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Human Health Risk-Based Screening Methodology

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Acronyms and Abbreviations

ABS	skin absorption factor
AF	adherence factor
ALARA	as low as reasonably achievable
CMS	corrective measures study
COPC	chemical of potential concern
CRDL	contract-required detection limit
Csat	soil saturation concentration
DOE	US Department of Energy
DOE-AL	US Department of Energy-Albuquerque Office
EPA	US Environmental Protection Agency
ER	environmental restoration
GI	gastrointestinal (tract)
HEAST	Health Effects Assessment Summary Tables
IEUBK	Integrated Exposure Uptake Biokinetic
IRIS	Integrated Risk Information System
IWP	installation work plan
NCEA	National Center for Environmental Assessment
NFA	No further action
NMED	New Mexico Environment Department
PAH	polycyclic aromatic hydrocarbon
PEF	particulate emission factor
PRG	preliminary remediation goal
PRS	potential release site
RAGS	Risk Assessment Guidance for Superfund
RCRA	Resource Conservation and Recovery Act
RfD	reference dose
RFI	Resource Conservation and Recovery Act (RCRA) Facility Investigation

SAL	screening action level
SOP	standard operating procedure
UCL	upper tolerance limit
VF	volatilization factor

1.0 INTRODUCTION

A human health screening assessment is the initial step used to evaluate the potential for risk to human health at sites potentially influenced by Laboratory activities and is part of the ER Project's approach to the Resource Conservation and Recovery Act (RCRA) corrective action process. The installation work plan (IWP) describes how the screening assessment relates to the overall corrective action approach employed by the ER Project (LANL 1998, 62060, Section 3.2). The main purpose of a screening assessment is to provide a simple but scientifically sound approach to streamlining the assessment process. The use of a screening assessment within the context of estimating potential risk at a site is entirely at the discretion of the risk assessor, i.e., a baseline risk assessment may be conducted at any site if the risk assessor deems it preferable to a screening assessment.

A screening assessment is most useful for sites that have a limited number of contaminants and where land use and potential exposures are fairly well defined. However, a screening assessment may also be useful in identifying key contaminants and exposure routes at a complex site. When contaminant transport and exposure assessment require complex modeling efforts, screening assessments have limited utility and a baseline risk assessment should be used. Based on the results of the screening assessment, a site may be proposed for no additional evaluation of potential human health risk; further site investigation requiring more sampling and a baseline risk assessment; remediation; or a corrective measures study (CMS). Because human health is only one aspect of site evaluation, the determination of acceptable human health risk does not translate into a final no further action (NFA) recommendation until other aspects of the investigation (e.g., ecological risk) are satisfactorily resolved.

This methodology document provides guidance to Environmental Restoration (ER) Project risk assessors as they perform human health risk-based screening assessments at Los Alamos National Laboratory (the Laboratory) sites. Implementation of this methodology will ensure consistency in screening assessment activities. A human health risk-based screening assessment may be the only assessment of potential risk conducted for a site, but could lead to a baseline risk assessment if results warrant it. This guidance is applicable to evaluating individual potential release sites (PRSs), consolidated PRSs, aggregated PRSs, watershed aggregates, or watersheds. In this document, the term "site" refers to all these areas.

2.0 SCREENING ASSESSMENT APPROACH

The ER Project's human health risk-based screening assessment approach follows the process described in the New Mexico Environment Department's (NMED's) Risk-Based Decision Tree (NMED 1998, 57761). It also follows guidance presented in NMED's technical background document for development of soil screening levels and the US Environmental Protection Agency (EPA) Region 6 human health medium-specific screening levels (NMED 2000, 68554; EPA 2000, 68410).

A fundamental requirement is that a screening assessment cannot be performed until the nature and extent of contamination at a site has been determined. Initially, data are assessed to determine whether the existing data and information are sufficient in quantity and quality to conduct a human health screening assessment (LANL 1998, 62060, Section 3.2.1). Subsequently, inorganic and radionuclide data are compared to background or fallout concentrations based on ER Project Standard Operating Procedures (SOPs) 15.12, Rev. 0, Performing Background Value Comparisons for Inorganic Chemicals, and 15.13, Rev. 0, Performing Background Value Comparisons for Radionuclides (LANL 1998, 59730). Organic chemicals are evaluated based on detection status because background data do not exist. These evaluations identify chemicals that are related to a release from a site and that may pose a potential risk to human health. These chemicals are referred to as "chemicals of potential concern" (COPCs). A human

health screening assessment for surface and near-surface soil and sediment is then conducted on the COPCs identified by the data review.

2.1 Screening Action Levels

The screening assessment process involves comparing chemical and radionuclide data to risk-based concentrations to determine if a potential for unacceptable risk to human health exists at the site (Section 2.2). The risk-based concentrations used in this comparison are referred to as screening action levels (SALs). A SAL is the concentration of a chemical or a radionuclide below which there is no unacceptable risk to human health as defined by a set of exposure assumptions. A site conceptual model is developed prior to conducting the screening assessment in order to identify likely contaminant source areas and mechanisms of transport, as well as to ensure that the exposure pathways, exposure assumptions, and potential receptors match those used in the calculation of the SAL. The site conceptual model represents links between the various aspects of the site in question and summarizes current understanding of the contamination. The information is used to determine the applicability of SALs at the site and whether an alternative approach is needed. For example, if additional, complex exposure pathways may be relevant in describing the site exposure scenario (e.g., meat-ingestion or drinking-water pathways), the site conceptual model does not coincide with the SAL assumptions and a screening assessment is not appropriate. In these instances, a site-specific baseline risk/dose assessment is conducted.

ER Project SALs have been developed for both chemicals and radionuclides based on a residential exposure. All screening evaluations are conducted only on this basis, i.e., there are no SALs representing other exposure scenarios. However, SALs are not necessarily protective of ecological endpoints and groundwater resources. SALs are intended for application to surface and near-surface soils and sediments and are generally applicable to a depth of approximately 10 to 12 ft, which accounts for possible construction-related excavation and/or residential activities. Contaminants occurring at depths greater than 10 to 12 ft may require evaluations other than or in addition to the SAL comparisons described below, e.g. transport to groundwater, and are at the discretion of the risk assessor.

Chemical SALs

Chemical SALs used in a screening assessment are based on chronic toxicity and not acute toxicity. A chemical SAL represents the concentration of a chemical that is associated with either a lifetime excess cancer risk of one in one million (10^{-6}) for carcinogens or a hazard quotient of one (1.0) for noncarcinogens (EPA 1989, 08021; EPA 1990, 55594). These standards are in agreement with EPA Region 6 human health medium-specific screening levels, but differ from NMED's technical background document for developing screening levels (EPA 2000, 68410; NMED 2000, 68554). The NMED guidance uses a 10^{-5} cancer-risk level for the soil screening level for each chemical carcinogen and an overall target level of 10^{-5} cancer risk. Regardless, the methodology for calculating chemical SALs uses current toxicity values, parameters, and equations.

Generally, if both noncarcinogenic and carcinogenic toxicity criteria have been established for a chemical, the criterion resulting in the lowest SAL is used in the screening assessment because if the lower value is not exceeded by the site concentration, neither toxicity-effect category is of concern. A screening assessment for chemical carcinogens uses the 10^{-6} ER Project SAL rather than the NMED soil screening level based on a 10^{-5} cancer risk. As a result, the NMED soil screening level for a chemical may be a noncarcinogenic value rather than a carcinogenic value (i.e., the value based on noncarcinogenic effect may be lower than that based on 10^{-5} cancer risk). For example, the ER Project SAL for chloroform, which is a carcinogen, is 0.24 mg/kg based on a 10^{-6} cancer risk, while the 10^{-5} level for chloroform is 2.4 mg/kg. The NMED soil screening level for chloroform is 0.38 mg/kg (NMED 2000, 68554) based on

noncarcinogenic effects. NMED uses the 0.38 mg/kg value as a soil screening level because it is lower than the 10^{-5} carcinogenic value of 2.4 mg/kg. Therefore, some COPCs are evaluated for both carcinogenic and noncarcinogenic effects using the ER Project SAL (e.g., for chloroform 0.24 mg/kg) and the NMED soil screening level (e.g., for chloroform 0.38 mg/kg), respectively to facilitate NMED's review of the screening assessment.

Radionuclide SALs

SALs for radionuclides are based on dose rather than on carcinogenic risk and are calculated using the residual radioactive materials (RESRAD) computer code developed by Argonne National Laboratory for the US Department of Energy (DOE) (at http://web.ead.anl.gov/resrad/). The RESRAD code has been continually revised and improved since it was issued in 1989. The RESRAD model and parameters used to calculate SALs are reviewed annually (at the start of each new fiscal year [October]) to determine whether the values need to be updated. Because RESRAD may be revised one or more times within a year, radionuclide SALs are revised only if the latest version of the RESRAD code affects the calculations. The ER Project starts using the revised radionuclide SALs (if any) at the beginning of a fiscal year; new values are not retroactive. The most recent manual describing the use of the RESRAD code was published in July 2001 (Yu et al. 2001, 71420).

The dose limit used as the basis for radionuclide SALs at a single site is 15 mrem/yr above background (EPA 1997, 58693; DOE-AL 2000, 67153). The dose limit is consistent with regulatory guidance and satisfies the "as low as reasonably achievable" (ALARA) principle to ensure that radiation dose is minimized (DOE-AL 2000, 67153). As a result, sites where radionuclide concentrations result in a dose greater than 15 mrem/yr (i.e., sites where radionuclide concentrations are greater than SAL or where exposure to multiple radionuclides results in an exceedance of the dose limit) are evaluated further. The derivation and use of SALs for radioactive COPCs is described by LANL (2001, 69683).

