RISK ASSESSMENT OF COMBUSTOR STACK EMISSIONS

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ABSTRACT

In May of 1993, the U.S. EPA announced that the permitting of hazardous waste combustors will depend, among other things, on the results of site-specific assessments of health risk (U.S. EPA, 1993). Current risk assessment methodologies attempt to trace the movement of pollutants throughout the environment and evaluate all means through which an individual might be exposed to chemicals released from a facility stack. Such multipathway risk assessments require the specification of a myriad of parameters. According to current regulatory guidance and practice, "reasonable maximum exposure" scenarios are constructed by selecting a mix of average and conservative parameter values: the goal is to estimate high-end risks to a limited population of individuals. Considerable uncertainty is associated with many parameters and assumptions, however, and the interpretation of risk estimates is confounded by the inability to gauge the overall degree of conservatism. In complex settings, uncertainties are so large that present methodologies are incapable of providing meaningful and reliable information. In these cases, probabilistic techniques offer a means to assess both uncertainty and variability and thereby provide a wealth of additional information that cannot be rendered by deterministic risk assessments. In this paper, we present a case study of a hazardous waste combustor in which probabilistic techniques will be employed extensively in a systematic, comprehensive manner. The direct treatment of uncertainty includes the detailed consideration of dose-response factors that are typically treated as definitive values in traditional risk assessments, but typically embody the largest degree of uncertainty.

INTRODUCTION

Combustors of hazardous waste and municipal solid waste release various potentially toxic compounds. Some of the chemicals emitted are constituents of the waste that travel through the combustion process and are not captured by pollution control equipment. These chemicals include metals such as mercury, arsenic, and cadmium, and organic compounds that escape combustion or are only partially oxidized. Other pollutants, such as polychlorinated dibenzo-p-dioxins and furans (PCDDs and PCDFs), are byproducts formed in the combustion train.

Risk assessment is a formal, mathematical tool that can be used to evaluate potential hazards introduced by pollutant emissions. The classic, four-step process (National Research Council, 1983) is illustrated in Figure 1.

In hazard identification, chemicals of concern are identified from stack test measurements or, in the case of a planned facility, from the operating experience of similar plants. For each chemical, emission rates are assigned based on measurements (if available) or projected with consideration of plant capacity, waste composition, pollution control equipment, and performance testing of existing facilities. Exposure assessment marries estimates of pollutant concentrations in environmental media with the anticipated rates these media will be contacted by receptors. Independently, the toxicity of each chemical is characterized quantitatively by a dose-response assessment that relates the degree of harm concomitant with varying levels of exposure. Finally, the exposure and dose-response assessments are compared to characterize potential risks.

As described further on, typical risk assessments rely

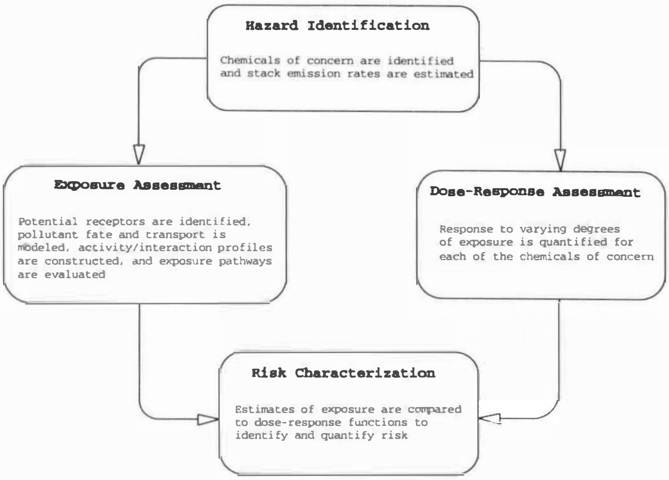

FIG. 1 THE BASIC ELEMENTS OF A HUMAN HEALTH RISK ASSESSMENT OF COMBUSTOR STACK EMISSIONS

on deterministic models that yield point estimates of risk. These estimates are generally thought to be conservative (high-end) values, but the degree of conservatism is unknown, as is the level of uncertainty of the estimate. In the final portions of this paper, we describe a risk assessment we are in the midst of conducting in which probabilistic methods are being utilized extensively. Our assessment is intended to characterize the risks (and uncertainty of risks) presented by the highly controversial WTI hazardous waste combustor in East Liverpool, Ohio. As described in section 5, the WTI facility is located in a narrow river valley, and potentially poor air pollutant dispersion emphasizes the importance of a thorough, carefully constructed risk assessment.

Evolution of Risk Assessment of Air Pollutant Sources

Risk assessment is a dynamic, evolving process. Changes and refinements have been introduced for two principal reasons. First, advances in scientific knowledge and applied methodologies have increased the sophistication of risk assessments. Second, this sophistication has been paralleled by the demands of regulators and others for risk assessments to provide information of ever-increasing breadth and detail.

The human health risk assessments of combustors that were performed a decade or so ago were of limited scope, typically considering a select number of chemicals and evaluating risks due only to inhalation. Pollutant concentrations in ambient air were predicted using standard dispersion models commonly used in regulatory analyses under the Clean Air Act. This process depended primarily on the specification of emission rates for all pollutants of interest.

The potential importance of exposure routes other than inhalation was recognized through the realization that for pollutants such as chlorinated dioxins and furans, most of an individual's body burden results from dietary intake, and inhalation constitutes only a small fraction of total exposure. Fortunately, most of the techniques needed to conduct multipathway risk assessments had already been developed by the Nuclear Regulatory Commission for the evaluation of nuclear power plants. As an example, Baes et al. (1984) provide a detailed review of the procedures and factors available for evaluating the fate and transport of pollutants in soil and agricultural products. Most of this work was devoted to inorganic compounds (radionuclides), however: more recent research has involved the estimation of fate and transport parameters for organic compounds, with special emphasis on chlorinated dioxins and furans.

