Utah Department of Environmental Quality
Division of Waste Management and Radiation Control
(DWMRC)

Technical Guide for Risk Assessments:
Utah Administrative Code R315-101
(TGRA)

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF ACRONYMS AND ABBREVIATIONS

AI	Adequate Intake
ALM	Adult Lead Methodology
AOC	Area of Contamination
ASTDR	Agency for Toxic Substances and Disease Registry
AUF	Area Use Factor
BAF	Bioaccumulation Factor
bgs	Below Ground Surface
BTV	Background Threshold Value
C	Celsius
CEC	Cation Exchange Capacity
CFR	Code of Federal Regulation
CMTP	Composite Model for Leachate Migration with Transformation Products
COC	Contaminant of Concern
COI	Contaminant of Interest
COPC	Contaminants of Potential Concern
COPEC	Contaminant of Potential Ecological Concern
CSM	Conceptual Site Model
DAF	Dilution Attenuation Factor
DERR	Department of Environmental Response and Remediation
DQO	Data Quality Objectives
DU	Decision Unit
DWMRC	Division of Waste Management and Radiation Control
EC	Electrical Conductivity
EPC	Exposure Point Concentration
ERA	Ecological Risk Assessment
ESA	Environmental Site Assessment
ESL	Ecological Screening Level
ESP	Exchangeable Sodium Percentage
FOD	Frequency of Detection
GPS	Global Positioning System
HEAST	Health Effects Assessment Summary Tables
HI	Hazard Index
HQ	Hazard Quotient
IEUBK	Integrated Exposure Uptake Biokinetic
IRIS	Integrated Risk Information System
ISL	Initial Screening Level
IUPAC	International Union of Pure and Applied Chemistry
J&E	Johnson and Ettinger
Kow	Octanol-Water Partition Coefficient
LANL	Los Alamos National Laboratory
LOAEC	Lowest Observed Adverse Effect Concentration
LOAEL	Lowest Observed Adverse Effect Level
MCL	Maximum Contaminant Level

LIST OF ACRONYMS AND ABBREVIATIONS, Cont.

MCLG	Maximum Contaminant Level Goal
MDL	Minimum Detection Limit
meq/100g	milliequivalents per 100 grams soil
meq/L	Milliequivalents per liter
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
mmhos/cm	millimhos per centimeter
MRL	Minimum Risk Level
NAPL	Non-aqueous Phase Liquid
NFA	No Further Action
NOAEC	No Observed Adverse Effect Concentration
NOAEL	No Observed Adverse Effect Level
NRCS	Natural Resources Conservation Service
OSWER	Office of Solid Waste and Emergency Response
PAH	Polycyclic Aromatic Hydrocarbon
PCB	Polychlorinated Biphenyl
PEF	Particulate Emission Factor
PFAS	Polyfluoroalkyl and Perfluoroalkyl Compounds
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctane Sulfonate
PPE	Probably Point of Entry
PPRTV	Provisional Peer-reviewed Toxicity Value
PUF	Plant Uptake Factor
QSAR	Quantitative Structure-activity Relationship
RCRA	Resource Conservation and Recovery Act
RfC	Reference Concentration
RfD	Reference Dose
RFI	RCRA Facility Investigation
RME	Reasonable Maximum Exposed
RSL	Regional Screening Level
SAP	Sampling and Analysis Plan
SLERA	Screening Level Ecological Risk Assessment
SLHQ	Screening Level Hazard Quotient
SQL	Sample Quantitation Level
SSL	Soil Screening Level
SU	Sample Unit
SWMU	Solid Waste Management Unit
TCDD	Tetrachlorodibenzo-p-dioxin
TCE	Trichloroethylene
TEF	Toxicity Equivalency Factor
TEQ	Toxicity Equivalent Concentration
TPH	Total Petroleum Hydrocarbon

LIST OF ACRONYMS AND ABBREVIATIONS, Cont.

TSS	Total Suspended Solids
TRV	Toxicity Reference Value
UAC	Utah Administrative Code
UCL	Upper Confidence Limit
US EPA	United States Environmental Protection Agency
USGS	United States Geologic Survey
UTL	Upper Tolerance Limit
VF	Volatilization Factor
VISL	Vapor Intrusion Screening Level
VOC	Volatile Organic Compound
WHO	World Health Organization

DEFINITIONS

Acceptable Risk Range: cancer risk greater than or equal to one additional cancer in 1,000,000 (1E-06), but less than or equal to one additional cancer in 10,000 (1E-04) or a noncancer hazard index less than or equal to one.

Action Level: the existence of a contaminant concentration in the environment that is high enough to warrant an action or trigger a response action.

Adverse Effect: any effect that causes harm to the normal functioning of plants, animals, or humans due to exposure to any contaminants of concern.

Appropriate Site Management Activities: measures that are reasonable and practical that will be taken to control and reduce risks greater than 1E-06 and less than 1E-04 for carcinogen and hazard index equal to or less than one for non-carcinogens under both current and reasonably anticipated future land use conditions, for example, institutional controls, engineering controls, groundwater monitoring, post-closure care, or corrective action and ensuring that assumptions made in the estimation of cancer risk and non-cancer hazard in the risk assessment report are not violated.

Assessment Endpoints: an explicit expression of environmental value that is to be protected. It is the part of the ecosystem that should be protected at a Superfund site, and it is generally some characteristic of a species of plant or animal, for example, reproduction, growth, that may be described numerically.

Background: measurements that are not influenced by releases from a site. Background constituents may be naturally occurring in the environment in forms that have not been influenced by human activity or may be natural and human-made substances present in the environment resulting from anthropogenic activities and not related to the site.

Background Threshold Value (BTV): a single value most often used to represent soil background levels. The BTV may be a default level established by the Division of Waste Management and Radiation Control (DWMRC), a surrogate level from another facility, or a site-specific level. This level is used to determine what constituents are present due to natural or anthropogenic levels or are representative of contamination.

Boundary: the furthest extent where contamination from a defined source has migrated in any medium when the release is first identified. This is often referred to as extent, when defining nature and extent of contamination.

Cancer Risk: the probability that an individual with contract cancer after lifetime exposure to a carcinogen.

Censored Data Sets: Data sets that contain one or more observations which are nondetects.

Cleanup: the range of corrective action activities that occur in the context of addressing environmental contamination at Resource Conservation and Recovery Act (RCRA) sites to lower contaminant concentration or decrease chemical toxicity. Activities may include but are not limited to waste removal, contaminated media removal or source reduction, such as excavation or pumping, in-place treatment of waste or contaminated media, such as bioremediation, monitored natural attenuation, hydraulic control and/or containment of waste or contaminated media, such as barrier walls, low permeability covers, liners or capping, or various combination of these approaches.

Complete Exposure Pathway: An exposure pathway is the link between a contaminant source and a receptor (United States Environmental Protection Agency, U.S. EPA, 1991). A complete exposure pathway is one in which the stressor can be traced or expected to travel from the source to a receptor that can be affected by that stressor and shall meet the following: (1) the presence of a source and transport; (2) exposure point or contact (receptor); and (3) exposure route. Otherwise, exposure is incomplete.

Conceptual Site Model (CSM): a written, illustrative, or both, representation of a site that documents the physical, chemical and biological processes that control the transport, migration, actual or potential, or both impacts of contamination in soil, air, ground water, surface water, sediments, to human or ecological receptors, or both, exposure pathways, at a site or at a reasonably anticipated site under both current and potential future land use scenarios.

Contaminate: to make a medium polluted through the introduction of hazardous waste or hazardous constituents as identified in Utah Administrative Code (UAC) R315-261-1092, which incorporates by reference 40 Code of Federal Regulations (CFR) Part 261, Appendix VIII.

Contaminant of Concern (COC): Constituents of Potential Concern that significantly contribute to a pathway in a land use scenario for a receptor that either exceeds a cumulative cancer risk of 1E-04 or exceeds a non-cancer hazard index of one.

Constituents of Potential Concern (COPC): constituents detected in a medium that are selected to be addressed in the risk assessment process because contact with humans may result in adverse effects.

Constituents of Potential Ecological Concern (COPEC): any constituent that is shown to pose possible ecological risk at a site. It is generally a constituent that may or may not be causing risk or adverse effects to plants and animals at a site.

Corrective Action: the cleaning up of environmental problems caused by the mismanagement of wastes, or the cleanup process or program under RCRA and any activities related to the investigation, characterization, and cleanup of release of hazardous waste or hazardous constituents from solid waste management units or hazardous waste management units at a permitted or interim status treatment storage or disposal facilities or voluntary cleanup sites or brownfield sites.

Corrective Action Complete With Controls: a condition of a solid waste management unit, a hazardous waste management unit, an area of contamination or a contaminated site where site characterization or risk assessment indicate corrective action is required and completed and the results of the risk assessment meet the closure standards and requirements specified in UAC R315-101-7(b), or a condition of a solid waste management unit, a hazardous waste management unit, area of contamination or a contaminated site where site characterization or risk assessment indicate corrective action is not required but also meets the closure standards and requirements specified in UAC R315-101-7(b).

Corrective Action Complete Without Controls: a condition of a solid waste management unit, a hazardous waste management unit, area of contamination or a contaminated site where site characterization or risk assessment indicate corrective action is required and completed and the results of the risk assessment meet the closure standards and requirements equivalent to a no further action or meeting the requirements of UAC R315-101-7(a) or a condition of a solid waste management unit, a hazardous waste management unit, area of contamination or a contaminated site when site characterization or risk assessment indicate corrective action is not required but also meets the closure standards and requirements equivalent to a no further action or meeting the requirements of UAC R315-101-7(a).

Corrective Action Level: the concentration of a contaminant in a medium after cleanup of a site that is protective of human health and the environment.

Data Quality Objectives (DQO): qualitative and quantitative statements of the quality of data needed to support specific decisions or regulatory actions.

Detection Limit: a measure of the capability of an analytical method to distinguish samples that do not contain a specific analyte from samples that contain low concentrations of the analyte. It is the lowest concentration or amount of the target analyte that can be determined to be different from zero by a single measurement at a stated level of probability. Detection limits are analyte and matrix-specific and are laboratory-dependent.

Dilution Attenuation Factor (DAF): the ratio of the contaminant concentration in soil leachate to the concentration in groundwater at the receptor point.

Dose-Response Assessment: this describes how the likelihood and severity of adverse health effects (the responses) are related to the amount and condition of exposure to an agent (the dose provided). Typically, as the dose increases, the measured response also increases, although not linear in relationship. At low doses there may be no response. At some level of dose, the responses begin to occur in a small fraction of the study population or at a low probability rate. Both the dose at which response begin to appear and the rate at which it increases given increasing dose can be variable between different pollutants, individuals, exposure routes, etc.

Environment: the surroundings or conditions in which a person, animal, or plant lives or operates.

Exposure: contact of an organism with a chemical or physical agent and it is the amount of the agent available at the exchange boundaries of the organism.

Exposure Assessment: the process of measuring or estimating the magnitude, frequency, and duration of exposure to a constituent in the environment or estimating future exposures for a constituent that has not yet been released. The exposure assessment answers the question of how much of the pollutant are receptors exposure to during a specific time period.

Exposure Pathway: the course a chemical or physical agent takes from a source to an exposed organism.

Exposure Point Concentration (EPC): either a maximum detected value or a statistical derivation of measured or modeled data that represents an estimate of the chemical concentration available from a particular medium or route of exposure. The exposure point concentration value is used to quantify potential cancer risks and non-cancer hazards.

Groundwater Cleanup Levels: site-specific groundwater chemical concentration levels based on groundwater use designation and exposure pathway established to ensure the protection of human health and the environment when defining groundwater cleanup objectives.

Groundwater Use: the current or reasonably expected maximum beneficial use of groundwater that warrants the most stringent cleanup levels, including drinking or other uses.

Hazard Identification: the process of determining whether exposure to a stressor can cause an increase in the incidence of specific adverse health effects. This process in the risk assessment answers the question of what health problems, either human or ecological, are caused by the pollutant.

Hazard Index (HI): the sum of hazard quotients.

Hazard Quotient: (HQ) the ratio of the potential exposure to a substance and the level at which no adverse effects are expected. The HQ is calculated to evaluate the potential for non-cancer health hazards to occur from exposure to a contaminant. An HQ may be calculated for human health or ecological receptors.

Lowest Observed Adverse Effects Level (LOAEL) or Lowest Observed Adverse Effects Concentration (LOAEC): the lowest level of a chemical stressor evaluated in a toxicity test that shows harmful effects on a plant or animal. A LOAEL is based on dose of a chemical ingested while a LOAEC refers to direct exposure to a chemical such as through the skin.

Maximum Contaminant Level (MCL): the highest level of a contaminant that is allowed in drinking water and is set as close to the "Maximum Contaminant Level Goal" as feasible using the best available treatment technology and taking cost into consideration. Maximum Contaminant Levels are enforceable standards.

Maximum Contaminant Level Goal (MCLG): the level of a contaminant in drinking water below which there is no known or expected risk to health. Maximum Contaminant Level Goals allow for a margin of safety and are non-enforceable public health goals.

Measures of Effects: quantitative measurements of effects expressed as statistical or numerical assessment endpoint summaries of the observations that make up the measurement.

Measurement End Point: a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint and it is a measure of biological effects such as death, reproduction, or growth, of a particular species.

Natural Resources: land, fish, wildlife, biota, air, water, ground water, drinking water supplies, and other similar resources.

No Further Action (NFA): the state of a solid waste management unit, a hazardous waste management unit, or a contaminated site at closure meeting the requirements in UAC R315-101-7(a) and it is equivalent to corrective action complete without controls if the site was under corrective action activities. No further action is equivalent to unrestricted land use.

Nonparametric: A term describing statistical methods that do not assume a particular population probability distribution and are therefore valid for data from any population with any probability distribution, which can remain unknown.

No Observed Adverse Effects Level (NOAEL) or No Observed Adverse Effects Concentration (NOAEC): the highest level of a chemical stressor in a toxicity test that did not cause a harmful effect in a plant or animal. A NOAEL refers to a dose of chemical that is ingested, while a NOAEC refers to direct exposure to a chemical such as through the skin.

Parametric: A term describing statistical methods that assume a probability distribution such as a normal, lognormal, or a gamma distribution.

Point of Departure: the target risk level that risk to an individual is considered insignificant.

Potentially Complete Exposure Pathway: a pathway that, due to current site conditions, is incomplete but could become complete at a future time because of changing site practices. For example, the ingestion pathway of groundwater from a residential well in a high total dissolved solids aquifer. This pathway could be complete if treatment technologies like reverse osmosis become economically feasible and are observed to be employed successfully in that aquifer.

Reasonable Maximum Exposure (RME): the highest exposure that is reasonably expected to occur at a site. Reasonable Maximum Exposure combines upper-bound and mid-range exposure factors so that the result represents an exposure scenario that is both protective and reasonable, not the worst possible case.

Regional Screening Levels (RSL): risk-based chemical concentrations derived from standardized equations combing exposure assumptions with US EPA chemical-specific toxicity values and target risk levels that are used for site screening and initial cleanup goals. For the residential receptor, the residential RSLs should be applied. For the industrial/commercial scenario, the composite worker RSLs should be applied. For the construction worker, the on-line calculator must be used to derive scenario specific RSLs.

Release: spill or discharge of hazardous waste, hazardous constituents, or material that becomes hazardous waste when released to the environment.

Regression on Order Statistic (ROS): A regression line is fit to the normal scores of the order statistics for the uncensored observations and is used to fill in values imputed from the straight line for the observations below the detection limit.

Responsible Party: the owner or operator of a site, or any other person responsible for the release of hazardous waste or hazardous constituents.

Risk-Based Clean Closure: closure of a site where hazardous waste was managed or any medium that has been contaminated by a release of hazardous waste or hazardous constituents, and where hazardous waste or hazardous constituents remain at the site in any medium at concentrations determined, in UAC R315-101, to cause minimal levels of risk to human health and the environment so as to require no further action or monitoring by the responsible party nor any notice of hazardous waste management on the record of title to the property.

Risk-Based Concentration: the concentration of a contaminant the values of which are derived from equations combining toxicity factors with standard exposure scenarios to calculate chemical concentrations corresponding to some fixed levels of risks in any medium, such as water, air, fish tissue, sediment, and soil.

Risk Characterization: summarize and integrate information from the hazard identification, dose-response, and exposure assessment phases of the risk assessment to synthesize an overall conclusion about risk. Risk characterization takes place in both human health risk assessments and ecological risk assessments.

Robust Statistic: a statistic that is resistant to errors in the results, produced by deviations from assumptions, such as normality. This means that the limits are not susceptible to outliers, or distributional assumptions. For example, if the limits are centered on the median, instead of on the mean, or on a modified, "robust mean," and constructed with suitable weighting, or influence, or function, they could be considered "robust."

Site: the area of contamination and any other area that could be impacted by the released contaminants, or could influence the migration of those contaminants, regardless of whether the site is owned by the responsible party.

Site Specific Screening Value: contaminant screening values derived for media, such as soil, sediment, water, at a site based on relevant site assumptions and factors.

Source Control: a range of actions, for example, removal, treatment in place, and containment, designed to protect human health and the environment by eliminating or minimizing migration of or exposure to significant contamination.

Target Risk: any acceptable specified risk level. the protective end of the acceptable risk range for screening of contaminants in risk assessment and considered to be the point of departure. The target risk is defined as 1E-06 and is appropriate for all human receptors.

Upper Confidence Limit (UCL): the upper boundary of a confidence interval. Because of the uncertainty associated with estimating the true average concentration at a site, the 95% UCL of the arithmetic mean is used to represent this variable and provides reasonable confidence that the true site average will not be underestimated.

Upper Tolerance Limit (UTL): A confidence limit on a percentile of the population rather than a confidence limit on the mean where a defined percentage of sample data will be less than or equal to that limit. For example, a 95% one-sided UTL for 95% coverage represents the value below which 95% of the population values are expected to fall with 95% confidence. In other words, a 95% UTL with coverage coefficient 95% represents a 95% UCL for the 95th percentile.

Utah Administrative Code (UAC) R315-101: outlines the cleanup actions and risk-based closure standards and applies to cleanup actions conducted voluntarily as well as corrective action at permitted sites. The complete rule may be found at https://adminrules.utah.gov/public/rule/R315-101/Current%20Rules.

TECHNICAL GUIDE FOR RISK ASSESSMENTS:
UTAH ADMINISTRATIVE CODE R315-101
(TGRA)

1.0 PURPOSE

1.1 Purpose

The Utah Division of Waste Management and Radiation Control (DWMRC) developed the *Technical Guide for Risk Assessments: Utah Administrative Code R315-101* (or the TGRA) to assist facilities within the State of Utah in navigating Utah Administrative Code (UAC) R315-101 (herein referred to as the Rule), which sets the standards for risk-based closure. TGRA outlines recommended approaches to both human health and ecological risk assessments based on current State and Federal risk assessment practices. The overarching objective of the TGRA is to allow for a consistent interpretation of the Rule when conducting risk assessments. The TGRA is focused on how to complete human health and ecologic risk assessments required under UAC R315-101.

1.2 Applicability

UAC R315-101 applies to sites in Environmental Cleanup Program, Corrective Action Sites, permitted facilities, releases from spills, and hazardous waste generators that are not cleaned up to background. UAC R315-101 risk-based cleanup standards apply to sites that will not or cannot be cleaned to background constituent levels. When some amount of contamination may be left in place, risk assessments are conducted to ensure the residual risks can be managed for the protection of human health and the environment. The process of conducting these risk assessments is outlined in the TGRA.

2.0 STABILIZATION OF RELEASES

In order to protect human health and the environment, when there has been a release, immediate action to stabilize the site either through source removal or source control must be taken by the responsible party. These actions apply to the spilled material, and any residue or contaminated media resulting from the spill and posing a hazard to human health or the environment.

Stabilization of releases is required for any hazardous waste handler, including transporters and sites under the Environmental Cleanup Program, Corrective Action sites, and permitted facilities. It is noted that permitted facilities will likely have permit conditions addressing spills, stabilization of releases and notification requirements. The facility-specific permit conditions should always be followed for permitted facilities.

If the DWMRC determines that the action taken to stabilize a release is insufficient to meet the requirements of the emergency control of spills as outlined UAC R315-263-30(c)(7) and cleanup requirements in UAC R315-263-31, additional corrective action will be required and is to be outlined in a work plan, to be submitted to the DWMRC, addressing the mitigation of the released waste.

The work plan will need to (1) define the scope of work to be performed, (2) include a description of the interim measures and other corrective actions to be taken, and (3) include a description of how the plan will meet the criteria of source removal or source control to residential levels requiring no long-term site controls.

> UAC R315-263-30(c)(7) states that: *in the event of a spill of hazardous waste or material which, when spilled, becomes hazardous waste, the person responsible for the material at the time of the spill shall immediately provide the emergency action taken to minimize the threat to human health and the environment when reporting the spill.*
>
> UAC R315-263-31 states that*: the person responsible for the material at the time of the spill shall clean up all the spilled material and any residue or contaminated media or other material resulting from the spill or take action as may be required by the DWMRC so that the spilled material, residue, or contaminated media no longer presents a hazard to human health or the environment as defined in UAC R315-101. The cleanup or other required actions shall be at the expense of the person responsible for the spill. If the person responsible for the spill fails to take the required action, the DWMRC may take action and bill the responsible person.*

If the responsible party is not able to clean up impacted media to background levels, they may perform human health and ecological risk assessments to verify that contamination has been removed or mitigated to residential closure levels and no ecological risks. If the responsible party is able to make these demonstrations, they may petition the DWMRC for a determination of Corrective Action Complete without Controls, or No Further Action (NFA). What this means is that no residual contamination may remain that would restrict future land use.

The removals will be considered complete and compliant with UAC R315-263-31 when the following conditions are shown in the risk assessment:

- The level of cumulative residential risk present at the site is less than or equal to 1E-06 for carcinogens and the hazard index is less than or equal to 1.0 for non-carcinogens (See Section 6);
- Ecological effects are insignificant (See Section 9); and
- Current and potential future impacts to groundwater are insignificant as determined by the soil-to-groundwater pathway screening assessment (see Section 8).

3.0 SITE CHARACTERIZATION AND DOCUMENTATION

The site characterization phase is intended determine the degree and extent of on-site contamination providing spatial and contextual information about the site, which may be used to determine if there is any reason to believe complete exposure pathways may exist at the site where a release of hazardous waste/constituents has occurred. The site characterization may be conducted as part of due diligence and include phased Environmental Site Assessments (ESA) for sites under the Environmental Cleanup Program. For Resource Conservation and Recovery Act (RCRA) part B permitted and corrective action sites, site characterization will likely consist of phased RCRA Facility Investigations (RFI). Regardless of the program, the elements and intent of site characterization are similar.

Details on how to conduct site characterization are outside the scope of the TGRA. However, some elements of site characterization are important in terms of data needs for risk assessments. During site characterization nature (chemical contaminants) and extent (horizontal and vertical) of contamination for all potentially impacted media are defined. Media may include soil, sediment, groundwater, surface water, biota, and air. During site characterization, the site history should be reviewed to determine contaminants that could potentially be present due to site history, identify sampling needs to determine background threshold values (BTVs), and develop a conceptual site model (CSM).

<u>It is important to note that risk assessments should not be submitted to the Division until the nature and extent of contamination are defined</u>.

3.1 Conceptual Site Model

A CSM is useful in planning the risk assessment process by providing information about the types of contamination known or suspected at the site, and the mechanisms by which human and ecological receptors could be exposed to the contaminants. Site-specific CSMs should be developed early in the site-specific risk assessment processes to aid in providing direction to sampling efforts and risk assessment objectives. The necessary components that will be included in the CSMs are (1) sources of contamination, (2) release mechanisms, (3) affected media, (4) potential receptors, and (5) exposure pathways. All five elements must be present for the exposure pathway to be considered complete.

A CSM is a graphical representation of site conditions that conveys what is known or suspected, at a discrete point in time, about the site-specific sources, releases, release mechanisms, contaminant fate and transport, exposure routes, and potential receptors. The CSM is generally documented by written descriptions and supported by maps, geological cross-sections, tables, diagrams and other illustrations to communicate site conditions. When preparing a CSM, the facility should decide the scope, quantity, and relevance of the information to be included, balancing the need to present as complete a picture as possible to illustrate current site conditions and establish risk management actions, with the need to keep the information focused and exclude extraneous data.

The CSM should identify all potential exposure pathways for both human health and ecological risk assessments. While each site may have unique pathways, common human health pathways include:

- Direct (and incidental) ingestion of soil,
- Dermal contact with soil,
- Inhalation of volatiles and fugitive dusts from contaminated soil,
- Ingestion of groundwater,
- Dermal contact with groundwater,
- Inhalation of volatile organic compounds (VOCs) volatilized from groundwater into indoor air, and
- Inhalation of volatiles in indoor air via the subsurface vapor intrusion pathway.

An example of a CSM showing both human health and ecological receptors is also provided as Figure 1.

Under some site-specific situations, additional complete exposure pathways may be identified. In these cases, a site-specific evaluation of risk or development of pathway-specific screening levels is warranted under which additional exposure pathways can be considered. If other land uses and exposure scenarios are determined to be appropriate for a site (e.g., farming, recreational land use, hunting, and/or Native American land use), the exposure pathways addressed in this document should be modified or augmented accordingly or a site-specific risk assessment should be conducted. Early identification of the need for additional information is important because it facilitates development of a defensible sampling and analysis strategy.

3.2 Receptors and Pathways

The three most common human receptors are a resident, industrial/commercial worker, and a construction worker. Most, if not all, risk assessments should evaluate these three receptors as part of the human health risk assessment. Note that receptors may be current, future/anticipated, or hypothetical. While a site may be slated for industrial use, the residential receptor would still be required to be assessed if NFA was desired. Ecological receptors are addressed in Section 9.

Receptors may primarily be exposed to contamination via several pathways, including soil, water (surface water or groundwater), and air.

3.2.1 Residential Receptors

A residential receptor may be actual or hypothetical. Evaluation of this receptor is required to achieve closure under NFA or to demonstrate the site risks are within an acceptable range to allow closure with controls.

A residential receptor is assumed to be a long-term receptor residing within the site boundaries. Adults and children exhibit different ingestion rates for soil. To account for changes in intake as the receptor ages, the US EPA Regional Screening Levels (RSLs) have incorporated age adjusted intakes in the derivation of the levels. Exposure to soil (to depths of zero to 10 feet below ground surface, bgs) is expected to occur during home maintenance activities and outdoor play activities.

Contaminant intake is assumed to occur via three exposure pathways – direct ingestion, dermal absorption, and inhalation of volatiles and fugitive dusts. The residential RSLs for soil include exposure via direct ingestion of soil, dermal absorption, and inhalation of fugitive dust.

The indoor air RSLs are compared to ambient air samples collected in a building or residence. However, in most cases, indoor air data are not available, and the vapor intrusion scenario is estimated using sub-slab soil gas or groundwater data. The residential RSLs for indoor air do not account for inhalation of volatiles indoors via vapor intrusion estimated from soil gas or groundwater. If VOCs are present at a site, and indoor air data are not available, the vapor intrusion pathway may require evaluation and the risks/hazards using VISLs and added to risk/hazard determined using the RSLs (see Equations 10 and 11). Refer to Section 7 on vapor intrusion.

Example:

- Indoor Air Data – use residential RSL indoor air screening level. As a side note, the VISL calculator may list an indoor air concentration in addition to VISLs for subslab and groundwater. The indoor air concentration listed in the VISL calculator is the same as the RSL indoor air screening level.
- Subslab Data Only – use VISL calculator (See Section 7) to determine an estimated indoor air concentration based on migration of VOCs through a building foundation.
- Groundwater Data Only – use VISL calculator (See Section 7) to determine an estimated indoor air concentration based on migration of VOCs from groundwater through soil and into a building.

The residential RSLs do not take into consideration ingestion of homegrown produce/meat/fish/dairy, vapor intrusion estimated from soil gas or groundwater, or other unique exposure pathways. If these pathways are complete, analysis of risks resulting from these additional exposure pathways must be determined and added to the total risk and hazard (refer to Section 6 and Equations 10 and 11).

3.2.2 Industrial/Commercial

The industrial/commercial scenario is considered representative of on-site workers who split their day between indoor and outdoor activities. Exposure to surface and shallow subsurface soils (i.e., at depths of zero to one ft bgs) is expected to occur during moderate digging associated with routine maintenance and ground-keeping activities. An industrial/commercial receptor is expected to be the most highly exposed receptor in the outdoor environment under generic or day-to-day industrial/commercial conditions. Thus, the industrial RSLs for this receptor are expected to be protective of other reasonably anticipated indoor and outdoor workers at a commercial/industrial facility. Note that RSLs for the industrial/commercial receptor are identified as “industrial” on the RSL table but are discussed in the RSL User’s Guide as a “composite worker”.

Similar to the resident, the industrial RSLs for soil include exposure via direct ingestion of soil, dermal absorption and inhalation of fugitive dust.

Similar to the resident, the industrial RSLs do not account for inhalation of volatiles via vapor intrusion. If vapor intrusion is complete, analysis of risks resulting from these additional exposure pathways must be determined and added to the total risk and hazard (refer to Section 7 and Equations 10 and 11). Industrial air RSLs are compared to indoor air samples collected in a building. However, in most cases, indoor air data are not available, and the vapor intrusion scenario is estimated using sub-slab soil gas or groundwater data. The industrial RSLs for indoor air do not account for inhalation of volatiles indoors via vapor intrusion estimated from soil gas or groundwater. If VOCs are present at a site, and indoor air data are not available, the vapor intrusion pathway may require evaluation and the risks/hazards using VISLs and added to risk/hazard determined using the RSLs (see Equations 10 and 11). Refer to Section 7 on vapor intrusion.

Example:

- Indoor Air Data – use RSL industrial air screening level. As a side note, the VISL calculator may list an indoor air concentration in addition to VISLs for subslab and groundwater. The indoor air concentration listed in the VISL calculator is the same as the RSL indoor air screening level.
- Subslab Data Only – use VISL calculator (See Section 7) to determine an estimated indoor air concentration based on migration of VOCs in soil through a building foundation.
- Groundwater Data Only – use VISL calculator (See Section 7) to determine an estimated indoor air concentration based on migration of VOCs from groundwater through soil and into a building.

3.2.2 Construction Worker

A construction worker is assumed to be a receptor that is exposed to contaminated soil during the workday for the duration of a single on-site construction project. If multiple construction projects are anticipated, it is assumed that different workers will be employed for each project. The activities for this receptor typically involve substantial exposures to surface and subsurface soils (i.e., at depths of zero to 10 feet bgs) during excavation, maintenance, and building construction projects (intrusive operations).

A construction worker is assumed to be exposed to contaminants via the following pathways: incidental soil ingestion, dermal contact with soil, and inhalation of contaminated outdoor air (volatile and particulate emissions). While a construction worker receptor is assumed to have a higher soil ingestion rate than a commercial/industrial worker due to the type of activities performed during construction projects, the exposure frequency and duration are assumed to be significantly shorter due to the short-term nature of construction projects.

Either lines of evidence need to be provided to demonstrate other scenarios are protective of the construction worker or the RSL on-line calculator will need to be run to derive construction worker screening levels. For example, if none of the RSLs for site contaminants are driven by inhalation toxicity (e.g., manganese), then it is possible the residential scenario RSLs are protective of the construction worker, and a qualitative analysis may be sufficient.

Refer to Section 6.1 for more details on deriving construction worker RSLs. Note, if site-specific RSLs for the construction worker are calculated, subchronic toxicity should be used when available.

3.2.3 Other Receptors

Other receptors may be present at a site, such as a trespasser or recreationalists and other unique exposure scenarios such as those for Native American communities, that may not be reflected in the generic RSLs. If other receptors are present at a site, either site-specific RSLs may be developed using the on-line calculator or lines of evidence may be provided to demonstrate the generic RSLs are protective of these additional receptors.

