**New Jersey**
**Department of Environmental Protection**

# SOIL AND SOIL LEACHATE REMEDIATION STANDARDS FOR THE MIGRATION TO GROUND WATER EXPOSURE PATHWAY

# BASIS AND BACKGROUND

**May 2021**

## CONTENTS

1. Introduction .... 4
  1.1. Purpose .... 4
2. Soil and Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway .... 5
  2.1. Soil Remediation Standards for the Migration to Ground Water Exposure Pathway .... 5
    2.1.1. Calculations .... 6
    2.1.2. Default Input Parameters .... 8
    2.1.3. Adjustment of Health-based Soil Criteria when Deriving Soil Remediation Standards for the Migration to Ground Water Exposure Pathway (SRS-MGW) .... 19
  2.2. Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway ($MGW_{LEACHATE}$) .... 20
    2.2.1. Calculations .... 20
    2.2.2. Adjustment of Health-based Soil Leachate Criteria when Deriving the Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway .... 20
3. Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway (ARS-MGW) .... 21
  3.1. Development of Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway Using Site-Specific Parameters in the Soil-Water Partition Equation other than the Dilution Attenuation Factor .... 21
  3.2. Development of Soil Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway Using a Site-Specific Dilution Attenuation Factor (DAF) .... 21
  3.3. Development of Soil Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway using the Synthetic Precipitation Leaching Procedure .... 21
    3.3.1. Calculation of Leachate Concentrations under Field Conditions Using the Results of the Synthetic Precipitation Leaching Procedure .... 22
  3.4. Development of Soil Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway (ARS-MGW) using the SESOIL Model .... 29
  3.5. Development of Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway using the Combined SESOIL/AT123D Model .... 30
  3.6. Immobile Chemicals .... 30
  3.7. Site Soil and Ground Water Data Evaluation .... 33

## TABLES

Table 1 Soil Remediation Standards - Migration To Ground Water Exposure Pathway (SRS-MGW) ........ 35

Table 2 Soil Leachate Remediation Standards - Migration To Ground Water Exposure Pathway (($MGW_{leachate}$) ........ 40

Table 3 Chemical and Physical Properties of Contaminants ........ 45

Table 4 Calculated Infiltration Rates For New Jersey Soils Using New Jersey Geological Survey GSR-32 Methodology ........ 50

## APPENDICES

APPENDIX A Sensitivity of the Soil-Water Partition Equation to Modification of Component Parameters ........ 53

APPENDIX B Maps of Water Table Elevation and Hydraulic Conductivity Used in the Assessment of the Kirkwood-Cohansey Aquifer ........ 60

**References** ........ 61

## 1. Introduction

The Department has readopted remediation standards for ground water, surface water and soil at N.J.A.C. 7:26D. Soil remediation standards are proposed for the ingestion-dermal exposure pathway, the inhalation exposure pathway and the, migration to ground water exposure pathway. In this Basis and Background document, the Department will describe how the Soil Remediation Standards for the Migration to Ground Water exposure pathway (SRS-MGW) are derived. This Basis and Background document includes Tables1 & 2 which list respectively the Soil Remediation Standards for the Migration to Ground Water Exposure Pathway (SRS-MGW) and the Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway ($MGW_{LEACHATE}$) for listed contaminants.

### 1.1. Purpose

The purpose of the soil and soil leachate remediation standards for the migration to ground water (MGW) exposure pathway is to prevent unacceptable risk to human health from the ingestion of contaminated ground water, caused by the migration of contaminants from the unsaturated soil zone to the ground water. The Class II-A ground water remediation standards (GWRS) at N.J.A.C. 7:26D-2.2 are developed considering the potable water exposure scenario, which is based on the protection of human health. The Class II-A (GWRS) are the applicable endpoints from which the soil and soil leachate remediation standards for the migration to ground water exposure pathway are calculated. These standards apply only to Class II-A aquifers.

The primary designated use for Class II-A ground water is potable water. Class II-A consists of all the ground water of the State, except for ground water designated in Classes I, II-B or III. The soil and soil leachate remediation standards for the MGW exposure pathway are developed to protect the Class II-A ground water. Site-specific soil remediation standards for the migration to ground water (SRS-MGW)exposure pathway using GWRS developed on a site-specific basis by the Department must be developed for all other aquifer classifications.

Ground water must be protected even if it is not being used for potable purposes. In accordance with the Water Pollution Control Act, N.J.S.A. 58:10A-1 et seq. “It is the policy of this State to restore, enhance and maintain the chemical, physical, and biological integrity of its waters, to protect public health, to safeguard fish and aquatic life and scenic and ecological values, and to enhance the domestic, municipal, recreational, industrial and other uses of water.” In order to maintain the integrity of ground water, no addition of contaminants that would result in an exceedance of the GWRS is allowed.

The MGW exposure pathway must be investigated, initially by using the SRS-MGW when a discharge to soil is known or suspected to have occurred in the unsaturated zone. Unlike the direct contact standards, the MGW exposure pathway does not apply below the water table.

The soil and soil leachate remediation standards for the migration to ground water exposure pathway do not address current ground water conditions. The Ground Water - SI/RI/RA Technical Guidance Document (NJDEP 2012) details when a ground water investigation is needed, and when to install a well (see https://www.nj.gov/dep/srp/guidance/#pa_si_ri_gw ). Soil and soil leachate remediation standards for the migration to ground water exposure pathway are designed to prevent future contamination of the ground water from current soil contamination or residual contamination remaining after remediation. In the past, the standards pertaining to this pathway were sometimes inappropriately used as a trigger for ground water investigation. If ground water on a site is currently clean, it may be because contamination in the soil has not yet made its way to the water table. Alternatively, it may be that contamination in the soil is at a low enough concentration that it will never impact the ground water in exceedance of the applicable GWRS. Whenever there is a discharge or suspected discharge, the migration to ground water exposure pathway must be investigated and addressed along with the other exposure pathways. Soil sampling and comparison to the soil and soil leachate remediation standards for the migration to ground water exposure pathway is the only way to begin investigation as to whether the migration to ground water exposure pathway is potentially an issue for the site or area of concern in question. The migration to ground water exposure pathway must be addressed even when an active ground water remedy is in place. Existing remediation systems address current ground water contamination. The migration to ground water exposure pathway addresses the potential for future ground water contamination from the current soil contamination in the vadose zone. Therefore, the two are not connected. The migration to ground water exposure pathway must be addressed such that future contamination of the ground water does not occur in the absence of active remediation.

## 2. Soil and Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway

The soil and soil leachate remediation stadnards for the migration to ground water exposure pathway are developed to address site investigations and remedial investigations when a contaminant discharge is known or suspected and where little or no site information is available. In such situations, the soil and soil leachate remediation standards listed in Tables 1 and 2 may be used. If the soil and soil leachate remediation standards are exceeded, the migration to ground water exposure pathway must be addressed, either by remediation or by utilizing the guidance document "Alternative Remediation Standards Technical Guidance for Soil and Soil Leachate for the Migration to Ground Water Exposure Pathway" to derive an Alternative Remediation Standard for the Migration to ground Water Exposure Pathway (ARS-MGW). This technical guidance document is located at
http://www.nj.gov/dep/srp/guidance/rs/ .

There are two sets of standards for this exposure pathway. The first set in Table 1, the Soil Remediation Standards for the Migration to Gound Water Exposure Pathway, are based on the Soil Water Partition Equation and are soil standards which should be compared to total soil concentrations obtained by soil sampling. The second set in Table 2, the Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway, are used for soil tested with the Synthetic Precipitation Leaching Procedure (SPLP). SPLP results, adjusted to field conditions, are compared to the Soil Leachate Remediation Standards.

### 2.1. Soil Remediation Standards for the Migration to Ground Water Exposure Pathway

The soil remediation standards for the MGW exposure pathway represent the acceptable concentrations of contaminants in the unsaturated zone soil that do not result in an exceedance of the New Jersey Ground Water Remediation Standards when soil moisture passing through this soil drains down into the underlying saturated zone. Concentrations of contaminants measured on site that are below the soil remediation standards do not represent a concern for the migration to ground water exposure pathway.

### 2.1.1. Calculations

The Department used modified versions of the USEPA Soil Screening Level Soil-Water Partition Equations (SSL SWPE) (USEPA, 1996a, Equation 22 and 24) to calculate the soil remediation standards for the migration to ground water exposure pathway (SRS-MGW). The methodology back-calculates an acceptable concentration in soil from an acceptable ground water concentration (the Class II-A GWRS). The standard calculated is protective even when the contaminant is immediately adjacent to the water table.

The USEPA SSL SWPE assumes that contaminants in soil exist in equilibrium between the sorbed phase (on soil solids), aqueous phase (in soil moisture) and vapor phase (in the soil airspace). The equation calculates the total amount of the contaminant that may be left behind in the soil such that the aqueous phase concentration of the contaminant will not result in an exceedance of a specified criterion after the soil moisture transports downward and mixes into the water table. The criteria for New Jersey are the Ground Water Remediation Standards (GWRS). As infiltrating precipitation containing leached contaminant recharges an aquifer at the water table, it mixes with ground water, reducing the concentration of the contaminant in the leachate. The amount of dilution and the resulting contaminant concentration in the ground water can be calculated with the Dilution-Attenuation Factor (*DAF*). The *DAF* is dependent on the rate of ground water recharge and the ground water flow rate. Dilution of the contaminant due to transport through the unsaturated soil zone is not included; the chemical in soil is assumed to be immediately adjacent to the water table. Volatilization and chemical degradation are also not included in this model. Additionally, the calculations assume that the GWRS must be achieved immediately after remediation.

The equations for calculating the SRS-MGW are given below (Equation 1a and 1b). For New Jersey purposes USEPA SSL Equations 22 and 24 were expanded to separate the target leachate concentration discussed in the USEPA SSL guidance document into its component parts. The target leachate concentration is the product of the New Jersey Ground Water Remediation Standard (GWRS) and the dilution-attenuation factor (*DAF*). This modification allows the New Jersey GWRS to be directly entered as an input parameter. To calculate the SRS-MGW, a migration to ground water soil criterion ($MGW_c$) is calculated using Equations 1a or 1b below. Equation 1a is used for inorganic contaminants and Equation 1b is used for organic contaminants. The $MGW_c$ is then compared to the soil reporting limit (RL) and soil saturation limit ($C_{sat}$) for the contaminant, listed in Table 1. If the criterion is greater than its soil saturation limit, a soil remediation standard for the migration to ground water exposure pathway does not apply (see discussion of Equation 4 below); however, free and residual product must be treated, removed or contained in accordance with the Technical Requirements for Site Remediation (N.J.A.C. 7:26E-5.1(e).) If the criterion is less than the soil reporting limit for the contaminant, the soil remediation standard for the migration to ground water exposure pathway defaults to the soil reporting limit. Otherwise, the criterion calculated using Equation 1a or 1b becomes the soil remediation standard for the migration to ground water exposure pathway.

For inorganic contaminants:

$$MGW_c = GWRS * \frac{mg}{1000\mu g} * \left\{ K_d + \frac{\theta_w + (\theta_a * H')}{\rho_b} \right\} * DAF$$ **Equation 1a**

For organic contaminants:

$$MGW_c = GWRS * \frac{mg}{1000\mu g} * \left\{ (K_{oc} * f_{oc}) + \frac{\theta_w + (\theta_a * H')}{\rho_b} \right\} * DAF$$ **Equation 1b**

Where:

$MGW_C$ = migration to ground water soil criterion (mg/kg)
$GWRS$ = New Jersey Ground Water Remediation Standard (mg/L)
$K_d$ = Soil-water partition coefficient (L/kg)
$K_{oc}$ = Soil organic carbon-water partition coefficient (L/kg)
$f_{oc}$ = Organic carbon content of soil (kg/kg)
$\theta_w$ = Water-filled soil porosity ($L_{water}/L_{soil}$)
$\theta_a$ = Air-filled soil porosity ($L_{air}/L_{soil}$)
$H'$ = Henry's law constant (dimensionless)
$\rho_b$ = Dry soil bulk density (kg/L)
$DAF$ = Dilution-attenuation factor, calculated below (Equation 2)

The *DAF* is described in the USEPA Soil Screening Level (USEPA SSL) document (USEPA 1996a). The *DAF* is used in the various options for calculating impact to ground water soil remediation standards, including calculation of the Leachate Criterion.

The *DAF* is calculated via Equation 2. In addition to aquifer and site physical parameters, this equation requires a value for the mixing zone depth in the aquifer, which is calculated using Equation 3. These two equations are taken from USEPA SSL guidance document (Equations 37 and 45), respectively.

Equation for calculating the dilution-attenuation factor (*DAF*):

$$DAF = 1 + \frac{Kid}{IL}$$ **Equation 2**

Where:

$i$ = hydraulic gradient (m/m)
$d$ = mixing zone depth (m), calculated below (Equation 3)
$I$ = infiltration rate (m/yr)
$L$ = length of area of concern parallel to ground water flow (m)

$K$ = aquifer hydraulic conductivity (m/yr)

Equation for calculating the aquifer mixing zone depth, $d$:

$$d = (0.0112L^2)^{0.5} + d_a\{1 - \exp[(-LI)/(Kid_a)]\}$$ **Equation 3**

Where:

$d_a$ = aquifer thickness (m)

The soil saturation limit ($C_{sat}$) is calculated via Equation 4, and it applies to organic contaminants.

$$C_{sat} = \frac{S}{\rho_b}(K_{oc} f_{oc} \rho_b + \theta_w + H'\theta_a)$$ **Equation 4**

Where $C_{sat}$ is the soil saturation limit (mg/kg), $S$ is the contaminant's water solubility (mg/L), and the remaining parameters are as defined earlier. Values for the input parameters are the same as those for Equation 1. The equation is taken from Equation 9 of the USEPA SSL guidance document. The soil saturation limit corresponds to the contaminant concentration in soil at which the absorptive limits of the soil particles, the solubility limits of the soil pore water, and saturation of soil pore air have been reached. Above this concentration, soil contaminant will be present in free phase, i.e., nonaqueous phase liquids (NAPLs) for contaminants that are liquid at ambient temperatures and pure solid phases for compounds that are solid at ambient soil temperatures. These free phase contaminants are considered to be trapped (filtered) by the soil particles and therefore immobile. The exception would be for free product liquids, which can move downward under the influence of gravity. For this reason, free and residual product must be treated, removed or contained in accordance with the Technical Requirements for Site Remediation (N.J.A.C. 7:26E-5.1(e).). Otherwise, free phase contaminants are not of concern for the migration to ground water exposure pathway, which is concerned with aqueous phase contaminant transport to the water table. A calculated soil remediation standard for the migration to ground water exposure pathway that is above the $C_{sat}$ value for a contaminant means that the aqueous concentration in downward infiltrating water that would be required to exceed the GWRS in ground water cannot be achieved (i.e. is greater than the water solubility of the contaminant.)

### 2.1.2. Default Input Parameters

The methodology to calculate default remediation standards necessitates assigning default values to all input parameters for Equations 1-4 above. The default values used for properties such as such as chemical properties which do not follow region-specific trends are from USEPA Regional Screening Levels Tables located https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables. For other parameters, such as soil properties and applicable ground water criteria, values more specific to New Jersey conditions are warranted. A comparison of the USEPA and the Department's default parameters are listed below, followed by an individual discussion of each input parameter. A sensitivity analysis showing the effect of varying parameters was conducted and the results are presented in Appendix A.

| Parameter | USEPA Default Value | NJDEP Default Value |
|---|---|---|
| Groundwater Remediation Standard, GWRS (mg/L) | MCL or MCLG | NJ Ground Water Remediation Standard |
| Soil organic carbon-water partition coefficient, $K_{oc}$ (L/kg) | chemical specific | chemical specific |
| Soil-water partition coefficient, $K_d$ (L/kg) | chemical specific | chemical specific |
| Henry's law constant at 25°C, $H'$ (dimensionless) | chemical specific | chemical specific |
| Water solubility, $S$, (mg/L) | chemical specific | chemical specific |
| Soil pH | 6.8 | 5.3 |
| Soil texture | loam | sandy loam |
| Soil porosity (v/v) | 0.43 | 0.41 |
| Organic carbon content of soil, $f_{oc}$ (kg/kg) | 0.002 | 0.002 |
| Water-filled soil porosity, $\theta_w$ ($L_{water}/L_{soil}$) | 0.3 | 0.23 |
| Air-filled soil porosity content, $\theta_a$ ($L_{air}/L_{soil}$) | 0.13 | 0.18 |
| Dry soil bulk density, $\rho_b$ (kg/L) | 1.5 | 1.5 |
| *DAF* | 1 | 20 |
| Aquifer hydraulic conductivity, $K$ (ft/day) | Monte Carlo | 142 |
| Aquifer gradient, $i$ | Monte Carlo | 0.003 |
| Mixing zone depth, $d$ (m) | Monte Carlo | 3.5 |
| Infiltration rate, $I$ (m/yr) | Monte Carlo | 0.28 |
| Length parallel to groundwater flow, $L$ (m) | 45 | 30.5 |
| Aquifer thickness, $d_a$ (m) | Monte Carlo | 3.5 |

## New Jersey Ground Water Remediation Standard (GWRS)

USEPA value: Maximum Contaminant Level Goal (MCLG) or Maximum Contaminant Level (MCL)

NJDEP value: Ground Water Remediation Standard

For ground water protection, New Jersey must use the Ground Water Remediation Standards (N.J.A.C. 7:26D-2.2) and its subsequent updates. These standards are often lower than USEPA's corresponding criteria. As shown in the sensitivity analysis (Appendix A), the remediation standard is directly proportional to the value of the ground water standard.

## Chemical Properties of Contaminants

Chemical properties for all regulated contaminants are listed in Table 3. Most of the values are from USEPA Regional Screening Levels Tables located at https://www.epa.gov/risk/regional-screening-levels-rsls.
The USEPA Regional Screening Levels Tables have become a de facto national reference source for regulatory chemical properties and toxicity factors. Values were drawn from the May 2018 listing of chemical properties.

