most types of food ingestion and inhalation. For soil pathways, however, an additional adjustment factor has to be added to account for significantly less absorption. Data suggests that the dose of dioxins absorbed from soil ingestion is about 30% and about 3% from soil dermal contact. For soil dermal contact, an absorbed dose is already calculated; thus the dermal contact pathway yields an absorbed dose already. Therefore, for these two pathways, Equation (1-3) needs an additional adjustment factor equal to 0.3/0.8, or 0.375, for soil ingestion and 1.00/0.8, or 1.25, for soil dermal contact. A full discussion on absorption of administered dioxin through the various pathways can be found in Chapter 1. Disposition and Pharmacokinetics, of the Health Reassessment Document (EPA, 2000b).

To evaluate incremental non-cancer effects in a risk assessment, EPA uses established Reference Doses (RfDs) for most contaminants. An RfD is defined as an estimate, with uncertainty spanning perhaps an order of magnitude, of a daily exposure of the human population, including sensitive subgroups, that is likely to be without an appreciable risk of deleterious non-cancer effects during a lifetime. The incremental dose from a particular source is compared with the RfD. However, an RfD has not been established for dioxin-like compounds. It was concluded in the Risk Characterization that setting an RfD using traditional Agency approaches would result in an RfD that would likely be 2-3 orders of magnitude below current background intakes and body burdens. For this reason, EPA concluded that establishing an RfD for assessing the potential for non-cancer effects would not be helpful to risk managers. Instead, it was suggested in the Risk Characterization that risk managers compare the increment of exposure to a specific source with background exposures for assessing the potential for non-cancer effects. For this reason, a “ROIE”, or Ratio of Incremental Exposure, was developed. The ROIE was defined as a ratio between the increment of daily exposure from the source in question, the term defined by ADD as noted above, to background daily exposure. A ROIE equaling 1.0 suggests that the increment of exposure from the source in question is equal to background exposure.

Background exposure could be considered to be a national average background exposure, or it could a quantity specifically developed for the site in question. The latter would be appropriate if certain behaviors, such as subsistence fishing or farming, or certain environmental conditions, such as a high local background of dioxin in soil or air due to industrial practices, were present in the vicinity of the specific source being evaluated.

1.5.3. Procedure for Estimating Exposure

Before making exposure estimates, the assessor needs to gain a more complete understanding of the exposure setting and the contamination source. The approach used for this assessment is termed the exposure scenario approach. There are 7 steps in this approach:

Step 1. Identify Source: Three principal sources are addressed in this document: contaminated soils, stack emissions, and effluent discharges.

Step 2. Estimate Release Rates: Estimating the release of contaminants from the initial source is the first step towards estimating the concentration in the exposure media. Releases from soil contamination include volatilization, and wind and soil erosion. Stack emissions and effluent discharges are point source releases into the environment.

Step 3. Estimate Exposure Point Concentrations: Contaminants released from soils, emitted from stacks, or discharged into surface waters move through the environment to points where human exposure may occur, and/or to impact environmental media to which humans are exposed. Various fate, transport, and transfer models are used to predict exposure media concentrations given source releases.

Step 4. Characterize Exposed Individuals and Exposure Patterns: Exposed individuals in the scenarios of this assessment are individuals who are exposed in their home environments. They are residents who breathe air at their residence, fish recreationally, have a home garden, farm, and are children ages 2-6 for the soil ingestion pathway. Exposures which occur at the workplace or other locations are not discussed in this assessment, although the procedures could be adapted for other exposure sites. Each pathway has a set of exposure parameters including contact rates, contact fractions, body weights, exposure durations, and a lifetime. An individual's total exposure is the sum of the exposures from individual pathways.

Step 5. Put It Together in Terms of Exposure Scenarios: A common framework for assessing exposure is with the use of "settings" and "scenarios." Settings are the physical aspects of an exposure area and the scenario characterizes the behavior of the population in the setting and determines the severity of the exposure. A wide range of exposures are possible depending on behavior pattern assumptions. An exposure scenario framework offers the opportunity to vary any number of assumptions and parameters to demonstrate the impact of changes to exposure and risk estimates.

Step 6. Estimate Exposure: The end result of having followed the above 5 steps are estimates of individual exposures to a characterized source of contamination.

Step 7. Assess Uncertainty: Uncertainties should be considered when applying procedures in this document to a particular site. Pertinent issues explored in this assessment include: 1) model predictions of exposure media concentrations compared to field measurements in a series of model validation exercises, 2) similarities and differences for alternate models for estimating exposure media concentrations, 3) sensitivity of model results to a range of values for methodology parameters, 4) mass balance checks, and 5) qualitative and quantitative discussions on the uncertainties with the model parameters and exposure estimates generated for the demonstration scenarios.

1.5.4. Estimating Exposure Media Concentrations

Literally hundreds of fate and transport models have been published which differ widely in their technical sophistication, level of spatial or temporal resolution, need for site specific parameterization, and so on. This makes selection of the most appropriate one for any particular situation very difficult. For this assessment, relatively simple, screening level models are used to model fate, transport, and transfer of dioxin-like compounds from the source to the exposure media. Simple assumptions are often made in order to arrive at the desired result, which is long-term average exposure media concentrations. Perhaps the most critical of the assumptions made is that the source strength remains constant throughout the period of exposure.

It is important to understand that EPA is not endorsing the algorithms of this assessment as the best ones for use in all dioxin assessments. They are suggested as reasonable starting points for site-specific or general assessments. All assumptions for the models and selection of parameter values are carefully described. If these assumptions do not apply to a particular situation, or where assessors require more spatial or temporal resolution, more complex models should be selected. Finally, it cannot be overemphasized that measured concentrations are generally more reliable than modeled ones. Assessors should use measured concentrations if available and if such measurements can be considered spatially and temporally representative for the exposed populations.

Chapter 4 provides algorithms used to evaluate the fate, transport, and transfer of dioxin-like compounds from contaminated soil, stack emissions, and effluent discharge. These three sources of dioxin release are referred to as "source categories" in this document. Algorithms are presented which link each of these sources to estimated concentrations in a number of media which may be contaminated as a result, and are therefore potential "exposure media": 1) surface soils, 2) surface-water associated media: suspended and bottom sediment and dissolved phase concentrations, 3) air including the vapor phase and in particulate form, and 4) biota including beef, milk, poultry, fruit and vegetables, and fish. The remainder of this section describes how each potential exposure medium can be affected by each source, and the algorithms used to make this link.

- **Surface soils:** Exposure to contaminated soil may be a result of direct contact with soil on the site of the contamination, or indirectly after the contaminated soil has been transported off the site of contamination and onto a nearby site of exposure. These cases are termed "on-site": the site of contamination and the site of exposure are the same, and "off-site": the site of exposure is distant from the site of contamination. In either case, soil concentrations are specified for the contaminated source. For the off-site case, dioxins reach the site of exposure via erosion. Mixing of contaminated and uncontaminated exposure site soil is into either a "tilled" 20-cm depth or a "non-tilled” 2-cm depth. The tilled concentrations are used to estimate concentrations in underground vegetables, and for outdoor dermal contact. The non-tilled concentrations are used for indoor dermal contact events, for childhood soil ingestion in residential and farm settings, and for cattle soil ingestion (used in estimation of beef and milk concentrations).

Exposure site soils can also be impacted from stack emissions due to air transport of particle-bound dioxins from the stack to the exposure site. Deposition modeling for dioxin-laden particles allows for estimation of tilled and non-tilled soil concentrations. When stack emissions are the source, the nontilled depth of mixing is again assumed to be 2 cm.

A key assumption for evaluating the exposure site as a result of both off-site erosion and stack emissions is that contaminants impact a thin layer of soil and do, in fact, dissipate; no dissipation is assumed if the site of contamination is also the site of exposure. A soil dissipation half-life of 25 years is assumed for all dioxin-like congeners for the shallow 2 cm depth and 100 years is assumed for the residues tilled to 20 cm.

- **Surface Water:** The principal assumption driving the solutions for the soil and stack emission source categories is that the suspended and bottom sediments of water bodies originate as watershed soils, which are subsequently eroded. For the stack emission source category, a portion of the sediments also originates from directly-depositing dioxins. The process of erosion transports soils within the watershed to the water body. Unit rates of erosion along with watershed size determine the total potential amount of soil which could be delivered to the water body. Sediment delivery ratios reduce that potential amount. A mass balance assures that soil eroding on an annual basis becomes either suspended or bottom sediment within an annualized volume of surface water. "Enrichment" of eroded soil is assumed, which means that eroded soil from a contaminated source is assumed to be higher in concentration of dioxin-like compounds than *in situ*, off-site soils. Once in the water body, a standard partitioning model based on the organic carbon partition coefficient, Koc, determines the concentration of contaminant in the water in truly dissolved form and the concentration on suspended sediments. The organic carbon normalized concentrations of suspended and bottom sediment are assumed to be equal. Watershed soil concentrations are model input parameters for determining the effect on surface water from contaminated soils. For stack emissions, a total (dry + wet) deposition rate of contaminant which represents average depositions onto the watershed is specified as an input parameter, as well as a mixing depth representing the watershed. In this way, average watershed soil concentrations are calculated for the stack emission source category.

For effluent discharges as sources, watershed soils are not considered. An amount of contaminant is discharged into an annual flow volume to obtain a simple dilution concentration. This total concentration is partitioned into a truly dissolved phase and a phase sorbed to suspended sediments using the organic carbon partition coefficient, the Koc. Bottom sediments are not considered for effluent discharges.

- **Soil to Air:** From contaminated soils, residues become airborne via the processes of volatilization and wind erosion. For on-site soil contamination, these vapor and particle phase fluxes are translated to ambient air concentrations using a near-field dispersion model. For the off-site scenario, the same approach is used to estimate ambient air exposure site concentrations, except that a far-field dispersion model is used. These airborne reservoirs are the basis for inhalation exposures, and are also used to estimate plant concentrations for vegetable ingestion and in grass and feed for estimating beef and milk concentrations.

- **Stack emissions and atmospheric transport modeling :** Air dispersion/deposition models consider the basic physical processes of advection, turbulent diffusion, and removal via wet and dry deposition to estimate the atmospheric transport, resulting ambient air concentration, and settling of particles. The ISCST3 model is used for air dispersion and deposition modeling. Besides discussions in Chapter 3 on theoretical underpinnings and parameter assignment, further discussions on the ISCST3 model can be found in EPA (1995b).