3.2 Soil Exposure Intervals

Based on current and potential/hypothetical land-use scenarios, receptors for completed exposure pathways can be exposed to varying depths of soil, or soil exposure intervals. Per the US EPA (US EPA, 1989), depth of samples should be considered, and surface soils should be evaluated separately from subsurface soils due to possible differences in exposure levels that would be encountered by different receptors. Exposure intervals for each receptor are based on the most likely types of activities and potential soil exposure. Default exposure intervals are summarized in Table 1.

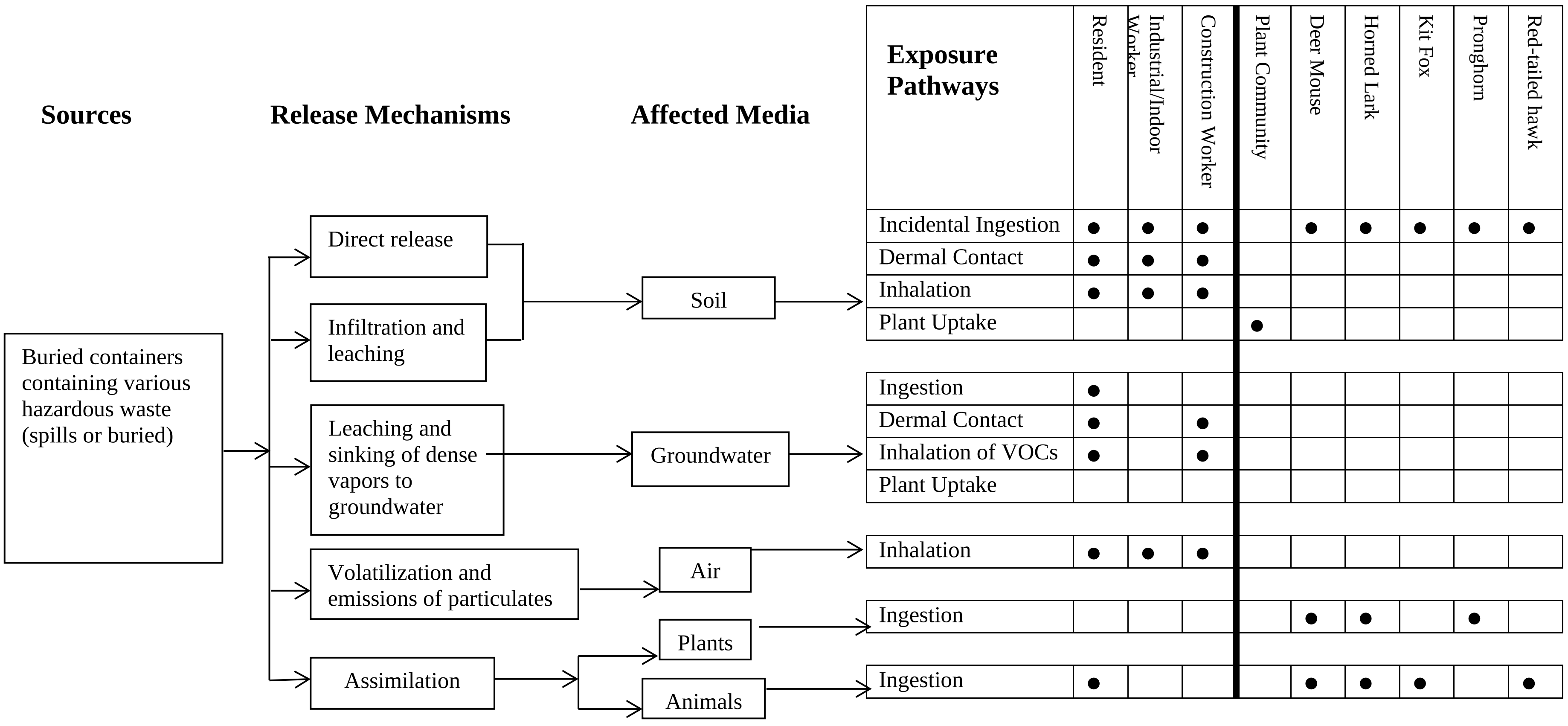
Residents could be exposed to surface and subsurface soils during home maintenance activities, yard work, landscaping, and outdoor play activities. Therefore, an exposure soil interval of 0-10 ft bgs should be assumed when evaluating soil exposure by a residential receptor.

It is assumed that industrial/commercial workers would only be exposed to surface soils (0-1 ft bgs). This receptor may be involved with routine maintenance and groundskeeping activities.

A construction worker is assumed to be exposed to surface and subsurface soils up to depths of 0-10 ft bgs. Construction workers are involved in digging, excavation, maintenance and building construction projects and could be exposed to surface as well as subsurface soil.

When evaluating the soil-to-groundwater pathway, refer to Section 8, concentrations are not restricted to a specific soil interval. Rather, the maximum detected concentration, regardless of depth, is used for the initial screening.

Exposure to soil by ecological receptors should be addressed separately in a tiered approach as outlined in Section 8. However, a discussion of soil exposure intervals for ecological receptors is warranted here because ecological receptors are considered in the CSM and depending on the types of ecological receptors, there could be a difference in exposure levels due to soil exposure intervals. Burrowing animals and deep rooted plants would be exposed to deeper soils, whereas all other animals/receptors would only be exposed to surface and shallow subsurface soils. Therefore, concentrations in soil 0-6 feet bgs should be assessed for burrowing animals while soil 0-1 ft bgs should be assessed for all other ecological receptors.



Exposure Pathways	Resident	Industrial/Indoor Worker	Construction Worker	Plant Community	Deer Mouse	Horned Lark	Kit Fox	Pronghorn	Red-tailed hawk
Incidental Ingestion	●	●	●		●	●	●	●	●
Dermal Contact	●	●	●						
Inhalation	●	●	●						
Plant Uptake				●					
Ingestion	●								
Dermal Contact	●		●						
Inhalation of VOCs	●		●						
Plant Uptake									
Inhalation	●	●	●						
Ingestion					●	●		●	
Ingestion	●				●	●	●		●

● = Exposure pathway potentially complete.

FIGURE 1. EXAMPLE CSM

Table 1. Soil Exposure Intervals

Receptor	Exposure Intervals (Soil)
Resident	0 – 10 ft bgs
Industrial/Commercial Worker	0 – 1 ft bgs
Construction Worker	0 – 10 ft bgs
Soil-to-Groundwater Migration	Depth of maximum detection
Ecological Receptors (non-burrowing)	0 – 1 ft bgs
Ecological Receptors (burrowing, deep rooted plants)	0 – 6 ft bgs

3.3 Background and Background Threshold Values (BTVs)

Whether conducting a human health or ecological risk assessment, determination of background concentrations is important to discern whether detected constituents are reflective of past operations or are present due to natural or other anthropogenic causes. Background metals and inorganics detected in soil can prove problematic for risk assessment purposes, as these elements may be naturally occurring metals and due to past historical operations. Inorganics and even some organics, such as polycyclic aromatic hydrocarbons (PAHs) and dioxin/furans, may also be present due to regional anthropogenic contributions, such as from runoff of asphalt, nearby industrial operations, or regional forest fires.

A background level is "the concentration of a hazardous substance that provides a defensible reference point that can be used to evaluate whether or not a release from the site has occurred. The background level should reflect the concentration of the hazardous substance in the medium of concern for the environmental setting on or near a site. A background level does not necessarily represent pre-release conditions, nor conditions in the absence of influence from source(s) at the site" (US EPA, 1992). It is important to note that background levels do not have to reflect pristine conditions (US EPA,).

A site attribution analysis looks at site concentrations and compares them to background or ambient levels. Constituents that are not present due to site activities, but are representative of background, are not carried forward into the risk analysis. Therefore, determination of background is a critical step to ensure the risk assessment reflects conditions as a result of site activities and avoids an overly conservative estimation of risk. Three types of background data are available for use as described below and include:

- Default county specific BTVs,
- Surrogate BTVs, and
- Site-specific BTVs.

<u>Establishment of a site-specific BTV is highly encouraged</u> for 1) areas where metals may have a greater range of ambient levels than the default county specific BTVs; 2) site data are above the background level(s) and a statistical comparison is needed (See Section 4); 3) differentiation of concentrations based on geology and/or depth in soil is needed, or 4) other project-specific needs

(e.g., geochemical evaluations or impact from other sources). In addition, site-specific background data will be needed if using incremental sampling (refer to Section 4.1.4).

As the BTVs represent an upper limit value, use of a 95 percent upper confidence level of the mean (95% UCL) is not appropriate in screening against the BTVs. The maximum detected site concentration should be used as the initial exposure point concentration (EPC), and if the site maximum is below the default BTV, the metal may be dropped as a constituent of potential concern or constituents of potential ecological concern (COPC/COPEC). Note – refer to Sections 4.2 and 9.2.2 on EPCs and Sections 4.1 and 9.2.1 on identifying COPCs/COPECs.

3.3.1 Default County-specific BTVs

Arsenic may be problematic in risk assessments since the RSLs are significantly lower than typical background soil concentrations in Utah. For many sites, especially Environmental Cleanup Sites located in highly developed areas, determination of background levels for arsenic as well as other metals may prove difficult. The DWMRC has established default county specific BTVs for RCRA metals plus a couple of other commonly detected metals. Note that insufficient data were available to derive default BTVs for selenium and silver. The data for the BTVs were taken from various databases, including United States Geological Society (USGS) databases, where specific global positioning system (GPS) locations, sampling methodology, and analytical methods were known. Only data that had the same sampling methodology and analytical method were compiled in deriving the county specific BTVs. These county specific BTVs may be used in lieu of site-specific background data and may be used for surface and subsurface data (up to 10 feet bgs).

The county specific BTV listed in Table 2 may be applied when using discrete or composite data. Variation of metals across Utah is well documented and it is recognized that metals may be present in site backgrounds at levels higher than the default BTVs listed in Table 2.

3.3.2 Surrogate BTVs

An alternative to the default county specific BTVs is to use surrogate BTVs. These are site-specific BTVs that were derived from a facility located within a six-mile radius of your site. The surrogate BTVs must have been derived following the methodology outlined in Section 3.4.3 and have been previously approved by the DWMRC.

3.3.3 Site-specific BTVs

If site-specific BTVs are to be developed, they should be established during site characterization and in accordance with the following sections. However, it is acceptable to initially go with the default county BTVs. Site-specific background levels may be derived if the site does not meet the county-specific level. It is acceptable to take a step-wise approach to background, as it may potentially be both time and cost effective.

3.3.3.1 Soil

Sample size, sample locations, as well as other site-specific parameters for background data sets should be outlined in a site characterization/facility investigation work plan. Guidance on the process of conducting a background soil study is beyond the scope of this document. However, the following criteria are representative of a defensible background data set:

- Includes enough data (minimum of 8) for statistical analyses;
- Free of statistically determined outliers;
- Reliably representative of the variations in background media (e.g., soil types or groundwater horizons);
- Collected from areas where there is no potential for site contamination based on site history;
- Collected from areas that are upwind of the site;
- Collected from soil types that are lithologically comparable to the samples that will be collected from contaminated areas; and
- Collected from depths that correspond to the exposure intervals that will be evaluated during human and ecological risk assessments.

An adequate sample size will likely capture a reliable representation of the background population while meeting the minimum sample size requirements for calculating BTVs and conducting hypothesis testing. US EPA (2020) recommends 8-10 samples for each background data set, but more are preferable. While it is possible to calculate BTVs with small data sets containing as few as three samples, these results are not considered representative and reliable enough to make cleanup or remediation decisions. Therefore, a minimum sample size of eight (8) is required to calculate BTVs and conduct hypothesis testing. The size of the background area and size of the site or facility under study should also be considered in determining sample size. That is, if the background and site areas are relatively large, then a larger background data set (e.g., > 8 samples) should be considered (US EPA, 2020). Background soil data are often grouped according to depth (e.g., surface vs. subsurface) or soil type. It is important to note that the minimum sample size of 8 should be met for each grouping of data to compute BTVs for each soil horizon or soil type.

Determination of BTVs should be conducted using current ProUCL software and guidance or other software as approved by DWMRC. In general, soil BTVs should be based on 95% upper tolerance limits (UTLs). Exceptions can occur on a case-by-case basis when the estimated 95% UTL is greater than the maximum detected concentration. This may be an indication that the 95% UTL is based on the accommodation of low-probability outliers (which may or may not be attributable to the background population) or highly skewed data sets and/or possibly inadequate sample size. In these cases, it may be warranted to evaluate the possibility of additional potential outliers or collection of more data. In lieu of collection of additional data to resolve the elevated UTL issue, the maximum detected concentration should be used as the BTV.

3.3.3.2 Surface Water Bodies

For moving surface water, such as a river or stream, background may be determined from upstream locations. Generally, sediment samples are preferred over aqueous samples for evaluating the surface water pathway because sediments are more likely to retain contaminants. In general, aqueous samples might represent current release conditions, whereas sediment samples might exhibit historical release conditions. Simple surface water pathway sampling generally consists of taking a minimum of one Probable Point of Entry (PPE) sample and one upstream background sample. If the surface water pathway has multiple PPEs, multiple background samples may be needed. The number of background samples collected depends on the complexity of the path of the surface water body. The presence of multiple tributaries upstream with multiple potential sources would require collecting multiple background samples in each tributary to differentiate the potential contribution of contamination from off-site sources [US EPA Office of Solid Waste and Emergency Response (OSWER) Directives 9345.1-05 and 9345.1-07].

Establishing a background level for a static water body (lake or spring) should be discussed with the DWMRC. For ponds and lakes, background samples may be collected near the inflow to the water body if it is not influenced by the site. A pond near the site may be selected for background sampling if it exhibits similar physical characteristics to the on-site pond. For large ponds and lakes, background samples may be collected from the water body itself, but as far away as possible from the influence of the PPE and other potential sources (OSWER Directive 9345.1-07).

3.3.3.3 Groundwater

Additional consideration may be given to determining background levels for groundwater, depending on intra-well or inter-well comparisons. In general, background samples should be collected from nearby wells that are not expected to be influenced by the source of contamination or by nearby sites. If there are other sites or potential local sources of groundwater contamination, additional background samples should be collected where possible to differentiate their contribution from that of the site under investigation (OSWER Directive 9345.1-05).

Aqueous release and background samples must be collected from comparable zones (e.g., saturated zone) in the same aquifer and, where possible, should be collected during the same sampling event. <u>Non-filtered samples should be collected to represent total dissolved metals</u>.

Table 2. Default County Specific BTVs

County	Arsenic	Barium	Cadmium	Chromium[1]	Lead	Mercury	Nickel	Zinc	Thallium	Copper
Beaver	18	724	0.4	58	33	0.05	22	110	0.9	24
Box Elder	9	631	0.9	61	41	0.02	27	96	1.0	28
Cache	9	606	1.0	46	30	0.10	14	95	0.975	24
Carbon	15	664	0.5	45	20	0.02	14	70	0.5	19
Daggett	7	380	0.1	22	11	0.01	7	31	0.3	7
Davis	13	454	0.4	30	25	0.02	10	47	0.35	18
Duchesne	22	749	0.5	55	25	0.02	14	73	0.6	25
Emery	14	508	3.0	80	15	0.03	35	102	1.0	36
Garfield	10	840	0.4	246	29	0.04	18	98	1.5	30
Grand	17	721	2.8	65	26	0.04	32	106	1.2	28
Iron	16	710	0.5	106	42	0.04	12	80	1.3	34
Juab	29	509	0.5	41	40	0.02	15	69	1.6	18
Kane	17	417	0.2	52	18	0.02	22	61	0.6	17
Millard	22	580	0.5	50	26	0.02	25	76	0.6	23
Morgan	7	508	0.7	42	27	0.03	18	83	0.5	26
Piute	4	937	0.3	44	22	0.03	17	98	0.5	29
Rich	5	818	0.5	63	21	0.02	18	69	0.6	22
Salt Lake	27	521	0.4	51	60	0.03	10	73	1.1	75
San Juan	6	469	0.4	42	21	0.02	19	53	0.5	39
Sanpete	9	400	0.7	49	27	0.02	19	84	0.6	14
Sevier	9	862	0.4	59	33	0.01	18	144	0.7	71
Summit	3	610	0.7	30	22	0.02	12	43	0.4	13
Tooele	25	581	1.1	53	64	0.05	25	111	0.9	64
Uintah	29	1060	0.5	66	26	0.04	25	75	0.5	22
Utah	14	376	0.9	36	20	0.02	12	57	0.475	17
Wasatch	8	1508	0.4	36	36	0.58	12	72	0.475	15
Washington	23	522	0.3	73	29	0.02	39	126	0.7	43
Wayne	8	477	0.4	37	18	0.01	10	58	0.6	16
Weber	8	400	0.4	31	15	0.02	11	49	0.4	16

Notes:
All data in units of milligrams per kilogram (mg/kg)
Sufficient numbers of detects were not available to derive a BTV for selenium and silver.

[1] Chromium is presented as total chromium. If hexavalent chromium is a COPC, and speciation of chromium is needed, additional site-specific background values based on valence state may be required.

4.0 IDENITIFICATION OF COPCS/COPECS AND EXPOSURE POINT CONCENTRATIONS (EPCS)

COPCs and COPECs are any substance likely to be present in environmental media affected by a release and past site history. Identification of COPCs/COPECs should begin with existing knowledge of the process, product, or waste from which the release originated. For example, if facility operations deal primarily with pesticide manufacturing, then pesticides should be considered COPCs/COPECs. Contaminants identified during current or previous site investigation activities should also be evaluated as COPCs/COPECs. A site-specific COPC/COPEC list for soil may be generated based on maximum detected (or, if deemed appropriate by DWMRC, the 95% UCL value) concentrations (US EPA 2002b) and a comparison of detection/quantitation limits for non-detect results to the DWMRC SSLs. This list may be refined through a site-specific risk assessment.

An initial reduction of COPCs/COPECs by a simple comparison to the RSL is not acceptable. All contaminants deemed present due to site activities must be carried forward as COPCs/COPECs for comparison to background, regardless of if the maximum detected concentration is less than the RSL. For example, if a contaminant has a concentration less than the RSL, the contaminant may not be dropped as a COPC prior to evaluating background and/or cumulative risk. Further, other lines of evidence, such as frequency of detection may not be used in the initial determination of COPCs/COPECs but may be addressed in the uncertainty discussion and/or revised assessment.

For the initial screening assessment, duplicates should be handled using the higher concentration as the EPC; averaging of the data is not appropriate for the initial screening assessment. If a refined EPC is needed, the duplicates may be averaged.

4.1 Soil/Sediment

4.1.1 Organics and Chemicals without Background Data

Per US EPA guidance (US EPA 1989), if there is site history to indicate a chemical was potentially used/present at a site, or if there is insufficient site history to demonstrate that a chemical could not be present, and the chemical was detected in at least one sample, this chemical must be included as a COPC/COPEC and evaluated in the screening assessment. Frequency of detection or other lines of evidence may not be used to eliminate a chemical as a COPC/COPEC if there is history to indicate it is potentially present due to site activities, although these lines of evidence may be addressed in the uncertainty analysis for the risk assessment.

It is possible a site may have been impacted by other anthropogenic sources. As one line of evidence to help assess site impact to certain organics, development of baseline levels for organics may be appropriate. For example, PAHs may be present due to runoff from nearby paved/industrial structures, and dioxins/furans may be ubiquitous due to natural fires. If there are other potential sources of organics, the site characterization work plan should include sampling to determine baseline organic levels. In lieu of baseline sampling, additional lines of evidence may be required to justify the organics as not being site related. Factors to consider are proximity to other source areas for contamination (e.g., paved roads), magnitude of detection, spatial variability.

4.1.2 Organics and Chemicals with Background Data

For organics and inorganics where background data are available, a comparison of site concentrations to appropriate background concentrations may be conducted prior to evaluation against SSLs. Those organics and inorganics that are present at levels indicative of natural background may be eliminated as COPCs/COPECs and not carried forward to the screening assessment calculations. Comparison to background must be conducted following current US EPA Guidance and as outlined following the tiered approach below.

4.1.3 Discrete Soil Sampling

For discrete data, the following tiered approach should be applied for determining if site data are reflective of background conditions.

<u>Step 1.</u> Compare the maximum detected site concentration to the site-specific background reference values (upper tolerance limit or upper threshold value) determined for each soil type and soil depth at the site. If the site maximum is less than the background reference value, it is assumed that the site concentrations are representative of background and the metal/inorganic/organic is not retained as a COPC/COPEC. If there is no background value for a constituent, then the constituent must be retained as a COPC/COPEC.

<u>Step 2:</u> If the maximum site concentration is greater than the background reference value, then a two-sample hypothesis test should be used to compare the distributions of the site data to the distributions of background data to determine if site concentrations are elevated compared with background. A simple comparison to the range of background is not acceptable. Background can vary across a site (especially larger sites) and not allow for soil type to be taken into consideration. Further, a range can mask low level contamination. <u>Comparisons of site data to the range of background values or comparison to the maximum detected concentration in the background data set may not be used as a line of evidence to eliminate site constituents as COPCs/COPECs.</u>

The most recent version of US EPA's ProUCL statistical software should be used for hypothesis testing. ProUCL should also be used to determine the most appropriate test (parametric or nonparametric) based on the distribution of the data. Appropriate methods in ProUCL will also be used to compute site-to-background comparisons based on censored data sets containing non-detect values. A review of graphical displays (e.g., box plots and Q-Q plots) may also be provided in addition to the results of the statistical tests to provide further justification in determining whether site concentrations are elevated compared with background. These graphical plots can also be generated by ProUCL software.

Note that the above two-sample test can only be used for site data sets that have sufficient samples (i.e., n ≥ 8) and number of detections (greater than 5 detected observations is preferred). While a minimum of 10 background data samples are now required, there may be sites where background has been previously determined from a data set that contains fewer than 10 samples. As stated in the current version of ProUCL User's Guide (US EPA, 2020), hypothesis testing is only considered to be reliable with sufficient sample size (n ≥ 8) and frequency of detection.

If there are not at least eight samples in the site data set and at least five detections, then the site maximum detected concentrations will be compared to the corresponding background value (i.e., 95% upper tolerance limit) as noted in Step 1 or additional data must be collected to conduct a two-tailed test.

Step 3: Additional lines of evidence may be used to justify exclusion of a constituent as being site related, such as site history, high percentage of non-detects, etc. However, these lines of evidence must be based on a sufficient number of samples to adequately define nature and extent of contamination and to clearly delineate potential hotspots. For areas where a hotspot may be present, additional actions are required (such as sampling and/or corrective actions) and the constituent(s) must be retained as a COPC/COPEC. Comparison of site data to regional data (such as USGS) databases not specific to the site and simple comparison to a range of data or quartiles are not acceptable lines of evidence.

4.1.4 Incremental Sampling Method

If incremental sampling (ISM) data are to be collected, a similar process as described above comparing site data to background may be conducted. However, the ISM BTVs must also be derived using the ISM approach. ISM data may not be compared to BTVs based on discrete sampling. ProUCL is being updated to include hypothesis testing and calculation of statistically derived upper thresholds for ISM data. However, until such statistical evaluations are available in ProUCL, the following approach should be conducted for comparing site ISM to background ISM data:

- If the site ISM maximum detected concentration is less than the background minimum ISM, the constituent may be considered present at ambient concentrations and does not require retention as a COPC/COPEC.
- If the site ISM maximum falls within the range of background ISM, a qualitative discussion and lines of evidence must be provided to justify exclusion of the constituent as a COPC/COPEC. Evaluation of triplicate data should be included. Note: collection of field triplicates or replicates helps to evaluate the effectiveness of the ISM sampling and to ensure more reliable estimates of the mean. ISM samples collected in triplicate, means soil aliquots are collected thrice following the same sample pattern within the same decision unit.

If the site ISM maximum is greater than the background ISM minimum, the constituent must be retained as a COPC/COPEC.

4.2 Exposure Point Concentration (EPCs)

4.2.1 Soil/Sediment

For the initial screening risk assessment, the maximum detected concentrations shall always be used as the EPCs. If using the maximum detected concentrations excess risk is a result, further assessment is warranted (see Section 5) using refined EPCs [e.g., 95 percent upper confidence limit (UCL)). US EPA (1989) recommends using concentration to represent "a reasonable estimate of the concentration likely to be contacted over time". US EPA's (1992b) *Supplemental Guidance to RAGS: Calculating the Concentration Term* states that, "because of the uncertainty associated with estimating the true average concentration at a site, the 95 percent UCL of the arithmetic mean should be used for this variable."

4.2.1.1 Discrete Samples

Upper confidence limits should only be calculated for data sets that meet the US EPA (2020) minimum requirements for calculating UCLs. The minimum requirements for calculating UCLs are: 1) each data set must contain at least eight samples (i.e., $n \geq 8$) for the analyte being evaluated; and 2) there must be a minimum of five detections (i.e., ≥ 5 detected observations) for the analyte being evaluated. Although it is possible to calculate UCLs with small datasets (i.e., $n \leq 8$) and low frequencies of detection (i.e., < 5 detected observations), these estimates are not considered reliable and representative enough to make defensible and correct cleanup and remediation decisions (US EPA, 2020). Therefore, UCLs should only be calculated for data sets that meet the minimum requirements for the calculation of UCLs. For datasets with less than four detects or datasets with less than eight samples and a low level of detection (less than 10%), the median concentration may be used as the EPC.

- UCLs should be calculated using the most current version of US EPA's ProUCL statistical software package. Statistical methods for calculating UCLs are dependent on the distribution of the data. Therefore, when calculating UCLs, ProUCL should be used to perform statistical tests in order to determine the distribution of the site data. If assumptions about the distribution cannot be made, then nonparametric methods can be utilized. ProUCL recommends a computational method for calculation of the 95% UCL based on the assumed distribution.
- Using parametric and nonparametric methods, ProUCL will typically return several possible values for the UCL. Professional judgment should be used in selecting the most appropriate UCL; however, the UCL recommended by ProUCL is based on the data distribution and is typically the most appropriate value to be adopted as the EPC for use in risk assessments. It is important to note that the UCL should not be greater than the maximum detected concentration.
- Non-detects (censored datasets) should be evaluated following the appropriate methodology outlined in the most recent version of US EPA's ProUCL Technical Guide. Currently, the ProUCL Technical Guide indicates that the Kaplan-Meier (KM) method

> yields more precise and accurate estimate of decision characteristics than those based upon substitution and regression on order statistics. Use of one-half the minimum detection limit (MDL) or sample quantitation limit (SQL), or other simple substitution methods, are not considered appropriate methods for handling non-detects.

4.2.1.2 ISM Samples

The Interstate Technology & Regulatory Council (ITRC) 2020 guidance states that "In theory, all of the UCL methods that are applied to discrete sampling results can also be applied to ISM. In practice, however, because fewer than eight replicate ISM samples are likely to be collected for a decision unit (DU), fewer options are typically available to calculate a UCL compared with discrete sampling data." For those DUs where there are eight or more sample units (SUs), the current version of US EPA's ProUCL should be used to calculate a UCL and the recommended UCL (if less than the maximum) used in the risk assessment. Triplicates should be conservatively represented in the calculation of the UCL as the maximum of the detected results, which will bias the UCL high.

For those DUs where there are three (3) to eight (8) sample units (SUs), Interstate Technology Regulatory Council (ITRC, 2020) and US EPA (2020) guidance indicate that not all of the UCL calculation methods provided in ProUCL are reliable. Instead, ITRC (2020) guidance indicates that either the Student's-t UCL or the Chebyshev UCL be used for DUs with 3-8 SUs. For these DUs (with 3-8 SUs), ProUCL should be run and the Student's t UCL used as the EPC if the data are determined to be normally distributed. If the data are determined to not be normally distributed, the 95% Chebyshev UCL should be used as the UCL. Triplicate data should be represented by the maximum of the detected values.

For DUs with 1-2 SUs, a UCL should not be calculated; the EPC should be the maximum detected concentration.

For chemicals with both non-detected results and detected results, the KM based UCLs (using Student's-t or Chebyshev) should be used, as recommended by US EPA (2020) guidance.

4.2.2 Groundwater EPCs

A workgroup comprised of members of two US EPA forums, the OSWER Human Health Regional Risk Assessors Forum and the Groundwater Forum, deliberated about how to determine groundwater exposure point concentration (GWEPC). The final consensus on how to determine groundwater exposure point concentration was published in a memorandum titled Determining Groundwater Exposure Point Concentrations, Supplemental Guidance, March 11, 2014. The objective of the memorandum was to reduce unwarranted variability in the exposure assumptions used by Regional Superfund staff to characterize exposures to human populations in baseline risk assessment.

UAC R315-101 has adopted this guidance in determining the GWEPC for evaluating risks from exposure to contaminated groundwater at all sites. GWEPC is a conservative estimate of the average chemical concentration in groundwater at a potential location and point in time. Note that ecological receptors are typically not exposed to groundwater. Groundwater that surfaces (such as a spring) is evaluated as surface water in an ecological risk assessment.

Data to be used in GWEPC calculations must be recent and from the core of the groundwater plume. For current risk, actual data should be used and is always preferred. While it is typically not appropriate to use modeled concentrations in GWEPC calculations for current risk, model data may be appropriate for assessing future risk. Representative samples should be from the core of the plume, where the three-dimensional core/center of the plume is the zone of highest concentration of each contaminant within a delineated groundwater plume. If a groundwater CSM has identified seasonal or temporal influences (e.g., drought patterns), the recommendation is to use data collected during times of higher detected concentrations.

If seasonality or temporal influences are not an issue, the recommendation is to use data collected from the latest two rounds of sampling for each selected well and preferably data collected within the last year to be representative of current site conditions. If data are not available within two-years of the assessment, additional groundwater data will be required to be collected to represent current conditions. Note: refer to the Unified Guidance for evaluating seasonal trends in data.

Non-detects are frequently an issue; consult the ProUCL Technical Guide (US EPA, 2022) on how to handle non-detects in the data set.

The following factors are to be considered when evaluating whether data are representative of current condition:

1. Movement - the faster the flow, the less representative older data will be to evaluate risk,
2. Fate and transport – the higher the attenuation rates, the less representative older data may be to evaluate future risks.

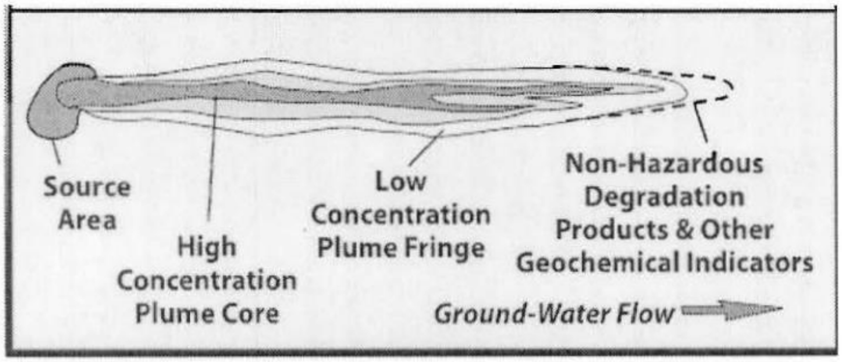
If there exists more than one aquifer, the recommendation is to consider each aquifer separately when calculating an EPC. There should be one EPC for each aquifer. If monitoring network provides sample concentration from multiple sample depths at a given location, the recommendation is to use the highest detected concentration from such samples at each location to calculate a GWEPC for each aquifer.

Data needs for site characterization focuses on the nature and extent of contamination. However, data needed for a GWEPC calculation focuses on the core or center of the contaminated plume. For groundwater there is the need to adequately characterize the entire plume to be able to identify the core of the plume which is distinguished by higher concentration levels when compared to the lower concentration levels at the fringes of the plume.