For ionizable organic contaminants, the Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites lists $K_{oc}$ values for any environmental pH value (USEPA 2002). The pH selected for New Jersey remediation standard calculations was 5.3 (see below). $K_d$ values (soil-water partition coefficients) were taken at pH 5.3 for those inorganic contaminants with pH dependent values reported in USEPA (2002). Diffusion coefficients are not used for calculation of the MGWSRS but are listed in Table 3 because they are used in the SESOIL model for calculation of alternative remediation standards.

The few exceptions to the above data sources are footnoted in the chemical property table (Table 3).

The remediation standard is most sensitive to a chemical's $K_{oc}$ value (Appendix A). The Henry's law constant has a relatively small effect (Appendix A), and the other chemical parameters are not used for calculation of remediation standards.

## Soil pH

USEPA value: 6.8

NJDEP value: 5.3

Although soil pH is not a direct input parameter in the soil standards calculations, it may affect the $K_{oc}$ value for ionizable organic contaminants and determines the $K_d$ value for several metals. The default pH of 6.8 used in the USEPA SSL guidance document is an overall average pH for United States soils. However, it is well known that soils in the eastern United States are more acidic than those in the western part of the country (Foth, 1984). Therefore, it is appropriate to use New Jersey-specific information regarding soil pH. The pH of New Jersey soils typically range from about pH 4 to pH 6.5 (Lee et al., 1996, Yin et al., 1996). A pH value of 5.3 is appropriate for New Jersey use.

## Soil Texture

USEPA: loam

NJDEP: sandy loam

Soil texture is not a direct input parameter in the soil standards calculations. However, several of the parameters in the soil-water partition equation are affected by soil texture. These are bulk density, air content, water content and infiltration rate. The first three parameters only minimally affect the calculated remediation standard (Appendix A). However, the infiltration rate does moderately affect the dilution-attenuation factor (*DAF*), which is used in Equations 1a and 1b. Heavier soils (such as loam soils, or those with significant clay content) retard water infiltration through soil relative to sandy loam soil or sand, and therefore moderately raise the value of the *DAF*. The USEPA uses loam as its default soil texture based on nationwide data. However, because the entire southern half of New Jersey is primarily composed of sand, loamy sand and sandy loam soils (Tedrow, 1986), it was felt that a loam soil texture was inadequately protective of the state. Sand is adequately protective for all soil types; however it was not used as the default soil texture because sand is too porous to be representative of northern New Jersey, which consists largely of sandy loam, loam and silt loam soils. Sandy loam soil was selected as a mid-range soil texture to represent the state when calculating remediation standards.

### Soil Porosity

USEPA value: 0.43

NJDEP value: 0.41

Soil porosity is not directly used in Equation 1a or 1b but is used when calculating soil air content. Soil porosity may range from 0.36 to 0.46 (v/v) (Carsel and Parrish, 1988). USEPA uses the porosity for loam soil, while the Department uses the value for sandy loam soil, which has been selected as the default soil texture for New Jersey. The value is selected from the same data source used by the USEPA (Carsel and Parrish, 1988).

### Dry Soil Bulk Density ($\rho_b$)

USEPA value: 1.5 kg/L

NJDEP value: 1.5 kg/L

Dry soil bulk densities vary over a relatively small range, from about 1.3-1.8 g/cc (Carsel et al., 1988). The USEPA default value was used because it agrees with the value listed for a sandy loam soil texture (Carsel et al., 1988). Remediation standards are only slightly affected by the value of this parameter (Appendix A).

### Water-filled Soil Porosity ($\theta_w$)

USEPA value: 0.3

NJDEP value: 0.23

Water-filled soil porosity (also known as soil moisture content, soil water content, volumetric soil water content) is highly specific to soil type and climate. The moisture content will vary according to season and short-term weather. In New Jersey, this variation for a sandy loam soil has been estimated to lie within the range of 0.18 to 0.26 (v/v) (Sanders and Talimcioglu, 1997).

For purposes of the New Jersey soil remediation standards calculations for the migration to ground water exposure pathway, it is best to use local climate data to determine average water content for a targeted soil. The targeted soil texture for New Jersey is sandy loam. USEPA's soil moisture value corresponds to a moisture level in between the field capacity of sandy loam soils and the saturation volume for loam soils and is higher than the actual average moisture level for sandy loam soil in New Jersey (Sanders and Talimcioglu, 1997).

For New Jersey, an average annual soil moisture content specific to sandy loam soil and New Jersey climate and weather conditions was determined using the simple relationship described in the USEPA Soil Screening Level User's Guide (USEPA, 1996b):

$$\theta_w = n(I / K_s)^{1/(2b+3)}$$

where $n$ is the total soil porosity, $I$ is the soil moisture infiltration rate (m/yr), $K_s$ is the saturated hydraulic conductivity of the soil (m/yr), and the factor 1/2b+3 is determined by the soil type and is provided in a lookup table in the User's Guide. Rather than calculating soil porosity as described in the User's Guide, which would yield a value of 0.43 (assuming a bulk density of 1.5 kg/L), a value of 0.41 for sandy loam was used, which was obtained from Carsel and Parrish (1988). This reference is USEPA's data source for soil properties for the USEPA SSL document. Additionally, a $K_s$ value of 387 m/yr for sandy loam soil (from Carsel and Parrish, 1988) was used instead of the lookup value of 230 m/yr provided in the user's guide because it is a more recent evaluation. The infiltration rate, $I$, was determined as discussed below to be 0.28 m/yr.

The average soil moisture determined via this method was 0.23 (v/v). If the alternative values for porosity and $K_s$ had been used, the soil moisture would have been calculated to be 0.25 (v/v). As shown in the sensitivity analysis (Appendix A), this alternative moisture value would have had a negligible effect on the calculated cleanup standard (only at the third significant digit).

### Air-filled Soil Porosity ($\theta_a$)

USEPA value: 0.13

NJDEP value: 0.18

Air-filled Soil Porosity (also known as volumetric soil air content) is determined from total porosity and soil moisture content. Refer to the discussion of Department values for those two parameters. This parameter has only a small effect on the remediation standard (Appendix A).

### Soil Organic Carbon Content ($f_{oc}$)

USEPA value: 0.002
NJDEP value: 0.002

Organic carbon content can vary from near zero (beach sands and other sandy soils at subsurface depths) to several percent (surface soils in forests). The USEPA judged that a fraction organic carbon content of 0.002 was appropriate for subsurface soils. The organic carbon content of soil has not been well documented below 1-2 meters in depth, but Carsel et al. (1988) performed statistical analyses of a large soil dataset and reported distributions of soil organic matter contents at various depth intervals up to 1.2 m depth. The average fraction organic carbon content of the three mean subsurface values for Class B and Class C soils was 0.002. These hydrologic soil groups include sandy loam soils. Therefore, the Department has also decided a default fraction organic carbon content of 0.002 is appropriate. Increasing the $f_{oc}$ value raises the soil remediation standard (Appendix A).

Saturated Hydraulic Conductivity (*K*) and Hydraulic Flow Gradient (*i*)

USEPA: Monte Carlo Distribution

NJDEP: *K*: 15,808 m/year (142 ft/day)
*i*: 0.003

Representative values for the hydraulic gradient (*i*) and hydraulic conductivity (*K*) for the Kirkwood-Cohansey aquifer were used to determine a default *DAF* value for New Jersey. This 3,069 square mile aquifer is relatively shallow, lies underneath soils with considerable sand content, and often exhibits low flow rates due to generally flat terrain. It comprises almost 75% of the N.J. Coastal Plain, and being a surficial water table aquifer, is vulnerable to contamination. Many water resource studies have been completed for this aquifer system because it is widespread and supplies potable water to many communities. Therefore, a significant amount of hydrogeologic data is available. For these reasons, it was selected as an appropriate aquifer for determining representative *K* and *i* values for New Jersey.

A statistical analysis of data for the Kirkwood-Cohansey aquifer was conducted. The hydraulic gradient (*i*) was calculated by measuring the potentiometric surface at 235 locations on maps of water table elevations in eight watershed basins (listed in Appendix B), and median, mean, geometric mean and mode values were calculated. Statistical analysis typically requires at least 30 data points, while 200 are considered sufficient for rigorous applications. The analysis indicated a tight grouping in the measurements over the entire Kirkwood-Cohansey system, leading to confidence in the results. The geometric mean is a “smoothing mean”. Its value was close to both the median and mean values and is evidence of the tight grouping in the measurements. The statistics for the Kirkwood-Cohansey gradient were as follows:

| | |
|---|---|
| Mean | 0.004 |
| Median | 0.003 |
| Geometric Mean | 0.003 |
| Mode | 0.002 |
| Std. Deviation | 0.002 |
| Coeff. of Variation | 0.67 |

Investigators report a geologic formation dip of 10-60 feet/mile (vertical gradient of 0.002-0.011) for the N.J. Coastal Plain (Volwinkel and Foster, 1981). On the scale of the Coastal Plain, one can reasonably assume this dip mirrors the hydraulic gradient in a water table aquifer; so the calculated values above appear compatible with this range. A gradient of 0.003 was selected as appropriate for this aquifer.

To determine a representative hydraulic conductivity for the Kirkwood-Cohansey aquifer, data was collected from the results of 67 aquifer stress tests. Thirty-three were compiled by the New Jersey Geological Survey (NJGS) (Canace and Sugarman, 2009), 13 were from the U. S. Geological Survey (USGS) (Martin, 1990); and 23 were taken from the eight individual watershed basins reported on the water table maps listed below. The NJGS and USGS reported median hydraulic conductivities for the compiled tests; individual test values were not reported. The basin tests were reported individually, but detailed statistical analysis was not attempted because less than 30 test results were available. Results were checked for redundancy; all 67 tests from these sources are unique. Results are as follows:

- 46 NJGS and USGS Tests Median (weighted) hydraulic conductivity: 138 ft./day.
- 21 Basin Tests from the aquifer maps: Median hydraulic conductivity: 150 ft./day

An overall weighted median value of 142 ft./day (15,808 m/year)was calculated from the two values above and is consistent with the surficial Kirkwood-Cohansey aquifer material type (medium to coarse sand) and high well yields.

*Additional considerations regarding New Jersey K and i values*

Hydraulic gradients were measured in eight of the eleven major watersheds for the Kirkwood-Cohansey aquifer in the Coastal Plain. Water resource investigations have not been completed for the remaining three, and potentiometric data is therefore unavailable for analysis.

Regarding other aquifers in the Coastal Plain, unconsolidated sediments make up the remainder of the surficial water table aquifer in the westernmost outcrop areas of the inner Coastal Plain. Hydraulic data for the underlying aquifers that outcrop in this area (between the Kirkwood-Cohansey and the Delaware River) were too few for reliable estimates. However, these outcrop areas are limited in geographic extent and were judged to be similar to the Kirkwood-Cohansey. Estimates of infiltration rates as an indicator of hydraulic conductivity in these surficial aquifers, as well as their material character, suggests that the Kirkwood-Cohansey surficial water aquifer is representative of these other aquifers for Default *DAF* purposes. Local variations are likely, but also occur in the Kirkwood-Cohansey.

While these *K* and *i* values are based on the coastal plain of New Jersey, the resulting *DAF* calculated using these values was judged to be representative for the entire state (see *DAF* section below).

The remediation standard is approximately linear with respect to hydraulic conductivity and the aquifer gradient (Appendix A).

Infiltration Rate (*I*)

USEPA: Monte Carlo Distribution

NJDEP: 0.28 meters/year (11 in/year)

The infiltration rate corresponds to the rate of recharge of precipitation to the ground water. The infiltration rate is an input parameter for calculating a *DAF*.

The New Jersey Geological Survey has published a methodology (GSR-32) for determining infiltration rates for New Jersey as a function of location, soil type and land use (Hoffman, 1999; Charles et al., 1993). Using several of the most commonly occurring soils in New Jersey (Tedrow, 1986), infiltration rates were calculated for each soil in each county where the soil had a significant presence (Table 4). For each calculation, data from a climate station from a municipality located in the area where the soil would occur was used. Three land uses were selected for each calculation: landscaped, bare soil, and agricultural soil. All three of these soil types assume 100% of the surface area is permeable. All sandy loam soils with significant acreage in the state (as mapped by Tedrow, 1986) were used, since this soil texture has been targeted as the default soil texture for New Jersey standards. In addition, other soil textures with a large presence in the state (as mapped by Tedrow, 1986) were also studied, in order to determine the overall variation of infiltration rates in the state, and to verify that sandy loam soil was appropriate as a default soil texture. One limitation of this method that should be mentioned is that the infiltration calculated (below the root zone) is assumed to be equal to ground water recharge (Charles et al., 1993).

Infiltration rates for New Jersey soils ranged from about 5-17 inches per year (Table 4). A summary table of the results is provided below:

| **Soil Texture** | **Average infiltration rate (in/yr)** | | | |
|---|---|---|---|---|
| | **Landscaped** | **Unvegetated** | **Agriculture** | **Overall** |
| Sandy loam | 12.8 | 8.4 | 11.3 | 10.8 |
| Sand | 16.5 | 14.0 | 15.6 | 15.4 |
| Loamy sand | 13.2 | 9.0 | 11.8 | 11.3 |
| Loam | 14.0 | 6.8 | 11.7 | 10.8 |
| Silt loam | 11.8 | 5.2 | 9.9 | 9.0 |
| All soils | 13.6 | 8.6 | 11.9 | 11.3 |

For sandy loam, loamy sand, and loam soils, an 11 in/yr (0.28 m/year) infiltration is representative. Silt loam soils have slightly lower infiltration rates, while sand soils yield rates a few inches higher. As discussed in the standards section of this document, it was decided to use sandy loam soil texture as the default soil type for New Jersey, as it was felt that use of a sand soil would be overly protective for much of the state. The results above confirm that assuming an infiltration rate of 11 in/yr (0.28 m/year) is adequately protective for sandy loam soil and the other remaining soil textures.

The remediation standard is inversely proportional with respect to the infiltration rate (Appendix A).

Source (Area of Concern) Length Parallel to Ground Water Flow (*L*)

USEPA default value: 45 m (148 feet)

NJDEP default value: 30.5 m (100 feet)

This parameter is equivalent to the length of the Area of Concern (AOC) parallel to ground water flow and is an input parameter in calculating a *DAF*. The Department's value results in higher remediation standards than if USEPA's value was used. The 30.5 m source length was judged to be larger than most Areas of Concern in New Jersey, and therefore adequately protective. This is also approximately equal to the length of a high-density residential lot size (¼ acre). The source length affects the *DAF* and remediation standard only if it results in a calculated mixing zone thickness that is larger than the aquifer thickness (see Appendix A)

Mixing Zone Depth (*d*)

USEPA: Monte Carlo Distribution

NJDEP: 3.4 m

The mixing zone depth corresponds to the depth to which the contaminant is diluted in ground water. It is calculated from the mixing zone depth equation (Equation 3) using several other field parameters. The mixing zone depth is then used in the *DAF* Equation (Equation 2). Using the default values for all the parameters that are used in this equation, the default mixing zone depth is calculated to be 3.4 m, which is slightly less than the default aquifer thickness. The parameter remains at this value under the default scenario even if the aquifer thickness is increased. Sensitivity analysis was not conducted for this parameter, because its dependent parameters are incorporated in the sensitivity analysis for the DAF equation (Appendix A).

Thickness of Affected Aquifer ($d_a$)

USEPA: Monte Carlo Distribution

NJDEP: 3.5 m (11.5 ft)

The aquifer thickness is used in calculating the aquifer mixing zone depth, which in turn is used in calculating the *DAF*. For the site size selected, the calculated mixing zone is independent of the aquifer thickness if it is 3.4 m or greater. Since an aquifer thickness of 3.5 m (11.5 ft) is considered a thin aquifer, this value was considered to be adequately protective and used as the default value. Varying this parameter has no effect on the calculated remediation standard under the default scenario unless its thickness is less than 3.4 m.

Dilution/Attenuation Factor (*DAF*)

USEPA value: 1 or 20

NJDEP value: 20

Substituting the parameters discussed above into Equations 1 and 2, a default *DAF* of 20 was calculated.

While most of the parameters for the *DAF* were based on a statewide representative values, the *K* and *i* values were based on an assessment of the coastal plain of New Jersey. The remainder of New Jersey has more complex geology. Unlike the New Jersey coastal plain, where aquifer properties viewed over a large scale are less variable and the hydraulic setting more uniform, northern New Jersey overburden aquifers are heterogeneous. Successive glaciations have reworked and re-deposited mixtures of clay, sand, silt, and glacial lake sediments. Accordingly, hydraulic properties can vary significantly over short distances. Furthermore, few regional studies have been completed in the northern part of the state, so data are limited relative to that available for the coastal plain. For this reason, representative *K* and *i* values could not be determined for the northern part of the state.

Since statewide *K* and *i* values are not available for northern New Jersey, it was of interest to examine other information that was available regarding dilution-attenuation factors that may apply to New Jersey, particularly the northern part of the state.

The USEPA conducted a nationwide assessment of dilution-attenuation Factors (*DAF*s) in its Soil Screening Guidance Document (USEPA 1996a). Nationally, *DAF* values for a half-acre site were found to range from one to several thousand. To derive a default *DAF* value, the USEPA used a “weight of evidence” approach to derive its default attenuation factor of 20. This was based on two studies where attenuation factors were estimated or calculated.

In the first study, USEPA’s Composite Model for Leachate Migration with Transformation Products (EPACMTP) model was used to derive *DAF* values by running the model in the Monte Carlo mode. *DAF* distributions were generated using expected variations in the input parameters that are used in its calculation. While this approach has its advantages, it resulted in a nationwide distribution of *DAF* values that were inappropriate for New Jersey use, based on policy considerations. This included the fact that the *DAF* values were calculated at the location of a receptor well, which was varied in its location and was often outside the main body of the contaminated groundwater plume. If the well was outside the plume, a high *DAF* was calculated. This is incompatible with New Jersey policy, since the probable location of a receptor well is not considered in the New Jersey Ground Water Quality Standards. All groundwater is to be protected for potential potable uses. Therefore, the *DAF* should be calculated within the plume itself. Additionally, the USEPA assumed a variable distance between the down gradient edge of the source and the receptor well. This is also incompatible with the Ground Water Quality Standards (N.J.A.C. 7:9C), which require compliance at the down gradient edge of the source.

