Permit 3/19/10

ENTERED

STATE OF NEW MEXICO
BEFORE THE SECRETARY OF ENVIRONMENT

IN THE MATTER OF:

APPLICATION OF THE UNITED STATES	)	
DEPARTMENT OF ENERGY AND	)	
LOS ALAMOS NATIONAL SECURITY, LLC	)	**No. HWB 09-37(P)**
FOR A HAZARDOUS WASTE FACILITY	)	**HWB 10-04(P)**
PERMIT FOR LOS ALAMOS NATIONAL	)	
LABORATORY	)	

TESTIMONY OF RICH MIRENDA

My testimony will discuss the human-health and ecological screening assessment conducted as part of LANL's permit application for the TA-16 open burn units.

Qualifications

I have over thirty years experience in environmental and risk assessment work. I have been at LANL for 15 years as a subcontractor, contractor, or LANL employee (a technical staff member/scientist since 2005). As part of my duties, I integrate and coordinate risk assessment activities for projects, and provide guidance on sampling and analysis, quality assurance, exposure scenario/pathway development, and report writing. I regularly work with project leaders, project managers, the regulatory authority, United States Department of Energy, and Los Alamos National Laboratory personnel to ensure consistency, accuracy, and appropriateness of sampling, analysis, and assessment of data/risk for each site, aggregate, and/or watershed being investigated. I also review and conduct human health and ecological risk assessments for the Environmental Programs Directorate. A copy of my qualifications is attached to this testimony.

Direct Testimony

I was asked to conduct human-health and ecological risk screening assessments as part of LANL's RCRA permit application for the TA-16 open burn units. Risk assessment is a tool used to evaluate the likelihood of adverse effects that may occur or are occurring as a result of exposure to one or more stressors. A risk assessment, whether human health or ecological, is conducted to determine if constituents in various environmental media could cause harm to humans or animals that come into contact with them. A risk assessment identifies who, either human or animal, could be exposed to potential contaminants in a particular environmental media such as air, soil or water; how and how often the exposure could occur; how the contaminants affect health; what is the potential for risk; and what level of risk is considered acceptable (regulatorily). The results are reported either as a cancer risk (presented as a probability) or as a hazard (presented as a hazard index).

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The results of the risk assessment for the TA-16 open burn units are provided in the report titled "Human-health and Ecological Screening Assessment for the TA-16 Burn Ground, Revision 1" (LA-UR-10-00086). The risk assessment, which included both human health and ecological screening assessments, was submitted to the New Mexico Environment Department (NMED) by letter dated January 8, 2010 (Exhibit 1). The risk assessment was conducted using NMED accepted guidance and methods. The report sets forth the results of the risk assessment and includes supporting data.

The human health risk assessment considered exposure for both an industrial worker and a hypothetical resident who could be exposed through ingestion, inhalation, and dermal contact with contaminants in the soil. The data were compared with screening levels derived by NMED using accepted toxicity data and exposure parameters designed to be protective of the receptors. The resulting risk calculations indicated that both risk and hazard were below the NMED target levels. The screening assessments indicate that there is no potential unacceptable risk to human health at the TA-16 burn unit. NMED agrees with the conclusion of the report as to the human health risk assessment.

The ecological risk screening assessment considered exposure to a series of receptors representing different trophic levels and included plants, soil invertebrates, mammals, and birds. The data were compared with ecological screening levels (ESLs) based on similar species and derived from experimentally determined no observed adverse effect levels, lowest observed-adverse-effect levels, or doses determined lethal to 50% of the test population. The ESLs are conservative and designed to be protective but are not meant to indicate actual risk. Because of the conservatism built into the ESL calculations, the ratio of the contaminant concentration to the ESL (i.e., the hazard quotient) and the hazard index (HI), which is the sum of the ratios or hazard quotients for a receptor, are often greater than 1.0 and overestimate the potential risk to receptors. Additional analyses are typically conducted to determine if there is a likelihood of risk and whether field studies need to be implemented to see if actual risks are present.

The US Environmental Protection Agency (EPA) guidance requires that initial screening-level assessments use conservative assumptions to evaluate the potential for adverse ecological impacts. The rationale behind this requirement is to provide a high confidence that all potential adverse impacts to ecological receptors resulting from exposure to contaminants are identified in the initial investigations. Thus, the screening-level assessment may be used to identify sites that clearly pose no threat to the environment as well as sites that need corrective action. However, for the many sites that do not fall into one of these two categories, screening-level evaluations must be followed by a series of progressively more in-depth and site-specific evaluations to characterize risks accurately and to provide adequate information for risk management decisions. The screening-level assessment helps to focus these more detailed (and often more complex) site-specific investigations by identifying important contaminants, receptors, ecological endpoints, and spatial scales.

The ecological risk screening assessment initially found four chemicals of potential ecological concern (COPECs) at the TA-16 open burn area; barium, cadmium, silver, and the tetrachlorodibenzodioxin (TCDD) [2,3,7,8-] equivalent concentration. The HIs ranged from <1.0

to 25 based on these four COPECs, which indicated either no potential risk, low potential risk, or moderate potential risk for some receptors. Further analysis of the results using comparisons to background concentrations and potential effects to populations found that HIs for the deer mouse, shrew, earthworm, and plant were 10, 4, 2, and 6, respectively. Given the conservative nature of the ESLs, as described in the report and indicated above, the HIs are biased high. In addition, Dourson and Stara (Exhibit 2) conducted a study of uncertainty factors incorporated in calculating ESLs for ecological receptors. Based on their study, HIs up to 10 may not adversely affect ecological receptors. To maintain conservatism, they stated that HIs less than 3 do not adversely affect ecological receptors. The Dourson and Stara conclusion has been verified by previous studies conducted by LANL in various canyons where field studies have been performed to determine if actual risks are present following screening assessments. These studies have found no effects to small mammal populations at similar metals and dioxin/furan concentrations. These previous canyon studies concluded there is no potential ecological risk to the earthworm, plant, American robin, American kestrel, Mexican spotted owl, deer mouse, shrew, desert cottontail, and red fox. Therefore, it was concluded that no potential risks to ecological receptors are present at the TA-16 burn units.

The NMED statement that the area of elevated contamination identified by both the model and soil sampling predicted dioxin/furan concentrations an order of magnitude higher than either the TA-16 upper confidence limit (UCL) of the mean or the other LANL canyons, adding uncertainty to assessment of the areas of highest impact around the burn units, is misleading. Exposure and risk are not determined based on an individual sample location but by exposure across an area, as represented by the UCL. A small mammal does not spend all of its time at any given location; it covers a larger area referred to as its home range. In addition, EPA guidance is to manage the ecological risk to populations rather than to individuals, with the exception of threatened and endangered species. The population area is larger than an individual's home range and the UCL represents the "true average" concentration to which the small mammal populations are realistically exposed. Therefore, a location or locations may contain elevated concentrations of a contaminant without resulting in adverse effects to the individual or to the population.

In conclusion the weight of evidence presented in the report to NMED indicates there is no risk to the ecological receptors evaluated for the TA-16 open burn units.

FURTHER AFFIANT SAYETH NAUGHT

Richard J. Mirenda

Richard J. Mirenda

STATE OF NEW MEXICO)
) ss.
COUNTY OF LOS ALAMOS)

SUBSCRIBED, SWORN TO AND ACKNOWLEDGED before me this 18th day of March, 2010

NOTARY PUBLIC

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My Commission Expires:

12/04/2011

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Date: January 8, 2010
Refer To: ENV-RCRA-10-006
LA-UR: 10-00086

Mr. James Bearzi, Chief
Hazardous Waste Bureau
New Mexico Environment Department
2905 Rodeo Park Drive East, Building 1
Santa Fe, New Mexico 87505-6303

Dear Mr. Bearzi:

SUBJECT: TRANSMITTAL OF HUMAN-HEALTH AND ECOLOGICAL SCREENING ASSESSMENT FOR THE TECHNICAL AREA 16 BURN GROUND, REVISION 1

The purpose of this letter is to transmit the revised screening assessment for the Technical Area (TA) 16 open burning treatment units included within the draft Los Alamos National Laboratory Hazardous Waste Facility Permit. This revision includes additional data and figures as requested by the New Mexico Environment Department- Hazardous Waste Bureau (NMED-HWB) on January 7, 2010 by teleconference and email. This assessment concludes that no potential unacceptable risks to human and ecological receptors are present at the TA-16 Burn Ground.

The enclosed human-health and ecological screening assessment for the TA-16 Burn Ground includes figures illustrating all sample locations; toxicity equivalent concentrations for dioxin and furan data; and total concentration values for barium, cadmium and silver. The assessment revision also incorporates further spatial analysis and avian receptor data. This advanced assessment also makes mention of the following considerations.

- Small mammal studies conducted within the area indicate that populations within the area are healthy.
- Elevated concentrations could likely be due to historic operations at the site.
- Previous studies have found no effects to small mammal populations at similar dioxin/furan concentrations in the canyons.

The following are additional factors that contribute to reducing the risk to human and ecological receptors at the TA-16 Burn Ground.

- Historical operations at the TA-16 Burn Ground included other open burning operations that were previously closed under the Resource Conservation and Recovery Act or are scheduled for clean-up under the corrective action program.

- The current and proposed future annual volumes of waste to be treated at the TA-16 Burn Ground are significantly decreased from past operations and are therefore unlikely to adversely impact the concentration of contaminants in the soil surrounding the units.
    - Decreases to the total volume of waste treated at the TA-16 Burn Ground has been detailed in previous submittals to the NMED-HWB and is due to strong efforts in total waste minimization, careful waste segregation, and the substitution of nonhazardous materials for high explosive research and processing operations.
- Types of waste streams treated and waste treatment methods used in the past were more likely to have impacts to soil than waste streams that are currently treated or slated for treatment within the future.
    - Past waste streams included more material such as wooden pallets, plastics, metal pieces, paper, and solvents. Waste treatment, prior to the installation of the propane burners at TA-16-388, included the addition of wood and a petroleum fuel such as kerosene. Current operations do not require fuel other than propane at TA-16-388 and only a small amount of a petroleum based fuel at TA-16-399. Waste streams have been eliminated through waste minimization efforts mentioned above, as well as increased waste generator education on minimizing the potential for high explosives contamination.

A compact disc is also included with this submittal that includes an electronic copy of the assessment discussed above.

Please contact Jack Ellvinger at (505) 667-0633 of the Water Quality and RCRA Group (ENV-RCRA) if you have questions.

Sincerely,

Anthony R. Grieggs
Group Leader
Water Quality & RCRA Group (ENV-RCRA)

ARG:LRVH/lm

Enclosures: a/s

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ENCLOSURE

HUMAN-HEALTH AND ECOLOGICAL SCREENING ASSESSMENT FOR THE TA-16 BURN GROUND

REVISION 1

(LA-UR-10-00086)

1.0 HUMAN-HEALTH SCREENING ASSESSMENT

A human-health risk screening assessment for Technical Area 16 (TA-16) Burn Ground is presented in the following sections.

1.1 Conceptual Site Model

Only authorized Laboratory workers currently have access to the area around the TA-16 Burn Ground so the predominant land use is industrial. Therefore, Laboratory workers are the primary receptors and the industrial scenario is the defining scenario for the human health risk screening assessment (i.e., the scenario on which decisions are based). Because the site is located within the boundaries of an operational facility (TA-16), the reasonably foreseeable future land use will continue to be industrial. Residential exposure is also assessed and provided for comparison purposes.

Potential exposure pathways for a site worker (as well as a hypothetical resident) include incidental ingestion of soil, inhalation of fugitive dust or vapors, dermal contact with soil, and external irradiation by gamma-emitting radionuclides. Inhalation of gas-phase contaminants, such as tritium and/or organic chemicals emanating from the site into the atmosphere, is not a potential means of exposure. Other potential pathways from subsurface releases to potential receptors would be complete only if soil were to be excavated and brought to the surface. In such a case, the potential contaminant migration pathways and potential exposure pathways would be the same as those of a surface soil release.

The primary ecological exposure pathways include root uptake, ingestion of contaminated soil, and food-web transport.

1.2 Identification of Chemicals of Potential Concern

1.2.1 Sampling

Thirty-seven surface samples (0-1 ft) were collected from 37 locations across the TA-16 Burn Ground (Figure 1.2-1) and analyzed for Resource Conservation and Recovery Act (RCRA) metals and/or dioxin and furan congeners. One sample (09RCRA462) is a background sample and is therefore not included in the calculation of the risks from dioxin and furan congeners. The data used are summarized in the following paragraphs and are evauated in the human health and ecological risk screening assessments.

1.2.2 Evaluation of Inorganic Chemicals

Thirty-one surface samples were collected from 31 locations and analyzed for RCRA metals. Table 1.2-1 presents the comparison of the inorganic chemicals to soil background values (BVs) (LANL 1998, 059730). Barium, cadmium, and silver were detected above BVs in at least one soil sample (Figures 1.2-2, 1.2-3, 1.2-4). These three inorganic chemicals are retained as chemicals of potential concern (COPCs).

1.2.3 Evaluation of Organic Chemicals

Thirty-six surface samples were collected from 36 locations and analyzed for dioxin/furan congeners (Figure 1.2-5). Dioxin/furan congeners were detected in each sample (Table 1.2-2). The number of congeners detected ranged from 3 to 17. All detected dioxin/furan congeners were retained as COPCs.

1.3 Screening Evaluation

The exposure point concentration (EPC) for the dioxin and furan congeners is the sum of the detected congeners weighted by the World Health Organization (WHO) 1995 Toxic Equivalent Factors (TEFs) (http://www.epa.gov/ncea/pdfs/dioxin/part2/drich9.pdf); the sum is expressed as the tetrachlorodibenzodioxin (TCDD) [2,3,7,8-] equivalent concentration. The TEFs used are presented in Table 1.3-1 and the TEF calculations are presented in Table 1.3-2.

The EPCs are the 95% upper confidence limit (UCL) of the arithmetic mean. All samples were collected from 0-1 ft and all of the data are used to calculate 95% UCLs for the industrial and residential risk screening assessments. The 95% UCL for dioxin and furan congeners was calculated using the TCDD[2,3,7,8-] equivalent concentration for each sample (Table 1.3-2).

The 95% UCLs for each COPC were calculated as described in U.S. Environmental Protection Agency (EPA) guidance (EPA 2002, 073593). Tests for distributions were performed using ProUCL 4.00.04 to determine the appropriate method for UCL calculations and the recommended UCLs were used (Figures 1.3-1, 1.3-2, 1.3-3, and 1.3-4). The 95% UCLs for each COPC are presented in Table 1.3-3. The following methods were used to calculate 95% UCL concentrations (depending on the type of distribution found for the data set):

- Student's t-statistic procedure – normal distributions
- Chebyshev (Mean, Sd) procedure – nonparametric distributions
- H-UCL procedure – lognormal distributions

The EPC for each COPC was compared with the industrial and residential soil screening levels (SSLs).The chemical SSLs used in the evaluations were obtained from New Mexico Environment Department (NMED) guidance (NMED 2009, 106420). The SSLs for carcinogens are equivalent to a 1 x 10^{-5} cancer risk and for noncarcinogens represent a hazard quotient (HQ) of 1.0. The comparisons with SSLs are conducted separately for carcinogens and noncarcinogens for industrial and residential receptors (Tables 1.3-4 through 1.3-7).

The EPCs for noncarcinogenic COPCs were less than their respective industrial and residential SSLs. The hazard indices (HIs) for the noncarcinogenic COPCs are approximately 0.004 and 0.06, respectively (Tables 1.3-4 and 1.3-5), which are less than NMED's target HI of 1.0 (NMED 2009, 106420).

The EPCs for carcinogenic COPCs were less than their respective industrial and residential SSLs. The total excess cancer risks from exposure to carcinogenic COPCs are approximately 3 x 10^{-7} and 2 x 10^{-6}, respectively (Tables 1.3-6 and 1.3-7), which are less than the NMED target risk level of 1 x 10^{-5} (NMED 2009, 106420).

1.3.1 Uncertainty Analysis

The analysis for human health is subject to uncertainties associated with data evaluation, exposure assessment, and toxicity values. Each or all of these uncertainties may affect the assessment results.

1.3.1.1 Data Evaluation

Data evaluation uncertainties may include errors in sampling, laboratory analysis, and data analysis. Although concentrations used in this risk assessment were less than estimated quantitation limits for some COPCs, data evaluation uncertainties are expected to have little effect on the assessment results.

The J (estimated) qualification of detected concentrations of some organic COPCs does not affect the assessment.

Another data evaluation uncertainty relates to the use of the 95% UCL as the EPC for each COPC. Use of the 95% UCL may result in an overestimation of risk for analytes that have elevated detection limits. Use of the maximum concentration also overestimates the exposure to contamination. Receptors are not exposed to these concentrations across the site.

1.3.1.2 Exposure Assessment

The receptors used in the assessment are subject to exposures in a different manner than the exposure assumptions used to derive the SSLs. Assumptions for the industrial SSLs are that the potentially exposed individual is a Laboratory worker who is outside for 8 h/d for 225 d/yr (NMED 2009, 106420) and spends the entire 8 h on-site within the contaminated area. Because it is unlikely that the worker is within the contaminated area for the entire work day, the screening assessment overestimates the exposure. As a result, risk, hazard, and dose may be overestimated.

Assumptions underlying the exposure parameters, routes of exposure, amount of contaminated media available for exposure, and intake rates for routes of exposure are consistent with NMED parameters and default values (NMED 2006, 106420). In the absence of site-specific data, several upper-bound values for the assumptions may be combined to estimate exposure for any one pathway, and the resulting risk estimate can exceed the 99th percentile. Therefore, uncertainties in the assumptions underlying the exposure pathways may contribute to risk assessments that exceed the reasonably expected range.

1.3.1.3 Toxicity Values

The primary uncertainty associated with the screening values is related to the derivation of toxicity values used in their calculation. Toxicity values (slope factors [SFs] and reference doses [RfDs]) were used to derive the risk-based screening values used in the screening evaluation (NMED 2009, 106420). Uncertainties were identified in four areas with respect to the toxicity values: (1) extrapolation from other animals to humans, (2) interindividual variability in the human population, (3) the derivation of RfDs and SFs, and (4) the chemical form of the COPC.

The SFs and RfDs are often determined by extrapolation from animal data to humans, which may result in uncertainties in toxicity values because differences exist between animals and humans in chemical absorption, metabolism, excretion, and toxic responses. Differences in body weight, surface area, and pharmacokinetic relationships between animals and humans are taken into account to address these uncertainties in the dose-response relationship. However, conservatism is usually incorporated in each of these steps, resulting in the overestimation of potential risk.

For noncarcinogenic effects, the degree of variability in human physical characteristics is important both in determining the risks that can be expected at low exposures and in defining the no-observed-adverse-effect level (NOAEL). The NOAEL uncertainty factor approach incorporates a 10-fold factor to reflect individual variability within the human population that can contribute to uncertainty in the risk assessment. This factor of 10 is generally considered to result in a conservative estimate of risk to noncarcinogenic COPCs.