Most of the early multipathway risk assessments were conducted for municipal solid waste combustors. By the mid-late 1980s, assessments evaluated exposure to contaminated soil, water, vegetables, beef and dairy products, fish, and the secondary exposure received by a nursing infant (Levin et al., 1990). Not all routes were common to all assessments, partly because of appropriateness (e.g., the absence of nearby farms allowed the omission of any assessment of beef and dairy products), and partly because there was no consensus regarding the list of exposure routes that should be evaluated.

Figure 2 is a conceptual representation of the multipathway exposure assessment to pollutants that may be released from the stack of a combustor. Two large boxes distinguish the "fate and transport modeling" and "human exposure estimates" sections. "Fate and transport modeling" utilizes mathematical algorithms to predict the travel of pollutants emitted from the stack. An air dispersion model typically combines information (1) about the plant (such as the size and configuration of the buildings, the height of the stack, and the properties of the flue gas), (2) the terrain of environs surrounding the facility, and (3) hourly measurements of meteorological parameters, to predict the dispersion of pollutants in the atmosphere. Additional algorithms predict the rates at which airborne pollutants are deposited to soil, water, and vegetative surfaces. Upon deposition, relevant physical and chemical processes are modeled in order to predict the behavior of contaminants in each of these media. Additional models predict the transfer and accumulation of pollutants in locally-produced vegetables, meats, fish, and dairy products.

The solid-filled arrows between the different environmental compartments indicate the pathways through which pollutants are assumed to travel. Some relationships are fairly simple. For example, pollutant concentrations in air are estimated in a straightforward manner by modeling the dispersion of emissions from the stack. Other routes

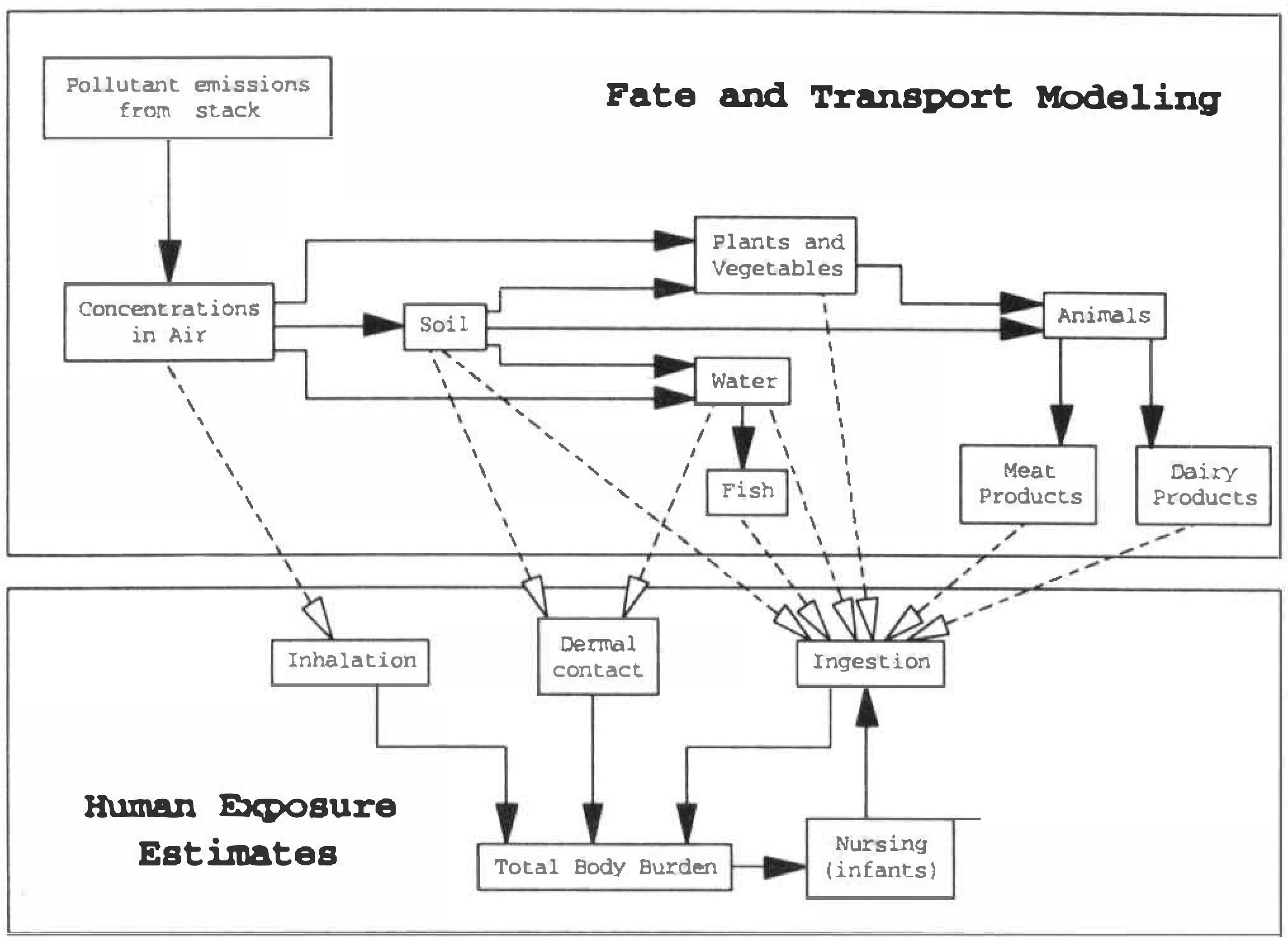

FIG. 2 A CONCEPTUAL VIEW OF A MULTIPATHWAY RISK ASSESSMENT FOR CUMBUSTOR STACK EMISSIONS

are much more complex and require the pursuit of contaminants through several environmental media.

Consider, for example, the estimation of pollutant concentrations in cow's milk that is produced within an area affected by the plant. Empirical measurements from other studies can be used to estimate the concentrations of pollutants in milk that will result from a cow's exposure to contaminants in her environment. In this case, the cow's intake of pollutants is derived from eating food and (incidentally) soil. The method used to estimate contaminant concentrations in air is as described above.

As indicated in Figure 2, pollutant concentrations in soil are modeled as the deposition of contaminants in air. Surface water supplies can be affected by (1) direct deposition of pollutants from the air and (2) contaminants conveyed in surface runoff of soil (which is in turn affected by deposition of pollutants from air). The cow's food supply (vegetation) can also become contaminated through two mechanisms: airborne pollutants can deposit to the surface of vegetation, and contaminants from soil can translocate into plants.