In the second study, the USEPA used data from two large surveys of hydrogeological site investigations, and calculated *DAF* values using the *DAF* equations presented in this document. The two surveys were the American Petroleum Institute’s (API’s) hydrogeologic database (HGDB) and USEPA’s database of conditions at Superfund sites contaminated with DNAPL (USEPA 1996a). Between these two databases, a total of 300 *DAF* calculations were made. The sites are classified according to hydrologic region in the United States. Three of these regions, the Northeast and Superior Uplands, the Glaciated Central Region, and the Atlantic and Gulf Coast, are appropriate for New Jersey.

Using information presented in the HGDB database of the USEPA Soil Screening Guidance (Appendix F, HGDB Database – National Average), the NJDEP examined USEPA-calculated *DAF* values for the two geologic regions found in northern New Jersey. Seventeen *DAF* values were chosen that best represented northern NJ geology (glaciated and upland regions) and the default size of the area of concern (0.5 acre source area). Statistics were applied to the selected values for *DAF* and yielded the following results:

- Arithmetic Mean: 36.5
- Geometric Mean: 17.8
- Median: 21
- Standard Deviation: 45.9
- Coefficient of Variation: 1.26

While this assessment is not as rigorous or quantitative as that conducted for the southern part of the state, the geometric mean and median *DAF* values shown above for northern New Jersey are similar to the default *DAF* of 20 determined using default *K* and *i* values for the coastal plain.

For the Atlantic Coast region, the HGDB database contained 15 values reported for the Atlantic Coast region for unconsolidated and semi-consolidated shallow surface aquifers (0.5 acre site size). Data were highly skewed, with a median *DAF* of 3 and mean of 30. While the variation in results was high, a *DAF* of 20 is near the middle of the range of values observed.

The DNAPL database yielded 50 sites in the Uplands region and 12 sites in the Atlantic Coast region with median *DAF* values (0.5 acre site size) of 22 and 20, respectively. (Glaciated Region data was inadequate for assessment.) While these median values support the NJDEP default *DAF* of 20, none of the sites in the database were in New Jersey, and less details were available regarding the hydrogeologic settings than for the HGDB database.

The NJDEP study of the Kirkwood-Cohansey aquifer provides the best data regarding dilution-attenuation factors values in New Jersey and led to selection of a default *DAF* of 20. The available USEPA data discussed above, while not specific to New Jersey, and not as quantitatively rigorous, also lend support to this default *DAF* value.

Therefore, it was judged that a *DAF* of 20 was suitable as a default dilution-attenuation factor for the entire state.

Increasing the *DAF* value raises the value of the remediation standard (Appendix A).

Avoiding redundant conservative assumptions in default parameter values

In accordance with the legislative mandate to avoid the use of redundantly conservative or unrealistic assumptions when calculating remediation standards, realistic, or typical, values, were used for the default input parameters, as shown in the following table:

| Parameter | Value selected is: |
|---|---|
| Ground Water Remediation Standard, GWRS (mg/L) | chemical-specific |
| Soil organic carbon-water partition coefficient, $K_{oc}$ (L/kg) | chemical-specific |
| Soil-water partition coefficient, $K_d$ (L/kg) | chemical-specific |
| Henry's law constant at 25°C, $H'$ (dimensionless) | chemical-specific |
| Soil pH | typical |
| Soil texture | typical |
| Soil porosity (v/v) | typical |
| Organic carbon content of soil, $f_{oc}$ (kg/kg) | typical subsurface |
| Water-filled soil porosity, $\theta_w$ ($L_{water}/L_{soil}$) | typical |
| Air-filled soil porosity, $\theta_a$ ($L_{air}/L_{soil}$) | typical |
| Dry soil bulk density, $r_b$ (kg/L) | typical |
| *DAF* | typical |
| Aquifer hydraulic conductivity, $K$ (m/yr) multiplied by Gradient, $i$ (m/m) | typical |
| Mixing zone depth, $d$ (m) | constant for all aquifer thicknesses > 3.4 m |
| Infiltration rate, $I$ (m/yr) | typical |
| Length parallel to groundwater flow, $L$ (m) | typical |
| Aquifer thickness, $d_a$ (m) | Does not affect results when > 3.4 m |

Some parameters, such as the chemical properties and applicable standards, are not subject to variation, and are listed as “chemical specific”, and others either had no effect on results or are constant. See the discussion of individual parameters above for more information.

### 2.1.3. Adjustment of Health-based Soil Criteria when Deriving Soil Remediation Standards for the Migration to Ground Water Exposure Pathway (SRS-MGW)

Soil remediation standards for the migration to ground water exposure pathway (SRS-MGW), along with the GWRS, health-based soil criteria, soil saturation limits, and soil reporting limits are listed in Table 1. As discussed in section 2.1.1, Migration to Ground Water Soil Criteria ($MGW_c$) are initially calculated using Equations 1a and 1b. These are health-based criteria. If no other adjustments to a criterion is necessary, it becomes the soil remediation standard for the migration to ground water exposure pathway. However, as discussed in Section 2.1.1, the soil remediation standard for the migration to ground water exposure pathway is sometimes limited by the soil reporting limit, which is the lowest enforceable regulatory concentration in soil. Additionally, an $MGW_c$ is not applicable as a standard when it is calculated to be above the soil saturation limit. Finally, in cases when a GWRS is not available or is not a health-based standard, a soil remediation standard for the migration to ground water exposure pathway was not calculated. The Brownfield and Contaminated Site Remediation Act (N.J.S.A 58:10B-12) requires calculated standards to be health-based. The GWRS for aluminum and manganese are secondary; that is, they are not based on health considerations, but primarily on aesthetic considerations such as taste, odor and appearance. Additionally, these elements may be found as background contaminants. Therefore, the Department has decided that the migration to ground water exposure pathway does not need to be addressed for aluminum and manganese unless there is cause to believe that their presence is due to a site discharge.

For arsenic, a state-wide natural background concentration of 19 mg/kg has been determined, after a review of available ambient background information from across the state. Since standards are not to be set below established natural background levels, the soil remediation standard for the migration to ground water exposure pathway defaults to 19 mg/kg for this contaminant.

## 2.2. Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway ($MGW_{LEACHATE}$)

The soil leachate remediation standards for the migration to ground water exposure pathway represent the acceptable concentrations of contaminants in the unsaturated zone soil aqueous phase that do not result in an exceedance of the New Jersey GWRS when this soil water drains down to and dilutes into the underlying saturated zone. The standards are used by comparing them to field leachate concentrations, which are calculated using sample results from the Synthetic Precipitation Leaching Procedure (SPLP). The soil leachate remediation standards for the migration to ground water exposure pathway, in conjunction with the SPLP procedure, are used to determine an ARS-MGW. See the Alternative Remediation Standards Technical Guidance for Soil and Soil Leachate for the Migration to Ground Water Exposure Pathway located at http://www.nj.gov/dep/srp/guidance/rs/ for further information.

### 2.2.1. Calculations

Since it has been determined that the appropriate default dilution-attenuation factor (DAF) is equal to a value of 20 for New Jersey purposes, (see section 2.1.2 above), health-based soil leachate criteria are initially calculated by multiplying the GWRS by a factor of 20. To derive soil leachate remediation standards for the migration to ground water exposure pathway, further adjustments are sometimes needed as discussed in Section 2.2.2

### 2.2.2. Adjustment of Health-based Soil Leachate Criteria when Deriving the Soil Leachate Remediation Standards for the Migration to Ground Water Exposure Pathway

Soil leachate remediation standards for the migration to ground water exposure pathway ($MGW_{LEACHATE}$), along with the GWRS, are listed in Table 2. As discussed in Section 2.1.1 above, some contaminants do not have soil leachate remediation standards for the migration to ground water exposure pathway because the health-based soil criteria ($MGW_c$) are above the soil saturation limit. For these same contaminants, health-based soil leachate criteria are above the contaminant's water solubility limit, which means that aqueous concentrations in the soil leachate corresponding to the health-based soil leachate criteria are not possible. Therefore, these contaminants also do not have soil leachate remediation standards. Other contaminants do not have ($MGW_{LEACHATE}$) for the MGW exposure pathway because a GWRS is not available. Finally, as discussed in Section 2.1.3 above, the GWRS for aluminum and manganese are not health-based, as required for the purposes of the New Jersey soil remediation standards. Therefore, these two contaminants do not have soil leachate remediation standards for the migration to ground water exposure pathway.

## 3. Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway (ARS-MGW)

Several options are available for determination of alternative remediation standards for the migration to ground water exposure pathway. Consult the available guidance document: "Alternative Remediation Standards Technical Guidance for Soil and Soil Leachate for the Migration to Ground Water Pathway", located at http://www.nj.gov/dep/srp/guidance/rs/ for more information. Following is a brief description of the ARS-MGW options available, along with supporting basis and background information where appropriate.

### 3.1. Development of Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway Using Site-Specific Parameters in the Soil-Water Partition Equation other than the Dilution Attenuation Factor

ARS-MGW are back calculated from the GWRS (N.J.A.C. 7:26D), using area of concern (AOC)-specific or site-specific values for one or more parameters in the USEPA Soil-Water Partition Equation (USEPA 1996a).

### 3.2. Development of Soil Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway Using a Site-Specific DilutionAttenuation Factor (DAF)

When AOC-specific or site-specific information is available regarding the ground water flow rate, it may be used to calculate a different *DAF* value. If the AOC-or site-specific *DAF* is higher than the default value of 20, this will result in a ARS-MGW that is greater than the soil remediation standard for the migration to ground water exposure pathway.

### 3.3. Development of Soil Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway using the Synthetic Precipitation Leaching Procedure

The Synthetic Precipitation Leaching Procedure (SPLP) determines the area of concern (AOC)- or site-specific leachability of the contaminant. The SPLP procedure will provide an accurate measure of this mobility for these types of contaminants and may be used to develop a soil alternative remediation standard for the migration to ground water exposure pathway.

The SPLP test uses Method 1312 from the USEPA SW-846 compendium of analytical and test methods (https://www.epa.gov/hw-sw846). Method 1312 measures the leaching potential of a contaminant in soil, and thus offers a quick and inexpensive method to assess chemical mobility. A particularly useful aspect of the SPLP procedure is that it measures desorption, rather than adsorption, of contaminants from soil. It is well known that desorption of many contaminants to soil decreases as contact time increases (Riley et al., 2010; Sander et al., 2006; Fendorf et al., 2004; Lock and Janssen, 2003, Loehr and Webster, 1996). In some cases, a portion of the contaminant may become irreversibly adsorbed to soil and therefore immobile. Default values for soil adsorption coefficients ($K_{oc}$ or $K_d$ values) used in the USEPA SSL partition equation do not consider these processes. While the USEPA partition equation could be used with desorption, rather than adsorption coefficients, values for desorption coefficients are not generally available and appear to be site-specific. For metals, an additional complication arises in that the $K_d$ values used in the USEPA soil-water partition coefficient for default screening levels assume that the most mobile species of the contaminant is present (USEPA 1996a). This is necessary because standard analytical methods usually measure total metals, so the actual species (redox state, salt, or complex) is not typically known. Because different species of a metal have widely varying mobility, the assumption of a mobile species is made in order to be adequately protective of all situations that may occur.

The SPLP procedure directly measures the current desorption (leaching) potential of the contaminant, because it uses actual aged, contaminated soil from the site. For metals, the leaching potential of the species present in the soil will be determined, whether or not it is actually known. While a detailed assessment of the results of this test may be complex when mixtures of different species of the same metal are present, the procedure nonetheless provides a much-improved estimate of leaching tendency over predictive methods using lookup $K_{oc}$ or $K_d$ adsorption constants.

The conditions of the SPLP test simulate actual environmental precipitation, in that the leaching solution is a simulation of mid-Atlantic rainfall (pH 4.2). Thus, the test more realistically estimates the leaching potential of contaminants that may occur under field conditions in New Jersey (Brown et al., 1996; Lackovic et al., 1997).

Since the SPLP test exhibits several advantages over other methods to determine the leaching potential of contaminants, and because it can be conducted in a time and cost-efficient manner during a site investigation, the use of this test is often recommended prior to pursuing other options when determining alternative impact to ground water remediation standards at contaminated sites.

The limitations of the SPLP test are as follows. First, because leachate is filtered through a 0.6 - 0.8 μm filter, the concentration of colloidal inorganics above this pore size may be underestimated. Second, because the oxidation/reduction potential of the sample is not preserved when the test is conducted, interconversion of metal species with multiple oxidation states may occur.

### 3.3.1. Calculation of Leachate Concentrations under Field Conditions Using the Results of the Synthetic Precipitation Leaching Procedure

When using the SPLP procedure to calculate a soil alternative remediation standard for the migration to ground water exposure pathway, the SPLP test results are used in conjunction with the soil leachate remediation standards for the MGW water exposure pathway. Leachate results from the SPLP procedure must be adjusted to field leachate concentrations in order to compare them with the ($MGW_{LEACHATE}$). This adjustment is derived using the USEPA Soil-Water Partition

Equation (Equation 1a), which in turn is based on the well-established linear adsorption-desorption model to describe partitioning of contaminants between soil solids, soil moisture, and soil air.

The SPLP Procedure is a batch equilibrium procedure which measures desorption of contaminant from soil after shaking overnight with an extracting solution. It is assumed that equilibrium is achieved between the sorbed and aqueous (leachate) phases at the end of the experiment. Specifically, the ratio of the sorbed and aqueous phase concentrations (known as the $K_d$ parameter) is constant over a range of contaminant concentrations and soil-to-water ratios. The SPLP test may conveniently be used to measure the $K_d$ value for a particular contaminant and soil type. The $K_d$ parameter also provides the foundation of the USEPA soil-water partition equation, which additionally includes a vapor phase. As will be demonstrated below, leachate concentrations measured under the conditions of the SPLP test do not necessarily represent leachate concentrations that would be observed in the field because the water to soil ratio affects the resulting leachate concentration. For chemicals that are not strongly adsorbed, a large percentage of the initial contaminant mass desorbs from soil during the SPLP extraction because of the large volume of extracting solution relative to soil. This results in the final soil sorbed concentration being much lower than the initial total concentration, which is balanced by a low leachate concentration in the large volume of extracting solution in order to maintain the correct equilibrium concentration ratio. Under field conditions, the much lower volume of water results in a much lower decrease in the final soil sorbed concentration, which is balanced by a higher leachate concentration in the small leachate volume. Leachate concentrations under field conditions are the relevant data needed to compare against the leachate criteria or to calculate site-specific impact to ground water remediation standards. These can be determined using the SPLP results, the known initial total soil concentration, and the assumptions underlying the basis of the USEPA soil water partition equation.

There are two fundamental relationships governing the equilibrium behavior of contaminants in soil. The first is that for the soil-water adsorption-desorption coefficient:

$$K_d = \frac{C_S}{C_W}$$ **Equation 5**

where, at equilibrium, $C_s$ is the concentration of the chemical in the soil sorbed phase, $C_w$ is the concentration of the chemical in the aqueous phase, and $K_d$ is the soil-water partition coefficient. For a particular soil, $K_d$ is assumed to be constant over a range of total concentrations. Note that the relationship requires the <u>ratio</u> of the two concentrations to be constant, <u>not the absolute concentration in each phase.</u> Therefore, as the relative amounts of soil and water change, the absolute concentrations in each phase must vary in order to keep the ratio constant. Since the $K_d$ is a constant, it remains the same under both field and SPLP leaching conditions.

The above equation may be expanded:

$$K_d = \frac{m_S/M_S}{C_L}$$ **Equation 6**

where $C_s$ is now expressed as the mass of contaminant, $m_s$, in a given mass of soil, $M_s$. $C_w$ has now been relabeled $C_L$, the leachate concentration at equilibrium.

The second relationship describes equilibrium partitioning between the air phase and the water (aqueous) phase:

$$H' = \frac{C_a}{C_w}$$ **Equation 7**

where, at equilibrium, $C_a$ is the concentration of the chemical in the air phase, $C_w$ is the concentration of the chemical in the water (aqueous) phase, and $H'$ is the dimensionless Henry's law constant, also assumed to be constant over a range of concentrations. This equation may also be expanded:

$$H' = \frac{m_a/V_a}{C_L}$$ **Equation 8**

where $C_a$ is now expressed as the mass of contaminant, $m_a$ in a given volume of air, $V_a$, and $C_L$ is again relabeled the leachate concentration. This equation may be rearranged:

$$m_a = C_L V_a H'$$ **Equation 9**

The total contaminant concentration in soil before leaching may be defined as $C_T$, and the volume of the extracting leaching solution as $V_L$. The mass of contaminant sorbed to soil at equilibrium after leaching may be calculated as the total mass of contaminant in the soil prior to the leaching experiment, $C_T M_S$, minus the mass of contaminant that ends up in the leachate at equilibrium, $C_L V_L$, minus the mass of contaminant in the air phase, $C_L V_a H'$. Substituting these terms into Equation 6 yields

$$K_d = \frac{(C_T M_S - C_L V_L - C_L V_a H')/M_S}{C_L}$$ **Equation 10**

Rearranging, the equation transforms to:

$$C_L K_d M_s = C_T M_S - C_L V_L - C_L V_a H'$$ **Equation 11**

$$C_T M_S = C_L \mathrm{K}_d M_S + C_L V_L + C_L V_a H'$$ **Equation 12**

$$C_T M_S = C_L(K_d M_s + V_L + V_a H')$$ **Equation 13**

$$C_T = C_L \frac{K_d M_S + V_L + V_a H'}{M_S}$$ **Equation 14**

$$C_T = C_L\left(K_d + \frac{V_L + V_a H'}{M_S}\right) \qquad \textbf{Equation 15}$$

Under soil conditions in the field, $V_L$ is the value for the water-filled soil porosity (that is $\theta_w$), $V_a$ is the value for the air-filled soil porosity (that is $\theta_a$), and $M_S$ is the mass of soil per unit volume (dry soil bulk density), $\rho_b$. Equation 15 thus transforms to the USEPA soil-water partition equation:

$$C_T = C_L\left(K_d + \frac{\theta_w + \theta_a H'}{\rho_b}\right) \qquad \textbf{Equation 16}$$

where

$\theta_w$ = 0.23 mL of water per mL of soil volume (NJ default field conditions)
= 2 L of water in 2.038 L volume in SPLP test (0.5 L water in 0.51L volume for volatiles)
$\theta_a$ = 0.18 mL of water per mL of soil volume (NJ default field conditions)
= 0 in SPLP test
$\rho_b$ = 1.5 g of soil per mL of soil volume (NJ default field conditions)
= 100 g of soil per 2.038 L volume in SPLP test (25 g soil in 0.51 L volume for volatiles)

The total volume of the SPLP test is approximately 2.038 L, due to the soil solids volume of approximately 0.039 L per 100 grams soil, assuming a soil particle density of 2.65 g/mL, added to the 2 L of extracting solution. For volatiles, these amounts are divided by four, due to the smaller sample size and extraction vessel. For purposes of the SPLP test, volatile organic chemicals are defined as contaminants with vapor pressures greater than 1 mm Hg at 25°C. These are identified in Table 2.