The SFs and RfDs are often determined by extrapolation from animal data to humans, which may result in uncertainties in toxicity values because differences exist between other animals and humans in chemical absorption, metabolism, excretion, and toxic response. Differences in body weight, surface area, and pharmacokinetic relationships between animals and humans are taken into account to address

these uncertainties in the dose-response relationship. However, conservatism is usually incorporated into each of these steps, resulting in the overestimation of potential risk.

COPCs may be bound to the environmental matrix and not available for absorption into the human body. However, the exposure scenarios default to the assumption that the COPCs are bioavailable. This assumption can lead to an overestimation of the total risk.

1.3.1.4 Additive Approach

For noncarcinogens, the effects of exposure to multiple chemicals are generally unknown and possible interactions could be synergistic or antagonistic, resulting in either an overestimation or underestimation of the potential risk. Additionally, RfDs used in the risk calculations typically are not based on the same endpoints with respect to severity, effects, or target organs. Therefore, the potential for noncarcinogenic effects may be overestimated for individual COPCs that act by different mechanisms and on different target organs but are addressed additively.

1.3.2 Interpretation

Based on an industrial scenario, the HI (0.004) is less than NMED's target level of 1.0 and the cancer risk (3×10^{-7}) is less than the NMED target level of 1×10^{-5}. For the residential scenario, the HI (0.06) is less than NMED's target level of 1.0 and the cancer risk (2×10^{-6}) is less than the NMED target level of 1×10^{-5}. The screening assessments indicate that there is no potential unacceptable risk to human health at the TA-16 Burn Ground.

2.0 ECOLOGICAL SCREENING ASSESSMENT

An ecological risk screening assessment for the TA-16 Burn Ground is presented in the following sections.

2.1 Screening Evaluation

The ecological risk-screening evaluation identifies chemicals of potential ecological concern (COPECs) and is based on the comparison of EPCs to ecological screening levels (ESLs) in accordance with Laboratory guidance (LANL 2004, 087630). The EPCs used in the assessment are presented in Table 1.3-3 and the calculation is described in section 1.3. The ESLs obtained from ECORISK Database, Version 2.3 (LANL 2008, 103352) are presented in Table 2.1-1. In addition, the avian ESLs for TCDD[2,3,7,8-] from ECORISK Database, Version 2.0 (LANL 2003, 080117) are presented in Table 2.1-1. The ESLs are based on similar species and are derived from experimentally determined NOAELs, lowest-observed-adverse-effect levels (LOAELs), or doses determined lethal to 50% of the test population. Information relevant to the calculation of ESLs, including concentration equations, dose equations, bioconcentration factors, transfer factors, and TRVs are presented in the ECORISK Database, Versions 2.0 and 2.3 (LANL 2003, 080117; LANL 2008, 103352).

The HQs calculated for each COPEC and screening receptor are the ratios of the EPC to the ESLs for each ecological receptor. The higher the contaminant levels relative to the ESLs, the higher the potential risk to receptors; conversely, the higher the ESLs relative to the contaminant levels, the lower the potential risk to receptors. The analysis begins with a comparison of the minimum ESL for each COPC to the EPC. HQs greater than 0.3 are used to identify COPECs requiring additional evaluation (LANL 2004, 087630). Individual HQs for a receptor are summed to derive an HI; an HI greater than 1.0 is an indication

that further assessment may be needed to be sure that exposure to multiple COPECs at a site will not lead to potential adverse impacts to a given receptor population. The HQ and HI analysis is a conservative indication of potential adverse effects and is designed to minimize the potential of overlooking possible COPECs at the site.

The HQs using the minimum ESLs exceeded 0.3 for barium, cadmium, silver, and TCDD[2,3,7,8-] (Table 2.1-2). All of the COPCs are retained as COPECs. An HQ for each COPEC/receptor combination was calculated and summed to obtain an HI for each receptor. The HI is the sum of HQs for chemicals with common toxicological endpoints for a given receptor. It is assumed for the purposes of ecological screening, that nonradionuclides have common toxicological effects and that HQs may be added. The calculations indicate that receptors, except the kestrel, robin (herbivore and omnivore), and cottontail, have HIs greater than 1.0 (Table 2.1-3). The results are discussed further in the uncertainty section.

2.2 Uncertainty Analysis

2.2.1 Chemical Form

The assumptions used in the ESL derivations are conservative and not necessarily representative of actual conditions. These assumptions include maximum chemical bioavailability, maximum receptor ingestion rates, minimum bodyweight, and additive effects of multiple COPECs. These factors tend to result in conservative ESL estimates, which may lead to an overestimation of the potential risk. The assumption of additive effects for multiple COPECs may result in an over- or underestimation of the potential risk to receptors.

The chemical form of the individual COPCs was not determined as part of the investigation. Toxicological data are typically based on the most toxic and bioavailable chemical species, which are not typically found in the environment. Inorganic, organic, and radionuclide COPECs are generally not 100% bioavailable to receptors in the natural environment because of interference from other natural processes, such as the adsorption of chemical constituents to matrix surfaces (e.g., soil) or rapid oxidation or reduction changes that render harmful chemical forms unavailable to biotic processes. The ESLs were calculated to ensure a conservative indication of potential risk (LANL 2004, 087630), and the values are biased toward overestimating the potential risk to receptors.

2.2.2 Exposure Assumptions

The EPCs used in the calculations of HQs are the 95% UCL concentrations. These EPCs are conservative estimates of exposure to each COPC. The sampling efforts focused on areas of known contamination, and receptors were assumed to ingest 100% of their food and spend 100% of their time at the site. These assumptions regarding the exposure for terrestrial receptors within the TA-16 Burn Ground are likely to result in an overestimation of potential ecological exposure and risk.

2.2.3 Toxicity Values

The HQs were calculated using ESLs, which are based on NOAELs as threshold effect levels; actual risk for a given COPEC/receptor combination occurs at a higher level, possibly somewhere between the NOAEL-based threshold and the threshold based on the LOAEL. The use of NOAELs leads to an overestimation of potential risk to ecological receptors. ESLs are based on laboratory studies requiring extrapolation to wildlife receptors. Laboratory studies are typically based on "artificial" and maintained populations with genetically similar individuals and are limited to single chemical exposures in isolated and controlled conditions using a single-exposure pathway. Wild species are concomitantly exposed to a

variety of chemical and environmental stressors, potentially rendering them more susceptible to chemical stress. On the other hand, wild populations are likely more genetically diverse than laboratory populations, making wild populations, as a whole, less sensitive to chemical exposure than laboratory populations. The uncertainties associated with the ESLs tend to lead to an overestimation of potential risk.

The avian ESLs for TCDD[2,3,7,8-] obtained from ECORISK Database, Version 2.0 (LANL 2003, 080117) are based on a toxicity value using intraperitoneal injections. This route of exposure does not occur naturally and assumes that 2,3,7,8-TCDD bioavailability and absorption from the gastrointestinal tract and the abdominal cavity are not significantly different. However, exposure by this route likely overestimates the potential absorption of TCDD[2,3,7,8-] by the receptor and thereby overestimates the potential effect on the receptor.

2.2.4 Comparison of EPCs to Background Concentrations

The ecological risk-screening assessments are based on the exposure of ecological receptors to contamination to a depth of 5 ft bgs. The EPCs of some of the inorganic COPECs are similar to background concentrations, indicating that exposure of receptors to these inorganic chemicals is similar to background.

The EPC for cadmium is similar to background concentrations for soil, indicating that exposure is similar to background (Table 2.2-1). Cadmium is not retained as a COPEC because the EPC is similar to background. Barium and silver are retained as COPECs for further evaluation.

2.2.5 Area Use Factors

In addition to the direct comparison of the EPC with the ESLs, area use factors (AUFs) are used to account for the amount of time that a receptor is likely to spend within the contaminated areas based on the size of the receptor's home range (HR). The AUF for an individual organism is calculated by dividing the size of the site by the HR for that receptor. Because threatened and endangered species must be assessed on an individual basis (EPA 1999, 070086), the AUF is applicable for the Mexican spotted owl. The kestrel (top carnivore) is used as the surrogate receptor for the Mexican spotted owl. Because the unadjusted HI for the kestrel (top carnivore) is 0.5 (Table 2.2-2) an AUF for the Mexican spotted owl is not warranted. There is no potential adverse impact to the Mexican spotted owl because the kestrel (top carnivore) HI is less than 1.0.

2.2.6 Population Area Use Factors

EPA guidance is to manage the ecological risk to populations rather than to individuals, with the exception of threatened and endangered species (EPA 1999, 070086). One approach to addressing the potential effects on populations is to estimate the spatial extent of the area inhabited by the local population that overlaps with the contaminated area. The population area for each receptor is based on the individual receptor home range and its dispersal distance (Bowman et al. 2002, 073475). Bowman et al. (2002, 073475) estimate that the median dispersal distance for mammals is 7 times the linear dimension of the HR (i.e., the square root of the HR area). If only the dispersal distances for the mammals with HRs within the range of the screening receptors are used, the median dispersal distance becomes 3.6 times the square root of the HR ($R^2 = 0.91$) (Bowman et al. 2002, 073475). If it is assumed that the receptors can disperse over the same distance in any direction, the population area is circular and the dispersal distance is the radius of the circle. Therefore, the population area for each receptor can be derived by $\pi(3.6\sqrt{HR})^2$ or approximately 40HR.

The population area use factor (PAUF) is calculated by dividing the site area (approximately 2.6 hectares [ha] for the TA-16 Burn Ground) by the population area of the receptor (Table 2.2-2). The HQs and HIs are recalculated minus the COPEC (cadmium) eliminated based on similarity to background (section 2.2.4) and adjusted by multiplying by the PAUFs. The HQs and HIs for the earthworm and plant are not adjusted by a PAUF because these receptors do not have HRs.

The adjusted HIs are less than 1.0 for the kestrel, robin, cottontail, and red fox (Table 2.2-3). The adjusted HI for the shrew (4) is above 1.0, but less than 10, and is approximately 10 for the deer mouse (Table 2.2-3). The elevated HIs for the deer mouse and shrew are due to TCDD[2,3,7,8-] equivalent concentrations.

The TCDD[2,3,7,8-] equivalent concentrations are relatively consistent across the site, except for the two highest concentrations. Based on the spatial distribution of the data, the TCDD[2,3,7,8-] equivalent concentrations in one area east of 16-0399 are higher than the rest of the area sampled. The outlier test in ProUCL 4.00.04 indicates that the maximum TCDD[2,3,7,8-] equivalent concentration is an outlier for this data set. This may indicate, at least in part, that the dioxins and furans in this area are from other sources, e.g., the SWMUs/AOCs to the east of the burn ground area. The concentrations also bias the EPC and overestimate the potential risk to the receptor populations. The area east of 16-0399 is approximately 0.1 ha. Comparing the maximum TCDD[2,3,7,8-] equivalent concentration in this area (3.71E-05 mg/kg) to the ESLs results in HQs of 64 and 128 for the deer mouse and shrew, respectively (Table 2.2-4). Adjusting the HQs with the PAUFs in Table 2.2-5 results in HQs of 1.9 and 0.8 (Table 2.2-6), respectively, for this area.

The rest of the area around 16-0388 and 16-0399 is approximately 2.5 ha. Recalculating the TCDD[2,3,7,8-] equivalent EPC without the two highest concentrations in the area east of 16-0399 results in a 95% UCL of approximately 1.2E-06 mg/kg (Figure 2.2-1). Comparing the revised 95% UCL to the ESLs results in HQs of 2 and 4 for the deer mouse and shrew, respectively (Table 2.2-4). Adjusting the HQs by the PAUFs in Table 2.2-5 results in HQs of 1.7 and 0.6 (Table 2.2-6), respectively, for the larger area.

Given the conservative nature of the ESLs as described above, the HQs for the deer mouse and shrew are overestimated. In addition, Dourson and Stara (1983, 073474) conducted a study of uncertainty factors incorporated in calculating ESLs for ecological receptors. Based on their study, the LOAEL to NOAEL adjustment indicates that HIs up to 10 may not adversely affect ecological receptors. To maintain conservatism, they state that HIs less than 3 do not adversely affect ecological receptors. Therefore, the adjusted HIs for the shrew and deer mouse do not indicate potential risks to these receptors across the site. As indicated below previous studies by the Laboratory have found no effects to small mammal populations at similar dioxin and furan congener concentrations in the canyons.

Biota investigations have been conducted in canyon reaches in Los Alamos/Pueblo Canyon (LANL 2004, 087390), Mortandad Canyon (LANL 2006, 094161; LANL 2007, 098279), and Pajarito Canyon (LANL 2008, 104909). Field and laboratory studies included collection and analysis of soil, sediment, and water samples; cavity-nesting bird monitoring and analysis of eggs; small mammal trapping and analysis of whole organisms; earthworm bioaccumulation tests—measures of growth and survival, and analysis of whole organisms; and seedling germination tests. The studies found no effects from exposure to TCDD in any of the canyon reaches.

The TCDD equivalent concentrations reported in Kraig et al. (2002, 085536, Table 5 and Table A-6) ranged from 4.7×10^{-7} mg/kg to 3.5×10^{-6} mg/kg in samples from lower Los Alamos Canyon. These levels are similar to the TCDD concentrations in Los Alamos and Pueblo Canyons (LANL 2005, 091818); the range of concentrations is 1.71×10^{-10} mg/kg to 4.96×10^{-6} mg/kg. Dioxins and furans, therefore, appear

to be present throughout the Los Alamos and Pueblo Canyons watershed at levels exceeding the screening levels for small mammals. The field studies conducted in the Los Alamos and Pueblo Canyons watershed included four locations where small-mammal populations were evaluated, and two small-mammal study areas were in the Pueblo Canyon watershed (reaches AC-3 and P-3W). No difference in population density, sex ratio, or reproductive classes was noted between these small-mammal study areas (LANL 2004, 087390). Because adverse ecological effects to mammals were not identified by the ecological risk assessment, the assessment implicitly demonstrated that there are no adverse ecological effects from dioxins and furans. In addition, TCDD concentrations ranged from 3.14×10^{-7} mg/kg to 3.09×10^{-6} mg/kg in Pajarito Canyon (dioxins and furans were not analyzed for in Mortandad Canyon) as part of the canyon investigation and no adverse effects were reported (LANL 2008, 104909).

The 95% UCL for TCDD[2,3,7,8-] equivalent across the site is 6.65×10^{-6} mg/kg, which is similar to the concentrations detected in Los Alamos, Pueblo, and Pajarito Canyons. Because no adverse ecological effects to mammals were identified following small mammal trapping and analysis of whole organisms in these canyons at similar concentrations, no adverse ecological effects are present within the TA-16 Burn Ground.

The HIs are above 1.0 for the earthworm (2) and the plant (6). Barium is the primary COPEC for the earthworm and plant. The barium ESLs for the earthworm (330 mg/kg) and plant (110 mg/kg) are similar to or less than the soil BV (295 mg/kg) and the maximum background concentration (410 mg/kg). A comparison of the barium EPC (704 mg/kg) to the maximum background concentration (410 mg/kg) results in a ratio of 1.7. Therefore, the EPC is less than twice background and not likely to impact the earthworm and plant. Furthermore, as noted above, the Dourson and Stara (1983, 073474) study indicated that HIs up to 10 may not adversely affect ecological receptors.

The plant community was observed to be typical of the surrounding area and appears healthy; no evidence was found that there are any adverse impacts of contamination to the plant community. No marked differences in vegetation were observed between this area and areas with similar topography that did not have elevated HIs. In addition, substantially higher concentrations of barium were reported in Cañon de Valle below the 260 Outfall as part of the Resource Conservation and Recovery Act facility investigations conducted from 1998-2002 (LANL 2003, 077965). Despite the substantially elevated barium concentrations (maximum concentration of 37300 mg/kg) in the canyon sediment, the plant community was observed to be typical of the surrounding area and is luxuriant and healthy; no evidence was found that there are any adverse impacts of any contamination in Cañon de Valle to the plant community. These observations support the conclusions that barium is not impacting the earthworm and plant at the TA-16 Burn Ground and the Dourson and Stara (1983, 073474) study. Because the plant community is not affected by the COPECs, the earthworm population is also likely not affected. Therefore, no COPECs are retained at this site.

3.0 CONCLUSIONS

The human-health screening assessment found that risks were below the NMED target levels for the industrial and residential scenarios. The ecological risk screening assessment found that no COPECs are retained for this area. The HIs calculated based on literature derived ESLs are conservative and overestimate the potential risk to receptors. Field observations and published studies indicate that the slightly elevated HIs do not reflect actual adverse ecological impacts to receptors at the site. Therefore, no potential unacceptable risks to human and ecological receptors are present.

4.0 REFERENCES

The following list includes all documents cited in this appendix. Parenthetical information following each reference provides the author(s), publication date, and ER ID. This information is also included in text citations. ER IDs are assigned by the Environmental Programs Directorate's Records Processing Facility (RPF) and are used to locate the document at the RPF and, where applicable, in the master reference set.

Copies of the master reference set are maintained at the New Mexico Environment Department Hazardous Waste Bureau and the Directorate. The set was developed to ensure that the administrative authority has all material needed to review this document, and it is updated with every document submitted to the administrative authority. Documents previously submitted to the administrative authority are not included.