Application of the ISCST3 model follows these steps:

Step 1. Emission factors: The first step in the use of the ISCST3 model is to determine "emission factors" for dioxin-like congeners. These factors are defined as the µg (or other mass unit) congener emitted per kg (or other mass unit) feed material combusted. Once assuming a rate of feed material combusted in appropriate units, kg/day, these emission factors can be translated to the units appropriate for atmospheric transport modeling, µg/sec. This assessment promotes the generation of specific congener emission factors, rather than emission factors for TEQ or homologue groups. Emission factors for the demonstration of stack emission sources in Chapter 5 were generated from actual test data from an incinerator burning organic wastes (source otherwise unspecified). Emission estimates for this example incinerator are similar to emissions that are known to be emitted from combustors employing sophisticated air pollution control devices (e.g., scrubbers combined with fabric filters).

Step 2. Vapor/particle (V/P) partitioning: The second step in atmospheric transport modeling is to determine the percent of totally emitted dioxin-like congener which is in a vapor phase, and the percent which is in the particle phase. The partitioning of stack emissions into these two phases was examined by reviewing stack testing data, ambient air sampling data, and a theoretical approach developed in Bidleman (1988). From this review, it was generally concluded that the most appropriate representation of partitioning of dioxins for purposes of fate modeling and exposure assessing was provided by the modeling approach, and the V/P partitioning scheme for dioxins and furans shown in Table 1-4 is the one adopted for this Assessment.

Step 3. Two runs of the ISCST3 model: In order to provide estimates of vapor and particle phase concentrations of dioxin-like compounds, as well as estimates of wet/dry particle deposition flux, it is necessary that to run the ISCST3 model twice. Both model runs should assume a "unit emissions release rate", e.g., 1 g/s. Results from these unit runs can easily be transformed to final outputs given the total emission rate of the congener and vapor/particle partitioning. A vapor phase run involves turning wet/dry deposition switches to the "off" position. This inactivates a plume depletion equation that subtracts out losses in ambient air concentration due to particle deposition. What is left are the Gaussian dispersion algorithms. The vapor phase concentrations are used for inhalation exposures and also for vapor transfers onto vegetation for food chain modeling. A second run of ISCST3 with wet/dry deposition switches turned to the "on" position is considered a simulation of particle-bound contaminant. Outputs from this run include wet and dry deposition rates, and air concentrations of contaminants in the particulate phase. The depositions are used in soil and food chain modeling, and the concentrations are added to the vapor phase concentrations from the first ISCST3 run to arrive at the total air-borne reservoir for inhalation exposures.

- **Biota:** Simple bioconcentration/biotransfer approaches are used to estimate biota concentrations in this assessment. Specifics for each biota considered are:

1. Fish - The soil contamination and stack emission source categories estimate the concentration of contaminant on bottom sediments of water bodies. A fish lipid concentration is estimated based the organic carbon normalized bottom sediment concentration and a BSAF, or Biota Sediment Accumulation Factor. Whole fish concentrations for exposure estimation then equal this lipid concentrations times a whole fish lipid content (or a fillet lipid content). For the effluent discharge source category, fish lipid concentrations are estimated as a function of organic carbon normalized concentrations and the closely related BSSAF, or Biota Suspended Solids Accumulation Factor. This recently introduced bioaccumulation factor (EPA, 1993) is analogous to the BSAF, and it is suggested in EPA (1993) that, as a first estimate, it take on the same chemical-specific numerical value as the BSAF.

2. Vegetation - Concentrations in three types of vegetation are considered in this assessment: below ground vegetables (carrots, potatoes, e.g.), above ground vegetables/fruits (tomatoes, apples), and above ground grass and cattle feed which are required for estimation of beef and milk concentrations. Assumptions critical to all three include: above ground vegetation is impacted by vapor phase transfers and particle deposition - there is no root to shoot translocation, outer portions of the vegetation are only impacted with minimal within plant translocation, particle bound contaminants deposit onto and mix in a vegetative reservoir and are subject to a fourteen-day dissipation half-life which represents particle washoff, and vegetables/fruits which have an outer protective layer (peas, citrus e.g.) are unimpacted by dioxin-like compounds. Below ground vegetable concentrations are estimated from soil water concentrations and a Root Concentration Factor, or RCF. Above ground concentrations in plants due to atmospheric vapor concentrations are modeled using a "biotransfer" approach, where the vapor concentrations are simply multiplied by an air-to-leaf transfer factor, B_{vpa}, and a surface area to volume reduction factor, VG, which is equal to 1.00 for grasses and other leafy vegetation and less than 1.00 for bulky vegetation. This B_{vpa} was found to be one of the most critical parameters for not only vegetation concentration modeling (i.e., above ground vegetations were found to be dominated by vapor transfers over particle phase depositions), but for subsequent terrestrial animal food chain models. The B_{vpa} was developed for the dioxins in a field calibration exercise.

3. Beef and Milk - Weighted average concentrations of dioxin-like compounds in the diets of cattle raised for beef or lactating cattle are multiplied by a congener-specific bioconcentration factor, BCF, which yields the concentrations in the fat of beef or milk. The same congener-specific BCF is used for beef and milk. This presumes that dioxin-like compounds bioaccumulate equally in body fat and milk fat of beef and dairy cattle. The difference between the two food products is mostly a function of the diets presumed for beef cattle and lactating cows. A set of BCFs for all dioxin-like congeners for this assessment were based on a set of data on a lactating cow (i.e., dietary intakes of dioxin congeners, concentrations in milk, and other pertinent quantities; McLachlan, et al., 1990). A later feeding experiment by Fries, et al. (1999) found BCFs very similar to the McLachlan, et al. (1990) single cow BCFs adopted for this assessment. Beef and dairy cattle diets are described in terms of proportions in pasture grass, cattle feed (silage, grains), and soil. Models described above estimate concentrations in these cattle intakes.

4. Chicken and Eggs - The algorithm to estimate the concentration of contaminant in chicken and/or eggs is essentially the same algorithm as in beef/milk above: the concentration in the lipid of chicken/eggs is a function of the weighted average concentration in the chicken diet (comprised of vegetation and soil) and chicken/egg bioconcentration factors. The experiments used to develop the chicken and egg bioconcentration factors were conducted by the Hazardous Materials Laboratory at the California EPA (Stephens, et al. 1995). Three key differences in the application of the chicken/egg bioconcentration model and beef/milk model were: 1) data was available and robust enough to assign different bioconcentration factors for chicken and eggs, 2) chickens, both layers and non-layers, were assumed to "free range" in the demonstrations of this pathway in Chapter 5, which translated to a higher exposure to soil in their diet - 10% for free range chickens vs. 4% for beef cattle and 2% for dairy cattle, and 3) based on information in

Stephens, et al. (1995), chicken feed was assumed to originate from protected vegetation and was therefore assumed to be dioxin-free.

1.5.5. Demonstration of Methodology

EPA (1992b) states, "In exposure scenario evaluation, the assessor attempts to determine the concentrations of chemicals in a medium or location and link this information with the time that individuals or populations contact the chemical. The set of assumptions about how this contact takes place is an exposure scenario." These assumptions can be made many different ways producing a wide variety of scenarios and associated exposure levels. The number of people exposed at different levels form a distribution of exposures. Ideally assessors would develop this entire distribution to fully describe the exposed population. Since the necessary information for developing a population distribution is rarely available, EPA (1992b) recommends developing a central and high end scenario to provide some idea of the possible range of exposure levels.

1.5.5.1. Description of the Demonstration Scenarios

The basic setting for which the methodologies are demonstrated is a rural setting which contains both farms and non-farm residences. The three principal sources of contamination, soil, stack emissions, and effluent discharges, are assumed to exist in such a setting. "Central" scenarios are based on typical behavior at a residence and "high end" scenarios are comprised of a farm family that raises a portion of its own food. Key distinguishing features between the high end and central scenarios include: 1) individuals in high end scenarios are assumed to be at their home a greater proportion of the day than the central scenarios (which impacts assignment of contact fraction), 2) individuals in high end scenarios are exposed to impacted beef from cattle which they raise on their farm while these exposures are not considered for the central scenarios, 3) in contrast, individuals in the central scenario recreationally fish, 4) the exposure duration for individuals in the high end scenario is 30 years compared to 9 years for the central scenario, and 5) certain exposure parameters, such as water ingestion rate which is 1.4 L/day for the central scenarios and 2 L/day for the high end scenario, are different.

The example scenarios were carefully crafted to be plausible and meaningful, considering key factors such as source strength, fate and transport parameterization, exposure parameters, and selection of exposure pathways. However, it should be clearly understood that the purpose of the demonstration scenarios is to provide users of this methodologies with a comprehensive example of their application. The demonstration exposure scenarios were:

Scenarios 1 and 2: Background conditions, Residence and Farm

Surface soils within the watershed are initialized to soil concentrations of the 17 dioxin-like congeners (no dioxin-like PCBs) which have been found in an actual rural setting. Also, air concentrations of the 17 congeners are initialized to air concentrations which have been found in this same rural setting. Scenario 1 is the central residential scenario, and Scenario 2 is the high end farming scenario. The exposure pathways for Scenario 1 are: water ingestion, air inhalation, fish ingestion, fruit/vegetable ingestion, soil dermal contact, and soil ingestion. The exposure pathways for Scenario 2 are: water ingestion, air inhalation, beef ingestion, fruit/vegetable ingestion, soil dermal contact, and soil ingestion. It is noted that for a background condition, it could be argued that all exposure is to background concentrations in exposure media. In other words, all contact fractions would be 1.00. However, if an assessor wished to compare the incremental impacts from a specific source of dioxin release with impacts an individual would receive by contact with the same exposure media which has only background concentrations of dioxins, than the assessor would assume all the same exposure behaviors (rates of contact, contact fractions). This demonstration takes this approach.