Standard residential default values similar to those used for chemical SAL calculations are incorporated into the RESRAD model with the doses summed over multiple pathways. Exposure pathways incorporated into the calculation of radionuclide SALs are the incidental ingestion of soil; inhalation of particulates, radon, and tritium; ingestion of homegrown fruits and vegetables; and external irradiation. Because certain radionuclides (particularly strontium-90) known to occur at some sites are susceptible to plant uptake, ingestion of homegrown produce (fruits and vegetables) is included among the potential pathways evaluated in calculating radionuclide SALs.

SALs and PRGs

Even though the numeric values may be the same, SALs and preliminary remediation goals (PRGs) generally are used for different purposes and the terms are not interchangeable. A SAL is used in the early stages of the RCRA corrective action process to identify chemicals that may present a threat to human health and, therefore, may warrant further investigation. SALs, unlike PRGs, are not cleanup values. PRGs or other cleanup criteria are used at later stages of the corrective action process to evaluate remedial alternatives and determine if remediation at a site has been successful. Although, the human health screening model for canyons has screening values (called PRGs) for several additional exposure scenarios and incorporates exposure pathways in addition to those used to calculate SALs for mesa-top sites (LANL 1998, 62049), the screening assessment approach is similar for mesa tops and canyons.

2.2 Screening Assessment

The screening assessment consists of four components: scoping, screening evaluation (comparison with SALs), uncertainty analysis, and interpretation. If a baseline risk assessment is conducted, the screening comparison with SALs may be incorporated into the risk assessment or the comparison with SALs is eliminated and all COPCs identified by the data review are evaluated in the baseline risk assessment.

2.2.1 Scoping

The scoping section of a screening assessment sets the stage for later components and follows a parallel evaluation for assessing potential ecological risk. The ecological scoping checklist can be a useful tool for human health screening, as well. The scoping section should include a discussion of both current and future land uses for a site as the key element that identifies appropriate receptors and exposure pathways. Potential land uses, transport and exposure pathways, and a review of the site history (focusing on release mechanisms, potential contaminants, and receiving media), are also important components of the scoping section.

In addition, the site conceptual model is compared to exposure pathways and assumptions in order to document that the SALs are applicable to evaluating potential human health risk at the site. The scoping section also should state whether the site is accessible and whether it is a secure area with limited and monitored access and egress; it should also be stated whether the area is expected to be used exclusively for Laboratory operations in the future, or there is a possibility of other land-use scenarios for the site (e.g., recreational or residential). These elements of the scoping section are critical to supporting the uncertainty analysis and interpretation discussions of a screening assessment (Sections 2.1.3, Uncertainty Analysis, and 2.1.4, Interpretation).

2.2.2 Screening Evaluation

The screening evaluation portion of a screening assessment compares either the maximum detected concentration or, if a sufficient number of samples have been collected (usually 10 or more samples), the 95% upper confidence limit (UCL) of the arithmetic mean concentration of each COPC with the appropriate SAL. Other approaches for estimating a representative concentration also may be used, including use of the mean concentration, the weighted average, and the median concentration; the value used must be representative of potential exposure and justified by the available data.

Comparisons to SALs for carcinogenic, noncarcinogenic, and radionuclide COPCs are performed separately:

- Carcinogens
  - All chemical carcinogens are compared with their respective SALs.

    Each ER Project SAL represents a 10^{-6} (one in one million) incremental excess cancer risk to an individual. The total incremental excess cancer risk (sum of cancer risk for all carcinogenic COPCs) is compared to a 10^{-5} cancer risk level (NMED's target risk level for carcinogens). In order to conclude that there is no potential unacceptable risk to human health the NMED target risk level of 10^{-5} should be met. However, EPA has indicated a target risk range for carcinogens of 10^{-4} to 10^{-6} (EPA 1990, 55594). If NMED's 10^{-5} target risk level is exceeded but is within the EPA target risk range, further evaluation may be warranted to determine whether potential unacceptable risk to human health exists at a site.

- Noncarcinogens
  - If the site has only one noncarcinogenic COPC, it is compared with the SAL.
  - If two or more COPCs are noncarcinogenic, each COPC is compared with 0.1 of its respective SAL.

    Each ER Project SAL is equivalent to a hazard quotient of 1.0, which represents the concentration at or below which no human health hazard is expected. The ratio of the exposure concentration for each noncarcinogenic COPC to its respective SAL or hazard quotient and the sum of individual ratios or hazard index for noncarcinogenic COPCs must be less than or equivalent to 1.0, in order to conclude that there is no potential unacceptable hazard to human health.

- Radionuclides
  - All radionuclides are compared with their respective SALs.

    Each ER Project SAL represents a dose of 15 mrem/yr. The total dose for all radionuclides detected above background/fallout levels must be less than or equivalent to 15 mrem/yr to safely conclude that there is no potential unacceptable dose.

These comparisons are based on guidance from NMED (NMED 2000, 68554), EPA Region 6 (EPA 2000, 68410), and DOE (DOE-AL 2000, 67153). If the above conditions are met, the site needs no further evaluation. As a result of the SAL comparison, each COPC falls into one of the following categories: greater than SAL, no SAL, or less than/equal to SAL.

"Greater than SAL" indicates that the representative concentration (e.g., the maximum detected concentration or the 95% UCL of the mean concentration) is greater than the SAL or 0.1 SAL, or the hazard index, total incremental cancer risk, or total dose are greater than the appropriate target level for a given chemical category. Any COPC in this category is evaluated further within the screening assessment or as part of a baseline risk assessment.

"No SAL" indicates no SAL currently is available for comparison. For many of these chemicals, it may be appropriate to use a SAL based on toxicological/molecular structural information for similar chemicals (i.e., a surrogate) or to calculate a SAL based on available toxicity information. The calculated SALs must be reviewed and approved by the ER Project Risk Assessment Team and submitted to the administrative authority for review. If SALs for surrogate chemicals are used in the screening evaluation, the rationale for their use must be presented in the screening evaluation. Commonly used surrogate chemicals are listed in an attachment to this document.

Samples are routinely analyzed for several inorganic chemicals (e.g., calcium, magnesium, potassium, and sodium) that are essential nutrients and for which SALs have not been developed due to a lack of sufficient low-dose toxicity information. These chemicals occasionally are detected at concentrations greater than background and therefore may be retained as COPCs. However, if the concentrations are only slightly elevated (e.g., less than twice the background value) above background levels, EPA's Risk Assessment Guidance for Superfund (RAGS), Part A, indicates that essential nutrients can be eliminated without further evaluation (EPA 1989, 08021). The potential importance of these chemicals as toxicants can be assessed based on human nutritional requirements. The evaluation of essential nutrients may be presented at several points within a report. However, the Resource Conservation and Recovery Act (RCRA) Facility Investigation (RFI) outline does not allow the elimination of COPCs other than by background comparison and detection status within the data review section. Therefore, for consistency

purposes the evaluation of nutrients should be presented in the screening evaluation section of a screening assessment.

"Less than SAL" indicates that the representative COPC concentration (e.g., the maximum detected concentration or the 95% UCL of the mean concentration) is less than the SAL or 0.1 SAL, or the hazard index, total incremental cancer risk, or total dose are less than the appropriate target level for a given category of chemicals. The COPCs in this category generally do not pose an unacceptable risk to human health and are not evaluated further.

Chemicals that are not detected in any sample generally are not included in a screening assessment when sample-specific detection limits do not exceed the contract-required detection limits (CDRLs) for EPA Contract Laboratory Program analytical methods or equivalent methods presented in the analytical statement of work (LANL 1995, 49738). Exceptions may be invoked if a detection limit is greater than background for inorganic chemicals or the detection limit is greater than SAL for organic chemicals and the inorganic or organic chemical is suspected to be present at a site. Factors to consider when determining if chemicals reported as "nondetects" are retained as COPCs in the screening assessment include

- site process knowledge that the chemical or related chemicals were used;
- the presence of related chemicals or chemicals of a degradation chain;
- the magnitude of the discrepancy between the detection limit and SAL (if the detection limit is close to the SAL, the chemical may be retained, but if the detection limit is several times less than the SAL, the chemical may be eliminated);
- the number of samples collected (analyzing only a few samples may not define the nature of contamination; therefore some chemicals may not be eliminated); and
- the applicability of the SAL assumptions at the site in question.

These factors also can be used to determine if a chemical whose sample-specific detection limits were elevated (e.g., due to matrix interference) is retained or eliminated as a COPC. When a chemical that has been reported as not detected is retained as a COPC, the maximum detection limit for that chemical is used in the screening assessment. The supporting rationale for eliminating or retaining a chemical with detection limits greater than a SAL is documented on a chemical-by-chemical basis.

2.2.3 Uncertainty Analysis

The uncertainty analysis presents a qualitative assessment of issues that potentially may affect the results of the screening evaluation and assists in interpretation of the results. The analysis includes issues related to data quality, receptors and exposure pathways, SAL development, exposure and toxicity assessments, and surrogate chemicals used in the SAL comparisons. The uncertainty analysis also evaluates COPCs retained by the screening evaluation to determine whether they are eliminated or carried forward to a risk assessment or corrective action, and details the rationale to eliminate any COPCs based on process knowledge (or other sound reasons). A qualitative discussion justifying the elimination or retention of COPCs that were not screened should also be included. The uncertainty analysis can result in the addition or deletion of COPCs.