For sites that have comingled plumes resulting from multiple sources, the aggregate risk needs to be evaluated based on the consideration of the combined effects, from each of the contaminants present. Data from a minimum of three wells in the core of the plume is recommended for calculations. GWEPC is calculated as the 95% UCL of the arithmetic mean concentration for each contaminant. The US EPA ProUCL is generally recommended for such calculations. It is desirable to use at least 10 data points for each contaminant, e.g., five wells and two rounds of data representative of current conditions equate to 10 data points to compute a 95% UCL.

If the computed 95% UCL is greater than the maximum detected concentration, the recommendation is to default to the maximum detected concentration for that contaminant. If less than three wells are within the core of the plume, the recommendation is to default to the maximum detected concentrations as the EPC for that contaminant and discuss this specifically in the uncertainty section of the risk assessment.

For an example of the plume core figure, refer to Figure 2 (from the Unified Guidance).



Idealized Plan View of a Groundwater Contaminant Plume for Purpose of Distinguishing the "Core" from Fringe Areas

Figure 2. Plume Core Figure (US EPA, 2009)

Well Types

Sampling data from monitoring wells are the only data acceptable for use in GWEPC calculations. If modeled data are to be used for GWEPC calculations, the data should be approved by the DWMRC prior to use.

- Monitoring wells in the core of the plume are the preferred source of data in GWEPC calculations for the purposes of characterizing a reasonable maximum exposure (RME) condition. There must be documentation that the wells have been properly constructed and maintained.
- Temporary well data such as from a hydropunch are not recommended for use in the calculations of GWEPC because the results are not reproducible. The exception may be a site-specific condition where temporary wells may be the only wells in the core of the plume. DWMRC approval of data from a temporary well is required prior to use.
- Piezometer data may or may not be acceptable for use in GWEPC calculations depending on the details of their construction. DWMRC approval of data from the piezometer is required prior to use.

Data Quality to be Addressed

In addition to well types, the following factors must be considered when evaluating data for inclusion in a data set for GWEPC development.

- Detection limits assure that laboratories can meet the Maximum Contaminant Level (MCL) and/or the tap water RSLs.
- Turbidity levels of samples must be stable and as low as possible, and generally less than 5-10 nephelometric turbidity units (NTUs) prior to sampling. If turbidity levels cannot be stabilized or adequately reduced, additional well development or well replacement may be considered before sample collection.
- Filtered vs. Unfiltered. Unfiltered data (i.e., total metals) are required for use in EPC calculations. It is noted that there are occasions where filtered sample data are needed, such as for geochemical modeling.
- All potential COPCs, including fate and transport process of VOCs breakdown products, non-aqueous phase liquid (NAPL), metals, the potential presence of contaminants of emerging concern, must be considered during sampling and analysis.

4.2.3 Non-detects

ProUCL Technical Guide should be consulted for handling non-detects. In general, ProUCL follows regression on order statistical (ROS) tests, where both detect and non-detect data are provided as inputs. However, to understand handling of non-detects, the following provides background on the evolution from simple substitution methods to more robust statistical evaluation of non-detects.

Measurements whose value are known only to be above or below a threshold are called censored data in the statistical literature. Censored data have been an integral part of several disciplines like medicine, industry, environment, etc., from which procedures have been developed to allow censored data to be incorporated into the computations of summary statistics, regression, and hypothesis tests. In the environmental field censored data are commonly encountered as values below a detection limit and are called “less thans” or “non-detects”. These values are not known exactly and because these low values are usually plotted to the left on a graph, nondetects are often labeled as “left-censored” with values lying somewhere to the left of the detection threshold.

In the environmental field, overly simplistic methods are commonly used when censored data are encountered. The first is to delete censored data values. Deleting the lowest values obviously produces biased results. The tests or statistics that result from this approach do not apply to the entire data set collected, but only to the part of the data on the higher end of the distribution. The argument for deletion is usually that the only interest is in detected observations. The second method commonly used for dealing with nondetects or censored data is to assign an arbitrary fraction of the detection limit. This is sometimes called “substitution” or “fabrication”. In several investigations, one-half the detection limit has been substituted for censored values. Substitution can induce a signal “not present” in the original data or result in a biased estimate of the mean with the highest variability.

Substitution of one-half the detection limit is not a reasonable method for interpreting censored data. The fundamental problem with this approach is in the statement that something is known that really is not known. This can be interpreted as the value of 0.5 times the detection limit is known about the observation, and not some other value below the detection limit. The true value may have been anywhere below the detection limit.

In truth, a great deal of information is available in censored data. If efficient methods are used, the information extracted from them is almost equal to that for data with single known values. The information is primarily contained in the proportion of data below the threshold values.

In summary there are three approaches for extracting information from datasets that include nondetects.

- ***Substitution or fabricating numbers***. These are widely used but have no theoretical basis and are not approved by DWMRC. Numerous papers have shown that substitution methods do not work well in comparison to other procedures.

- ***Maximum Likelihood estimation (MLE).*** MLE uses data both below and above the detection limit that are assumed to follow a certain distribution such as the lognormal. Parameters are computed that best match a fitted distribution to the observed values above each detection limit and the percentage of data below each limit. The most crucial consideration for MLE is how well data fits the assumed distribution. For small data sets there is often insufficient information to determine the validity and reliability of the assumed distribution and the estimated parameters.

  The US EPA ProUCL program computes summary statistics for raw as well as log-transformed data sets with and without nondetects observations. For uncensored data sets, mathematical algorithms and formulae used in the program are discussed. The ProUCL program also computes the MLE and the minimum variance unbiased estimates (MVVUEs) of the population parameters of normal, lognormal and gamma distributions. Critical values for gamma goodness of fit (GOF) for various decision statistics (e.g., UCL and BTVs) are computed using MLE estimates.

- ***Regression on Order Statistics (ROS).*** A regression line is fit to normal scores of the order statistics for uncensored observations and is used to fill in values imputed from the straight line for observations below the detection limit. ProUCL imputes nondetects based upon a hypothesized distribution such as gamma or lognormal distribution. The ROS method yields a data set of a certain size (N) which is used to compute the various summary statistics, and to estimate EPCs and BTVs.

5.0 RISK EVALUATION CRITERIA AND SPECIAL CONSIDERATIONS

Target risk and hazard levels for human health are risk management-based criteria for carcinogenic and noncarcinogenic responses, respectively, to determine: (1) whether site-related contamination poses an unacceptable risk to human health and requires corrective action or (2) whether implemented corrective action(s) sufficiently protects human health. If an estimated risk or hazard falls within the target range, the risk manager must decide whether or not the site poses an unacceptable risk. This decision should consider the degree of inherent conservatism or level of uncertainty associated with the site-specific estimates of risk and hazard. An estimated risk that exceeds these targets, however, does not necessarily indicate that current conditions are not safe or that they present an unacceptable risk. Rather, a site risk calculation that exceeds a target value may simply indicate the need for further evaluation or refinement of the exposure model.

For cumulative exposure for soil via ingestion, inhalation, and dermal pathways, DWMRC uses the US EPA RSLs based on a carcinogenic risk level of one-in-one million (1E-06) and a noncarcinogenic hazard quotient of 1.0. A carcinogenic risk level is defined as the incremental probability of an individual developing cancer over a lifetime, as a result of exposure to a potential carcinogen. The noncarcinogenic hazard quotient assumes that there is a level of exposure below which it is unlikely for even sensitive populations to experience adverse health effects.

<u>For the initial screening assessment, the RSLs may be used in lieu of calculating dose for exposure pathways</u>.

5.1 Hierarchy of Human Health Toxicity Data

The toxicity values used in calculating residential and composite worker (industrial/commercial) RSLs are based on chronic exposure while those for a construction worker will be based on subchronic. The default RSLs already have the preferred toxicity built into them.

However, the following hierarchy of toxicity data should be followed when refined assessments, to include target organ analysis, are conducted. The primary sources for the human health benchmarks follow the US EPA Superfund programs tiered hierarchy of human health toxicity values (US EPA 2003). Although the US EPA 2003 identified several third tiered sources, a hierarchy among the third-tier sources was not assigned by the US EPA. The- hierarchy of sources to be applied is as follows (US EPA, 2016a):

The below hierarchy should be followed when selecting target organs for a refined hazard assessment.

1) Integrated Risk Information System (IRIS) (US EPA, 2023) (www.epa.gov/iris),

2) Provisional peer reviewed toxicity values (PPRTVs) (https://www.epa.gov/pprtv),

3) Agency for Toxic Substances and Disease Registry (ATSDR) (http://www.atsdr.cdc.gov/) and minimal risk levels (MRLs) (http://www.atsdr.cdc.gov/mrls/index.asp),

4) California EPA’s Office of Environmental and Health Hazard Assessment values (CalEPA) (https://dtsc.ca.gov/assessing-risk/), and

5) Health Effects Assessment Summary Tables (HEAST) (US EPA 1997a).

5.2 Special Considerations

Special assumptions were also applied in determining appropriate toxicological data for certain chemicals.

5.2.1 Lead

The US EPA RSL Table recommends levels for lead, based on blood-lead modeling applied for the residential scenarios (Integrated Exposure Uptake Biokinetic Model, IEUBK) and industrial/construction workers (Adult Lead Methodology, ALM). If a site-specific screening level is needed, note that neither the IEUBK nor the ALM are appropriate for acute exposures. For short-term exposure less than 90 days, periodic exposure, or acute exposure, alternative modeling approaches should be applied (USEPA 2016).

Exposure to lead can result in neurotoxic and developmental effects. The primary receptors of concern are children, whose nervous systems are still undergoing development and who also exhibit behavioral tendencies that increase their likelihood of exposure (e.g., pica). These effects may occur at exposures so low that they may be considered to have no threshold and are evaluated based on a blood lead level [rather than an external dose as reflected in the reference dose/reference concentration (RfD/RfC) methodology]. Therefore, US EPA views it to be inappropriate to develop noncarcinogenic “safe” exposure levels (i.e., RfDs) for lead. Instead, US EPA’s lead assessment workgroup has recommended the use of the IEUBK model that relates measured lead concentrations in environmental media with an estimated blood-lead level for assessing risks to residential receptors (US EPA 2016h). The model is used to calculate a blood lead level in children when evaluating residential land use and in adults (based on a pregnant mother’s capacity to contribute to fetal blood lead levels). However, US EPA recommends the use of the ALM for adults in evaluating occupational scenarios at sites where access by children is reliably restricted (US EPA 2016h). The DWMRC soil concentration for each receptor that would not result in an estimated blood-lead concentration of 10 micrograms per deciliter (µg/dL) or greater (residential adult of 400 mg/kg and industrial and construction worker of 800 mg/kg) can be calculated. If the screening levels for lead are exceeded, it is recommended that site-specific bioavailability of lead using the US EPA’s *in-vitro* bioaccessibility assay for lead be used to refine the screening levels. Note that if site-specific screening levels are defined, the exposure to a typical/hypothetical child resident must not have an estimated risk exceeding 5%, or a resulting blood lead level of more than 10 µg/dL (US EPA 2016h).

The ALM along with site-specific exposure parameters may be used if a site-specific screening level for a landfill worker is proposed as part of a contained-out request. Refer to the DWMRC guidance on contained-out (DWMRC, 2022) for more detail. https://documents.deq.utah.gov/waste-management-and-radiation-control/hazardous-waste/DSHW-2020-015943.pdf

5.2.2 Chemical Agents

Chemical agents may be suspected to be present in soil at some sites. RSLs are not available for chemical agents; therefore, chemical agents will be evaluated by comparing EPCs with the health-based screening levels (HBSLs) provided in the US Army Public Heath Command (USAPHC, 2011) the report entitled *Chemical Agent Health-Based Standards and Guidelines Summary Table 2: Criteria for Water, Soil, Waste, as of July 2011* or as updated. The chemical agent data are updated from the Oak Ridge National Laboratory (ORNL) *Reevaluation of 1999 Health Based Environmental Screening Levels (HBESLs) for Chemical Warfare Agents.* The HBSLs shown in Table 3 were calculated using standard US EPA methodology and exposure and represent RME conditions.

Table 3. Health-Based Screening Levels (HBSLs) for Chemical Agents

Agent	Residential HBSL (mg/kg)	Industrial Worker HBSL[a] (mg/kg)
HD (Mustard)	0.01	0.3
L (Lewisite)	0.3	3.7
GA (Tabun)	2.8	68
GB (Sarin)	1.3	32
GD/GF (Soman/Cyclosarin)	0.22	5.2
VX	0.042	1.1

[a] Note: Industrial worker HBSLs were not converted from units of g/kg as listed in USAPHC, 2011 (assume unit error in source document)

The HBSL for HD is the only HBSL that is based on a carcinogenic endpoint. All other agents listed in Table 3 have HBSLs that are based on noncarcinogenic endpoints. Risks and hazards will be calculated for chemical agents and added to the cumulative risk and hazard index calculations for respective scenarios.

The industrial worker is synonymous with an indoor worker and the industrial screening levels. Residential HBSLs are considered protective of a construction worker scenario as both scenarios require evaluation of soil from zero to 10 ft bgs (Table 1).

Risk-based screening levels have not been established for chemical agents in drinking water. The U.S. Army Center for Health Promotion and Preventive Medicine (USACHPPM, 1999) evaluated the potential for groundwater contamination from chemical agent and found that the groundwater contamination scenario was not plausible due to hydrolysis, degradation, and dilution of the agents. However, if toxicity data become available to qualitatively address this pathway, this document will be updated to reflect the methodology and data.

5.2.3 Chromium

Elemental chromium (Cr) is naturally present and considered stable in the ambient environment in one of two valence states: chromium (III) and chromium (VI). Chromium (III) occurs in chromite compounds or minerals and concentrations in soil/groundwater result from the weathering of minerals. Chromium (III) is the most stable state of environmental chromium; chromium (VI) in the environment is man-made, present in chromate and dichromate compounds, and is the more toxic of the oxidation states (RAIS, 1992).

The oxidation state of Cr has a significant effect on its transport and fate in the environment. The equilibrium distribution of the Cr between the two oxidation states is controlled by the reduction/oxidation potential (redox) environment. Oxidation depends on a variety of factors and is a function of pH and the rate of electron exchange, or standard reduction potential (Eh). Chromium (VI) is converted to the less toxic and much less mobile form of chromium (III) by reduction reactions. The corresponding oxidation of chromium (III) to chromium (VI) can also occur under oxidizing conditions.

The degree to which chromium (III) can interact with other soil constituents is limited by the fact that most chromium (III) is present in the form of insoluble chromium oxide precipitates rendering chromium (III) relatively stable in most soils. Oxidation of chromium (III) to chromium (VI) can occur under specific environmental conditions with influencing factors including the soil pH, chromium (III) concentration, presence of competing metal ions, availability of manganese oxides, presence of chelating agents (i.e., low molecular weight organic compounds), and soil water activity. Chromium (III) oxidation is favored under acidic conditions, where the increased solubility of chromium (III) at lower pH enables increased contact with oxidizing agents. Aside from decreasing soil pH, chromium (III) solubility is enhanced by chelation to low molecular weight compounds such as citric or fulvic acids. Conversely, factors influencing the reduction of chromium (VI) to chromium (III) in soil include soil pH, the presence of electron donors such as organic matter or ferrous ions, and soil oxygen levels (CEQG, 1999). Chromium reducing action of organic matter increases with decreasing pH.

Figure 3 (TCEQ, 2002) shows a generalized Eh-pH diagram for the chromium-water system. Chromium (III) exists over a wide range of Eh and pH conditions [e.g., Cr^{3+}, $Cr(OH)_3$, and CrO_2^-] while chromium (VI) exists only in strongly oxidizing conditions (e.g., $HCrO^-_4$ and CrO^2_4).

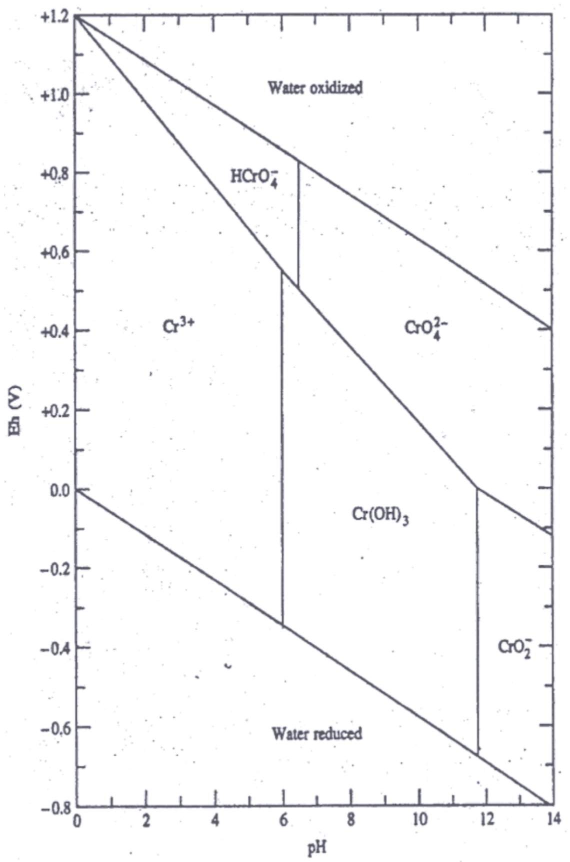


Figure 3. Eh-pH Diagram for Chromium

Generally, groundwater containing high concentrations of chromium is more likely to be comprised of chromium (VI) than chromium (III) because chromium (III) is more likely to have precipitated as Cr_2O_3 x H_2O and, to a lesser extent, adsorbed. Chromium (VI) is highly mobile in groundwaters with neutral to basic pH. In acidic groundwaters chromium (VI) can be moderately adsorbed by pH-dependent minerals such as iron and aluminum oxides. Under favorable conditions, chromium (VI) reduces to chromium (III) rapidly via ferrous iron, organic matter, and microbes. The oxidation of chromium (III) to chromium (VI) by dissolved oxygen and monoxides is kinetically slower (TCEQ, 2002). Redox conditions and pH dominate Cr speciation and thus are important parameters required for assessment of groundwater data.

The RSL tables no longer contain risk-based screening levels for total chromium (except for air). The US EPA deleted the total chromium values due to uncertainty associated with the previously applied ratio of trivalent to hexavalent chromium. The concern was that an assumed ratio (1:6) had the potential to both under- and over-estimate risk.

For sites where chromium is to be included for analysis, a tiered process should be applied. If a review of site-specific geology and geochemistry indicates conditions are not favorable for the possible presence of chromium (VI), additional sampling may be conducted to demonstrate that total chromium is representative of only chromium (III). If site-specific speciated data demonstrate the absence of chromium (VI) in background and/or site soil, the use of the chromium (III) SSLs may be warranted. However, if there is site history sufficient to identify chromium (VI) as a potential site contaminant, such as the site previously housed a plating operation or soil/water chemistry may allow for speciation, analyses of media (soil and/or groundwater) should include hexavalent and total chromium in the analytical suite along with determination of pH (water samples) and Eh to assess chemical state. Comparison of the species-specific data can be compared to representative background concentrations.

If site history does not indicate a known source for chromium (VI), the data (soil and/or groundwater) should be analyzed for total chromium. If the site levels of total chromium are within background, no additional analyses would be required (chromium would drop from the risk assessment as a constituent of concern). However, if the total chromium concentrations are statistically different (using a 95% confidence level) from background for soil or if chromium appears to be a site contaminant in groundwater, a two-tiered approach should be applied.

A more detailed review of the site history should be conducted to see if there were any potential sources for chromium (VI) or any processes that could have resulted in an alteration of speciation (such as introduction of acids). If there is no potential source, or it does not appear that any other chemicals or contaminants are present that may have altered the speciation of Cr, and this can be documented, no additional analyses will be required, and the data may be evaluated as total chromium.

<u>If there is a potential source for chromium (VI) or the data are statistically different (using a 95% confidence level) from background, additional sampling should be conducted to determine speciation. The species-specific data will then be compared to the trivalent and hexavalent chromium EPA RSL screening levels</u>.

5.2.4 Dioxin/Furans

<u>Dioxins/Furans</u>. Toxicity data for the dioxin and furan congeners were assessed using the 2005 World Health Organization's (WHO) toxicity equivalency factors (TEF) (Van den berg, et al 2006) and are summarized in Table 4. When screening risk assessments are performed for dioxins/furans at a site, the TEFs in Table 4 should be applied to the analytical results and summed for each sample location; the sum, or toxicity equivalent (TEQ) as calculated using Equations 1 and 2, should be compared to the EPA RSL 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD).

$$TEF_i \times C_i = TEC_i \qquad \text{Equation 1}$$

$$\sum TEC_i = TEQ \qquad \text{Equation 2}$$

Where:

TEF_i = Congener-specific toxicity equivalency factor (Table 4)
C_i = Congener-specific concentration
TEQ = Toxicity Equivalent Concentration

Table 4. Dioxin and Furan Toxicity Equivalency Factors

Dioxin and Furan Congeners	TEF
Chlorinated dibenzo-p-dioxins	
2,3,7,8-TCDD	1
1,2,3,7,8-PeCDD	1
1,2,3,4,7,8-HxCDD	0.1
1,2,3,6,7,8-HxCDD	0.1
1,2,3,7,8,9-HxCDD	0.1
1,2,3,4,6,7,8-HpCDD	0.01
OCDD	0.0003
Chlorinated dibenzofurans	
2,3,7,8-TCDF	0.1
1,2,3,7,8-PeCDF	0.03
2,3,4,7,8-PeCDF	0.3
1,2,3,4,7,8-HxCDF	0.1
1,2,3,6,7,8-HxCDF	0.1
1,2,3,7,8,9-HxCDF	0.1
2,3,4,6,7,8-HxCDF	0.1
1,2,3,4,6,7,8-HpCDF	0.01
1,2,3,4,7,8,9-HpCDF	0.01
OCDF	0.0003

5.2.5 Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs consist of chemicals that belong to the same family and exhibit similar toxicological properties. However, they differ in their degree of toxicity and a relative potency factor (RPF) is sometimes applied to adjust the oral slope factor or inhalation unit risk factor and basing the RPF on benzo(a)pyrene.

Provisional Guidance for Quantitative Risk Assessment recommends that a RPF be used to convert concentration of carcinogenic PAHs (cPAHs) to an equivalent concentration of benzo(a)pyrene when assessing risks posed by these substances from oral exposures. The RPFs are based on the potency of each compound relative to that of benzo(a)pyrene.

<u>The toxicity values contained in the RSL tables have already been adjusted using the RFPs.</u> <u>The RSL SSLs for each PAH may be used and adjustment with RPFs is not required</u>. Computationally it makes no difference or little difference whether the RFPs are applied to the concentrations of PAHs found in the environmental samples or to the toxicity values as long as the RFPs are not applied to both.

5.2.6 Polychlorinated Biphenyls (PCBs)

PCBs refer to complex man-made mixtures of chlorinated hydrocarbons. PCBs were specifically manufactured for their insulating properties and have historically been used in capacitors, transformers, and other electrical equipment as they do not easily burn, evaporate nor conduct electricity. The term "Aroclor" refers to a PCB mixture of individual PCB compounds called PCB congeners. Theoretically, Aroclor mixtures can contain up to 209 different individual PCB congeners; however, most Aroclors contain only about 130 individual congeners.

Historically, it was appropriate to screen sites as well as estimate risks based on Aroclor data for both human health risk and ecological risk assessments. Recent guidance, however, requires that much more detailed information on polychlorinated biphenyl (PCB) congener data be collected at PCB-contaminated sites.

For PCB risk assessment under UAC R315-101, Aroclor analysis can be used as a preliminary screen and to investigate the nature and extent of contamination where release is suspected, i.e., for presence or absence of PCBs. If site history indicates no release or use of PCBs, congener analysis will not be required. However, at sites where site history indicates PCB release especially from PCB transformers or used oil recycling sites where the potential exists for a mixture or used oil and PCB oil, congener analysis must be performed to conduct a human health or ecological risk assessment.

The results of Aroclor analysis, however, must be interpreted carefully because a preliminary data indicating no PCB contamination may be a false negative result. This is true when PCB mixture has undergone extensive weathering and thereby changing the Aroclor composition for which the analytical method was based. To confirm that no PCB congeners are present, it may be necessary to conduct congener analysis on a limited number of samples. In addition, it is also possible to have Aroclor non-detect but have the dioxin-like PCB congener present at levels that pose unacceptable risk to human health and the environment.

The toxicity of a particular PCB mixture, whether it is the original commercial Aroclor or weathered environmental mixture analyzed in a sample, is dependent on the type and quantity of individual PCB congeners present in the PCB mixture. Although information on homologue composition can provide general information, it does not provide congener-specific information that is necessary to quantify toxicity and potential risks. This is because the toxicity of specific individual PCB congeners *within* each homologue group can vary by several orders of magnitude. In other words, knowledge of homologue composition is not particularly useful in quantifying the toxicity of the PCB mixture. While the *number* of chlorines represented in each homologue group is important, it is the three-dimensional *position* of chlorines and the conformation of the biphenyl rings that ultimately govern the toxicity of each of the 209 PCB congeners. Thus, it is not possible to assign toxicity values to homologue groups. Therefore, to evaluate the toxicity and health risks associated with environmental PCB mixtures, the composition and concentration of individual PCB *congeners* must be quantified. In addition, the position of the chlorination on the biphenyl ring governs toxicity.

A small subset of PCB congeners evokes dioxin-like toxic effects, which should be target analytes if an HHRA or ERA is conducted. There are 13 different PCB congeners in this group that have been identified by Ahlborg et al. (1994) and U.S.EPA (1996) that are structurally similar to chlorinated dibenzo-p-dioxins (CDDs) and chlorinated dibenzo furans (CDFs). These can be present in Aroclors 1242, 1248, 1254, and 1260. Like dioxin, these PCB congeners all bind to the aryl hydrocarbon receptor and elicit dioxin-specific biochemical and toxic responses. These toxic responses are exacerbated because these congeners have a long half-life in the body (for many decades) and persist and accumulate in the food chain.

Ahlborg, et al. (1994) have derived TEF for each of the 13 congeners as a fraction of the toxicity of 2,3,7,8-TCDD.

Toxicity data for the dioxin-like PCBs relative to 2,3,7,8-TCDD toxicity can be found on the EPA RSL Tables. TEFs for non-ortho [International Union of Pure and Applied Chemistry (IUPAC) numbers 77, 81, 126, and 169)] and mono-ortho congeners (IUPAC numbers 105, 114, 118, 123, 156, 157, 167, and 189) were assessed using the 2005 WHO TEFs (Van den Berg, et al 2006) while TEFs for di-ortho congeners (IUPAC numbers 170 and 180) are taken from Ahlborg, et al, 1993 (see Table 2-2).

The toxicity information (cancer potency factors) listed in the RSL Tables for the numbered PCB congeners are derived by applying the respective TEFs to the toxicity data for 2,3,7,8-TCDD. This means there should be no modification of sample data and/or the RSL values when conducting PCB risk assessment.

High Risk, Low Risk, Lowest Risk in Calculating Risk

The US EPA RSL Table contains PCB screening levels designated as “high risk”, “low risk” and “lowest risk”. However, as noted above, the screening hierarchy for PCBs is that Aroclors may be used for an initial presence/absence determination, but individual congener data are required if PCBs are confirmed present or a known COPC. Therefore the individual Aroclor and/or congener RSLs are used and preferred over total PCB data and High/Low/Lowest risk RSLs should not be used.

5.2.7 Total Petroleum Hydrocarbon (TPH)

Traditionally, hydrocarbon-impacted soils at sites contaminated by releases of petroleum fuels have been managed based on their total petroleum hydrocarbon (TPH) content. TPH refers to the total mass of hydrocarbons present without identifying individual compounds. In practice, TPH is defined by the analytical method that is used to measure the hydrocarbon content in contaminated media. Since the hydrocarbon extraction efficiency is not identical for each method, the same sample analyzed by different TPH methods will produce different TPH concentrations.

The hazard and health risk assessments that are typically conducted to support risk management decisions at contaminated sites generally require some level of understanding of the hydrocarbon chemical composition present in the contaminated media. Traditional TPH measurement techniques, however, provide no specific information about the detected hydrocarbons. Because TPH is not a consistent entity, the assessment of health effects and development of toxicity values for mixtures of hydrocarbons are problematic.

On that basis, DWMRC assesses risk from TPH by analyzing and assessing the individual chemical constituents rather than relying on TPH fraction data. Use of the Utah Department of Remediation and Environmental Response (DERR) Underground Storage Tank Initial Screening Levels (ISLs) and/or Tier 1 Screening Levels are not appropriate to use in risk assessments conducted for UAC R315-101.

The EPA RSL Table contains a listing of TPH fractions based on the PPRTV assessment. However, to circumvent problems associated with analytical methods and toxicity values for hydrocarbon mixtures, UAC R315-101 requires using the individual chemical constituents to evaluate risk from TPH release. All the TPH indicator compounds including most of the carcinogens in the TPH carbon range are listed in the EPA RSL Table. Table 5 below shows typical listings of TPH indicator compounds.

Table 5. Indicator Compounds Associated with Common TPH Mixtures

Indicator Compounds
Benzene
Toluene
Ethylbenzene
Xylene
Acenaphthene
Anthracene
Benzo(a)pyrene
Chrysene
Dibenz(a,h)anthracene
Indeno(1,2,3-cd)pyrene
Benzo(k)fluoranthene
Benzo(b)fluoranthene
Benzo(a)anthracene
Fluoranthene
Fluorene
Naphthalene
Pyrene
Lead (inorganic)
Metals
Methyl tert butyl ether (MTBE)
Methyl ethyl ketone (MEK)
Methyl isobutyl ketone

5.2.8 Polyfluoroalkyl and Perfluoroalkyl Compounds (PFAS) – RESERVED, For Informational Purposes Only

Polyfluoroalkyl and perfluoroalkyl compounds (PFAS), which are synthetic chemicals that do not occur naturally. However, once released, they are persistent and mobile in the environment. These compounds (and other PFAS) repel oil, grease, and water and have been used in many consumer, commercial and industrial products (Gaines, 2022).

Perfluorinated compounds are considered an emerging contaminant. These include perfluorohexane sulfonic acid (PFHxS), perfluorooctane sulfonate (PFOS), and perfluorooctanoic acid (PFOA).

PFAS may be divided into two primary categories: polymer (or potential precursors) and non-polymer PFAS. Table 6 lists the most common PFAS that should be included in analytical suites. In addition, to the listed PFAS, four replacement chemicals, GenX, Adona, and F53b major and minor should be included in the analytical suite as appropriate based upon site history.

Table 6. PFAS Analyte List

Analytical Name	Acronym	CAS Number
Perfluorotetradecanoic acid	PFTeA	376-06-7
Perfluorotridecanoic acid	PFTriA	72629-94-8
Perfluorododecanoic acid	PFDoA	307-55-1
Perfluoroundecanoic acid	PFUnA	2058-94-8
Perfluorodecanoic acid	PFDA	335-76-2
Perfluorononanoic acid	PFNA	375-95-1
Perfluorooctanoic acid	PFOA	335-67-1
Perfluoroheptanoic acid	PFHpA	375-85-9
Perfluorohexanoic acid	PFHxA	307-24-4
Perfluoropentanoic acid	PFPeA	2706-90-3
Perfluorobutanoic acid	PFBA	375-22-4
Perfluorodecanesulfonic acid	PFDS	335-77-3
Perfluorononanesulfonic acid	PFNS	68259-12-1
Perfluorooctanesulfonic acid	PFOS	1763-23-1
Perfluoroheptanesulfonic acid	PFHpS	375-82-8
Perfluorohexanesulfonic acid	PFHxS	355-46-4
Perfluoropentanesulfonic acid	PFPeS	2706-91-4
Perfluorobutanesulfonic acid	PFBS	375-73-5
Perfluoroictabesylfonamide	PFOSA	754-91-6
Fluorotelomer sulphonic acid 8:2	FtS 8:2	39108-34-4
Fluorotelomer sulphonic acid 6:2	FtS 6:2	27619-97-2
Fluorotelomer sulphonic acid 4:2	FtS 4:2	757124-72-4
2-(N-Ethylperfluoroactanesulfonamido) acetic acid	N-EtFOSAA	2991-50-6
2-(N-Methylperfluoroactanesulfonamido) acetic acid	N-MeFOSAA	2355-31-9

Despite the large number of potentially present substances, toxicity studies have only been conducted on a few PFAS. While PFAS are a class of emerging compounds, there is much focus on these substances by State and Federal regulatory communities. It is anticipated that there will be changes and updates to preliminary screening levels as more data become available.