Rearrangement of Equation 16 yields the equation to predict field leachate concentrations from the total contaminant concentration in a soil sample:

$$C_L = \frac{C_T}{K_d + \frac{\theta_w + \theta_a H'}{\rho_b}} \qquad \textbf{Equation 17}$$

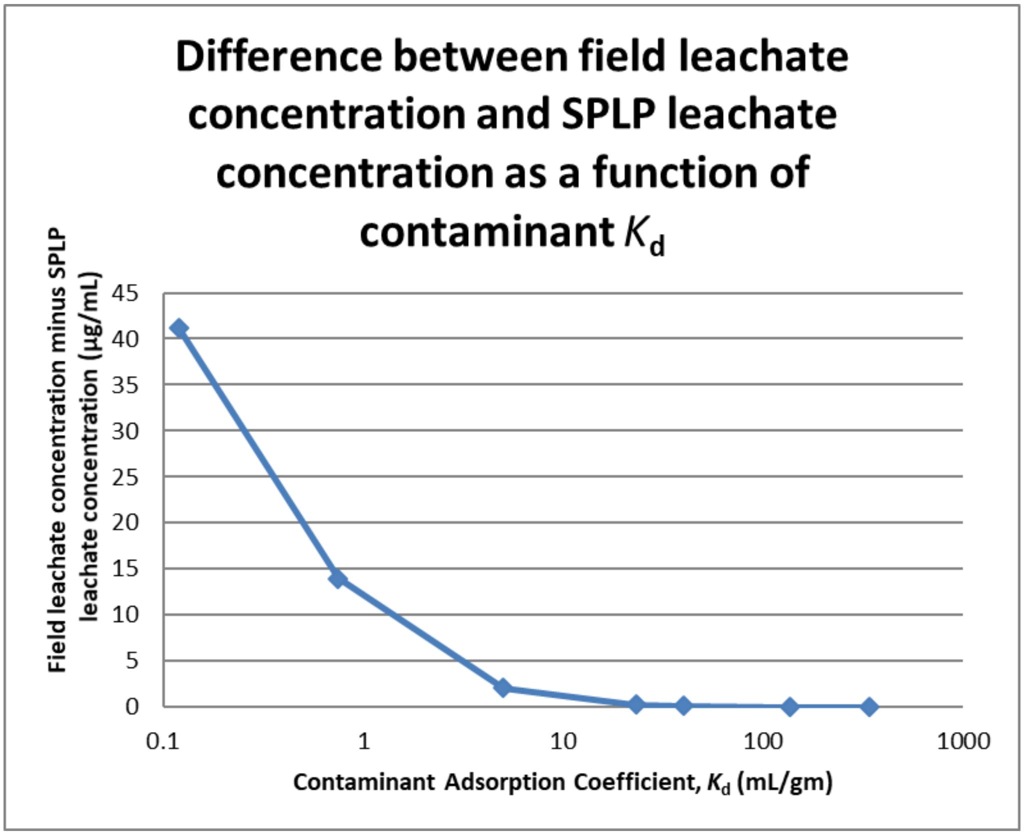
Since this equation is derived from the soil-water partition coefficient and the Henry's law constant, the assumptions governing those parameters apply to the use of this equation. Note that as $K_d$ becomes large relative to the other term in the denominator, $C_L$ becomes <u>independent</u> of the relative amounts of water, soil, and air, and $C_L$ becomes <u>constant</u>. Therefore, for chemicals with a high $K_d$ values, leachate concentrations measured under SPLP and field conditions will be similar. In contrast, chemicals with <u>low</u> $K_d$ values will exhibit leachate concentrations that are dependent on the amounts of water, soil and air. The table below illustrates the behavior of this equation for selected contaminants with widely varying $K_d$ values. The table shows the relative amounts of soil, water and air in 1 mL of soil under environmental conditions in the field, and also the relative amounts of soil and water used in the SPLP test. To avoid confusion when looking at this table, volatile contaminants are assumed to use the same SPLP apparatus (and the same amounts of water and soil) as the nonvolatile contaminants. This is acceptable because it is the water-to-soil ratio that affect the results of these calculations, not the absolute amounts of water and soil. (The water-to-soil ratio is 20:1 for both SPLP apparatus.) The table then shows the mass of soil normalized to 1 gram for both environmental and SPLP conditions, and the corresponding proportional amounts of water and air. For both scenarios, the total mass of contaminant is held constant at 13.33 µg. Looking first at the results for trichloroethene, a contaminant with a very low $K_d$ value, the difference between the leachate concentration under SPLP conditions and environmental conditions in the field is quite large. The leachate concentration under SPLP conditions does not reflect the much higher leachate concentration that would be observed in the field. This is because the soil-sorbed concentration after equilibrium under SPLP conditions is only a small fraction of the initial total soil concentration, which requires a lower leachate concentration to maintain the proper equilibrium ratio. In contrast, DDT exhibits virtually the same leachate concentration under both conditions because of its very high $K_d$ contaminant. The soil-sorbed concentration essentially remains at the initial total soil concentration, because very little desorbs. In this case, the SPLP leachate concentration could be used directly to approximate field conditions.

The figure below illustrates that when the $K_d$ becomes less than about 25-50 mL/g, the SPLP leachate concentration cannot be used directly to estimate the field leachate concentration. In this case, the $K_d$ value, along with the total soil concentration, $C_T$, and the field values for soil moisture and soil vapor volume, should be used in Equation 17 to estimate the leachate concentration for the sample under field conditions. Since Equation 17 is the technically complete calculation, it is used for all chemicals when evaluating SPLP test results.

To do this, first a sample-specific $K_d$ value is determined, using the SPLP test results. Since the $K_d$ is a constant, it can be measured using the SPLP test and then used to predict leachate concentrations under field conditions. The SPLP leachate ($C_{SPLP}$) concentration and a modified version of Equation 10 is used, since the SPLP test does not contain an air phase:

$$K_d = \frac{(C_T M_S - C_{SPLP} V_L)/M_S}{C_{SPLP}} \qquad \textbf{Equation 18}$$

The volume of the extracting solution and the mass of soil in the SPLP test, the total soil concentration, and the SPLP leachate concentration are used in Equation 18 to determine the sample-specific $K_d$. This $K_d$ is then used in Equation 17 along with $C_T$, $H'$, and the field values of $\theta_w$, $\theta_a$ and $\rho_b$ to calculate the leachate concentration under field conditions.


Difference between field leachate concentration and SPLP leachate concentration as a function of contaminant $K_d$
Field leachate concentration minus SPLP leachate concentration (µg/mL)
45
40
35
30
25
20
15
10
5
0
0.1
1
10
100
1000
Contaminant Adsorption Coefficient, $K_d$ (mL/gm)

## Leachate concentrations under SPLP and field conditions for various contaminants

| *Representative contaminant* | $K_d$ *Soil adsorption coefficient with default fraction organic carbon content of 0.002 (mL/g)* | $H'$ *Henry's law constant, dimensionless, 25°C* | *Conditions* | *Liquid to air to solid ratio (mL water/mL air/g soil)* | $\theta_w$ *Volume of water (mL)* | $\rho_b$ *Mass of soil (g)* | $\theta_a$ *Volume of air (mL)* | *Total mass in sample (µg)* | $C_{total}$ *Total (initial) soil concentration (µg/g)* | *Equilibrium soil sorbed concentration (µg/g)* | $C_L$ *Equilibrium leachate concentration (µg/mL)* | *% of total mass in leachate at equilibrium* | *Difference between field leachate and SPLP concentration (µg/mL)* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| TCE | 0.12 | 0.403 | Field | 0.23/0.18/1.5 | 0.15 | 1 | 0.12 | 13.33 | 13.33 | 5.02 | 41.87 | 47 | |
| | | | SPLP | 2000/0/100 | 20 | 1 | 0 | 13.33 | 13.33 | 0.08 | 0.66 | 99 | 41.21 |
| 1,4 Dichlorobenzene | 0.75 | 0.0985 | Field | 0.23/0.18/1.5 | 0.15 | 1 | 0.12 | 13.33 | 13.33 | 10.96 | 14.62 | 16 | |
| | | | SPLP | 2000/0/100 | 20 | 1 | 0 | 13.33 | 13.33 | 0.48 | 0.64 | 96 | 13.98 |
| 2-Methylnaphthalene | 5 | 0.0212 | Field | 0.23/0.18/1.5 | 0.15 | 1 | 0.12 | 13.33 | 13.33 | 12.94 | 2.59 | 2.9 | |
| | | | SPLP | 2000/0/100 | 20 | 1 | 0 | 13.33 | 13.33 | 2.67 | 0.53 | 80.0 | 2.05 |
| Cadmium | 23 | 0 | Field | 0.23/0.18/1.5 | 0.15 | 1 | 0.12 | 13.33 | 13.33 | 13.24 | 0.58 | 0.6 | |
| | | | SPLP | 2000/0/100 | 20 | 1 | 0 | 13.33 | 13.33 | 7.13 | 0.31 | 46.5 | 0.27 |
| Dieldrin | 40 | 4.09E-04 | Field | 0.23/0.18/1.5 | 0.15 | 1 | 0.12 | 13.33 | 13.33 | 13.28 | 0.33 | 0.4 | |
| | | | SPLP | 2000/0/100 | 20 | 1 | 0 | 13.33 | 13.33 | 8.89 | 0.22 | 33.3 | 0.11 |
| Chlordane | 135 | 1.99E-03 | Field | 0.23/0.18/1.5 | 0.15 | 1 | 0.12 | 13.33 | 13.33 | 13.32 | 0.099 | 0.11 | |
| | | | SPLP | 2000/0/100 | 20 | 1 | 0 | 13.33 | 13.33 | 11.61 | 0.086 | 12.90 | 0.013 |
| DDT | 337 | 3.40E-04 | Field | 0.23/0.18/1.5 | 0.15 | 1 | 0.12 | 13.33 | 13.33 | 13.32 | 0.03954 | 0.044 | |
| | | | SPLP | 2000/0/100 | 20 | 1 | 0 | 13.33 | 13.33 | 12.58 | 0.03734 | 5.602 | 0.00220 |

## 3.4. Development of Soil Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway (ARS-MGW) using the SESOIL Model

The Seasonal Soil Compartment Model (SESOIL) was originally formulated by Arthur D. Little and Associates for the USEPA (Bonazountas and Wagner, 1984). Since that time it has been revised and updated several times (Bonazountas et al., 1997). The model now has nearly a 20-year history and has been subject to several laboratory and field validation studies (Bonazountas et al., 1997; Melancon et al., 1986; Sanders, 1995).

The SESOIL model has become fairly well established and has been accepted by several state agencies and the USEPA for calculating remediation standards. Some of the states routinely using SESOIL exclusively or SESOIL with AT123D (a ground water transport model) are Oregon, New Hampshire, Wisconsin, Massachusetts and Hawaii. The models may be useful for site-specific remediation standard calculations when 1) there is a clean zone of soil between the contamination and the water table, or 2) when ground water plume modeling is desired.

The SESOIL model accounts for the contaminant processes of advection, volatilization, degradation, and surface runoff (Bonazountas and Wagner, 1984, Bonazountas et al., 1997). Precipitation is generated using a statistical formula that incorporates monthly New Jersey climate data. Water transport is calculated via the statistical water balance dynamics theory of Eagleson (1978). The model includes the mechanisms of runoff, infiltration, evapotranspiration, and ground water recharge. The capillary fringe effect is also included. Contaminant transport downward is calculated via advection using the retardation factor. The factor allows calculation of the distance of contaminant transport during each time step. The soil column is considered to be layered into several compartments. Contaminant entering a soil compartment is considered to be immediately mixed through the entire compartment. Vapor phase transport is also modeled (upward direction only) to allow calculation of contaminant volatilization. Various options for contaminant degradation are also included. However, with one exception, the Department does not allow for contaminant degradation. The exception applies to benzene, toluene, ethylbenzenes and xylene, and other hydrocarbons which may degrade rapidly in the vadose zone under certain conditions (DeVaull et al., 2002; Howard et al., 1991). Biodegradation is a highly variable, highly site-specific parameter that is not amenable to routine determination in the unsaturated soil zone without research-level investigation. However, volatile hydrocarbons generally exhibit fairly rapid degradation in the soil. Based on a review of reported unsaturated soil zone degradation rates reported by Devaull et al. (2002), it was observed that for volatile hydrocarbons, half-lives were generally one month or less. Picking contaminant-specific half-lives for these chemicals implies a more accurate value for each contaminant than actually exists without site-specific investigation. A one-month half life (rate constant of 0.023 days$^{-1}$) was judged to be adequately protective for these contaminants while still providing rapid attenuation for volatile hydrocarbons in the unsaturated soil zone. For these contaminants, a default half-life of 1 month may be used in the SESOIL model.

The Seasonal Soil Compartment Model (SESOIL) may be used to demonstrate that a specified existing or proposed concentration distribution of contaminant in soil will not result in future contamination of ground water above the GWRS. This contaminant concentration distribution may then be used to define a soil alternative remediation standard for the migration to ground water exposure pathway.

### 3.5. Development of Alternative Remediation Standards for the Migration to Ground Water Exposure Pathway using the Combined SESOIL/AT123D Model

The Analytical Transient 1-, 2- and 3-Dimensional Ground Water model (AT123D) is an analytical one, two or three-dimensional ground water contaminant transport model originally developed by Yeh (1981). It computes the spatial-temporal concentration distribution of contaminants in the aquifer system and predicts the transient spread of a contaminant plume through a ground water aquifer. It accounts for 1) advection of the chemical with the water flowing through the aquifer, 2) dispersion of the chemical via hydrodynamic dispersion and molecular diffusion, 3) adsorption of contaminant to aquifer solids, and 4) contaminant decay.

When ground water is already impacted and a ground water Classification Exception Area (CEA) is in place, the combined Seasonal Soil Compartment Model/Analytical Transient 1-, 2- and 3-Dimensional Ground Water (SESOIL/AT123D) model may be used to demonstrate that a specific vertical contaminant concentration distribution in the unsaturated zone will not result in an increasing ground water plume size or unacceptable ground water impacts beyond the time frame specified in the CEA. This contaminant concentration distribution may then be used to define a soil alternative remediation standard for the migration to ground water exposure pathway.

The default one-month half-life for degradable hydrocarbons is allowed in the unsaturated zone (SESOIL model), but not allowed in the saturated zone (AT123D model). To use degradation rates in the unsaturated zone, alternative procedures from the New Jersey Bureau of Ground Water Pollution Abatement must be used. Consult the "Alternative Remediation Standards Technical Guidance for Soil and Soil Leachate for the Migration to Ground Water Exposure Pathway" for further details.

### 3.6. Immobile Chemicals

Scientific evidence suggests that chemicals become more resistant to desorption from soil as contact time increases (Loehr and Webster, 1996; Alexander, 1995; Pavlostathis and Mathavan, 1992). Highly adsorbed chemicals, such as chlorinated pesticides, may become irreversibly adsorbed to soil and therefore immobile (Alexander, 1995). This behavior would be added to the already low transport potential of these contaminants (due to their high soil adsorption coefficients). Therefore, it can be assumed that these chemicals do not pose a threat to ground water if an adequate zone of clean soil exists between the contamination and the ground water.

For purposes of the New Jersey soil remediation standards for the migration to ground water exposure pathway (SRS-MGW), a chemical is classified as an immobile chemical if a 2-foot clean zone between the contamination and the water table is an adequate separation distance to prevent contaminant transport to the water table over a 100-year time period. To simulate contaminant transport, the Department used the Seasonal Soil Compartment Model (SESOIL). In the previous version of the immobile chemical guidance (2008), it was determined that when a chemical had a soil organic carbon-water partition coefficient ($K_{oc}$) of 50,000 L/kg or greater (or a soil-water partition coefficient ($K_d$) of 100 L/kg or greater), the vertical transport distance over 100 years was 11 inches or less. Since that assessment was conducted, average annual rainfall amounts have increased in New Jersey, and the weather database in the SESOIL model has been updated. Furthermore, contaminant transport distance in the SESOIL model is more complex than originally assumed in 2008, being affected by the depth to the water table and the depth of the contaminant below the soil surface. Therefore, updated SESOIL model runs were conducted on a model chemical with a $K_{oc}$ value of 50,000 L/kg.