Areas of uncertainty that may need to be discussed include but are not limited to

- the definition of the physical setting (likelihood that the receptor and pathways are appropriate);

- data quality issues, such as lab qualifiers indicating high/low bias or use of professional judgment;
- analytical methods used;
- environmental transport models;
- chemicals that have detection limits that exceed their respective SALs;
- toxicity values used (e.g., use of any surrogate chemicals in calculation of SALs);
- intake/exposure parameters and their assumed values;
- multiple pathway exposure assumptions; and
- differences in the toxic endpoints (target organs) among noncarcinogens.

2.2.4 Interpretation

The interpretation portion of the screening assessment summarizes the results of the human health screening assessment with respect to the uncertainty analysis. The uncertainty analysis permits the formulation of conclusions regarding the potential risk to human receptors and the confidence level in the assessment results. The interpretation portion also identifies COPCs that are retained by the screening evaluation and evaluates the subsequent steps for further evaluation. For example, a COPC above its SAL may indicate that subsequent action at the site is warranted (e.g., remediation) or it may represent that no potential unacceptable risk to human health exists under actual site conditions. If the latter is true, an analysis of the data that shows no unacceptable risk may be presented. If a risk assessment is not conducted, a comparison to values representing a more realistic exposure scenario, e.g., a site worker or recreational user, may be appropriate. The results of the comparison are then used to justify no further action if concentrations indicate no potential unacceptable risk.

2.3 Evaluation of Chemicals Not Related to a Process Release

Certain classes of chemicals, e.g., polycyclic aromatic hydrocarbons (PAHs) and pesticides, which have been detected in soil, often are not related to a release from the PRS and therefore are not necessarily the responsibility of the ER Project. Non-process related sources in the form of runoff from asphalt-paved areas and rooftops and/or incomplete combustion associated with forest fires, vehicle exhaust, and similar events can result in PAH deposition in the soil. Pesticides may be present at a site as a result of herbicides/insecticides applied during routine facility maintenance (past and/or present). In other cases, residues from maintenance activities, such as paint chips from bridges and buildings, may be present in the soil and result in elevated levels of inorganic chemicals. In some cases, the presence of non-process related chemicals (inorganic and organic) may affect ER Project site decisions and investigators may choose to treat these chemicals separately.

If treated separately, non-process related chemicals should be discussed in Section 2.3.5, Revised Site Conceptual Model, of the risk assessment. The conceptual model should describe where the chemicals occur, how they are distributed over the site, and what potential sources may be responsible for their presence. Information on site history, operational processes, and the possible use of appropriate local data sets (e.g., samples collected upgradient or upstream from Laboratory activities) are presented to support the rationale for identifying the chemicals that result from non-process related activities/sources. If the potential risk from non-process related chemicals is evaluated using a comparison with SALs, the assessment as well as the rationale for their elimination should be presented in Section 2.4.2.1(c).

Uncertainty Analysis, of the screening assessment; this comparison should be separate from the screening evaluation of process-related COPCs. Estimated potential risk may be based either on the maximum detected concentration or a representative concentration that is appropriate for the area over which an individual might be exposed (e.g., arithmetic mean or 95% UCL of the mean). Alternatively, a risk assessment that presents both total potential risk (process related and non-process related) and potential incremental, process-related risk may be conducted.

3.0 DERIVATION OF SALS FOR NONRADIOLOGICAL CHEMICALS

The approach described is intended to ensure consistency in the calculation of SALs and in their application for evaluating specific media. Independent calculation of SALs that incorporates updated toxicity values in real time rather than on periodic updates from outside sources is also acceptable. However, any deviations from the procedure described in this document may be made only with approval from the ER Project Risk Assessment and peer review team leader and the administrative authority.

The equations and parameter values presented in this methodology are consistent with those of NMED and EPA Region 6 in the derivation of their soil screening levels and medium-specific human health screening levels, respectively (NMED 2000, 68554;EPA 2000, 68410). Lead is the exception to the SAL calculations described in this document because EPA-approved toxicity values for lead have not been published. The SAL for lead is 400 mg/kg, which represents the soil lead level that limits exposure of a child to no more than a 5% chance of exceeding a 10 μg/dL blood lead level (EPA 1994, 59509). The SAL was calculated using EPA's Integrated Exposure Uptake Biokinetic (IEUBK) model (EPA 1994, 59894). In addition, because certain inorganic and organic chemicals have relatively low toxicity, a non-risk based maximum concentration of 10^5 (100,000) mg/kg is used when the calculated risk-based SAL exceeds this concentration. COPCs that have SALs of this nature include benzoic acid, chromium III, and dimethyl phthalate.

The ER Project's primary resource for calculating SALs is NMED soil screening level guidance. If chemical information is not available from NMED guidance, the project follows EPA Region 6 guidance. While the differences between the guidance documents are small, there is a modification in the SAL calculations depending upon the information available. Eventually, as the NMED guidance becomes refined and expanded to include all relevant chemicals, it will become the ER Project's sole resource for calculating SALs.

The ER Project has adopted the approaches of NMED and EPA Region 6 for three reasons: (1) as a quality assessment tool to check values that the ER Project calculates; (2) to allow other risk assessors, who are familiar with the NMED and Region 6 approaches, a greater understanding of the ER Project's guidelines and procedures; and (3) to provide continuity within the ER Project for calculating SALs.

3.1 Toxicity Values

Toxicity values (reference doses [RfD] and cancer slope factors) associated with chronic exposure are preferentially used for calculating SALs. The preferred source of toxicity values is EPA's Integrated Risk Information System (IRIS) located on the World Wide Web site at http://www.epa.gov/iris. EPA's Health Effects Assessment Summary Tables (HEAST) are used as a source of toxicity values if values are not published in IRIS (EPA 1997, 58968). Finally, provisional toxicity values may be obtained for some chemicals and routes of exposure from EPA's National Center for Environmental Assessment (NCEA). NCEA publishes issue papers on certain chemicals' toxicity values for use by EPA regional offices. However, the NCEA provisional values have not been subjected to rigorous scientific review and therefore cannot be used with the confidence of values obtained from IRIS or HEAST. The NCEA values

are used in calculating SAL values for performing screening assessments because (1) they reflect the state of knowledge within NCEA at the time of their publication and therefore incorporate a level of review beyond peer-reviewed publications and (2) incorporate several upper-bound exposure estimates and conservatively biased submodels for dermal absorption and dust resuspension. Therefore, the uncertainty associated with the provisional toxicity value is balanced by a bias toward protecting the public. If provisional values are used in SAL calculations for chemicals that are potential risk-drivers, consequences to the confidence of the screening decision are discussed in the uncertainty analysis of the screening assessment.

EPA specifies separate toxicity values for ingestion and inhalation intake routes. Toxicity values for metals are not extrapolated between ingestion and inhalation exposure routes, when a value has been published for one route only, due to the potential differences in absorption efficiencies between these intake routes. Because absorption of organic chemicals more closely approximates 100% for both ingestion and inhalation, route-to-route extrapolation is performed for organic chemicals. If extrapolated toxicity values (between ingestion and inhalation exposure routes) are used in SAL calculations for chemicals that are potential risk-drivers, effects on the confidence of the screening decision are discussed in the report.

EPA toxicity values for chemical ingestion also are used to evaluate risks associated with dermal absorption. Depending on the chemical and on the method of administration in the studies from which oral toxicity values are derived, oral toxicity values may reflect varying absorption efficiencies from the gastrointestinal (GI) tract into the bloodstream. Because specific information on absorption efficiency is rarely available, oral toxicity values used to evaluate risks associated with dermal absorption are not adjusted to account for the chemical-specific oral absorption fraction associated with the oral toxicity value. Using oral toxicity values without adjustment for GI absorption efficiency may result in an under- or overestimate of intake via the dermal pathway. However, this uncertainty is balanced by the conservative assumptions incorporated into the dermal absorption model, as discussed in Section 3.3.3, Modeling Inhalation and Dermal Pathways, in this document.

The relative bioavailability of a chemical in the GI tract is not incorporated into the SAL calculations for the soil ingestion pathway. In general, chemical bioavailability in soil is lower than in water due to the time required for a chemical to desorb from a soil particle and/or diffuse from within pores in the soil particle. Desorption rates are also generally longer for soil contamination than for "spiked" food used in laboratory toxicity studies because desorption rates are correlated with the length of time that a chemical has been sorbed. Some fraction of a chemical adsorbed onto soil may desorb at so slow a rate as to be effectively unavailable during the transit time through the GI tract. Factors such as chemical form, soil-particle diameter, geochemical factors, and individual nutritional status affect the degree of bioavailability from soil. The assumption of equivalent bioavailability from soil and from the administration vehicle used in the toxicity studies on which many toxicity values are based (e.g., food or water) results in an overestimate of uptake from soil ingestion.

Some chemicals that are routinely analyzed for and detected do not have EPA-approved or provisional toxicity values. The approach to this issue is to identify a similar chemical for which toxicity values are available and incorporate it in the screening assessment as a surrogate. Similarity may be based on chemical structure or that it is a degradation product of the chemical. Identification of an appropriate surrogate value, and whether the evaluation is performed within the context of a screening assessment or a risk assessment, is a chemical-specific and assessment-specific decision that is beyond the scope and purpose of this document. Surrogates and the assumptions affecting their choice and use are documented in each report, where applicable (see the attachment to this document).

3.2 Exposure Scenarios, Media, and Pathways

The SALs represent a residential exposure that is generally more protective than other exposure scenarios, e.g., industrial or recreational. Therefore, sites screened and released on the basis of a residential land use are also safe for recreational and commercial/industrial activities. The restrictions of the SALs with respect to media and depth were mentioned in Section 2.0. Exposure pathways for chemical SALs include incidental ingestion of soil, inhalation of particulates and vapors in ambient air, and dermal absorption of chemicals in soil. If other pathways are thought to be important for a given site/receptor, chemical SALs might not be applicable and further analysis is warranted in a baseline risk assessment. The pathway for ingesting homegrown vegetables and fruit is not typically included in the assessment of chemical carcinogens and noncarcinogens and is not included in the NMED methodology or in any EPA regional soil screening methodologies.