Exposure Scenario 3: Soil Contamination, Farm

A 40,000 m^2 rural farm is located 150 m (500 ft roughly) from a 40,000 m^2 area of bare soil contamination; an area that might be typical of contaminated industrial property. The surface soil at this property is contaminated with three example dioxin-like compounds to the same concentration of 1 part per billion (ppb). These compounds are: 2,3,7,8-TCDD, 2,3,4,7,8-PCDF, and 2,3,3',4,4',5,5'-HPCB. The 1 ppb soil concentration is reasonable for industrial sites of contamination of dioxin-like compounds, and generally about three orders of magnitude higher than the concentrations of these congeners in background settings. As in the above and all scenarios, bottom sediment in a nearby river is impacted, which impacts the water and fish. The exposure pathways include: water ingestion, air inhalation, beef ingestion, fruit/vegetable ingestion, soil dermal contact, and soil ingestion.

Exposure Scenarios 4 and 5: Stack Emissions, Residence and Farm

A 4,000 m^2 rural residence (Scenario 4) is located 5000 meters from an incinerator, and a 40,000 m^2 (Scenario 5) rural farm is located 500 meters downwind from an incinerator. Emission data of the suite of 17 dioxin-like dioxin and furan congeners (no dioxin-like PCBs) is available from stack testing of an actual incinerator. This allows for estimation of impacts from each congener individually, and estimation of WHO$_{98}$-TEQ$_{DF}$ impacts. The exposure pathways for Scenario 4 are: water ingestion, air inhalation, fish ingestion, fruit/vegetable ingestion, soil dermal contact, and soil ingestion. The exposure pathways for Scenario 5 are: water ingestion, air inhalation, beef ingestion, fruit/vegetable ingestion, soil dermal contact, and soil ingestion.

Exposure Scenario 6: Effluent Discharge into a River

Exposure parameters associated with central behaviors for the water and fish ingestion pathways were chosen to demonstrate this source category. The source strength was developed from data on pulp and paper mill discharges of 2,3,7,8-TCDD. The discharges of the other two example compounds are assumed to be the same for purposes of demonstration. Obviously, however, there is less of a tie to real data for the discharge rate for these other two example compounds.

Food pathway analyses outside of the scenario framework: The food consumption pathways of fish, milk, chicken, and eggs are demonstrated using source strength characteristics of the three high end scenarios above: Scenarios 2 (background conditions), 3 (soil contamination), and 5 (stack emission). These food pathways were not modeled in the scenarios themselves. In these analyses, exposure media concentrations are calculated for each source and the pathway exposure estimates are provided. The purpose of these external pathway analyses was to provide further demonstration and to compare impacts from the various food pathways where methodologies have been provided in this assessment.

1.5.5.2. Results from the Demonstration Scenarios

For brevity, only a subset of results from the demonstrations will be summarized. Table 1-5 gives the exposure media concentrations for WHO_{98}-TEQ_{DF} for Example Scenarios #1 and #2, demonstrating a background setting, and Scenarios #4 and #5, the central and high end scenarios for the stack emission source category. Table 1-6 gives the estimated Lifetime Average Daily Doses, LADDs, and the cancer risk for Scenarios #2 and #5, the high end scenarios for the background setting and stack emission source.

Much of the differences between exposure pathways and scenarios is due to differences in exposure media estimation. Following are some of the observations on exposure media concentrations, LADDs, cancer and non-cancer risks for the background and stack emission scenario demonstrations:

1) Concentrations in environmental and exposure media for the stack emission central and high end scenario were about 3 and 2 orders of magnitude lower than the central and high end scenarios demonstrating background conditions, respectively. For example, the background WHO_{98}-TEQ_{DF} air concentration was 0.021 pg/m^3. In contrast, the WHO_{98}-TEQ_{DF} air concentration for the stack emission source was 2 orders of magnitude lower at 500 meters from the stack, at 0.00024 pg/m^3, and was lower still at 5000 meters from the stack, at 0.000085 pg/m^3. This suggests that the example stack emission source, which was a single emission source with a high level of pollution control, would contribute little to overall background exposure levels.

2) For both the background scenarios, 1 and 2, and the stack emission scenarios, 4 and 5, WHO_{98}-TEQ_{DF} soil concentrations were over an order of magnitude higher than 2,3,7,8-TCDD concentrations. The difference in 2,3,7,8-TCDD and WHO_{98}-TEQ_{DF} impacts to all media mirrors the difference in stack emissions of 2,3,7,8-TCDD and stack emissions of WHO_{98}-TEQ_{DF}. This trend in differences between 2,3,7,8-TCDD and TEQ impacts occurs in all exposure media estimations for both the background scenarios and the stack emission scenarios.

3) Within each demonstration scenario, there appears to be a reasonably narrow range of predicted lipid concentrations among beef, milk, chicken, and egg fat. The difference is about a factor of 3 to 4. The lowest concentrations are noted for the stack emission demonstration scenarios, in the 10^{-3} to 10^{-2} pg WHO_{98}-TEQ_{DF}/g (ppt) range, lipid basis. The background concentrations were next highest, about two orders of magnitude higher in the 10^{-1} to 10^{0} ppt range. The average concentration of WHO_{98}-TEQ_{DF} in lipids of terrestrial animal food products measured in US food products, as described in Volume II of the Exposure Reassessment Document, is similar to the background concentration predictions. This is not unexpected since concentrations of dioxins in terrestrial food animals in the background scenarios were modeled based on a profile of dioxins and furans found in air in an actual rural setting.

4) Table 1-6 shows the percent of total scenario exposure which is accounted for by each pathway. The total scenario LADD was calculated simply as the sum of the pathway LADDs in the scenario, without accounting for any differences in body absorption. As discussed above, inhalation and food/water ingestion pathways have an absorption in the range of 80%, soil ingestion has an absorption of 30%, and soil dermal contact has absorption already considered, so LADD estimates are already at 100% absorption. From Table 1-6, it is seen that the beef pathway dominates the scenarios. For the central scenarios (not summarized in this section), which included fish ingestion but not beef or milk ingestion, the fish ingestion dominated. Interestingly, the beef ingestion pathway LADD was over an order of magnitude higher than the fish ingestion pathway LADD. This was more due to differences in the exposure parameters including the ingestion and contact rates, and the differences in the lipid content of the full product, rather than lipid concentrations themselves since the fish lipid concentrations tended to be higher than the beef lipid concentrations for a given source.

5) Differences between analogous "central" and "high end" exposure pathway estimates for the background demonstration scenarios, 1 and 2, were near or less than an order of magnitude (inhalation exposure for the central background scenario and the inhalation exposure for high end on-site scenario are analogous exposures). This is because the exposure parameters used to distinguish typical and high end exposures, the contact rates, contact fractions, and exposure durations, themselves did not differ significantly, and these were the only

distinguishing features for analogous pathways in the background demonstrations. For the total exposure, however, there was a difference of a factor of 20 between high end and central exposure in the background demonstration scenarios. This is because the high end scenario included consumption of beef, which was the highest exposure pathway and exceeded the fish pathway of the central scenario by over an order of magnitude.

6) In the stack emission scenarios, placing exposed individuals either 500 or 5000 meters away from the incinerator did significantly impact the results. The order of magnitude difference in distance added about an order of magnitude difference in exposure media concentrations and hence LADD estimates. Therefore, the full difference in analogous pathways between the central and high end was closer to 2 orders of magnitude for the stack emission demonstration scenarios.

7) The LADD of 0.093 WHO_{98}-TEQ_{DF}/kg-day for the background high scenario is about an order of magnitude lower than the adult background dose of 0.64 pg WHO_{98}-TEQ_{DF}/kg-day generated in Volume II, Chapter 4. The reasons for this difference are: 1) the Volume II background exposure estimate was an average daily dose, ADD, not an LADD calculated in the demonstration scenarios here. The LADD estimated in this chapter assumes 30 years of exposure. The ADD during the exposure period would be just over twice, or 70/30, as high as the LADD; 2) the Volume II background exposure estimate considered additional pathways including fish, dairy ingestion (milk and otherwise), eggs, pork, and poultry. If one adds the additional pathways for the background high scenario - milk, chicken, egg, and fish, the LADD (and ADD) roughly doubles; 3) the exposure factors are different, with the most important difference being that in the exposure scenarios considered in this chapter, contact fractions of less than 1.0 were assumed - less than 0.5 for the terrestrial animal pathways, in fact.

Some of these differences between the Volume II background exposure estimate and the LADD estimates for the background high scenario also are relevant for the procedures demonstrated here to characterize non-cancer risk. As described earlier, a “ratio of incremental exposure”, or ROIE, is the current recommended approach for evaluating non-cancer risk. This is defined as the ratio of the incremental dose due to the source being evaluated and the background dose, multiplied by 100%. The background dose can be a generic US background dose, or a site-specific dose. The generic adult background dose of WHO_{98}-TEQ_{DF}, as noted above, is 0.64 pg TEQ/kg-day. A background dose for the specific site being evaluated here has not yet been developed. All of the exposures and risks displayed in previous tables assumed a less-than-lifetime exposure, a limited set of exposure pathways, and contact fractions less than 1.0 (meaning that a fraction of their total consumption was from home-produced and impacted food). Producing a site-specific background exposure requires an assessor to estimate the total exposure of an individual (or individuals) to dioxins if the nearby source were not in existence. In that circumstance, the family would be still be consuming home produced foods. But they would also be consuming store bought or restaurant bought foods. The "total" exposure would include all pathways considered in the scenarios of this chapter, but other pathways as well.

For the purposes of the demonstration in Chapter 5, it was assumed that the farming family in the background scenario consume foods at similar rates whether or not they are consuming home produced or store bought food products, and that their exposure is characterized by all the pathways in the formal demonstration scenarios, as well as the additional scenarios that were demonstrated in Chapter 5, including milk, chicken, eggs, and fish. To estimate their average background daily dose over a lifetime, the exposure duration will increase from 30 to 70 years, and the contact fractions will all rise to 1.00. The resulting daily exposure is 1.16 pg WHO_{98}-TEQ_{DF}/kg-day. This 1.16 pg WHO_{98}-TEQ_{DF}/kg-day will be used here as the "site-specific background dose" against which one can develop ROIEs for the incinerator source.