The lower box (with the heading "human exposure estimates") in Figure 2 illustrates the three mechanisms through which humans are exposed to pollutants—inhalation of vapors and particles, ingestion of a variety of media (both purposeful and incidental), and dermal contact with chemicals in soil or water. Incorporating contaminant uptake by these three routes allows the estimation of a total intake for each pollutant.

Within Figure 2, the "fate and transport modeling" and "human exposure estimates" are linked with open-headed arrows. The connections indicate the various pathways through which humans may be exposed to pollutants that originate from stack emissions.

Detailed guidance for conducting multipathway risk assessments has been developed by the U.S. EPA (1990) that incorporates all of the elements of Figure 2. There are, however, a number of technical and policy issues that are currently under examination. Thus, risk assessment methodologies will continue to evolve as new data are collected, scientific uncertainties are resolved, and guidance is refined. The remainder of this section is devoted to the discussion of several issues pertinent to current risk assessment practices.

Estimating Pollutant Deposition

Particle-bound pollutants deposit to the ground via two mechanisms. Dry deposition is the process whereby particles settle and adhere to the surface. Wet deposition refers to the particle scavenging that occurs as precipitation falls in the atmosphere. Of these two processes, dry deposition is better understood, and modeling algorithms for dry deposition have been more commonly applied than those for wet deposition. In fact, many multipathway risk assessments have ignored explicit consideration of wet deposition, in part because of the immaturity of available models, and in part because of uncertainties in its contribution to food-chain pathways that generally are found to be the most critical exposure routes.[1] More specifically, one of the key steps in estimating pollutant concentrations in vegetation is assessing the level of particle bound pollutants deposited onto the surface of leaves. Consider particles washed out of the air by precipitation. Little is known about the fraction of pollutants that may be transferred from rain to leaf surfaces as falling hydrometers careen through vegetative cover. In fact, arguments could be made that precipitation serves to clean vegetative surfaces. Countering this notion, particles may remain on leaves upon termination of precipitation and subsequent evaporation of the thin film of water that had adhered to the surface of the plant.

Models are available ior estimating wet deposition of particles (EPA, 1990), but none have been validated adequately in field tests. Qualitatively, estimates of wet deposition flux can be much greater than dry deposition flux, especially close to the stack location. In fact, since wet deposition models assume particle scavenging over the entire cross-section of the plume, the maximum wet deposition rate is modeled at the stack location. Dry deposition, however, depends upon the ground-level concentration, and thus is greatest at the point where the plume touches down. As a result, the indiscriminate summation of maximum wet and dry deposition rates (which we have seen in some assessments) is an erroneous simplification. Furthermore, as discussed above, assumptions concerning the role of

[1]Evaluation of wet deposition has not been addressed by most regulatory authorities. The New York State Department of Health, for example, requires the consideration of only dry deposition in their draft exposure assessment guidance for municipal solid waste combustors (NYSDOH, 1991).

wet deposition in fate and transport modeling may be of crucial importance.

Recent research (e.g., McCrady and Maggard, 1993) has demonstrated that plant tissues can assimilate vapors from the atmosphere and recent laboratory studies show this to be a potentially important mechanism (Welsch-Pausch et al., 1993; Rippen & Wesp, 1993). For pollutants such as PCDDs and PCDFs that exhibit an affinity for the particle-bound phase, even a small vapor phase component can enhance air-to-plant deposition rates. Additional research is necessary, however, to characterize the factors that govern vapor-particle partitioning of these compounds in the atmosphere.

Defining Exposure Profiles

Even though multipathway risk assessments are highly quantitative and entail numerous models and calculations, many aspects of risk assessment remain a craft and demand skilled professional judgement. Perhaps the greatest sense of discretion afforded the risk assessor is the choice of the factors that determine the nature and degree of exposure assumed in the risk assessment. Typically, hypothetical individuals are constructed who contact facility-released pollutants through multiple exposure routes — often through all exposure routes considered in the analysis. Since many assumptions are multiplicative, particular combinations of parameters and choices can have dramatic effects on risk estimates. General choices include where the hypothetical receptor lives (and remains living for extended periods), sources of foodstuffs (home-grown or store-bought), and rates of contact with contaminated media.

The Maximally Exposed Individual (MEI) is a common regulatory construct of early multipathway risk assessments. Frequently, MEIs were assumed to live indefinitely at the location of maximum facility impact and partake heavily in activities so as to assume a high degree of exposure. Under the impression that MEIs were overly conservative, the concept of Reasonable Maximum Exposed Individual (RMEI) has evolved. Adopted from Superfund risk assessment methodologies, the goal of constructing an RMEI is to derive an appropriate mix of assumptions and parameters at average, conservative, and extreme values so that the overall exposure profile is conservative yet still plausible. Quoting from U.S. EPA (1989) guidance:

> ". . . *For Superfund exposure assessments, intake variable values for a given pathway should be selected so that the combination of all intake variables results in an estimate of reasonable maximum exposure for that pathway.* As defined previously, reasonable maximum exposure (RME) is the maximum exposure that is reasonably expected to occur at a site. Under this approach, some intake variables may not be at their individual maximum values but when in combination with other variables will result in estimates of RME. . . ." (*emphasis* in the original)

The construction of an RMEI exposure profile is thus subjective, since specific guidance on the mix of assumptions is not provided (although there has been an attempt to formulate some standard assumptions, as discussed in the following Section). However, there is no easy way (if any) to measure the overall degree of conservatism embedded within the RMEI. Consequently, RMEI risk estimates may fail to convey meaning since it may be impossible to gauge where the RMEI fits into the population at risk. Perhaps more importantly, RMEI analyses are not what most numbers of the public want or comprehend. In our experience, people who live near an existing or proposed combustor don't want to know what their risks are *less than* — they want to know what their risks are.