Exposure equations and parameter values for SAL calculations are provided in this document (Section 3.4, Screening Action Level Equations and Parameter Values). The primary sources of exposure parameters used in SAL calculations are EPA's standard default exposure factors (EPA 1991, 59893) and EPA's exposure factors handbook, Volumes 1, 2, and 3 (EPA 1997, 66596, 66597, 66598). Parameter values are intended to provide estimates of the "reasonable maximum exposure" for a resident incorporating the pathways listed above. Many of these exposure parameters describe the intensity, frequency, and duration of exposure. For the dermal absorption exposure route, parameter values for exposed body surface area, soil adherence factor, and skin absorption factors were obtained from EPA Region 6 (EPA 2000, 68410) and on the World Wide Web at http://www.epa.gov/earth1r6/6pd/rcra_c/pd-n/screen.htm. EPA Region 6 obtained these values from a draft version of EPA's dermal risk assessment guidance. Although the guidance is still in interim draft form and has not been released for general use, these parameter values reflect current EPA guidance for evaluating the dermal exposure route based on EPA review of relevant published research and are unlikely to change.

3.3 Modeling Inhalation and Dermal Pathways

3.3.1 Inhalation – Volatile Organic Compounds

The concentration of volatile organic chemical vapors in the ambient air breathing zone associated with volatile organic chemicals in site soils is calculated using a steady-state volatilization model. The model used is Hwang and Falco's volatilization factor (VF) model, originally described by EPA (1991, 58234).

The version of the VF model that is used for calculating SALs is presented in the user's guide and technical background document of EPA's soil screening guidance documents (EPA 1996, 58917; EPA 1996, 59902). The primary difference between versions of the VF model is that the output of a separate air dispersion model (based on 1 yr of meteorological data) has replaced the earlier box model component. From a table of dispersion model output ordered by area and regional location, users select a value most applicable to the site under consideration for use in their assessment.

The VF model assumes an effectively infinite depth of contaminated soil and no cover of clean soil. The first assumption may contribute to overestimates of risk for sites with a relatively small volume of contamination because calculated volatile organic chemical emissions over a chronic exposure period of many years can easily violate conservation of mass. The assumption of no cover potentially results in an underestimation of the diffusion path length, which consequently increases the estimate of flux to the atmosphere. However, the ambient air VF model is used to screen sites for residential land use and situations in which a building may be constructed over the affected soils. Indoor air volatile organic chemical concentrations at a site may be considerably higher than in ambient air. Thus, the conservative

biases associated with applying the VF model to ambient air impacts are balanced by the model's potential application to sites where indoor air impacts may be of concern.

The VF model is valid for sites where a volatile organic chemical is present at concentrations below soil particle, pore water, and pore air saturation phases. For conditions in which soil is saturated with one or more organic chemicals, a SAL calculated using the VF model output is not reliable. The screening value for a volatile organic chemical under these conditions is the soil saturation concentration (Csat). This value is used to identify the possible presence of nonaqueous phase volatile organic chemical liquid (which may result in greater likelihood of off-site migration) and is not associated with toxicological endpoints.

Because VF model output is not reliable above the Csat of a volatile organic chemical, the ER Project recommends using the Csat value for initially screening volatile organic chemicals in saturated soil conditions and when the calculated value from the VF model exceeds Csat. The possibility of enhanced migration in the environment is assessed when site concentrations of a volatile organic chemical exceed the Csat value and the volatile organic chemical exists as a nonaqueous phase liquid. All volatile organic chemicals with site concentrations greater than Csat are subsequently evaluated for potential risk by comparing concentrations to a SAL calculated using only the soil ingestion and dermal absorption pathways. The elimination of the inhalation pathway from this SAL calculation, and the use of Csat to identify volatile organic chemicals for which migration is an enhanced concern, is consistent with EPA soil screening guidance, EPA Region 6 soil screening guidelines and NMED guidance (EPA 1996, 58917; EPA 2000, 68410; NMED 2000, 68554). Supporting documentation for eliminating the inhalation pathway under saturated soil conditions is provided in EPA's soil screening guidance document (EPA 1996, 59902).

VF and Csat model equations, and parameter values for SAL calculations, are documented below. Parameter values for site-related factors such as soil porosities, density, and amount of organic carbon are default values recommended in EPA's soil screening guidance (EPA 1996, 58917). Chemical-specific parameter values are required for chemical diffusivity in air and water, Henry's Law constant, solubility in water, and organic carbon partition coefficient. The sources that are used to obtain these values, in order of prioritization, are (1) EPA's soil screening guidance (EPA 1996, 58917; EPA 1996, 59902) and (2) EPA's Superfund chemical data matrix (EPA 1996, 64708). Other references that may be employed if data are unavailable in the primary references include the "Handbook of Environmental Fate and Exposure Data for Organic Chemicals" (Howard 1990, 59892), EPA's subsurface contamination reference guide (EPA 1990, 59926), and EPA's Superfund exposure assessment manual (EPA 1988, 59901).

3.3.2 Inhalation–Fugitive Dust

The concentration of dust in air above contaminated soils and sediment is calculated using a screening-level soil resuspension model, EPA's particulate emission factor (PEF) model. This model originally was described in "Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites" (EPA 1985, 59903). The version of the PEF model that is used for calculating ER SALs is presented in the user's guide and technical background document of EPA's soil screening guidance documents (EPA 1996, 58917; EPA 1996, 59902). The primary difference between the two versions of the PEF model is that the output of a separate air dispersion model (based on 1 yr of meteorological data) has replaced the earlier box model component. From a table of dispersion model output ordered by area and regional location, users select a value most applicable to the site under consideration for their assessment.

The PEF model used for screening the dust inhalation pathway is based on wind erosion of surfaces that have an unlimited reservoir of particles. The model calculates the concentration of respirable particles in

the air due to wind erosion. Depending on site soil conditions, there may not be an unlimited supply of particles of this size available throughout the exposure period, which may overestimate the intake via dust inhalation. A limitation of the model is that it does not address particulate resuspension caused by mechanical forces. Therefore, fugitive dust concentrations calculated using this model are not applicable for activities such as construction. If a construction scenario is used during site assessment, an alternative approach for estimating airborne dust concentrations is selected in coordination with the administrative authority.

The PEF model equations and parameter values for SAL calculations are documented in Section 3.4, Screening Action Level Equations and Parameter Values, of this document. Parameter values for the PEF model, including the dispersion term (Q/C), vegetative cover, and wind speeds are default values recommended in EPA's soil screening guidance (EPA 1996, 58917). No chemical-specific parameter values are required in the PEF model.

3.3.3 Dermal Absorption

The amount of soil residing on a unit area of skin is described using an adherence factor (AF). A layer of soil is assumed to cover 100% of a specified body surface area corresponding to the AF. The literature on AFs recognizes that they are dependent upon body part, soil type, particle size, soil moisture content, and other variables. Information for quantifying these variables often is unavailable single default values are used for the AFs when SALs are calculated (0.2 mg/cm^2 for a child and 0.07 mg/cm^2 for an adult) (EPA 2000, 68410).

Dermal absorption from soil is evaluated using a skin absorption factor (ABS) to model desorption of a chemical from soil, absorption through skin, and transfer to the bloodstream. According to EPA Region 6 soil screening guidelines, a default ABS value for semivolatile organic chemicals is 0.1 (EPA 2000, 68410). NMED guidance also uses a default ABS value for organic chemicals of 0.1 and a default ABS of 0.01 for inorganic chemicals (NMED 2000, 68554). Chemical-specific ABS values obtained from EPA Region 6 guidance (EPA 2000, 68410) are used in SAL calculations for the following chemicals: arsenic (0.03); cadmium (0.01); chlordane (0.04); 2,4-dichlorophenoxyacetic acid (2,4-D) (0.05); p,p'-dichlorodiphenyltrichloroethane/ p,p'-dichlorodiphenyldichloroethylene/p,p'-dichlorodiphenyldichloroethane (DDT/DDD/DDE) (0.03); hexachlorocyclohexane (lindane) (0.04); 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) (0.03); PAHs (0.13); PCBs (0.14); and pentachlorophenol (0.25).

The approach used to model dermal absorption incorporates several conservative assumptions that may result in an overestimation of actual absorption. The ABS value reflects an assumption that absorption is independent of concentration and does not change with time. A chemical is assumed to be 100% available for absorption from adhered soil. For example, no loss of volatile or semivolatile chemicals is assumed to occur due to volatilization when soil is present on the skin.

The dermal absorption pathway also includes assumptions that may result in an over- or underestimation of absorption. As described in Section 3.1, oral toxicity values are not corrected for GI absorption efficiency resulting in the potential for over- or underestimating risk through the dermal pathway. An additional assumption is that skin is presumed to be intact; abrasions or cuts on the skin surface that could result in greater absorption on an individual basis are not considered. These two assumptions that may underestimate dermal risk are balanced by the conservative assumptions already discussed in this section and do not compromise the protective quality of SALs.

3.4 SAL Equations and Parameter Values

Equations 1 and 2 are used to calculate SALs for noncarcinogenic and carcinogenic chemicals, respectively, via direct soil ingestion, inhalation of chemical vapors or airborne dust, and dermal absorption from soil. Table 1 gives parameters for these equations.