As discussed in Chapter 2, a pertinent issue for generation of ROIEs is the use of LADDs or ADDs. Chapter 2 recommends the use of ADD for ROIEs, but this demonstration will show a ROIE calculation for both LADD and ADD. The total LADD for the stack emission high end scenario, as displayed in Table 1-7, is $1.01 * 10^{-3}$ pg WHO_{98}-TEQ_{DF} /kg-day. The ADD can be simply calculated as this LADD times 70/30, or $2.36 * 10^{-3}$ pg WHO_{98}-TEQ_{DF}/kg-day. The ROIEs are then easily calculated as:

using LADD: $[(1.01*10^{-3})/(1.16)] * 100\% = 0.09\%$
using ADD: $[(2.36*10^{-3})/(1.16)] * 100\% = 0.20\%$

As seen by these two calculations, the ROIE is less than 1%. If the generic US background dose of 0.64 pg WHO_{98}-TEQ_{DF}/kg-day were used instead of the site-specific background dose of 1.16 pg WHO_{98}-TEQ_{DF}/kg-day, the ROIEs would not be significantly different: 0.16% using LADD and 0.37% for ADD.

1.5.6. Sensitivity Analysis

This section discusses three sensitivity analysis issues pertinent to use of the site-specific methodologies promoted in this document: 1) the appropriate use and categorization of model parameters, 2) a sensitivity analysis exercise on the parameters required for algorithms estimating exposure media concentrations, and 3) the issue of mass balance with regard to the source strength terms of the four source categories.

1.5.6.1. Categorization and Use of Model Parameters

Table 6.1 in Chapter 6 lists all the parameters, including names, definitions, and units, that are required for the methodologies of this assessment except the exposure parameters. Exposure parameters are given in Table 2.1 in Chapter 2. Table 6.1 also gives four additional pieces of information for each parameter listed. Three are numerical values which were used in the sensitivity analysis exercises that are described below. One of those parameters is labeled "selected", which were the ones used in the demonstration exposure scenarios. High and low values of parameters selected for sensitivity analysis were carefully developed and might be considered a reasonable range of values for other uses of the methodology (with obvious exceptions such as areas of contamination, distances from contaminated to exposure site, and so on). The fourth piece of information is a qualitative judgement on the part of the authors of this document as to the appropriateness of using the "selected" parameter values for other assessments. This judgement is categorized in three ways:

1) **First Order Defaults:** As defaults, these parameters are independent of site specific characteristics. As first order defaults, it is felt that the values selected for the demonstration scenarios carry a sufficient weight of evidence from current literature such that these values are recommended for other assessments. Several of the chemical specific parameters, such as the Henry's Constant, H, and the organic carbon partition coefficient, Koc, fall into this category. The qualifier above, "current literature", indicates that new information could lead to changes in these values.

2) **Second Order Defaults:** Like the above category, these parameters are judged to be independent of site specific characteristics. However, unlike the above category, the current scientific weight of evidence is judged insufficient to describe values selected for demonstration purposes as first order defaults. Parameters of principal note in this category are the bioconcentration parameters specific to the chemicals, such as the Biota Sediment Accumulation Factor, or BSAF. This parameter translates a bottom sediment concentration to a fish tissue concentration. Users should carefully review the justification for the SOD values selected for the demonstration scenarios before using the same values.

3) **Site Specific:** These parameters should or can be assigned values based on site-specific information. The information provided on their assignment for the demonstration scenarios, and for selection of high and low values for sensitivity analysis testing, is useful for determining alternate values for a specific site. A key class of SS parameters which are the source strength terms - the soil concentrations, effluent discharge rates, and stack emission rates. If users are unable to obtain site-specific information, or their use of the methodologies is for general purposes, they should review the justification for selection of values for methodology

demonstration, as well as information provided giving ranges of likely values for model parameters.

The exposure parameters can be categorized as have the contaminant fate and transport/transfer parameters. Assignment of these values are critical as LADD estimates are linearly related to parameter assignments - doubling exposure duration assumptions double LADDs, and so on. Some of the exposure parameters are appropriately described as first order defaults. These include: lifetime, body weights, water ingestion rates, inhalation rates, and an exposure duration for a childhood pattern of soil ingestion. All of the other exposure parameters are better described as either second order defaults or site-specific. All exposure parameters were developed based on information and recommendations in EPA's *Exposure Factors Handbook* (EPA, 1997b) and *Dermal Exposure Assessment: Principals and Applications* (EPA, 1992c). Attaining site-specific information is recommended for exposure parameters.

1.5.6.2. Sensitivity Analysis

Sensitivity analysis was undertaken in order to evaluate the impact to exposure media concentration estimations with changes in fate and transport/transfer model parameters. Figure I-5 shows an example of sensitivity analysis conducted. This figure describes the impact of key factors for the stack emission source category for determining impacts to beef. The x-axis contains the names of the parameters evaluated. The key below the figure gives the definition of the parameters and the values selected for the demonstration scenarios. The y-axis shows the fractional change to the key model result, in this case, beef fat concentrations, to the changes made in the parameter. These actual changes in model predictions are noted above and below the bars. For example, beef fat concentrations increase by about a factor 2 with a 10-fold increase in the soil concentration. In contrast, there appears to be almost a linear relationship between an increase in vegetation concentration with beef fat concentration, as shown by the sensitivity test displayed next to the soil concentration test. There, a 10-fold increase in grass and other feed concentration resulted in a 10-fold increase in beef fat concentration. This was not the same trend that was tested for the soil contamination source category. There, soil dioxin provides the source term for vegetation and cattle impacts, not the air source of the stack emission category. In the soil contamination source category, the soil-to-cattle pathway dominates the prediction of beef fat concentration, whereas in the stack emission source category, the air-to-plant-to-cattle dominates. The right side of Figure 1 shows how the beef fat predictions are effected by changes in the assumptions regarding how the beef cattle are exposed, as expressed in the exposure parameters BCSDF (beef cattle soil diet fraction), BCFDF (beef cattle feed diet fraction), and the others listed in Figure 1-2. When....

Following are key overall observations from the several sensitivity analysis exercises undertaken in Chapter 6:

1) **Source terms are the most critical for exposure media impacts.** Source terms include soil concentrations, stack emission rates, and effluent discharge rates. In all cases, the impact to exposure media is linear with changes to source terms. Proximity to the source term can be important as well, as demonstrated with differences in distance from the stack emission source.

2) **Chemical-specific parameters, particularly the bioconcentration/biotransfer parameters, are the second most critical model inputs.** Some of these have lesser impacts within the range tested, such as the organic carbon partition coefficient, Koc, for surface water impacts. Generally, at least an order of magnitude in range in possible media concentrations is noted with the range of chemical-specific parameter ranges tested. The impact of changes to bioconcentration/biotransfer parameters is mostly linear. This is because these transfer factors estimate media concentrations as a linear transfer from one media to another. For example, fish lipid concentrations are a linear function of the organic carbon normalized concentration of contaminants in sediments. These transfer parameters are also identified as uncertain parameters. Tested ranges sometimes spanned over an order of magnitude for 2,3,7,8-TCDD.

3) **All other parameters had less of an impact as compared to source strength and chemical specific parameters; nearly all impacts were within an order of magnitude for the range of tested values.** Part of the reason for this trend is that there is a reasonably narrow range for many of the non-chemical specific or source term parameters - soil properties, wind speeds, vegetation yields, and others.

4) **The sensitivity analysis exercises unearthed a dichotomy in model performance, and likely therefore behavior in the real world, when soil is the source of dioxins as compared to when stack emissions are the source of dioxins.** The on-site soil source category was demonstrated with a 1 ppt soil concentration of 2,3,7,8-TCDD, a concentration similar to measured concentrations of 2,3,7,8-TCDD in rural settings. Air concentrations are estimated to be $4*10^{-5}$ pg/m^3 (vapor+particle phases summed). Atmospheric transport modeling in the demonstration of the stack emission source category resulted in an exposure site air concentration (vapor+particle phases summed also) at 500 meters from the stack to be $1*10^{-5}$ pg/m^3. With similar air concentrations predicted to occur at the exposure site for the demonstration of the soil and stack emission categories, one might hypothesize that all subsequent impacts would be similar. That was not the case. The stack emission source algorithms deposited particulates onto soil to estimate a soil concentration that was in the 10^{-3} ppt range for the 1-cm untilled depth and the 10^{-5} range for the 20-cm tilled depth. This

compares to the 1 ppt concentration for the on-site soil source category demonstration. With similar air concentrations but a 3+ order of magnitude difference in soil concentrations in the demonstration of the soil and the stack emission sources, the following trends were noted: 1) below ground vegetables had much higher concentrations for the soil source demonstration scenario; 2) soil-related exposures (dermal contact and soil ingestion) were much higher for the soil source demonstration scenario; 3) soil was significantly more critical in predicting beef and milk fat concentrations in the soil source category. The following shows the relative impact of soil versus vegetation (grass and cattle feed) for the on-site soil demonstration and the stack emission demonstration:

	Percent impact due to ingestion of:		
Description	Soil	Grass	Feed
Soil contamination, beef	90	7	3
Soil contamination, milk	87	2	11
Stack emission, beef	5	59	32
Stack emission, milk	3	15	82

Subsequently, beef and milk concentrations were almost two orders of magnitude higher for the soil source category as compared to the stack emission source category, 4) because above ground vegetation are driven by air concentrations, above ground vegetables/fruit and grass/cattle feed concentrations were similar for both demonstrations.

1.5.6.3. Mass Balance

A mass balance exercise was undertaken to evaluate whether a principal of mass balance will be violated with the models and parameters used for the demonstration of the soil source category - that principal being that dioxin releases from a site cannot exceed the original amount at the site (assuming no replenishment). A simplifying assumption for the soil source category was that the soil concentration remained constant over the period of exposure - there was not a systematic depletion of the reservoir over time due to modeled dissipation processes.

First, an estimate of the "reservoir" of 2,3,7,8-TCDD that is implied with the demonstration parameters was made. Then, an estimate of the rate at which this reservoir dissipated using the solution algorithms for dissipation: volatilization and wind erosion flux from soils, soil erosion, the soil ingestion by cattle and children, losses in runoff and leaching, the loss via dermal contact, and the removal via harvest of below ground vegetation. The premise examined was that, if it takes substantially more time than the exposure period to dissipate the reservoir, then the assumption of a constant soil concentration may be suitable for purposes of exposure assessments. On the other hand, complete dissipation within a time period less than or even near to the period of exposure would mean that exposures and risks are being overestimated. This analysis led to a conclusion that the reservoir modeled in the exercise above would take more than 90 years to dissipate.

This was not a definitive exercise, by any means, but it does lend some confidence that a principal of mass balance may not have been violated for the soil source categories, and for the assumption of 30 years exposure duration.