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Table 1.2-1
Background Comparisons for Inorganic Chemicals at the TA-16 Burn Ground

Sample ID	Depth (ft)	Media	Arsenic (mg/kg)	Barium (mg/kg)	Cadmium (mg/kg)	Chromium (mg/kg)	Lead (mg/kg)	Mercury (mg/kg)	Selenium (mg/kg)	Silver (mg/kg
Soil Background Value (mg/kg)			**8.17**	**295**	**0.4**	**19.3**	**22.3**	**0.1**	**1.52**	**1**
09RCRA696	0-1	Soil	1.94	**1180**	0.33	7.85	11	0.0154	1.01(U)	0.574
09RCRA698	0-1	Soil	2.23	**470**	**0.493**	6.25	11.5	0.0163	1.07(U)	0.634
09RCRA700	0-1	Soil	0.773(J)	242	0.326	3.05	6.03	0.00568(J)	1.05(U)	0.398(J)
09RCRA702	0-1	Soil	3.39	196	**0.487**	11.1	13	0.0196	1.11(U)	0.452(J)
09RCRA704	0-1	Soil	2.42	229	**0.555**	8.8	19.7	0.0122(J)	1.05(U)	0.537
09RCRA706	0-1	Soil	1.94	298	**0.562**	5.2	11.5	0.0103(J)	1.03(U)	0.39(J)
09RCRA708	0-1	Soil	0.944(J)	**970**	0.319	4.59	9.16	0.0061(J)	1.01(U)	0.692
09RCRA710	0-1	Soil	0.767(J)	**1780**	**0.453**	5.24	13.5	0.04	1.05(U)	**7.95**
09RCRA712	0-1	Soil	0.981(J)	**1730**	**0.425**	5.16	9.48	0.0203	1.17(U)	**1.59**
09RCRA714	0-1	Soil	1.53	105	0.325	5.72	8.74	0.0123	1.04(U)	0.373(J)
09RCRA716	0-1	Soil	2.15	**434**	**0.579**	7.83	15.3	0.0152	1.03(U)	0.446(J)
09RCRA718	0-1	Soil	1.8	**1260**	**0.68**	8.89	17.4	0.0209	1.13(U)	0.876
09RCRA720	0-1	Soil	2.5	199	0.313	13.1	10.9	0.00946(J)	1.10(U)	0.439(J)
09RCRA722	0-1	Soil	1.8	243	**0.46**	8.24	11.1	0.00976(J)	1.04(U)	0.375(J)
09RCRA724	0-1	Soil	1.96	**356**	0.362	6.77	12.5	0.0116	1.02(U)	0.622
09RCRA726	0-1	Soil	1.76	98.8	0.358	6.06	11.9	0.00683(J)	1.05(U)	0.352(J)
09RCRA728	0-1	Soil	1.34	**314**	0.389	6.01	11.1	0.00848(J)	1.02(U)	0.413(J)
09RCRA730	0-1	Soil	1.46	230	0.366	4.79	7.11	0.00794(J)	1.05(U)	0.331(J)
09RCRA732	0-1	Soil	1.84	267	**0.414**	6.85	11.8	0.0118	1.03(U)	0.297(J)
09RCRA734	0-1	Soil	1.79	**345**	0.381	7.24	13.3	0.00854(J)	1.05(U)	0.471(J)
09RCRA736	0-1	Soil	1.45	275	0.385	6.03	15.9	0.00817(J)	1.02(U)	0.449(J)
09RCRA738	0-1	Soil	1.24	141	0.219	11.6	10	0.00888(J)	1.05(U)	0.164(J)
09RCRA740	0-1	Soil	1.85	**612**	**0.439**	9.55	10.4	0.0184	1.03(U)	**1.12**
09RCRA742	0-1	Soil	1.17	**488**	**0.408**	7.41	9.64	0.0115(J)	1.06(U)	0.923
09RCRA744	0-1	Soil	2.26	**948**	**0.621**	9.74	11.4	0.0412	1.09(U)	**1.17**
09RCRA746	0-1	Soil	1.47	270	0.317	8.47	10.5	0.00913(J)	1.04(U)	0.509(J)
09RCRA748	0-1	Soil	1.54	**417**	**0.487**	8.05	11.2	0.0112(J)	1.05(U)	0.771
09RCRA750	0-1	Soil	1.21	**414**	0.32	6.3	10.7	0.008(J)	1.06(U)	0.821
09RCRA752	0-1	Soil	2.1	**486**	0.306	7.13	11.9	0.0135	1.05(U)	0.64
09RCRA754	0-1	Soil	1.48	**401**	0.373	6.52	12.9	0.0115(J)	1.07(U)	0.552
09RCRA756	0-1	Soil	1.77	**915**	**0.438**	9.12	13.4	0.0229	1.11(U)	**1.7**

Note: Source of BVs is LANL (1998, 059730). Bolded values are above the soil BV.

Table 1.2-2
Dioxin and Furan Congener Concentrations in Samples Collected from the TA-16 Burn Ground

Congener	09RCRA460 (mg/kg)	09RCRA461 (mg/kg)	09RCRA463 (mg/kg)	09RCRA464 (mg/kg)	09RCRA465 (mg/kg)	09RCRA695 (mg/kg)	09RCRA697 (mg/kg)	09RCRA699 (mg/kg)	09RCRA701 (mg/kg)	09RCRA703 (mg/kg)
2,3,7,8-TCDD	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	1.03E-07	1.33E-07	Not detected	6.18E-07
1,2,3,7,8-PeCDD	Not detected	Not detected	Not detected	Not detected	1.92E-06	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	4.19E-07	Not detected	Not detected	Not detected	5.38E-06	Not detected	Not detected	Not detected	Not detected	7.33E-07
1,2,3,6,7,8-HxCDD	7.15E-07	Not detected	Not detected	Not detected	1.06E-05	Not detected	4.67E-07	5.61E-07	Not detected	1.58E-06
1,2,3,7,8,9-HxCDD	7.26E-07	Not detected	Not detected	Not detected	1.14E-05	Not detected	5.18E-07	5.20E-07	Not detected	1.65E-06
1,2,3,4,6,7,8-HpCDD	2.08E-05	4.22E-06	4.41E-06	1.08E-05	2.92E-04	4.50E-06	8.35E-06	1.31E-05	8.37E-06	3.67E-05
OCDD	1.41E-04	2.07E-05	2.70E-05	3.22E-05	1.55E-03	3.41E-05	8.61E-05	1.02E-04	5.06E-05	2.09E-04
2,3,7,8-TCDF	1.83E-07	Not detected	Not detected	Not detected	2.01E-07	3.14E-06	6.59E-07	5.72E-07	1.14E-06	Not detected
1,2,3,7,8-PeCDF	Not detected	Not detected	Not detected	Not detected	Not detected	1.59E-06	4.83E-07	8.30E-07	Not detected	Not detected
2,3,4,7,8-PeCDF	Not detected	Not detected	Not detected	6.33E-07	7.15E-07	Not detected	5.13E-07	Not detected	Not detected	4.66E-07
1,2,3,4,7,8-HxCDF	4.95E-07	Not detected	Not detected	7.30E-07	3.21E-06	6.73E-07	5.88E-07	5.83E-07	Not detected	9.55E-07
1,2,3,6,7,8-HXCDF	5.39E-07	Not detected	Not detected	1.02E-06	3.96E-06	Not detected	4.53E-07	4.82E-07	Not detected	8.90E-07
2,3,4,6,7,8-HxCDF	7.23E-07	Not detected	Not detected	1.09E-06	5.33E-06	Not detected	6.30E-07	5.67E-07	Not detected	1.13E-06
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	1.04E-05	1.63E-06	1.20E-06	5.09E-06	8.44E-05	2.33E-06	3.50E-06	4.30E-06	3.05E-06	1.27E-05
1,2,3,4,7,8,9-HpCDF	5.35E-07	Not detected	Not detected	1.38E-06	5.95E-06	Not detected	Not detected	Not detected	Not detected	1.02E-06
OCDF	1.77E-05	2.54E-06	2.83E-06	1.07E-05	1.87E-04	3.98E-06	1.25E-05	1.27E-05	5.82E-06	2.93E-05

Table 1.2-2
Dioxin and Furan Congener Concentrations in Samples Collected from the TA-16 Burn Ground (continued)

Congener	09RCRA705 (mg/kg)	09RCRA707 (mg/kg)	09RCRA709 (mg/kg)	09RCRA711 (mg/kg)	09RCRA713 (mg/kg)	09RCRA715 (mg/kg)	09RCRA717 (mg/kg)	09RCRA719 (mg/kg)	09RCRA721 (mg/kg)	09RCRA723 (mg/kg)
2,3,7,8-TCDD	Not detected	1.69E-07	5.49E-07	1.34E-06	1.38E-07	2.52E-07	1.50E-07	Not detected	1.28E-07	2.22E-07
1,2,3,7,8-PeCDD	Not detected	Not detected	5.13E-07	6.99E-06	6.73E-07	5.39E-07	6.33E-07	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	7.41E-07	Not detected	7.71E-07	1.79E-05	1.47E-06	1.21E-06	5.53E-07	Not detected	Not detected	Not detected
1,2,3,6,7,8-HxCDD	1.40E-06	7.53E-07	1.28E-06	3.33E-05	2.80E-06	2.45E-06	1.01E-06	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDD	1.49E-06	8.03E-07	1.40E-06	4.07E-05	3.42E-06	3.06E-06	1.01E-06	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDD	3.66E-05	2.01E-05	3.23E-05	9.00E-04	5.81E-05	6.85E-05	2.02E-05	6.22E-07	4.64E-06	1.95E-05
OCDD	2.08E-04	1.29E-04	1.85E-04	4.80E-03	2.38E-04	3.69E-04	1.01E-04	2.92E-06	2.68E-05	1.06E-04
2,3,7,8-TCDF	5.45E-07	9.19E-07	5.95E-07	1.51E-06	3.84E-07	3.38E-07	1.65E-06	1.46E-07	3.25E-07	3.23E-07
1,2,3,7,8-PeCDF	Not detected	5.72E-07	4.76E-07	1.65E-06	Not detected	Not detected	1.87E-06	Not detected	Not detected	Not detected
2,3,4,7,8-PeCDF	Not detected	6.77E-07	4.97E-07	1.97E-06	Not detected	Not detected	1.82E-06	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDF	7.64E-07	7.87E-07	7.66E-07	1.21E-05	9.65E-07	1.01E-06	2.02E-06	Not detected	Not detected	Not detected
1,2,3,6,7,8-HXCDF	7.60E-07	7.41E-07	7.37E-07	1.44E-05	1.52E-06	1.28E-06	1.50E-06	Not detected	Not detected	Not detected
2,3,4,6,7,8-HxCDF	9.33E-07	9.19E-07	8.63E-07	1.78E-05	1.69E-06	1.56E-06	1.82E-06	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	2.02E-06	Not detected	Not detected	4.86E-07	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	1.13E-05	7.27E-06	8.71E-06	2.88E-04	2.34E-05	2.38E-05	1.01E-05	Not detected	1.49E-06	3.03E-06
1,2,3,4,7,8,9-HpCDF	9.20E-07	7.30E-07	7.41E-07	1.92E-05	8.20E-07	1.29E-06	7.29E-07	Not detected	Not detected	Not detected
OCDF	2.79E-05	1.95E-05	2.19E-05	6.57E-04	2.33E-05	4.71E-05	1.62E-05	Not detected	3.42E-06	6.36E-06

Table 1.2-2
Dioxin and Furan Congener Concentrations in Samples Collected from the TA-16 Burn Ground (continued)

Congener	09RCRA725 (mg/kg)	09RCRA727 (mg/kg)	09RCRA729 (mg/kg)	09RCRA731 (mg/kg)	09RCRA733 (mg/kg)	09RCRA735 (mg/kg)	09RCRA737 (mg/kg)	09RCRA739 (mg/kg)	09RCRA741 (mg/kg)	09RCRA743 (mg/kg)
2,3,7,8-TCDD	Not detected	2.76E-07	9.45E-08	4.61E-07	1.73E-07	Not detected	Not detected	Not detected	2.41E-07	1.33E-07
1,2,3,7,8-PeCDD	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	Not detected	5.22E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,6,7,8-HxCDD	Not detected	9.35E-07	5.88E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDD	Not detected	1.12E-06	7.07E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDD	6.76E-07	2.26E-05	1.36E-05	5.93E-06	2.00E-06	3.55E-06	8.76E-07	3.76E-06	3.17E-06	4.19E-06
OCDD	4.00E-06	1.36E-04	7.97E-05	3.18E-05	1.10E-05	2.29E-05	3.57E-06	1.91E-05	1.38E-05	1.65E-05
2,3,7,8-TCDF	2.22E-07	3.17E-07	4.13E-07	2.45E-07	2.00E-07	3.19E-07	2.65E-07	2.86E-07	3.83E-07	3.42E-07
1,2,3,7,8-PeCDF	Not detected	Not detected	6.28E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
2,3,4,7,8-PeCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDF	Not detected	5.51E-07	5.90E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,6,7,8-HXCDF	Not detected	5.39E-07	4.38E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
2,3,4,6,7,8-HxCDF	Not detected	6.73E-07	5.17E-06	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	Not detected	8.01E-06	5.17E-06	2.02E-06	7.45E-07	1.68E-06	Not detected	7.53E-07	6.40E-07	6.87E-07
1,2,3,4,7,8,9-HpCDF	Not detected	5.06E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
OCDF	Not detected	1.55E-05	8.65E-06	3.64E-06	1.19E-06	3.59E-06	Not detected	1.91E-06	1.51E-06	1.34E-06

Table 1.2-2
Dioxin and Furan Congener Concentrations in Samples Collected from the TA-16 Burn Ground (continued)

Congener	09RCRA745 (mg/kg)	09RCRA747 (mg/kg)	09RCRA749 (mg/kg)	09RCRA751 (mg/kg)	09RCRA753 (mg/kg)	09RCRA755 (mg/kg)
2,3,7,8-TCDD	Not detected	1.82E-07	1.76E-07	1.33E-07	Not detected	Not detected
1,2,3,7,8-PeCDD	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	Not detected	Not detected	Not detected	Not detected	Not detected	6.08E-07
1,2,3,6,7,8-HxCDD	Not detected	Not detected	Not detected	Not detected	Not detected	8.00E-07
1,2,3,7,8,9-HxCDD	Not detected	Not detected	Not detected	Not detected	Not detected	8.24E-07
1,2,3,4,6,7,8-HpCDD	1.54E-06	7.12E-06	6.39E-06	4.88E-06	4.89E-06	1.20E-05
OCDD	8.98E-06	4.47E-05	3.34E-05	2.83E-05	2.89E-05	3.76E-05
2,3,7,8-TCDF	3.25E-07	3.83E-07	4.38E-07	4.23E-07	3.77E-07	4.11E-07
1,2,3,7,8-PeCDF	Not detected	Not detected	Not detected	7.18E-07	5.05E-07	Not detected
2,3,4,7,8-PeCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDF	Not detected	Not detected	Not detected	4.88E-07	Not detected	Not detected
1,2,3,6,7,8-HXCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
2,3,4,6,7,8-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	Not detected	1.61E-06	1.38E-06	8.28E-07	1.12E-06	1.38E-06
1,2,3,4,7,8,9-HpCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
OCDF	Not detected	5.33E-06	3.29E-06	2.57E-06	3.11E-06	4.04E-06

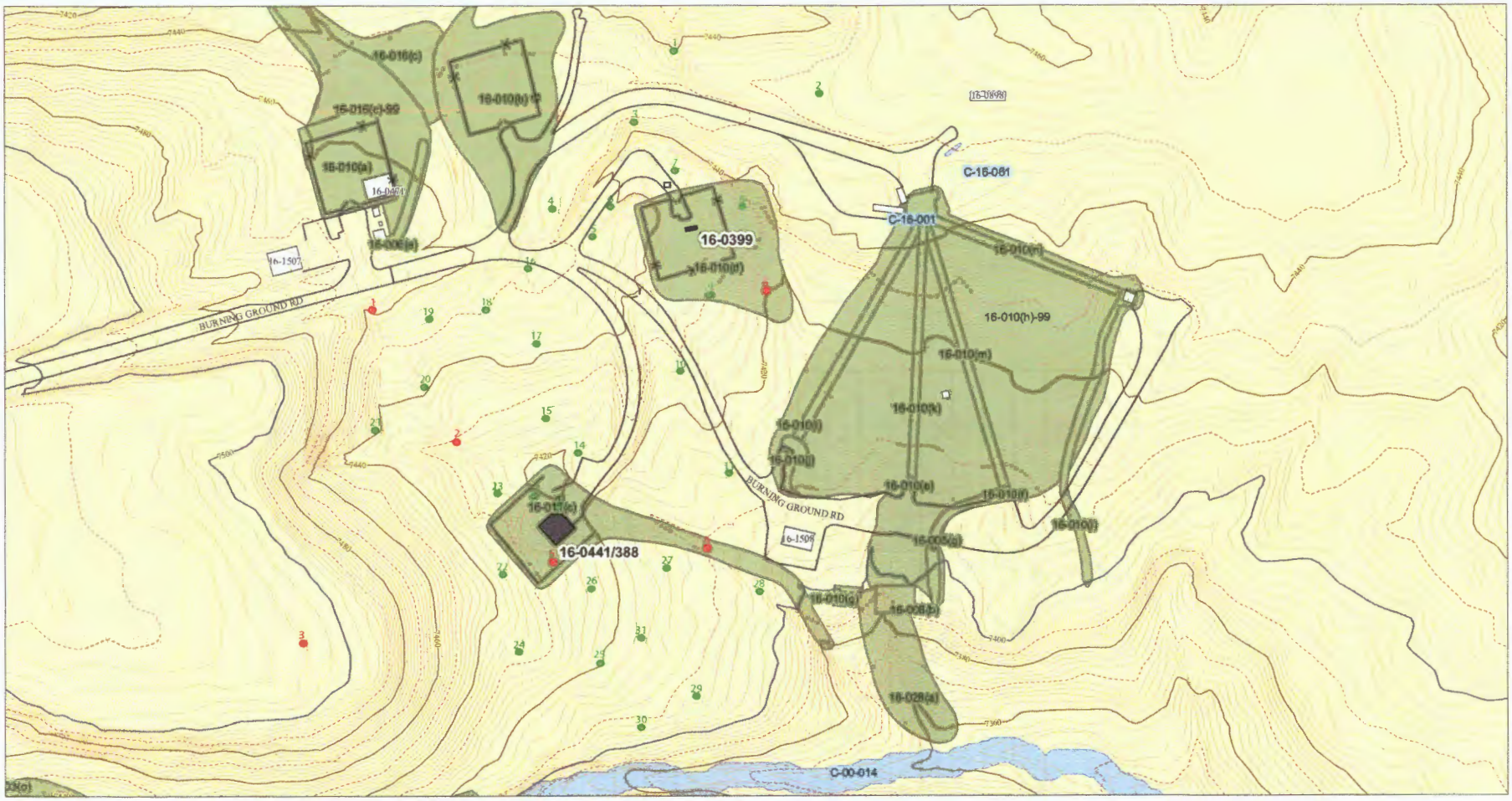

Figure 1.2-1: Technical Area 16 Burn Ground Soil Sample Locations For 2009 Soil Sample Collection Events

Map Created By: Kathryn Bennett, WES-RDA-GIS GIS Team, July 15, 2009, Map Reference #09-0030-12-TA16BurnGround

Fences
100-ft Contour
Roads, Paved
20-ft Contour
10-ft Contour
2-ft Contour

PRS and Areas of Concern
AOC
SWMU
08/26/09 Soil Sample Location
06/08/09 Soil Sample Location

RCRA waste management units
RWMU Structure
Other Structures

Los Alamos NATIONAL LABORATORY

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1:1,600

0 45 90 180 Feet

Meters
0 8.57 14 21 28

1983 North American Datum
State Plane Coordinate System
New Mexico, Central Zone (US Feet)

DATA SOURCES:

PATH	NAME	DATE ACCESSED	DATE PUBLISHED
[illegible]	[illegible]	[illegible]	[illegible]