Combined Exposures for Noncarcinogenic Chemicals in Soil

$$C = \frac{THQ \times BW_c \times AT_n}{EF \times ED_c \times \left[\left(\frac{1}{RfD_o} \times \frac{IRS_c}{10^6\,mg/kg}\right) + \left(\frac{1}{RfD_o} \times \frac{SA_c \times AF_c \times ABS}{10^6\,mg/kg}\right) + \left(\frac{1}{RfD_i} \times \frac{IRA_c}{(VF_s \text{ or } PEF)}\right)\right]} \quad (1)$$

Note: Use VF_s for volatile chemicals and PEF for nonvolatile chemicals.

Combined Exposures for Carcinogenic Chemicals in Soil

$$C = \frac{TR \times AT_c}{EF \times \left[\left(\frac{IFS_{adj} \times CSF_o}{10^6\,mg/kg}\right) + \left(\frac{SFS_{adj} \times ABS \times CSF_o}{10^6\,mg/kg}\right) + \left(\frac{InhF_{adj} \times CSF_i}{(VF_s \text{ or } PEF)}\right)\right]} \quad (2)$$

Note: Use VF_s for volatile chemicals and PEF for nonvolatile chemicals.

Because contact rates may be different for children and adults, carcinogenic risks during the first 30 yr of life are calculated using age-adjusted factors ("adj"). Use of age-adjusted factors is especially important for soil ingestion exposures, which are higher during childhood and decrease with age. However, for purposes of combining exposures across pathways, additional age-adjusted factors are used for inhalation and dermal exposures.

Equation 3 is used (mg·yr)/(kg-day), for ingestion:

$$IFS_{adj} = \frac{ED_c \times IRS_c}{BW_c} + \frac{(ED_t - ED_c) \times IRS_a}{BW_a} \quad (3)$$

Equation 4 is used (mg·yr)/(kg-day), for dermal contact:

$$SFS_{adj} = \frac{ED_c \times AF_c \times SA_c}{BW_c} + \frac{(ED_t - ED_c) \times AF_a \times SA_a)}{BW_a} \quad (4)$$

Equation 5 is used (m^3-yr)/(kg·day), for inhalation:

$$InhF_{adj} = \frac{ED_c \times IRA_c}{BW_c} + \frac{(ED_t - ED_c) \times IRA_a)}{BW_a} \quad (5)$$

Table 1
Parameters Used in the SAL Equations

C	=	chemical SAL in soil (mg/kg)	
THQ	=	target hazard quotient	1
TR	=	target cancer risk	10^{-6}
AT_c	=	averaging time (carcinogen)	70 yr x 365 days
AT_n	=	averaging time (noncarcinogen)	exposure duration (ED) x 365 days
ABS	=	skin absorption factor	organic = 0.1
			inorganic = 0.01[a]
			Chemical-specific values (Section 3.3.3 of this document)
AF_c	=	adherence factor–child	0.2 mg/cm^2
BW_c	=	body weight–child	15 kg
CSF_o	=	cancer slope factor–oral	(mg/kg-day)$^{-1}$ (see Section 3.1 of this document)
CSF_i	=	cancer slope factor–inhalation	(mg/kg-day)$^{-1}$ (see Section 3.1 of this document)
EF	=	exposure frequency	350 day/yr
ED_c	=	exposure duration–child	6 yr
IFS_{adj}	=	age-adjusted ingestion factor	114 mg-yr/kg-day
$InhF_{adj}$	=	age-adjusted inhalation factor	11 m^3-yr/kg-day
IRA_c	=	inhalation rate–child	10 m^3/day
IRS_c	=	soil ingestion rate–child	200 mg/day
PEF	=	particulate emission factor	(m^3/kg) (see below)
RfD_o	=	reference dose–oral	(mg/kg-day) (see Section 3.1 of this document)
RfD_i	=	reference dose–inhalation	(mg/kg-day) (see Section 3.1 of this document)
SA_c	=	exposed surface area–child	2800 cm^2/day
SFS_{adj}	=	age-adjusted skin contact factor for carcinogens	361 mg-yr/kg-day
VF_s	=	volatilization factor for soil	(m^3/kg) (see below)
BW_a	=	body weight–adult	70 kg
ED_r	=	exposure duration–residential	30 yr[b]
AF_a	=	adherence factor–adult	0.07 mg/cm^2
IRS_a	=	soil ingestion rate–adult	100 mg/day
SA_a	=	exposed surface area–adult	5700 cm^2/day
IRA_a	=	inhalation rate–adult	20 m^3/day

[a]Absorption values for inorganic chemicals not used if the SAL is calculated using EPA Region 6 guidance.

[b]Exposure duration for lifetime resident is 30 years. For carcinogens, the exposures are combined for child (6 yr) and adult (24 yr).

Derivation of the Volatilization Factor

Equation 6 is used to derive the VF for volatile organic chemicals used to calculate SALs; parameters are presented in Table 2.

$$VF_s = \left(\frac{Q}{C}\right) \times \frac{(3.14 \times D_A \times T)^{1/2}}{2 \times \rho_b \times D_A} \times 10^{-4} \left(m^2/cm^2\right) \quad (6)$$

where:

$$D_A = \frac{\left(\Theta_a^{10/3} D_i H' + \Theta_w^{10/3} D_w\right)/n^2}{\rho_b K_d + \Theta_w + \Theta_a H'}$$

Table 2
Parameters Used in Derivation of Volatilization Factor

VF_s	=	volatilization factor	(m^3/kg)
D_A	=	apparent diffusivity	(cm^2/sec)
Q/C	=	inverse of mean concentration at the center of a 0.5-ac^2 source	68.18 g/m^2-sec per kg/m^{3a}
T	=	exposure interval	9.5×10^8 sec
ρ_b	=	dry soil bulk density	1.5 g/cm^3
Θ_a	=	air filled soil porosity (L_{air}/L_{soil})	0.18[b] or $n - \Theta_w$
Θ_w	=	water filled soil porosity (L_{water}/L_{soil})	0.26[c]
D_i	=	diffusivity in air (cm^2/sec)	chemical-specific
H'	=	dimensionless Henry's Law constant	chemical-specific
D_w	=	diffusivity in water (cm^2/sec)	chemical-specific
n	=	total soil porosity (L_{pore}/L_{soil})	0.43 or $1 - (\rho_b/\rho_s)$
ρ_s	=	soil particle density	2.65 g/cm^3
K_d	=	soil-water partition coefficient (cm^3/g)	$K_{oc} f_{oc}$ (chemical-specific)
K_{oc}	=	soil organic carbon/water partition coefficient (L/kg)	chemical-specific
f_{oc}	=	fraction organic carbon content of soil	0.0015[d] (g/g)

Note: $H' = \dfrac{\text{Henry's Law Constant}}{\text{universal gas constant} \times \text{temperature}}$

[a] The Q/C value used in the derivation of the volatilization factor is 68.81 g/m^2-sec per kg/m^3 if EPA Region 6 guidance is used.

[b] The air filled soil porosity (Θ_a) value used in the derivation of the volatilization factor is 0.28 if EPA Region 6 guidance is used.

[c] The water filled soil porosity (Θ_w) value used in the derivation of the volatilization factor is 0.26 if EPA Region 6 guidance is used.

[d] The fraction organic carbon content of soil (f_{oc}) value used in the derivation of the volatilization factor is 0.006 (g/g) if the Region 6 guidance is used.

Derivation of the Soil Saturation Concentration

Equation 7 is used to derive the soil saturation concentration (Csat) for organic chemicals; parameters are presented in Table 3.

$$sat = \frac{S}{\rho_b}\left(K_d \rho_b + \Theta_w + H' \Theta_a\right) \quad (7)$$

Table 3
Parameters used in the Derivation of the Soil Saturation Concentration (C_{sat})

sat	=	soil saturation concentration	(mg/kg)
S	=	solubility in water (mg/L)	chemical-specific
K_d	=	soil-water partition coefficient (cm^3/g)	$K_{oc}f_{oc}$ (chemical-specific)
K_{oc}	=	soil organic carbon/water partition coefficient (L/kg)	chemical-specific
f_{oc}	=	fraction organic carbon content of soil	0.0015[a] (g/g)
ρ_b	=	dry soil bulk density	1.5 g/cm^3
Θ_w	=	water filled soil porosity (L_{water}/L_{soil})	0.26[b]
H'	=	dimensionless Henry's Law constant	chemical-specific
Θ_a	=	air filled soil porosity (L_{air}/L_{soil})	0.18[c] or $n - \Theta_w$

Note: $H' = \dfrac{\text{Henry's Law Constant}}{\text{Universal gas constant} \times \text{temperature}}$

[a] The fraction organic carbon content of soil (f_{oc}) value used in the derivation of the volatilization factor is 0.006 (g/g) if EPA Region 6 guidance is used.

[b] The water filled soil porosity (Θ_w) value used in the derivation of the volatilization factor is 0.26 if EPA Region 6 guidance is used.

[c] The air filled soil porosity (Θ_a) value used in the derivation of the volatilization factor is 0.28 if EPA Region 6 guidance is used.

See Section 3.3.1 for guidance on the application of the soil saturation concentration for development of a SAL for volatile organic chemicals.

Derivation of the Particulate Emission Factor

Equation 8 is used to derive the particulate emission factor (PEF) for non-volatile organic chemicals and inorganic chemicals for use in the calculation of SALs; parameters are presented in Table 4.

$$PEF(m^3/kg) = \frac{Q}{C} \times \frac{3{,}600\ sec/h}{0.036 \times (1 - V) \times (U_m/U_t)^3 \times F(x)} \qquad (8)$$

Table 4
Parameters used to Derive the Particulate Emission Factor

PEF	=	particulate emission factor	1.36×10^9 m^3/kg[a]
Q/C	=	inverse of the mean concentration at the center of a 0.5-ac^2 source	93.77 g/m^2-sec per kg/m^3[b]
V	=	fraction of vegetative cover	0.5 (unitless)
U_m	=	mean annual wind speed	4.69 m/sec
U_t	=	equivalent threshold value of wind speed at 7 m	11.32 m/sec
F(x)	=	function dependent on U_m/U_t (derived using EPA 1985, 59903)	0.194 (unitless)

[a] A PEF value of 1.316×10^9 is calculated using the default parameter values in EPA Region 6 guidance.