1.5.7. Model Comparisons and Model Validations

Chapter 7 contains a series of tests of the fate models, including comparisons with other available models and model validation exercises. Brief summaries of these exercises providing an overview and qualitative statement about the results are provided below; Chapter 7 provides all the detail and the quantitative results. Overall, model comparison and validation tests in Chapter 7 showed that: 1) the empirical models selected for this methodology compared well with most other models, many of them empirical as well, 2) since many of the key models of this assessment, in particular the bioconcentration models relating a biota concentration (vegetation, terrestrial animal lipid concentration) to an adjacent media concentration (air, animal feeds), were developed from field data, it is not surprising that when tested against other field data, they were shown to reproduce the field data reasonably well. It can be concluded that, with careful parameter assignment, model predictions of environmental/exposure media concentrations of dioxin-like compounds should be reasonably realistic for most uses. Nonetheless, it should be understood that model testing is an ongoing process. The model comparisons and validations summarized here are, by no means, expected to establish model validity beyond any doubt. Users of this methodology are encouraged to subject the models to any number of tests, validation or otherwise, as they use the models described in this document to conduct site-specific assessments for dioxin-like compounds.

1.5.7.1. Model Comparisons

1. Evaluation of alternative air-to-leaf modeling approaches Three empirical air-to-leaf models for estimating grass concentrations from air concentrations are described and tested against two field data sets. Both field data sets contained simultaneous air concentration and grass concentration measurements of dioxin-like congeners. One set was in a rural and the other in an industrial setting in the United Kingdom. Therefore, this test was both a model comparison test as well as a validation of the air-to-plant modeling developed in this methodology. A principal finding of this exercise was that the model selected for this assessment provided the best fit of the data to the model.

2. Estimating water concentrations given a steady input load from overland sources

The WASP4 model, a substantially more complicated aquatic fate model than the one developed in this methodology, was tested in a dynamic and a steady-state mode for Lake Ontario (EPA, 1990a). Conditions in the steady state run were duplicated for the simple dilution model used in the effluent discharge source category of this assessment. Results suggest that the simple dilution model of this assessment produces reasonably similar results as the more complicated WASP4 model.

3. Estimating fish tissue concentrations based on water column concentrations rather than bottom sediment concentrations A water column measure of the potential for a contaminant to accumulate in fish tissue is termed the Bioaccumulation Factor, or BAF. Bioaccumulation refers to the net accumulation of a chemical from exposure via food and sediments as well as water, and is calculated as the ratio of the chemical concentration in the fish to that in the water. A $ssBAF_l^t$ and a $ssBAF_l^d$ (defined respectively as the steady state BAF, lipid- and total water concentration-based, and steady state BAF, lipid- and dissolved water concentration-based) were developed for lake trout, 2,3,7,8-TCDD, and for Lake Ontario 1987 contamination conditions (EPA, 1990a). WASP4 model runs assuming steady loadings to Lake Ontario were duplicated using the watershed modeling approach of this assessment, where water body concentrations were a function of soil erosion loading, followed by simple partitioning and dilution algorithms.

The prediction of whole fish tissue concentrations using $ssBAF_1^t$ and $ssBAF_1^d$ in the WASP4 modeling exercise was similar to fish concentrations predicted using the simple dilution model and the use of the BSAF of this assessment. Differences were further studied using changes in key modeling parameters including sediment organic carbon fraction and others.

4. Other modeling approaches and considerations for air concentrations resulting from soil volatilization For the soil contamination source category, air concentrations result from volatilization of vapor phase dioxins and suspension of particle-bound dioxins. Dioxins released in these two ways disperse over the soils using simple dispersion models. Two alternate modeling approaches for soil volatilization were tested, and one alternate air dispersion model was tested. One of the alternate volatilization approaches was developed by Jury and co-workers (Jury, et al., 1983, 1984a,b). If one assumes that the contaminant moves through the soil column in only the vapor phase, a simplification of the fundamental equations used by Jury offers another option for modeling soil volatilization. This model comparison test showed that the volatilization model chosen for this methodology predicted an average flux over 30 years roughly four times higher than the average flux predicted by the Jury model. The exact reason for this four-fold difference was not ascertained, and could lie in differences in assumed boundary conditions. In any case, it is judged that both models predict comparable volatilization fluxes. On the other hand, the vapor diffusion model predicted volatilization rates that were 100 times less than the Jury models and about 250 times lower than the model of this methodology. The reason for this discrepancy also could not be ascertained.

The alternate approach to estimating on-site dispersion is the "box-model" approach. This is a simple dilution approach similar to the dilution model used to model the dispersion of dioxins emitted from a pipe effluent discharge. Model testing showed that the box model predicted air concentrations above a soil that was 10 times higher than the near-field dispersion model used in this methodology and 100 times higher than the far-field solution.

5. An alternate model for estimating plant concentrations from soil concentrations For plants grown in contaminated soils, plant concentrations are modeled as a two-step process: vapor and particle-phase releases from soil disperse in the air and settle (particle-bound) or transfer (vapor-phase) to the vegetation in the models of this assessment. An alternate and simpler approach was developed from field data on above ground vegetation concentrations correlated to soil concentrations of contaminants and the octanol water partition coefficient in Travis and Arms (1988). This correlation led to an empirical bioconcentration factor for vegetation, B_v, regressed against the contaminant log Kow, and defined by the authors as the concentration in above ground plant parts divided by the concentration in soil. The B_v calculated for 2,3,7,8-TCDD is 0.0041. This compares with two plant:soil ratios calculated using the soil-to-air-to-plant algorithms of this assessment: they were in the range of 10^{-5} for bulky vegetables and 10^{-3} for leafy vegetation.

6. Alternate modeling approaches for estimating beef and milk concentrations

Webster and Connett (1990) compared five models which estimated the 2,3,7,8-TCDD content of cow's milk from 2,3,7,8-TCDD air contamination. All five models have the same basic framework. Particulate-bound 2,3,7,8-TCDD deposits onto the ground and vegetation, cattle feed and pasture grass, to which the cattle are exposed. Algorithms to estimate soil concentrations in these models are the same ones used in this approach, but the vegetation algorithms importantly do not consider vapor phase dioxins. Model validation exercises done in Chapter 6, and model sensitivity analysis exercises done in Chapter 5 both show that neglecting the vapor phase will result in a significant underprediction in vegetation impacts for the lower chlorinated dioxins.

These models use a “biotransfer factor” approach to estimating dioxin concentrations in cow’s milk. This approach converts a daily dosage of dioxin, in units of mass/day, into a dioxin concentration. The “bioconcentration” approach in this assessment first calculates an average dioxin concentration in the whole diet, considering fractions of the diet in vegetation and soil and the concentrations in these diet components, and then multiplies this average concentration by a bioconcentration factor. The most sophisticated biotransfer factor was developed by Travis and Arms (1988), who developed a beef and a milk biotransfer factor, B_b and B_m, as a function of the log Kow of the contaminant. Given a log Kow of 6.8 for 2,3,7,8-TCDD (assumed in this assessment), B_b is solved for as 0.16 and B_m is solved for as 0.03. The beef/milk bioconcentration factors of this document were developed from a data in a study by McLachlan, et al. (1990). This same data can be used to calculate biotransfer factors. The 2,3,7,8-TCDD B_m from that data set is calculated at 0.01. While this 0.01 looks similar to the 0.03 calculated using the empirical relationship developed by Travis and Arms (1988), this does not imply that the Travis and Arms relationship is valid for dioxin-like compounds. In fact, the Travis and Arms relationship calculates a larger B_m as log Kow increases, while the McLachlan data suggest that B_m decreases as the degree of chlorination increases. It was concluded that the Travis and Arms relationship is not valid for compounds with high log Kow, and definitely not valid for dioxin-like compounds.

1.5.7.2. Model Validations

1. The impact of dioxin soil contamination to nearby soils The demonstration of the "soil source" category in Chapter 5 suggested that the concentration at a site of exposure 100 meters away would be 40% of the concentration at the contaminated site for the 2 cm depth and 6% for the 20-cm depth. The literature contained several, somewhat anecdotal, information on elevations of dioxin in soils near a contaminated site, and these were reviewed, including derivation of a similar percentage when possible. One of those sites was the Dow Chemical site in Midland, MI, and percentages derived with their data ranged from 3.5% to 15%, depending on what soil concentration was assumed to represent contamination and what represented impacted.

2. Background soil concentration to bottom sediment concentration ratio A "sediment enrichment ratio" of 3.00 in the model increases the concentration of dioxins in eroding soil as compared to the concentration in the basin draining into a water body. The concentrations of dioxins in the water body will be predicted to be higher than watershed soils, but not necessarily three times higher. Other factors, particularly the organic carbon partition coefficient of the dioxin compound and organic carbon fraction of the water body sediments, also influence the prediction of water body sediment concentration. In the demonstration of background conditions in Chapter 5, the ratio of the concentration of 2,3,7,8-TCDD in water soil to the water body sediment was 2.8. A set of data on 2,3,7,8-TCDD in bottom sediment and background soil concentration from several water bodies and nearby soils was used to derive several similar ratios. Although there was variability among the various water bodies, the average ratio derived using all soil samples (n = 77) and all sediment concentrations (n = 346) was also 2.8.

3. Soil-to-Air and Air-to-Soil Modeling Air and soil measurements of the 17 dioxin congeners at a rural site were available for model testing. Two tests were conducted - one assumed air was the source of dioxins in the soil and soil concentrations were predicted based on deposition of dioxins from the air, and the other assumed that soil was the source of dioxins in the air and modeled the release and dispersion of dioxins. It was found that, when air dioxin was assumed to be source of dioxin in the soil, a good match between predicted and observed soil concentrations were found. When soil was instead assumed to be the source of dioxins in the air, it was found that air concentrations were underpredicted by about three orders of magnitude. This supports findings described earlier in the fate and transport overview of Volume III that long range transport of dioxins by air is the cause for widespread dispersal of dioxins in the environment.