Standard Assumptions and Site-Specific Factors

The latitude permitted for the construction of RMEIs conflicts with one of the prime objectives of many regulators — that of consistency. Especially germane to large programs such as Superfund, fairness dictates that, with all other factors being equal, similarly contaminated sites in identical locales should be treated the same, and hence, should yield equal estimates of risk. Also, the risk estimates for different sites (or for different remedial scenarios of the same site) should, to the degree appropriate, be amenable to relative comparison.

For these and other reasons, the U.S. EPA (1991) has developed standard default exposure assumptions to be used in the absence of site-specific information. Although this guidance provides a degree of uniformity to risk assessments that can be useful to regulators, there are disadvantages. The specification of standard default values offers convenience and easy justification, which diminishes the incentive to gather site specific data that could augment the relevance of the risk assessment. Also, the repeated use of the same parameters and assumptions makes them more familiar over time, and this familiarity can lead to an unjustified sense of confidence. Many parameters and assumptions used in risk assessments remain highly uncertain. For example, widely varying estimates of incidental soil ingestion rates have been reported. But insisting on the use of default values curtails the motivation to even consider uncertainty, let alone to research and understand it.

Modeling PCDDs/PCDFs

In addition to the recent research on vapor absorption by plants described in Section 2.1, various other aspects of PCDDs and PCDFs are under investigation, and studies can be expected to provide more detailed information on fate and transport behavior in the environment and on toxicity to humans and others. Perhaps the most significant trend is the distinction of the individual dioxin and furan congeners. Traditionally, all dioxin and furan congeners have been assigned relative toxicity factors based on structure-activity relationships and treated as a single compound (2,3,7,8-tetrachlorodibenzo-p-dioxin [2,3,7,8-TCDD]) in environmental modeling. Recent studies, however, are providing information on the behavior of individual congeners, and there is a growing trend to use this information and model each congener separately in multi-step food-chain pathways. In most cases, the distribution of dioxin and furan congeners emitted from the facility is assumed to persist to the point of deposition. Further research may evaluate the potential for selective degradation of congeners in the atmosphere.

EVALUATING ACCEPTABILITY FOR DETERMINISTIC RISK ASSESSMENTS

Having been a witness to the evolution of risk assessments for municipal solid waste combustors, one of our clients remarked, with frustration, that although the projected emission rates of new facilities have dropped dramatically over time (primarily because of advances in control technologies), estimates of human health risk from combustors have remained more or less the same. Although there are a number of factors that have led to this paradox, the foremost has been the fact that while the level of acceptable risks has remained relatively constant, the typical scope of risk assessments has increased significantly over time.

Despite the many complexities of risk management, the 10^{-6}–10^{-4} range of acceptable excess cancer risk to RMEIs as defined by the Superfund program is both pervasive and widely applied. An RMEI excess cancer risk of 10^{-5} has been adopted in a number of regulatory frameworks and, as a rule of thumb, can be thought of as a generic target level.

A decade or less ago, it would have been sufficient to demonstrate that risks from the inhalation pathway alone would meet target risk criteria. Today, the combined risks of a multipathway risk assessment are expected to meet the same criteria. Since, for many chemicals, indirect exposures are much larger than inhalation exposure, more stringent emission rates are necessary to satisfy the target risk criteria.

As the process of encompassing more detailed exposure calculations continues (e.g., the need to consider wet deposition, vapor absorption, higher bioconcentration factors, etc.), additional pressure is placed on reducing the overall conservatism of the risk assessment so that the same target criteria can be met. For example, RMEIs may be restricted to the locations of actual farms, relaxing a previous assumption that a person living at the point of maximum facility impact may at some point decide to keep livestock. At some point, the mix of exposure assumptions may appear to lose its conservative edge. Unfortunately, current deterministic methods do not provide any means to gauge the degree of conservatism in risk estimates, and arguments over the appropriateness of assumptions cannot be resolved rationally.

There are various ways to address to this dilemma. One solution is to redefine target risk criteria as new burdens are placed upon risk assessments. This solution is not likely to be embraced by either regulators or the public; target risk criteria are generally perceived as absolute goals to be met in all circumstances. A second solution is to limit and/or fix the requirements of the risk assessment, but this alternative also fails to satisfy the notion that target risk criteria should be absolute (inclusive) goals.

Another alternative, which we see as the most promising, is the application of probabilistic risk assessment techniques. By dealing directly with uncertainties, probabilistic risk assessments can provide a wealth of information that is not accessible from deterministic analyses. The advantages of probabilistic techniques are elaborated in the following sections.

PROBABILISTIC RISK ASSESSMENT

Probabilistic risk assessment techniques have been available for some time, and have been used extensively by the nuclear power industry (U.S. NRC, 1983). Probabilistic techniques differ from deterministic algorithms by explicitly considering variability and uncertainty in parameter values and models. So doing, the results of the risk assessment are no longer limited to point (deterministic) estimates, but rather are a distribution of possible values.

Compared with a point estimate, a distribution of values provides a greater amount of information. Percentile values, ranges, and other statistical measures can be used to characterize likelihood and uncertainty. For example, 95th or 99th percentile values may be used to characterize the risks due to reasonable maximum exposure; the risk due to average-case exposure can be gauged by the median of the distribution.

Each estimate of risk has its different uses. Worst-case estimates provide information as to extreme patterns of facility operations, resulting exposures, and expected toxicities. Median-case estimates are more useful for comparing to risks posed by other sources and types of air pollution, especially when those risks are based on quantifiable effects in exposed populations. Properly combined with demographic information, probabilistic methods can be used to derive estimates of population-weighted risk and distributions of risk to specific segments of the population.

The Monte Carlo method is perhaps the best-known probabilistic technique. In a Monte Carlo simulation, distributions are specified for each parameter that account for both variability and uncertainty. A single instance of risk is computed by: (1) selecting random values for each parameter from the distributions and accounting for any correlations between variables (e.g., food consumption rates may be partially related to body weight); and (2) calculating the value of risk (using deterministic models and relationships) with the set of random values for that instance. By conducting a large number of determinations, a distribution of risk values is generated from which it is possible to determine relevant statistical measures.