[b] The Q/C value used in the derivation of the particulate emission factor is 90.8 g/m^2-sec per kg/m^3 if EPA Region 6 guidance is used.

4.0 DERIVATION OF SALS FOR RADIOLOGICAL CHEMICALS

This section summarizes the approach used to calculate ER Project SALs for radionuclides in soils. The ER Project document, "Derivation and Use of Radionuclide Screening Action Levels," provides much more detail on the derivation of the radionuclide SALs, including the RESRAD run with all input parameters and a table of SALs (LANL 2001, 69683).

RESRAD incorporates the dose assessment methodology described in DOE Order 5400.5,"Radiation Protection of the Public and the Environment" and is used to implement residual radioactive material guidelines contained in this DOE Order (DOE 1990, 58980). As stated in a DOE Albuquerque Operations Office (DOE-AL) memorandum regarding the release of real property with residual radioactive materials, "DOE requires the use of RESRAD computer models for evaluating potential doses resulting from the presence of residual contamination" (DOE-AL 2000, 67153).

4.1 Target Dose Limit

DOE-AL has published guidance addressing the selection of annual dose limits for dose assessments under its jurisdiction: "DOE-AL staff will approve proposed releases of real property where the modeled dose is up to and including 15 mrem/yr. Proposed releases resulting in doses greater than 15 mrem/yr require concurrence with DOE-HQ" (DOE-AL 2000, 67153). A 15-mrem/yr target dose limit is consistent with published EPA guidance (EPA 1997, 58693) and is well below the basic dose limit of 100 mrem/yr above background established in DOE Order 5400.5 (DOE 1990, 58980). The EPA has determined that a target dose limit of 15 mrem/yr equates to an approximate increased lifetime cancer risk of 10^{-4}, and "is consistent with levels generally considered protective in other governmental actions, particularly regulations and guidance developed by EPA in other radiation control programs" (EPA 1997, 58693). Therefore, the ER Project uses a target dose limit of 15 mrem/yr to calculate radionuclide SALs.

4.2 Exposure Scenario, Media, and Pathways

As with the chemical SALs, the radionuclide SALs represent a residential exposure that is protective of other land-use options. Radionuclide SALs are designed to predict the maximal on-site soil dose over time and do not account for radionuclide migration with surface water and/or infiltration to groundwater. Radionuclide migration with water may be of equal or greater concern as soil dose at some canyon-bottom sites and at mesa-top sites where potentially contaminated water was discharged. Therefore, at these types of sites in particular, radionuclide SALs may need to be supplemented with an evaluation of migration potential.

The exposure pathways used to calculate radionuclide SALs are incidental soil ingestion; inhalation of particulates, radon, and tritium; external irradiation; and ingestion of homegrown produce. Unique exposure pathways for a gaseous form of tritium (as water vapor) include ambient inhalation of tritiated water vapor and absorption of tritiated water vapor through the skin. Ingestion of homegrown produce is a pathway that is commonly evaluated in radionuclide dose assessments but generally is excluded in screening-level chemical risk assessment calculations (e.g., plant ingestion is not included in the soil-screening criteria published by NMED and EPA Regions 3, 6, and 9). In keeping with common practice and because certain radionuclides (particularly strontium-90) known to occur at some sites are susceptible to plant uptake, ingestion of homegrown produce is included among the pathways the ER Project uses to calculate radionuclide SALs.

The primary sources of exposure parameters used in radionuclide SAL calculations are EPA's standard default exposure factors and EPA's Exposure Factors Handbook (EPA 1991, 56140; EPA 1997, 66596,

66597, 66598). The parameter values are intended to provide an estimate of the "reasonable maximum exposure" for a residential exposure scenario incorporating the pathways listed above (EPA 1991, 56140).

4.3 Key Input Parameters and Assumptions

To determine if radionuclide SALs are applicable to a particular site it is necessary to determine if the assumptions underlying their calculation are consistent with the site conceptual model. The site conceptual model includes what is known or assumed regarding the spatial distribution of radionuclides in soil, the potential for radionuclide migration over time, and characteristics of the applicable receptor population. If potentially important site-specific transport or exposure pathways are not included in the derivation of the radionuclide SALs, a comparison of site data to radionuclide SALs either may be inappropriate or require site-specific calculation of screening values that are representative of the site. If screening is determined to be inappropriate for a site, a site-specific dose assessment is conducted.

Radionuclide migration from a contaminated site may occur by dissolution in surface or groundwater, erosion of soil, resuspension as airborne particulate, biotic uptake, and/or volatilization as a vapor. Radionuclide migration to an off-site location is not incorporated in the calculation of radionuclide SALs nor does RESRAD directly support such evaluations. Although potential off-site radionuclide concentrations are generally lower than those observed on the site, modeling remote concentrations over time may be important because of (1) public concerns, (2) different routes and/or intensities of exposure than are considered for on-site receptors, and (3) regulatory requirements to assess off-site impacts. An example of a potentially substantial off-site impact is uptake of strontium-90 by plants rooted in contaminated soil and the subsequent redistribution in leaf litter or in fine particulates if the plant or plant material burns. Appendix K of Yu et al. (2001, 71420) provides guidance for evaluating off-site migration using RESRAD output.

Certain parameters are critical to defining site conditions and how these conditions may change over time. These parameters generally are subject to great site-specific variability, such as the size, thickness, and soil erosion rate of the contaminated zone; the evapotranspiration coefficient at the site; and the site-related soil particulate concentration in air. In addition, the length of the modeling period can influence radionuclide SALs for certain radionuclides associated with ingrowth of radioactive progeny. These parameter values have been defined so that the calculation of radionuclide SALs effectively results in static soil conditions (LANL 2001, 69683). As a result, radionuclides in soil are modeled so that they do not leach from the contaminated zone or become depleted due to erosion.

Although the contaminated zone is modeled as static with respect to radionuclide loss through erosion and leaching, RESRAD accounts for radon and tritium diffusing as gases from contaminated soil (lost to the atmosphere). As a result, the modeled tritium dose declines rapidly over time. Radon isotopes, however, will be in equilibrium with their parent nuclides; changes over time for dose rates through radon inhalation will depend on site-specific conditions with respect to the concentrations of these parent nuclides. Radon dose rates also can be influenced by many radon pathway-specific parameters that govern the diffusion of radon through soil and into a building. Tritium diffusion from soil as water vapor (resulting in the inhalation dose) is greatly affected by the absolute humidity of the air above the contaminated zone (a local value for absolute humidity is used in radionuclide SAL calculations) (LANL 2001, 69683).

5.0 EVALUATION OF COPCS RETAINED BY THE SCREENING ASSESSMENT

If the screening assessment demonstrates acceptable risk, the site needs no further evaluation for potential risk to human health. If there is a potential unacceptable risk based on the screening assessment, a risk management decision must be made whether to conduct a baseline risk assessment, proceed with an accelerated action, or inaugurate a CMS of the site. All COPCs identified and retained by the screening assessment are carried forward for further evaluation. Chemicals screened out using SALs or otherwise eliminated in the screening assessment may be excluded in a site-specific risk assessment. This is an appropriate course of action if it is apparent from the screening assessment that chemicals eliminated by this process contribute only negligible risks in a baseline risk assessment (e.g., less than 10% of the total risk, hazard, and/or dose). These chemicals should be discussed in the uncertainty section of the risk assessment with regard to their contribution to risk but are not used in the risk calculations. In other cases, many chemicals may be present at a site or a few chemicals may be present slightly below their SALs, in which case it may be prudent to include them in a baseline risk assessment. Such decisions are made on a site-by-site basis and in consultation with the administrative authority.

6.0 REFERENCES

The following list includes all references cited in this chapter. The parenthetical information following the reference provides the author, publication date, and ER Project identification (ER ID) number. This information is also included in the citation in the text and can be used to locate the document.

ER ID numbers are assigned by the Laboratory's ER Project to track all material associated with Laboratory potential release sites. These numbers can be used to locate copies of the documents at the ER Project's Records Processing Facility and, where applicable, within the ER Project reference library. The references cited in this work plan can be found in the volumes of the reference library titled "Reference Set for Analysis and Assessment Focus Area."

Copies of the reference library are maintained at the New Mexico Environment Department Hazardous Waste Bureau, the Los Alamos Area Office of the US Department of Energy, and the ER Project Office. This library is a living document that was developed to ensure that the administrative authority has all the necessary material to review the decisions and actions proposed in this work plan. However, documents previously submitted to the administrative authority are not included in the reference library.

ATSDR (Agency of Toxic Substances and Disease Registry) 1997. "Toxicological Profiles on CD-ROM: Toxicological Profiles of Endosulfan, Endrin, and Endrin Aldehyde" (ATSDR 1997, 56531.3)

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Attachment

Surrogate Chemicals in the Screening Assessment

Some chemicals do not have a published toxicity value (i.e., cancer slope factor or reference dose) in the US Environmental Protection Agency's (EPA's) Integrated Risk Information System (IRIS) database, Health Effects Assessment Summary Tables (HEAST), or National Center for Environmental Assessment (NCEA) publications. As part of the screening assessment, the risk assessor may either qualitatively discuss the presence of the chemical and the implications of the absence of the chemical from the risk estimate or evaluate the chemical quantitatively. If it is determined that a baseline risk assessment is appropriate, the chemical is retained as a chemical of potential concern (COPC) and evaluated as part of the potential overall risk.