It is noted that the June 2022 tap water screening levels for PFOA, PFOS and Perfluorobutanesulfonic acid (PFBS), which are based on the US EPA's updated Lifetime Health Advisories (US EPA 2022c) in drinking water, are very low and the new tap water screening levels may be below the capability of instrument MDLs. Until labs revised methods to obtain lower MDLs, the risk assessment should discuss any detections between the new tap water screening levels and detection limits in the Uncertainty Analysis and include lines of evidence to support any claim on risk.

US EPA has proposed PFAS national primary drinking water regulation and if finalized will regulate PFOA and PFOS as individual contaminants at four parts per trillion (ppt), or 4.0 nanograms per liter (ng/L) and will regulate four other PFAS – Perfluorononanoic acid (PFNA), PFHxS, PFBS and GenX Chemicals as a mixture using the hazard index (HI) approach where the HI must be less than or equal to one, as shown in Table 7.

Table 7. Proposed MCLs for Select PFAS

Compounds	Proposed MCLs
PFOS	4 ppt (4.0 ng/L)
PFOA	4 ppt (4.0 ng/L)
PFHxS	Hazard Index = 1.0 (unitless)*
GenX Chemicals	
PFNA	
PFBS	
*Learn more about the hazard index calculation, and the specific levels for these four PFAS below	

EPA is proposing to regulate four PFAS – PFHxS, GenX Chemicals, PFNA, and PFBS – as a mixture, using an established approach called a hazard index. The Hazard Index is a tool used to evaluate health risks from simultaneous exposure to mixtures of certain chemicals. Many PFAS are found together and in different levels and combinations. Estimating risk by considering one chemical at a time may underestimate the health risks associated with exposure to many PFAS at the same time. To prevent health risks from mixtures of certain PFAS in drinking water, EPA is proposing to use this Hazard Index calculation to regulate PFHxS, GenX Chemicals, PFNA, and PFBS in public water systems. To determine the Hazard Index for these four PFAS, water systems would monitor and compare the amount of each PFAS in drinking water to its associated Health Based Water Concentration (HBWC), which is the level at which no health effects are expected for that PFAS. Water systems would add the comparison values for each PFAS contained within the mixture. If the value is greater than 1.0, it would be an exceedance of the proposed Hazard Index MCL for PFHxS, GenX Chemicals, PFNA, and PFBS.

How is the Hazard Index for PFHxS, GenX Chemicals, PFNA, and PFBS calculated?

To determine the HI for these four PFAS, water systems would monitor and use those sampling results as inputs into a formula with their Health-Based Water Concentration (HBWC) (i.e., the level at which no health effects are expected for that PFAS). The proposed HBWCs for each of the four PFAS are in Table 8.

Table 8. Health Based Water Concentrations for Select PFAS

Compound	HMWC (ppt)
PFHxS	9.0
GenX Chemicals	10
PFNA	10
PFBS	2000

Water systems would use a calculator tool provided by US EPA to easily determine their HI result. The tool performs the calculation explained below.

For each of the four PFAS, the calculation first divides the results of the drinking water sample by the HBWC and then adds all the values for each PFAS. If the total value is greater than 1.0, it would be an exceedance of the proposed HI MCL as follows:

$$HI = \left(\frac{GenX_w}{HBWC\ ppt}\right) + \left(\frac{PFBS_w}{HBWC\ ppt}\right) + \left(\frac{PFNA_w}{HBWC\ ppt}\right) + \left(\frac{PFHxS_w}{HBWC\ ppt}\right) \qquad \text{Equation 3}$$

Where:
$GenX_w$ = monitored concentration of GenX in water
$PFBS_w$ = monitored concentration of PFBS in water
$PFNA_w$ = monitored concentration of PFNA in water
$PFHxS_w$ = monitored concentration of PFHxS in water

For example, if the mixture contains the following levels of these four PFAS, the HI for that mixture would exceed the proposed MCL.

$$2.1 = \left(\frac{5\ ppt}{10\ ppt}\right) + \left(\frac{200\ ppt}{2000\ ppt}\right) + \left(\frac{5\ ppt}{10\ ppt}\right) + \left(\frac{9\ ppt}{9\ ppt}\right) \qquad \text{Equation 4}$$

Following recent peer-reviewed science that indicates that mixtures of PFAS can pose a health risk greater than each chemical on its own, a HI helps to account for the increased risk from mixtures of PFAS that may be found in contaminated drinking water and or the environment. The HI is a long-established tool that US EPA regularly uses in addressing risks of chemical mixtures and it is used at contaminated sites under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and sites under the RCRA.

PFOA and PFOS were not included in the HI calculation because US EPA has determined that PFOA and PFOS are likely carcinogens (i.e., cancer causing). This means that there is no level of these contaminants that is without a risk of adverse health effects. Therefore, US EPA is proposing to set the MCL for these two contaminants at 4 ppt, the lowest feasible level based on the ability to reliably measure and remove these contaminants from drinking water.

US EPA used the practical quantitation limits (PQLs) for the six PFAS proposed for regulation in determining the proposed MCLs. US EPA has identified the PQLs for the six PFAS proposed for regulation as shown in Table 9 below. The PQL is defined as the lowest concentration of a contaminant that can be reliably achieved within specified limits of precision and accuracy during routine laboratory operating conditions. This level provides the precision and accuracy that US EPA estimates can be achieved across laboratories nationwide.

Table 9. Practical Quantitation Limits for Select PFAS

Compound	PQL (ppt)
PFOS	4.0
PFOA	4.0
PFHxS	3.0
GenX Chemicals	5.0
PFNA	4.0
PFBS	3.0

5.2.9 Salts

Salts are immensely soluble in ground and surface water. Salinity is the measure of the amount of salt present in soil and water. Salinity is broadly classified into primary and secondary. Primary salinity is the product of natural processes that deposit salts for an extended period on land and water like weathering, rain, and strong wind. Whereas secondary salinity is the action of anthropogenic activity such as releasing of oil and gas production water, well development fluids, hydraulic fracturing fluids, and flowback waters on the ground (Neff, et.al, 2011).

In arid regions, such as Utah, soil drainage is often poor and evaporation rates are high. Soils with sandy topsoil and dense clay subsoils may have severe problems at depth without any surface signs. The clay disperses because of an excessive proportion of sodium in the exchangeable cations attached to the surface of the clay. Soils with six percent or more of sodium as a percentage of the total exchangeable cations are sodic. Sodicity in soils has a strong influence on the soil structure of the layer in which it is present. A high proportion of sodium within the soil can result in dispersion, where the clay particles swell strongly and separate from each other on wetting. On drying, the soil becomes dense, cloddy and without structure. This dense layer is often impermeable to water and plant roots. In addition, scalding can occur when the topsoil is eroded and sodic subsoil is exposed to the surface, increasing erodibility. Thus, sodic soils adversely affect the plants' growth. (Wiesman, 2009).

If salts are released to surface soil, and sufficient precipitation does not wash the salts to below plant root levels, the increased soil salinity will stunt growth and eventually kill most of the native plants. High salt levels hinder water absorption, inducing physiological drought. The soil may contain adequate water, but plant roots are unable to absorb the water due to unfavorable osmotic potential. This is referred to as osmotic or water-deficit effect of salinity (Greenway and Munns, 1980). Plants are generally most sensitive to salinity during germination and early growth. Salinity inhibits seed germination, plant growth, development, and yield and lowers soil water potential and leaf water potential disturbing plant water relations and reducing the turgor of plant, which ultimately leads to osmotic stress (Arif, et. al, 2020). Soil salinity imposes ion toxicity, nutrient deficiencies, nutritional imbalances, osmotic stress, and oxidative stress on plants (Pichtel, 2016). With native plants unable to thrive in saline conditions, the soil is either left barren and subject to erosion or non-native invader species may also move into the area.

Runoff from saline soils into surface water bodies, the salts will tend to sink towards the bottom of the water body, creating a dense layer that can inhibit gas exchange with the overlying water. This can lead to the development of low oxygen conditions that are detrimental to fish and other aquatic organisms (Arif, etc. al, 2020).

When there has been a release of salts to either soil or a water body, the ecological toxicity of the increased salts and salinity must be evaluated as part of the risk assessment. For oil and gas production water, well development fluids, hydraulic fracturing fluids, and flowback waters, in addition to salts, other common contaminants include water-soluble low molecular weight organic acids and monocyclic aromatic hydrocarbons, total PAHs, and higher molecular weight alkyl phenols.

As noted above, with time, and continued natural precipitation, the issue of adsorption, complexation, lability of contaminants in soils, and the corresponding reduction in toxicity over time is an important issue in understanding the fate of salts in soils.

Sufficient ecological toxicity data are available for most salts, to include sodium, chloride, bromide, nitrate/nitrite, and phosphate. If a release of saline waters has occurred, remediation may be needed along with a site-specific ecological risk assessment.

5.2.9.1 Salt Affected Soil

A soil-affected soil is defined as a soil that has been adversely modified for the growth of plants by the presence of or actions of soluble salts. This group of soils includes both *sodic* and *saline* soils (Nomenclature Committee Report, 1958). *Saline soil* contains sufficient soluble salts to interfere with growth of most crop plants. *Sodic soil* contains sufficient exchangeable sodium to interfere with the growth of most crop plants. *Saline-sodic* soil contains sufficient salt and exchangeable sodium to interfere with the growth of most crop plants.

Most salt-affected soils are associated with semiarid and arid climates. It should be noted that not all soils in arid regions are salt-affected. Under a dry-climate regime such as in Utah, the potential evaporation rates greatly exceed precipitation over most of the year (James et al., 1982). This climate condition dictates that essentially no water percolates through the soil under natural conditions.

5.2.9.2 Classification of Salt-affected Soil

Salt-affected soils may be classified into *normal, saline, sodic and saline-sodic* categories. The criteria used to classify salt-affected soils are:

1. *Electrical Conductivity (EC):*

   Measures the ability of the soil solution to conduct electricity. Salinity of the saturation extract as measured by the electrical conductivity at 25 °C and expressed in reciprocal ohms or ohm^{-1} and referred to as mho (ohm spelled backwards). Conductivity is expressed as specific conductance or conductance of a unit volume of solution as millimhos per centimeter (mmhos/cm). According to the US Salinity Lab (US SLS, 1954) a *saline salt* has an EC of 4 ds/m or greater. Plants vary in their tolerance to salinity which influences water uptake or available water.

2. *Total Soluble Salts (TSS)*

   Refers to the total amount of salts in a soil-saturated paste extract expressed in milligrams per liter (mg/L). The total soluble salts (TSS, in mg/L) are approximately equivalent to 640 times the electrical conductivity (EC, in mmhos/cm).

3. *Exchangeable-sodium percentage (ESP)*

   ESP is the sodium adsorbed on soil particles as a percentage of the Cation Exchange Capacity (CEC). CEC is the estimated sum of the major exchangeable cations, including hydrogen and expressed as milliequivalent per 100 grams of soil (meq/100g). *Sodic soil* has an ESP greater than 15% (US Salinity Lab, 1954). ESP is used to characterize sodicity of soils only.

   Sodicity is manifested in the swelling and subsequent deflocculation (dispersion) of the clay minerals, resulting in retardation of both air and water entry into the soil. Sodicity is particularly serious in heavy-textured soils that contain 2:1 expanding clay minerals. Sandy oils are affected less due to their low clay content.

4. *Sodium Adsorption Ratio (SAR)*

   SAR describes the proportion of sodium to calcium and magnesium in soil solution. Concentrations are expressed in milliequivalents per liter (meq/L) analyzed from a saturated paste extract. When the SAR is greater than 13, the soil is called *sodic soil.* Excess sodium in sodic soils causes soil particles to repel each other preventing the formation of soil aggregates. The result is a very tight soil structure with poor infiltration, poor aeration and surface crusting making tillage difficult and restricts seedling emergence and root growth (Munshower, 1994; Seelig, 2000; Horneck et al., 2007).

Figure 4 provides the classification of salt-affected soils using the saturated paste extraction method for determining the amount of salt in soil.

Class	**EC (mmhos/cm)**	**SAR**	**ESP**	**Typical soil structural condition**[1]
Normal	Below 4.0	Below 13	Below 15	Flocculated[2]
Saline	Above 4.0	Below 13	Below 15	Flocculated
Sodic	Below 4.0	Above 13	Above 15	Dispersed[3]
Saline-Sodic	Above 4.0	Above 13	Above 15	Flocculated

[1] Soil structural condition also depends on other factors not included in the Natural Resources Conservation Service (NRCS) classification system, including soil organic matter, soil texture, and EC of irrigation water.

[2] Flocculated soil – soil stuck together, aggregated. Allows for water to move through large pores and plant roots to grow mainly in pore spaced.

[3] Dispersed soil – soil that is plugged with no aggregate formation. Impedes water movement and soil drainage.

Figure 4. Classification of Salt-affected Soils (Saha, 2022)

5.2.9.3 Visual Diagnosis of Salt-affected Soil

The three soil conditions - saline, saline-sodic and sodic soils resulting from accumulation of salts have distinct characteristics that can be observed in the field. These characteristics are useful and helpful for diagnosing salinity problems. Completely white soils, or soils with a white crust are saline. Plants may exhibit leaf tip burn. Soils with brown-black crust or a black powdery residue are sodic and are indicative of poor drainage. Grey colored soils with stressed plants are generally saline-sodic.

5.2.10 Contaminants of Emerging Concern

Contaminants of emerging concern are those contaminants possibly present in environmental media that are suspected to elicit adverse effects to human and ecological receptors but may or may not have established health standards or established analytical methods. As many agencies, including the US EPA, are working to understand the types of effects and levels of concern in environmental media, it is important to consider whether emerging contaminants may be present at facilities in Utah.

For facilities where a regulated contaminant of emerging concern is detected in site media and RSLs are available, a quantitative analysis is required if RSLs are available. If RSLs are not available, a qualitative discussion of potential exposure and impact on overall risk/hazard must be included in the risk assessment. If the detected contaminant of emerging concern is not regulated e.g., PFAS, only a qualitative assessment will be required in a risk assessment describing potential impacts on human health and the environment in a risk assessment until such time that the contaminant of emerging concern becomes regulated, and RSLs become available.

6.0 HUMAN HEALTH RISK ASSESSMENT

The methodology and exposure assumptions that are utilized in order to quantify risks and hazards to current and future human receptors at sites in accordance with UAC R315-101, follows the standard exposure scenarios at the site: 1) hypothetical residential land use and construction; or 2) actual (industrial) land use and construction. Risk assessments must be conducted at sites where the nature and extent of contamination has been fully characterized. This applies to sites in Environmental Cleanup Program, Corrective Action Sites, and permitted facilities.

Note that if the nature and extent of contamination has not been defined for a site, a risk assessment should not be submitted to the DWMRC.

6.1 RSLs

The RSLs for the resident and composite worker (i.e., industrial/commercial worker) are tabulated and available for soil, indoor air, and tap water. The RLS based on a cancer risk of 1E-06 and a HQ of 1.0 should be applied. The HQ of 1.0 over 0.1 is acceptable, as DWMRC requires all COPCs to be evaluated for a combined assessment and total HI (See Equation 11).

Table 10. What RSL to Use for a Given Medium

Exposure Medium	RSL to Use
Soil	Soil
Indoor air	Indoor air
Groundwater	Tapwater[1]
Vapor intrusion (soil gas or groundwater)	VISL (see Section 7)

[1] While an MCL may be used for site characterization and as a protection standard for corrective action, the tapwater RSL is applied for risk assessments. This is because the MCL is not derived purely on toxicity but rather incorporates technology constraints. The risk assessment only evaluates toxicity. The tapwater RSL applies to both residential and industrial/commercial receptors.

RSLs are not available for a construction worker and the on-line calculator should be used to derive construction worker screening levels.

6.1.1 Construction Worker RSLs

The RSL calculator may be used to calculate the Construction Worker screening level values. The default values in the calculator may be used to calculate the SSLs. However, the on-line calculator requires a particulate emission factor (PEF) and volatilization factor (VF). A default PEF has been calculated as shown in Equation 5. This equation can also be used to develop a site-specific PEF for a construction worker scenario as needed.

US EPA toxicity data indicate that risks from exposure to some chemicals via the inhalation pathway far outweigh the risk via ingestion or dermal contact. To address the soil/sediment-to-air pathways, the RSL calculations incorporate a VF for volatile contaminants and a PEF for semi-volatile and inorganic contaminants.

Inhalation of chemicals absorbed to suspended respirable particles in ambient air is assessed by calculating a site-specific PEF, which is calculated based on modeled fugitive dust emissions from contaminated soils. The PEF addresses dust generated from open sources, which is termed "fugitive" because it is not discharged into the atmosphere in a confined flow stream. For further details on the methodology associated with the PEF model, the reader is referred to US EPA's *Soil Screening Guidance: Technical Background Document* (US EPA 1996), *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* (US EPA 2002a) and *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (US EPA 1998a).

It is important to note that the PEF for use in evaluating exposures of industrial worker receptors addresses only windborne dust emissions and does not consider emissions from traffic or other forms of mechanical disturbance, which could lead to a greater level of exposure. The PEF for use in evaluating the construction worker exposures considers windborne dust emissions and emissions from vehicle traffic associated with construction activities. Therefore, the fugitive dust pathway must be considered carefully when developing the CSM at sites where receptors may be exposed to fugitive dusts by other mechanisms.

Equation 5. Derivation of the Particulate Emission Factor

$$\mathrm{PEF_{CW}} = \mathrm{Q/C_{CW}} \times \frac{1}{\mathrm{F_D}}\left[\frac{\mathrm{T} \times \mathrm{A_R}}{556 \times \left(\frac{\mathrm{W}}{3}\right)^{0.4} \times \frac{(365\ \mathrm{days/yr}\ -\ \mathrm{P})}{365\ \mathrm{days/yr}} \times \sum \mathrm{VKT}}\right]$$

Parameter	**Definition (units)**	**Value**	**Reference**
PEF_{CW}	Particulate emission factor for a construction worker (m^3/kg)	2.1E+06	Calculated (Default)
Q/C_{CW}	Inverse of a mean concentration at center of a 0.5-acre-square source (g/m^2-s per kg/m^3)	23.02	US EPA 2002b
F_D	Dispersion correction factor (unitless)	0.185	US EPA 2002b
T	Total time over which construction occurs (s)	7.2E+06	US EPA 2002b
A_R	Surface area of road segment (m^2)	274.2	US EPA 2002b
W	Mean vehicle weight (tons)	8	US EPA 2002b
P	Number of days with at least 0.01 inches of precipitation (days/yr)	60	US EPA 2002b
ΣVKT	sum of fleet vehicle kilometers traveled during the exposure duration (km)	168.75	US EPA 2002b

The soil-to-air VF is used to define the relationship between the concentration of the contaminant in soil and the flux of the volatilized contaminant to ambient air. The volatilization factor is applicable to COPCs that are VOCs. VOCs are defined as those chemicals having a Henry's Law constant greater than 1 x 10^{-5} atmospheres-cubic meters per mole (atm-m^3/mole) and a molecular weight less than 200 grams per mole (g/mole). The emission terms used in the VF are chemical-specific and will be calculated from physical-chemical information obtained from sources including US EPA's *Soil Screening Guidance: Technical Background Document* (US

EPA, 1996 and 2001), US EPA Master Physical and Chemical Parameter table for development US EPA RSLs (US EPA 2011a), US EPA's *Basics of Pump and Treat Groundwater Remediation Technology* (US EPA 1990), US EPA's *Dermal Exposure Assessment* (US EPA 1992a*)*, *Superfund Public Health Evaluation Manual* (US EPA 1986), US EPA's *Additional Environmental Fate Constants* (US EPA 1995), Hazardous Substance Release/Health Effects Database (ATSDR 2003), the Risk Assessment Information System database (DOE 2005), and/or the CHEMFACTS database (US EPA 2000c). The VF is calculated using Equation 6.

Equation 6
Derivation of the Volatilization Factor for Construction Worker Scenario

$$VF_{s-cw} = \left(\frac{\left(3.14 \times D_A \times T\right)^{0.5}}{2 \times \rho_b \times D_A} \right) \times 10^{-4} \times Q/C \times (1/F_D)$$

Where:

$$D_A = \frac{\left[\dfrac{\left(\theta_a^{10/3} D_a H' + \theta_w^{10/3} D_w\right)}{n^2} \right]}{\rho_b K_d + \theta_w + \theta_a H'}$$

Parameter	Definition (units)	Default
VF_{s-cw}	Volatilization factor for soil, construction worker (m^3/kg)	Chemical-specific
D_A	Apparent diffusivity (cm^2/s)	Chemical-specific
Q/C	Inverse of the mean concentration at the center of a 0.5- acre-square source (g/m^2-s per kg/m^3)	Salt Lake
T	Exposure interval (s)	3.15E+07
10^{-4}	Conversion factor (m^2/cm^2)	1E-04
F_D	Dispersion correction factor (unitless)	0.185
ρ_b	Dry soil bulk density (g/cm^3)	1.5
n	Total soil porosity $1 - (\rho_b/\rho_s)$	0.43
θ_a	Air-filled soil porosity ($n - \theta_w$)	0.17
θ_w	Water-filled soil porosity	0.26
ρ_s	Soil particle density (g/cm^3)	2.65
D_a	Diffusivity in air (cm^2/s)	Chemical-specific
H'	Dimensionless Henry's Law constant	Chemical-specific
D_w	Diffusivity in water (cm^2/s)	Chemical-specific
K_d	Soil-water partition coefficient (cm^3/g) = $K_{oc} \times f_{oc}$ (organics)	Chemical-specific
K_{oc}	Soil organic carbon partition coefficient (cm^3/g)	Chemical-specific
f_{oc}	Fraction organic carbon in soil (g/g)	0.0015

6.1.2 Construction Worker - Dermal Contact with Groundwater

If VOCs are present, follow the methodology for a trench scenario outlined in Section 7.2.

If shallow groundwater is present at a site at depths less than 10 ft bgs, it is possible that a construction worker could come into contact with potentially contaminated groundwater during intrusive activities. Incidental ingestion of groundwater may occur, but the amount of groundwater accidentally ingested is assumed to be negligible and evaluation of this scenario would not result in significant risk. However, exposure through dermal contact with groundwater must be evaluated if: 1) groundwater is less than 10 ft bgs, and 2) groundwater has been impacted by site activities.

Equation 7 below is used to estimate the dermally absorbed dose (DAD) from accidental contact with contaminated groundwater (US EPA, 2004).

Equation 7. Dermal Absorbed Dose (DAD) – Incidental Contact with Groundwater

$$DAD = \frac{DA_{event} \times EV \times ED \times EF \times SA}{BW \times AT}$$

Parameter	**Definition (units)**	**Value**	**Reference**
DAD	Dermally Absorbed Dose (mg/kg-day)	--	--
DA_{event}	Absorbed dose per event (mg/cm^2-event)	Chemical-specific	Equations 8 or 9
EV	Event Frequency (events/day)	1	US EPA 2004
ED	Exposure Duration (yr)	1	DSHW, 2008
EF	Exposure Frequency (days/year)	125	DSHW, 2008
SA	Skin surface area available for contact (cm^2)	3,470	US EPA 2014
BW	Body Weight (kg)	80	US EPA 2014
AT_c	Averaging Time, carcinogens (days)	ED x 365 days/yr	US EPA 2004
AT_n	Averaging Time, noncarcinogens (days)	70yr x 365 day/yr	US EPA 2004

The absorbed dose per event is dependent on the lag time and the permeability of the chemical into the skin and is evaluated differently for organics and inorganics. Equation 8 shows the calculation methods for organic constituents. US EPA 2004 guidance will be followed for determining lag times and times to reach steady state and site-specific data will be used where available.

Equation 8. Dermal Absorbed Dose per event for Organic Constituents

If $t_{event} \leq t^*$, then:

$$DA_{event} = 2FA \times K_p \times C_w \sqrt{\frac{6\tau_{event} \times t_{event}}{\pi}}$$

If $t_{event} > t^*$, then:

$$DA_{event} = FA \times K_p \times C_w \left[\frac{t_{event}}{1+B} + 2\tau_{event}\left(\frac{1+3B+3B^2}{(1+B)^2}\right)\right]$$

Parameter	Definition (units)	Value	Reference
DA_{event}	Absorbed dose per event (mg/cm^2-event)	Chemical-specific	US EPA 2004
FA	Fraction absorbed water (unitless)	Chemical-specific	US EPA 2004
Kp	Dermal permeability coefficient in water	Chemical-specific	US EPA 2004
Cw	Chemical concentration in water (mg/cm^3)	Site-specific	EPC
τ_{event}	Lag time per event (hours/event)	Chemical-specific	US EPA 2004
t_{event}	Event duration (hours/event)	1	US EPA 2004
t^*	Time to reach steady state (hours)	2.4 x τ_{event}	US EPA 2004
B	Dimensionless ratio of permeability coefficient through the stratum corneum relative to its permeability coefficient across the viable epidermis (unitless)	Chemical-specific	US EPA 2004

Per US EPA 2004, the absorbed dose per event for inorganics is calculated using Equation 9.

Equation 9. Dermal Absorbed Dose per event for Inorganic Constituents

$$DA_{event} = K_p \times C_w \times t_{event}$$

Parameter	Definition (units)	Value	Reference
DA_{event}	Absorbed dose per event (mg/cm^2-event)	Chemical-specific	US EPA 2004
K_p	Dermal permeability coefficient in water	Chemical-specific	US EPA 2004
C_w	Chemical concentration in water (mg/cm^3)	Site-specific	EPC
t_{event}	Event duration (hours/event)	1	US EPA 2004

6.2 Quantifying Risk

The process used by the RSL calculator to calculate carcinogenic risk and hazard quotient uses a simple method that relies on the linear nature of the relationship between concentration and risk.

Cancer risks are added together to calculate cumulative risk using Equation 10 below, while noncancer HIs is calculated using Equation 11. If a COPC has both carcinogenic and noncarcinogenic endpoints, both of these endpoints will be evaluated against appropriate screening levels in the screening level calculations shown in Equations 10 and 11. The RSL summary tables only present the screening level that is most conservative; however, the supporting tables provide screening levels for both endpoints when a chemical may exhibit both cancer and non-cancer effects.

Equation 10. General Cumulative Risk for Carcinogenic COPCs

$$Cumulative\ Risk = \left[\left(\frac{EPC_1}{RSL_1}\right) + \left(\frac{EPC_2}{RSL_2}\right) + \cdots + \left(\frac{EPC_i}{RSL_i}\right)\right] \times (TR)$$

$$Individual\ Cancer\ Risk = \left(\frac{EPC}{RSL}\right) \times TR$$

Note: Risk for each exposure route will be added for an overall risk (soil, water, and air). RSLs may not include all exposure pathways. Vapor intrusion risks are added to this calculation to result in total risk.

Parameter	**Definition (units)**
Cumulative Risk	Sum of individual constituents' risks (unitless; expressed as incremental probability of developing cancer over a lifetime)
$EPC_{1,2\ldots i}$	Exposure Point Concentration. Maximum detected concentration for constituents 1 through *i* (mg/kg for soil [0-10 ft bgs]; µg/L for groundwater, and µg/m^3 for indoor air); or revised EPC (95UCL)
$RSL_{1,2\ldots i}$	US EPA residential RSL for constituents 1 through *i* (carcinogenic endpoint) (mg/kg for soil; µg/L for tap water, and µg/m^3 for indoor air)
TR	DWMRC target risk level (1 x 10^{-6}) (unitless; incremental probability)

µg/L – micrograms per liter
µg/m^3 – micrograms per cubic meter

For COPCs with noncarcinogenic endpoints, the maximum detected concentration is divided by the RSL for each constituent and multiplied by the target hazard quotient (HQ) of one, resulting in the HQ. The HQs will be added together to calculate the HI (See Equation 11). Since all HQs are initially considered to be additive, the RSLs based on a target level of 1.0 are applied.

In the event that the hazard index results in a value above the target level of 1.0, noncarcinogenic effects may be evaluated for those chemicals with the same toxic endpoint and/or mechanism of action. While for carcinogens, the effect is response-addition (meaning the end result is cancer regardless of type), for noncarcinogens, toxicity is unique to specific organs and only chemicals with the same mode of action exhibit response-addition. For a refined noncarinogenic assessment, chemicals are separated by similar mode of action. This is referred to as a target organ analysis. The sources of information on toxic end point or mechanism of action follow the US EPA toxicity hierarchy, as outlined in Section 5.1, with the IRIS database being the first tier. This information may be used to evaluate the additive health effects resulting from simultaneous exposure to multiple contaminants.

Equation 11. General Hazard Index for Noncarcinogenic COPCs

$$HI = [(HQ_1) + (HQ_2) + \cdots + (HQ_i)] \times THQ$$

$$HQ = \frac{EPC}{RSL}$$

Note: HIs for each exposure route will be added for an overall HI (soil, water, and air). RSLs may not include all exposure pathways. Vapor intrusion risks are added to this calculation to result in total hazard index.

Parameter	**Definition (units)**
HI	Hazard index; sum of HQs (unitless)
HQ	Hazard quotient (unitless)
THQ	Target hazard quotient (1.0) (unitless)
EPC	Exposure Point Concentration. Maximum detected concentration for constituents 1 through *i* (mg/kg for soil; µg/L for groundwater; and µg/m^3 for indoor air); or revised EPC (95UCL)
RSL	US EPA residential RSL (noncarcinogenic endpoint) (mg/kg for soil; µg/L for tap water; µg/m^3 for indoor air), based on target level of 1.0

When calculating a revised HI, only those HQs for chemicals with the same mode of action (e.g., target organ) are summed. This potentially results in several HIs. Refer to Table 11 for an example.

Table 11. Target Organ Analysis Example

				Target Organ Systems[1]	**Non-Cancer RSL (mg/kg)**	**Non-Cancer Hazard Quotient**
					Soil	**Soil**
Chemical	**CASRN**	**EPC (mg/kg)**		**Chronic Exposure**	**Industrial Worker**	**Industrial Worker**
Benzene	71-43-2	533.1	95 UCL	HM, IM	4.2E+02	1.0
Ethylbenzene	100-41-4	62.0	Max	HP; UR; DV	1.7E+04	0.004
Naphthalene	91-20-3	306.0	95 UCL	NV, RS	5.9E+02	0.5
				Non-Cancer HI:		**1.6**
				Developmental (DV) HI:		0.004
				Hematological (HM) HI:		1.0
				Hepatic (HP) HI:		0.004
				Immune (IM) HI:		1.0
				Nervous (NV) HI:		0.5
				Respiratory (RS) HI:		0.5
				Urinary (UR) HI:		0.004

RSL =Composite Worker (Target Cancer Risk = 1E-06 and Target Hazard Quotient = 1.0)
CASRN = Chemical Abstract Services Registry Number
EPC = Exposure Point Concentration
mg/kg = milligrams per kilogram
(1) US EPA's IRIS and the Risk Assessment Information System were consulted for target organ groups.