4. Soil to Below Ground Vegetation A laboratory test in which carrots and potatoes were grown in soil at various 2,3,7,8-TCDD levels provided a test for the soil-to-below ground vegetation model. In that model, an RCF (root concentration factor) is multiplied by soil water concentration to predict root concentration. For bulky below ground vegetables, an additional VG_{bg} reduced the concentration predicted only with RCF since this RCF was developed from experiments on barley roots grown in solution. It was found that the measured concentrations in the carrots and potatoes exceeded the predictions made using the RCF and a VG_{bg} initially set at 0.01. It was found that the measured peel concentrations were similar to model predictions without the VG_{bg}. This suggested that the dioxins translocated more than expected into the carrots and potatoes, and on the basis of these tests, the VG_{bg} was increased from 0.01 to 0.25 for application of this model to below ground vegetables.

5. Paper and Pulp Mill Discharges and the Subsequent Impact to Fish Emissions of 2,3,7,8-TCDD measured in the 101 mill study (EPA, 1990b) were combined with average streamflow data from the rivers and streams into which the paper mills discharged. Predictions of 2,3,7,8-TCDD in fish were compared with measurements of TCDD in fish from EPA's National Bioaccumulation Study (EPA, 1992c). These fish were known to be downstream of the pulp and paper mills. First, it was found that the model significantly underpredicted fish concentrations when the mill discharged into the largest water bodies where paired data was available - those with streamflows averaging $2.8 * 10^{10}$ L/hr, with a narrow range of 1 to $4 * 10^{10}$ L/hr. The predicted concentrations were about two orders of magnitude lower than measured. It was speculated that there were many more sources of dioxin input to the rivers in these larger systems; this modeling exercise assumed that the pulp mill was the only source of dioxins to impact the fish. The predictions of TCDD in fish were much better in other paired data sets when the streamflows averaged $5.4 * 10^8$ L/hr, with a range of 10^7 to 10^9 L/hr. These lower streamflows characterized 38 of 47 mills used in this model validation exercise. The average of 38 mills and 74 fish for modeled and observed fish concentrations is 7 ppt and 15 ppt, respectively. Also of note and perhaps not ironically, the highest observed fish concentration of 143.3 ppt is matched by the highest predicted fish concentration of 89.2 ppt, for these lower streamflow conditions. The average of 9 mills and 21 fish associated with large receiving water bodies for modeled and observed fish concentrations is 0.7 and 5.3 ppt, respectively.

6. ISCST3 Modeling of the Release of Large Amounts of Dioxin from a Municipal Solid Waste Incinerator and the Subsequent Impacts to Air and Soil Measurements of dioxin TEQ emissions at nearly 1 kg/yr from a municipal solid waste incinerator in Columbus, OH, in 1994 caused EPA's Region 5 to issue an Emergency Order requiring operators of this incinerator to install MACT, Maximum Achievable Control Technology, pollution devices on an accelerated basis. The incinerator had been in operation 11 years at this time, and a soil study clearly showed elevations of dioxins in soils to about 3 km from the incinerator. Air measurements taken at the same time additional stack measurements were taken were also available, and the stack, air, soil, and obtained meteorological data allowed for a comprehensive test of the ISCST3 model to predict short-term air concentrations, and with predicted depositions, long-term soil concentrations. It was found that the prediction of highest 48-hour air concentrations were generally in the same direction and magnitude as was found in the air monitoring program. With specific congener differences, the model also showed the highest soil concentrations near the incinerator, with reduced concentrations in both measured and modeled concentrations as the distance increased. An important observation was that the modeled soil concentrations retained a similar relative profile as the stack-emitted dioxins. Differences between the soil and stack profiles were only due to vapor/particle partitioning - there was no modeling of air degradation (there was plume depletion due to deposition), and all congeners were modeled to dissipate in soil at a similar half-life of 25 years. On the other hand, the measured soil concentrations, while unambiguously and significantly elevated near the incinerator, had a relative profile very similar to dioxin profiles found in a typical background urban or rural soil profile, where concentrations are much lower. This shows that, while high emissions clearly resulted in high soil concentrations near the Ohio incinerator, differential dissipation mechanisms for the congeners in the soil and/or in the air resulted in a very typical profile that was not modeled by ISCST3 and the soil concentration model. Future modeling exercises for dioxins in air and depositing to soil using the ISCST3 or similar models should focus on this disparity.

1.5.8. Uncertainty

Some discussion of the issues commonly lumped into the term "uncertainty" is needed at the outset. The following questions capture the range of issues typically involved in uncertainty evaluations:

(1) How certain are site specific exposure predictions that can be made with the methods?

(2) How variable are the levels of exposure among different members of an exposed local population?

(3) How variable are exposures associated with different sources of contamination?

The emphasis in Volume III is in providing the technical tools needed to perform site-specific exposure assessments. For the assessor focusing on a particular site, question (1) will be of preeminent importance. Therefore, the emphasis of the uncertainty evaluation is to elucidate those uncertainties inherent to the exposure assessment tools presented. This chapter examines the capabilities and uncertainties associated with estimating exposure media concentrations of the dioxin-like compounds using the fate, transport, and transfer algorithms, and also identifies and discusses uncertain parameters associated with human exposure patterns (contact rates and fractions, exposure durations, etc.).

A site specific assessment will also need to address the variability of risks among different members of the exposed population, the second key question above. The level of detail with which this can be done depends on the assessors knowledge about the actual or likely activities of the exposed population. In this document, one approach to evaluating this variability is demonstrated. Separate "central" and "high end" scenario calculations are presented to reflect different patterns of human activities within a hypothetical rural population.

A key issue with regard to intra-population variability is that it is best (if not only) addressed within the context of a specifically identified population. If such information is available, a powerful tool that can be used to evaluate the variability within a population is Monte Carlo Analysis. Three recent Monte Carlo studies which have been done for exposure to 2,3,7,8-TCDD were reviewed. Assumptions on distributions of exposure patterns and fate and transport parameter distributions are described, as are the results of their analyses.

With regard to question (3), this document does not present a detailed evaluation of how exposure levels will vary between different sources of release of dioxin-like compounds into the environment. While Volume III does demonstrate the methodologies developed for sources of release of dioxin-like compounds into the environment with source strengths and environments crafted to be plausible and meaningful, there is still a great deal of variability on both the source strengths and on the environments into which the releases occur. For example, the frequency with which farms and rural residences are near stack emissions of dioxin-like compounds is not addressed. Such analysis is beyond the scope of this document.

Chapter 7 identifies some of the key uncertainties, as well as the key supporting evidence, for the procedures and parameters associated with all the pathways. A summary of discussions from the uncertainty evaluation is now presented. First is a summary of three exposure parameters common to all pathways:

1. Lifetime, Body Weights, and Exposure Durations: Of these three parameters, the exposure duration is the most uncertain. The estimates of 9 and 30 years were made in this assessment for non-farming residents in rural settings, and farming residents in rural settings.

An adult body weight of 70 kilograms and a lifetime of 70 years are standard assumptions for exposure and risk and, although variability is recognized for these parameters, these variations are not expected to add significant uncertainty in exposure estimates. The same is true for the 17 kg child body weight in the childhood exposure pattern of soil ingestion.

2. Soil Ingestion and Soil Dermal Contact: Soil ingestion for older children and adults were not considered, which may have underestimated lifetime soil ingestion exposures. Pica soil ingestion patterns were not evaluated in this assessment. The ingestion rates (100 mg/day for central scenarios and 600 mg/day for high end scenarios, during ages 2-6) considering this appear reasonable. For the soil dermal contact pathway, key uncertain parameters include the contact rate, (0.005 and 0.1 mg/cm^2-event for indoor and outdoor events for the high end farming scenario) and the absorption fraction (0.03 for dioxin-like compounds).

A major area of uncertainty for both pathways is the estimation of soil concentrations where the source of contamination is located distant from the site of exposure. For this assessment, this includes the off-site soil source category and the stack emission source category. Validation exercises described above seem to suggest that prediction of soil concentrations from airborne depositions or from soil erosion appear reasonable. Key uncertain parameters identified include the dissipation rate (0.0693 yr^{-1}), the mixing depth (2 cm), and the use of an enrichment ratio (equal to 3.0) which increases the concentration of dioxin-like compound on eroded soil relative to in-situ soil.

3. Ingestion of Water: A comparison of alternate modeling approaches for estimating water concentrations showed similar results to the models adopted for this assessment. There also does not appear to be a wide range of possible values for water ingestion rate (1.4 L/day for central scenarios and 2.0 L/day for high end scenarios) and contact fraction (0.75 for central scenarios and 0.90 for high end scenarios), and these are not expected to introduce significant uncertainty into water ingestion exposure estimates.

4. Inhalation: The inhalation rate assumed for both central and high end scenarios was 20 m^3/day. The distinction in the scenarios was in the contact fractions: central scenarios assumed a contact fraction of 0.75 and high end scenarios had a 0.90 contact fraction. These fractions correspond to time at the home environment. These fractions and the inhalation rate are not expected to add significant uncertainty in inhalation exposure estimates.

Sensitivity analysis showed air concentrations resulting from soil emissions to be sensitive to Koc and H, and also to key source strength and delivery terms such as areas of contamination and wind speed. Assuming these non-chemical specific parameters can be known with reasonable certainty for site-specific applications, the most uncertainty lies with chemical specific data.

Alternate approaches for volatilization and air dispersion tested included the volatilization approach developed by Jury, et al. (1983, 1984a,b) and the box model for dispersion calculations. The Jury model predicted about 1/3 as much volatilization flux (given the selection of parameters, made equal to or most analogous to the models of this assessment) as the Hwang, et al. (1986) model of this assessment. The box model predicted about 6 times higher air concentrations than the near-field dispersion approach of this assessment. This reasonable comparison lends some credibility to the models selected.

Approaches to estimate particulate phase concentrations are empirical and based on field data. They are based on highly erodible soils but are specific to inhalable size particles, those less than 10 μm. As such, they may overestimate inhalation exposures, but may underestimate the total reservoir of particulates, which becomes critical for the particle deposition to vegetation algorithms. Another area of uncertainty is the assumption that volatilized contaminants do not become sorbed to airborne particles - this is also critical because vapor phase transfers dominate plant concentration estimation. A final key area of uncertainty is that transported contaminants from a contaminated to an exposure site via erosion are assumed not to volatilize or resuspend at the exposure site or from soils between the contaminated and the exposure site - air borne exposure site concentrations may be underestimated as a result.

5. Fruit and Vegetable Ingestion: All ingestion parameters assumed are evaluated as reasonable for general exposure to broad categories of fruits and vegetables. However, great variability is expected if using these procedures on a specific site where home gardening practices can be more precisely ascertained. Concepts of below and above ground vegetations were developed to accommodate soil to root algorithms and soil to air to vegetation algorithms.