The quantitative evaluation may involve the calculation of a screening action level (SAL) using toxicity data from sources other than those described above along with the equations and parameters presented in this methodology. Alternatively, the SAL of an appropriately similar chemical based on toxicity and/or chemical structure, i.e., a surrogate, may be used in the screening assessment. Subsequently, a discussion of the uncertainty related to the use of a surrogate is necessary for a risk management decision. The use of a surrogate may cause an under- or overestimation of risk.

The chemicals presented in Table A-1 have been detected but do not have SALs because adequate toxicity information is not available. These chemicals can potentially be evaluated using the SALs for similar chemicals listed in the table. This list is not intended to be all-inclusive.

Table A-1
Chemicals with Inadequate Toxicity Information and Corresponding Surrogate Chemicals

Chemical	Surrogate Chemical	Rationale
acenaphthylene	acenaphthene	structure similarity
2-amino-4,6-dinitrotoluene	2,6- dinitrotoluene	structure similarity
4-amino-2,6-dinitrotoluene	2,6- dinitrotoluene	structure similarity
benzo(g,h,i)perylene	pyrene	structure similarity
endosulfan I	endosulfan	Endosulfan I is an isomer of endosulfan
endosulfan II	endosulfan	Endosulfan II is an isomer of endosulfan
endosulfan sulfate	endosulfan	Endosulfan sulfate is the primary metabolite of endosulfan
endrin aldehyde	endrin	Endrin aldehyde is an impurity and breakdown product of endrin
endrin ketone	endrin	Endrin ketone is the primary metabolite of endrin
2-hexanone	2-butanone	structure similarity
isopropyltoluene	isopropylbenzene	structure similarity
2-methylnaphthalene	naphthalene	structure similarity
3-nitroaniline	2-nitroaniline	structure similarity
4-nitroaniline	2-nitroaniline	structure similarity
2-nitrophenol	4-nitrophenol	structure similarity
phenanthrene	anthracene*	structure similarity

*NMED uses pyrene as a surrogate for phenanthrene based on structural similarity.

acenaphthylene The surrogate chemical is acenaphthene. The difference between the two chemicals is a double bond located on the 5-carbon ring (Figure A-1). The double bond strengthens the structure making acenaphthylene less reactive than acenaphthene.

2-amino-4,6-dinitrotoluene The surrogate chemical is 2,6-dinitrotoluene. The difference between the two chemicals is the addition of an amino group (NH_2), the position of that amino group and the position of the nitro group (NO_2) (Figure A-2). Because there are fewer sites for reactions to occur on the surrogate 2,6-dinitrotoluene, the surrogate will be less reactive than 2-amino-4,6-dinitrotoluene.

4-amino-2,6-dinitrotoluene The surrogate chemical is 2,6-dinitrotoluene. The difference between the two chemicals is the addition of an amino group (NH_2) (Figure A-2). Because there are fewer sites for reactions to occur on the surrogate 2,6-dinitrotoluene, the surrogate will be less reactive than 4-amino-2,6-dinitrotoluene.

benzo(g,h,i)perylene The surrogate chemical is pyrene. The difference between the two chemicals is the addition two benzene rings (Figure A-3). The additional benzene rings add stability to the structure making the surrogate pyrene more reactive than benzo(g,h,i)perylene.

endosulfan I The surrogate chemical is endosulfan. Endosulfan I is an isomer of endosulfan. These chemicals have the same chemical weight and the same molecules. The melting point of endosulfan and endosulfan I is 106°C and 108°C, respectively. A comparison of structure is presented in Figure A-4.

endosulfan II The surrogate chemical is endosulfan. Endosulfan II is an isomer of endosulfan. These chemicals have the same chemical weight and the same molecules. The melting point of endosulfan and endosulfan II is 106°C and 207°C, respectively. A comparison of structure is presented in Figure A-4.

endosulfan sulfate The surrogate chemical is endosulfan. Endosulfan is converted to endosulfan sulfate and endosulfan diol in laboratory animals. Endosulfan and its primary metabolite, endosulfan sulfate, exhibit similar toxicities and are both believed to be responsible for the toxicity observed in animals. The dose that kills 50% of treated laboratory animals (LD50) for endosulfan is listed in the Merck Index as 43 mg/kg for male rats and 18 mg/kg for female rats; endosulfan sulfate was not listed. The Registry of Toxic Effects of Chemical Substances (RTECS) lists only one citation for endosulfan sulfate, which is titled "Analytical Reference Standards and Supplemental Data: The Pesticides and Industrial Chemicals Repository" (USDHHS 1987, 70943). This source listed the LD50 value of endosulfan sulfate as 18 mg/kg for rats. While there were no specifics given on the gender of the rats, this LD50 supports the statement that endosulfan sulfate is toxicologically similar to endosulfan (ATSDR 1997, 56531.3). The use of endosulfan as a surrogate for endosulfan sulfate could lead to an overestimation of the risk posed by endosulfan sulfate because the other primary metabolite (endosulfan diol) could be mainly responsible for the toxicity instead of endosulfan sulfate. The use of endosulfan as a surrogate for endosulfan sulfate is not likely to lead to an underestimate of the risk because the two chemicals exhibit similar toxicities (LD50s) observed in animals. A comparison of structures is presented in Figure A-5.

endrin aldehyde The surrogate chemical is endrin. Endrin aldehyde is an impurity and breakdown product of Endrin. Photochemical isomerization of endrin was observed after exposure of thin layers of solid endrin on glass to sunlight (ATSDR 1997, 56531.3). Minor amounts of endrin aldehyde were formed in this reaction. Endrin is also sensitive to transformation by heat, yielding primarily the pentacyclic ketone and endrin aldehyde at temperatures greater than 230°C (ATSDR 1997, 56531.3). A comparison of structures is presented in Figure A-6.

endrin ketone The surrogate chemical is endrin. Endrin ketone is a breakdown product of endrin. Photochemical isomerization of endrin, primarily to the pentacyclic ketone commonly called endrin

ketone, was observed after exposure of thin layers of solid endrin on glass to sunlight (ATSDR 1997, 56531.3). Results of seasonal studies indicated that this isomerization proceeds with a half-life of 5 to 9 days in intense summer sunlight, with complete conversion to endrin ketone in 15 to 19 days. A comparison of structures is presented in Figure A-6.

2-hexanone The surrogate chemical is 2-butanone. The difference between the two chemicals is that 2-hexanone has a longer chain with two carbon and four hydrogen atoms. Under the same conditions, these chemicals react similarly. 2-hexanone will be less stable due to the elongated shape causing a polarity difference between the oxygen and the carbon in the number six position (Figure A-7).

4-isopropyltoluene The surrogate chemical is isopropylbenzene based on structural similarity. The difference between the two chemicals is a carbon molecule in the number two position for 2-isopropyltoluene (Figure A-8). Isopropyltoluene is more reactive because the additional carbon affords an additional reaction site.

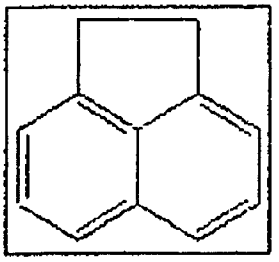
2-methylnaphthalene The surrogate is naphthalene based on structural similarity. The difference between the two chemicals is a carbon molecule in the number three position for 2-methylnaphthalene (Figure A-9). 2-methylnaphthalene is more reactive because the additional carbon affords an additional reaction site.

3-nitroaniline The surrogate chemical is 2-nitroaniline based on structural similarity. The difference between the two chemicals is the position of the nitro group (NO_2) (Figure A-10). Under the same conditions, these chemicals react similarly.

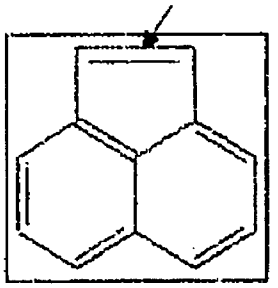
4-nitroaniline The surrogate chemical is 2-nitroaniline based on structural similarity. The difference between the two chemicals is the position of the nitro group (NO_2) (Figure A-10). Under the same conditions, these chemicals react similarly.

2-nitrophenol The surrogate chemical is 4-nitrophenol based on structural similarity. The difference between the two chemicals is the position of the nitro group (NO_2) (Figure A-11). Under the same conditions these chemicals react similarly.

phenanthrene The surrogate chemical is anthracene based on structural similarity. The difference between the two chemicals is the position of the benzene rings (Figure A-12). Under the same conditions, these chemicals react similarly. Phenanthrene is more polar and therefore more reactive than anthracene. NMED uses pyrene as a surrogate (Figure A-13).



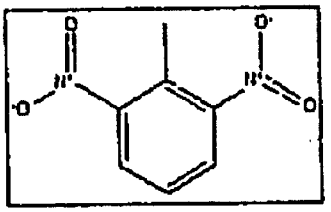
Acenaphthene
(Surrogate)



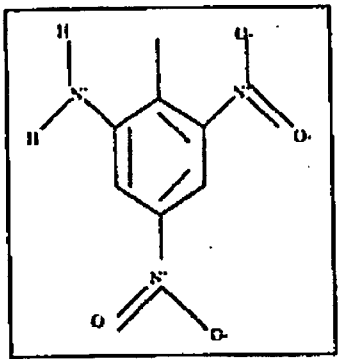
Acenaphthylene

Comment: The difference between the two chemicals, a double bond located on the 5-carbon ring, is indicated by an arrow.