Variability and Uncertainty

Variability and uncertainty are associated with practically all exposure parameters. The differentiation of these terms is important. Variability is the measured variation between members of a defined population that leads to potential differences in risks. Uncertainty is the combination of all other effects that lead to variations in risk estimates for the defined population. These definitions are necessarily somewhat arbitrary, since the distinction between the two can become blurred (it may not be known whether different measurements arise because of variability in the parameter being measured or from uncertainty in the measurement of that parameter), and some effects that contribute to variability for some populations instead contribute to uncertainties in others.[2]

As discussed in Section 2.2, the standard way of accounting for both uncertainty and variability has been to choose point estimates for some parameters that come from relatively extreme values of the variability or uncertainty distributions characterizing some of those parameters, and central values for others (U.S. EPA, 1989). This method leads to the concept of selection of individuals or populations who are RME (reasonably maximally exposed) receptors. Incorporating complete variability and uncertainty distribution allows evaluation of how extreme the selected RME individuals or populations are; and also allows explicit estimation of the remaining uncertainty and variability for the RME receptors themselves.

A Simple Example of a Monte Carlo Simulation

As a simple example, consider the problem of estimating the risk to one of the 400,000 or so individuals in California affected by dibromochloropropane (DBCP) in their water supply.[3] DBCP has been shown to cause cancers in laboratory animals given reasonably large doses, and so it is assumed to also be a human carcinogen, and to be so at even very small doses.

Measurements made in Californian water supplies show that DBCP contaminates the water supplies used by about 400,000 people, although the concentration in those water supplies varies considerably. For any individual chosen at random from the exposed population, the likelihood that one's exposure lies in any given range can be well approximated by a lognormal distribution. In particular, the probability that the concentration in their water lies below a concentration C mg/liter can be approximated by

$$\Phi(0.562 \ln (C/C_0))$$

where Φ is the standard normal function and $C_0 = 1.62 \times 10^{-4}$ mg/liter

Individuals using the contaminated water will be exposed to DBCP principally through drinking, showering (DBCP escapes into the air, and may be inhaled), and through dermal contact (principally in showers). Different individuals drink different quantities of water, and have differing showering habits, so that their exposures differ even for similar concentrations in the water supply. These differences can be approximated by lognormal probability distributions for the effective amounts of contaminated water that result in exposure by these three routes:

Ingestion:	Median 0.028 liters/kg-day, geometric standard deviation a factor of 1.4
Inhalation:	Median 0.03 liters/kg-day, geometric standard deviation a factor of 3
Skin contact:	Median 0.022 liters/kg-day, geometric standard deviation a factor of 2.

The total effective rate of contact (E_e, in liters/kg-day) with contaminated water is obtained by summing these three ($E_e = E_{ing} + E_{inh} + E_{sk}$).

The State of California has estimated the carcinogenic potency to humans of DBCP from a mouse experiment by applying various extrapolations, all of which involve some uncertainty. By adopting a relatively conservative attitude, they obtained an estimate for P, the human carcinogenic potency, of 7 kg-day/mg. This value is explicitly designed to be high, so that it is likely that the actual carcinogenic potency in humans is lower. It is possible to evaluate the uncertainty distribution for all the steps taken in California's procedure of estimating carcinogenic potency in humans, although the resulting combined uncertainty distribution cannot be as easily described as in the preceding case.

A typical approach to obtaining a point estimate of risk (R) for an individual (analogous to the estimates that would be obtained for a Superfund site) may be obtained by making point estimates for each of the variables described above, and multiplying them together.

[2]As an example, the variation in individual bodyweight contributes to variability of a risk estimate in a population if the distribution of bodyweights in that population is known. For an individual *of a known bodyweight in* the population, this variability is removed. However, for an individual *of unknown* bodyweight, the distribution of bodyweights contributes to the uncertainty in risk estimates.

[3]DBCP is a formerly widely used pesticide—not a contaminant emitted from combustors. The example is offered to illustrate a simple Monte Carlo analysis. Such analysis could be applied to any contaminant in any medium.

$$R = PE_eC = P(E_{ing} + E_{inh} + E_{sk})C$$

This also incorporates extra assumptions that we are not concerned with here: the individual stays in the same place for a lifetime; the measured concentration does not change with time; and so forth. The point estimates used would depend on the type of estimate required, but would usually be conservative (i.e. high). For a Reasonably Maximally Exposed (RME) individual, as used in the Superfund program, one would probably select the 95th percentile of the concentration distribution (0.003 mg/liter in the samples from the water supplies), or an even higher percentile, with the means of the ingestion, inhalation, and skin contact exposures (the mean of the sum is 0.094 liters/k-day), together with the point estimate of potency. The resultant point estimate of lifetime risk is simply the product $7 \times 0.003 \times 0.094 = 0.002$.

A more complete picture of individual risk emerges, however, if account is taken of all the uncertainties and variability. In this problem, the main source of variability between individuals arises from the exposure concentration distribution, and the main source of uncertainty is in the estimate of human carcinogenic potency. For a random individual in the exposed population, the inter-individual variability has to be interpreted as an uncertainty (since for a random individual, the actual exposure concentration is not known except with an uncertainty corresponding to the variability between individuals), so that both of these distributions may be interpreted as uncertainty distributions. The complete uncertainty distribution for individual risk may thus be obtained as

$$R = P \otimes (E_{ing} \oplus E_{inh} \oplus E_{sk}) \otimes C$$

where the symbol $\oplus$ indicates a convolution integral for the sum of two distribution functions, and $\otimes$ indicates a convolution integral for the product of two distribution functions.

Exact evaluation of such convolution integrals is not usually possible, so that an approximation is necessary. One standard and very convenient approximation method is the Monte Carlo simulation. A sample is drawn from each distribution going into the risk estimate, and the risk estimate evaluated using those sample values. This is repeated sufficiently often that a numerical approximation to the distribution of risk estimates is obtained, with the accuracy of the approximation inversely proportional to the number of samples. Figure 3 shows the results of such a simulation on our example, using 10,000 samples. The cumulative distribution for individual risk is shown as a solid line (left scale), and the differential distribution as a dotted line (arbitrary scale). The location of the point estimate is also shown (it is at approximately the 99th percentile of the complete distribution).