Addendum to Figure 1.2-1

	Dioxin-Furan Analysis	Metals Analysis
Sample Location on Figure	Sample ID	Sample ID
06/08/09 Location 1	09RCRA460	Not Applicable
06/08/09 Location 2	09RCRA461	Not Applicable
06/08/09 Location 3	09RCRA462	Not Applicable
06/08/09 Location 4	09RCRA463	Not Applicable
06/08/09 Location 5	09RCRA464	Not Applicable
06/08/09 Location 6	09RCRA465	Not Applicable
08/26/09 Location 1	09RCRA695	09RCRA696
08/26/09 Location 2	09RCRA697	09RCRA698
08/26/09 Location 3	09RCRA699	09RCRA700
08/26/09 Location 4	09RCRA701	09RCRA702
08/26/09 Location 5	09RCRA703	09RCRA704
08/26/09 Location 6	09RCRA705	09RCRA706
08/26/09 Location 7	09RCRA707	09RCRA708
08/26/09 Location 9	09RCRA709	09RCRA710
08/26/09 Location 8	09RCRA711	09RCRA712
08/26/09 Location 28	09RCRA713	09RCRA714
08/26/09 Location 29	09RCRA715	09RCRA716
08/26/09 Location 25	09RCRA717	09RCRA718
08/26/09 Location 30	09RCRA719	09RCRA720
08/26/09 Location 24	09RCRA721	09RCRA722
08/26/09 Location 26	09RCRA723	09RCRA724
08/26/09 Location 31	09RCRA725	09RCRA726
08/26/09 Location 27	09RCRA727	09RCRA728
08/26/09 Location 11	09RCRA729	09RCRA730
08/26/09 Location 10	09RCRA731	09RCRA732
08/26/09 Location 14	09RCRA733	09RCRA734
08/26/09 Location 12	09RCRA735	09RCRA736
08/26/09 Location 13	09RCRA737	09RCRA738
08/26/09 Location 22	09RCRA739	09RCRA740
08/26/09 Location 23	09RCRA741	09RCRA742
08/26/09 Location 16	09RCRA743	09RCRA744
08/26/09 Location 21	09RCRA745	09RCRA746
08/26/09 Location 20	09RCRA747	09RCRA748
08/26/09 Location 19	09RCRA749	09RCRA750
08/26/09 Location 18	09RCRA751	09RCRA752
08/26/09 Location 17	09RCRA753	09RCRA754
08/26/09 Location 15	09RCRA755	09RCRA756

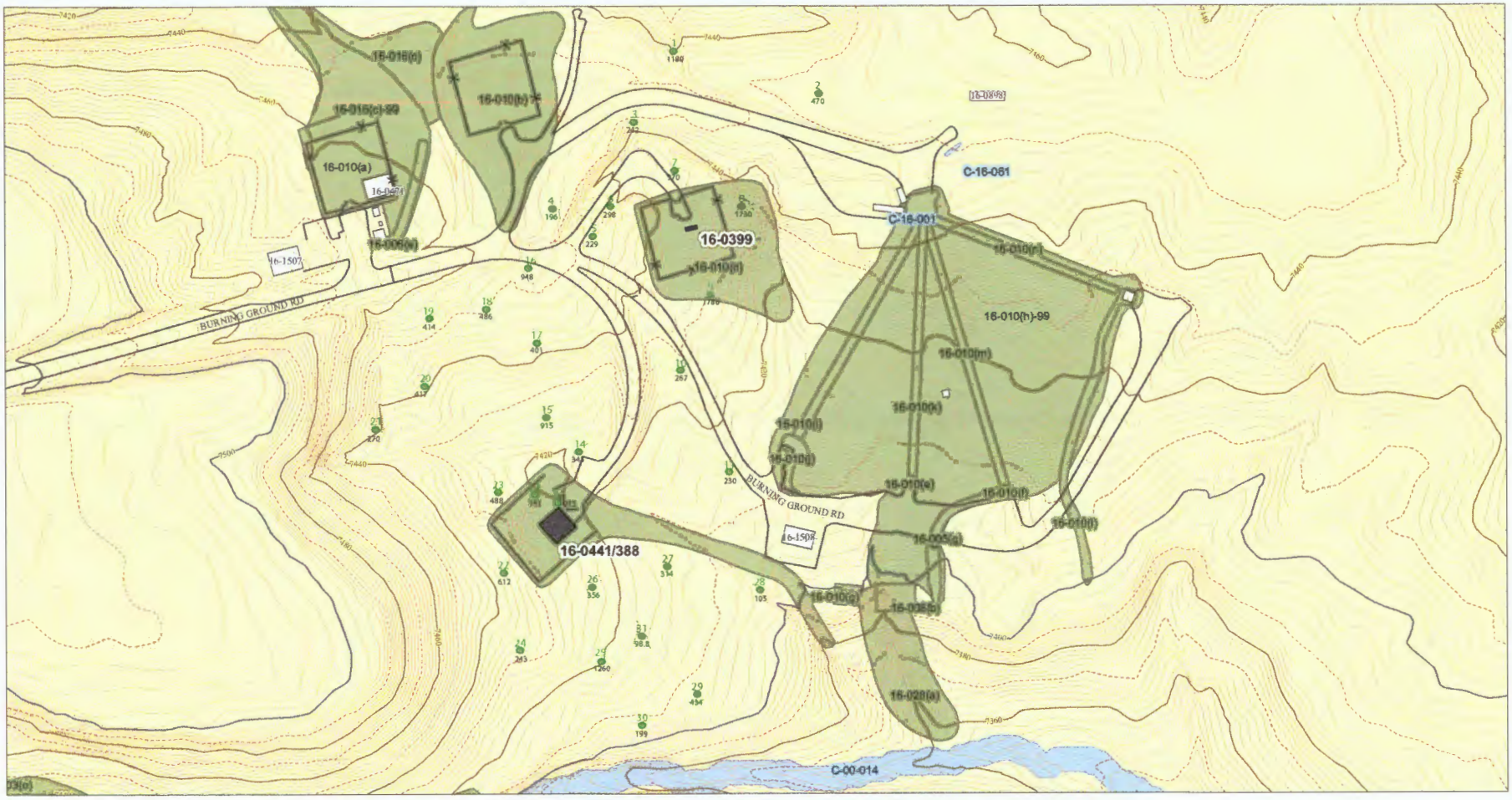

Figure 1.2-2: Technical Area 16 Burn Ground Barium Soil Concentrations

Map Created By: Kathryn Bennett, WES-EDA-GIS GIS Team, July 15, 2009, Map Reference #09-0030-12-TA16BurnGround

Fences
Roads, Paved
100-ft Contour
20-ft Contour
10-ft Contour
2-ft Contour
PRS and Areas of Concern
AOC
SWMU
08/26/09 Soil Sample Location
RCRA waste management units
RWMU Structure
Other Structures

Los Alamos NATIONAL LABORATORY EST. 1943

* Concentrations values in black for each sample location are barium concentrations in milligrams per kilogram (mg/Kg).

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1:1,600

0 45 90 180 Feet

0 8 .57 14 21 28 Meters

1983 North American Datum
State Plane Coordinate System
New Mexico, Central Zone (US Feet)

DATA SOURCES:

DATA	NAME	DATE ACCESSED	DATE PUBLISHED
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[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]

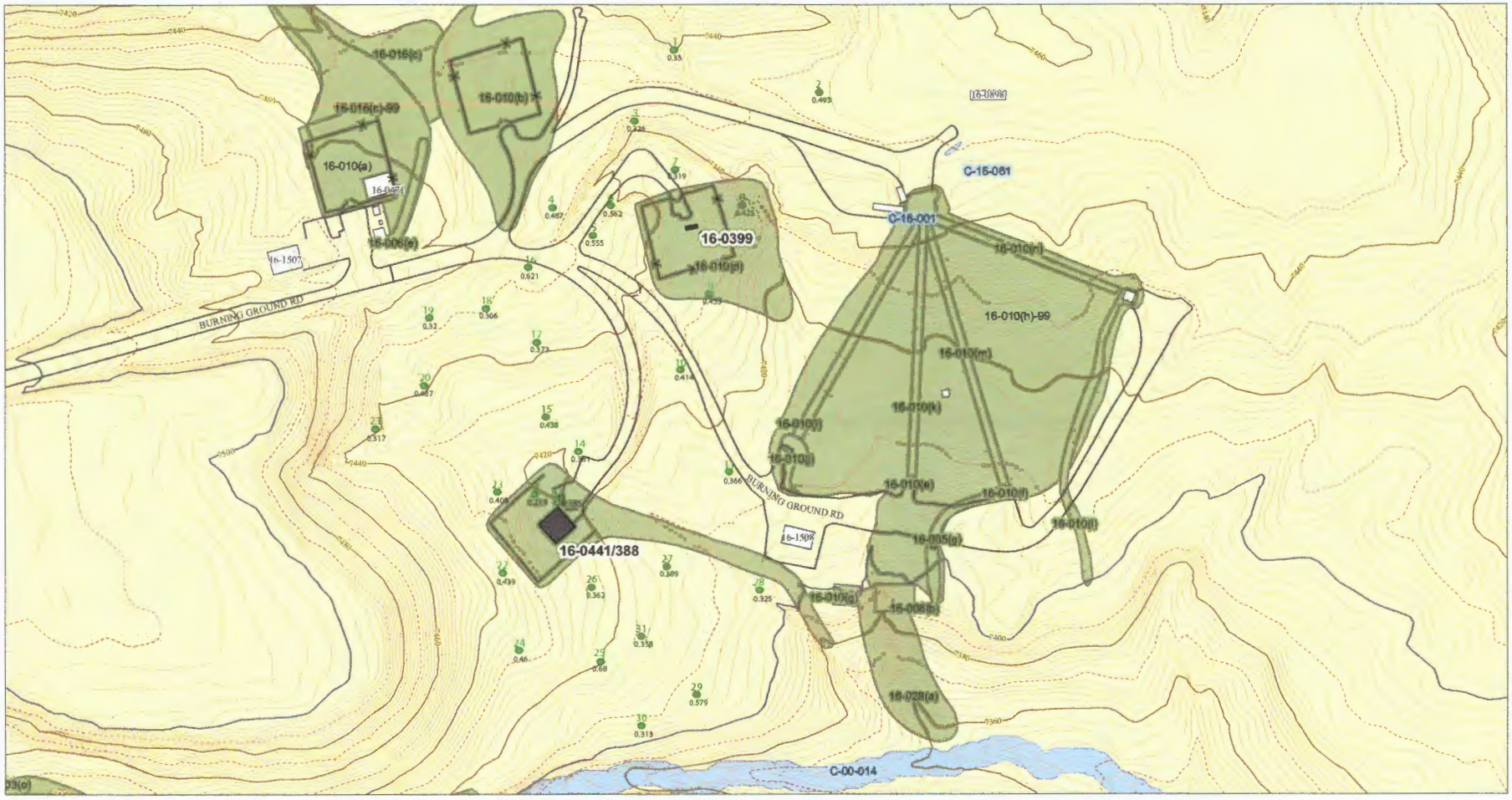

Figure 1.2-3: Technical Area 16 Burn Ground Cadmium Soil Concentrations

Map Created By: Kathryn Bennett, WES-EDA-GIS GIS Team, July 15, 2009, Map Reference #09-0030-12-TA16BurnGround

Fences
Roads, Paved
100-ft Contour
20-ft Contour
10-ft Contour
2-ft Contour

PRS and Areas of Concern
AOC
SWMU
08/26/09 Soil Sample Location

RCRA waste management units
RWMU Structure
Other Structures

Los Alamos NATIONAL LABORATORY

* Concentrations values in black for each sample location are cadmium concentrations in milligrams per kilogram (mg/Kg).

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1:1,600

0 45 90 180 Feet

Meters
0 3.57 14 21 28

1983 North American Datum
State Plane Coordinate System
New Mexico, Central Zone (US Feet)

DATA SOURCES:

PATH	NAME	DATE ACCESSED	DATE PUBLISHED
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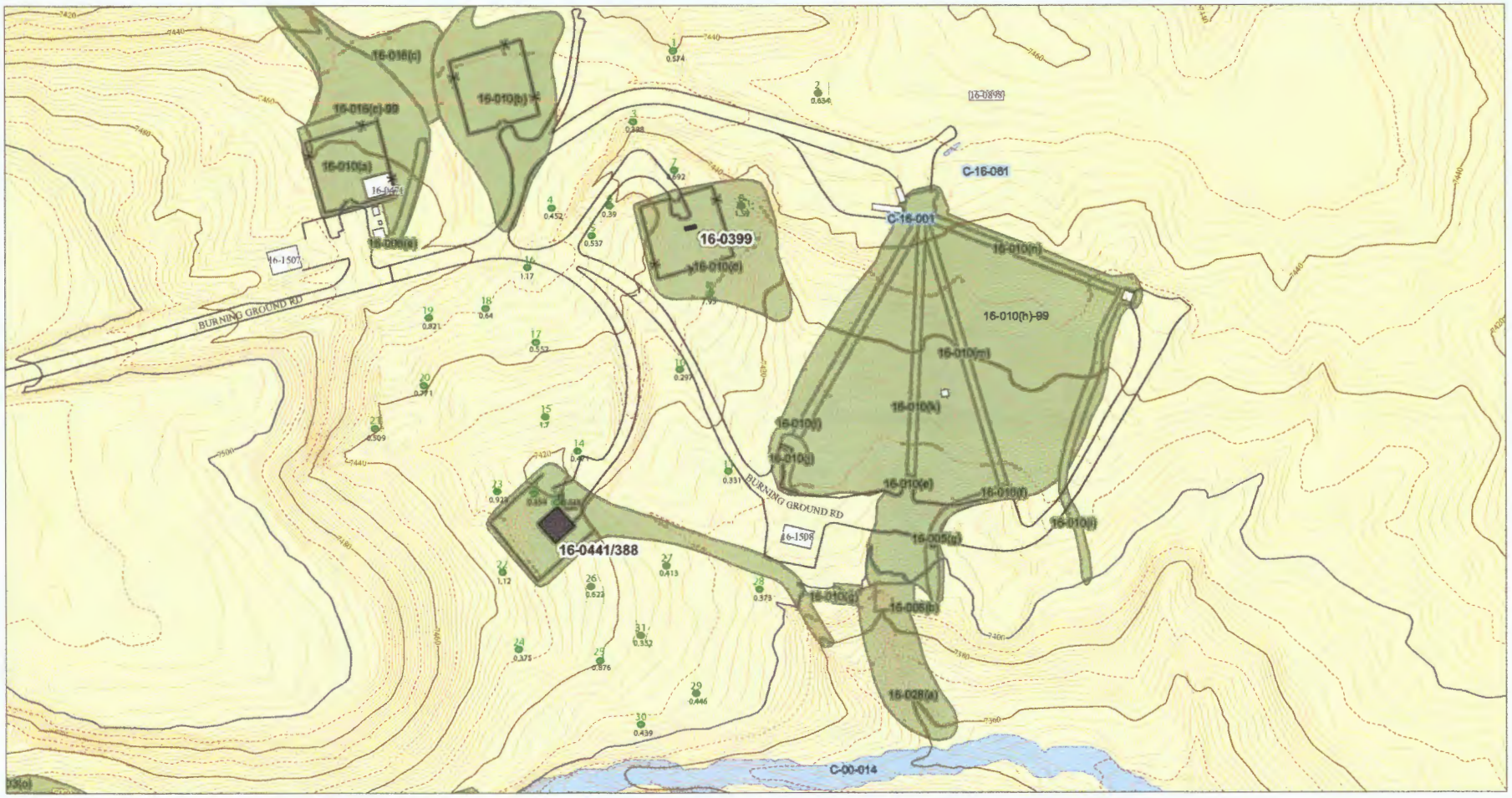

Figure 1.2-4: Technical Area 16 Burn Ground Silver Soil Concentrations

Map Created By: Kathryn Bennett, WES-EDA-GIS GIS Team, July 15, 2009, Map Reference #09-0030-12-TA16BurnGround

Fences
Roads, Paved
100-ft Contour
20-ft Contour
10-ft Contour
2-ft Contour

PRS and Areas of Concern
AOC
SWMU
08/26/09 Soil Sample Location

RCRA waste management units
RWMU Structure
Other Structures

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* Concentrations values in black for each sample location are silver concentrations in milligrams per kilogram (mg/Kg).

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1:1,600

0 45 90 180 Feet

0 7.5 14 21 28 Meters

1983 North American Datum
State Plane Coordinate System
New Mexico, Central Zone (US Feet)

DATA SOURCES:

PATH	NAME	DATE ACCESSED	DATE PUBLISHED
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[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]
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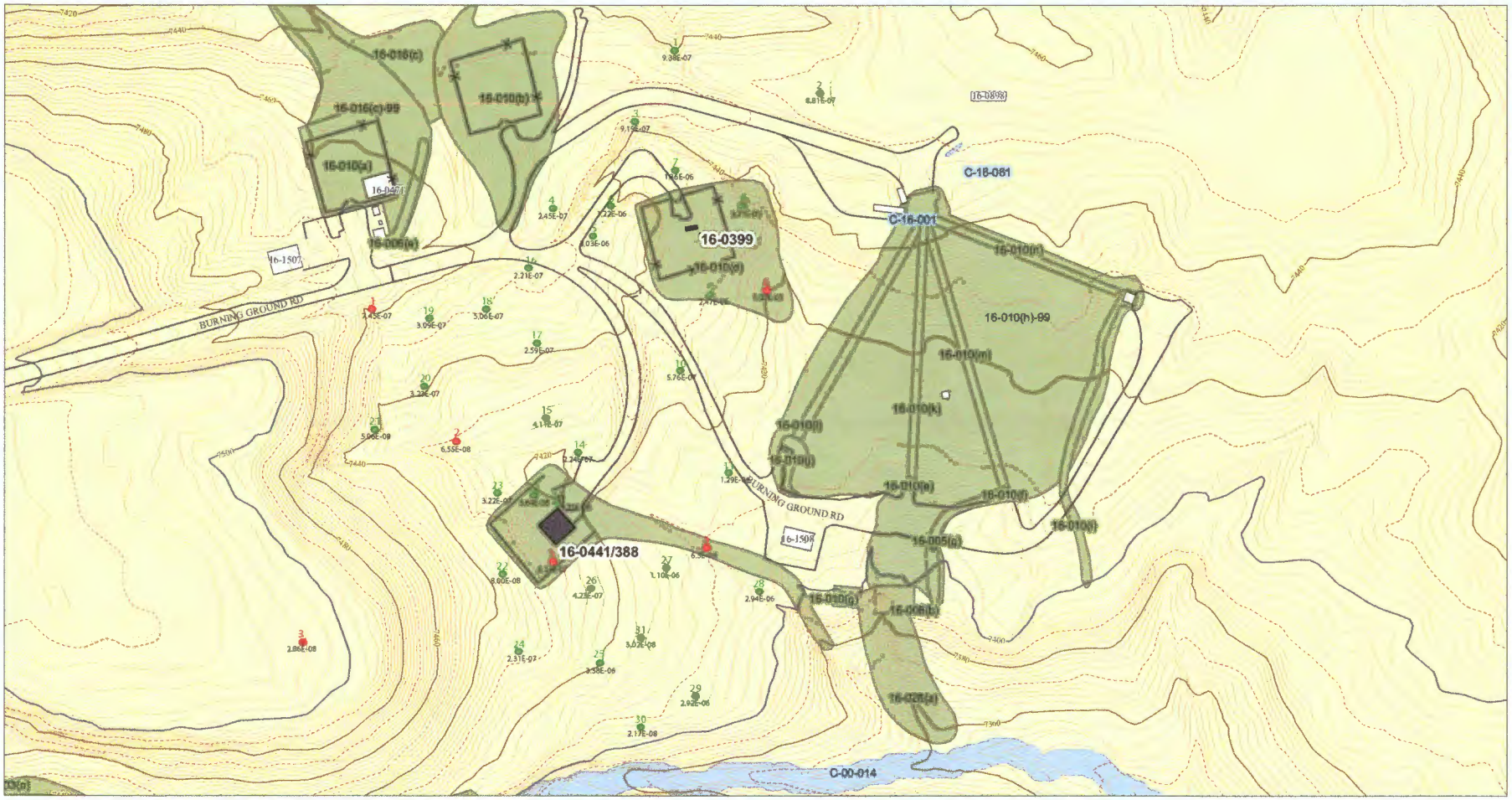