Protected vegetation - those with outer inedible protections such as citrus or corn - were assumed not to be impacted by dioxin-like compounds.

A key assumption in the vegetation algorithm, that dioxin-like compounds do not translocate from root to shoot, was verified by two experiments. Vapor-phase contributions to vegetation dominated the contaminated soil and stack emission source categories, with one exception. Particle depositions were more important for above ground fruit/vegetable concentrations for the stack emission source.

Critical empirical parameters were the above and below ground correction factors, VG_{ag} and VG_{bg}, which were set at 0.01 and 0.25, respectively, for fruits and vegetables. The VG_{ag} was justified based on the fact that the above-ground transfer algorithm was based on experiments for the azalea leaf, and this leafy vegetation would differ for fruits such as apples where translocation into the fruit would be minimal; the VG_{ag} for grass was 1.00 for this same reason. Support for this VG_{ag} for bulky above ground vegetation came from independent experiments by McCrady (1994). The VG_{bg} was given a value of 0.25 based on the testing of the root transfer algorithm on experiments on carrots and potatoes.

Important experimentally derived empirical factors describing the transfer of dioxin-like compounds from one media to plants include the RCF, the soil to below ground vegetables transfer factor, and B_{vpa}, the vapor-phase to above ground plant transfer factor. Validation exercises described above on both of these factors lent a degree a credibility to the independent use of these transfer factors.

6. Ingestion of Fish: The key exposure parameter for this pathway was the fish ingestion rate. The rates assumed in the demonstration scenarios were low in comparison to estimates given for subsistence fisherman or others who live near large water bodies where fish are commercially caught. The justification for the lower ingestion rate for demonstration purposes was that the setting demonstrated was described as rural, containing farms and non-farm residences, where the emphasis is on agriculture. A relatively small watershed with a small impacted water body was assumed. Daily ingestion rates of 8 (central) and 25 (high end) g/day were assumed, based on data from recreational fisherman surveys summarized in EPA (1997b) and on subsequent recommendations for these values in EPA (1997b).

Other models for estimating fish concentration based on water column concentrations, rather than suspended sediment concentrations, were described in EPA (1993) and demonstrated in this assessment. Results indicated that the water column approaches would predict similar whole fish concentrations compared with the sediment concentration approaches of this assessment. A key uncertain parameter for estimating fish tissue concentrations is the Biota Sediment Accumulation Factor, or BSAF, and the Biota Suspended Sediment Accumulation Factor, or BSSAF. A range of 0.03 to 0.30 for 2,3,7,8-TCDD is hypothesized for column feeding fish, while the Connecticut data (CDEP, 1992) and some other data on bottom feeding fish indicate higher BSAFs ranging up to 0.86 for 2,3,7,8-TCDD. A value of 0.09 for 2,3,7,8-TCDD for BSAF and BSSAF is used in this assessments. A "Bioequivalency Factor", or BEF, approach is used to assign values to other dioxin-like congeners. In this approach, a relative rating scheme provides factors to multiply the 2,3,7,8-TCDD bioconcentration parameter to get this same factor for the other dioxin congeners. Another key parameter is the fish lipid content, which can vary from below 0.05 to above 0.20. The model estimates a fish lipid concentration. Multiplying fish lipid concentration by fish lipid content arrives at a whole fish concentration or an edible fish concentration, depending on the user's assignment and characterization of the fish lipid content variable. For this assignment, the fish lipid content was assigned a value of 0.07 for the demonstration scenarios, based on lipid content of fish in EPA's Lake Ontario study (EPA, 1990a).

7. Beef and Milk Ingestion: The rates of beef and milk fat ingestion are 2.45 and 14.0 g/day, respectively. Beef fat and milk fat contents are assumed to be 22% and 3.5%, respectively. An additional factor of 0.55 accounts for cooking and post cooking loss of beef. The assumptions for contact fractions for beef and milk (fractions of their total consumption that comes from home supplies) was 0.48 and 0.21, respectively. Since exposure estimates from these pathways are linearly related to ingestion rate and contact fraction, these are critical exposure parameters for site specific applications.

Comparison with earlier modeling approaches showed that the current approach to estimating beef and milk concentrations is the same as earlier approaches, although mathematically formulated differently. Early efforts in the literature did not consider vapor transfers to vegetation; one later assessment did include vapor transfers, and a key result in that assessment, as well as this one, is that vapor transfers are critical for beef impacts. Finally, earlier assessments considered the practice of fattening beef cattle prior to slaughter by feeding them residue-free grains. These efforts estimated over a 50% reduction in beef concentration due to residue degradation or elimination and/or dilution with increases in body fat. The demonstration scenarios in this assessment did not consider this practice. However, this practice was considered in the air-to-beef food chain validation exercise. There, a 50% reduction in beef concentrations due to feedlot fattening was assumed.

Key uncertain and variable parameters for beef/milk concentrations include: 1) the beef and milk fat bioconcentration factor, BCF, 2) the beef cattle and dairy cow exposure assumptions (fractions of their feed in which feed categories), 3) the assumptions concerning vapor/particle partitioning for the stack emission source category, and 4) the air-to-leaf transfer parameter, B_{vpa}, for vapor phase contaminants.

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Table 1-1. The TEF scheme for I-TEQ$_{DF}$

Dioxin (D) Congener	TEF	Furan (F) Congener	TEF
2,3,7,8-TCDD	1.0	2,3,7,8-TCDF	0.1
1,2,3,7,8-PeCDD	0.5	1,2,3,7,8-PeCDF	0.05
1,2,3,4,7,8-HxCDD	0.1	2,3,4,7,8-PeCDF	0.5
1,2,3,6,7,8-HxCDD	0.1	1,2,3,4,7,8-HxCDF	0.1
1,2,3,7,8,9-HxCDD	0.1	1,2,3,6,7,8-HxCDF	0.1
1,2,3,4,6,7,8-HpCDD	0.01	1,2,3,7,8,9-HxCDF	0.1
OCDD	0.001	2,3,4,6,7,8-HxCDF	0.1
		1,2,3,4,6,7,8-HpCDF	0.01
		1,2,3,4,7,8,9-HpCDF	0.01
		OCDF	0.001

Table 1-2. The TEF scheme for dioxin-like coplanar PCBs, as determined by the World Health Organization in 1994

Chemical Structure	IUPAC Number	TEF
3,3',4,4'-TeCB	PCB-77	0.0005
2,3,3',4,4'-PeCB	PCB-105	0.0001
2,3,4,4',5-PeCB	PCB-114	0.0005
2,3',4,4',5-PeCB	PCB-118	0.0001
2',3,4,4',5-PeCB	PCB-123	0.0001
3,3',4,4',5-PeCB	PCB-126	0.1
2,3,3',4,4',5-HxCB	PCB-156	0.0005
2,3,3',4,4',5'-HxCB	PCB-157	0.0005
2,3',4,4',5,5'-HxCB	PCB-167	0.00001
3,3',4,4',5,5'-HxCB	PCB-169	0.01
2,2',3,3',4,4',5-HpCB	PCB-170	0.0001
2,2',3,4,4',5,5'-HpCB	PCB-180	0.00001
2,3,3',4,4',5,5'-HpCB	PCB-189	0.0001

Table 1-3. The TEF scheme for TEQ_{DFP}-WHO_{98}

Dioxin Congeners	TEF	Furan Congeners	TEF
2,3,7,8-TCDD	1.0	2,3,7,8-TCDF	0.1
1,2,3,7,8-PeCDD	1.0	1,2,3,7,8-PeCDF	0.05
1,2,3,4,7,8-HxCDD	0.1	2,3,4,7,8-PeCDF	0.5
1,2,3,6,7,8-HxCDD	0.1	1,2,3,4,7,8-HxCDF	0.1
1,2,3,7,8,9-HxCDD	0.1	1,2,3,6,7,8-HxCDF	0.1
1,2,3,4,6,7,8-HpCDD	0.01	1,2,3,7,8,9-HxCDF	0.1
OCDD	0.0001	2,3,4,6,7,8-HxCDF	0.1
		1,2,3,4,6,7,8-HpCDF	0.01
		1,2,3,4,7,8,9-HpCDF	0.01
		OCDF	0.0001

Chemical Structure	IUPAC Number	TEF
3,3',4,4'-TeCB	PCB-77	0.0001
3,4,4',5-TCB	PCB-81	0.0001
2,3,3',4,4'-PeCB	PCB-105	0.0001
2,3,4,4',5-PeCB	PCB-114	0.0005
2,3',4,4',5-PeCB	PCB-118	0.0001
2',3,4,4',5-PeCB	PCB-123	0.0001
3,3',4,4',5-PeCB	PCB-126	0.1
2,3,3',4,4',5-HxCB	PCB-156	0.0005
2,3,3',4,4',5'-HxCB	PCB-157	0.0005
2,3',4,4',5,5'-HxCB	PCB-167	0.00001
3,3',4,4',5,5'-HxCB	PCB-169	0.01
2,3,3',4,4',5,5'-HpCB	PCB-189	0.0001

Table 1-4. Particle fractions, ϕ, in four airsheds at 20°C for the dioxin-like congeners.