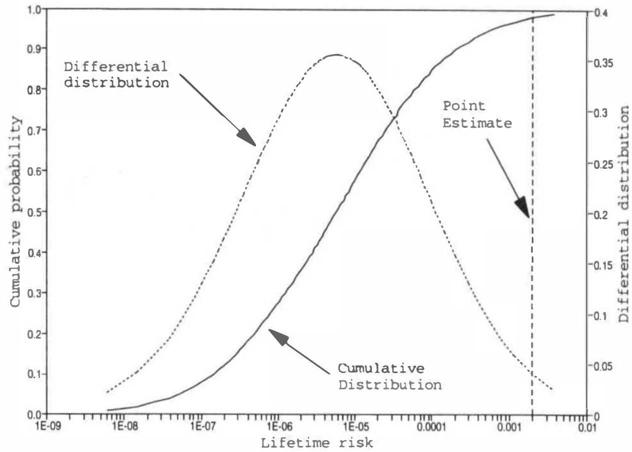

FIG. 3 RESULTS OF A MONTE CARLO SIMULATION FOR INDIVIDUAL RISK FROM DBCP

The distribution shown in Figure 3 must be interpreted carefully. For each value of risk (x-axis value) the cumulative curve shows the probability (y-axis value) that the lifetime risk to a randomly chosen exposed individual is smaller than the given value, subject to the assumptions going into this simulation (which include the caveats mentioned above, about the individual staying in the same place, the exposure staying the same, and so forth, together with various others that are more technical and even more important, such as assumptions about carcinogenic dose-response curves). This distribution applies only to randomly chosen individuals — it does not correspond to the uncertainty distribution for an individual exposed to a known concentration, for example, nor can it be used to evaluate the distribution of total population risks. In both the latter examples, the inter-individual variability in concentration either does not enter the problem, or enters it in a different way that must be correctly accounted for.

A PROBABILISTIC RISK ASSESSMENT OF THE WASTE TECHNOLOGIES INDUSTRIES (WTI), EAST LIVERPOOL, OHIO

The WTI hazardous waste combustor is situated in the Ohio River Valley about 25 miles northwest of Pittsburgh, Pennsylvania, on the northwest bank of the Ohio River. The facility lies between the river and CONRAIL tracks in the East End of East Liverpool, Ohio, some $1\frac{1}{2}$ to $1\frac{3}{4}$ miles east to northeast of the main part of East Liverpool. Figure 4 shows the rail line of the East Liverpool area and was based on topological maps of the East Liverpool North and South quadrangle published by the U.S. Geological Survey. The Ohio River runs approximately ENE to WSW by the facility, at which point the valley is approximately 4000 feet wide and 300 feet deep, as shown in the approximate cross-section shown

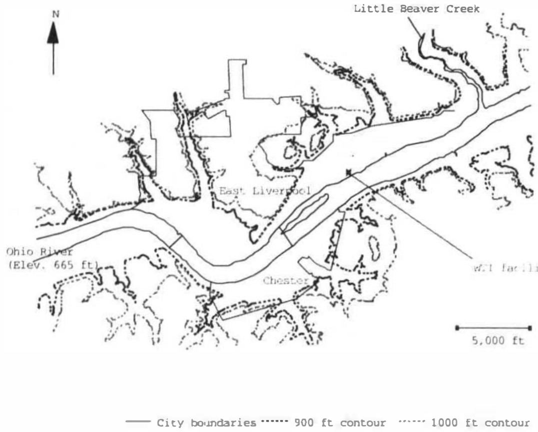

FIG. 4 EXISTING RAIL LINE CONNECTING TRANSFER STATION WITH FACILITY

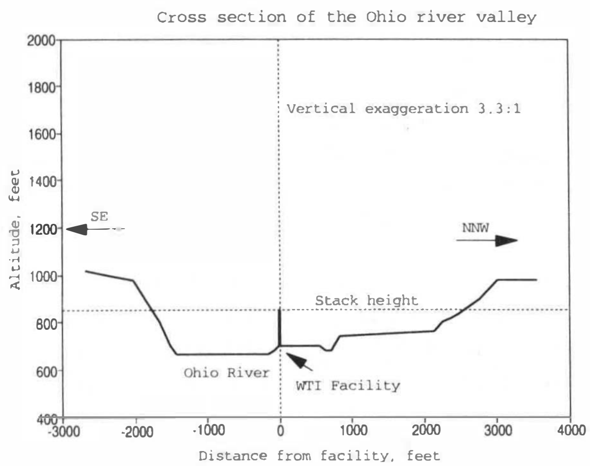

FIG. 5 APPROXIMATE CROSS-SECTION OF THE OHIO RIVER VALLEY AT THE LOCATION OF THE FACILITY

in Figure 5. The top of the 150-foot stack of the facility, about 850 feet above sea level, is 100 to 150 feet below the top of the nearby valley scarps that rise 250 to 350 feet above the river. Most of the main part of East Liverpool itself lies hidden from the facility by a steep valley wall rising to an altitude of 900 to 1000 feet above sea level.

Land use immediately adjacent to the WTI facility is industrial, but East End neighborhoods are quickly encountered to the west and north. Further downriver, the City of Chester, West Virginia is situated on the southwest bank of the Ohio River, and the main part of East Liverpool rises out of the valley to the north. Upriver, various communities are located on the banks of the Ohio River in West Virginia and Pennsylvania. Near the facility, especially to the south and directly across the River, the steep walls of the Ohio River Valley are largely forested. Outside of the valley, land use varies from residential in areas such as the outskirts of East Liverpool and small communities such as Lawrenceville, West Virginia, to the sparsely populated rural lands common in West Virginia and Pennsylvania.

The facility burns mixtures of hazardous waste, the nature of which is limited by operating permits and regulations. Hazardous waste streams are burned in a combustor that consists of a rotary kiln and a secondary combustion chamber. Combustor gases are routed through a heat recovery boiler, then through a spray dryer, an electrostatic precipitator (ESP), and a wet scrubber before being reheated (by steam) and exhausted through the stack. In addition, at various points downstream of the combustors, activated carbon is injected into the gas stream to help remove organic and some inorganic materials. The slurry from the wet scrubber is neutralized and fed into the spray dryer, while combustor slag and fly ash from the boilers, spray dryer, and ESP are disposed of off-site as solid waste.