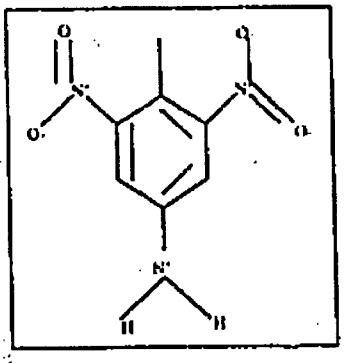
Figure A-1. Structure comparison between acenaphthene and acenaphthylene



2,6-DNT
(Surrogate)

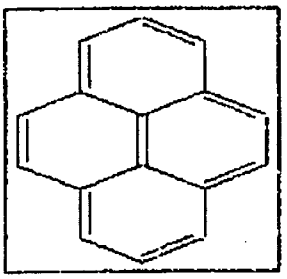


2-Amino-4,6-DNT

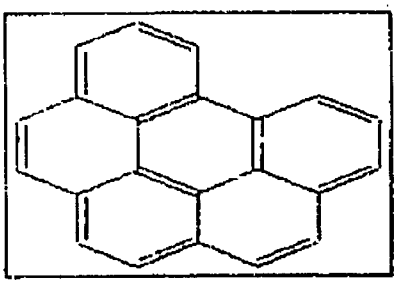


4-Amino-2,6-DNT

Figure A-2. Structure comparison among 2,6-DNT, 2-amino-4,6-DNT, and 4-amino-2,6-DNT



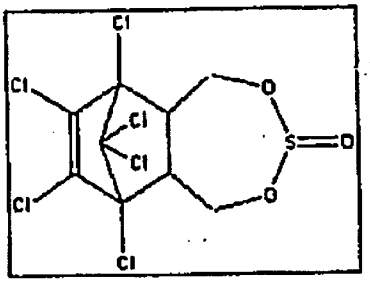
Pyrene
(Surrogate)



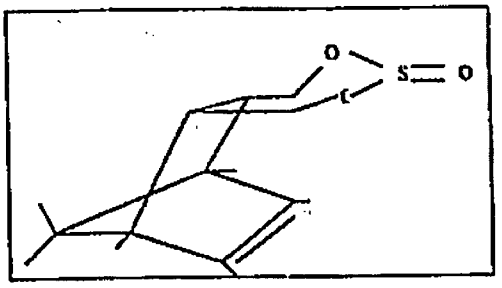
Benzo(g,h,i)perylene

Comment: The difference between the two chemicals, two additional benzene rings, is indicated by an arrow.

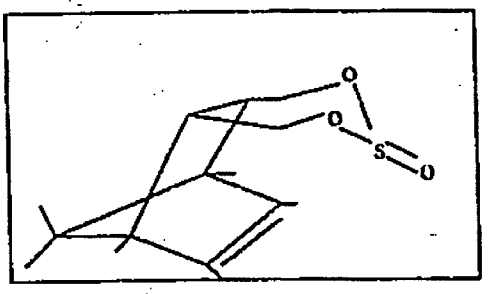
Figure A-3. Structure comparison between pyrene and benzo(g,h,i)perylene



Endosulfan
(Surrogate)

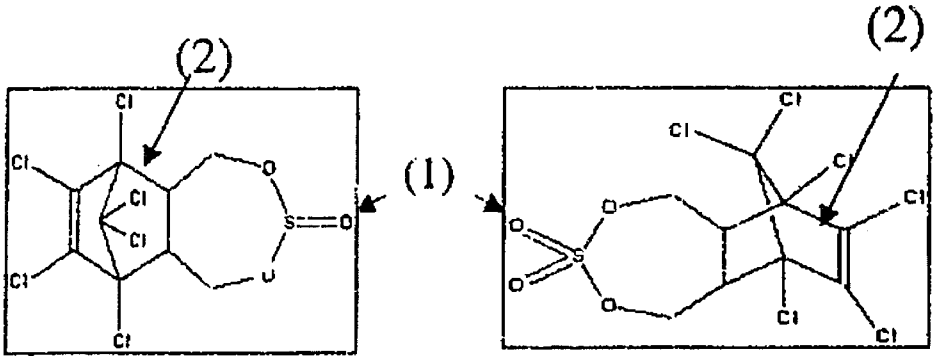


Endosulfan I



Endosulfan II

Figure A-4. Structure comparison among endosulfan and isomers endosulfan I and endosulfan II

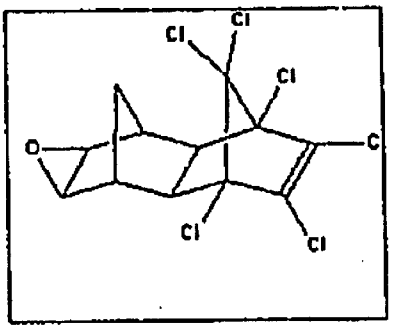


Endosulfan (Surrogate)

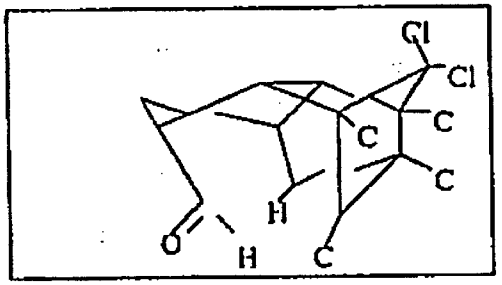
Endosulfan Sulfate

Comment: The differences between the two chemicals, (1) the number of oxygen molecules off the sulfate on the eight-sided ring, and (2) the shape of the six-sided ring, is indicated by arrows.

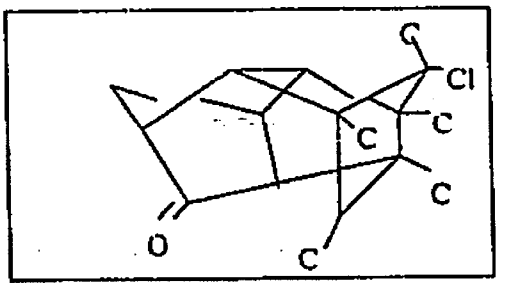
Figure A-5. Structure comparison between endosulfan and endosulfan sulfate



Endrin
(Surrogate)

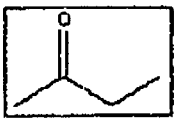


Endrin Aldehyde

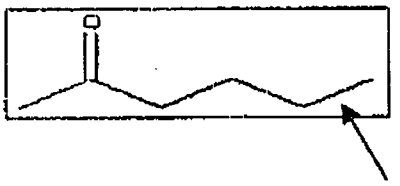


Endrin Ketone

Figure A-6. Structure comparison among endrin, endrin aldehyde, and endrin ketone



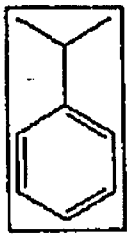
2-Butanone
(Surrogate)



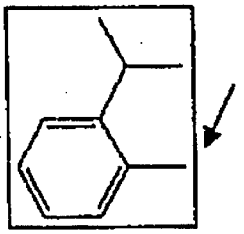
2-Hexanone

Comment: The difference between the two chemicals, two carbon and four hydrogen atoms, is indicated by an arrow.

Figure A-7. Structure comparison between 2-butanone and 2-hexanone



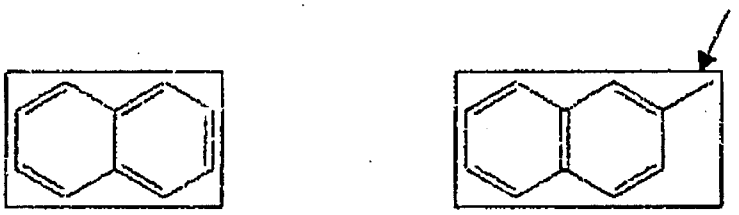
Isopropylbenzene
(Surrogate)



4-Isopropyltoluene

Comment: The difference between the two chemicals, a carbon molecule and two hydrogen molecules in the number two position, is indicated with an arrow.

Figure A-8. Structure comparison between isopropylbenzene and 4-isopropyltoluene

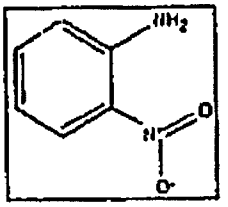


Naphthalene (Surrogate)

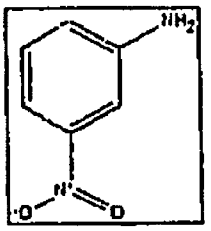
2-Methylnaphthalene

Comment: The difference between the two chemicals, a carbon molecule and two hydrogen molecules in the number three position, is indicated by an arrow.

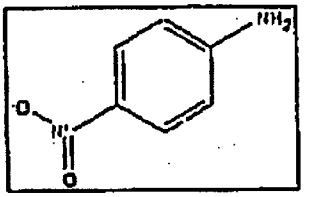
Figure A-9. Structure comparison between naphthalene and 2-methylnaphthalene



2-Nitroaniline
(surrogate)

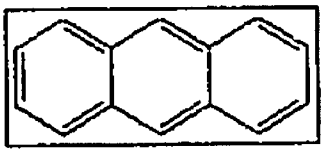


3-Nitroaniline

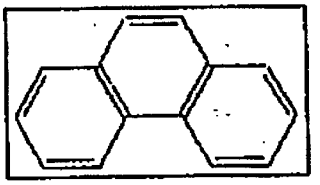


4- Nitroaniline

Figure A-10. Structure comparison among 2-nitroaniline, 3-nitroaniline, and 4-nitroaniline



Anthracene
(Surrogate)



Phenanthrene

Figure A-12. Structure comparison between anthracene and phenanthrene