Figure 1.2-5: Technical Area 16 Burn Ground Dioxin/Furan Soil Concentrations

Map Created By: Kathryn Bennett, WBI-EDA-GIS GIS Team, July 15, 2009, Map Reference #09-0010-12-TA16BurnGround

100-ft Contour
20-ft Contour
10-ft Contour
2-ft Contour

Fences
Roads, Paved

PRS and Areas of Concern
AOC
SWMU

08/26/09 Soil Sample Location
06/08/09 Soil Sample Location

RCRA waste management units
RWMU Structure
Other Structures

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1:1,600

0 45 90 180 Feet

Meters
0 8.57 14 21 28

1983 North American Datum
State Plane Coordinate System
New Mexico, Central Zone (US Feet)

DATA SOURCES:

PATH	NAME	DATE ACCESSED	DATE PUBLISHED
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C:\Documents and Settings\1[illegible]\Application Data\ESRI\ArcCatalog\Connection to GIS-SQL P.sde\eng\ER.LANL_hydrographic\eng.ER.LANL_hydrographic	Water Features	2009-06-15	1997
C:\Documents and Settings\1[illegible]\Application Data\ESRI\ArcCatalog\Connection to GIS-SQL P.sde\eng\ER.LANL_structure\eng.ER.LANL_structure	Structure	2009-06-15	1997
C:\Documents and Settings\1[illegible]\Application Data\ESRI\ArcCatalog\Connection to GIS-SQL P.sde\eng\ER.LANL_vegcover\eng.ER.LANL_vegcover	Vegetation Cover	2009-06-15	1997
C:\Documents and Settings\1[illegible]\Application Data\ESRI\ArcCatalog\Connection to GIS-SQL P.sde\eng\ER.Point_Source\eng.ER.Point_Source	Point Source	2009-06-15	1997
C:\Documents and Settings\1[illegible]\Application Data\ESRI\ArcCatalog\Connection to GIS-SQL P.sde\eng\ER.PWD_roadbase\eng.ER.PWD_roadbase	Roads, Paved	2009-06-15	1997
X:\Projects\R\Projects\R-0100\Data\Lab_arc_shmt_sfls	Shmt Fields	2009-06-15	10-Oct-08
X:\Projects\R\Projects\R-0100\TSC_data\TSC_solid_settling_sfls	TSC solid settling sfls	2009-06-15	06-Apr-09
C:\Documents and Settings\1[illegible]\Application Data\ESRI\ArcCatalog\Connection to GIS-SQL P.sde\eng\ER.PWD_structure\eng.ER.PWD_structure	Structure	2009-06-15	01-Jul-09
X:\secure\cur\Residential\R\Data_all_reg.mdb\dbo\vw\vw_form_gas_analy_detail\vw_ER_gas	PRS and Areas of Concern	2009-06-15	2009-06-15

Concentrations values in black for each sample location are TCDD toxicity equivalent concentrations in milligrams per kilogram (mg/Kg).

Table 1.3-1
Toxic Equivalency Factors (TEFs) Used for Calculating TCDD Equivalent Concentrations

Dioxin and Furan Congeners	WHO 1995* TEF
Tetrachlorodibenzodioxin[2,3,7,8-]	1
Pentachlorodibenzodioxin[1,2,3,7,8-]	1
Hexachlorodibenzodioxin[1,2,3,4,7,8-]	0.1
Hexachlorodibenzodioxin[1,2,3,6,7,8-]	0.1
Hexachlorodibenzodioxin[1,2,3,7,8,9-]	0.1
Heptachlorodibenzodioxin[1,2,3,4,6,7,8-]	0.01
Octachlorodibenzodioxin	0.0003
Tetrachlorodibenzofuran[2,3,7,8-]	0.1
Pentachlorodibenzofuran[1,2,3,7,8-]	0.03
Pentachlorodibenzofuran[2,3,4,7,8-]	0.3
Hexachlorodibenzofuran[1,2,3,4,7,8-]	0.1
Hexachlorodibenzofuran[1,2,3,6,7,8-]	0.1
Hexachlorodibenzofuran[1,2,3,7,8,9-]	0.1
Hexachlorodibenzofuran[2,3,4,6,7,8-]	0.1
Heptachlorodibenzofuran[1,2,3,4,6,7,8-]	0.01
Heptachlorodibenzofuran[1,2,3,4,7,8,9-]	0.01
Octachlorodibenzofuran	0.0003

*http://www.epa.gov/ncea/pdfs/dioxin/part2/drich9.pdf.

Table 1.3-2
Dioxin and Furan Congener Concentrations Converted Using the Toxicity Equivalency Factors

Congener	09RCRA460 (mg/kg)	09RCRA461 (mg/kg)	09RCRA463 (mg/kg)	09RCRA464 (mg/kg)	09RCRA465 (mg/kg)	09RCRA695 (mg/kg)	09RCRA697 (mg/kg)	09RCRA699 (mg/kg)	09RCRA701 (mg/kg)	09RCRA703 (mg/kg)
2,3,7,8-TCDD	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	1.03E-07	1.33E-07	Not detected	6.18E-07
1,2,3,7,8-PeCDD	Not detected	Not detected	Not detected	Not detected	1.92E-06	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	4.19E-08	Not detected	Not detected	Not detected	5.38E-07	Not detected	Not detected	Not detected	Not detected	7.33E-08
1,2,3,6,7,8-HxCDD	7.15E-08	Not detected	Not detected	Not detected	1.06E-06	Not detected	4.67E-08	5.61E-08	Not detected	1.58E-07
1,2,3,7,8,9-HxCDD	7.26E-08	Not detected	Not detected	Not detected	1.14E-06	Not detected	5.18E-08	5.20E-08	Not detected	1.65E-07
1,2,3,4,6,7,8,-HpCDD	2.08E-07	4.22E-08	4.41E-08	1.08E-07	2.92E-06	4.50E-08	8.35E-08	1.31E-07	8.37E-08	3.67E-07
OCDD	4.23E-08	6.21E-09	8.10E-09	9.66E-09	4.65E-07	1.02E-08	2.58E-08	3.06E-08	1.52E-08	6.27E-08
2,3,7,8-TCDF	1.83E-08	Not detected	Not detected	Not detected	2.01E-08	3.14E-07	6.59E-08	5.72E-08	1.14E-07	Not detected
1,2,3,7,8-PeCDF	Not detected	Not detected	Not detected	Not detected	Not detected	4.77E-07	1.45E-07	2.49E-07	Not detected	Not detected
2,3,4,7,8-PeCDF	Not detected	Not detected	Not detected	1.90E-07	2.15E-07	Not detected	1.54E-07	Not detected	Not detected	1.40E-07
1,2,3,4,7,8-HxCDF	4.95E-08	Not detected	Not detected	7.30E-08	3.21E-07	6.73E-08	5.88E-08	5.83E-08	Not detected	9.55E-08
1,2,3,6,7,8-HXCDF	5.39E-08	Not detected	Not detected	1.02E-07	3.96E-07	Not detected	4.53E-08	4.82E-08	Not detected	8.90E-08
2,3,4,6,7,8-HxCDF	7.23E-08	Not detected	Not detected	1.09E-07	5.33E-07	Not detected	6.30E-08	5.67E-08	Not detected	1.13E-07
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	1.04E-07	1.63E-08	1.20E-08	5.09E-08	8.44E-07	2.33E-08	3.50E-08	4.30E-08	3.05E-08	1.27E-07
1,2,3,4,7,8,9-HpCDF	5.35E-09	Not detected	Not detected	1.38E-08	5.95E-08	Not detected	Not detected	Not detected	Not detected	1.02E-08
OCDF	5.31E-09	7.62E-10	8.49E-10	3.21E-09	5.61E-08	1.19E-09	3.75E-09	3.81E-09	1.75E-09	8.79E-09
TCDD[2,3,7,8-] equivalent concentration	**7.45E-07**	**6.55E-08**	**6.50E-08**	**6.59E-07**	**1.05E-05**	**9.38E-07**	**8.81E-07**	**9.19E-07**	**2.45E-07**	**2.03E-06**

Table 1.3-2
Dioxin and Furan Congener Concentrations Converted Using the Toxicity Equivalency Factors (continued)

Congener	09RCRA705 (mg/kg)	09RCRA707 (mg/kg)	09RCRA709 (mg/kg)	09RCRA711 (mg/kg)	09RCRA713 (mg/kg)	09RCRA715 (mg/kg)	09RCRA717 (mg/kg)	09RCRA719 (mg/kg)	09RCRA721 (mg/kg)	09RCRA723 (mg/kg)
2,3,7,8-TCDD	Not detected	1.69E-07	5.49E-07	1.34E-06	1.38E-07	2.52E-07	1.50E-07	Not detected	1.28E-07	2.22E-07
1,2,3,7,8-PeCDD	Not detected	Not detected	5.13E-07	6.99E-06	6.73E-07	5.39E-07	6.33E-07	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	7.41E-08	Not detected	7.71E-08	1.79E-06	1.47E-07	1.21E-07	5.53E-08	Not detected	Not detected	Not detected
1,2,3,6,7,8-HxCDD	1.40E-07	7.53E-08	1.28E-07	3.33E-06	2.80E-07	2.45E-07	1.01E-07	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDD	1.49E-07	8.03E-08	1.40E-07	4.07E-06	3.42E-07	3.06E-07	1.01E-07	Not detected	Not detected	Not detected
1,2,3,4,6,7,8,-HpCDD	3.66E-07	2.01E-07	3.23E-07	9.00E-06	5.81E-07	6.85E-07	2.02E-07	6.22E-09	4.64E-08	1.05E-07
OCDD	6.24E-08	3.87E-08	5.55E-08	1.44E-06	7.14E-08	1.11E-07	3.03E-08	8.76E-10	8.04E-09	3.18E-08
2,3,7,8-TCDF	5.45E-08	9.19E-08	5.95E-08	1.51E-07	3.84E-08	3.38E-08	1.65E-07	1.46E-08	3.25E-08	3.23E-08
1,2,3,7,8-PeCDF	Not detected	1.72E-07	1.43E-07	4.95E-07	Not detected	Not detected	5.61E-07	Not detected	Not detected	Not detected
2,3,4,7,8-PeCDF	Not detected	2.03E-07	1.49E-07	5.91E-07	Not detected	Not detected	5.46E-07	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDF	7.64E-08	7.87E-08	7.66E-08	1.21E-06	9.65E-08	1.01E-07	2.02E-07	Not detected	Not detected	Not detected
1,2,3,6,7,8-HXCDF	7.60E-08	7.41E-08	7.37E-08	1.44E-06	1.52E-07	1.28E-07	1.50E-07	Not detected	Not detected	Not detected
2,3,4,6,7,8-HxCDF	9.33E-08	9.19E-08	8.63E-08	1.78E-06	1.69E-07	1.56E-07	1.82E-07	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	2.02E-07	Not detected	Not detected	4.86E-08	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	1.13E-07	7.27E-08	8.71E-08	2.88E-06	2.34E-07	2.38E-07	1.01E-07	Not detected	1.49E-08	3.03E-08
1,2,3,4,7,8,9-HpCDF	9.20E-09	7.30E-09	7.41E-09	1.92E-05	8.20E-09	1.29E-08	7.29E-09	Not detected	Not detected	Not detected
OCDF	8.37E-09	5.85E-09	6.57E-09	1.97E-07	6.99E-09	1.41E-08	4.86E-09	Not detected	1.03E-09	1.91E-09
TCDD[2,3,7,8-] equivalent concentration	**1.22E-06**	**1.36E-06**	**2.47E-06**	**3.71E-05**	**2.94E-06**	**2.92E-06**	**3.38E-06**	**2.17E-08**	**2.31E-07**	**4.23E-07**

Table 1.3-2
Dioxin and Furan Congener Concentrations Converted Using the Toxicity Equivalency Factors (continued)

Congener	09RCRA725 (mg/kg)	09RCRA727 (mg/kg)	09RCRA729 (mg/kg)	09RCRA731 (mg/kg)	09RCRA733 (mg/kg)	09RCRA735 (mg/kg)	09RCRA737 (mg/kg)	09RCRA739 (mg/kg)	09RCRA741 (mg/kg)	09RCRA743 (mg/kg)
2,3,7,8-TCDD	Not detected	2.76E-07	9.45E-08	4.61E-07	1.73E-07	Not detected	Not detected	Not detected	2.41E-07	1.33E-07
1,2,3,7,8-PeCDD	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	Not detected	5.22E-08	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,6,7,8-HxCDD	Not detected	9.35E-08	5.88E-08	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDD	Not detected	1.12E-07	7.07E-08	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8,-HpCDD	6.76E-09	2.26E-07	1.36E-07	5.93E-08	2.00E-08	3.55E-08	8.76E-09	3.76E-08	3.17E-08	4.19E-08
OCDD	1.20E-09	4.08E-08	2.39E-08	9.54E-09	3.30E-09	6.87E-09	1.07E-09	5.73E-09	4.14E-09	4.95E-09
2,3,7,8-TCDF	2.22E-08	3.17E-08	4.13E-08	2.45E-08	2.00E-08	3.19E-08	2.65E-08	2.86E-08	3.83E-08	3.42E-08
1,2,3,7,8-PeCDF	Not detected	Not detected	1.88E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
2,3,4,7,8-PeCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDF	Not detected	5.51E-08	5.90E-08	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,6,7,8-HXCDF	Not detected	5.39E-08	4.38E-08	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
2,3,4,6,7,8-HxCDF	Not detected	6.73E-08	5.17E-07	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	Not detected	8.01E-08	5.17E-08	2.02E-08	7.45E-09	1.68E-08	Not detected	7.53E-09	6.40E-09	6.87E-09
1,2,3,4,7,8,9-HpCDF	Not detected	5.06E-09	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
OCDF	Not detected	4.65E-09	2.60E-09	1.09E-09	3.57E-10	1.08E-09	Not detected	5.83E-10	4.53E-10	4.02E-10
TCDD[2,3,7,8-] equivalent concentration	**3.02E-08**	**1.10E-06**	**1.29E-06**	**5.76E-07**	**2.24E-07**	**9.21E-08**	**3.64E-08**	**8.00E-08**	**3.22E-07**	**2.21E-07**

Table 1.3-2
Dioxin and Furan Congener Concentrations Converted Using the Toxicity Equivalency Factors (continued)

Congener	09RCRA745 (mg/kg)	09RCRA747 (mg/kg)	09RCRA749 (mg/kg)	09RCRA751 (mg/kg)	09RCRA753 (mg/kg)	09RCRA755 (mg/kg)
2,3,7,8-TCDD	Not detected	1.82E-07	1.76E-07	1.33E-07	Not detected	Not detected
1,2,3,7,8-PeCDD	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDD	Not detected	Not detected	Not detected	Not detected	Not detected	6.08E-08
1,2,3,6,7,8-HxCDD	Not detected	Not detected	Not detected	Not detected	Not detected	8.00E-08
1,2,3,7,8,9-HxCDD	Not detected	Not detected	Not detected	Not detected	Not detected	8.24E-08
1,2,3,4,6,7,8-HpCDD	1.54E-08	7.12E-08	6.39E-08	4.88E-08	4.89E-08	1.20E-07
OCDD	2.69E-09	1.34E-08	1.00E-08	8.49E-09	8.67E-09	1.13E-08
2,3,7,8-TCDF	3.25E-08	3.83E-08	4.38E-08	4.23E-08	3.77E-08	4.11E-08
1,2,3,7,8-PeCDF	Not detected	Not detected	Not detected	2.15E-07	1.52E-07	Not detected
2,3,4,7,8-PeCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,7,8-HxCDF	Not detected	Not detected	Not detected	4.88E-08	Not detected	Not detected
1,2,3,6,7,8-HXCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
2,3,4,6,7,8-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,7,8,9-HxCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
1,2,3,4,6,7,8-HpCDF	Not detected	1.61E-08	1.38E-08	8.28E-09	1.12E-08	1.38E-08
1,2,3,4,7,8,9-HpCDF	Not detected	Not detected	Not detected	Not detected	Not detected	Not detected
OCDF	Not detected	1.60E-09	9.87E-10	7.71E-10	9.33E-10	1.21E-09
TCDD[2,3,7,8-] equivalent concentration	**5.06E-08**	**3.23E-07**	**3.09E-07**	**5.06E-07**	**2.59E-07**	**4.11E-07**

Table 1.3-3
Exposure Point Concentrations for the Industrial and Residential Scenarios and Ecological Receptors

COPC	Number of Analyses	Minimum Concentration (mg/kg)	Maximum Concentration (mg/kg)	Mean Concentration (mg/kg)	Distribution	EPC (mg/kg)	EPC Method
Barium	31	98.8	1780	526.3	Lognormal	704.4	95% H-UCL
Cadmium	31	0.219	0.68	0.416	Normal	0.448	95% Student's-t UCL
Silver	31	0.164	7.95	0.872	Nonparametric	1.94	95% Chebyshev (Mean, Sd) UCL
TCDD[2,3,7,8-] equivalent	36	0.0000000217	0.0000371	0.00000212	Nonparametric	0.00000665	95% Chebyshev (Mean, Sd) UCL

Table 1.3-4
Industrial Screening Evaluation of Noncarcinogenic COPCs

COPC	EPC (mg/kg)[a]	Industrial SSL (mg/kg)[b]	Hazard Quotient
Barium	704.4	224000	0.003
Cadmium	0.448	1120	0.0004
Silver	1.94	5680	0.0003
		HI	**0.004**

[a] The EPC is the 95% UCL.
[b] SSLs are from NMED (2009, 106420).

Table 1.3-5
Industrial Screening Evaluation of Carcinogenic COPCs

COPC	EPC (mg/kg)[a]	Industrial SSL (mg/kg)[b]	Cancer Risk
TCDD[2,3,7,8-] equivalent	0.00000665	0.000204	3×10^{-7}
		Total Excess Cancer Risk	**3×10^{-7}**

[a] The EPC is the 95% UCL.
[b] SSL is from NMED (2009, 106420).

Table 1.3-6
Residential Screening Evaluation of Noncarcinogenic COPCs

COPC	EPC (mg/kg)[a]	Residential SSL (mg/kg)[b]	Hazard Quotient
Barium	704.4	15600	0.05
Cadmium	0.448	77.9	0.006
Silver	1.94	391	0.005
		HI	**0.06**

[a] The EPC is the 95% UCL.
[b] SSLs are from NMED (2009, 106420).