6.3 Chemicals with No RSLs

The RSL tables do not address all constituents that may potentially be present at a site. The absence of a RSL does not preclude the evaluation of that constituent in the risk assessment. For each compound that does not have an RSL, an effort must be made to determine if there are available toxicity data to derive a screening level following the preferred toxicity database hierarchy (See Section 5.1). Methodologies and assumptions consistent with those used to develop the RSLs should be applied.

In addition, quantitative structure-activity relationship (QSAR) can be used to find relationships between chemical structure or structural properties and biological activity of target property based on structural similarities. Toxicity data for these chemicals can be used as surrogates for chemicals with no US EPA RSLs. Biological effects of compounds can often be predicted from their molecular structure using data about other similar compounds. This is because there is a relationship between molecular structures and their biological activity.

6.4 One Hit Model

The one-hit equation is only applied to scenarios where the exposure dose is high, and it assumes any single “hit” of an amount of a carcinogen at a cellular target (e.g., DNA), can initiate a series of events leading to a tumor. The one-hit equation is an exponential model that limits the single chemical risk to less than one, whereas the regular linear cancer model may calculate values greater than one. The equation (12) is as follows:

$$Risk = 1 - e^{(-intake \times toxicity)} \qquad \text{Equation 12}$$

The reassessment of risk is typically only focused on the receptor of concern (e.g., residential or industrial) and the critical exposure pathway driving risks at the site. It is noted that excluding non-critical exposure pathways may underestimate total risk.

The resulting risk is assessed to determine if the initial risk that concluded adverse health impact is valid.

Example: Calculated risk to benzene

Assume the carcinogenic risk to an industrial worker for benzene exceeded 1.0E-02 and the risk was driven by ingestion of soil. The intake (soil ingestion) would be calculated using US EPA intake equation, for example from the Human Health Evaluation Manual Supplemental Guidance.

$$Intake = \frac{Concentration \times Ingestion\ Rate \times Conversion\ Factor}{Body\ Weight} \qquad \text{Equation 13}$$

Assuming the concentration is soil is 3.9E+05 mg/kg, ingestion rate of 100 mg/kg, body weight of 80 kg, exposure frequency of 225 days/year and an exposure duration of 25 years, the intake would be 4.88E-01 mg/kg-day. The oral cancer slope factor for benzene is 5.5E-02 mg/kg-day.

$$Risk = 1 - Exp(-Intake \times CSF) \qquad \text{Equation 14}$$

The risk estimated for exposure by the One-hit model is 2.64E-02, which is above the acceptable risk range. The conclusion is that exposure to benzene in soil to an industrial work is outside the risk range, and the initial conclusion that excess risk is present is valid.

As with any risk-based tool, the potential exists for misapplication. In most cases the root cause will be a lack of understanding of the intended use of screening levels. In order to prevent misuse of the RSLs, the following should be avoided:

- Applying RSLs to a site without adequately developing a conceptual site model that identifies relevant exposure pathways and exposure scenarios,
- Use of RSLs as cleanup levels without verifying numbers with a toxicologist or risk assessor,
- Not considering the effects of additivity when screening multiple chemicals, and
- Applying RSLs and risk determinations on sites where the nature and extent of contamination has not been defined.

6.5 Discussion of Uncertainties

All risk assessments involve many assumptions that may or may not accurately reflect site conditions. A discussion of the uncertainties associated with the risk assessments must be included in each site-specific risk assessment conducted at a site. Typical uncertainties in human health risk assessments that may over- or underestimate risks and/or hazards that may be applicable at any site may include:

- Data collection and evaluation – insufficient number of samples; loss of contaminant during sampling; high method detection limits; and field or laboratory contamination.
- Exposure assessment – exposure assumptions that may not accurately reflect actual exposures; representativeness of fate and transport models; use of maximum detected concentrations as the EPC; assumption of uniform concentration over entire site; and assumption of 100% bioavailability of COPCs.
- Toxicity assessment – availability and accuracy of toxicity data; use of surrogate toxicity data; extrapolation of results of toxicity studies from animals to humans; and assumption of linearity of dose-response relationships.
- Risk Characterization – assumption of additivity of risk/hazard estimates; use of surrogate toxicity information; and unavailable toxicity information.

7.0 VAPOR INTRUSION

If volatiles are present in subsurface media (e.g., soil-gas or groundwater), volatilization through the vadose zone and into indoor air could occur. If indoor air data are available, the indoor air RSLs may be used for direct comparison. However, if indoor air concentrations are not available, the US EPA vapor intrusion screening levels (VISLs) and the VISL calculator are used for estimating the indoor air concentration based on groundwater or soil gas data. VOCs are considered those chemicals having a Henry's Law constant greater than 1E-05 atmospheres –

cubic meter per mole (atm-m^3/mole) and a molecular weight less than 200 grams per mole (g/mole) and determined to be sufficiently volatile and toxic to pose inhalation risk via vapor intrusion from either a soil or groundwater source.

Residential receptors and industrial workers could be exposed to VOCs volatilized from subsurface media (soil and/or groundwater) through pore spaces in the vadose zone and building foundations (or slab) into indoor air. Construction workers may be exposed via build-up of VOCs in trenches.

Incomplete pathway; no action required.

If no VOCs are detected in site media, then the vapor intrusion pathway is considered incomplete.

Potentially complete pathway - qualitative discussion

If during investigation sampling the following criteria are met, the pathway is considered potentially complete, and a qualitative discussion of the vapor intrusion pathway will be required:

- VOC detections are minimally (e.g., once or twice) detected in site media (soil, soil gas, and/or groundwater),
- Concentrations are below screening levels,
- There is no suspected source(s) for VOCs, and/or
- Concentrations are decreasing with depth (for soil).

In addition, if VOCs were present at a site but the source(s) and associated contaminated soil have been removed and the following criteria have been met, only a qualitative assessment of the vapor intrusion pathway will be required:

- Confirmation sampling indicates removal of the source with minimal VOCs detected in soil/soil gas or groundwater data,
- Concentrations are below screening levels, and
- Concentrations decrease with depth.

Complete pathway; quantitative assessment

If during investigation sampling or confirmation sampling VOCs are detected consistently in site media, concentrations are detected at depth or show increasing concentrations with depth in soil, and/or there is potentially a source(s) for the VOCs based on site history, a quantitative assessment of the vapor intrusion pathway is required following a tiered approach.

US EPA guidance no longer supports the use of bulk soil data for evaluation of the vapor intrusion pathway (US EPA, 2002). If VOCs are present and this pathway is complete, active soil gas and/or groundwater data must be used as appropriate. Note that passive soil gas data may be used to assess the presence or absence of VOCs, but active soil gas data are required for assessing the risk pathway.

Step 1. Compare the maximum detected concentration for soil gas or groundwater against the EPA's VISL calculator (EPA, 2023) using the default attenuation factors (0.03 soil gas and 0.001 groundwater). Attenuation is the reduction in concentrations that occurs through migration in the subsurface combined with the dilution that occurs when vapor enters a building and mix with indoor air. The attenuation factor is expressed as the ratio of concentrations of chemicals in indoor air to the concentrations in subsurface vapor. Attenuation factors are site specific and can vary depending on several variables (e.g., soil type, depth of contamination, building characteristics and indoor air exchange rates). The US EPA default attenuation factors are based on conservative assumptions and empirical data. If active soil gas data are collected from soils located outside of a structure or below a slab, the VISL target sub slab and exterior soil gas concentrations for a target cancer risk of 1E-06 and a target HQ of 1.0 should be applied. The VISL target groundwater concentrations for a target cancer risk of 1E-06 and a target HQ of 1.0 should be applied for groundwater data.

It is recommended that conditions at the site are consistent with the assumptions underlying the generic VISL conceptual model. Specific factors may result in unattenuated or enhanced transport of vapors towards a receptor, and consequently are likely to render the VISL screening target subsurface concentrations inappropriate. If the following conditions apply, then the use of VISL is not appropriate and evaluation should follow the processes in Step 2:

- Very shallow groundwater sources (for example, depths to water less than five feet below foundation level);

- Shallow soil contamination vapor sources (for example, sampled at levels within a few feet of the base of the foundation); or

- Buildings with significant openings to the subsurface (for example, sumps, unlined crawlspaces, earthen floors) or significant preferential pathways, either naturally occurring or anthropogenic (not including typical utility perforations present in most buildings).

Step 2. Use other suitable, acceptable, and well calibrated mathematical models to estimate indoor air concentration and vapor intrusion. Model results (i.e., predicted indoor air or sub slab soil gas concentration) must be in good agreement with measured data.

The US EPA and State risk assessors and toxicologists participate in a quarterly Risk Assessor meeting. The topics of the April 2022 meeting were the numerical/calculational problems with Version 6.0 of the US EPA Johnson and Ettinger (J&E) Model Spreadsheet Tool (September 2017) and the applicability of the J&E modeling in risk assessments.

US EPA indicated that the 2017 J&E Model Spreadsheet Tool has been noted to have some limitations as well as producing some calculation errors. The current on-line version has a programming error in the calculation of lifetime cancer risk for mutagenicity and this is especially concerning for contaminants such as trichloroethylene (TCE). In summary, US EPA does not recommend the use of the current model for soil vapor intrusion assessment particularly for the purpose of demonstrating that a response action is not needed.

US EPA further stated that when suitably constructed, documented, and verified mathematical models can provide an acceptable line of evidence supporting risk management decisions pertaining to vapor intrusion. This may suggest that to use the J&E Model which is a predictive model, one must collect sub slab data or other site-specific data to perform a model calibration to fit the data. The output data or predicted data must be in good agreement with measured data for the use of the J&E Model to be acceptable for vapor intrusion assessment.

Nonetheless, US EPA contends that until such time the US EPA addresses the programming issues identified in the 2017 J&E Model, its use is considered unacceptable for vapor intrusion assessment at any site. In lieu of the use of the J&E Model, DWMRC recommends the use of the US EPA VISL calculator in vapor intrusion assessment at any site. DWMRC may consider the use the J&E Model for use in vapor intrusion assessment but only with the approval of the Director, (DWMRC Position Paper on the J&E Model, https://documents.deq.utah.gov/waste-management-and-radiation-control/corrective-action/DSHW-2022-022911.pdf).

7.1 Vapor Intrusion Screening Levels

Residential receptors and commercial/industrial workers could be exposed to volatile compounds vaporized from subsurface media (soil gas and/or groundwater) through pore spaces in the vadose zone and building foundations (or slabs) into indoor air. Per US EPA guidance (US EPA, 2015 and errata 2018), this pathway must be evaluated if: 1) there are vapor-forming compounds present in subsurface media that are sufficiently volatile and toxic, and 2) there are existing or planned buildings where exposure could occur. If volatile and toxic constituents are detected in site media and are not listed, VISLs should be calculated following the methodologies in the US EPA Vapor Intrusion Screening Level Guidance Document.

The US EPA (2015 and errata 2018) vapor intrusion guidance does not support the use of bulk soil data for evaluation of the vapor intrusion pathway; active soil gas and/or groundwater data must be used as appropriate. As such, VISLs are neither available nor recommended for soil. It is noted; however, that bulk soil data can be used in a qualitative sense to determine delineation of a vapor source or in determining if soil has been impacted and additional evaluation (e.g., soil gas) is needed. Conversely, it must not be assumed that non-detect results of volatile compounds in soil equates to an absence of a vapor source.

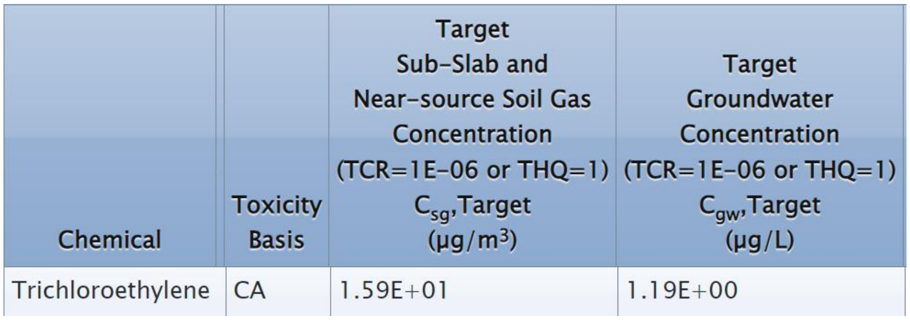
However, if site concentrations exceed the VISLs, it is recommended that the assumptions underlying the US EPA VISL calculations be reviewed and a determination made as to whether they are applicable at each site. Site-specific factors may result in unattenuated or enhanced transport of vapors towards a receptor, and consequently are likely to render the VISLs target subsurface concentrations overly or underly conservative.

Application of the VISLs is appropriate as a first-tier screening assessment for all sites except those where the following conditions apply. If any of the below are applicable to a site, a site-specific evaluation must be conducted:

- Very shallow groundwater sources [e.g., depth to water is less than five feet below foundation level];
- Shallow soil contamination resulting in vapor sources (e.g., VOCs are found at significant levels within 10 ft of the base of the foundation);
- Buildings with significant openings to the subsurface (e.g., sumps, unlined crawlspaces, earthen floors) or significant preferential pathways, either naturally occurring or anthropogenic (not including typical utility perforations present in most buildings);
- Vapor sources originating in landfills where methane is generated in sufficient quantities to induce advective transport into the vadose zone;
- Vapor sources originating in commercial or industrial settings where vapor-forming chemicals can be released within an enclosed space and the vapor density of a chemical may result in significant advective transport of the vapors downward through cracks and openings in floors and into the vadose zone; and/or
- Leaking vapors from gas transmission lines.

US EPA VISLs should be used as a tool to estimate potential cumulative risks and/or hazards from exposure to volatile and toxic chemicals at a site where the underlying assumptions are deemed appropriate and if further evaluation is required.

Below is a screenshot from the VISL calculator as an example of a soil gas and a groundwater vapor intrusion screening level for a residential scenario for TCE, based on a carcinogenic risk level of 1E-06 and a HI of 1.0. Once obtained, the VISLs are applied in a similar fashion to RSLs and incorporated into Equations 10 and 11, as appropriate.



Chemical	Toxicity Basis	Target Sub-Slab and Near-source Soil Gas Concentration (TCR=1E-06 or THQ=1) C_{sg},Target (µg/m^3)	Target Groundwater Concentration (TCR=1E-06 or THQ=1) C_{gw},Target (µg/L)
Trichloroethylene	CA	1.59E+01	1.19E+00

Figure 5. Example Output VISL Calculator

7.2 Construction Worker Trench Model

The following is excerpted from the Virginia Unified Risk Assessment Model – VURAM User Guide for Risk Assessors, August 2022

There are no well-established models available for estimating migration of volatiles from groundwater into a construction/utility trench. Virginia Department of Environmental Quality (VDEQ) recommends the following trench model, developed by VDEQ, for evaluating construction groundwater and soil gas. As construction workers are presumed to be adults, age-adjusted and mutagenic equations, as well as TCE and vinyl chloride specific equations, do not apply to the construction worker computations.

The models are based on a two-step process. First, a simple fate and transport equation of a vadose zone model to estimate volatilization of gases (emission flux of VOCs) from contaminated groundwater into the air of the trench. Then a box model is used to estimate dispersion of the contaminants from the air inside the trench into the above-ground atmosphere to estimate the EPC for air in a construction trench (Ctrench). For chemicals that are not included in the RSL table, calculate EPCs for air in a construction trench, following the soil gas equations. References should be provided for all chemical-specific parameters.

In October 2017, VDEQ revised the parameterization of the soil gas equations underlying the Construction Worker Trench Model. During a review of the equations and approaches utilized in the VDEQ's construction worker trench model, risk assessment staff identified the need for a modification to the soil gas trench model that evaluates risks from soil vapor to construction workers in a trench. Currently, VDEQ's application of the groundwater trench model assumes that the distance from the bottom of the trench to a vapor source is 31 centimeters (cm). This value is adjusted to 1 cm for the soil gas trench model; this change applies ONLY to the soil gas portion of the trench model. This modification is made because soil gas analytical results are direct measurements of vapors within the soil column that could be directly adjacent to the trench and diffusing directly through the trench walls.

It is a reasonable assumption that the contaminated source materials or soil gas would intersect with the trench walls. The change is also consistent with US EPA's recent acknowledgment that contaminated groundwater is not the only source of vapor and that soils saturated with volatiles can also be a significant driver of vapor contamination. As a result, there is a substantial change in the construction worker soil gas screening levels. Modifying the model in this way provides a more accurate representation of both exposures and risks to construction workers in these scenarios and is consistent with other regulatory agencies' approaches and their application of VDEQ's Construction Worker Trench Model.

VDEQ's Construction Worker Trench Model (groundwater) has been adopted by other state agencies because it captures scenarios involving the exposure of a construction worker to vapors from contaminated groundwater. With the 2017 revision of the soil gas portion, the Construction Worker Trench Model also captures scenarios involving exposure to gases directly measured in the trench and incorporates vapor concentrations directly measured in the subsurface.

Table 12. Trench Exposure Parameters (VDEQ, 2022)

Symbol	Description	Value	Units
TR-ACH	Trench Air Changes per Hour	2	(h)-1
TR-ACvad	Trench Advection Coefficient Groundwater greater than 15ft	0.25	(cm3/cm3)
TR-CF1	Trench Conversion Factor-1	0.001	(L/cm3)
TR-CF2	Trench Conversion Factor-2	10000	(cm2/m2)
TR-CF3	Trench Conversion Factor-3	3600	(s/hr)
TR-CF4	Trench Conversion Factor-4	1000000	(cm3/m3)
TR-D-dir	Trench Depth - groundwater less Than 15ft	2.44	(m)
TR-D-ind	Trench Depth - groundwater greater than 15ft	4.57	(m)
TR-Dsg	Trench - Depth to soil gas vapor source	1	(cm)
TR-EFcw	Trench Construction Worker Exposure Frequency	125	(days/yr)
TR-ETcw	Trench Construction Worker Exposure Time	4	(hrs/day)
TR-EVcw	Trench Construction Worker Events	1	(events/day)
TR-F	Trench Fraction of floor through which contaminant can enter	1	(unitless)
TR-HV	Trench Thickness of Vadose Zone - groundwater greater than 15 ft	30	(cm)
TR-IRcw	Trench Construction Worker Groundwater Ingestion Rate	0.02	(L/day)
TR- KGH2O	Trench Gas-phase mass transfer coefficient of water vapor at 25deg C	0.833	(cm/s)
TR-KLO2	Trench Liquid-phase mass transfer coefficient of oxygen at 25deg C	0.002	(cm/s)
TR-L	Trench Length	2.44	(m)
TR-Lgw	Trench Depth to groundwater	488	(cm)
TR-MWH2O	Trench Molecular Weight of Water	18	(unitless)
TR-MWO2	Trench Molecular Weight of Oxygen	32	(unitless)
TR-Porvad	Trench Porosity in Vadose Zone - groundwater greater than 15ft	0.44	(cm3/cm3)
TR-R	Trench Ideal Gas Constant	0.000082	(atm-m3/mol-K)
TR-Temp-F	Trench Temperature Fahrenheit	77	(F)
TR-Temp-K	Trench Temperature -	298	(K)
TR-W	Trench Width	0.91	(m)
TR-W/D	Trench Width to Depth Ratio	0.38	(unitless)

7.3 Groundwater

Two exposure scenarios are evaluated based on the site-specific depth of the groundwater: indirect contact based on contaminant transport through the vadose zone groundwater depth greater than 15 feet and direct contact based on groundwater pooling in the trench groundwater depth less than or equal to 15 feet. Two unique volatilization factors (VF) are computed for each chemical. For indirect contact, where the groundwater is greater than 15 feet, the VF Equation 16 is used. For direct contact, where the groundwater is less than 15 feet, VF Equation 18 is applied. VDEQ assumes that a construction project could result in an excavation as deep as 15 feet. At some sites there is a high probability that construction projects with deeper excavations may occur. Contact the DWMRC project manager and risk assessor to discuss the appropriate assumptions for site-specific parameters.

Equations 16 or 18 are used to calculate chemical-specific VF. Residential groundwater equations for noncancer adult and cancer and construction worker exposure values, are used to compute screening levels or hazard/risk values. The appropriate groundwater VF replaces the Andelman Volatilization Factor (K=0.5) in the residential groundwater equations. Airborne concentration of a contaminant in a trench can be estimated using Equation 15

$$C_{trench} = C_{gW} \times VF \qquad \text{Equation 15}$$

Where:

C_{trench} = Concentration of contaminant in trench, µg/m³
C_{gW} = Concentration of contaminant in groundwater, µg/L
VF = Volatilization factor (See Equations X and X), chemical-specific, L/ m^3

Groundwater Greater than 15 Feet Deep

$$VF = \frac{\left(H_i \times D_{air} \times AC_{vad}^{3.33} \times A \times F \times 10^{-3} \times 10^4 \times 3600\right)}{\left(R \times T \times L_d \times ACH \times V \times Pro_{vad}^2\right)} \qquad \text{Equation 16}$$

Where:

H_i = Henry's Law constant for contaminant (RSL table), atm-m³/mol
D_{air} = Diffusion coefficient in air (RSL table), cm²/s
AC_{vad} = Volumetric air content in vadose zone soil, cm³/cm³
A = Area of trench, m²
F = Fraction of floor through which contaminant can enter, unitless
R = Ideal gas constant, atm-m³/mole-°K
T = Average system absolute temperature, degree Kelvin (°K)
L_d = Distance between trench bottom and groundwater Equation X, cm
ACH = Air changes per hour, h-1
V = Volume of trench, m³
Por_{vad} = Total soil porosity in vadose zone, cm³/cm³
10^{-3} = Conversion factor, L/cm³
10^{4} = Conversion factor, cm²/m²
3600 = Conversion factor, s/hr

The value for R is 8.2 x 10^{-5} atm-m³/mole-°K. A default value of 298°K may be used for the average system absolute temperature.

Studies of urban canyons suggest that if the ratio of trench width -- relative to wind direction -- to trench depth is less than or equal to 1.0, a circulation cell or cells will be set up within the trench that limits the degree of gas exchange with the atmosphere. VDEQ has assumed an ACH in this case of 2/hr - based upon measured ventilation rates of buildings.

$$L_d = L_{gw} - D_{trench} \qquad \text{Equation 17}$$

Where:

L_{gw} = depth to groundwater, cm
D_{trench} = depth of trench, cm

Groundwater Less Than or Equal to 15 Feet Deep

If the depth to groundwater at a site is less than 15 feet, VDEQ assumes that a worker would encounter groundwater when digging an excavation or a trench. The worker would then have direct exposure to the groundwater. The worker would also be exposed to contaminants in the air inside the trench that would result from volatilization from the groundwater pooling at the bottom of the trench. VDEQ assumes that the trench would only intercept the groundwater for a few inches since a groundwater pool of more than a few inches would likely require dewatering. Therefore, trench depth should be set to equal the actual depth to groundwater at the site. Equation 18 is used to calculate VF for groundwater less than 15 feet deep.

$$VF = \frac{K_i \times A \times F \times 10^{-3} \times 10^4 \times 3600}{ACH \times V} \qquad \text{Equation 18}$$

Where:

K_i = Overall mass transfer coefficient of contaminant (Equation 19), cm/s
A = Area of trench, m^2
F = Fraction of floor through which contaminant can enter, unitless
ACH = Air changes per hour, h^{-1}
V = Volume of trench, m^3
10^{-3} = Conversion factor, L/cm^3
10^4 = Conversion factor, cm^2/m^2
3600 = Conversion factor, s/hr

$$K_i = \left[\frac{1}{k_{iL}} + \frac{(R \times T)}{H_i \times k_{iG}}\right]^{-1} \qquad \text{Equation 19}$$

Where:

k_{iL} = Liquid-phase mass transfer coefficient of i (Equation 20), cm/s
R = Ideal gas constant, atm-m^3/mole-°K
T = Average system absolute temperature, °K
H_i = Henry's Law constant for contaminant (RSL table), atm-m^3/mol
KiG = Gas-phase mas transfer coefficient of i (Equation 21), cm/s

The value for R is 8.2×10^{-5} atm-m^3/mole-°K. A default value of 298°K may be used for the average system absolute temperature.

$$k_{iL} = \left(\frac{MW_{O2}}{MW_i}\right)^{0.5} \times \frac{T}{298} \times k_{L,O2} \qquad \text{Equation 20}$$

Where:

k_{iL} = Liquid-phase mass transfer coefficient of I,cm/s
MW_{O2} = Molecular weight of O2, g/mol
MW_i = Molecular weight o component i, g/mol
T = Absolute temperature of system, °K
$k_{L,O2}$ = Liquid-phase mass transfer coefficient of oxygen at 25°C, cm/s

The value of $k_{L,\,O2}$ is 0.002 cm/s.

$$k_{iG} = \left(\frac{MW_{H2O}}{MW_i}\right)^{0.335} \times \left(\frac{T}{298}\right)^{1.005} \times k_{G,H2O} \qquad \text{Equation 21}$$

Where:

k_{iG} = Gas-phase mass transfer coefficient of component I, cm/s
MW_{H2O} = Molecular weight of water, g/mol
MW_i = Molecular weight o component i, g/mol
T = Absolute temperature of system, °K
$kG_{,H2O}$ = Gas-phase mass transfer coefficient of water vapor at 25°C, cm/s

The value of $k_{G,\,H2O}$ is 0.833 cm/s. (*Superfund Exposure Assessment Manual*, EPA, Office of Remedial Response, April, 1988.)

7.4 Soil Gas

This model can be used to estimate the contaminant concentration in soil vapor (C_{SV}) partitioning from the groundwater concentration. The contaminant is then transported by diffusion to the trench base or face (where applicable) and diluted by mixing within the trench. In order to accommodate the assumption that the construction worker could intersect with the sample collection depth, distance between the trench bottom and vapor source (L_d) is modified to 1 cm.

A unique, chemical-specific, dimensionless volatilization factor for soil vapor (VF_{SV}) is developed based on the groundwater VF_{gt} Equation 16. Trench dimensions remain consistent with groundwater equations. Apply the construction exposure parameters to the residential air equations for noncancer adult and cancer. The resulting hazard/risk is multiplied by the chemical- specific VF_{SV} as an attenuation factor to obtain a final hazard/risk value. Screening values are likewise computed by using the residential equations and then divided by VF_{SV}. The final screening value is the lower of the calculated noncancer/cancer screening values. Soil gas volatilization factor is based on groundwater depth greater than 15 feet, Equations 15 and 16. Combining these two equations yields:

$$C_{gw} = \frac{H_i \times D_{air} \times AC_{vad}^{3.33} \times A \times F \times 10^{-3} \times 10^{4} \times 3600}{R \times T \times L_d \times ACH \times V \times Por_{vad}^{2}} \qquad \text{Equation 22}$$

Where:

H_i = Henry's Law constant for contaminant (RSL table), atm-m³/mol
D_{air} = Diffusion coefficient in air (RSL table), cm²/s

AC_{vad} = Volumetric air content in vadose zone soil, cm³/cm³
A = Area of trench, m²
F = Fraction of floor through which contaminant can enter, unitless
R = Ideal gas constant, atm-m³/mole-°K
T = Average system absolute temperature, °K
L_d = Distance between trench bottom and groundwater Equation 17, cm
ACH = Air changes per hour, h-1
V = Volume of trench, m³
Por_{vad} = Total soil porosity in vadose zone, cm³/cm³
10^{-3} = Conversion factor, L/cm³
10^4 = Conversion factor, cm²/m²
3600 = Conversion factor, s/hr

Soil gas concentrations are estimated from groundwater concentrations using the following equations:

$$C_{sg} = \frac{HLC}{C_{gw}} \qquad \text{Equation 23}$$

$$HCL = \frac{H_i}{R \times T} \qquad \text{Equation 24}$$

Where:

Csg =Concentration in soil gas, µg/m³
HLC =Dimensionless Henry's Law Constant, (unitless)

Combining Equations 23 and 24 and solving for the groundwater concentration yields:

$$C_{gw} = C_{sg} \times \frac{R \times T}{H_i} \qquad \text{Equation 25}$$

Substituting Equation 25 in trench concentration equation yields:

$$C_{trench} = C_{sg} \times \frac{R \times T}{H_i} \times \frac{H_i \times D_{air} \times AC_{vad}^{3.33} \times A \times F \times 10^{-3} \times 10^4 \times 3600}{R \times T \times L_d \times ACH \times V \times Por_{vad}^2} \qquad \text{Equation 26}$$

Equation 26 simplifies to the following:

$$C_{trench} = C_{sg} \times \frac{D_{air} \times AC_{vad}^{3.33} \times A \times F \times 10^4 \times 3600}{L_d \times ACH \times V \times Por_{vad}^2 \times 10^6} \qquad \text{Equation 27}$$

Since the concentration in the trench is equal to the soil gas concentration times VF_{SV}:

$$V_{SV} = \frac{D_{air} \times AC_{vad}^{3.33} \times A \times F \times 10^4 \times 3600}{L_d \times ACH \times V \times Por_{vad}^2 \times 10^6} \qquad \text{Equation 28}$$

Where:

D_{air} = Diffusion coefficient in air (RSL table), cm²/s
AC_{vad} = Volumetric air content in vadose zone soil, cm³/cm³
A = Area of trench, m²

F = Fraction of floor through which contaminant can enter, unitless
L_d = Distance between trench bottom and groundwater Equation 17, cm
ACH = Air changes per hour, h^{-1}
V = Volume of trench, m^3
Por_{vad} = Total soil porosity in vadose zone, cm^3/cm^3
10^6 = Conversion factor, cm^3/cm^3
10^4 = Conversion factor, cm^2/m^2
3600 = Conversion factor, s/hr

8.0 SOIL-TO-GROUNDWATER

When closing or managing a contaminated site, the mass of contaminants in the source area should not increase. This means that levels of contamination in soil should not act as a continuing source for groundwater contamination. It is understood that naturally occurring variations in groundwater contaminant concentrations, natural groundwater flow, and dispersion of plumes will occur, but there should not be an on-going source for new contamination (e.g., contamination continuing to leach through soil or buried/leaking waste).

Future impacts to groundwater can be addressed by evaluating the potential for detected concentrations in soil at each site to contaminate groundwater via the soil-to-groundwater migration pathway. This may be achieved by following a stepwise approach.

8.1 Step 1 – Generic SSLs

To assess the potential of contamination migrating through soil to groundwater, the Protection of Groundwater soil screening levels (SSLs) from the US EPA RSL tables should be used. The RSL tables may list two protection of groundwater SSLs: risk-based and/or MCL-based. <u>If the RSL table lists a value for both a risk-based and an MCL-based SSL, the least conservative (greater of the two values) may be used for comparison to site data</u>.

The SSLs listed in the RSL tables are based on a dilution attenuation factor (DAF) of 1. The DAF is a function of the hydraulic conductivity of the aquifer, infiltration rate, mixing zone, and length of the source area parallel to groundwater flow. A DAF of one assumes that no dilution or attenuation occurs within the unsaturated zone to the water table. Adsorption and degradation are not considered, and the assumption is that the contaminant in soil comes into immediate contact with groundwater. The higher the DAF value, the greater the degree of dilution and attenuation of contaminants along the flow path. The DWMRC has established that a DAF of 20 (US EPA, 2002a) is protective of groundwater for most sites in Utah.

Because of assumptions used in SSL model approach, use of the DAF model may be inappropriate for certain conditions, including sites where:

- Adsorption or degradation processes are expected to significantly attenuate contaminant concentrations in the soil or aquifer media;
- Saturated thickness is significantly less than 12 meters thick;

- Fractured rock or karst aquifer types exist (violates the unconfined, unconsolidated, homogeneous, isotropic assumptions);
- Facilitated transport is significant (colloidal transport, transport via dissolved organic matter, or transport via solvents other than water); and/or
- NAPLs are present.