The model was run using the default environmental conditions and soil properties used for calculation of generic soil cleanup standards (see documentation on the inhalation pathway of the soil standards regulations). This scenario assumes a sandy loam soil. Sandy loam soil properties were entered using recommended values from the model authors. In addition, sand soil runs were conducted, since this soil is more porous to water infiltration and ground water recharge than sandy loam soil. The soil was assumed to be homogeneous. The organic carbon content was set to the generic value of 0.2% (w/w), and the Freundlich exponent was set to one. Two central New Jersey weather stations were selected (Trenton, NJ and Hightstown, NJ) as intermediate locations between northern and southern New Jersey. The vadose zone was run at depths ranging from 1 foot in thickness to 20 feet in thickness. 1-foot soil sublayers were used. Chemical was applied to depths ranging from 1 foot to 8 feet. The Henry's law constant and the diffusion coefficients were set to 1E-6. This was done because this minimizes contaminant volatilization. The chemicals of concern do not exhibit significant vapor phase behavior, so volatilization was not modeled in these simulations. Other input parameter values relevant to contaminant transport potential in the soil column are given in the table below:

| **SESOIL Modeling Parameters** | |
|---|---|
| *Parameter* | *Value* |
| Groundwater Depth (ft) | 1 to 20 feet |
| Bulk density of soil (g/cc) | 1.5 |
| Number of soil layers | 2 |
| Soil sublayer thickness (ft) | 1 |
| Intrinsic permeability ($cm^2$) | 2E-9 (sandy loam); 1E-8 (sand) |
| Effective porosity (v/v) | 0.25 (sandy loam); 0.30 (sand) |
| Disconnectedness index | 4 (sandy loam); 3.7 (sand) |
| Time increment (days) | 1 |
| Length of run (years) | 100 |
| Climate station | Trenton, NJ or Hightstown, NJ |
| Organic carbon content | 0.20% |
| Freundlich exponent | 1 |

Simulations were conducted for a 100-year time period. This time period was selected because 1) it is near the upper limit of human life expectancy and 2) transport is not likely after this length of time because these chemicals would become irreversibly adsorbed. (The time period for reduced desorption to occur has been reported to be on the order of weeks or months for several chemicals (Loehr and Webster, 1996; Alexander, 1995; Pavlostathis and Mathavan, 1992)). The distance that the model contaminant was predicted to move over a 100-year time period was tabulated for various model runs, as tabulated below.

| Soil texture | Weather Station | Depth to Ground Water (feet) | Contaminant Application Depth (feet) | Vertical Transport distance (inches) |
|---|---|---|---|---|
| Sandy loam | Hightstown | 10 | 8 | 17 |
| Sandy loam | Hightstown | 10 | 4 | 19 |
| Sandy loam | Hightstown | 10 | 1 | 26 |
| Sand | Hightstown | 10 | 8 | 22 |
| Sand | Hightstown | 10 | 4 | 23 |
| Sand | Hightstown | 10 | 1 | 28 |
| Sand | Hightstown | 20 | 4 | 27 |
| Sandy loam | Trenton | 10 | 8 | 18 |
| Sandy loam | Trenton | 10 | 4 | 20 |
| Sandy loam | Trenton | 10 | 1 | 27 |
| Sandy loam | Trenton | 20 | 8 | 26 |
| Sandy loam | Trenton | 20 | 4 | 27 |
| Sandy loam | Trenton | 20 | 1 | 30 |
| Sandy loam | Trenton | 5 | 2 | 22 |
| Sandy loam | Trenton | 5 | 1 | 28 |

The vertical transport distance is affected not only by relevant chemical, environmental, and soil properties, but also by the depth of the water table and the depth of the contaminant application below the soil surface. Contaminant transport distance was sometimes greater than two feet (but no more than 2.5 feet) when it was only applied to the top foot of the vadose zone, or when the depth to the ground water was large. However, this only occurred for larger separation distances between the contaminant and the water table, and contaminant never reached the water table in these simulations. The results indicate that as long as there is a minimum separation distance of two feet between the contamination and the water table, the contaminant does not reach the water table in a 100-year simulation period. Therefore, the conditions to qualify for an immobile chemical ($K_{oc}$ value of 50,000 L/kg or a $K_d$ value of 100 L/kg) remain unchanged from those specified in 2008. The eligible chemicals are listed below.

**List of Immobile Chemicals**

Lead
Aldrin
Benzo(a)anthracene
Bis(2-ethylhexyl phthalate)
Chlordane
DDD
DDE
DDT
PCBs
2,3,7,8 -Tetrachlorodibenzo-p-dioxin
Toxaphene

For these contaminants, if the investigator can demonstrate that a minimum two-foot clean zone is present between the contamination and the water table, no remediation may be required for the MGW pathway. Some chemicals have been eliminated from the list provided with the 2008 guidance document because they are no longer regulated or have updated $K_{oc}$ or $K_d$ values that are now lower than the cutoff values stated above.

## 3.7. Site Soil and Ground Water Data Evaluation

In situations where the highest concentrations of a contaminant are located at the water table, and where ground water monitoring demonstrates that the GWRS are not exceeded, the Department considers the migration to ground water exposure pathway satisfactorily addressed. See the Alternative Remediation Standards Technical Guidance for Soil and Soil Leachate for the Migration to Ground Water Exposure Pathway for further details.

# TABLES

## Table 1

## SOIL REMEDIATION STANDARDS - MIGRATION TO GROUND WATER EXPOSURE PATHWAY (SRS-MGW)

| Contaminant | CAS No. | Ground Water Remediation Standard (µg/L) | Soil Remediation Criterion Migration to Ground Water (mg/kg) | Soil Saturation Limit ($C_{sat}$) (mg/kg) | Soil Reporting Limit (RL) (mg/kg) | Migration to Ground Water Soil Remediation Standard (SRS-MGW) (mg/kg) |
|---|---|---|---|---|---|---|
| Acenaphthene | 83-32-9 | 400 | 82 | 40 | 0.17 | NA[1] |
| Acetone (2-Propanone) | 67-64-1 | 6,000 | 19 | 160,000 | 0.010 | 19 |
| Acetophenone | 98-86-2 | 700 | 3.6 | 1,600 | 0.33 | 3.6 |
| Aldrin | 309-00-2 | 0.04 | 0.13 | 2.8 | 0.0017 | 0.13 |
| Aluminum (total) | 7429-90-5 | NA | NA | NA | 20 | NA[2] |
| Anthracene | 120-12-7 | 2,000 | 1,300 | 1.4 | 0.17 | NA[1] |
| Antimony (total) | 7440-36-0 | 6 | 5.4 | NA | 1.0 | 5.4 |
| Arsenic (total) | 7440-38-2 | 3 | 1.6 | NA | 0.50 | 19[3] |
| Atrazine | 1912-24-9 | 3 | 0.036 | 21 | 0.33 | 0.33[4] |
| Barium (total) | 7440-39-3 | 6,000 | 2,100 | NA | 5.0 | 2,100 |
| Benzaldehyde | 100-52-7 | NA | NA | 1,200 | 0.33 | NA[5] |
| Benzene | 71-43-2 | 1 | 0.0094 | 850 | 0.0050 | 0.0094 |
| Benzo(a)anthracene (1,2-Benzanthracene) | 56-55-3 | 0.1 | 0.71 | 3.3 | 0.17 | 0.71 |
| Benzo(a)pyrene | 50-32-8 | 0.1 | 2.3 | 1.9 | 0.17 | NA[1] |
| Benzo(b)fluoranthene (3,4-Benzofluoranthene) | 205-99-2 | 0.2 | 4.8 | 1.8 | 0.17 | NA[1] |
| Benzo(k)fluoranthene | 207-08-9 | 0.5 | 12 | 0.94 | 0.17 | NA[1] |
| Beryllium | 7440-41-7 | 1 | 0.70 | NA | 0.50 | 0.70 |
| 1,1'-Biphenyl | 92-52-4 | 400 | 83 | 78 | 0.17 | NA[1] |
| Bis(2-chloroethoxy)methane | 111-91-1 | NA | NA | 1,400 | 0.17 | NA[5] |
| Bis(2-chloroethyl)ether | 111-44-4 | 7 | 0.030 | 3,700 | 0.33 | 0.33[4] |
| Bis(2-ethylhexyl)phthalate | 117-81-7 | 3 | 14 | 65 | 0.17 | 14 |
| Bromodichloromethane (Dichlorobromomethane) | 75-27-4 | 1 | 0.0045 | 690 | 0.0050 | 0.0050[4] |
| Bromoform | 75-25-2 | 4 | 0.018 | 680 | 0.0050 | 0.018 |
| Bromomethane (Methyl bromide) | 74-83-9 | 10 | 0.043 | 3,300 | 0.0050 | 0.043 |
| 2-Butanone (Methyl ethyl ketone) (MEK) | 78-93-3 | 300 | 0.98 | 36,000 | 0.010 | 0.98 |
| Butylbenzyl phthalate | 85-68-7 | 100 | 29 | 39 | 0.17 | 29 |
| Cadmium | 7440-43-9 | 4 | 1.9 | NA | 0.50 | 1.9 |

| Contaminant | CAS No. | Ground Water Remediation Standard (µg/L) | Soil Remediation Criterion Migration to Ground Water (mg/kg) | Soil Saturation Limit ($C_{sat}$) (mg/kg) | Soil Reporting Limit (RL) (mg/kg) | Migration to Ground Water Soil Remediation Standard (SRS-MGW) (mg/kg) |
|---|---|---|---|---|---|---|
| Caprolactam | 105-60-2 | 4,000 | 16 | 160,000 | 0.33 | 16 |
| Carbon disulfide | 75-15-0 | 700 | 3.7 | 580 | 0.0050 | 3.7 |
| Carbon tetrachloride | 56-23-5 | 1 | 0.0075 | 300 | 0.0050 | 0.0075 |
| Chlordane (alpha and gamma forms summed) | 57-74-9 | 0.5 | 1.4 | 7.6 | 0.0017 | 1.4 |
| 4-Chloroaniline | 106-47-8 | 30 | 0.23 | 1,500 | 0.17 | 0.23 |
| Chlorobenzene | 108-90-7 | 50 | 0.64 | 320 | 0.0050 | 0.64 |
| Chloroethane (Ethyl chloride) | 75-00-3 | NA | NA | 1,700 | 0.0050 | NA[5] |
| Chloroform | 67-66-3 | 70 | 0.33 | 1,900 | 0.0050 | 0.33 |
| Chloromethane (Methyl chloride) | 74-87-3 | NA | NA | 1,200 | 0.0050 | NA[5] |
| 2-Chloronaphthalene | 91-58-7 | 600 | 61 | 60 | 0.17 | NA[1] |
| 2-Chlorophenol (o-chlorophenol) | 95-57-8 | 40 | 0.76 | 11,000 | 0.17 | 0.76 |
| Chrysene | 218-01-9 | 5 | 36 | 0.72 | 0.17 | NA[1] |
| Cobalt (total) | 7440-48-4 | 100 | 90 | NA | 0.50 | 90 |
| Copper (total) | 7440-50-8 | 1,300 | 910 | NA | 1.0 | 910 |
| Cyanide | 57-12-5 | 100 | 20 | NA | 0.50 | 20 |
| Cyclohexane | 110-82-7 | NA | NA | 65 | 0.0050 | NA[5] |
| 4,4'-DDD (p,p'-TDE) | 72-54-8 | 0.1 | 0.47 | 21 | 0.0033 | 0.47 |
| 4,4'-DDE (p,p'-DDX) | 72-55-9 | 0.1 | 0.47 | 9.4 | 0.0033 | 0.47 |
| 4,4'-DDT | 50-29-3 | 0.1 | 0.67 | 1.9 | 0.0033 | 0.67 |
| Dibenz(a,h)anthracene | 53-70-3 | 0.3 | 23 | 9.5 | 0.17 | NA[1] |
| Dibromochloromethane (Chlorodibromomethane) | 124-48-1 | 1 | 0.0044 | 600 | 0.0050 | 0.0050[4] |
| 1,2-Dibromo-3-chloropropane | 96-12-8 | 0.02 | 0.00015 | 470 | 0.0050 | 0.0050[4] |
| 1,2-Dibromoethane (Ethylene dibromide) | 106-93-4 | 0.03 | 0.00014 | 920 | 0.0050 | 0.0050[4] |
| 1,2-Dichlorobenzene (o-Dichlorobenzene) | 95-50-1 | 600 | 11 | 140 | 0.0050 | 11 |
| 1,3-Dichlorobenzene (m-Dichlorobenzene) | 541-73-1 | 600 | 11 | 110 | 0.0050 | 11 |
| 1,4-Dichlorobenzene (p-Dichlorobenzene) | 106-46-7 | 75 | 1.4 | 74 | 0.0050 | 1.4 |
| 3,3'-Dichlorobenzidine | 91-94-1 | 30 | 3.9 | 20 | 0.33 | 3.9 |
| Dichlorodifluoromethane (Freon 12) | 75-71-8 | 1,000 | 38 | 540 | 0.0050 | 38 |

| Contaminant | CAS No. | Ground Water Remediation Standard (µg/L) | Soil Remediation Criterion Migration to Ground Water (mg/kg) | Soil Saturation Limit ($C_{sat}$) (mg/kg) | Soil Reporting Limit (RL) (mg/kg) | Migration to Ground Water Soil Remediation Standard (SRS-MGW) (mg/kg) |
|---|---|---|---|---|---|---|
| 1,1-Dichloroethane | 75-34-3 | 50 | 0.24 | 1,200 | 0.0050 | 0.24 |
| 1,2-Dichloroethane | 107-06-2 | 2 | 0.0095 | 2,000 | 0.0050 | 0.0095 |
| 1,1-Dichloroethene (1,1-Dichloroethylene) | 75-35-4 | 1 | 0.0069 | 830 | 0.0050 | 0.0069 |
| 1,2-Dichloroethene (cis) (c-1,2-Dichloroethylene) | 156-59-2 | 70 | 0.35 | 1,600 | 0.0050 | 0.35 |
| 1,2-Dichloroethene (trans) (t-1,2-Dichloroethylene) | 156-60-5 | 100 | 0.56 | 1,300 | 0.0050 | 0.56 |
| 2,4-Dichlorophenol | 120-83-2 | 20 | 0.19 | 2,600 | 0.17 | 0.19 |
| 1,2-Dichloropropane | 78-87-5 | 1 | 0.0058 | 810 | 0.0050 | 0.0058 |
| 1,3-Dichloropropene (total) | 542-75-6 | 1 | 0.0063 | 880 | 0.0050 | 0.0063 |
| Dieldrin | 60-57-1 | 0.03 | 0.024 | 7.9 | 0.0033 | 0.024 |
| Diethylphthalate | 84-66-2 | 6,000 | 44 | 390 | 0.17 | 44 |
| 2,4-Dimethylphenol | 105-67-9 | 100 | 2.3 | 8,900 | 0.17 | 2.3 |
| Di-n-butyl phthalate | 84-74-2 | 700 | 35 | 28 | 0.17 | NA[1] |
| 2,4-Dinitrophenol | 51-28-5 | 40 | 0.12 | 430 | 0.33 | 0.33[4] |
| 2,4-Dinitrotoluene/2,6-Dinitrotoluene (mixture) | 25321-14-6 | 10 | 0.27 | 360 | 0.17 | 0.27 |
| Di-n-octyl phthalate | 117-84-0 | 100 | 560 | 6.2 | 0.33 | NA[1] |
| 1,4-Dioxane | 123-91-1 | 0.4 | 0.0013 | 160,000 | 0.067 | 0.067[4] |
| Endosulfan I and Endosulfan II (alpha and beta) (summed) | 115-29-7 | 40 | 11 | 4.4 | 0.0033 | NA[1] |
| Endrin | 72-20-8 | 2 | 1.6 | 10 | 0.0033 | 1.6 |
| Ethylbenzene | 100-41-4 | 700 | 15 | 180 | 0.0050 | 15 |
| Extractable Petroleum Hydrocarbons (Category 1) | various | NA | NA | NA | 80 | NA[5] |
| Extractable Petroleum Hydrocarbons (Category 2) | various | NA | NA | NA | 80 | NA[5] |
| Fluoranthene | 206-44-0 | 300 | 670 | 29 | 0.33 | NA[1] |
| Fluorene | 86-73-7 | 300 | 110 | 31 | 0.17 | NA[1] |
| alpha-HCH (alpha-BHC) | 319-84-6 | 0.02 | 0.0023 | 12 | 0.0017 | 0.0023 |
| beta-HCH (beta-BHC) | 319-85-7 | 0.04 | 0.0046 | 1.4 | 0.0017 | 0.0046 |
| Heptachlor | 76-44-8 | 0.05 | 0.083 | 15 | 0.0017 | 0.083 |
| Heptachlor epoxide | 1024-57-3 | 0.2 | 0.081 | 4.1 | 0.0017 | 0.081 |
| Hexachlorobenzene | 118-74-1 | 0.02 | 0.0050 | 0.078 | 0.17 | 0.17[4] |
| Hexachloro-1,3-butadiene | 87-68-3 | 1 | 0.038 | 6.1 | 0.17 | 0.17[4] |
| Hexachlorocyclopentadiene | 77-47-4 | 40 | 2.5 | 5.6 | 0.33 | 2.5 |