Table 1.3-7
Residential Screening Evaluation of Carcinogenic COPCs

COPC	EPC (mg/kg)[a]	Residential SSL (mg/kg)[b]	Cancer Risk
TCDD[2,3,7,8-] equivalent	0.00000665	0.0000414	2×10^{-6}
		Total Excess Cancer Risk	**2×10^{-6}**

[a] The EPC is the 95% UCL.
[b] SSL is from NMED (2009, 106420).

Dioxin/Furan

General Statistics

Number of Valid Observations	36	Number of Distinct Observations	36

Raw Statistics		**Log-transformed Statistics**	
Minimum	2.17E-08	Minimum of Log Data	-17.65
Maximum	0.0000371	Maximum of Log Data	-10.2
Mean	2.082E-06	Mean of log Data	-14.56
Median	4.645E-07	SD of log Data	1.645
SD	6.281E-06		
Coefficient of Variation	N/A		
Skewness	5.299		

Relevant UCL Statistics

Normal Distribution Test		**Lognormal Distribution Test**	
Shapiro Wilk Test Statistic	0.329	Shapiro Wilk Test Statistic	0.978
Shapiro Wilk Critical Value	0.935	Shapiro Wilk Critical Value	0.935
Data not Normal at 5% Significance Level		Data appear Lognormal at 5% Significance Level	

Assuming Normal Distribution		**Assuming Lognormal Distribution**	
95% Student's-t UCL	3.85E-06	95% H-UCL	4.476E-06
95% UCLs (Adjusted for Skewness)		95% Chebyshev (MVUE) UCL	4.366E-06
95% Adjusted-CLT UCL	4.791E-06	97.5% Chebyshev (MVUE) UCL	5.523E-06
95% Modified-t UCL	4.005E-06	99% Chebyshev (MVUE) UCL	7.795E-06

Gamma Distribution Test		**Data Distribution**
k star (bias corrected)	0.421	Data appear Lognormal at 5% Significance Level
Theta Star	4.951E-06	
MLE of Mean	2.082E-06	
MLE of Standard Deviation	3.21E-06	
nu star	30.28	
Approximate Chi Square Value (.05)	18.71	
Adjusted Level of Significance	0.0428	
Adjusted Chi Square Value	18.3	
Anderson-Darling Test Statistic	2.058	
Anderson-Darling 5% Critical Value	0.826	
Kolmogorov-Smirnov Test Statistic	0.206	
Kolmogorov-Smirnov 5% Critical Value	0.156	
Data not Gamma Distributed at 5% Significance Level		

Nonparametric Statistics	
95% CLT UCL	3.804E-06
95% Jackknife UCL	3.85E-06
95% Standard Bootstrap UCL	3.756E-06
95% Bootstrap-t UCL	1.202E-05
95% Hall's Bootstrap UCL	1.052E-05
95% Percentile Bootstrap UCL	4.079E-06
95% BCA Bootstrap UCL	5.553E-06
95% Chebyshev(Mean, Sd) UCL	6.645E-06
97.5% Chebyshev(Mean, Sd) UCL	8.619E-06
99% Chebyshev(Mean, Sd) UCL	1.25E-05

Assuming Gamma Distribution	
95% Approximate Gamma UCL	3.369E-06
95% Adjusted Gamma UCL	3.445E-06

Potential UCL to Use	
Use 95% Chebyshev (Mean, Sd) UCL	6.645E-06

Figure 1.3-1. ProUCL printout of dioxin/furan UCL calculations for the TA-16 Burn Ground.

Barium

General Statistics

Number of Valid Observations	31

Raw Statistics

Minimum	98.8
Maximum	1780
Mean	526.3
Median	356
SD	446.9
Coefficient of Variation	0.849
Skewness	1.654

Number of Distinct Observations	31

Log-transformed Statistics

Minimum of Log Data	4.593
Maximum of Log Data	7.484
Mean of log Data	5.979
SD of log Data	0.752

Relevant UCL Statistics

Normal Distribution Test

Shapiro Wilk Test Statistic	0.777
Shapiro Wilk Critical Value	0.929

Data not Normal at 5% Significance Level

Lognormal Distribution Test

Shapiro Wilk Test Statistic	0.958
Shapiro Wilk Critical Value	0.929

Data appear Lognormal at 5% Significance Level

Assuming Normal Distribution

95% Student's-t UCL	662.5

95% UCLs (Adjusted for Skewness)

95% Adjusted-CLT UCL	683.8
95% Modified-t UCL	666.5

Assuming Lognormal Distribution

95% H-UCL	704.4
95% Chebyshev (MVUE) UCL	851.8
97.5% Chebyshev (MVUE) UCL	996.1
99% Chebyshev (MVUE) UCL	1280

Gamma Distribution Test

k star (bias corrected)	1.734
Theta Star	303.5
MLE of Mean	526.3
MLE of Standard Deviation	399.6
nu star	107.5
Approximate Chi Square Value (.05)	84.58
Adjusted Level of Significance	0.0413
Adjusted Chi Square Value	83.45

Anderson-Darling Test Statistic	1.001
Anderson-Darling 5% Critical Value	0.76
Kolmogorov-Smirnov Test Statistic	0.185
Kolmogorov-Smirnov 5% Critical Value	0.16

Data not Gamma Distributed at 5% Significance Level

Assuming Gamma Distribution

95% Approximate Gamma UCL	668.9
95% Adjusted Gamma UCL	678

Data Distribution

Data appear Lognormal at 5% Significance Level

Nonparametric Statistics

95% CLT UCL	658.3
95% Jackknife UCL	662.5
95% Standard Bootstrap UCL	657.1
95% Bootstrap-t UCL	694.7
95% Hall's Bootstrap UCL	687.1
95% Percentile Bootstrap UCL	661.1
95% BCA Bootstrap UCL	689.7
95% Chebyshev(Mean, Sd) UCL	876.1
97.5% Chebyshev(Mean, Sd) UCL	1028
99% Chebyshev(Mean, Sd) UCL	1325

Potential UCL to Use

Use 95% H-UCL	704.4

Figure 1.3-2. ProUCL printout of barium UCL calculations for the TA-16 Burn Ground.

Cadmium

General Statistics

Number of Valid Observations	31
Number of Distinct Observations	30

Raw Statistics

Minimum	0.219
Maximum	0.68
Mean	0.416
Median	0.389
SD	0.104
Coefficient of Variation	0.251
Skewness	0.723

Log-transformed Statistics

Minimum of Log Data	-1.519
Maximum of Log Data	-0.386
Mean of log Data	-0.907
SD of log Data	0.246

Relevant UCL Statistics

Normal Distribution Test

Shapiro Wilk Test Statistic	0.948
Shapiro Wilk Critical Value	0.929

Data appear Normal at 5% Significance Level

Lognormal Distribution Test

Shapiro Wilk Test Statistic	0.975
Shapiro Wilk Critical Value	0.929

Data appear Lognormal at 5% Significance Level

Assuming Normal Distribution

95% Student's-t UCL	0.448

95% UCLs (Adjusted for Skewness)

95% Adjusted-CLT UCL	0.449
95% Modified-t UCL	0.448

Assuming Lognormal Distribution

95% H-UCL	0.451
95% Chebyshev (MVUE) UCL	0.497
97.5% Chebyshev (MVUE) UCL	0.532
99% Chebyshev (MVUE) UCL	0.601

Gamma Distribution Test

k star (bias corrected)	15.48
Theta Star	0.0269
MLE of Mean	0.416
MLE of Standard Deviation	0.106
nu star	959.8
Approximate Chi Square Value (.05)	888.9
Adjusted Level of Significance	0.0413
Adjusted Chi Square Value	885.1
Anderson-Darling Test Statistic	0.387
Anderson-Darling 5% Critical Value	0.745
Kolmogorov-Smirnov Test Statistic	0.0982
Kolmogorov-Smirnov 5% Critical Value	0.158

Data appear Gamma Distributed at 5% Significance Level

Data Distribution

Data appear Normal at 5% Significance Level

Nonparametric Statistics

95% CLT UCL	0.447
95% Jackknife UCL	0.448
95% Standard Bootstrap UCL	0.446
95% Bootstrap-t UCL	0.449
95% Hall's Bootstrap UCL	0.45
95% Percentile Bootstrap UCL	0.448
95% BCA Bootstrap UCL	0.449
95% Chebyshev(Mean, Sd) UCL	0.498
97.5% Chebyshev(Mean, Sd) UCL	0.533
99% Chebyshev(Mean, Sd) UCL	0.602

Assuming Gamma Distribution

95% Approximate Gamma UCL	0.449
95% Adjusted Gamma UCL	0.451

Potential UCL to Use

Use 95% Student's-t UCL 0.448

Figure 1.3-3. ProUCL printout of cadmium UCL calculations for the TA-16 Burn Ground.

Silver

General Statistics

Number of Valid Observations	31	Number of Distinct Observations	31

Raw Statistics		**Log-transformed Statistics**	
Minimum	0.164	Minimum of Log Data	-1.808
Maximum	7.95	Maximum of Log Data	2.073
Mean	0.872	Mean of log Data	-0.496
Median	0.537	SD of log Data	0.69
SD	1.361		
Coefficient of Variation	1.561		
Skewness	5.001		

Relevant UCL Statistics

Normal Distribution Test		**Lognormal Distribution Test**	
Shapiro Wilk Test Statistic	0.386	Shapiro Wilk Test Statistic	0.872
Shapiro Wilk Critical Value	0.929	Shapiro Wilk Critical Value	0.929
Data not Normal at 5% Significance Level		Data not Lognormal at 5% Significance Level	

Assuming Normal Distribution		**Assuming Lognormal Distribution**	
95% Student's-t UCL	1.287	95% H-UCL	1.005
95% UCLs (Adjusted for Skewness)		95% Chebyshev (MVUE) UCL	1.211
95% Adjusted-CLT UCL	1.509	97.5% Chebyshev (MVUE) UCL	1.404
95% Modified-t UCL	1.323	99% Chebyshev (MVUE) UCL	1.784

Gamma Distribution Test		**Data Distribution**
k star (bias corrected)	1.41	Data do not follow a Discernable Distribution (0.05)
Theta Star	0.619	
MLE of Mean	0.872	
MLE of Standard Deviation	0.734	
nu star	87.39	
Approximate Chi Square Value (.05)	66.84	**Nonparametric Statistics**
Adjusted Level of Significance	0.0413	95% CLT UCL 1.274
Adjusted Chi Square Value	65.84	95% Jackknife UCL 1.287
		95% Standard Bootstrap UCL 1.268
Anderson-Darling Test Statistic	2.718	95% Bootstrap-t UCL 2.337
Anderson-Darling 5% Critical Value	0.763	95% Hall's Bootstrap UCL 2.699
Kolmogorov-Smirnov Test Statistic	0.212	95% Percentile Bootstrap UCL 1.322
Kolmogorov-Smirnov 5% Critical Value	0.161	95% BCA Bootstrap UCL 1.626
Data not Gamma Distributed at 5% Significance Level		95% Chebyshev(Mean, Sd) UCL 1.937
		97.5% Chebyshev(Mean, Sd) UCL 2.398
Assuming Gamma Distribution		99% Chebyshev(Mean, Sd) UCL 3.304
95% Approximate Gamma UCL	1.14	
95% Adjusted Gamma UCL	1.157	
Potential UCL to Use		Use 95% Chebyshev (Mean, Sd) UCL 1.937

Figure 1.3-4. ProUCL printout of silver UCL calculations for the TA-16 Burn Ground.

Table 2.1-1
Ecological Screening Levels for Terrestrial Receptors

Chemical	Kestrel (Avian insectivore)	Kestrel (carnivore)	Robin (herbivore)	Robin (insectivore)	Robin (omnivore)	Deer mouse	Desert cottontail	Earthworm	Plant	Montane shrew	Red fox
Barium	11000	37000	820	1000	930	1800	3300	330	110	1300	41000
Cadmium	2	580	4.4	0.29	0.54	0.51	9.9	140	32	0.27	510
Silver	19	840	11	2.6	4.3	24	150	na[a]	560	14	4100
TCDD[2,3,7,8-] equivalent	0.000014[b]	0.000014[b]	0.00024[b]	0.0000041[b]	0.0000081[b]	0.00000058	0.000048	5	na	0.00000029	0.0000012

Note: ESLs from ECORISK Database, Version 2.3 (LANL 2008, 103352) unless otherwise noted.
[a] na = Not available.
[b]Avian ESLs for TCDD taken from ECORISK Database, Version 2.0 (LANL 2000, 080117).

Table 2.1-2
Comparison of EPCs with the Minimum ESLs

COPC	EPC[a] (mg/kg)	Minimum ESL[b] (mg/kg)	Receptor	Hazard Quotient
Barium	704.4	110	Plant	**6.4**
Cadmium	0.448	0.27	Montane shrew	**1.7**
Silver	1.94	2.6	Robin (insectivore)	**0.7**
TCDD[2,3,7,8-] equivalent	0.00000665	0.00000029	Montane shrew	**22.9**

[a] The EPC the 95% UCL.
[b] ESLs from ECORISK Database, Version 2.3 (LANL 2008, 103352).

Table 2.1-3
Hazard Quotient Analysis

COPECs	EPC (mg/kg)	Kestrel (Avian insectivore)	Kestrel (carnivore)	Robin (herbivore)	Robin (insectivore)	Robin (omnivore)	Deer mouse	Desert cottontail	Earthworm	Plant	Montane shrew	Red fox
Barium	704.4	0.06	0.02	**0.9**	**0.7**	0.08	**0.4**	0.2	**2.1**	**6.4**	**0.5**	0.02
Cadmium	0.448	0.2	0.0008	0.1	**1.5**	**0.8**	**0.9**	0.05	0.003	0.01	**1.7**	0.0009
Silver	1.94	0.1	0.002	0.2	**0.7**	**0.5**	0.08	0.01	na*	0.003	0.1	0.0005
TCDD[2,3,7,8-] equivalent	6.65E-06	**0.5**	**0.5**	0.03	**1.6**	**0.8**	**11.5**	0.1	0.000001	na	**22.9**	**5.5**
	HI	0.9	0.5	1	**5**	**2**	**13**	0.4	**2**	**6**	**25**	**6**

Note: Bolded values indicate HQs greater than 0.3 or HIs greater than 1.0.

* na = Not available.

Table 2.2-1
Comparison of EPCs to Background Concentrations in Soil

COPEC	EPC (mg/kg)	Soil Background Concentrations (mg/kg)
Barium	704.4	21-410
Cadmium	0.448	0.2–2.6
Silver	1.94	1

Table 2.2-2
Population Area Use Factors for Ecological Receptors

Receptor	Home Range[a] (ha)	Population Area[b] (ha)	PAUF[c]
Kestrel	106	4240	0.0006
Robin	0.42	16.8	0.15
Deer mouse	0.077	3.0	0.87
Desert cottontail	3.1	124	0.02
Montane shrew	0.39	15.6	0.17
Red fox	1038	41,520	0.00006

[a] Values from EPA 1993, 059384.
[b] Derived by 40HR.
[c] PAUF is calculated as the area of the site (2.6 ha) divided by the population area.

Table 2.2-3
Adjusted Hazard Quotient Analysis

COPECs	EPC (mg/kg)	Kestrel (Avian insectivore)	Kestrel (carnivore)	Robin (herbivore)	Robin (insectivore)	Robin (omnivore)	Deer mouse	Desert cottontail	Earthworm	Plant	Montane shrew	Red fox
Barium	704.4	0.00004	0.00001	0.1	0.1	0.01	0.3	0.004	**2.1**	**6.4**	0.09	0.000001
Silver	1.94	0.00006	0.000001	0.03	0.1	0.08	0.07	0.0002	na*	0.003	0.02	0.00000003
TCDD[2,3,7,8-] equivalent	6.65E-06	0.0003	0.0003	0.005	0.2	0.1	**10**	0.002	0.000001	na	**4**	0.0003
	Adjusted HI	0.0004	0.0003	0.1	0.4	0.2	**10**	0.0006	**2**	**6**	**4**	0.0003

Note: Bolded values indicate HQs greater than 0.3 or HIs greater than 1.0.

* na = Not available.

Table 2.2-4
Area Adjusted Hazard Quotient Analyses for TCDD[2,3,7,8-] equivalent

COPECs	EPC (mg/kg)	Deer mouse	Montane shrew
TCDD[2,3,7,8-] equivalent for area east of 16-0399	3.71E-05	64	128
TCDD[2,3,7,8-] equivalent for rest of area around 16-388 and 16-0399	1.2E-06	2	4

Table 2.2-5
Population Area Use Factors for Ecological Receptors

Receptor	Home Range[a] (ha)	Population Area[b] (ha)	PAUF for Area East of 16-0399[c]	PAUF for Rest of Area Around 16-0388 and 16-0399[d]
Deer mouse	0.077	3.0	0.03	0.83
Montane shrew	0.39	15.6	0.006	0.16

[a] Values from EPA 1993, 059384.
[b] Derived by 40HR.
[c] PAUF calculated as the area of the site (0.1 ha) divided by the population area.
[d] PAUF calculated as the area of the site (2.5 ha) divided by the population area.

Table 2.2-6
Area and PAUF Adjusted Hazard Quotient Analyses for TCDD[2,3,7,8-] equivalent

COPECs	EPC (mg/kg)	Deer mouse	Montane shrew
TCDD[2,3,7,8-] equivalent for area east of 16-0399	3.71E-05	1.9	0.8
TCDD[2,3,7,8-] equivalent for rest of area around 16-0388 and 16-0399	1.2E-06	1.7	0.6

Dioxin/Furan data w/o highs

General Statistics

Number of Valid Observations	34	Number of Distinct Observations	34
Raw Statistics		**Log-transformed Statistics**	
Minimum	2.17E-08	Minimum of Log Data	-17.65
Maximum	3.38E-06	Maximum of Log Data	-12.6
Mean	8.042E-07	Mean of log Data	-14.78
Median	4.17E-07	SD of log Data	1.398
SD	9.224E-07		
Coefficient of Variation	N/A		
Skewness	1.574		

Relevant UCL Statistics

Normal Distribution Test		**Lognormal Distribution Test**	
Shapiro Wilk Test Statistic	0.778	Shapiro Wilk Test Statistic	0.954
Shapiro Wilk Critical Value	0.933	Shapiro Wilk Critical Value	0.933
Data not Normal at 5% Significance Level		Data appear Lognormal at 5% Significance Level	
Assuming Normal Distribution		**Assuming Lognormal Distribution**	
95% Student's-t UCL	1.072E-06	95% H-UCL	2.086E-06
95% UCLs (Adjusted for Skewness)		95% Chebyshev (MVUE) UCL	2.22E-06
95% Adjusted-CLT UCL	1.11E-06	97.5% Chebyshev (MVUE) UCL	2.765E-06
95% Modified-t UCL	1.079E-06	99% Chebyshev (MVUE) UCL	3.836E-06
Gamma Distribution Test		**Data Distribution**	
k star (bias corrected)	0.745	Data appear Gamma Distributed at 5% Significance Level	
Theta Star	1.08E-06		
MLE of Mean	8.042E-07		
MLE of Standard Deviation	9.319E-07		
nu star	50.64		
Approximate Chi Square Value (.05)	35.3	**Nonparametric Statistics**	
Adjusted Level of Significance	0.0422	95% CLT UCL	1.064E-06
Adjusted Chi Square Value	34.67	95% Jackknife UCL	1.072E-06
		95% Standard Bootstrap UCL	1.06E-06
Anderson-Darling Test Statistic	0.335	95% Bootstrap-t UCL	1.133E-06
Anderson-Darling 5% Critical Value	0.786	95% Hall's Bootstrap UCL	1.107E-06
Kolmogorov-Smirnov Test Statistic	0.0924	95% Percentile Bootstrap UCL	1.056E-06
Kolmogorov-Smirnov 5% Critical Value	0.157	95% BCA Bootstrap UCL	1.116E-06
Data appear Gamma Distributed at 5% Significance Level		95% Chebyshev(Mean, Sd) UCL	1.494E-06
		97.5% Chebyshev(Mean, Sd) UCL	1.792E-06
Assuming Gamma Distribution		99% Chebyshev(Mean, Sd) UCL	2.378E-06
95% Approximate Gamma UCL	1.154E-06		
95% Adjusted Gamma UCL	1.175E-06		
Potential UCL to Use		Use 95% Approximate Gamma UCL	1.154E-06

Figure 2.2-1. ProUCL printout of dioxin/furan UCL calculations without the highest values for the TA-16 Burn Ground.