For sites that have these types of conditions, consideration should be given to application of a more detailed site-specific analysis than either the generic or site-specific models described herein.

The use of the SSL based on a DAF of 20 is advised for Step 1. Therefore, RSL SSLs, which are based on a DAF of 1.0, will require modification to reflect values based on a DAF of 20 (i.e., multiply the RSL SSL by 20).

1. Compare the maximum detected concentration for COPCs in soil to the US EPA RSL SSLs based on the DAF of 20. This is simply a point-to-point comparison, as shown in the example below. The maximum detected site soil concentration regardless of depth should be used.

Table 13. SSL Example

Constituent	**Max (mg/kg)**	**SSL DAF 20 (mg/kg)**	**Site Max> SSL?**
Barium	5.14E+02	3.20E+03	No
Mercury	2.35E+00	2.00E+00	**Yes**
Benzo(a)pyrene	2.21E+00	4.80E+00	No
Naphthalene	3.47E-01	1.08E-02	**Yes**

2. If the maximum detected concentration exceeds the SSL DAF 20, the potential exists for future impacts to groundwater. If the potential for future groundwater contamination exists, additional lines of evidence and a re-evaluation using a refined EPC (95UCL) may be provided. If sufficient data are not available to calculate a 95UCL, the maximum constituent of potential concern concentration value shall be used for evaluation, or an alternate value for a revised EPC may be proposed.

Table 14. Refined SSL Evaluation Example

Constituent	**95UCL[1] (mg/kg)**	**SSL DAF 20 (mg/kg)**	**95UCL[1]> SSL?**
Mercury	1.25E+00	2.00E+00	No
Naphthalene	1.88E-01	1.08E-02	**Yes**
[1] Less than four detections were available in the dataset for naphthalene, the median concentration was used as the refined EPC (US EPA, 2022).			

If the results of the comparison to the SSLs using the refined EPC are acceptable, no additional analysis is warranted. If the analysis shows potential for contamination of groundwater, Step 2 should be followed.

8.2 Step 2 – Site-specific DAF

If maximum detected concentrations and/or revised EPCs in soil at a site exceed the generic soil-to-groundwater SSLs (Step 1), then site-specific soil-to-groundwater SSLs may be estimated, and the Step 2 approach followed. As stated in US EPA (1996a), the calculation of soil-to-groundwater SSLs is most sensitive to the DAF. Unless sufficient data are available to calculate a site-specific DAF, there is little benefit derived from using the site-specific SSLs instead of the generic SSLs.

The development of the site-specific dilution attenuation factor should follow US EPA's *Supplemental Guidance for Developing Soil Screening Levels*. US EPA's *Supplemental Soil Screening Guidance: Technical Background Document* (US EPA 1996a) and *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* (US EPA 2002a), or the most current US EPA guidance. Estimation of contaminant release in soil leachate is based on the Freundlich adsorption isotherm. The Freundlich equation was modified to relate the sorbed concentration to the total concentration measured in a soil sample (which includes contaminants associated with solid soil, soil-water and soil-air components) (Feenstra 1991). Equation 29, given below, is used to calculate SSLs corresponding to target soil leachate concentrations (C_w).

Equation 29
Soil Screening Level for Leaching to Groundwater Pathway

$$SSL = C_w \ x \left[K_d + \left(\frac{\theta_w + \theta_a H'}{\rho_b} \right) \right]$$

Parameter	Definition (units)	Default
SSL	Soil Screening Level for migration to groundwater pathway (mg/kg)	Chemical-Specific
C_w	Target soil leachate concentration (mg/L)	Chemical-Specific
K_d	Soil /water partition coefficient (L/kg)	Chemical-Specific
θ_w	Water-filled soil porosity (L_{water}/L_{soil})	0.26
θ_a	Air-filled soil porosity (L_{air}/L_{soil}), n - θ_w	0.17
n	Total soil porosity (L_{pore}/L_{soil}), 1 - (ρ_b/ρ_s)	0.43
ρ_s	Soil particle density (kg/L)	2.65
ρ_b	Dry soil bulk density (kg/L)	1.5
H´	Dimensionless Henry's Law constant	Chemical-Specific

Target soil leachate concentrations (C_w) are equivalent to either the tap water SSLs or an MCL multiplied by a DAF, as follows:

$$C_w = \text{Tap Water SSL} \times \text{DAF}$$
or
$$C_w = \text{MCL} \times \text{DAF}$$
Equation 30

Contaminants transported as a leachate through soil to groundwater are affected by physical, chemical, and biological processes that can significantly reduce their concentration. These processes include adsorption, biological degradation, chemical transformation, and dilution from mixing leachate with groundwater. The total reduction in concentration between the source of the contaminant (vadose zone soil) and the point of groundwater withdrawal is defined as the ratio of contaminant concentration in soil leachate to the concentration in groundwater at the point of withdrawal. This ratio is termed a dilution/attenuation factor (DAF; US EPA 1996a and 1996b). The higher the DAF value the greater the degree of dilution and attenuation of contaminants along the migration flow path. A DAF of one implies no reduction in contaminant concentration occurs.

Development of the RSL SSLs considers only the dilution of contaminant concentration through mixing with groundwater in the aquifer directly beneath the source. This is consistent with the conservative assumptions used in the SSL methodology including an infinite source, soil contamination extending from surface to groundwater and the point of exposure occurring at the downgradient edge of the source. The ratio of contaminant concentration in soil leachate to the concentration in groundwater at the point of withdrawal that considers only dilution processes is calculated using the simple water balance equation (Equation 31), described below.

Equation 31
Dilution/Attenuation Factor (DAF)

$$\text{DAF} = 1 + \left(\frac{K \times i \times D}{I \times L} \right)$$

Where:

$$D = \left(0.0112 \times L^2\right)^{0.5} + D_a \left(1 - \exp\left[\frac{-L \times I}{K \times i \times D_a}\right]\right)$$

Parameter	Definition (units)	Default
DAF	Dilution/attenuation factor (unitless)	Site-Specific
K	Aquifer hydraulic conductivity (m/yr)	Site-Specific
i	Hydraulic gradient (m/m)	Site-Specific
D	Mixing zone depth (m)	Site-Specific
I	Infiltration rate (m/yr)	Site-Specific
L	Source length parallel to groundwater flow (m)	Site-Specific
D_a	Aquifer thickness (m)	Site-Specific

Most of these parameters are available from routine environmental site investigations. The mixing zone depth incorporates one additional parameter, the aquifer thickness (D_a).

If the 95% UCL concentration exceeds the calculated groundwater protection soil screening level, the potential exists for future impacts to groundwater. The groundwater protection soil screening level value shall be the greater of either the maximum contaminant level or the risk-based groundwater protection soil screening level value for evaluation. If the potential for future groundwater contamination exists, the responsible party may choose to submit a work plan for approval by the director describing actions that will be taken to protect groundwater from future impacts due to soil contamination. In addition, the work plan shall include a proposal for collection of sufficient monitoring data to evaluate both current and future groundwater conditions. Alternatively, an alternative method as outlined in Step 3 may be applied.

8.3 Step 3 – Alternative Methods

An alternate method for evaluating potential future impacts to groundwater due to soil contamination may be proposed to the DWMRC for approval. If it is determined that the potential for future groundwater contamination exists, a work plan should be submitted for approval by the DWMRC describing actions that will be taken to protect groundwater from future impacts due to soil contamination. In addition, the work plan should include a proposal for collection of sufficient monitoring data to evaluate both current and future groundwater conditions.

Alternative methods may include site-specific fate and transport modeling (using commercially available programs capable of reproducing known groundwater contamination). In addition, weight of evidence may be provided. Discussions should include frequency of detection, magnitude of detected concentrations, soil profiles and extent of contamination, and history of the contamination at the site. Other site-specific issues may include the potential for dense, sinking vapors acting as a source for contamination.

9.0 ECOLOGICAL RISK ASSESSMENT

Ecological risk assessments (ERAs) are required at sites where it has been determined that exposure pathways are potentially complete for ecological receptors. A complete exposure pathway consists of 1) a source; 2) a mechanism of contaminant release; 3) a receiving or contact medium; 4) a potential receptor population; and 5) an exposure route. In order for a potential receptor population to exist, sites must contain open areas that would allow plant growth and suitable habitat for wildlife. Pathways may be incomplete for ecological receptors at sites in industrial areas or are filled in with concrete or pavement; in these cases, an ecological waiver may be granted (refer to Section 6.1).

In accordance with US EPA ERA guidance (US EPA 1997c), the objectives of the ERAs are to 1) document whether actual or potential ecological risks exist at a site; 2) identify which contaminants present at a site pose an ecological risk; and 3) generate data to be used in evaluating cleanup options, if warranted. The ERAs should be conducted in accordance with US EPA guidance and general processes consisting of four main components: 1) problem formulation; 2) exposure assessment; 3) toxicity assessment; and 4) risk characterization.

The ERAs may follow a tiered approach, with each tier including problem formulation, exposure assessment, toxicity assessment, and risk characterization. The Tier 1 assessment is a screening level assessment that utilizes conservative assumptions. If the results of the Tier 1 assessment indicate potential for adverse risk, then a Tier 2 assessment will be conducted. The Tier 2 assessment provides a more refined screening analysis utilizing some site-specific information. If the results of the Tier 2 assessment indicate potential for adverse risk, then a Tier 3 site-specific risk assessment or additional site actions may be warranted.

The Texas Commission on Environmental Quality has a website that provides several useful links to US EPA guidance, screening levels, wildlife exposures, and identification of species, that may help in completed an ecological assessment.
https://www.tceq.texas.gov/remediation/eco/eco_links.html

9.1 Ecological Waiver

Site investigations must include an evaluation of human health and ecological risk to support risk-based closure. An ecological risk assessment is warranted when it has been determined that exposure pathways are potentially complete for ecological receptors. A complete pathway consists of 1) a source, 2) a mechanisms of contaminant release, 3) a receiving or contact medium, 4) a potential receptor population, and 5) and exposure route. Of these five criteria, the most fundamental is the fourth criterion. In order for a potential receptor population to exist, a site must contain open areas that would allow plant growth and suitable habitat for wildlife. Pathways are incomplete for ecological receptors at sites that are completely filled-in with buildings, concrete, or pavement. For these areas, a risk assessment cannot be completed, and a waiver may be requested in lieu of a quantitative risk assessment.

Environmental conditions at the site may be used to eliminate the need for ecological risk assessment and support an ecological waiver include:

- The affected property is not a viable habitat,
- The site cannot be used by potential ecological receptors as a habitat, and/or
- Complete or potentially complete exposure pathways do not exist due to prevailing conditions or property setting.

Photographs of the property are useful in showing the state of potential habitat, such as if the site is completely paved and/or covered in structures with minimal or no vegetation and devoid of habitat. In addition, the property may be in an area that is highly industrialized, consisting of paved/cemented lots and industrial-use buildings. A discussion of surrounding lots and the potential or lack of potential for nearby habitat is also helpful as well as a discussion of any observations of wildlife using the property for permanent habitat or food. While it is possible that some species could be casually present (such as birds resting in nearby trees or crossing the property), it may not plausible that any receptor would forage, nest or den on the property itself due to a complete lack of vegetation.

Using the above, it can be demonstrated that there is no complete exposure pathway, and an ecological risk assessment is not deemed required. A formal request for a waiver for conducting a quantitative ecological risk assessment should be submitted along with all potential lines of evidence to justify minimal impact on ecological receptors. A waiver may be submitted as a standalone document or contained within a site characterization report.

9.2 Tier 1

The objective of the Tier 1 screening-level ERA (SLERA) is to determine whether: 1) there are any potential adverse effects for ecological receptors; and 2) there may be potential adverse health effects to ecological receptors, and further evaluation of ecological risk is warranted. The SLERA should contain a detailed discussion of each of these items.

- Characterization of the environmental setting, including current and future land uses. Ecological assessments must include the evaluation of present-day conditions and land uses but also evaluate future land uses.
- Identification of known or likely chemical stressors (chemicals of potential ecological concern, COPECs). The characterization data from the site (e.g., facility investigation) is evaluated to determine what constituents are present in which media.
- Identification of the fate and transport pathways that are complete. This includes an understanding of how COPECs may be mobilized from one medium to another.
- Identification of the assessment endpoints that should be used to assess impact of the receptors; what is the environmental value to be protected.
- Identification of the complete exposure pathways and exposure routes. What is the impacted medium/media (soil, surface water, sediment, groundwater, and/or plants) and how might the representative receptors be exposed (direct ingestion, inhalation, and/or direct contact)?
- Species likely to be impacted and selection of representative receptors. From the list of species likely to be present on-site, what species are to be selected to represent specific trophic levels?

9.2.1 COPECs

The identification of COPECs for the ecological risk assessments will follow the same methodology as presented in Section 4.0 for organic and inorganic constituents. For ecological assessments, the potential for a chemical to be bioaccumulative should be considered when identifying COPECs.

Burrowing animals and plant roots would be exposed to COPECs in deeper soils, whereas all other animals would only be exposed to surface soils. Concentrations of contaminants in soil 0-6 ft bgs will be assessed for burrowing animals and deep-rooted plants while concentrations in soil 0-1 ft bgs will be assessed for all other receptors.

9.2.2 EPCs

The Tier 1 exposure assessment consists of estimating exposure doses based on conservative exposure assumptions and maximum detected concentrations within the defined exposure intervals. EPCs are discussed in Section 4.2.

9.2.3 Receptors

Sites may include a wide range of terrestrial, semi-aquatic, and aquatic wildlife. A generalized food web for soil is shown in Figure 6. Wildlife receptors for the SLERA should be selected to represent the trophic levels and habitats present or potentially present at the site and include any Federal threatened and endangered species and State sensitive species.

As there are typically numerous species of wildlife and plants present at a given facility or site and in the surrounding areas, only a few key receptors need to be selected for quantitative evaluation in the SLERA, which are representative of the ecological community and varying trophic levels in the food web. Possible receptors that may be evaluated in the SLERAs at each site include the following:

- Plant community,
- Deer mouse,
- Horned lark,
- Kit fox (evaluated at sites greater than 267 acres),
- Pronghorn (evaluated at sites greater than 342 acres), and
- Red-tailed hawk (evaluated at sites greater than 177 acres).

The above key receptors selected as the representative species represent the primary producers as well as the three levels of consumer (primary, secondary, and tertiary) for the most common receptors found at hazardous waste sites in Utah. If water bodies are present, and aquatic receptors are viable, DWMRC should be consulted to discuss appropriate identification of receptor species, pathways, and SLERA methodologies.

9.2.3.1 Plants

The plant community will be evaluated quantitatively in the SLERAs at all sites. Specific species of plants will not be evaluated separately; rather the plant community will be evaluated as a whole. The plant community provides a necessary food source directly or indirectly through the food web for wildlife receptors.

9.2.3.2 Deer Mouse

The deer mouse (Peromyscus maniculatus) is a common rodent throughout much of North America and it can thrive in a variety of habitats. The deer mouse was selected as a representative receptor because it is prevalent near most sites in Utah, and it represents one of the several species of omnivorous rodents that may be present at sites. Small rodents are also a major food source for larger omnivorous and carnivorous species. The deer mouse receptor will be evaluated at all sites, regardless of size. The deer mouse has a relatively small home range and could therefore be substantially exposed to COPECs at sites if their home range is located within a solid waste management unit (SWMU) or other corrective action sites.

Based on a review of literature (OEHHA, 1999) and from the Natural Diversity Information Source (CDW, 2011), a dietary composition consisting of 26% invertebrates and 74% plant matter will be assumed for the deer mouse.

9.2.3.3 Horned Lark

The horned lark (*Eremophila alpestris*) is a common widespread terrestrial bird. It spends much of its time on the ground and its diet consists mainly of insects and seeds. The horned lark receptor was chosen because it is prevalent in Utah and represents one of the many small terrestrial bird species that could be present. Since the horned lark spends most of its time on the ground, it also provides a conservative measure of effect since it has a higher rate of incidental ingestion of soil than other songbirds. The horned lark is also a major food source for omnivorous intermediate species, and top avian carnivores. The horned lark will be evaluated based on an omnivorous diet of invertebrates and plant matter. The horned lark receptor will be evaluated at all sites, regardless of size. The horned lark has a relatively small home range and could therefore be substantially exposed to COPECs at sites if their home range is located within a SWMU or other corrective action units.

It will be assumed that the horned lark’s diet consists of 75% plant matter, and 25% animal matter based on a study conducted by Doctor, *et al*, 2000.

9.2.3.4 Kit Fox

The kit fox (*Vulpes macrotis*) is native to the western United States and Mexico. Its diet consists of mostly small mammals. Although the kit fox’s diet may also consist of plant matter during certain times of the year, the kit fox will be evaluated as a carnivore, with a diet consisting of 100% prey items. It was selected as a key receptor because it is a sensitive species and is common in Utah, and the surrounding area at most sites in Utah provides suitable habitat for the kit fox. The kit fox also is representative of a mammalian carnivore within the food web.

The kit fox will only be evaluated at sites that are larger than 276 acres. A kit fox has a large home range size (2767 acres) (Zoellick & Smith, 1992) and it is assumed that risks are negligible from exposure to COPECs at sites that are less than 10% of the receptors home range. Unless the area use factor (AUF) is at least 10%, food items potentially contaminated with COPECs and incidental soil ingestion at the site would not contribute significantly to the receptor’s diet and exposure to COPECs. The kit fox diet will be based on composition of 100% prey.

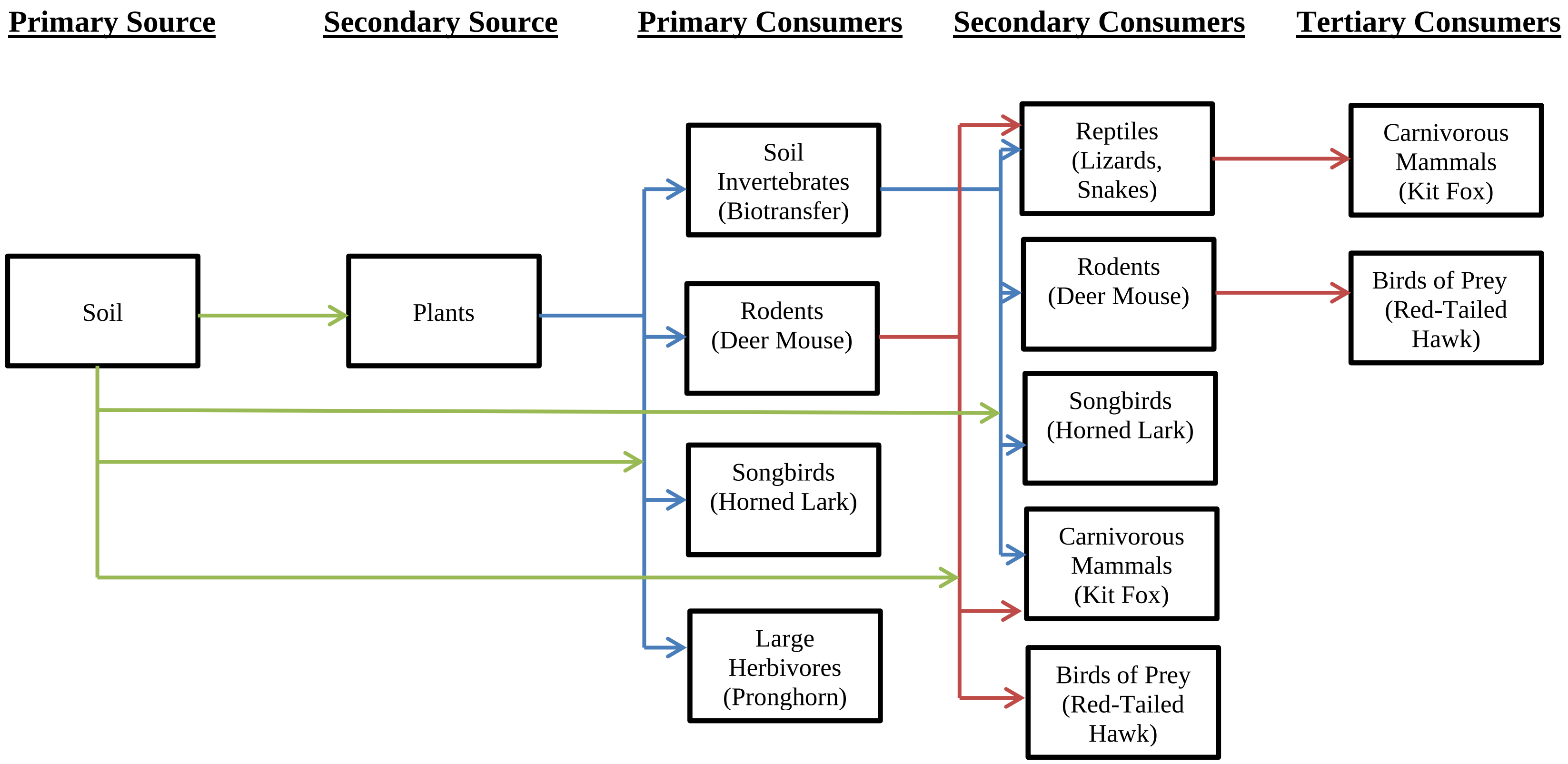


Figure 6. Generalized Food Web for Soil

9.2.3.5 Red-Tailed Hawk

The red-tailed hawk (*Buteo jamaicensis*) was selected as a top carnivore avian key receptor. The red-tailed hawk is widespread throughout Utah and is one of the most common birds of prey. It hunts primarily rodents, rabbits, birds, and reptiles. The red-tailed hawk was chosen as a key receptor since it is a common species throughout Utah. The red-tailed hawk will only be evaluated at sites that are larger than 177 acres. The red-tailed hawk has a large home range size (1770 acres) (US EPA, 1993b), and risks to the red-tailed hawk from exposure to COPECs at sites smaller than 177 acres (10% of the home range) would be negligible. The red-tailed hawk diet will be based on composition of 100% prey.

9.2.3.6 Pronghorn Antelope

The pronghorn (*Antilocapra Americana*) is a popular big game species that occurs in western Canada, United States, and northern Mexico. Its diet consists mainly of sagebrush and other shrubs, grasses, and forbs. The pronghorn was selected as a key receptor representative of large herbivorous species of wildlife. The pronghorn will only be evaluated at sites that are larger than 342 acres. The pronghorn has a large home range size (3422 acres) (Reynolds, 1984), and risks to the pronghorn from exposure to COPECs at sites smaller than 342 acres (10% of the home range) would be negligible. It is assumed that 100% of the diet is from grazing.

9.2.4 Exposure Pathways

A CSM (refer to Section 3.1) provides a summary of potentially complete exposure pathways, along with potentially exposed receptor types. A complete exposure pathway is defined as a pathway having all the following attributes:

- A source and mechanism for hazardous waste/constituent release to the environment,
- An environmental transport medium or mechanism by which a receptor can encounter the hazardous waste/constituent,
- A point of receptor contact with the contaminated media or via the food web, and
- An exposure route to the receptor.

If any of the above components are missing from the exposure pathway, it is not a complete pathway for the site. A discussion regarding all possible exposure pathways and the rationale/justification for eliminating any pathways will be included in the risk assessment.

Affected media that ecological receptors may be exposed to at sites are soil, biota, and surface water or groundwater (through springs). Surface water, sediment, and groundwater should be evaluated based on site-specific conditions.

Wildlife receptors could be exposed to COPECs that have been assimilated into biota. Ingestion of contaminated plant and animal matter, as a necessary component of the receptor's diet, will be evaluated quantitatively in the SLERAs. However, for the Tier 1 SLERA, it will conservatively be assumed that 100% of the wildlife receptors' dietary intake consists of site soil.

For soil, two soil intervals should be evaluated:

- For all non-burrowing receptors and for shallow-rooted plants, the soil exposure interval is typical of surface conditions and is considered to be between zero (0) and one (1) foot bgs.
- For all burrowing receptors (and receptors that may use borrows) and deep-rooted plants, the soil interval to be evaluated is 0 – 6 feet bgs.

Table 15. Ecological Soil Exposure Intervals

Receptor	Exposure Intervals (Soil)
Ecological Receptors (non-burrowing and shallow rooted plants)	0 – 1 foot bgs
Ecological Receptors (burrowing and deep-rooted plants)	0 – 6 feet bgs

9.2.5 Exposure Assessment

The effects assessment evaluated the potential toxic effects on the receptors being exposed to the COPECs. The effects assessment includes selection of appropriate toxicity reference values (TRVs) for the characterization and evaluation of risk. TRVs are receptor and chemical specific exposure rates at which no adverse effects have been observed, or at which low adverse effects are observed. TRVs that are based on studies with no adverse effects are called no observed adverse effects levels (NOAELs). TRVs that are based on studies with low adverse effects are termed lowest observed adverse effects levels (LOAELs).

For the initial SLERA, the preference for TRVs is based on chronic or long-term exposure, when available. The TRVs should be selected from peer-reviewed toxicity studies and from primary literature. Initial risk characterization should be conducted using the lowest appropriate chronic NOAEL for non-lethal or reproductive effects. If a TRV is not available and/or no surrogate data could be identified, the exclusion of potential toxicity associated with the COPEC will be qualitatively addressed in the uncertainty analysis of the risk assessment. Other factors that may be included in this discussion are frequency of detection, depth of detections, and special analysis of the detections.

9.2.6 Dose

For the initial SLERA, conservative assumptions should be applied as follows:

- The maximum detected concentrations for the exposure interval listed in Table 13 will be utilized in calculating exposure doses.

- 100% of the diet is assumed to contain the maximum concentration of each COPEC detected in the site media.
- Minimum reported body weights should be applied.
- Maximum dietary intake rates should be used.
- It will be assumed that 100% of the diet consists of direct ingestion of contaminated soil.
- It is assumed that the bioavailability is 100% at each site.
- Foraging ranges are initially set equal to the size of the site being evaluated. This means that the AUF in the SLERA is set to a value of one.
- Because body weight is reported as wet-weight (kg), and soil concentrations are reported as dry-weight (mg/kg), a wet-weight to dry-weight conversion factor of 0.22 (assuming 78% moisture content) will also be applied when calculating exposure doses.

The equation and exposure assumptions for calculating the Tier 1 exposure doses for the deer mouse are presented in Equation 32.

Equation 32. Calculation of Tier 1 Exposure Dose for COPECs in Soil; Deer Mouse

$$Exposure\ Dose = \frac{(C_s \times (IR * ww{:}dw)\ \times AUF)}{BW}$$

Parameter	**Definition (units)**	**Value**	**Reference**
Exposure Dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	calculated	--
C_s	Chemical concentration in soil (mg/kg)	Site-specific	Maximum detected concentration (0-10 ft bgs)
IR	Ingestion rate (kg food [ww]/day)	0.007	Maximum reported total dietary intake (US EPA, 1993g)
ww:dw	Wet-weight to dry weight conversion factor for ingested matter (kg [dw]/kg [ww])	0.22	78-percent moisture
AUF	Area use factor (the ratio of the site exposure area to the receptor foraging range) (unitless)	1	Maximum possible value
BW	Body weight (kg)	0.014	Minimum reported adult body weight (CDW, 2011)

The equation and exposure assumptions for calculating the Tier 1 exposure dose for the horned lark are presented in Equation 33.

Equation 33. Calculation of Tier 1 Exposure Dose for COPECs in Soil; Horned Lark

$$Exposure\ Dose = \frac{(C_s \times (IR * ww{:}dw) \times AUF)}{BW}$$

Parameter	Definition (units)	Value	Reference
Exposure Dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	Calculated	--
C_s	Chemical concentration in soil (mg/kg)	Site-specific	Maximum detected concentration (0-1 ft bgs)
IR	Ingestion rate (kg food [ww]/day)	0.024	Maximum reported total dietary intake; American robin (US EPA, 1993g)
ww:dw	Wet-weight to dry weight conversion factor for ingested matter(kg [dw]/kg [ww])	0.22	78-percent moisture
AUF	Area use factor (the ratio of the site exposure area to the receptor foraging range) (unitless)	1	Maximum possible value
BW	Body weight (kg)	0.025	Minimum reported adult body weight (Trost, 1972)

The equation and exposure assumptions for calculating the Tier 1 exposure doses for the kit fox are presented in Equation 34.

Equation 34. Calculation of Tier 1 Exposure Dose for COPECs in Soil; Kit Fox

$$Exposure\ Dose = \frac{(C_s \times (IR * ww{:}dw) \times AUF)}{BW}$$

Parameter	Definition (units)	Value	Reference
Exposure Dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	calculated	--
C_s	Chemical concentration in soil (mg/kg)	Site-specific	Maximum detected concentration (0-10 ft bgs)
IR	Ingestion rate (kg food [ww]/day)	0.18	Maximum reported total dietary intake (OEHHA, 2003)
ww:dw	Wet-weight to dry weight conversion factor for ingested matter (kg [dw]/kg [ww])	0.22	78-percent moisture
AUF	Area use factor (the ratio of the site exposure area to the receptor foraging range) (unitless)	1	Maximum possible value
BW	Body weight (kg)	1.6	Minimum reported adult body weight (OEHHA, 2003)

The equation and exposure assumptions for calculating the Tier 1 exposure doses for the red-tailed hawk are presented in Equation 35.

Equation 35 Calculation of Tier 1 Exposure Dose for COPECs in Soil; Red-tailed Hawk

$$Exposure\ Dose = \frac{(C_s\ \times (IR * ww{:}dw) \times AUF)}{BW}$$

Parameter	Definition (units)	Value	Reference
Exposure Dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	Calculated	--
C_s	Chemical concentration in soil (mg/kg)	Site-specific	Maximum detected concentration (0-1 ft bgs)
IR	Ingestion rate (kg food [ww]/day)	0.12	Maximum reported total dietary intake (US EPA, 1993g)
ww:dw	Wet weight to dry weight conversion factor for ingested matter (kg [dw]/kg [ww])	0.22	78-percent moisture
AUF	Area use factor (the ratio of the site exposure area to the receptor foraging range) (unitless)	1	Maximum possible value
BW	Body weight (kg)	0.96	Minimum reported adult body weight (US EPA, 1993g)

The equation and exposure assumptions for calculating the Tier 1 exposure doses for the pronghorn are presented in Equation 36.

Equation 36. Calculation of Tier 1 Exposure Dose for COPECs in Soil; Pronghorn

$$Exposure\ Dose = \frac{(C_s \times (IR * ww{:}dw) \times AUF)}{BW}$$

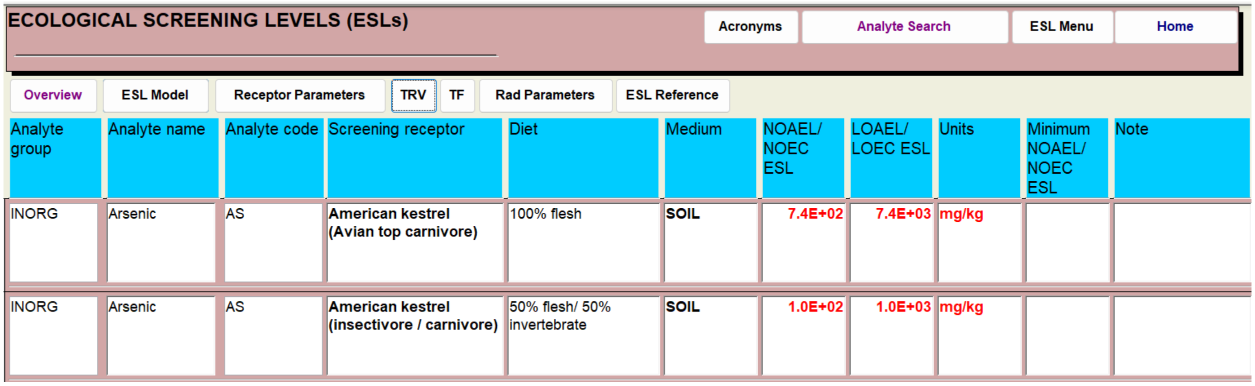
Parameter	Definition (units)	Value	Reference
Exposure Dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	calculated	--
C_s	Chemical concentration in soil (mg/kg)	Site-specific	Maximum detected concentration (0-1 ft bgs)
IR	Ingestion rate (kg wet matter/day) Based on equation: $IR=a(BW)^b$ where: a=2.606, b=0.628	0.74	Dry matter intake rate for herbivores (based on Nagy, 2001)
ww:dw	Wet weight to dry weight conversion factor for ingested matter (kg [dw]/kg [ww])	0.22	78-percent moisture
AUF	Area use factor (the ratio of the site exposure area to the receptor foraging range) (unitless)	1	Maximum possible value
BW	Body weight (kg)	47	Minimum reported adult body weight (O'Gara, 1978)

Exposure doses will not be calculated for plants. For the Tier 1 exposure assessment, it will be assumed that the exposure concentrations for plants are equal to the maximum detected concentrations of COPECs in soil.