Several risk assessments have already been performed for the WTI facility. The air permit, for example, is based in part on a risk assessment — an assessment demonstrating that permitted metals will be emitted at rates that will not cause exceedances of health-based air quality guidelines. A more detailed, preliminary risk assessment, performed by U.S. EPA Region V and ENVIRON Corporation (U.S. EPA, 1992), evaluated potential effects on an individual "maximally exposed" via inhalation to emissions from the WTI facility. More recently, various risk assessments were presented in Federal court (in the matter of Greenpeace, Inc., et al. vs. Waste Technologies Industries, et al., U.S. District Court, Northern District of Ohio, Eastern Division, Case No. 4:93CV0083), although these were limited to determining whether emissions of polychlorinated dibenzo-p-dioxins and furans during one year of operations would pose an imminent hazard to human health.

The U.S. EPA intends to perform a long-term multipathway risk assessment for the WTI facility based upon emission rates measured during trial operations. The results of the risk assessment may be used, in part, to determine final permit limitations. We anticipate that the U.S. EPA's risk assessment will be deterministic, and will include models and methodologies that are new to the U.S. EPA and relatively novel in the risk assessment field as a whole (such as wet deposition). An extreme degree of uncertainty may be associated with the risk estimates, and it is likely that the deterministic risk assessment will fail to provide meaningful measures of this uncertainty. In this case, an extreme burden will be placed on risk managers to judge the relevance of risk estimates. Should risk managers lack the technical training necessary to understand and interpret the complex series of models that are employed in the multipathway analysis, the risk estimates may appear uninformative, and important decisions may rest on non-technical matters.

As an alternative, we have derived a protocol for a probabilistic risk assessment of stack emissions from the WTI facility. Our goal is to perform the most informative, defensible risk assessment feasible — one that lays plain actual and plausible impacts.

To provide a broad characterization of the risks to health associated with operation of the WTI facility, two types of risk estimates will be calculated. The first, generated by the probabilistic methods, will be estimates for individuals in the population surrounding the WTI facility and for that population as a whole. The purpose of this estimate is to convey the range of risks associated with the range of exposure conditions anticipated to exist within the neighboring communities, and to illustrate the uncertainties of such risk estimates.

The second type of estimate (again generated by the probabilistic methods) will be for individuals and a population that might be highly exposed — in this case, those we designate as "farmers." This nomenclature is not meant to limit the analysis to those who farm, but indicates a lifestyle in which a substantial fraction of the foodstuffs consumed (vegetables, meat, and dairy products) comes from areas immediately adjacent to the place of residence. This population corresponds to the one that would be targeted by the standard "reasonably maximally exposed individual" (RMEI) approach. Parameter distributions used throughout the proposed risk assessment will include the values that might be used in such a standard RMEI evaluation, so we can easily generate the RMEI risk estimates by turning off all uncertainty and variability and always using the default point values.

The initial stages of our assessment are underway, and the key features of our assessment are described in the following sections.

Pollutant Emissions

The chemicals of concern must be first identified, and the emission rate of each chemical must then be characterized. As a general rule, chemicals will be considered if (1) they have been detected in stack testing of the WTI facility and other hazardous waste combustors and (2) toxicological data are available for their evaluation. A preliminary list of chemicals includes 49 organic compounds (counting all PCDD/PCDF congeners as a single class), 12 metals, and the gases HCl and NOx. For most chemicals of concern, we will develop an expected distribution of the long-term average emission rate from the WTI facility for use in the Monte Carlo assessment. Data to be considered include the test results of trial burns conducted at the WTI facility and measurements taken at other hazardous waste combustors. For some compounds, insufficient information is available to estimate a meaningful distribution, so that only a point estimate of the emission rate will be used.

Air Dispersion Modeling and Deposition Modeling

The WTI facility is located on the Ohio River in a narrow valley, the walls of which, with respect to the surface of the river, are roughly twice the height of the stack of the WTI facility and rise rapidly from the valley floor. The setting is typical of other valleys in the region in which air stagnation episodes (inversions), which can trap pollutants and build-up significant concentrations of contaminants, are frequent. The heavy industry of the past — combined with uncontrolled emissions from cars and trucks — at times led to severe pollution of air within the valleys. One of the worst air pollution catastrophes in the twentieth century occurred in 1948 in Donora, PA, which is located south of Pittsburgh on the Monongahela River. Consequently, the air dispersion modeling study is of critical importance and demands careful consideration.

Since it serves as an integral step in the evaluation of every exposure pathway, the importance of the air dispersion analysis cannot be understated. However, air dispersion modeling is often granted only cursory attention within the framework of a multipathway risk assessment. This mis-match between significance and consideration stems from the fact that the methods of air dispersion modeling are well-established relative to other aspects of risk assessment. Although seldom discussed, there is some degree of uncertainty associated with even the most widely accepted models. This uncertainty is exacerbated by the valley setting of the WTI facility. There are no standard, validated models available to predict processes such as wet deposition of pollutants, or for estimating impacts beyond the first ridge of terrain, which is quickly encountered in the narrow Ohio River Valley. In addition, the complex meteorological factors that influence pollutant transport within and out of the valley preclude the straightforward application of standard models.

Consequently, hybrid approaches will be developed that will combine and modify several existing models. Several site-specific factors will be addressed by the modeling strategy. The limitations of standard complex terrain models will be recognized. In-valley meteorology and synoptic (regional-out-of-valley) flow conditions will be distinguished, including differences in both travel characteristics (wind speed, wind direction) and plume dispersion. The frequency of inversion conditions will be evaluated and modeled appropriately. Finally, meteorological data taken at (or near) the WTI facility will be used in an appropriate manner to characterize dispersion at plume height, and (if suitable) these data will be supplemented by data taken at a nearby facility — the Beaver Valley Power Station — and the Pittsburgh Airport to characterize synoptic conditions.