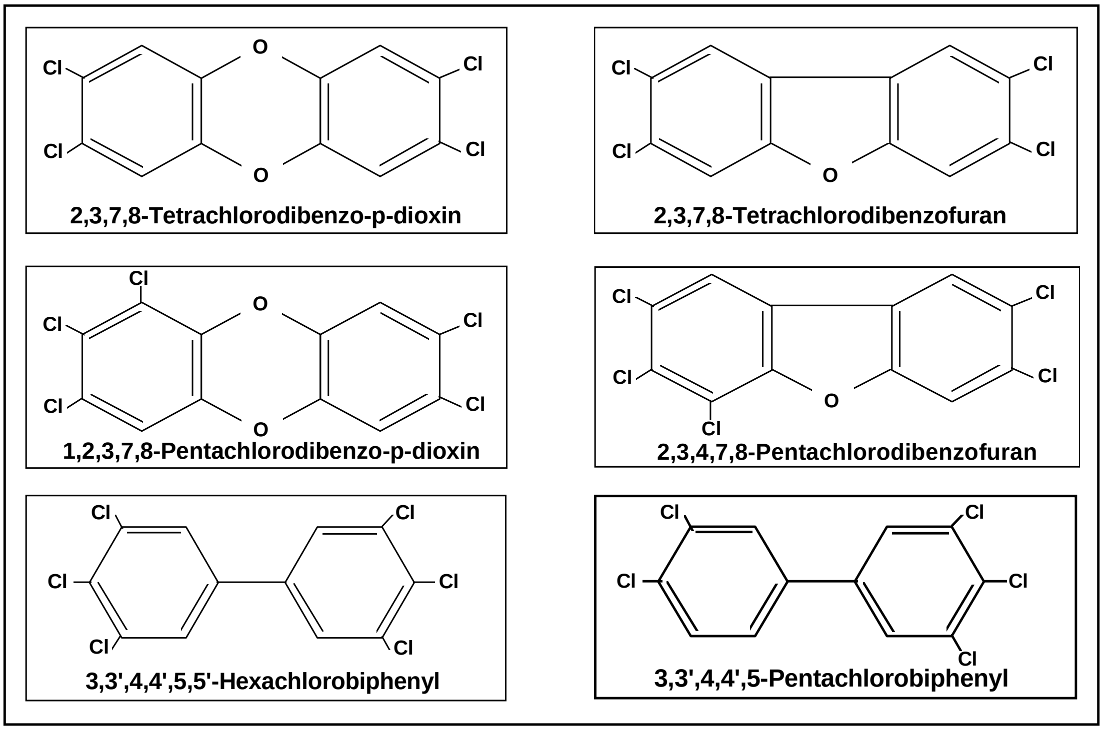
Congener	Clean Continental	Average Background	Background Plus Local Sources	Urban
2378-TCDD	0.10	0.29	0.49	0.75
12378-PCDD	0.44	0.74	0.87	0.95
123478-HxCDD	0.78	0.93	0.97	0.99
123678-HxCDD	0.78	0.93	0.97	0.99
123789-HxCDD	0.78	0.93	0.97	0.99
1234678-HpCDD	0.93	0.98	0.99	0.997
OCDD	0.98	0.995	0.998	0.999
2378-TCDF	0.09	0.27	0.47	0.73
12378-PCDF	0.27	0.57	0.75	0.91
23478-PCDF	0.38	0.69	0.84	0.94
123478-HxCDF	0.63	0.86	0.93	0.98
123678-HxCDF	0.63	0.86	0.93	0.98
123789-HxCDF	0.74	0.91	0.96	0.99
234678-HxCDF	0.74	0.91	0.96	0.99
1234678-HpCDF	0.86	0.96	0.98	0.99
1234789-HpCDF	0.92	0.98	0.99	0.997
OCDF	0.98	0.995	0.998	0.999

Table 1-5. WHO_{98}-TEQ_{DF} environmental and exposure media concentrations for the background conditions scenarios, #1 and #2, and the stack emissions demonstration scenarios, #4 and #5.

Description	Background, Scenarios 1 and 2	Emission, Central Scenario 4	Emission, High End Scenario 5
Air, vapor phase, pg/m^3	$2.59*10^{-3}$	$2.45*10^{-5}$	$6.94*10^{-5}$
Air, particle phase, pg/m^3	$1.87*10^{-2}$	$6.04*10^{-5}$	$1.74*10^{-4}$
Soil, untilled, pg/g	1.29	$4.46*10^{-3}$	$3.51*10^{-2}$
Soil, tilled, pg/g	0.65	$4.46*10^{-4}$	$3.51*10^{-3}$
Soil, watershed, pg/g	1.29	$8.91*10^{-4}$	$8.91*10^{-4}$
Surface water, pg/L	$2.63*10^{-3}$	$3.80*10^{-5}$	$3.80*10^{-5}$
Sediment, pg/g	3.37	$2.39*10^{-3}$	$2.39*10^{-3}$
fish lipid, pg/g*	6.33	$5.64*10^{-3}$	$5.64*10^{-3}$
leafy vegetation, pg/g dry	0.45	$1.86*10^{-3}$	$6.39*10^{-3}$
above ground fruit/veg, pg/g fresh	$5.74*10^{-3}$	$1.20*10^{-5}$	$6.37*10^{-5}$
below ground vegetables, pg/g fresh	$1.94*10^{-2}$	$1.63*10^{-5}$	$1.29*10^{-4}$
beef fat, pg/g	1.58	$4.35*10^{-3}$	$1.65*10^{-2}$
milk fat, pg/g	1.10	$3.05*10^{-3}$	$1.11*10^{-3}$
chicken fat, pg/g	0.61	$2.02*10^{-3}$	$1.38*10^{-2}$
egg fat, pg/g	0.71	$2.25*10^{-3}$	$1.55*10^{-2}$

Table 1-6. Lifetime average daily doses, LADD, and cancer risk estimates, of WHO_{98}-TEQ_{DF} for the high end background scenario, #2, and for the high end stack emission scenarios, #5.

Scenario/Pathway	LADD, ng/kg-day	Percent of total scenario exposure	Cancer Risk
Scenario 2 - Background High			
Soil Ingestion	$3.25*10^{-6}$	3	$1.22*10^{-6}$
Soil Dermal Contact	$1.40*10^{-6}$	2	$5.27*10^{-8}$
Inhalation	$2.34*10^{-6}$	3	$2.34*10^{-6}$
Water Ingestion	$2.90*10^{-8}$	<1	$2.90*10^{-7}$
Beef Ingestion	$8.30*10^{-5}$	89	$8.30*10^{-5}$
Vegetable Ingestion	$2.32*10^{-6}$	2	$2.36*10^{-6}$
Fruit Ingestion	$3.65*10^{-7}$	<1	$3.65*10^{-7}$
Total	$9.30*10^{-5}$	100	$8.96*10^{-5}$
Scenario 5 - Stack Emission High			
Soil Ingestion	$8.86*10^{-8}$	9	$3.32*10^{-8}$
Soil dermal contact	$8.51*10^{-9}$	1	$3.19*10^{-10}$
Inhalation	$2.68*10^{-8}$	2	$2.68*10^{-8}$
Water ingestion	$4.19*10^{-10}$	<1	$4.19*10^{-10}$
Beef ingestion	$8.65*10^{-7}$	86	$8.65*10^{-7}$
Vegetable ingestion	$1.83*10^{-8}$	2	$1.83*10^{-8}$
Fruit ingestion	$4.05*10^{-9}$	<1	$4.05*10^{-9}$
Total	$1.01*10^{-6}$	100	$9.48*10^{-7}$



2,3,7,8-Tetrachlorodibenzo-p-dioxin

2,3,7,8-Tetrachlorodibenzofuran

1,2,3,7,8-Pentachlorodibenzo-p-dioxin

2,3,4,7,8-Pentachlorodibenzofuran

3,3',4,4',5,5'-Hexachlorobiphenyl

3,3',4,4',5-Pentachlorobiphenyl

Figure 1-1. Chemical structure of 2,3,7,8-TCDD and related compounds

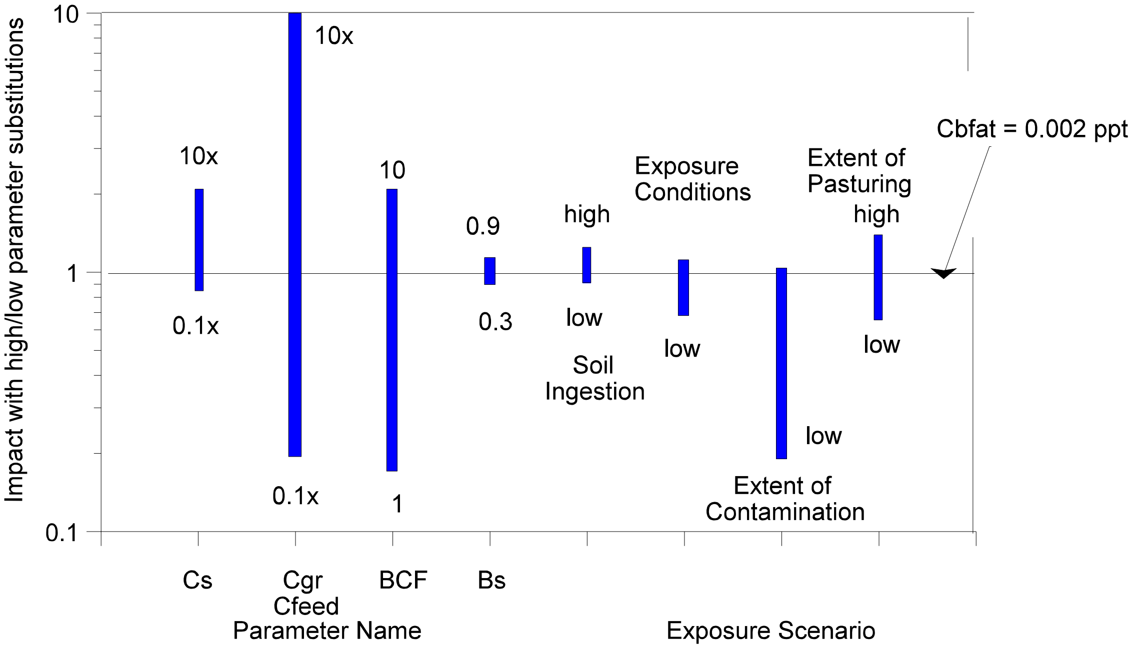


Figure 1-2. Results of sensitivity analysis of algorithms estimating beef fat concentrations in the stack emission source category.

Parameter Name	Definition	Selected
Cs	2,3,7,8-TCDD soil concentration, pg/g	0.001
Cgr	2,3,7,8-TCDD grass concentration, pg/g dry wt.	0.0004
Cfeed	2,3,7,8-TCDD feed concentration, pg/g dry wt.	0.0002
Cbfat	2,3,7,8-TCDD beef fat concentration, pg/g	0.002
BCF	beef/milk bioconcentration factor, unitless	5.76
Bs	bioavailability of contaminant on soil relative to vegetation	0.65

Exposure Scenario Parameters:

BCSDF	beef cattle soil diet fraction	0.04
BCFDF	beef cattle feed diet fraction	0.48
BCGDF	beef cattle grass diet fraction	0.48
BCGRA	beef cattle fraction of contaminated grazing land	1.00
BCFOD	beef cattle fraction of contaminated feed	1.00