| Contaminant | CAS No. | Ground Water Remediation Standard (µg/L) | Soil Remediation Criterion Migration to Ground Water (mg/kg) | Soil Saturation Limit ($C_{sat}$) (mg/kg) | Soil Reporting Limit (RL) (mg/kg) | Migration to Ground Water Soil Remediation Standard (SRS-MGW) (mg/kg) |
|---|---|---|---|---|---|---|
| Hexachloroethane | 67-72-1 | 7 | 0.079 | 28 | 0.17 | 0.17[4] |
| n-Hexane | 110-54-3 | 30 | 5.5 | 88 | NA | 5.5 |
| 2-Hexanone | 591-78-6 | 40 | 0.15 | 3,200 | 0.010 | 0.15 |
| Indeno(1,2,3-cd)pyrene | 193-39-5 | 0.2 | 16 | 0.74 | 0.17 | NA[1] |
| Isophorone | 78-59-1 | 40 | 0.23 | 3,400 | 0.17 | 0.23 |
| Isopropylbenzene | 98-82-8 | 700 | 22 | 98 | 0.0050 | 22 |
| Lead (total) | 7439-92-1 | 5 | 90 | NA | 0.50 | 90 |
| Lindane (gamma-HCH)(gamma-BHC) | 58-89-9 | 0.03 | 0.0035 | 42 | 0.0017 | 0.0035 |
| Manganese (total) | 7439-96-5 | NA | NA | NA | 0.50 | NA[2] |
| Mercury (total) | 7439-97-6 | 2 | 0.014 | NA | 0.10 | 0.10[4] |
| Methoxychlor | 72-43-5 | 40 | 43 | 5.4 | 0.017 | NA[1] |
| Methyl acetate | 79-20-9 | 7,000 | 22 | 39,000 | 0.0050 | 22 |
| Methylene chloride (Dichloromethane) | 75-09-2 | 3 | 0.013 | 2,800 | 0.0050 | 0.013 |
| 2-Methylnaphthalene | 91-57-6 | 30 | 3.1 | 130 | 0.17 | 3.1 |
| 4-Methyl-2-pentanone (MIBK) | 108-10-1 | NA | NA | 3,400 | 0.010 | NA[5] |
| 2-Methylphenol (o-cresol) | 95-48-7 | 50 | 0.77 | 20,000 | 0.33 | 0.77 |
| 4-Methylphenol (p-cresol) | 106-44-5 | 50 | 0.75 | 16,000 | 0.33 | 0.75 |
| Methyl tert-butyl ether (MTBE) | 1634-04-4 | 70 | 0.25 | 9,100 | 0.0050 | 0.25 |
| Naphthalene | 91-20-3 | 300 | 19 | 100 | 0.17 | 19 |
| Nickel (total) | 7440-02-0 | 100 | 48 | NA | 0.50 | 48 |
| 4-Nitroaniline | 100-01-6 | NA | NA | 270 | 0.33 | NA[5] |
| Nitrobenzene | 98-95-3 | 6 | 0.073 | 1,300 | 0.17 | 0.17[4] |
| N-Nitrosodi-n-propylamine | 621-64-7 | 10 | 0.14 | 9,200 | 0.17 | 0.17[4] |
| N-Nitrosodiphenylamine | 86-30-6 | 10 | 1.1 | 190 | 0.17 | 1.1 |
| 2,2'-oxybis(1-chloropropane) | 108-60-1 | 300 | 1.9 | 540 | 0.33 | 1.9 |
| Pentachlorophenol | 87-86-5 | 0.3 | 0.062 | 140 | 0.33 | 0.33[4] |
| Phenol | 108-95-2 | 2,000 | 21 | 44,000 | 0.33 | 21 |
| Polychlorinated biphenyls (PCBs) | 1336-36-3 | 0.5 | 1.6 | 110 | 0.030 | 1.6 |
| Pyrene | 129-00-0 | 200 | 440 | 15 | 0.17 | NA[1] |
| Selenium (total) | 7782-49-2 | 40 | 11 | NA | 2.5 | 11 |
| Silver (total) | 7440-22-4 | 40 | 0.33 | NA | 0.50 | 0.50[4] |
| Styrene | 100-42-5 | 100 | 2.1 | 330 | 0.0050 | 2.1 |

| Contaminant | CAS No. | Ground Water Remediation Standard (µg/L) | Soil Remediation Criterion Migration to Ground Water (mg/kg) | Soil Saturation Limit ($C_{sat}$) (mg/kg) | Soil Reporting Limit (RL) (mg/kg) | Migration to Ground Water Soil Remediation Standard (SRS-MGW) (mg/kg) |
|---|---|---|---|---|---|---|
| Tertiary butyl alcohol (TBA) | 75-65-0 | 100 | 0.32 | 160,000 | 0.10 | 0.32 |
| 1,2,4,5-Tetrachlorobenzene | 95-94-3 | NA | NA | 2.7 | 0.17 | NA[5] |
| 2,3,7,8-Tetrachlorodibenzo-p-dioxin | 1746-01-6 | 0.00001 | 0.00010 | 0.10 | 0.0000010 | 0.00010[6] |
| 1,1,2,2-Tetrachloroethane | 79-34-5 | 1 | 0.0069 | 980 | 0.0050 | 0.0069 |
| Tetrachloroethene (PCE) (Tetrachloroethylene) | 127-18-4 | 1 | 0.0086 | 89 | 0.0050 | 0.0086 |
| 2,3,4,6-Tetrachlorophenol | 58-90-2 | 200 | 26 | 150 | 0.17 | 26 |
| Toluene | 108-88-3 | 600 | 7.8 | 340 | 0.0050 | 7.8 |
| Toxaphene | 8001-35-2 | 2 | 6.2 | 85 | 0.17 | 6.2 |
| 1,2,4-Trichlorobenzene | 120-82-1 | 9 | 0.52 | 140 | 0.0050 | 0.52 |
| 1,1,1-Trichloroethane | 71-55-6 | 30 | 0.20 | 420 | 0.0050 | 0.20 |
| 1,1,2-Trichloroethane | 79-00-5 | 3 | 0.017 | 1,300 | 0.0050 | 0.017 |
| Trichloroethene (TCE) (Trichloroethylene) | 79-01-6 | 1 | 0.0065 | 410 | 0.0050 | 0.0065 |
| Trichlorofluoromethane (Freon 11) | 75-69-4 | 2,000 | 29 | 790 | 0.0050 | 29 |
| 2,4,5-Trichlorophenol | 95-95-4 | 700 | 68 | 5,800 | 0.20 | 68 |
| 2,4,6-Trichlorophenol | 88-06-2 | 20 | 0.86 | 1,700 | 0.20 | 0.86 |
| 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon TF) | 76-13-1 | 20,000 | 1,300 | 530 | 0.0050 | NA[1] |
| 1,2,4-Trimethylbenzene | 95-63-6 | NA | NA | 80 | 0.076 | NA[5] |
| Vanadium (total) | 7440-62-2 | NA | NA | NA | 2.5 | NA[5] |
| Vinyl chloride | 75-01-4 | 1 | 0.0067 | 2,900 | 0.0050 | 0.0067 |
| Xylenes (total) | 1330-20-7 | 1,000 | 19 | 100 | 0.0050 | 19 |
| Zinc (total) | 7440-66-6 | 2,000 | 930 | NA | 1.0 | 930 |

NA = Not available/not applicable
1 – Standard not applicable because criterion is above soil saturation limit
2 – Standard not applicable because Ground Water Remediation Standard is a secondary standard
3 – Standard is based on natural background
4 – Standard set to soil reporting limit
5 – Ground Water Remediation Standard not available
6 – This standard is used for comparison to site soil data that have been converted to sample-specific TCDD-TEQ values through application of the Toxicity Equivalence Factor Methodology (USEPA 2010) and using the WHO 2005 Mammalian Toxic Equivalency Factors (TEFs).

## Table 2

## SOIL LEACHATE REMEDIATION STANDARDS - MIGRATION TO GROUND WATER EXPOSURE PATHWAY ((MGW$_{LEACHATE}$)

| Contaminant | CAS No. | Ground Water Remediation Standard (GWRS)(µg/L) | Soil Leachate Remediation Standard - Migration to Ground Water (MGW$_{LEACHATE}$) (µg/L) |
|---|---|---|---|
| Acenaphthene | 83-32-9 | 400 | NA[1] |
| Acetone (2-Propanone)* | 67-64-1 | 6,000 | 120,000 |
| Acetophenone | 98-86-2 | 700 | 14,000 |
| Aldrin | 309-00-2 | 0.04 | 0.80 |
| Aluminum (total) | 7429-90-5 | NA | NA[2] |
| Anthracene | 120-12-7 | 2,000 | NA[1] |
| Antimony (total) | 7440-36-0 | 6 | 120 |
| Arsenic (total) | 7440-38-2 | 3 | 60 |
| Atrazine | 1912-24-9 | 3 | 60 |
| Barium (total) | 7440-39-3 | 6,000 | 120,000 |
| Benzaldehyde* | 100-52-7 | NA | NA[3] |
| Benzene* | 71-43-2 | 1 | 20 |
| Benzo(a)anthracene (1,2-Benzanthracene) | 56-55-3 | 0.1 | 2.0 |
| Benzo(a)pyrene | 50-32-8 | 0.1 | NA[1] |
| Benzo(b)fluoranthene (3,4-Benzofluoranthene) | 205-99-2 | 0.2 | NA[1] |
| Benzo(k)fluoranthene | 207-08-9 | 0.5 | NA[1] |
| Beryllium | 7440-41-7 | 1 | 20 |
| 1,1'-Biphenyl | 92-52-4 | 400 | NA[1] |
| Bis(2-chloroethoxy)methane | 111-91-1 | NA | NA[3] |
| Bis(2-chloroethyl)ether* | 111-44-4 | 7 | 140 |
| Bis(2-ethylhexyl)phthalate | 117-81-7 | 3 | 60 |
| Bromodichloromethane (Dichlorobromomethane)* | 75-27-4 | 1 | 20 |
| Bromoform* | 75-25-2 | 4 | 80 |
| Bromomethane (Methyl bromide)* | 74-83-9 | 10 | 200 |
| 2-Butanone (Methyl ethyl ketone) (MEK)* | 78-93-3 | 300 | 6,000 |
| Butylbenzyl phthalate | 85-68-7 | 100 | 2,000 |
| Cadmium | 7440-43-9 | 4 | 80 |
| Caprolactam | 105-60-2 | 4,000 | 80,000 |
| Carbon disulfide* | 75-15-0 | 700 | 14,000 |
| Carbon tetrachloride* | 56-23-5 | 1 | 20 |

| Contaminant | CAS No. | Ground Water Remediation Standard (GWRS)(µg/L) | Soil Leachate Remediation Standard - Migration to Ground Water (MGWLEACHATE) (µg/L) |
|---|---|---|---|
| Chlordane (alpha and gamma forms summed) | 57-74-9 | 0.5 | 10 |
| 4-Chloroaniline | 106-47-8 | 30 | 600 |
| Chlorobenzene* | 108-90-7 | 50 | 1,000 |
| Chloroethane (Ethyl chloride)* | 75-00-3 | NA | NA[3] |
| Chloroform* | 67-66-3 | 70 | 1,400 |
| Chloromethane (Methyl chloride)* | 74-87-3 | NA | NA[3] |
| 2-Chloronaphthalene | 91-58-7 | 600 | NA[1] |
| 2-Chlorophenol (o-Chlorophenol)* | 95-57-8 | 40 | 800 |
| Chrysene | 218-01-9 | 5 | NA[1] |
| Cobalt (total) | 7440-48-4 | 100 | 2,000 |
| Copper (total) | 7440-50-8 | 1,300 | 26,000 |
| Cyanide | 57-12-5 | 100 | 2,000 |
| Cyclohexane* | 110-82-7 | NA | NA[3] |
| 4,4'-DDD (p,p'-TDE) | 72-54-8 | 0.1 | 2.0 |
| 4,4'-DDE (p,p'-DDX) | 72-55-9 | 0.1 | 2.0 |
| 4,4'-DDT | 50-29-3 | 0.1 | 2.0 |
| Dibenz(a,h)anthracene | 53-70-3 | 0.3 | NA[1] |
| Dibromochloromethane (Chlorodibromomethane)* | 124-48-1 | 1 | 20 |
| 1,2-Dibromo-3-chloropropane | 96-12-8 | 0.02 | 0.40 |
| 1,2-Dibromoethane (Ethylene dibromide)* | 106-93-4 | 0.03 | 0.60 |
| 1,2-Dichlorobenzene (o-Dichlorobenzene)* | 95-50-1 | 600 | 12,000 |
| 1,3-Dichlorobenzene (m-Dichlorobenzene)* | 541-73-1 | 600 | 12,000 |
| 1,4-Dichlorobenzene (p-Dichlorobenzene)* | 106-46-7 | 75 | 1,500 |
| 3,3'-Dichlorobenzidine | 91-94-1 | 30 | 600 |
| Dichlorodifluoromethane (Freon 12)* | 75-71-8 | 1,000 | 20,000 |
| 1,1-Dichloroethane* | 75-34-3 | 50 | 1,000 |
| 1,2-Dichloroethane* | 107-06-2 | 2 | 40 |
| 1,1-Dichloroethene (1,1-Dichloroethylene)* | 75-35-4 | 1 | 20 |
| 1,2-Dichloroethene (cis) (c-1,2-Dichloroethylene)* | 156-59-2 | 70 | 1,400 |

| Contaminant | CAS No. | Ground Water Remediation Standard (GWRS)(µg/L) | Soil Leachate Remediation Standard - Migration to Ground Water (MGWLEACHATE) (µg/L) |
|---|---|---|---|
| 1,2-Dichloroethene (trans) (t-1,2-Dichloroethylene)* | 156-60-5 | 100 | 2,000 |
| 2,4-Dichlorophenol | 120-83-2 | 20 | 400 |
| 1,2-Dichloropropane* | 78-87-5 | 1 | 20 |
| 1,3-Dichloropropene (total)* | 542-75-6 | 1 | 20 |
| Dieldrin | 60-57-1 | 0.03 | 0.60 |
| Diethylphthalate | 84-66-2 | 6,000 | 120,000 |
| 2,4-Dimethylphenol | 105-67-9 | 100 | 2,000 |
| Di-n-butyl phthalate | 84-74-2 | 700 | NA[1] |
| 2,4-Dinitrophenol | 51-28-5 | 40 | 800 |
| 2,4-Dinitrotoluene/2,6-Dinitrotoluene (mixture) | 25321-14-6 | 10 | 200 |
| Di-n-octyl phthalate | 117-84-0 | 100 | NA[1] |
| 1,4-Dioxane* | 123-91-1 | 0.4 | 8.0 |
| Endosulfan I and Endosulfan II (alpha and beta) (summed) | 115-29-7 | 40 | NA[1] |
| Endrin | 72-20-8 | 2 | 40 |
| Ethylbenzene* | 100-41-4 | 700 | 14,000 |
| Extractable Petroleum Hydrocarbons (No. 2 Fuel Oil and Diesel) | various | NA | NA[3] |
| Extractable Petroleum Hydrocarbons (Other) | various | NA | NA[3] |
| Fluoranthene | 206-44-0 | 300 | NA[1] |
| Fluorene | 86-73-7 | 300 | NA[1] |
| alpha-HCH (alpha-BHC) | 319-84-6 | 0.02 | 0.40 |
| beta-HCH (beta-BHC) | 319-85-7 | 0.04 | 0.80 |
| Heptachlor | 76-44-8 | 0.05 | 1.0 |
| Heptachlor epoxide | 1024-57-3 | 0.2 | 4.0 |
| Hexachlorobenzene | 118-74-1 | 0.02 | 0.40 |
| Hexachloro-1,3-butadiene | 87-68-3 | 1 | 20 |
| Hexachlorocyclopentadiene | 77-47-4 | 40 | 800 |
| Hexachloroethane | 67-72-1 | 7 | 140 |
| n-Hexane* | 110-54-3 | 30 | 600 |
| 2-Hexanone* | 591-78-6 | 40 | 800 |
| Indeno(1,2,3-cd)pyrene | 193-39-5 | 0.2 | NA[1] |
| Isophorone | 78-59-1 | 40 | 800 |
| Isopropylbenzene* | 98-82-8 | 700 | 14,000 |
| Lead (total) | 7439-92-1 | 5 | 100 |

| Contaminant | CAS No. | Ground Water Remediation Standard (GWRS)(µg/L) | Soil Leachate Remediation Standard - Migration to Ground Water (MGWLEACHATE) (µg/L) |
|---|---|---|---|
| Lindane (gamma-HCH) (gamma-BHC) | 58-89-9 | 0.03 | 0.60 |
| Manganese (total) | 7439-96-5 | NA | NA[2] |
| Mercury (total) | 7439-97-6 | 2 | 40 |
| Methoxychlor | 72-43-5 | 40 | NA[1] |
| Methyl acetate* | 79-20-9 | 7,000 | 140,000 |
| Methylene chloride (Dichloromethane)* | 75-09-2 | 3 | 60 |
| 2-Methylnaphthalene | 91-57-6 | 30 | 600 |
| 4-Methyl-2-pentanone (MIBK)* | 108-10-1 | NA | NA[3] |
| 2-Methylphenol (o-cresol) | 95-48-7 | 50 | 1,000 |
| 4-Methylphenol (p-cresol) | 106-44-5 | 50 | 1,000 |
| Methyl tert-butyl ether (MTBE)* | 1634-04-4 | 70 | 1,400 |
| Naphthalene | 91-20-3 | 300 | 6,000 |
| Nickel (total) | 7440-02-0 | 100 | 2,000 |
| 4-Nitroaniline | 100-01-6 | NA | NA[3] |
| Nitrobenzene | 98-95-3 | 6 | 120 |
| N-Nitrosodi-n-propylamine | 621-64-7 | 10 | 200 |
| N-Nitrosodiphenylamine | 86-30-6 | 10 | 200 |
| 2,2'-oxybis(1-chloropropane) | 108-60-1 | 300 | 6,000 |
| Pentachlorophenol | 87-86-5 | 0.3 | 6.0 |
| Phenol | 108-95-2 | 2,000 | 40,000 |
| Polychlorinated biphenyls (PCBs) | 1336-36-3 | 0.5 | 10 |
| Pyrene | 129-00-0 | 200 | NA[1] |
| Selenium (total) | 7782-49-2 | 40 | 800 |
| Silver (total) | 7440-22-4 | 40 | 800 |
| Styrene* | 100-42-5 | 100 | 2,000 |
| Tertiary butyl alcohol (TBA)* | 75-65-0 | 100 | 2,000 |
| 1,2,4,5-Tetrachlorobenzene | 95-94-3 | NA | NA[3] |
| 2,3,7,8-Tetrachlorodibenzo-p-dioxin | 1746-01-6 | 0.00001 | 0.00020[4] |
| 1,1,2,2-Tetrachloroethane* | 79-34-5 | 1 | 20 |
| Tetrachloroethene (PCE) (Tetrachloroethylene)* | 127-18-4 | 1 | 20 |
| 2,3,4,6-Tetrachlorophenol | 58-90-2 | 200 | 4,000 |
| Toluene* | 108-88-3 | 600 | 12,000 |
| Toxaphene | 8001-35-2 | 2 | 40 |
| 1,2,4-Trichlorobenzene | 120-82-1 | 9 | 180 |
| 1,1,1-Trichloroethane* | 71-55-6 | 30 | 600 |
| 1,1,2-Trichloroethane* | 79-00-5 | 3 | 60 |

| Contaminant | CAS No. | Ground Water Remediation Standard (GWRS)(µg/L) | Soil Leachate Remediation Standard - Migration to Ground Water (MGW LEACHATE) (µg/L) |
|---|---|---|---|
| Trichloroethene (TCE) (Trichloroethylene)* | 79-01-6 | 1 | 20 |
| Trichlorofluoromethane (Freon 11)* | 75-69-4 | 2,000 | 40,000 |
| 2,4,5-Trichlorophenol | 95-95-4 | 700 | 14,000 |
| 2,4,6-Trichlorophenol | 88-06-2 | 20 | 400 |
| 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon TF)* | 76-13-1 | 20,000 | NA[3] |
| 1,2,4-Trimethylbenzene* | 95-63-6 | NA | NA[3] |
| Vanadium (total) | 7440-62-2 | NA | NA[3] |
| Vinyl chloride* | 75-01-4 | 1 | 20 |
| Xylenes (total)* | 1330-20-7 | 1,000 | 20,000 |
| Zinc (total) | 7440-66-6 | 2,000 | 40,000 |

* = Contaminant is a volatile
NA = Standard not available/not applicable
1 – Standard not applicable because soil criterion is above soil saturation limit
2 – Standard not applicable because Ground Water Remediation Standard is a secondary standard
3 – Ground Water Remediation Standard not available
4 – This standard is used in conjunction with soil data that have been converted to sample-specific TCDD-TEQ values through application of the Toxicity Equivalence Factor Methodology (USEPA 2010) and using the WHO 2005 Mammalian Toxic Equivalency Factors (TEFs).