9.2.7 Toxicity and Risk Characterization

For the Tier 1 ERAs, toxicity reference values will be selected based on NOAELs. TRVs will be obtained from literature and available databases such as Los Alamos National Laboratory's (LANL) EcoRisk database.

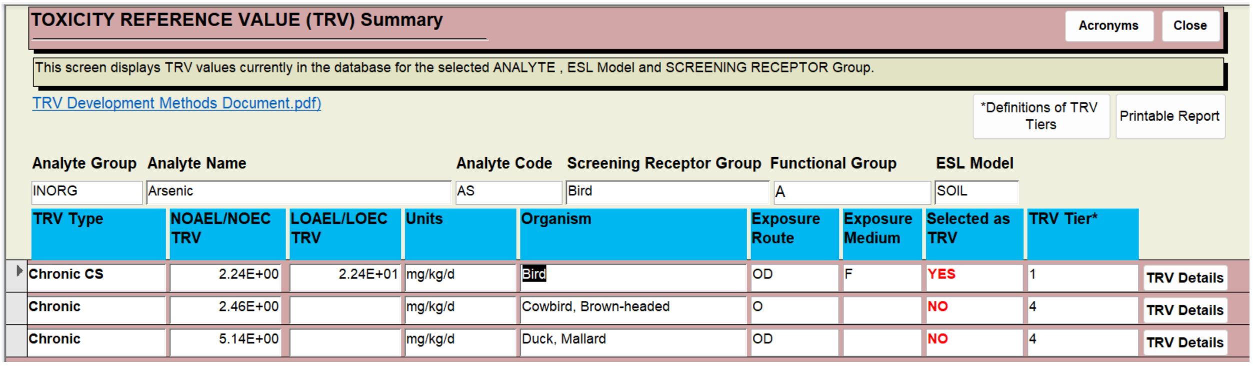
Note: If using the LANL EcoRisk database, caution must be taken. EcoRisk provides pre-calculated ecological screening levels that are based on LANL-specific assumptions. The main search engine will provide ecological screening levels (ESLs) for receptors based on NOAEL and LOAEL toxicity data (see Figure 7). When calculating doses using the above equations, the TRVs and not the ESLs must be used (as shown in Figure 8).



ECOLOGICAL SCREENING LEVELS (ESLs)

Analyte group	Analyte name	Analyte code	Screening receptor	Diet	Medium	NOAEL/ NOEC ESL	LOAEL/ LOEC ESL	Units	Minimum NOAEL/ NOEC ESL	Note
INORG	Arsenic	AS	**American kestrel (Avian top carnivore)**	100% flesh	**SOIL**	**7.4E+02**	**7.4E+03**	**mg/kg**		
INORG	Arsenic	AS	**American kestrel (insectivore / carnivore)**	50% flesh/ 50% invertebrate	**SOIL**	**1.0E+02**	**1.0E+03**	**mg/kg**		

Figure 7. Snapshot of LANL EcoRisk and ESLs



TOXICITY REFERENCE VALUE (TRV) Summary

This screen displays TRV values currently in the database for the selected ANALYTE , ESL Model and SCREENING RECEPTOR Group.

Analyte Group	Analyte Name	Analyte Code	Screening Receptor Group	Functional Group	ESL Model
INORG	Arsenic	AS	Bird	A	SOIL

TRV Type	NOAEL/NOEC TRV	LOAEL/LOEC TRV	Units	Organism	Exposure Route	Exposure Medium	Selected as TRV	TRV Tier*	
Chronic CS	2.24E+00	2.24E+01	mg/kg/d	Bird	OD	F	**YES**	1	**TRV Details**
Chronic	2.46E+00		mg/kg/d	Cowbird, Brown-headed	O		**NO**	4	**TRV Details**
Chronic	5.14E+00		mg/kg/d	Duck, Mallard	OD		**NO**	4	**TRV Details**

Figure 8. Snapshot of LANL EcoRisk and TRVs

In lieu of using EcoRisk or other US EPA ecological databases, a review of literature may be conducted to determine if data are available to either derive a TRV or if an appropriate surrogate can be applied. If a new TRV is derived, the TRV and supporting data will be provided to the DWMRC for approval.

If a TRV is not available and/or no surrogate data could be identified, the exclusion of potential toxicity associated with the COPEC will be qualitatively addressed in the uncertainty analysis of the risk assessment. Other factors that may be included in this discussion are frequency of detection, depth of detections, and special analysis of the detections.

For plants, the Tier 1 screening level hazard quotients for plants will be calculated by comparing exposure doses (i.e., maximum detected concentrations of COPECs; 0-1 ft bgs for shallow rooted plans or 0-10 ft bgs for deep rooted plants) to an effect concentration. The equation for screening level hazard quotient (SLHQ) for plants is shown in Equation 37.

Equation 37. Calculation of Screening-Level Hazard Quotients for Plant Receptors

$$SLHQ = \frac{C_s}{Effect\ Concentration}$$

Parameter	Definition (units)
SLHQ	Screening level hazard quotient (unitless)
C_s	Chemical concentration in soil (mg COPEC / kg soil dry weight), (0-1 ft bgs shallow-rooted and 0-6 ft bgs deep rooted plants)
Effect Concentration	Concentration at which adverse effects are not expected (mg/kg)

Tier 1 SLHQs for wildlife receptors will be calculated by comparing estimated exposure doses derived using Equations 32 through 36 for each of the key receptors determined to have complete habitat and exposure pathways at the site to NOAEL-based TRVs. The derivation of SLHQ for the key receptors (except plants) is shown in Equation 38.

Equation 38. Calculation of Screening-Level Hazard Quotients for Wildlife Receptors

$$SLHQ = \frac{Dose}{TRV}$$

Parameter	Definition (Units)
SLHQ	Screening-level hazard quotient (unitless)
Dose	Estimated receptor-specific contaminant intake, from Equations 1 through 5 (mg/kg of body weight/day)
TRV	NOAEL-based TRV (mg/kg/day)

An ESL can be derived for comparison to chemical concentrations in soil, as shown in Equation 39. As discussed above, pre-calculated ESLs may be available from various sources. However, for soil calculation of dose is preferred over a generic ESL that may not reflect Utah-specific parameters. While not comprehensive, the following is a list of commons sources for ESLs and ecological toxicity data:

- LANL EcoRisk Database (LANL, 2020)
- Region 4 Ecological Risk Assessment Supplemental Guidance (US EPA, 2018)

- Region 5 Ecological Screening Levels (US EPA, 2003)
- ECOTOX (US EPA, 2023)

Aquatic community organisms are exposed to chemicals in their natural environment primarily through direct contact with water and sediment. As defined in the LANL EcoRisk documentation, the aquatic organism spends at least part of their life in close association with sediment. For comparison to surface water data, ESLs based on a generic aquatic community organism may be applied. Aquatic organisms for sediment ESLs are broadly representative of the adverse effects of COPECs on the aquatic community and apply to both aquatic plants and invertebrates. Water quality standards listed in UAC R317-2 may not be used as ESLs.

Sediment ESLs do not apply to fish or other wildlife. If fish or other organisms are identified as receptors, the approach and ESLs/TRVs should be discussed with DWMRC. A useful tool for water and sediment pathways is the US EPA EcoBox (https://www.epa.gov/ecobox/epa-ecobox-tools-exposure-pathways-water-and-sediment).

Equation 39 reflects the relationship between dose and chemical concentration in soil under Tier 1 as well as the relationship between the TRV and ESL.

Equation 39. Use of the ESLs to Determine the SLHQ

$$SLHQ = \frac{C_s}{ESL}$$

Parameter	**Definition (Units)**
SLHQ	Screening-level hazard quotient (unitless)
C_s	Chemical concentration in soil or sediment (mg COPEC / kg soil dry weight) or other medium (e.g., surface water)
ESL	Ecological Screening Level

SLHQs are calculated for each receptor and each COPEC. For each receptor, additive risk must be evaluated. For the initial screening assessment, it is assumed that all COPECs have equal potential risk to the receptor. The overall HI is then calculated for each receptor using Equation 40:

$$HI = SLHQ_x + SLHQ_Y + ... + SLHQ_z \qquad \text{Equation 40}$$

Where:

HI = Hazard Index (unitless)
$SLHQ_x$ = Hazard quotient for each COPEC (unitless)

DWMRC applies a target risk level for ecological risk assessments of 1.0. If the HI for any receptor is above this target risk level, then there is a potential for adverse effects on ecological receptors and additional evaluation following the Tier 2 SLERA process is required.

As with all risk assessments, the SLERA should include a discussion of the uncertainties. More detailed information may be found in the *Guidance for Assessing Ecological Risks Posed by Chemicals: Screening-Level Ecological Risk Assessment (NMED, 2014).*

9.3 Tier 2

The refined Tier 2 ERA will follow the same steps as taken in the Tier 1 SLERA, only with more realistic exposure assumptions likely to be encountered by each ecological receptor. Although the Tier 2 assessment is more site-specific than Tier 1, the Tier 2 assessment also employs many assumptions that would provide conservative estimates of ecological risk and is more conservative than a site-specific Tier 3 assessment.

The first step in the Tier 2 problem formulation will be to refine the list of ecological COPECs. This will be accomplished by reviewing the results of the Tier 1 assessment. COPECs which had a receptor specific SLHQ less than one will not be retained as a COPEC for that receptor for assessment in the refined analysis.

The following assumptions will apply to Tier 2 exposure doses:

- EPC – 95UCLs will be utilized as the EPC for determination of EPCs and UCLs).
- AUF – Site-specific value between 0 and 1, based on the ratio of the exposure area (size of SWMU or corrective action site) to the receptor's average home range size, as shown in Equation 41; if a receptor's home range size is less than the exposure area, a value of 1 will be assumed.

$$AUF = \frac{Exposure\ Area\ of\ Site\ (acres)}{Average\ Home\ Range\ (acres)}$$ **Equation 41**

- Bioavailability – It will be assumed that the bioavailability is 100% at each site.
- Body weight – The average reported adult body weight will be applied.
- Ingestion rate – The average reported ingestion rate will be applied.
- Dietary composition – Receptor-specific percentages of plant, animal, and soil matter will be considered. Concentrations of COPECs in dietary elements (plant and animal matter) will be predicted using bio-uptake and bioaccumulation modeling.
- Wet-weight to dry-weight conversion factor – Because body weight is reported as wet-weight (kg), and soil concentrations are reported as dry-weight (mg/kg), a wet-weight to dry-weight conversion factor will also be applied when calculating exposure doses.

The Tier 2 exposure doses for wildlife receptors will include one, two or all three of the following elements, depending on the receptor being evaluated: 1) ingestion of plant matter; 2) ingestion of animal (or invertebrate) matter; and 3) incidental ingestion of soil. Bio-uptake and bioaccumulation modeling will be utilized to predict the concentrations of COPECs in plants and animal/invertebrate matter that could be ingested by wildlife receptors.

Plant uptake factors (PUFs) will be used to predict the concentrations of COPECs in plants. PUF values and the equation that should be used to calculate PUF values for inorganic constituents are summarized in Table 16. PUF values and equations for selected organic constituents are listed in Table 17. For organic COPECs, the PUFs are based on the octanol-water partition coefficient (K_{ow}), which will be obtained from US EPA databases or primary literature.

If a PUF is not available, then a value of one (1) will be applied which assumes 100% assimilation. The equation and variables that will be used to predict COPEC concentrations in plants are shown in Equation 42.

Equation 42. Calculation of COPEC Concentrations in Plants

$$C_{plant} = C_{soil} \times PUF$$

Parameter	Definition (Units)	Value
C_{plant}	COPEC concentration in plant (mg/kg dry weight)	Calculated
C_{soil}	Concentration of COPEC in soil (EPC) (mg/kg dry weight)	Site-specific
PUF	Plant-uptake factor (unitless)	For inorganics (see Table 16) For organic constituents (Travis and Arms, 1988): PUF = $10^{(1.588 - 0.578 \log K_{ow})}$ or Table 17 K_{ow}- obtain from EPA, 2011b or most current

Table 16. Plant Uptake Factors for Inorganics

Analyte	Plant Uptake Factors (PUFs) and Equations[2]
Aluminum[1]	4.0E-03
Antimony	ln(Cp) = 0.938 * ln(Cs) - 3.233
Arsenic	Cp = 0.03752 * Cs
Barium	Cp = 0.156 * Cs
Beryllium	ln(Cp) = 0.7345 * ln(Cs) - 0.536
Boron	4.0E+00[1]
Cadmium	ln(Cp) = 0.546 * ln(Cs) - 0.475
Calcium	3.5E+00[1]
Chromium	Cp = 0.041 * Cs
Cobalt	Cp = 0.0075 * Cs
Copper	ln(Cp) = 0.394 * ln(Cs) + 0.668
Iron	4.0E-03[1]
Lead	ln(Cp) = 0.561 * ln(Cs) - 1.328
Magnesium	1.0E+00[1]
Manganese	Cp = 0.079 * Cs

Analyte	Plant Uptake Factors (PUFs) and Equations[2]
Mercury	9.0E-01[1]
Molybdenum	2.5E-01[1]
Nickel	$\ln(Cp) = 0.748 * \ln(Cs) - 2.223$
Potassium	1.0E+00[1]
Selenium	$\ln(Cp) = 1.104 * \ln(Cs) - 0.677$
Silver	$Cp = 0.014 * Cs$
Sodium	7.5E-02[1]
Thallium	4.0E-03[1]
Tin	3.0E-02[1]
Vanadium	$Cp = 0.00485 * Cs$
Zinc	$\ln(Cp) = 0.554 * \ln(Cs) + 1.575$

[1] From Baes, *et.al*, 1994

[2] US EPA, 2007

Cp – concentration in plant

Cs -concentration in soil

Table 17. Plant Uptake Equations for Select Organics

Analyte	Plant Uptake Factor (PUF) Equation[1]
Dieldrin	$Cp = 0.41 * Cs$
TNT	$Cp = 4.23 * Cs$
RDX	$Cp = 0.43 * Cs$
Acenaphtene	$\ln(Cp)= -0.8556 * \ln(Cs) - 5.562$
Acenaphthylene	$\ln(Cp) = 0.791 * \ln(Cs) -1.144$
Anthracene	$\ln(Cp)= 0.7784 * \ln(Cs) - 0.9887$
Fluoranthene	$Cp = 0.50 * Cs$
Fluorene	$\ln(Cp)= -0.8556 * \ln(Cs) - 5.562$
Naphthalene	$Cp = 12.2 * Cs$
Phenanthrene	$\ln(Cp)= 0.6203 * \ln(Cs) - 0.1665$
Benzo(a)anthracene	$\ln(Cp)= 0.5944 * \ln(Cs) - 2.7078$
Benzo(a)pyrene	$\ln(Cp)= 0.9750 * \ln(Cs) - 2.0615$
Benzo(b)fluoranthene	$Cp = 0.310 * Cs$
Benzo(k)fluoranthene	$\ln(Cp)= 0.8595 * \ln(Cs) - 2.1579$
Chrysene	$\ln(Cp)= 0.5944 * \ln(Cs) - 2.7078$
Dibenz(a,h)anthracene	$Cp = 0.13 * Cs$
Indeno(1,2,3-cd)pyrene	$Cp = 0.11 * Cs$
Pyrene	$Cp = 0.72 * Cs$
Pentachlorophenol	$Cp = 5.93 * Cs$

Analyte	Plant Uptake Factor (PUF) Equation[1]
[1] US EPA, 2007 Cp – concentration in plant Cs -concentration in soil	

Concentrations of COPECs in animal matter (invertebrates and prey species) will be predicted by applying bioaccumulation or biomagnification factors (BAFs). The BAFs will be selected from primary literature sources. If BAF data are not available, a default value of 1.0 will be used, which will conservatively assume 100% assimilation. Methodology for determining BAFs for soil to plants, soil to earthworms, and soil to small mammals may be found in US EPA (2003(g), 2005, and 2007). The equation and variables for predicting concentrations in animal matter are shown in Equation 43.

Equation 43. Calculation of COPEC Concentrations in Prey/Invertebrate

$$C_{prey/invert} = C_{soil} \times BAF$$

Parameter	Definition (Units)	Value
$C_{prey/invert}$	COPEC concentration in prey/invertebrate (mg/kg dry weight)	Calculated
C_{soil}	Concentration of COPEC in soil (EPC) (mg/kg dry weight)	Site-specific
BAF	Bioaccumulation/Biomagnification factor	Chemical-specific (see US EPA 2003(g), 2005, and 2007)

The equation and exposure assumptions that will be used to calculate the Tier 2 exposure doses for the deer mouse are shown in Equation 44.

The equation and exposure assumptions that will be used to calculate the Tier 2 exposure doses for the horned lark are shown in Equation 45.

Equation 44. Calculation of Tier 2 Exposure Dose for COPECs in Soil; Deer Mouse

$$Exposure\ Dose = \frac{\left[\left(C_{plant} \times \left(IR_{plant} \times ww{:}dw\right)\right) + \left(C_{invert} \times \left(IR_{invert} \times ww{:}dw\right)\right) + \left(C_{soil} \times IR_{soil} \times ST\right) \times AUF\right]}{BW}$$

Parameter	Definition (Units)	Value	Reference
Exposure dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	Calculated	--
C_{plant}	COPEC concentration in plants (mg final COPEC/kg plant dry weight)	Calculated	See Equation 42
IR_{total}	Receptor-specific average ingestion rate based on total dietary intake (kg wet weight/day)	0.004	US EPA 1993g
IR_{plant}	Receptor-specific plant-matter ingestion rate (kg food wet weight/day)	0.003	Based on an average ingestion rate of 0.004 kg/day (US EPA, 1993g) and a diet of 74% plant matter (OEHHA, 1999)
ww:dw	Wet weight to dry weight conversion factor for ingested matter	0.22	78-percent moisture
C_{invert}	Invertebrate EPC (mg final COPEC/kg invertebrate dry weight)	Calculated	See Equation 43
IR_{invert}	Receptor-specific animal matter ingestion rate (kg food wet weight/day)	0.001	Based on an average ingestion rate of 0.004 kg/day (US EPA, 1993g) and a diet of 26% invertebrate matter (OEHHA, 1999)
C_{soil}	Surface-soil EPC (mg final COPEC/kg soil dry weight)	Site-specific	95% UCL if available, or maximum (0-10 ft bgs)
IR_{soil}	Receptor-specific incidental soil ingestion rate (kg soil dry weight/day)	0.000018	Based on < 2% (Beyer et. al, 1994); Average ingestion rate of (0.004 kg/day wet weight * 0.22 ww:dw) * 2%.
ST	Bioavailability factor for constituents ingested in soil (assumed to be 1.0 for all constituents)	1.0	Conservative default (assume 100% bioavailability)
AUF	area use factor (maximum value = 1); ratio of area of site to average receptor foraging range (0.3 acres for deer mouse)	Site-specific	US EPA, 1993g
BW	average adult body weight (kg)	0.02	CDW, 2011

Equation 45. Calculation of Tier 2 Exposure Dose for COPECs in Soil; Horned Lark

$$Exposure\ Dose = \frac{\left[\left(C_{plant} \times \left(IR_{plant} \times ww{:}dw\right)\right) + \left(C_{invert} \times \left(IR_{invert} \times ww{:}dw\right)\right) + \left(C_{soil} \times IR_{soil} \times ST\right) \times AUF\right]}{BW}$$

Parameter	Definition (Units)	Value	Reference
Exposure dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	Calculated	--
C_{plant}	COPEC concentration in plants (mg final COPEC/kg plant dry weight)	Calculated	See Equation 42
IR_{plant}	Receptor-specific plant-matter ingestion rate (kg food wet weight/day)	0.026	Based on average ingestion rate of 0.035 kg/day (US EPA 1993b) and a diet of 75% plant matter (Doctor, *et* al, 2000) and US EPA, 1993g
ww:dw	Wet weight to dry weight conversion factor for ingested matter	0.22	78-percent moisture
C_{invert}	Invertebrate EPC (mg final COPEC / kg invertebrate dry weight)	Site-specific	See Equation 43
IR_{invert}	Receptor-specific animal matter ingestion rate (kg food wet weight/day)	0.009	Based on average ingestion rate of 0.035 kg/day (US EPA 1993b) and a diet of 25% invertebrates (Doctor, et al, 2000) and US EPA, 1993g
C_{soil}	Surface-soil EPC (mg final COPEC / kg soil dw)	Site-specific	95% UCL if available, or maximum (0-1 ft bgs)
IR_{soil}	Receptor-specific incidental soil ingestion rate (kg/day dry weight)	0.00077	Based on 10% (Baer, *et al*, 1994). Average ingestion rate of (0.035 kg/day (wet weight) * 0.22 ww:dw) * 10%).
ST	Bioavailability factor for constituents ingested in soil (assumed to be 1 for all constituents)	1	Conservative default (assume 100% bioavailability)
AUF	Area use factor (maximum value = 1); ratio of area of site to average receptor foraging range (4 acres for horned lark)	Area of site (acres) / 4 acres	Beason, 1995
BW	Average adult body weight (kg)	0.033	Trost, 1972

The equation and exposure assumptions that will be used to calculate the Tier 2 exposure doses for the kit fox are shown in Equation 46.

Equation 46. Calculation of Tier 2 Exposure Dose for COPECs in Soil; Kit Fox

$$Exposure\ Dose = \frac{\left[\left(C_{prey} \times \left(IR_{prey} \times ww\colon dw\right)\right) + \left(C_{soil} \times IR_{soil} \times ST\right) \times AUF\right]}{BW}$$

Parameter	Definition (Units)	Value	Reference
Exposure dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	Calculated	--
C_{prey}	Prey EPC (mg final COPEC / kg prey dry weight)	Calculated	See Equation 43
IR_{prey}	Receptor-specific animal matter ingestion rate (kg food wet weight/day)	0.13	Based on an average ingestion rate of 0.13 kg/day (OEHHA, 2003) and a diet of 100% animal matter
ww:dw	Wet weight to dry weight conversion factor for ingested matter	0.22	78-percent moisture
C_{soil}	Surface and subsurface-soil (0-10 ft bgs) EPC (mg final COPEC / kg soil dw)	Site-specific	95% UCL if available, or maximum (0-10 ft bgs)
IR_{soil}	Receptor-specific incidental soil ingestion rate (kg soil dry weight/day)	0.0008	Based on 2.8% (Beyer et.al., 1994). Average ingestion rate of (0.13 kg/day (wet weight) *0.22 ww:dw) * 2.8%).
ST	Bioavailability factor for constituents ingested in soil (assumed to be 1for all constituents)	1	Conservative default (assume 100% bioavailability)
AUF	Area use factor (maximum value = 1); ratio of area of site to average receptor foraging range (1713 acres for kit fox)	Site-specific	--
BW	Average adult body weight (kg)	2.0	OEHHA, 2003

The equation and exposure assumptions that will be used to calculate the Tier 2 exposure doses for the red-tailed hawk are shown in Equation 47.

Equation 47. Calculation of Tier 2 Exposure Dose for COPECs in Soil; Red-Tailed Hawk

$$Exposure\ Dose = \frac{\left[\left(C_{prey} \times \left(IR_{prey} \times ww{:}dw\right)\right) + \left(C_{soil} \times IR_{soil} \times ST\right) \times AUF\right]}{BW}$$

Parameter	Definition (Units)	Value	Reference
Exposure dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	Calculated	--
C_{prey}	Prey EPC (mg final COPEC / kg prey dry weight)	Calculated	See Equation 43
IR_{prey}	receptor-specific animal matter ingestion rate (kg food wet weight/day)	0.1	Based on an average ingestion rate of 0.1 kg/day (US EPA 1993g) and a diet of 100% animal matter
ww:dw	Wet weight to dry weight conversion factor for ingested matter	0.22	78-percent moisture
C_{soil}	surface-soil EPC (mg final COPEC / kg soil dw)	Site-specific	95% UCL if available, or maximum (0-1 ft bgs)
IR_{soil}	receptor-specific incidental soil ingestion rate (kg soil dry weight/day)	0.0004	Based on < 2% (Beyer et. al., 1994). Average ingestion rate of (0.12 kg/day (wet weight) *0.22 kg/kg) * 2%).
ST	bioavailability factor for constituents ingested in soil (assumed to be 1 for all constituents)	1	Conservative default (assume 100% bioavailability)
AUF	area use factor (maximum value = 1); ratio of area of site to average receptor foraging range (1770 acres for red-tailed hawk)	Site-specific	--
BW	average adult body weight (kg)	1.1	US EPA, 1993g

The equation and exposure assumptions that will be used to calculate the Tier 2 exposure doses for the pronghorn are shown in Equation 48.

Equation 48. Calculation of Tier 2 Exposure Dose for COPECs in Soil; Pronghorn

$$Exposure\ Dose = \frac{\left[\left(C_{plant} \times \left(IR_{plant} \times ww{:}dw\right)\right) + \left(C_{soil} \times IR_{soil} \times ST\right) \times AUF\right]}{BW}$$

Parameter	Definition (Units)	Value	Reference
Exposure dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)	Calculated	--
C_{plant}	COPEC concentration in plants (mg final COPEC/kg plant dry weight)	Calculated	See Equation 42
IR_{plant}	receptor-specific plant-matter ingestion rate (kg food wet weight/day)	1.4	Based on an average ingestion rate of 1.4 kg/day (US FWS, 2005) and a diet of 100% plant matter
ww:dw	Wet weight to dry weight conversion factor for ingested matter	0.22	78-percent moisture
C_{soil}	surface-soil EPC (mg final COPEC / kg soil dw)		95% UCL if available, or maximum (0-1 ft bgs)
IR_{soil}	receptor-specific incidental soil ingestion rate (kg soil dry weight/day)	0.006	Based on < 2% (Beyer et. al., 1994). Average ingestion rate of (1.4 kg/day (wet weight) * 0.22 ww:dw) * 2%).
ST	bioavailability factor for constituents ingested in soil (assumed to be 1.0 for all constituents)	1	Conservative default (assume 100% bioavailability)
AUF	area use factor (maximum value = 1); ratio of area of site to average receptor foraging range (3422 acres for pronghorn)	Site-specific	Zoellick & Smith, 1992
BW	Average adult body weight (kg)	50	O’Gara, 1978

9.3.1 Toxicity

The Tier 2 TRVs will be based on LOAELs. The LOAEL will be used as it is more representative of population risks.

9.3.2 Risk Characterization

Risk characterization for Tier 2 will be conducted by calculating HQs for plant and wildlife receptors using a similar method as in the Tier 1 SLERA. The equation and assumptions for calculating the Tier 2 HQs for wildlife receptors are shown in Equation 49.

Equation 49. Calculation of Tier 2 Hazard Quotients for Wildlife Receptors

$$HQ = \frac{Dose}{TRV}$$

Parameter	Definition (Units)
HQ	Hazard quotient (unitless)
Dose	Estimated receptor-specific contaminant intake (mg/kg of body weight/day)
TRV	Toxicity reference value (mg/kg/day) based on lowest observed adverse effects level (LOAEL)

For plants, a qualitative discussion of the potential for adverse risk will be provided in the assessment. Comparison of TRVs to soil concentrations based on the 95% UCL may be provided.

Summation of HQs will be added for COPECs that have a similar receptor-specific mode of toxicity. If the Tier 2 HI is less than one, adverse ecological effects are not expected, and no further action will be taken.

If a HQ exceeds 1.0, this is not necessarily indicative that an adverse risk will occur (Menzie, *et.al*. 1993, Tannenbaum, *et.al* 2003, and Tannenbaum, 2004). As reproductive impacts are most critical to assessing risk to populations, the HQs should be carefully evaluated. If the HQ is equal to or greater than 1.0, but less than 10, there may be a low potential for adverse effects. If the HQ is greater than 10, there may be a higher potential for adverse effects to occur based on experimental evidence.

For sites that have an HI equal to or greater than one, the site may require: 1) additional evaluation under a weight-of-evidence analysis; 2) a Tier 3 risk assessment; or 3) a corrective measures study or other remedial action.

Per US EPA (1997c), Tier 2 ecological risk characterization should include a discussion of the uncertainties since many assumptions may or may not accurately reflect site conditions. Therefore, a discussion of the uncertainties associated with Tier 2 SLERA will be included in the report.

9.4 Tier 3

If the Tier 2 ERA does not show that levels of contamination in the impacted media are below the target level of 1.0, additional quantitative analyses (e.g., biota studies to evaluate impacts at the site) or even corrective actions (e.g., removals) may be warranted. DWMRC should be consulted before proceeding with additional analyses and/or corrective actions and a cost-benefit analysis that weighs corrective actions (removals) versus additional investigations should be performed. If the SLERA, consultation with DWMRC, and the cost-benefit analysis support further evaluation of the contaminated site, site-specific data that supports formulation of a problem statement for a Tier 3 site-specific ecological risk assessment should be conducted.

9.4.1 Performing a Tier 3 Site Specific Ecological Risk Assessment

After problem formulation is completed and an integrated conceptual exposure model is developed and discussed with DWMRC, a Work Plan should be developed and submitted to DWMRC for approval. Site specific data should be collected and used, wherever practicable, to determine whether site releases present unacceptable risks and to develop quantitative cleanup levels that are protective. As in all risk assessments, the scope of the Tier 3 site-specific risk assessment should be tailored to the nature and complexity of the issues present at the site and all response alternatives being considered, including their costs and implementability.

9.4.2 Problem Formulation for Tier 3

Like a Tier 1 or Tier 2 SLERA, a Tier 3 assessment begins with a problem formulation step. By combining information on: (1) the site COPECs; (2) the ecotoxicity of the COPECs; (3) the ecological setting; (4) environmental fate and transport; and (5) complete exposure pathways, those aspects of the site ecosystem potentially at risk as well as the responses to that risk are identified. Based on that information, the risk assessment team and DWMRC agree on assessment endpoints and specific risk questions or testable hypotheses that, together with an integrated CSM, form the basis for the site investigation.

Problem Formulation for a Tier 3 assessment includes the following elements:

- Refinement of the COPECs by examining the assumptions used in the SLERA.
- Further characterization of the ecological effects associated with the contaminants.
- Reviewing and refining information on contaminant fate and transport, complete exposure pathways, and ecosystems potentially at risk.
- Selection of site-specific assessment endpoints.
- Development of an integrated CSM and associated risk questions.

If the problem formulation step indicates additional sampling is required for the Tier 3 assessment, a separate sampling and analysis plan (SAP) may also be required. In addition to documenting the approaches, procedures, and expectations for the Tier 3 site-specific ecological risk assessment, the Work Plan should also summarize all agreements between the facility and DWMRC regarding the contaminants of concern, assessment endpoints, exposure pathways, and risk questions.