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REGULATORY TOXICOLOGY AND PHARMACOLOGY 3, 224–238 (1983)

Regulatory History and Experimental Support of Uncertainty (Safety) Factors[1]

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Received April 2, 1983

A synthesis of available literature on uncertainty (safety) factors which are used to estimate acceptable daily intakes (ADIs) for toxicants is presented. This synthesis reveals reasonable qualitative biological premises, as well as specific biological data that [illegible] gestion is made in ord[illegible] tainty are also identified.

INTRODUCTION

Sensible regulation of industrial or agricultural chemicals by governmental agencies to protect public health demands that all appropriate toxicity data available on a specific chemical be used to estimate a "safe" environmental or industrial level of exposure to humans. The scientific support of such public health regulations requires a two-phased approach by toxicologists: the compilation of adequate dose–response data, usually from animal experiments, but whenever possible from available human observations, to obtain "no-effect" levels; and the assessment of these data to provide "safe" levels or to define risk levels. For a toxic chemical (i.e., noncarcinogen)[2] the "safe" level for humans is termed the acceptable daily intake (ADI).[3] Uncertainty (also called safety) factors are used extensively with human or animal toxicity data to estimate these ADIs by the general formula

[1] Although the research (or other work) described in this article has been funded wholly or in part by the United States Environmental Protection Agency, it has not been subjected to the Agency's required peer and administrative review and, therefore, does not necessarily reflect the view of the Agency and no official endorsement should be inferred.

[2] In regulatory parlance "toxicants" (i.e., noncarcinogenic chemicals) are postulated to exert their toxic effects by mechanisms which exhibit thresholds. Therefore, derivation of an ADI is appropriate. No such threshold mechanism has been universally accepted for carcinogens. Therefore, derivation of an ADI for these chemicals has not been recommended. (See text footnote 6.)

[3] An ADI is defined as the amount of toxicant in milligrams per kilogram body weight per day (or in milligrams per day for a 70-kg person) which is not anticipated to result in any adverse effects after chronic exposure to the general population of humans, including sensitive subgroups. Adverse effects are considered as functional impairment or pathological lesions which may affect the performance of the whole organism, or which reduce an organism's ability to respond to an additional challenge (U. S. EPA, 1980). Operationally, ADIs are calculated by dividing a NOEL, NOAEL, or LOAEL derived from human or animal toxicity

0273-2300/83 $3.00
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$$\text{ADI} = \frac{\text{"no-effect" level}}{\text{uncertainty factor}}.$$

The purpose of this paper is to present a brief regulatory history of uncertainty factors and to discuss supporting experimental observations. It is emphasized that uncertainty factors are adjustments of the NOEL, NOAEL, or LOAEL reported for small populations of humans or experimental animals in order to estimate the comparable NOAEL from chronic contaminant exposure for a large human population which includes sensitive subgroups (this level being synonymous with an ADI). However, some of these factors also incorporate a degree of safety. Other recent publications which discuss uncertainty factors are available (Calabrese, 1982; Food Safety Council, 1982). The former manuscript delves primarily into additional areas of extrapolation from experimental animals to humans; the latter review also discusses other areas pertinent to food safety.

REGULATORY HISTORY

Scientific guidance and recommendations on the use of ADIs have been adopted by several United States governmental and international bodies such as the U. S. Environmental Protection Agency (U. S. EPA), the Food and Drug Administration (FDA), the Joint Food and Agricultural Organization/World Health Organization (FAO/WHO) Food Standards Programme (Codex Alimentarius Committee on Food Additives), and by the FAO Committee on Pesticide Residues and the WHO Expert Committee on Pesticide Residues.

Initial publications in this area of regulation appear to be by Lehman and Fitzhugh (1954) of the Food and Drug Administration. They suggested that ADIs for food additives or contaminants be derived from a chronic animal NOEL or NOAEL (measured in mg/kg of diet) by dividing by a 100-fold uncertainty factor. These authors reasoned that this factor accounted for several areas of uncertainty: intra-(human) or inter-(animal to human) species variability or intrastrain variability in response to the toxicity of a chemical, allowance for sensitive human subpopulations due to illness as compared to healthy experimental animals, and possible synergistic action of any one of the many intentional or unintentional food additives or contaminants in the human diet.

Similar areas of uncertainty have also been addressed by other authors. For example, Bigwood (1973) (associated with the WHO/FAO) justified the 100-fold uncertainty factor for food additives on the basis of differences in body size of the laboratory animal vs that of man, differences in food requirements varying with age, sex, muscular expenditure, and environmental conditions within a species, differences in water

studies by one or more uncertainty factors. These acronyms are defined as follows. NOEL: no-observed-effect level. That dose of chemical at which there are no statistically or biologically significant increases in frequency or severity of effects between the exposed population and its appropriate control. NOAEL: no-observed-adverse-effect level. That dose of chemical at which there are no statistically or biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control. Effects are produced at this dose, but they are not considered to be adverse. LOAEL: lowest-observed-adverse-effect level. The lowest dose of chemical in a study or group of studies which produces statistically or biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control.

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balance of exchange between the body and its environment among species, and differences in susceptibility to the toxic effect of a given contaminant among species. A similar approach has been adopted by the WHO Expert Committee for Pesticide Residues (Lu, 1979). Vettorazzi (1976, 1980) substantiated the use of the 100-fold uncertainty factor by discussing differences in susceptibility between animals and humans to toxicants, variations in sensitivities in the human population, the fact that the number of animals tested is small compared with the size of the human population that may be exposed, the difficulty in estimating human intake, and the possibility of synergistic action among chemicals within the human diet.

Although the specific areas of uncertainty described by these authors (Lehman and Fitzhugh, 1954; Bigwood, 1973; Vettorazzi, 1976, 1980) to support a 100-fold uncertainty factor differ somewhat, they can be generally viewed as due to intra- or interspecies variability. It has been suggested that two 10-fold uncertainty factors, one for each type of variability, be used to describe the 100-fold uncertainty factor in some instances (Bigwood, 1973; Klassen and Doull, 1980; Food Safety Council, 1982).

The FDA expanded their initial approach in the derivation of ADIs when chronic data were unavailable. In such cases where subchronic animal NOELs or NOAELs were available in two species the FDA recommended a factor of 1000 instead of 100, the additional 10-fold was ostensibly due to the added uncertainty when estimating an ADI from adequate shorter-term toxicity data (Kokoski, 1976). If subchronic data were available for only one species a 2000-fold uncertainty factor was recommended as it seemed likely that the extra margin of uncertainty would probably encompass the range of sensitivity of two species which is normally required (Shibko, 1981). The National Academy of Sciences (NAS, 1977) recommended a similar approach to uncertainty factors when estimating ADIs for pollutants in drinking water. However, the NAS recommendation differed from the FDA's in two regards: first, the NAS suggested that a NOEL or NOAEL be measured in milligrams per kilogram body weight per day versus milligrams per kilogram of diet, and second, the NAS outlined the use of a 10-fold uncertainty factor to estimate and ADI if valid experimental results from studies on prolonged ingestion by man were available. This latter idea is consistent with the general view that the 100-fold uncertainty factor is composed of two 10-fold units (*v. supra*).

The U. S. EPA (1980) recommended uncertainty factors for estimating ADIs of pollutants in ambient waters based on the NAS reasoning. The U. S. EPA also recommended an additional uncertainty factor between 1 and 10 when an ADI was estimated from a LOAEL (if a NOAEL was unavailable) in order to adjust the LOAEL into the range of a NOAEL. For example, if an ADI was calculated from an animal chronic LOAEL (other data being unavailable), an uncertainty factor of between 100 and 1000 would be recommended. Each of these latter recommendations (FDA, NAS, and U. S. EPA) were based on the 100-fold uncertainty factor, as discussed previously, when calculating an ADI from a NOEL or NOAEL found in animals.

INTRASPECIES ADJUSTMENT

Figure 1 is a plot of frequency versus an intraspecies adjustment factor obtained by raising 10 to the power (3 ÷ probit, log-dose slope) using 490 individual probit,

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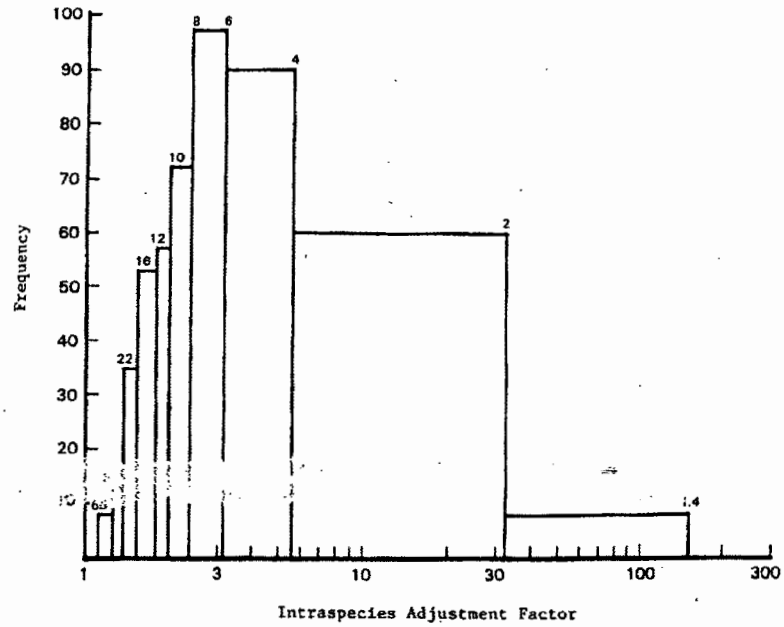

FIG. 1. Frequency vs an intraspecies adjustment factor obtained by raising 10 to the power (3 standard deviations ÷ the probit, log-dose slope). Probit, log-dose slopes are shown within the figure. Adapted from Fig. 1 (Weil, 1972).

log-dose slopes from Weil (1972). These slopes were for acute lethality and varied from approximately 1.4 to 65.

The adjustment factors of Fig. 1 can be considered as reductions in milligrams per kilogram body weight (b.w.) dose needed to scale down a median response (in this case an LD_{50}) three probits. A three-probit reduction places the median response in the general range expected for a potential sensitive subgroup of the population under study (e.g., $LD_{0.13}$). Numerical values associated with the frequencies of Fig. 1 are the slopes from Weil (1972). The most frequently occurring slopes lie within the range of 6 to 8 (97 occurrences out of 490).

Figure 1 indirectly supports[4] a 10-fold uncertainty factor to account for intraspecies variability when estimating an ADI. Approximately 92% of the probit, log-dose slopes analyzed by Weil (1972) had values of greater than 3; for these chemicals a 10-fold decrease in dose would drop a median response (e.g., an LD_{50}) below the general range expected to result in death for only the most sensitive members of this rather homogeneous population. For the remaining chemicals (i.e., those with slopes of less than 3) a 10-fold reduction in dose would not achieve this concurrent reduction in expected response.

Based on Fig. 1, a 10-fold reduction in milligrams per kilogram b.w. dose for toxicants to account for intraspecies variability when estimating an ADI at first seems

[4] The support is indirect because the endpoints (percentage mortality versus a NOEL, NOAEL, or LOAEL) are not strictly comparable.

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conservative. The average probit, log-dose slope of 7.8 is associated with only a 2.4 reduction in dose to effect a three-probit drop in response. However, these probit, log-dose slopes are garnered on laboratory rats which are generally expected to be less heterogeneous in response to the toxicity of a contaminant when compared to the human population. Greater heterogeneity in response is associated with lower slopes and correspondingly greater dose reductions. Such greater heterogeneity in humans is supported by Krasovskii (1976) who claimed a 6-fold difference in sensitivity to the action of fluorine and nitrates in children, and a general 3- to 5-fold difference in sensitivity between children and adults. Thus, the intraspecies variability for humans to the toxicity of chemicals might be estimated from these data to be between 18 and 30.

Mantel and Bryan (1961) discussed this issue of probit, log-dose slopes in some detail and concluded, for purposes of extrapolation for carcinogens, that a slope of 1.0 is likely to be conservative. Such a slope would correspond to a 1000-fold reduction in dose needed to obtain a three-probit drop in response. Other authors have also discussed this issue of probit, log-dose slopes (Munro and Krewski, 1981; Oser, 1969). [illegible] comments are addre[illegible]

From this brief presentation of data it seems somewhat reasonable to employ a 10-fold uncertainty factor to account for intraspecies variability in lieu of chemical-specific toxicity data. However, it is also necessary to examine this area of uncertainty experimentally or theoretically in much greater detail.

INTERSPECIES ADJUSTMENT

Figure 2 is a plot of experimental animal weight (w) versus an interspecies adjustment factor, calculated as the cube root of the assumed average human body weight (70 kg) divided by w

$$\text{i.e.,} \quad \sqrt[3]{\frac{70}{w}}.$$

These factors account for differences in milligrams per kilogram b.w. doses due to different body-surface areas between experimental animals and man, based on the assumption that different species are equally sensitive to the effects of a toxin on a dose per unit surface area. When this surface area dose is converted to corresponding units of milligrams per kilogram b.w., species with greater body weight (e.g., humans) appear to be more sensitive to the toxicity of a contaminant than species of smaller body weight (e.g., rodents). Dose conversions based on body-surface area are generally thought to more accurately reflect differences among species in several biological parameters when compared to conversions based on milligrams per kilogram b.w. (Mantel and Schneiderman, 1975). For acute toxicity to alkylating agents, equivalent doses among mammals are more accurately estimated by dose per body-surface area rather than dose per kilogram b.w. (Rall, 1969; Homan, 1972).

These factors in Fig. 2 can be thought of as reductions in experimental animal dose (in milligrams per kilogram b.w.) needed to estimate a comparable human milligram per kilogram b.w. dose. For example, a comparable milligram per kilogram b.w. dose for the average person (70 kg) estimated from a rat (0.33 kg) given an experimental dose of 100 mg/kg b.w. is 17. This human dose is derived by dividing

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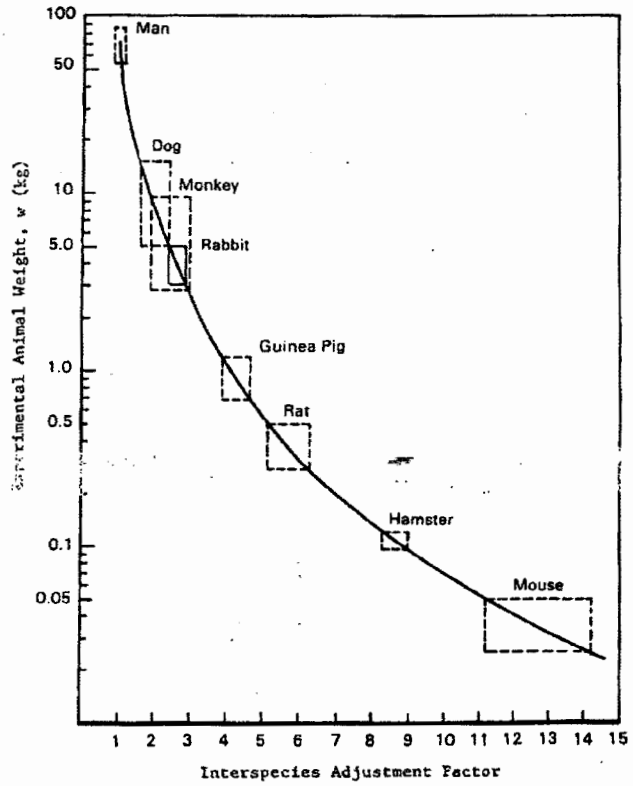

FIG. 2. Experimental animal weight (w) vs an interspecies adjustment factor calculated as the cubed root of the ratio between the assumed average human body weight (70 kg) and w. Enclosed areas along the function represent general ranges of average body weights of experimental adult animals. Rabbit values are represented by the box with solid lines. Values are from Altman and Dittmer (1962).

the animal dose, 100 mg/kg b.w., by an interspecies adjustment factor of about 6.0 (i.e., the cube root of the expression: 70 kg/0.33 kg). The enclosed areas along the function represent ranges of average adult weights for different experimental animals (Altman and Dittmer, 1962).

Figure 2 can be construed as support of a 10-fold uncertainty factor to account for interspecies variability to the toxicity of a chemical when estimating an ADI from animal doses measured in milligrams per kilogram b.w. The NAS (1977) confirms this contention by stating that man is generally more vulnerable than experimental animals on the basis of body weight by a factor between 6 and 12, but displays no supporting data. Evans *et al.* (1944) found that humans were more sensitive on a milligrams per kilogram b.w. basis than rats to a number of metallic poisons. Ratios of toxic doses between rats and humans varied between 2.5 and 152, with a geometric mean of approximately 12. Hayes (1967) compared either the smallest acute dose (milligrams per kilogram b.w.) with serious effects or the largest acute nonfatal dose for six pesticides between rats and humans. Ratios varied from 1.9 to 100 with a

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geometric mean of approximately 11. Six comparisons of chronic doses which yielded similar effects varied from 0.58 to 9.4 with a geometric mean of approximately 2.9. The ratio between a 70-kg person and a 0.33-kg rat described in Fig. 2 is approximately 6.0. Evans *et al.* (1944) also described ratios of maintenance doses of vitamins between rats and humans. Such ratios varied between 2.6 and 12.9, with a geometric mean of 4.3. Lehman and Fitzhugh (1954) mention that humans were 4 or 10 times as sensitive to arsenic or fluorine in their diet as dogs or rats, respectively. These latter doses, however, were not measured in milligrams per kilogram b.w. Apparently little additional quantitative work has been done comparing the toxicity of chemicals between animals and humans, at least for the purpose of estimating safe ambient exposures. Publications in the area of estimating therapeutic doses for antineoplastic agents are available (Goldsmith *et al.*, 1975).