One strategy under consideration involves the use of a numerical model (e.g., WyndValley) to estimate pollutant dispersion within the valley using local meteorological data. Vertical (upward) fluxes of contaminants will be estimated out of top layer of the numerical model, which will be set to coincide with the height of the valley. These fluxes will then be used to construct virtual emission sources for use in standard gaussian plume models and, along with regional meteorological data, to estimate pollutant concentrations at locations outside of the valley.

All three modes of deposition discussed in Section 2.1 will be modeled. Both wet and dry deposition of particle-bound pollutants will be weighted by a particle size distribution measured during stack testing of the WTI facility. Wet deposition will consider precipitation patterns measured in the local area. Finally, the uptake of vapor-phase chemicals by vegetation will consider vapor-particle partitioning of pollutants and chemical-specific bioaccumulation factors (which are especially relevant to PCDD/PCDF congeners).

Fate and Transport Modeling

Additional mathematical models will be used to estimate pollutant concentrations in soil, water, vegetables and crops, beef and dairy products, and fish. As indicated in Figure 2, some of these models are interdependent, and all depend on the predictions of air dispersion and deposition modeling. The literature contains varying degrees and qualities of information for these models, and these data will be used to construct distributions for as many parameters as possible.

Evaluating Exposures and Doses

The risks to be assessed are to people presumed to be exposed to the chemicals of concern in the various media. People encounter different media, however, and even their exposures from similar environments differ. Exposures and risks will vary from person to person because people are dissimilar in age, recreational habits, occupation, the types of food they eat and where they obtain them, and their length of stay in various locations.

These various parameters (and others) may be evaluated for individuals in a population. But there is substantial variability among individuals of the population who may be exposed, and we cannot identify all the individuals and measure the relevant parameters for each one. Instead, we will rely on anonymous profiles for "individuals" in the population, and choose these profiles to be representative or otherwise relevant. Even for individuals with similar superficial characteristics, there may nevertheless be substantial variation from one individual to another.

Evaluating the Potential for Toxic Effects

For each chemical of concern and for each individual, there is a relationship between exposure or dose and probability (risk) of harm to health. The intensity, type, or risk of any particular health effect is likely to increase as the exposure or the dose increases — in other words, there is a dose-response relationship. With each chemical of concern, we associate an exposure or dose that is likely to be safe for non-carcinogenic effects, and another parameter that measures how likely is any given exposure or dose to cause cancer.

In essentially all cases, however, there is a tremendous uncertainty in the estimation of the dose that will cause a given effect in any particular individual, and there are other uncertainties due to the substantial variability among individuals. Even for large populations, there is uncertainty in the dose that will cause some effect in some fraction of the population, or the number of people that may be affected.

These uncertainties will be considered using the results of toxicological studies and employing the standard models used to evaluate the potential for carcinogenic and non-carcinogenic health effects. As a preliminary step, all non-carcinogenic effects will be grouped together; if necessary, these effects will be evaluated separately for different toxicological endpoints.

SUMMARY

Multipathway risk assessments are commonly performed for combustors of hazardous waste and municipal solid waste in an attempt to estimate their risks to human health. The deterministic methods that are typically employed, however, provide point estimates of risk that are difficult, if not impossible, to interpret because of the unknown degree of uncertainty embedded within the assessment. Probabilistic methods can provide a much greater level of information that can be used to assess (1) risks to populations, (2) the variability of risks to subpopulations, and (3) the level of uncertainty associated with risk estimates. We are currently engaged in a probabilistic multipathway risk assessment for the WTI hazardous waste combustor in East Liverpool, Ohio. We anticipate that the results of our assessment will provide meaningful information that could not be derived from a deterministic assessment, and may suggest changes to the typical methods used to perform risk assessments of combustors.

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REFERENCES

Baes, C. F., Sharp, R. D., Sjoreen, A. L., and Shor, R. E. (1984). A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture. Oak Ridge, TN: Oak Ridge National Laboratory. ORNL-5786.

Levin A., Fratt, D. B., Leonard, A., Bruins, R. J. F., and Fradkin, L.

(1991). Comparative analysis of health risk assessments for municipal waste combustors. *J. Air & Waste Mgmt. Assoc.* 41(1): 20–30.

McCrady, J. K., and Maggard, S. P. (1993). Uptake and photodegradation of 2,3,7,8-tetrachlorodibenzo-p-dioxin sorbed to grass foliage. *Environ. Sci. Technol.* 27: 343–350.

National Research Council (1983). Risk Assessment in the Federal Government: Managing the Process. Washington, DC: National Academy Press.

NYSDOH (1991). New York State Department of Health Guidance for Exposure Assessment of Municipal Solid Waste and Hospital Waste Incinerator Emissions. Albany, NY: New York State Department of Health, Bureau of Toxic Substance Assessment.

Rippen, G., and Wesp, H. (1993). Kale uptake of PCDD/PCDF, PCB and PAH under field conditions: Importance of gaseous dry deposition. In: *Proceedings of Dioxin '93, 13th International Symposium Chlorinated Dioxins and Related Compounds.* Vienna, Austria, September, 1993.

U.S. EPA (1989). Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual, Part A, Interim Final. Washington, DC: Office of Emergency and Remedial Response. EPA/540-1-89/002.

U.S. EPA (1990). Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions. Office of Health and Environmental Assessment. EPA/600/6 90/003.

U.S. EPA (1992). Preliminary Risk Assessment of Inhalation Exposures to Stack Emissions from the WTI Incinerator. U.S. Environmental Protection Agency, Region 5. July 1992.

U.S. EPA (1993). Draft Strategy for Combustion of Hazardous Waste.

U.S. NRC, (1983). PRA Procedure Guide — A Guide to the Performance of Probabilistic Risk Assessments for Nuclear Power Plants. Office of Nuclear Regulatory Research, Washington, D.C. NUREG/CR-2300-V1 and V2.

Welsch-Pausch, K., Umlauf, G., and McLachlan, M.S. (1993). An investigation of the deposition pathways of atmospheric PCDD/F to standardized grass culture. In: *Proceedings of Dioxin '93, 13th International Symposium Chlorinated Dioxins and Related Compounds.* Vienna, Austria, September, 1993.