# Table 3

## CHEMICAL AND PHYSICAL PROPERTIES OF CONTAMINANTS

| Contaminant | CAS No. | Water Solubility (mg/L) | Henry's Law Constant (atm-$m^3$/mol, 25°C) | Henry's Law Constant (dimensionless, 25°C) | Air Diffusivity ($cm^2$/sec) | Water Diffusivity ($cm^2$/sec) | Soil Organic Carbon-Water Partition Coefficient, $K_{oc}$ (L/kg) | Soil-Water Partition Coefficient, $K_d$ (L/kg) |
|---|---|---|---|---|---|---|---|---|
| Acenaphthene | 83-32-9 | 3.9 | 1.84E-04 | 7.5224E-03 | 5.0614E-02 | 8.3300E-06 | 5027 | NA |
| Acetone (2-Propanone) | 67-64-1 | 1000000 | 3.50E-05 | 1.4309E-03 | 1.0592E-01 | 1.1471E-05 | 2.364 | NA |
| Acetophenone | 98-86-2 | 6130 | 1.04E-05 | 4.2518E-04 | 6.5222E-02 | 8.7228E-06 | 51.85 | NA |
| Aldrin | 309-00-2 | 0.017 | 4.40E-05 | 1.7989E-03 | 2.2812E-02 | 5.8402E-06 | 82020 | NA |
| Aluminum (total) | 7429-90-5 | NA | NA | NA | NA | NA | NA | 1500 |
| Anthracene | 120-12-7 | 0.0434 | 5.56E-05 | 2.2731E-03 | 3.8973E-02 | 7.8522E-06 | 16360 | NA |
| Antimony (total) | 7440-36-0 | NA | NA | NA | NA | NA | NA | 45 |
| Arsenic (total) | 7440-38-2 | NA | NA | NA | NA | NA | NA | 26[1] |
| Atrazine | 1912-24-9 | 34.7 | 2.36E-09 | 9.6484E-08 | 2.6466E-02 | 6.8378E-06 | 224.5 | NA |
| Barium (total) | 7440-39-3 | NA | NA | NA | NA | NA | NA | 17[1] |
| Benzaldehyde | 100-52-7 | 6950 | 2.67E-05 | 1.0916E-03 | 7.4393E-02 | 9.4627E-06 | 11.09 | NA |
| Benzene | 71-43-2 | 1790 | 5.55E-03 | 2.2690E-01 | 8.9534E-02 | 1.0263E-05 | 145.8 | NA |
| Benzo(a)anthracene (1,2-Benzanthracene) | 56-55-3 | 0.0094 | 1.20E-05 | 4.9059E-04 | 2.6144E-02 | 6.7495E-06 | 176900 | NA |
| Benzo(a)pyrene | 50-32-8 | 0.00162 | 4.57E-07 | 1.8683E-05 | 4.7583E-02 | 5.5597E-06 | 587400 | NA |
| Benzo(b)fluoranthene (3,4-Benzofluoranthene) | 205-99-2 | 0.0015 | 6.57E-07 | 2.6860E-05 | 4.7583E-02 | 5.5597E-06 | 599400 | NA |
| Benzo(k)fluoranthene | 207-08-9 | 0.0008 | 5.84E-07 | 2.3875E-05 | 4.7583E-02 | 5.5597E-06 | 587400 | NA |
| Beryllium | 7440-41-7 | NA | NA | NA | NA | NA | NA | 35[1] |
| 1,1'-Biphenyl | 92-52-4 | 7.48 | 3.08E-04 | 1.2592E-02 | 4.7059E-02 | 7.5618E-06 | 5129 | NA |
| Bis(2-chloroethoxy)methane | 111-91-1 | 7800 | 3.85E-06 | 1.5740E-04 | 6.1186E-02 | 7.1492E-06 | 14.38 | NA |
| Bis(2-chloroethyl)ether | 111-44-4 | 17200 | 1.70E-05 | 6.9501E-04 | 5.6719E-02 | 8.7070E-06 | 32.21 | NA |
| Bis(2-ethylhexyl)phthalate | 117-81-7 | 0.27 | 2.70E-07 | 1.1038E-05 | 1.7340E-02 | 4.1807E-06 | 119600 | NA |
| Bromodichloromethane (Dichlorobromomethane) | 75-27-4 | 3032 | 2.12E-03 | 8.6672E-02 | 5.6263E-02 | 1.0731E-05 | 31.82 | NA |
| Bromoform | 75-25-2 | 3100 | 5.35E-04 | 2.1872E-02 | 3.5732E-02 | 1.0356E-05 | 31.82 | NA |
| Bromomethane (Methyl bromide) | 74-83-9 | 15200 | 7.34E-03 | 3.0008E-01 | 1.0050E-01 | 1.3468E-05 | 13.22 | NA |
| 2-Butanone (Methyl ethyl ketone) (MEK) | 78-93-3 | 223000 | 5.69E-05 | 2.3262E-03 | 9.1446E-02 | 1.0193E-05 | 4.51 | NA |
| Butylbenzyl phthalate | 85-68-7 | 2.69 | 1.26E-06 | 5.1513E-05 | 2.0832E-02 | 5.1733E-06 | 7155 | NA |
| Cadmium | 7440-43-9 | NA | NA | NA | NA | NA | NA | 23[1] |
| Caprolactam | 105-60-2 | 772000 | 2.53E-08 | 1.0343E-06 | 6.9242E-02 | 8.9994E-06 | 24.5 | NA |
| Carbon disulfide | 75-15-0 | 2160 | 1.44E-02 | 5.8872E-01 | 1.0644E-01 | 1.2977E-05 | 21.73 | NA |
| Carbon tetrachloride | 56-23-5 | 793 | 2.76E-02 | 1.1284E+00 | 5.7143E-02 | 9.7849E-06 | 43.89 | NA |
| Chlordane (alpha and gamma forms summed) | 57-74-9 | 0.056[2] | 4.86E-05[2] | 1.9869E-03[2] | 1.7900E-02[3] | 4.3700E-06[3] | 67540[5] | NA |
| 4-Chloroaniline | 106-47-8 | 3900 | 1.16E-06 | 4.7424E-05 | 7.0385E-02 | 1.0253E-05 | 112.7 | NA |

| Contaminant | CAS No. | Water Solubility (mg/L) | Henry's Law Constant (atm-$m^3$/mol, 25°C) | Henry's Law Constant (dimensionless, 25°C) | Air Diffusivity ($cm^2$/sec) | Water Diffusivity ($cm^2$/sec) | Soil Organic Carbon-Water Partition Coefficient, $K_{oc}$ (L/kg) | Soil-Water Partition Coefficient, $K_d$ (L/kg) |
|---|---|---|---|---|---|---|---|---|
| Chlorobenzene | 108-90-7 | 498 | 3.11E-03 | 1.2715E-01 | 7.2130E-02 | 9.4765E-06 | 233.9 | NA |
| Chloroethane (Ethyl chloride) | 75-00-3 | 6710 | 1.11E-02 | 4.5380E-01 | 1.0376E-01 | 1.1619E-05 | 21.73 | NA |
| Chloroform | 67-66-3 | 7950 | 3.67E-03 | 1.5004E-01 | 7.6920E-02 | 1.0891E-05 | 31.82 | NA |
| Chloromethane (Methyl chloride) | 74-87-3 | 5320 | 8.82E-03 | 3.6059E-01 | 1.2396E-01 | 1.3648E-05 | 13.22 | NA |
| 2-Chloronaphthalene | 91-58-7 | 11.7 | 3.20E-04 | 1.3082E-02 | 4.4691E-02 | 7.7301E-06 | 2478 | NA |
| 2-Chlorophenol (o-Chlorophenol) | 95-57-8 | 11300 | 1.12E-05 | 4.57890E-04 | 6.6118E-02 | 9.4784E-06 | 398[1] | NA |
| Chrysene | 218-01-9 | 0.002 | 5.23E-06 | 2.1382E-04 | 2.6114E-02 | 6.7495E-06 | 180500 | NA |
| Cobalt (total) | 7440-48-4 | NA | NA | NA | NA | NA | NA | 45 |
| Copper (total) | 7440-50-8 | NA | NA | NA | NA | NA | NA | 35 |
| Cyanide | 57-12-5 | NA | NA | NA | NA | NA | NA | 9.9 |
| Cyclohexane | 110-82-7 | 55 | 1.50E-01 | 6.1325E+00 | 7.9973E-02 | 9.1077E-06 | 145.8 | NA |
| 4,4'-DDD (p,p'-TDE) | 72-54-8 | 0.09 | 6.60E-06 | 2.6983E-04 | 4.0608E-02 | 4.7447E-06 | 117500 | NA |
| 4,4'-DDE (p,p'-DDX) | 72-55-9 | 0.04 | 4.16E-05 | 1.7007E-03 | 2.3000E-02 | 5.8592E-06 | 117500 | NA |
| 4,4'-DDT | 50-29-3 | 0.0055 | 8.32E-06 | 3.4015E-04 | 3.7933E-02 | 4.4322E-06 | 168600 | NA |
| Dibenz(a,h)anthracene | 53-70-3 | 0.00249 | 1.41E-07 | 5.7645E-06 | 4.4567E-02 | 5.2073E-06 | 1912000 | NA |
| Dibromochloromethane (Chlorodibromomethane) | 124-48-1 | 2700 | 7.83E-04 | 3.2011E-02 | 3.6636E-02 | 1.0561E-05 | 31.82 | NA |
| 1,2-Dibromo-3-chloropropane | 96-12-8 | 1230 | 1.47E-04 | 6.0098E-03 | 3.2135E-02 | 8.9048E-06 | 115.8 | NA |
| 1,2-Dibromoethane (Ethylene dibromide) | 106-93-4 | 3910 | 6.50E-04 | 2.6574E-02 | 4.3035E-02 | 1.0439E-05 | 39.6 | NA |
| 1,2-Dichlorobenzene (o-Dichlorobenzene) | 95-50-1 | 156 | 1.92E-03 | 7.8496E-02 | 5.6170E-02 | 8.9213E-06 | 382.9 | NA |
| 1,3-Dichlorobenzene (m-Dichlorobenzene) | 541-73-1 | 125[2] | 2.63E-03[2] | 1.0751E-01[2] | 6.9200E-02[4] | 7.8600E-06[4] | 375.3[5] | NA |
| 1,4-Dichlorobenzene (p-Dichlorobenzene) | 106-46-7 | 81.3 | 2.41E-03 | 9.8528E-02 | 5.5043E-02 | 8.6797E-06 | 375.3 | NA |
| 3,3'-Dichlorobenzidine | 91-94-1 | 3.1 | 2.84E-11 | 1.1611E-09 | 4.7482E-02 | 5.5478E-06 | 3190 | NA |
| Dichlorodifluoromethane (Freon 12) | 75-71-8 | 280 | 3.43E-01 | 1.4023E+01 | 7.6029E-02 | 1.0839E-05 | 43.89 | NA |
| 1,1-Dichloroethane | 75-34-3 | 5040 | 5.62E-03 | 2.2976E-01 | 8.3645E-02 | 1.0621E-05 | 31.82 | NA |
| 1,2-Dichloroethane | 107-06-2 | 8600 | 1.18E-03 | 4.8242E-02 | 8.5722E-02 | 1.0995E-05 | 39.6 | NA |
| 1,1-Dichloroethene (1,1-Dichloroethylene) | 75-35-4 | 2420 | 2.61E-02 | 1.0670E+00 | 8.6311E-02 | 1.0956E-05 | 31.82 | NA |
| 1,2-Dichloroethene (cis) (c-1,2-Dichloroethylene) | 156-59-2 | 6410 | 4.08E-03 | 1.6680E-01 | 8.8406E-02 | 1.1335E-05 | 39.6 | NA |
| 1,2-Dichloroethene (trans) (t-1,2-Dichloroethylene) | 156-60-5 | 4520 | 9.38E-03 | 3.8348E-01 | 8.7609E-02 | 1.1191E-05 | 39.6 | NA |
| 2,4-Dichlorophenol | 120-83-2 | 5500 | 4.29E-06 | 1.7538E-04 | 4.8577E-02 | 8.6786E-06 | 159[1] | NA |
| 1,2-Dichloropropane | 78-87-5 | 2800 | 2.82E-03 | 1.1529E-01 | 7.3340E-02 | 9.7252E-06 | 60.7 | NA |
| 1,3-Dichloropropene (total) | 542-75-6 | 2800 | 3.55E-03 | 1.4513E-01 | 7.6272E-02 | 1.0123E-05 | 72.17 | NA |
| Dieldrin | 60-57-1 | 0.195 | 1.00E-05 | 4.0883E-04 | 2.3286E-02 | 6.0062E-06 | 20090 | NA |
| Diethylphthalate | 84-66-2 | 1080 | 6.10E-07 | 2.4939E-05 | 2.6074E-02 | 6.7227E-06 | 104.9 | NA |
| 2,4-Dimethylphenol | 105-67-9 | 7870 | 9.51E-07 | 3.8879E-05 | 6.2245E-02 | 8.3140E-06 | 491.8 | NA |
| Di-n-butyl phthalate | 84-74-2 | 11.2 | 1.81E-06 | 7.3998E-05 | 2.1436E-02 | 5.3255E-06 | 1157 | NA |