However, a 10-fold uncertainty factor based on these discussions to account for interspecies extrapolation appears to incorporate a margin of safety if the underlying assumption of dose equivalence among species per unit of surface area is correct. For instance, with most experimental animals a 10-fold reduction in milligrams per kilogram [illegible] would underestimate [illegible] factor between 1 and [illegible] (see Fig. 2). With mice this 10-fold dose reduction would actually predict a higher ADI. Therefore, it might be more accurate to replace this 10-fold factor with a dose adjustment between the experimental animal and man, as in Fig. 2.

In contrast, Hoel *et al.* (1975) feel that the quantitative extrapolation in the area of chronic toxic effects (ostensibly carcinogenesis) from animal to human should include both an adjustment factor based on body weight as in Fig. 2, and another factor determined by information on the contaminant and species and strain of the test animal. These authors support their suggestion by a discussion on the expected larger differences in response within the population of humans as compared to the test animal because of the heterogeneity of the human population, in addition to the differences in response among humans and animals due to different body-surface areas. However, Hoel *et al.* do not display any supporting data, and their discussion appears similar to those evoked by Lehman and Fitzhugh (1954), Bigwood (1973), or Vettorazzi (1976, 1980) for the use of a 100-fold uncertainty factor (*v. supra*). Thus, the Hoel *et al.* (1975) proposal, while perhaps reasonable for carcinogenesis, lacks specificity when estimating ADIs for toxicants. This weakness is especially evident when the available toxicity data on a contaminant are sparse.

Although data exist to support the contention that a 10-fold decrease in milligrams per kilogram b.w. animal dose is adequate to adjust to humans when chemical-specific data are not available, this area of uncertainty could profit from additional investigation.

SUBCHRONIC TO CHRONIC EXPOSURE ADJUSTMENT

Figure 3 is a plot of frequency versus ratios of subchronic to chronic exposure for either NOAELs, LOAELs, or their combination. These frequency plots are derived from a series of toxicity experiments for various compounds compiled by Weil and McCollister (1963). The subchronic exposures reported by these authors varied between 30 and 210 days; the mean value was 92 days. The chronic exposures were all 2 years. All effect levels (i.e., NOAELs or LOAELs) were determined for rats or dogs.

These experimentally determined ratios can be considered as reductions in subchronic NOELs, NOAELs, or LOAELs in order to yield the corresponding chronic

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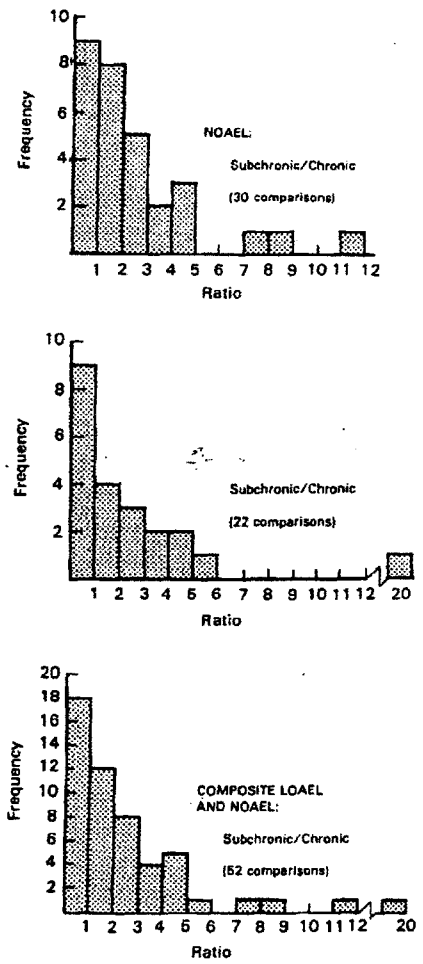

FIG. 3. Frequency vs the ratio of subchronic to chronic exposures for either NOAELs, LOAELs, or composite NOAEL–LOAEL values. Adapted from Table 1 (Weil and McCollister, 1963).

effect level. For example, a ratio of 5.0 indicates that the chronic NOAEL or LOAEL was 5-fold less than the corresponding NOAEL or LOAEL for the given chemical after subchronic exposure. It is evident from Fig. 3 that for more than half of the observed chemicals ratios are 2.0 or less. Approximately 96% of these ratios are below a value of 10.

Figure 3 supports a 10-fold uncertainty factor to account for estimating an ADI from a subchronic effect level for a chronic level is unavailable. However, the average subchronic to chronic NOAEL or LOAEL ratio is approximately 2, which indicates that this uncertainty factor also incorporates a margin of safety. For example, if an uncertainty factor of 1000 is used to estimate an ADI from a subchronic animal

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NOAEL, the ADI will be underestimated by 5-fold in over half the cases when compared to using the average ratio of 2. [An uncertainty factor of 1000 (i.e., $10_1 \times 10_2 \times 10_3$) as compared to 200 (i.e., $10_1 \times 10_2 \times 2$) in the denominator.][5]

McNamara (1976) reported the frequency of experimentally determined ratios of subchronic to chronic exposure for NOAELs on a different series of chemicals. Values of 1.0 or less were reported in 34 of 41 ratios; the remaining ratios were all less than 3.0. His compiled data suggest that dose reductions of 3.0 or less will be adequate to estimate a chronic NOAEL from a corresponding subchronic NOAEL.

The State of Michigan (1981) recommends a dose reduction of approximately 4.8 in order to adjust a mammalian subchronic NOAEL to a corresponding chronic NOAEL. This recommendation is based on a percentile-rank analysis of selected data from Weil and McCollister (1963). The NAS (1965) recommends 5% of an ADI (established by a NOAEL from a 90-day feeding study and a 100-fold uncertainty factor) as a negligible-residue level for pesticides in foodstuffs. Assuming contaminated foodstuffs will be consumed over a lifetime this recommendation can be seen as a 20-fold reduction in dose to adjust a NOAEL observed after subchronic exposure (90 days) to that expected after chronic exposure. McNamara (1976) suggests a 10-fold reduction in dose to adjust a 3-month (subchronic) no-effect dose (NOAEL) to an expected lifetime NOAEL based on both his work and Weil and McCollister (1963).

These recommendations indicate that unless contaminant-specific data are available, it seems reasonable to employ a 10-fold uncertainty factor to account for differences between subchronic and chronic effect levels. Based on Fig. 3 such ratios are likely to be less than 10, 96% of the time.

LOAEL TO NOAEL ADJUSTMENT

Figure 4 is a plot of frequency versus ratios of LOAEL to NOAEL for either subchronic or chronic exposure, or their combination. The data for this figure are also adapted from Weil and McCollister (1963). These experimentally determined ratios can be thought of as reductions in a LOAEL found after subchronic or chronic exposure in order to yield the corresponding NOAEL. For example, a ratio of 3.0 indicates that the NOAEL found after subchronic or chronic exposure is 3-fold less than the corresponding LOAEL for a particular chemical. It is evident from Fig. 4 that all chemicals have values of 10 or less. Of these ratios 96% have values of 5 or less.

Figure 4 supports an uncertainty factor between 1 and 10 to account for estimating an ADI from a LOAEL if a NOAEL is unavailable. These data prompted Weil (1972) to suggest an additional 5-fold reduction in dose when estimating a corresponding maximum no-ill-effect level (or NOAEL) from a minimum effect level (or LOAEL). The U. S. EPA (1980) recommends that this variable uncertainty factor reflects a scientific judgment of the difference between the observed LOAEL and the hypothesized NOAEL. This difference will not necessarily be the same from experiment to experiment (as is apparent from the ratios in Fig. 4). In practice the value for this variable uncertainty factor has been chosen by the U. S. EPA (1980) from values among 1 through 10 based on the severity of the adverse effect of the LOAEL. For example,

[5] Subscripts on 10's refer to Guideline Nos. 1 through 3. See Table 1.

FIG. 4. I
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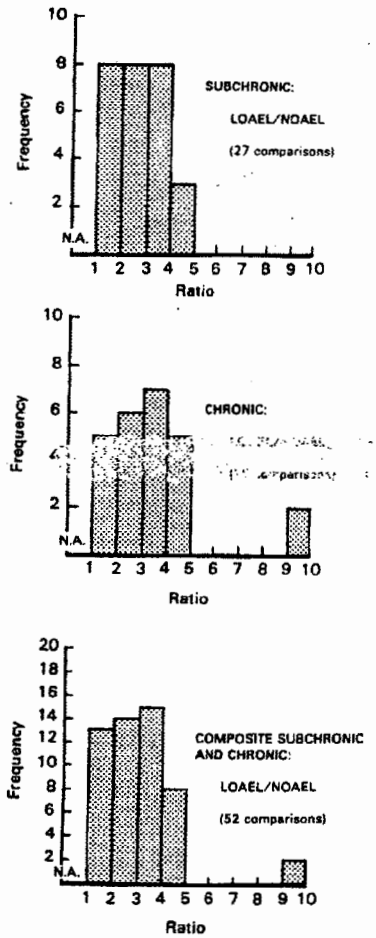

FIG. 4. Frequency vs the ratio of LOAEL to NOAEL after either subchronic, chronic, or composite subchronic and chronic exposures. A ratio of 1.0 or less is not allowable (N.A.) by definition. Adapted from Table 1 (Weil and McCollister, 1963).

if the LOAEL represents liver cell necrosis, a higher value is suggested for this uncertainty factor (perhaps 10). If the LOAEL is fatty infiltration of the liver, then a lower value is suggested (perhaps 3). The hypothesized NOAEL should be closer to the LOAEL showing less severe effects.

This concept of using variable uncertainty factors based on the severity of the observed effects if firmly established in deriving threshold limit values (TLVs) for industrial chemical exposures (Stokinger, 1972). This experience, in conjunction with the experimental data (Fig. 4) indicates that it is reasonable to employ a variable uncertainty factor between 1 and 10 when estimating an ADI from a LOAEL, in lieu of chemical-specific data.

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DISCUSSION

As summarized in Table 1, several uncertainty factors are currently recommended to estimate ADIs for toxicants depending on the available human or animal toxicity data. These factors are 10, 100, or 1000 (U. S. EPA, 1980; NAS, 1977). However a perusal of the literature that discusses these factors indicates that 10, 100, and 1000 generally represent different categories resulting in multiples of 10 (i.e., 10_1, $10_1 \times 10_2$, $10_1 \times 10_2 \times 10_3$)[5] applied to the type of available data used for the extrapolation. For example, an uncertainty factor of 10 is used to estimate ADIs with appropriate chronic human data and reflects intraspecies variability to the adverse effects of a chemical (i.e., 10_1). An uncertainty factor of 100 is used to estimate ADIs with sufficient chronic animal data (supported by fragmentary human data). It accounts for both intra- and interspecies variability (i.e., $10_1 \times 10_2$). An uncertainty factor of 1000 is used to estimate ADIs with satisfactory subchronic animal data (if chronic data are unavailable). It incorporates the uncertainty in extrapolating data from sub- [illegible] e., 10_3), as well as [illegible] variable uncertainty factor between 1 and 10 is applied to estimate ADIs using LOAELs (if NOAELs are unavailable). This uncertainty factor reduces the LOAEL into the range of a NOAEL.

In cases where data do not completely fulfill the conditions for a category of uncertainty factors (either 10, 100, or 1000), or appear to be intermediate between two categories, intermediate uncertainty factors can be used to estimate the ADI. This approach is discussed by the U. S. EPA (1980). Such intermediate uncertainty factors may be developed on a logarithmic scale (e.g., 33 being halfway between 10 and 100). This modification of the NAS (1977) approach allows scientists to judge whether, for example, dog is a more appropriate species than mouse to extrapolate to man in case of a particular chemical, and on that basis to assign an intermediate uncertainty factor instead of a uniform 10 for interspecies variability (i.e., 10_2).

Furthermore, intimate knowledge of a chemicals mechanism of toxicity, critical effect and/or pharmacokinetics in humans and experimental animals allows for the use of smaller uncertainty factors. For example, U. S. EPA (1981) suggests a 10-fold uncertainty factor to calculate an ADI for cholinesterase-inhibiting insecticides when adequate human dose–response data are available on blood cholinesterase inhibition regardless of the length of exposure. This recommendation is based on the extensive knowledge on the mechanism of toxic action and critical effect of these insecticides. U. S. EPA (1980) does not use the "no effect"/uncertainty factor approach in estimating environmental exposures when sufficient data are available on a chemical's critical effect and human pharmacokinetics. These latter procedures, however, can be used for only a few chemicals because a fairly complete data base is required.

A possible modification to the standard approach would be to present a range for the ADI rather than one value. The range could be based at the high end on the average reductions in dose needed to estimate the ADI (from Figs. 1 and 3) and the body-surface area ratio (Fig. 2), and at the low end on the standard 10-fold reductions (i.e., 10_1, 10_2, 10_3). As an example, an ADI estimated from a subchronic mouse NOAEL of 100 mg/kg/day would range from 0.10 to approximately 1.6 mg/kg/day (or 7.0 to approximately 110 mg/day for a 70-kg person). In this case the value 0.10 is equal to: 100 mg/kg/day $\div$ ($10_1 \times 10_2 \times 10_3$); whereas the higher value of 1.6 represents: 100 mg/kg/day $\div$ ($2.4 \times 13.3 \times 2.0$). In this latter calculation 2.4 is the

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TABLE 1

GUIDELINES, EXPERIMENTAL SUPPORT, AND REFERENCES FOR THE USE OF UNCERTAINTY (SAFETY) FACTORS[a]

Guidelines[b]	Experimental support	References
(1) **Use a 10-fold factor when extrapolating from valid experimental results from studies on prolonged ingestion by man.** This 10-fold factor protects the sensitive members of the human population estimated from data garnered on average healthy individuals	Log-probit analysis; Log probit analysis; Comparison human senstivity	Mantel and Bryan, 1961; Weil, 1972; Krasovskii, 1976
(2) **Use a 100-fold factor when extrapolating from valid results of long-term feeding studies on experimental animals with results of studies of human ingestion not available or scanty (e.g., acute exposure only).** This represents an additional 10-fold uncertainty factor in extrapolating data from the average animal to the average man.	Body-surface area dose equivalence; Toxicity comparison between humans and rats; between humans and rats or dogs	Rall, 1969; Evans *et al.*, 1944, and Hayes, 1967; Lehman and Fitzhugh, 1954
(3) **Use a 1000-fold factor when extrapolating from less than chronic results on experimental animals with no useful long-term or acute human data.** This represents an additional 10-fold uncertainty factor in extrapolating from less than chronic to chronic exposures.	Subchronic/chronic NOAEL comparison; Subchronic/chronic NOAEL or LOAEL comparison	McNamara, 1976; Weil and McCollister, 1963
(4) **Use an additional uncertainty factor of between 1 and 10 depending on the sensitivity of the adverse effect when deriving an ADI from a LOAEL.** This uncertainty factor drops the LOAEL into the range of a NOAEL.	LOAEL/NOAEL comparison	Weil and McCollister, 1963

[a] These factors are to be applied to the highest valid NOAEL or NOEL which does not have a valid LOAEL equal to or below it, in calculating an ADI when no indication of carcinogenicity of a chemical exists.

[b] Guidelines are in bold print. Guidelines 1 and 2 are supported by the FDA and the WHO/FAO deliberations (Lehman and Fitzhugh, 1954; Bigwood, 1973; Vettorazzi, 1976, 1980); Guidelines 1–3 have been established by the NAS (1977) and are used in a similar form by the FDA (Krasovskii, 1976); Guidelines 1–4 are recommended by the U. S. EPA (1980).

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reduction in dose based on the average probit, log-dose slope of 7.8 (Fig. 1), 13.3 is the mouse to human reduction in dose based on the interspecies adjustment factor in Fig. 2 (mouse weight assumed to be 0.03 kg), and 2.0 is the assumed average subchronic to chronic ratio (Fig. 3).

Uncertainty factors have generated much discussion because they have been used to estimate ADIs for toxicants whose data bases vary widely in both completeness and discrepancy. Several reports have been critical. For instance, Golberg (1975) asserts that agreement over the issue of uncertainty factors is tantamount to an admission of lack of essential information for risk assessment. Unfortunately, such lack of essential information is commonplace for many of the chemicals in our environment, and yet regulatory decisions on these chemicals are necessary.

Munro and Krewski (1981) criticize the uncertainty factor approach to human health risk estimation first on the grounds that the NOEL will depend on sample size, and second that it does not account for the slope of the dose–response curve. This latter criticism is also discussed by Oser (1969). As an example of this latter criticism, a 10-fold uncertainty factor may provide a reasonable approximation of [illegible] sensitive individual [illegible] conservative if the slope is steeper and not protective enough if the slope is shallower (see also previous discussion under Intraspecies Adjustment).

The first criticism is somewhat mitigated by requiring statistically or biologically significant differences (or lack of) when determining NOELs, NOAELs or LOAELs, but as Munro and Krewski (1981) indicate 0/10 and 0/100 still have different interpretations. The U. S. EPA (1980) outlines in some detail the proper choice of an effect level when faced with several, but the outline still does not completely address this first criticism.

The second criticism, that uncertainty factors do not account for the slope of the dose–response curve, raised by both Munro and Krewski (1981) and Oser (1969), has not been addressed in any systematic way. Perhaps this should not be expected. Chronic and subchronic toxicity tests are seldom conducted with a sufficient number of closely spaced doses such that a probit, log-dose slope can be determined, unless such tests are for carcinogenicity.[6] This area of uncertainty could use much additional investigation.

However, scientists associated with the WHO/FAO (Bigwood, 1973; Vettorazzi, 1976, 1980), the FDA (Lehman and Fitzhugh, 1954; Kokoski, 1976), the NAS (1977), the U. S. EPA (1980), and independent groups such as the Food Safety Council (1982) have endorsed the use of uncertainty factors. Moreover, the data discussed in this paper suggest that these factors are not arbitrary as is commonly perceived, although several of them incorporate a margin of safety that may vary.

Thus, as long as toxicant-specific human health data are meager or nonexistent, or comparable pharmacokinetic studies in humans and animals have not been conducted, uncertainty factors seem necessary for estimating ADIs of toxicants for long-term, low-level exposure. Their use in schemes for estimating acceptable intakes for

[6] Uncertainty factors have not been recommended with carcinogenesis data (U. S. EPA, 1980; NAS, 1977; Mantel and Schneiderman, 1975; State of Michigan, 1981). Weil (1972) suggested, however, the use of a 5000-fold uncertainty factor when estimating a safe level of exposure to a carcinogen from a minimum effect level (i.e., a LOAEL). This factor incorporates the standard 100-fold factor for chronic animal data (i.e., $10_1 \times 10_2$), a 5-fold factor because the extrapolation starts from a LOAEL (as discussed in the text) and an additional 10-fold factor because of the general irreversibility of the mechanisms of carcinogenesis.

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exposures of shorter duration, or of multiple chemicals are also being investigated. The lack of data on chemical toxicity is even more apparent in these areas.

ACKNOWLEDGMENTS

The authors acknowledge the typing and editorial assistance of Patricia A. Daunt and Annette T. Pressley in the preparation of this document, and the helpful discussion of Dr. Linda S. Erdreich on scientific issues.

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