9.4.3 Refining Contaminants of Concern

Because of the conservative assumptions used during the SLERA, some of the COPECs retained for the Tier 3 assessment might pose negligible risk. At this stage of the ecological risk assessment process, the risk assessment team should review the assumptions used in the SLERA (e.g., bioavailability assumed to be 100 percent) against COPEC-specific values reported in the literature and consider how the hazard quotients or indices would change if more realistic, yet conservative, assumptions were applied.

New information may become available that indicates the initial assumptions that screened some contaminants out of the SLERA are no longer valid (e.g., site contaminant levels are higher than originally reported). In this case, contaminants can be placed back on the list of COPECs to be investigated.

After consultation with DWMRC, one or more of the following supplemental components (background concentrations, frequency and magnitude of detection, dietary considerations) may be included in the Problem Formulation step for the Tier 3 assessment. These components need not be implemented in the order presented herein, nor do all the components need to be implemented. However, any COPEC identified for potential exclusion from the Tier 3 assessment through application of any supplemental component must also be evaluated for its potential to bioaccumulate, biomagnify, and bioconcentrate.

Those components included in the assessment should be identified and discussed in the Work Plan. In addition, the Tier 3 ecological risk assessment report should fully address the issues associated with each supplemental component included in the Tier 3 assessment and describe the rationale underlying its selection for inclusion in the assessment.

9.4.4 Frequency and Magnitude of Detection

The SAP needs to provide for characterization of the full range of variability and distribution in the data while meeting the project criteria for completeness, comparability, representativeness, precision, and accuracy. Given data of adequate quality, reduction of COPECs through application of this component may be determined acceptable following consultation with DWMRC. A frequency of detection (FOD) evaluation should re-examine the original results considering:

- The information and data considered in the evaluation performed for the SLERA;
- The results of the SLERA; and
- The information and data gathered in performing the problem formulation activities associated with the Tier 3 site-specific ecological risk assessment.

The rationale, criteria, and methodology to be employed should be discussed with DWMRC. For a Tier 3 assessment, these discussions should be expanded to address additional issues including: 1) the influence of random and/or biased sampling on the frequency and magnitude of detected values within the distribution of data; 2) the spatial and temporal pattern of contaminants identified as low frequency and/or low magnitude; 3) comparison of risk-based detection limits with toxicity benchmarks; and 4) the relationship of detected values to toxicity benchmarks. The agreed upon approach should be documented in the Work Plan.

9.4.5 Dietary Considerations

Some site-related chemicals such as calcium, iron, magnesium, sodium, and potassium can function as nutrients in organisms serving as physiological electrolytes. When present at concentrations that allow them to function in this manner, they typically pose little ecological risk. However, some nutrients (e.g., selenium, copper, molybdenum, and boron) can transition from essential to toxic at slightly higher concentrations. As part of the Tier 3 assessment, the suite of nutrients relevant to the range of ecological receptors (wildlife versus plants) at the site should be identified. The potential for toxic effects resulting from site concentrations relative to the toxicological benchmarks for nutrients should be evaluated. In addition, the assessment should determine whether exposure to site contamination could result in a nutrient deficiency for organisms of concern. As part of the analysis, the nutrient deficiency level and the toxicity benchmark should be compared to determine if they are similar in magnitude.

9.4.6 Bioaccumulation, Bioconcentration and Biomagnification

For those COPECs identified by applying any of the supplemental components discussed above, it is essential to evaluate their potential to bioaccumulate, bioconcentrate, and/or biomagnify prior to eliminating them from further consideration in the Tier 3 assessment. Compounds with a high potential to accumulate and persist in the food chain should be carried out through the risk assessment process.

Additionally, the Tier 3 assessment should address the likelihood that contaminants identified for removal from the list of COPECs could exert adverse effects on higher trophic level organisms. A determination that bioaccumulation and biomagnification have been satisfactorily addressed through methods developed in consultation with the DWMRC and documented in the Tier 3 assessment Work Plan (e.g., modeling, site-related tissue measurements) should be included in the site-specific risk assessment report.

9.4.7 Further Characterization of Ecological Effects

The literature searches conducted as part of the SLERA should be expanded to obtain the information needed for the more detailed problem formulation phase of the Tier 3 site-specific ecological risk assessment. The literature search should identify NOAELs, LOAELs, exposure-response functions, and the mechanisms of toxic responses for those contaminants that were not addressed in the SLERA. Appendix C of US EPA's 1997 *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (US EPA 1997a) presents additional details on the factors that are important in conducting a literature search. For all chemicals on the refined list of COPECs, it is important to obtain and review the primary literature to ensure potential data gaps are addressed and that the most recently available information is used is Tier 3 risk assessment.

9.4.8 Reviewing and Refining Information on Contaminant Fate and Transport, Complete Exposure Pathways, and Ecosystems Potentially at Risk

The exposure pathways and the ecosystems associated with the assessment endpoints that were retained in the SLERA are evaluated in more detail. Additional information should be compiled on:

- The environmental fate and transport of the COPECs;
- The ecological setting and general flora and fauna of the site (including habitat, potential receptors, etc.); and

- The magnitude and extent of contamination, including its spatial and temporal variability relative to the assessment endpoints.

It is frequently possible to reduce the number of exposure pathways that require evaluation to one or a few "critical exposure pathways" which (1) reflect maximum exposures of receptors within the ecosystem, or (2) constitute exposure pathways to ecological receptors sensitive to specific COPECs. If multiple critical exposure pathways exist at a site, each should be evaluated as part of the Tier 3 assessment.

9.4.9 Contaminant Fate and Transport

Information on how the COPECs will or could be transported or transformed in the environment by physical, chemical, and biological processes should be used to identify the exposure pathways that could produce significant ecological impacts. Physically, COPECs move through the environment by volatilization, erosion, deposition (contaminant sinks), weathering of parent material with subsequent transport, and/or water transport. Chemically, COPECs can undergo several processes in the environment such as degradation, complexation, ionization, precipitation, and/or adsorption. Several biological processes also affect COPEC fate and transport in the environment including bioaccumulation, biodegradation, biological transformation, food chain transfers, and/or excretion. Degradation product(s) and biological transformation products may be more or less toxic than the parent compound.

The above information is used to evaluate how COPECs will partition in the environment and determine the bioavailability of site contaminants. Note that at this point in the process, it may be possible for the risk assessment team and DWMRC to use this information to replace some of the conservative assumptions employed in the SLERA and eliminate some COPECs from further evaluation. Such negotiations should be summarized in the Work Plan and must be documented in the Tier 3 site-specific ecological risk assessment report.

9.4.10 Complete Exposure Pathways

The potentially complete exposure pathways identified in the SLERA must be evaluated in more detail in the Tier 3 assessment based on the refined contaminant fate and transport evaluation and the refined evaluation of potential ecological receptors.

Some of the potentially complete exposure pathways identified in the SLERA may be ruled out from further consideration at this time. Conversely, additional exposure pathways might be identified particularly those originating from secondary sources of contamination. Any data gaps that result in questions about whether an exposure pathway is complete should be identified, and the type of data needed to answer those questions should be described to assist in developing the Work Plan and SAP. During the re-examination of the exposure pathways, the potential for food-chain exposures deserves particular attention as some COPECs are effectively transferred through food chains while others are not.

9.4.11 Ecosystems Potentially at Risk

The ecological setting information collected during the SLERA should provide answers to several questions including:

- What habitats are present?
- What types of water bodies are present, if any?
- Do any other habitats exist on or adjacent to the site?

If the questions above cannot be effectively answered using the information from the SLERA, an additional site visit should be considered to supplement the one conducted during the Scoping Assessment.

Available information on the ecological effects of contaminants as well as observations made during the initial and subsequent site visits can help focus the Tier 3 assessment on specific ecological resources that should be evaluated more thoroughly. For example, some groups of organisms can be more sensitive than others to a particular COPEC; alternatively, an already-stressed population (e.g., due to habitat degradation) could be particularly sensitive to any added stressor.

9.4.12 Selection of Site-Specific Assessment Endpoints

The selection of assessment endpoints includes discussion between the risk assessment team and DWMRC concerning management policy goals and ecological values. Input should be sought from all stakeholders associated with a site when identifying assessment endpoints. Stakeholder input at this stage helps ensure that DWMRC can readily defend the assessment endpoints when making decisions for the site.

If a Tier 2 screening assessment has been performed for the site, the selection of assessment endpoints should be re-examined. The endpoints selected for the Tier 3 assessment should reflect:

- Contaminants and concentrations at the site;
- Mechanisms of toxicity of the contaminants to different groups of organisms;
- Ecologically relevant receptor groups potentially sensitive or highly exposed to site contaminants and attributes of their natural history; and
- Potentially complete exposure pathways.

In addition, the risk assessment team should determine if any of the COPECs can adversely affect organisms in direct contact with contaminated media (e.g., direct exposure to water, sediment, soil) or if the contaminants accumulate in food chains, resulting in adverse effects in organisms that are not directly exposed or are minimally exposed to the original contaminated media (i.e., indirect exposure). Also, the risk assessment team must decide if the Tier 3 assessment should focus on toxicity resulting from direct or indirect exposures, or if both should be evaluated.

In specifying assessment endpoints, a broad specification (e.g., protecting aquatic communities) is generally of less value in problem formulation than a focused specification (e.g., maintaining aquatic community composition and structure downstream of a site similar to that upstream of the site). Focused assessment endpoints define the ecological value in sufficient detail to identify the measures needed to answer specific questions about the site or to test specific hypotheses.

Once assessment endpoints have been selected, testable hypotheses should be developed to determine whether or not a potential threat to the assessment endpoints exists. Measurement endpoints can also be developed or if developed as part of a Tier 2 screening assessment, refined based on the activities associated with the problem formulation step of the Tier 3 assessment. Note that testable hypotheses and measurement endpoints cannot be finalized without agreement on the assessment endpoints among DWMRC, the risk assessment team, and other stakeholders.

9.4.13 Development of a Conceptual Site Model and Associated Risk Questions

Conceptual Site Model

Based on the information obtained from the SLERA, knowledge of the contaminants present, the screening CSM, including the exposure pathway model, and the assessment endpoints, an integrated CSM should be developed. The integrated CSM should include a contaminant fate-and-transport diagram that traces the movement of COPECs from sources through the ecosystem to receptors associated with the assessment endpoints.

Exposure pathways that do not lead to a species or group of species associated with the proposed assessment endpoint indicate that: (1) there is an incomplete exposure pathway to the receptor(s) associated with the proposed assessment endpoint; or (2) there are missing components or data necessary to demonstrate a complete exposure pathway. If case (1) is true, the proposed assessment endpoint should be reevaluated to determine if it is an appropriate endpoint for the site. If case (2) is true, then additional field data may be needed to reevaluate contaminant fate and transport at the site.

Assessment endpoints differ from site to site and can represent one or more levels of biological organization. At any particular site, the appropriate assessment endpoints might involve local populations of a particular species, community-level integrity, and/or habitat preservation. The integrated CSM must encompass the level of biological organization appropriate for the assessment endpoints for the site.

Risk Questions

Ecological risk questions are inquiries into the relationship between an assessment endpoint and its expected response when exposed to site contamination. Risk questions should be based on the assessment endpoints selected for the site and lead to answers that establish a foundation for the study design and evaluation of the results of the site investigation in the analysis and risk characterization phases of the risk assessment process. The most basic question applicable to virtually every site asks whether site-related contaminants are causing or have the potential to cause adverse effects on the assessment endpoint(s). To ensure the Tier 3 assessment is useful in a feasibility study, it is helpful if the specific contaminant(s) posing the most significant threat(s) can be identified. Thus, the question is refined to ask "does (or could) chemical X cause adverse effects on the assessment endpoint?" In general, four lines of evidence are used to answer this question:

- Comparison of estimated or measured exposure levels for a given chemical with levels that are known from the literature to be toxic to receptors associated with the assessment endpoints;
- Comparison of laboratory bioassays of media from the site and bioassays of media from a reference site;
- Comparison of *in situ* toxicity tests at the site with *in situ* toxicity tests in a reference body of water; and
- Comparison of observed effects in the receptors associated with the site with similar receptors at a reference site.

9.4.14 Finalization of the CSM

The problem formulation step for the Tier 3 assessment is considered complete once the risk assessment team and DWMRC reach agreement on four items: the ecological contaminants of concern, the assessment endpoints, the exposure pathways, and the risk questions. These items should be presented and summarized in the integrated CSM for the site and the CSM should be presented and discussed in the Work Plan and SAP (if a separate SAP is developed) for the Tier 3 site-specific assessment.

9.4.15 Develop a Work Plan and SAP for Tier 3

Based on the information assembled during problem formulation, the risk assessment team and DWMRC agree on assessment endpoints, risk questions and/or testable hypotheses that, together with the rest of the integrated CSM, form the basis for the site investigation. At this stage, site-specific information on exposure pathways and/or the presence of specific species is likely to be incomplete. By using the integrated CSM, measurement endpoints can be selected/verified and a plan for filling information gaps can be developed and written into the Work Plan and SAP.

Field verification of the SAP is important to ensure that the data quality objectives (DQOs) for the site investigation will be met. This step verifies that the selected assessment endpoints, testable hypotheses, exposure pathway model, measurement endpoints, and study design are appropriate and implementable at the site. By verifying the field sampling plan prior to conducting the full site investigation, well-considered alterations can be made to the study design and/or its implementation if necessary. If changing conditions identified during field verification force changes to the Work Plan and/or SAP (e.g., selection of a different reference site), the changes should be agreed to and documented by the risk assessment team in consultation with DWMRC.

Site investigation activities and sampling and analysis procedures should be clearly documented in the Work Plan and/or SAP. However, the Work Plan and SAP should allow for instances where unexpected conditions arise in the field that indicate a need to change the study design. The Work Plan and SAP should indicate that should the need arise, the ecological risk assessment team will reevaluate the feasibility or adequacy of the sampling design and any resulting changes to the Work Plan or SAP will be agreed upon by both the risk assessment team and DWMRC and will be documented in the Tier 3 site-specific ecological risk assessment report.

When possible, any field sampling efforts for the ecological risk assessment should overlap with other site data collection efforts to reduce sampling costs and to prevent redundant sampling. The Work Plan and/or the SAP should specify the methods by which the collected data will be analyzed. Both plans should address all food chain exposure model parameters, data reduction techniques, data interpretation methods, and statistical analyses that will be used. Once completed, the documents should be submitted to DWMRC. At the successful conclusion of the review process, DWMRC will issue approvals or approvals with modifications for the Work Plan and SAP and the site investigation, data evaluation, and risk characterization can proceed.

<u>Recommended Information for Tier 3 site-specific Ecological Risk Assessment Work Plan and/or Sampling and Analysis Plan</u>

At a minimum, the Tier 3 site-specific ecological Work Plan and accompanying SAP (if needed) should include:

- A brief and concise summary of the information contained in the SLERA Report.
- The results of the problem formulation step for the Tier 3 site-specific ecological risk assessment including:
- Summary of discussion and agreements with DWMRC regarding the use of FOD in the assessment.
- Refined list of COPECs.
- Further characterization of the ecological effects associated with site contaminants.
- Review and refinement of information on contaminant fate and transport, complete exposure pathways, and ecosystems potentially at risk at the site.
- Review and refinement of the selection of site-specific assessment endpoints.
- Development of the integrated CSM and associated risk questions.
- Identification and discussion of the Supplemental Components i.e., background concentrations, frequency and magnitude of detection, dietary considerations, and any additional considerations used in refining the list of COPECs.
- Presentation and discussion of the integrated CSM.
- Detailed presentation of all site investigation activities and sampling and analysis procedures including quality assurance/quality control requirements.
- Presentation and discussion of all assessment endpoints, risk questions, and testable hypotheses.
- The SAP should specify the relationship between measurement and assessment endpoints, the necessary number, volume, and types of samples to be collected, and the sampling techniques to be used.
- The SAP should specify the data reduction and interpretation techniques and the DQOs for the site investigation.

- Contingency plan(s) that anticipate situations that may arise during the site investigation that require modification of the approaches documented in the Work Plan and/or SAP.
- Detailed presentation of procedures for analyzing site-specific data collected during the site investigation.
- Identification and discussion of the methodology to be employed in the analysis of exposure response.
- Identification and discussion of statistical techniques to be used in the Tier 3 assessment.
- Quantified exposure for each measurement receptor for each pathway.
- Technical Decision Point summarizing agreement between the risk assessment team and DWMRC on the list of COPECs, assessment endpoints, exposure pathways, and risk questions.

9.4.16 Analysis of Ecological Exposures and Effects

Analysis of exposure and effects is performed interactively, with one analysis informing the other. These analyses are based on the information collected during the SLERA, problem formulation activities conducted in preparation for the Tier 3 assessment, and additional information collected in developing the Work Plan and SAP. Both analyses are performed in accordance with the data interpretation and analysis methods outlined in the Work Plan and SAP.

In the analysis phase, the site-specific data obtained during the site investigation replace many of the assumptions made for the SLERA. For the exposure and ecological effects characterizations, the uncertainties associated with the field measurements and with the assumptions made where site-specific data are not available must be documented in the Tier 3 site-specific ecological risk assessment report.

9.4.17 Characterizing Exposures

In the exposure analysis, both the ecological stressor and the ecosystem must be characterized on similar temporal and spatial scales. The result of the analysis is an exposure profile that quantifies the magnitude and spatial and temporal patterns of exposure as they relate to the assessment endpoints and risk questions developed during problem formulation. This exposure profile along with a description of the associated uncertainties and assumptions serves as input to the risk characterization.

Stressor characterization involves determining the stressor's distribution and pattern of change. The analytic approach for characterizing ecological exposures should follow the methodology specified in the Work Plan and SAP. For chemical stressors, a combination of fate-and-transport modeling and sampling data from the site are typically used to predict the current and likely future nature and extent of contamination at a site. Any site-specific information that can be used to replace previous assumptions based on literature searches or information from other sites should be incorporated into the description of ecological conditions at the site. This information and all remaining assumptions and uncertainties associated with the characterization of exposures at the site should be documented in the Tier 3 site-specific ecological risk assessment report.

Specifically, exposure to COPECs released from facility contaminant sources is evaluated through consideration of the exposure pathways included in the integrated CSM. All exposure pathways identified as potentially complete should be evaluated in the exposure assessment. The summation of this potential exposure across all pathways for a measurement receptor defines the exposure of that measurement receptor to a COPEC. Exposure assessments are conducted separately for each community and each measurement receptor.

9.4.18 Characterizing Ecological Effects

Following the methods for analyzing site-specific data specified in the Work Plan and SAP, the assembled information on ecological effects is integrated with any evidence of existing impacts gathered during the site investigation (e.g., toxicity testing).

Exposure-response Analysis

In this phase of the analysis, measurement endpoints are related to the assessment endpoints using the logical structure provided by the integrated CSM. Any extrapolations required to relate measurement to assessment endpoints (e.g., between species, between response levels, from laboratory to field) should be explained. Finally, an exposure-response relationship is described to the extent possible (e.g., by a regression equation), including the confidence limits (quantitative or qualitative) associated with the relationship. Statistical techniques such as those available in US EPA's ProUCL software (US EPA, 2022) and other methods used to identify and/or describe the relationship between exposure and response from the field data should follow the analysis procedure specified in the Work Plan and SAP.

When exposure-response data are not available or cannot be developed, a threshold for adverse effects can be developed instead, as in the SLERA. For the Tier 3 assessment, however, site-specific information should be used instead of conservative assumptions used in the SLERA. If a site is analyzed using this approach, the methodology should be described in the Work Plan and, as necessary, the SAP.

Evidence of Causality

Demonstrating a correlation between the contaminant gradient at the site and ecological impacts is an important component of establishing causality. Thus, it is important to evaluate the strength of the causal association between the site contaminants and their impact on the measurement and assessment endpoints. However, other lines of evidence should be presented in support or in the absence of such a demonstration. Note that by itself, an exposure-response correlation at a site is not sufficient to demonstrate causality. The correlation must be supported by one or more lines of evidence as well as an analysis of potential confounding factors at the site. Criteria for evaluating causal associations are outlined in the US EPA's *Framework for Ecological Risk Assessment* (US EPA, 1992d).

9.4.19 Risk Characterization

The risk characterization section of the Tier 3 site-specific ecological risk assessment report should include a qualitative and quantitative presentation of the risk results and associated uncertainties.

9.4.20 Risk Estimation

For population measurement receptors, HQs and HIs should reflect the actual diet of the receptor; the exposure and risk to multiple contaminants are additive (i.e., two or more contaminants may affect the same target organs or organ systems and/or act by similar mechanisms). Therefore, HQs and HIs calculated using TRVs based on different effects (e.g., survivorship vs. reproductive ability), toxicity endpoints (e.g., NOAEL, LOAEL), and/or exposure durations (e.g., acute, chronic) should not be summed to derive HIs. In these cases, risk assessment efforts should be focused on the highest contributing COPEC or class of COPECs which can reasonably be summed across effects, toxicity endpoints, and exposure durations (US EPA, 1999a).

Documentation of the risk estimates should describe how inferences are made from the measurement endpoints to the assessment endpoints established during problem formulation. For ecological risk assessments that rely upon multiple lines of evidence, a strength-of-evidence approach is used to integrate different types of data to support the conclusions of the assessment. The lines of evidence might include toxicity test results, assessments of existing impacts at a site, or risk calculations comparing exposures estimated for the site with toxicity values from the literature. Balancing and interpreting these different types of data can be a major task and require professional judgment. As already noted, the strength of evidence provided by different types of tests and the precedence that one type of study might have over another should have been established in the Work Plan. Taking this approach will ensure that data interpretation is objective and not biased to support a preconceived result. Additional strength-of-evidence considerations at this stage include the degree to which DQOs were met and whether confounding factors became evident during the site investigation and analysis phase of the risk assessment process.

For some biological tests (e.g., toxicity tests, benthic macroinvertebrate studies), all or some of the data interpretation process should be outlined in existing documents, such as in toxicity testing manuals. In most cases, however, the Work Plan or SAP (if available) must describe how resulting data will be interpreted for a site. The data interpretation methods also should be presented in the risk characterization documentation. For example, if the triad approach was used to evaluate contaminated sediments, the risk estimation section should describe how the three types of studies (i.e., toxicity test, benthic invertebrate survey, and sediment chemistry) are integrated to draw conclusions about risk.

Where exposure-response functions are not available or developed, the quotient method of comparing an estimated exposure concentration to a threshold for response can be used, as used in the SLERA. If possible, presentation of full exposure-response functions is preferred as these functions provide DWMRC with more information on which to base site decisions. This guidance has recommended the use of on-site contamination gradients to demonstrate on-site exposure-response functions. Where such data have been collected, they should be presented along with the risk estimates in the Tier 3 site specific ecological risk assessment report. HQs and HI s (for contaminants with the same mechanism of toxicity), the results of in situ toxicity testing, or community survey data can be mapped along with analytic chemistry data to provide a clear picture of the relationship between areas of contamination and observed or expected ecological effects.

In addition to developing point estimates of exposure concentrations (as provided by the hazard quotient approach), it may be possible to develop a distribution of exposure levels based on the potential variability in various exposure parameters. Probabilities of exceeding a threshold for adverse effects can then be estimated. As previously stated, the risk assessment team and DWMRC should agree on the specific analyses to be used in characterizing risks and document the procedures for the analyses in the Work Plan.

9.4.21 Risk Description

Risk descriptions for Tier 3 assessments should document the environmental contamination levels that bound the threshold for adverse ecological effects for each assessment endpoint. The lower bound of the threshold should be based on consistent conservative assumptions and NOAEL toxicity values while the upper bound should be based on observed impacts or predictions that ecological impacts could occur. This upper bound should be developed using consistent assumptions, site-specific data, LOAEL toxicity values, or an impact evaluation.

The approach for estimating environmental contaminant concentrations that represent thresholds for adverse ecological effects should be specified in the study design and documented in the Work Plan. When higher trophic-level organisms are associated with assessment endpoints, the study design should describe how monitoring data and contaminant-transfer models will be used to back-calculate an environmental concentration representing a threshold for effect. If the site investigation identifies a gradient of ecological effects along a contamination gradient, the risk assessment team should identify and document the levels of contamination below which no further improvements in the assessment endpoints are discernable or expected. If departures from the original analysis plan are necessary based on information obtained during the site investigation or data analysis phase, the reasons for the change should be discussed with DWMRC and the results of those discussions documented in the Tier 3 risk assessment report.

9.4.22 Additional Risk Information

In addition to developing numerical estimates of existing impacts, risks, and thresholds for ecological effects, the risk assessment team should establish the context of the estimates by describing their extent, magnitude, and potential ecological significance. Additional ecological risk descriptors are listed below:

- The location and areal extent of existing contamination above a threshold for adverse effects;

- The degree to which the threshold for contamination is exceeded or is likely to be exceeded in the future, particularly if exposure-response functions are available; and
- The expected half-life (qualitative or quantitative) of contaminants in the environment (e.g., sediments, food chain) and the potential for natural recovery once the sources of contamination are removed.

9.4.23 Uncertainty Analysis

There are several sources of uncertainties associated with ecological risk estimates. One is the initial selection of substances of concern based on the sampling data and available toxicity information. Other sources of uncertainty include estimates of toxicity to ecological receptors at the site based on limited data from the laboratory (usually on other species), from other ecosystems, or from the site over a limited period. Additional uncertainties result from the exposure assessment, because of the uncertainty in chemical monitoring data and models used to estimate exposure concentrations or doses. Further uncertainties are included in risk estimates when simultaneous exposures to multiple substances occur.

Within the analysis each source of uncertainty should be identified and its impact on the risk estimates and risk characterization discussed. Uncertainty should be distinguished from variability. Variability arises from true heterogeneity or variation in environmental characteristics and receptors. Uncertainty, on the other hand, represents lack of knowledge about certain factors, which can sometimes be reduced through additional study.

In general, there are two approaches to tracking uncertainties through a risk assessment:

- Using various point estimates of exposure and response to develop one or more point estimates of risk; and
- Conducting a distributional analysis to predict a distribution of risks based on a distribution of exposure levels and exposure-response information. Whether one or the other or both approaches are taken should have been agreed to by the risk assessment team and DWMRC and documented in the Work Plan.

9.4.24 Recommended Content of the Tier 3 Ecological Risk Assessment Report

In addition to the information delineated below, the report should include any other information about the site which the risk assessors consider relevant to evaluating the ecological risk at the site. For purposes of clarity, it is recommended that this additional information be included in an appendix to the Tier 3 Report and merely referenced in the main body of the report text.

The results of the Tier 3 COPECs selection process should be presented in a tabular format showing the final list of COPECs from the SLERA, the refined list of COPECs developed during Tier 3 problem formulation and technically defensible justification for each COPEC eliminated from or added to the refined list of site contaminants.

The following items should also be included in the Tier 3 Ecological Risk Assessment Report:

- A brief and concise but comprehensive summary of the information contained in the SLERA Report.
- The list of refined COPECs addressed in the Tier 3 assessment.
- A comprehensive summary of the results of all Tier 3 problem formulation activities.
- A description of all deviations from the Work Plan and SAP, including the circumstances that led to the deviations and the agreements with DWMRC on how to address those circumstances.
- A description of all in-field modifications to the approaches outlined in the Work Plan and/or SAP, including the circumstances that led to the need for in-field modifications and the agreements with DWMRC regarding the appropriate modifications for addressing those circumstances.
- Identification and discussion of the assumptions and uncertainties associated with the analysis of ecological exposures and ecological effects.
- A demonstration of the correlation between the contaminant gradients at the site and the ecological effects of the contaminant gradients, including any supporting lines of evidence needed to establish causality.
- Presentation and discussion of qualitative and quantitative risk results and the uncertainties reflected in the results.
- Number, type and size of habitats present in the assessment area.
- Sources of information are used to determine habitats.
- Plant and animal species typical of those habitats.
- All food webs developed for habitats occurring in the assessment area including:
    - Media for which web is constructed,
    - Division into trophic levels,
    - Class-specific guild designations for each trophic level, and
    - Major dietary interactions.
- Assessment endpoints selected for guilds and communities (and rationale).
- Measurement endpoints associated with identified assessment endpoints.
- Measures of effect selected for guilds and communities (and rationale).
- Integrated conceptual site exposure model.
- Estimated COPEC concentration in each component of each trophic level.
- Quantified exposure for each measurement receptor for each pathway.
- Summary of toxicity values used in the Tier 3 assessment.
- Results of HQ and HI calculations for each receptor if this approach is used in the Tier 3 assessment.
- Evaluation of nature/magnitude of risk at each site.
- Qualitative analysis of impact of all identified uncertainties on the ecological risk assessment process.

10.0 INTREPRETING RESULTS AND SITE MANAGEMENT

10.1 No Further Action

If it is determined that the site qualifies for NFA or risk-based clean-closure under the requirements of UAC R315-101, then no corrective measures are required, and the site will not be evaluated further.

It is possible that soil may qualify for NFA or risk-based clean-closure, but groundwater does not meet these criteria. In these cases, special post closure restrictions may be required limiting future use of groundwater.

A site qualifies for an NFA, or Corrective Action Complete without Controls, unrestricted land use, or risk-based clean closure and no site management plan (SMP) status when the level of risk present at the site is less than or equal to 1E-06 as the point of departure for carcinogens and the hazard index is less than or equal to one for non-carcinogens. This evaluation is based on only the residential land used exposure scenario. The NFA status in addition to the residential exposure scenario has to meet other criteria (1) ecological effects as the site should be insignificant (2) current impacts to groundwater at the site must meet the groundwater protection standards, (3) residual soil contamination present at the site should pose no future threat to groundwater.

Alternatively, where the soil medium meets the criteria of an NFA, but the groundwater medium does not meet the NFA criteria, the site may be divided into two media, the soil medium and the groundwater medium. The soil medium can be designated as NFA, and the groundwater medium will undergo further evaluation and may be restricted for its use or qualify as corrective action complete with controls.

10.2 Mixed Media NFA.

If it is determined that the site qualifies for NFA or risk-based clean-closure under the requirements of UAC R315-101, then no corrective measures are required, and the site will not be evaluated further. It is possible that soil may qualify for NFA or risk-based clean-closure, but groundwater does not meet these criteria. In these cases, the soil medium can be designated as NFA, and the groundwater medium will undergo further evaluation and may be restricted for its use or qualify as corrective action complete with controls.

10.3 Corrective Action Complete with Controls/Restricted Land/Use along with Requirements for Site Management Plan, (SMP)

A site may be considered for corrective action complete with controls or restricted land use if the level of risk present as the site is greater than 1E-06 but less than 1E-04. This risk range is considered the risk management range or site management range for carcinogens. For non-carcinogens, the hazard index must be less than or equal to one. These risk levels and hazard levels may be based on the actual land use exposure scenario evaluations could be residential or industrial/commercial or construction exposure land use.

Please note that a site cannot enter into site management unless risks and hazards meet these standards. Certain controls can be introduced into the SMP to mitigate risks. For residential land development where vapor intrusion may be driving risks, the main floor of the building could be parking garages while the upper-level floors would be residential. On the other hand, vapor intrusion mitigation system may be constructed on the main floor for residential dwellings to mitigate risks from vapor inhalation. These types of controls are considered engineering controls which are a subsect of land use controls (LUCs).

Other controls such as institutional controls (ICs) that may be unique to development at a site when the risk management range or site management range is attained, could be mixed-use development. Here development may be restricted to having commercial/industrial development on the main floor of the building while residential dwelling could be confined to the upper floors. Note that several other land use control options may come into play depending on site-specific conditions.

10.4 Corrective Action Requirements

Corrective action may be required at a site if the level of risk present at the site is greater than 1E-04 for carcinogens or a hazard index greater than one for non-carcinogens for any of the land use exposure scenarios. Also, the following conditions at a site may trigger corrective action at a site (1) ecological effects are significant at the site, or (2) groundwater contamination standards are exceeded on-site or off-site, or (3) residual contamination present at the site poses a potential threat to groundwater.

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