| **Contaminant** | **CAS No.** | **Water Solubility (mg/L)** | **Henry's Law Constant (atm-m$^3$/mol, 25°C)** | **Henry's Law Constant (dimensionless, 25°C)** | **Air Diffusivity (cm$^2$/sec)** | **Water Diffusivity (cm$^2$/sec)** | **Soil Organic Carbon-Water Partition Coefficient, $K_{oc}$ (L/kg)** | **Soil-Water Partition Coefficient, $K_d$ (L/kg)** |
|---|---|---|---|---|---|---|---|---|
| 2,4-Dinitrophenol | 51-28-5 | 2790 | 8.60E-08 | 3.5159E-06 | 4.06670E-02 | 9.0756E-06 | 0.0178[1] | NA |
| 2,4-Dinitrotoluene/2,6-Dinitrotoluene (mixture) | 25321-14-6 | 270 | 3.97E-07 | 1.6230E-05 | 5.9131E-02 | 6.9090E-06 | 587.4 | NA |
| Di-n-octyl phthalate | 117-84-0 | 0.022 | 2.57E-06 | 1.0506E-04 | 3.5559E-02 | 4.1548E-06 | 140800.00 | NA |
| 1,4-Dioxane | 123-91-1 | 1000000 | 4.80E-06 | 1.9624E-04 | 8.7374E-02 | 1.0541E-05 | 2.633 | NA |
| Endosulfan I and Endosulfan II<br>(alpha and beta) (summed) | 115-29-7 | 0.325 | 6.50E-05 | 2.6574E-03 | 2.2484E-02 | 5.7628E-06 | 6761 | NA |
| Endrin | 72-20-8 | 0.25 | 6.36E-06 | 2.600E-04 | 3.6158E-02 | 4.2248E-06 | 20090 | NA |
| Ethylbenzene | 100-41-4 | 169 | 7.88E-03 | 3.2216E-01 | 6.8465E-02 | 8.4558E-06 | 446.1 | NA |
| Extractable Petroleum Hydrocarbons (Category 1) | various | NA | NA | NA | NA | NA | NA | NA |
| Extractable Petroleum Hydrocarbons (Category 2) | various | NA | NA | NA | NA | NA | NA | NA |
| Fluoranthene | 206-44-0 | 0.26 | 8.86E-06 | 3.6222E-04 | 2.7596E-02 | 7.1827E-06 | 55450 | NA |
| Fluorene | 86-73-7 | 1.69 | 9.62E-05 | 3.9329E-03 | 4.3974E-02 | 7.8890E-06 | 9160 | NA |
| alpha-HCH (alpha-BHC) | 319-84-6 | 2 | 6.70E-06 | 2.7392E-04 | 4.3284E-02 | 5.0574E-06 | 2807 | NA |
| beta-HCH (beta-BHC) | 319-85-7 | 0.24 | 4.40E-06 | 1.7988E-05 | 2.7667E-02 | 7.3955E-06 | 2807 | NA |
| Heptachlor | 76-44-8 | 0.18 | 2.94E-04 | 1.2020E-02 | 2.2344E-02 | 5.6959E-06 | 41260 | NA |
| Heptachlor epoxide | 1024-57-3 | 0.2 | 2.10E-05 | 8.5854E-04 | 2.4001E-02 | 6.2475E-06 | 10110 | NA |
| Hexachlorobenzene | 118-74-1 | 0.0062 | 1.70E-03 | 6.9501E-02 | 2.8974E-02 | 7.8497E-06 | 6195 | NA |
| Hexachloro-1,3-butadiene | 87-68-3 | 3.2 | 1.03E-02 | 4.2110E-01 | 2.6744E-02 | 7.0264E-06 | 845.2 | NA |
| Hexachlorocyclopentadiene | 77-47-4 | 1.8 | 2.70E-02 | 1.1038E+00 | 2.7238E-02 | 7.2170E-06 | 1404 | NA |
| Hexachloroethane | 67-72-1 | 50 | 3.89E-03 | 1.5904E-01 | 3.2094E-02 | 8.8904E-06 | 196.8 | NA |
| n-Hexane | 110-54-3 | 9.5 | 1.80E+00 | 7.3590E+01 | 7.3108E-02 | 8.1658E-06 | 131.5 | NA |
| 2-Hexanone | 591-78-6 | 17200 | 9.32E-05 | 3.8103E-03 | 7.0356E-02 | 8.4404E-06 | 14.98 | NA |
| Indeno(1,2,3-cd)pyrene | 193-39-5 | 0.00019 | 3.48E-07 | 1.4227E-05 | 4.4784E-02 | 5.2327E-06 | 1951000 | NA |
| Isophorone | 78-59-1 | 12000 | 6.64E-06 | 2.7146E-04 | 5.2505E-02 | 7.5296E-06 | 65.15 | NA |
| Isopropylbenzene | 98-82-8 | 61.3 | 1.15E-02 | 4.7016E-01 | 6.0304E-02 | 7.8566E-06 | 697.8 | NA |
| Lead (total) | 7439-92-1 | NA | NA | NA | NA | NA | NA | 900 |
| Lindane<br>(gamma-HCH)<br>(gamma-BHC) | 58-89-9 | 7.3 | 5.14E-06 | 2.1014E-04 | 4.3284E-02 | 5.0574E-06 | 2807 | NA |
| Manganese (total) | 7439-96-5 | NA | NA | NA | NA | NA | NA | 65 |
| Mercury (total) | 7439-97-6 | NA | NA | NA | NA | NA | NA | 0.20[1] |
| Methoxychlor | 72-43-5 | 0.1 | 2.03E-07 | 8.2993E-06 | 2.2085E-02 | 5.5926E-06 | 26890 | NA |
| Methyl acetate | 79-20-9 | 243000 | 1.15E-04 | 4.7016E-03 | 9.5776E-02 | 1.1008E-05 | 3.064 | NA |
| Methylene chloride (Dichloromethane) | 75-09-2 | 13000 | 3.25E-03 | 1.3287E-01 | 9.9936E-02 | 1.2512E-05 | 21.73 | NA |
| 2-Methylnaphthalene | 91-57-6 | 24.6 | 5.18E-04 | 2.1177E-02 | 5.2432E-02 | 7.7811E-06 | 2478 | NA |
| 4-Methyl-2-pentanone (MIBK) | 108-10-1 | 19000 | 1.38E-04 | 5.6419E-03 | 6.9780E-02 | 8.3477E-06 | 12.6 | NA |
| 2-Methylphenol (o-cresol) | 95-48-7 | 25900 | 1.20E-06 | 4.9060E-05 | 7.2835E-02 | 9.3168E-06 | 306.5 | NA |
| 4-Methylphenol (p-cresol) | 106-44-5 | 21500 | 1.00E-06 | 4.0883E-05 | 7.2394E-02 | 9.2397E-06 | 300.4 | NA |

| Contaminant | CAS No. | Water Solubility (mg/L) | Henry's Law Constant (atm-$m^3$/mol, 25°C) | Henry's Law Constant (dimensionless, 25°C) | Air Diffusivity ($cm^2$/sec) | Water Diffusivity ($cm^2$/sec) | Soil Organic Carbon-Water Partition Coefficient, $K_{oc}$ (L/kg) | Soil-Water Partition Coefficient, $K_d$ (L/kg) |
|---|---|---|---|---|---|---|---|---|
| Methyl tert-butyl ether (MTBE) | 1634-04-4 | 51000 | 5.87E-04 | 2.3998E-02 | 7.5267E-02 | 8.5904E-06 | 11.56 | NA |
| Naphthalene | 91-20-3 | 31 | 4.40E-04 | 1.7988E-02 | 6.0499E-02 | 8.3770E-06 | 1544 | NA |
| Nickel (total) | 7440-02-0 | NA | NA | NA | NA | NA | NA | 24[1] |
| 4-Nitroaniline | 100-01-6 | 728 | 1.26E-09 | 5.153E-08 | 6.3660E-02 | 9.7545E-06 | 109.1 | NA |
| Nitrobenzene | 98-95-3 | 2090 | 2.40E-05 | 9.8119E-04 | 6.8054E-02 | 9.4494E-06 | 226.4 | NA |
| N-Nitrosodi-n-propylamine | 621-64-7 | 13000 | 5.38E-06 | 2.1995E-04 | 5.6440E-02 | 7.7580E-06 | 275.4 | NA |
| N-Nitrosodiphenylamine | 86-30-6 | 35 | 1.21E-06 | 4.9648E-05 | 5.5886E-02 | 6.5299E-06 | 2632 | NA |
| 2,2'-oxybis (1-chloropropane) | 108-60-1 | 1700 | 7.42E-05 | 3.0335E-03 | 3.9889E-02 | 7.3606E-06 | 82.92 | NA |
| Pentachlorophenol | 87-86-5 | 14 | 2.45E-08 | 1.0016E-06 | 2.9520E-02 | 8.0121E-06 | 5100[1] | NA |
| Phenol | 108-95-2 | 82800 | 3.33E-07 | 1.3614E-05 | 8.3398E-02 | 1.0254E-05 | 187.2 | NA |
| Polychlorinated biphenyls (PCBs) | 1336-36-3 | 0.7 | 4.15E-04 | 1.6966E-02 | 2.4340E-02 | 6.2671E-06 | 78100 | NA |
| Pyrene | 129-00-0 | 0.135 | 1.19E-05 | 4.8651E-04 | 2.7787E-02 | 7.2479E-06 | 54340 | NA |
| Selenium (total) | 7782-49-2 | NA | NA | NA | NA | NA | NA | 14[1] |
| Silver (total) | 7440-22-4 | NA | NA | NA | NA | NA | NA | 0.26[1] |
| Styrene | 100-42-5 | 310 | 2.75E-03 | 1.1243E-01 | 7.1114E-02 | 8.7838E-06 | 446.1 | NA |
| Tertiary butyl alcohol (TBA) | 75-65-0 | 1000000[2] | 9.05E-06[2] | 3.6996E-04[2] | 9.8500E-02[3] | 1.1400E-05[3] | 2.111[5] | NA |
| 1,2,4,5-Tetrachlorobenzene | 95-94-3 | 0.595 | 1.00E-03 | 4.0883E-02 | 3.1896E-02 | 8.7531E-06 | 2220 | NA |
| 2,3,7,8-Tetrachlorodibenzo-p-dioxin | 1746-01-6 | 0.0002 | 5.00E-05 | 2.0442E-03 | 4.7028E-02 | 6.7568E-06 | 249100 | NA |
| 1,1,2,2-Tetrachloroethane | 79-34-5 | 2830 | 3.67E-04 | 1.5004E-02 | 4.8921E-02 | 9.2902E-06 | 94.94 | NA |
| Tetrachloroethene (PCE) (Tetrachloroethylene) | 127-18-4 | 206 | 1.77E-02 | 7.2363E-01 | 5.0466E-02 | 9.4551E-06 | 94.94 | NA |
| 2,3,4,6-Tetrachlorophenol | 58-90-2 | 23 | 8.84E-06 | 3.6140E-04 | 5.0338E-02 | 5.8816E-06 | 2969[1] | NA |
| Toluene | 108-88-3 | 526 | 6.64E-03 | 2.7146E-01 | 7.7804E-02 | 9.2043E-06 | 233.9 | NA |
| Toxaphene | 8001-35-2 | 0.55 | 6.00E-06 | 2.4530E-04 | 3.2439E-02 | 3.7902E-06 | 77200 | NA |
| 1,2,4-Trichlorobenzene | 120-82-1 | 49 | 1.42E-03 | 5.8054E-02 | 3.9599E-02 | 8.4033E-06 | 1356 | NA |
| 1,1,1-Trichloroethane | 71-55-6 | 1290 | 1.72E-02 | 7.0319E-01 | 6.4817E-02 | 9.5990E-06 | 43.89 | NA |
| 1,1,2-Trichloroethane | 79-00-5 | 4590 | 8.24E-04 | 3.3688E-02 | 6.6890E-02 | 1.0026E-05 | 60.7 | NA |
| Trichloroethene (TCE) (Trichloroethylene) | 79-01-6 | 1280 | 9.85E-03 | 4.0270E-01 | 6.8662E-02 | 1.0221E-05 | 60.7 | NA |
| Trichlorofluoromethane (Freon 11) | 75-69-4 | 1100 | 9.70E-02 | 3.9657E+00 | 6.5356E-02 | 1.0048E-05 | 43.89 | NA |
| 2,4,5-Trichlorophenol | 95-95-4 | 1200 | 1.62E-06 | 6.6230E-05 | 3.1394E-02 | 8.0893E-06 | 3140[1] | NA |
| 2,4,6-Trichlorophenol | 88-06-2 | 800 | 2.60E-06 | 1.0630E-04 | 3.1395E-02 | 8.0896E-06 | 999[1] | NA |
| 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon TF) | 76-13-1 | 170 | 5.26E-01 | 2.1504E+01 | 3.7566E-02 | 8.5920E-06 | 196.8 | NA |
| 1,2,4-Trimethylbenzene | 95-63-6 | 57 | 6.16E-03 | 2.5184E-01 | 6.0675E-02 | 7.9208E-06 | 614.3 | NA |
| Vanadium (total) | 7440-62-2 | NA | NA | NA | NA | NA | NA | 1000 |
| Vinyl chloride | 75-01-4 | 8800 | 2.78E-02 | 1.1365E+00 | 1.0712E-01 | 1.2004E-05 | 21.73 | NA |
| Xylenes (total) | 1330-20-7 | 106 | 6.63E-03 | 2.7105E-01 | 6.8515E-02 | 8.4640E-06 | 382.9 | NA |

| Contaminant | CAS No. | Water Solubility (mg/L) | Henry's Law Constant (atm-m$^3$/mol, 25°C) | Henry's Law Constant (dimensionless, 25°C) | Air Diffusivity (cm$^2$/sec) | Water Diffusivity (cm$^2$/sec) | Soil Organic Carbon-Water Partition Coefficient, $K_{oc}$ (L/kg) | Soil-Water Partition Coefficient, $K_d$ (L/kg) |
|---|---|---|---|---|---|---|---|---|
| Zinc (total) | 7440-66-6 | NA | NA | NA | NA | NA | NA | 23[1] |

NA = Not applicable
All values from USEPA Regional Screening Level Tables, May 2018 (http://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables ), unless otherwise indicated
1 – Kd or Koc value listed for pH 5.3 in Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites, United States Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC, December 2002 (http://www.epa.gov/superfund/superfund-soil-screening-guidance)
2 – Experimental values from USEPA's Estimation Program Interface Suite, V 4.11 (https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface)
3 – Calculated using USEPA's WATER9 calculator, V 3.0. (https://www3.epa.gov/ttn/chief/software/water/index.html)
4 – From USEPA's WATER9 calculator, V 3.0 database (https://www3.epa.gov/ttn/chief/software/water/index.html)
5 – Molecular Connectivity Index values from USEPA's Estimation Program Interface Suite, V 4.11 (https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface)

## Table 4

## CALCULATED INFILTRATION RATES FOR NEW JERSEY SOILS USING NEW JERSEY GEOLOGICAL SURVEY GSR-32 METHODOLOGY

| Soil Name | Primary Counties of Occurrence | Representative Municipality | Recharge (in/yr) | | |
|---|---|---|---|---|---|
| | | | Landscaped Open Space | Unvegetated | General Agriculture |
| Sassafras sandy loam | Mercer | Washington Twp. | 13.6 | 8.9 | 11.9 |
| Sassafras sandy loam | Middlesex | South River Boro | 13.8 | 9.1 | 12.2 |
| Sassafras sandy loam | Burlington | Delran Twp. | 13.2 | 8.7 | 11.6 |
| Sassafras sandy loam | Salem | Alloway Twp. | 11.3 | 7.7 | 9.9 |
| Sassafras sandy loam | Cumberland | Bridgeton City | 10.7 | 7.5 | 9.4 |
| Freehold sandy loam | Monmouth | Millstone Twp. | 13.8 | 8.9 | 12.1 |
| Freehold sandy loam | Burlington | Chesterfield Twp. | 13.8 | 8.9 | 12.1 |
| Freehold sandy loam | Camden | Runnemede Boro | 11.5 | 7.7 | 10 |
| Freehold sandy loam | Gloucester | Swedesboro Boro | 10.8 | 7.4 | 9.4 |
| Collington sandy loam | Monmouth | Holmdel Twp. | 13.7 | 8.7 | 12 |
| Colts Neck sandy loam | Monmouth | Colts Neck Twp. | 14.6 | 9.1 | 12.7 |
| Westphalia sandy loam | Camden | Lindenwold Boro | 12 | 7.5 | 10.4 |
| Westphalia sandy loam | Gloucester | Harrison Twp. | 10.5 | 6.8 | 9.1 |
| Aura sandy loam | Gloucester | Elk Twp. | 11.2 | 7.7 | 10 |
| Aura sandy loam | Salem | Pittsgrove Twp. | 11.3 | 7.8 | 10 |
| Aura sandy loam | Cumberland | Upper Deerfield Twp. | 11.2 | 7.7 | 9.9 |
| Dunnellen sandy loam | Bergen | Oradell Boro | 16 | 10.1 | 14.1 |
| Dunnellen sandy loam | Union | Plainfield City | 15.9 | 10.1 | 14 |
| Dunnellen sandy loam | Middlesex | Piscataway Twp. | 14.8 | 9.5 | 13.1 |
| Galestown sand | Mercer | Trenton City | 16.4 | 13.9 | 15.6 |
| Galestown sand | Burlington | Burlington City | 16.1 | 13.7 | 15.2 |
| Lakewood sand | Monmouth | Neptune Twp. | 17.3 | 14.5 | 16.4 |
| Lakewood sand | Ocean | Manchester Twp. | 16.8 | 14.2 | 15.9 |
| Lakewood sand | Burlington | Pemberton Twp. | 15.9 | 13.6 | 15.1 |
| Downer loamy sand | Monmouth | Neptune Twp. | 16 | 10.6 | 14.3 |
| Downer loamy sand | Ocean | Manchester Twp. | 15.5 | 10.4 | 13.9 |
| Downer loamy sand | Burlington | Pemberton Twp. | 14.7 | 9.9 | 13.2 |
| Downer loamy sand | Atlantic | Galloway Twp. | 12.3 | 8.4 | 11 |
| Downer loamy sand | Cumberland | Vineland City | 12.5 | 8.5 | 11.1 |
| Hammonton loamy sand | Atlantic | Estelle Manor City | 12.3 | 8.5 | 10.8 |
| Hammonton loamy sand | Cumberland | Hopewell Twp. | 11.7 | 8.2 | 10.3 |
| Hammonton loamy sand | Cape May | Lower Twp. | 10.6 | 7.6 | 9.7 |
| Boonton loam | Passaic | Hawthorne Boro | 14.1 | 6.4 | 11.8 |
| Boonton loam | Hudson | Harrison Town | 11 | 5 | 9.2 |
| Boonton (Peckmantown) loam | Essex | Newark City | 10.9 | 5 | 9.1 |
| Boonton loam | Union | Roselle Park | 12 | 5.5 | 10 |
| Boonton loam | Middlesex | Perth Amboy City | 11.5 | 5.3 | 9.6 |
| Boonton loam | Bergen | Ramsey Boro | 14.9 | 6.8 | 12.4 |
| Rockaway (Paxton) loam | Passaic | Ringwood Boro | 16.6 | 8.3 | 14.1 |
| Rockaway loam | Morris | Rockaway Twp. | 16.7 | 8.4 | 14.2 |
| Rockaway loam | Sussex | Franklin Boro | 16.4 | 8.2 | 13.9 |
| Annandale loam | Morris | Chester Twp. | 16.6 | 8.2 | 13.7 |

| **Soil Name** | **Primary Counties of Occurrence** | **Representative Municipality** | **Recharge (in/yr)** | | |
|---|---|---|---|---|---|
| | | | **Landscaped Open Space** | **Unvegetated** | **General Agriculture** |
| Annandale loam | Warren | Pohatcong Twp. | 12 | 6.6 | 9.9 |
| Annandale loam | Hunterdon | Tewksbury Twp. | 15.6 | 7.9 | 12.9 |
| Penn silt loam | Somerset | Hillsborough Twp. | 12.1 | 5.3 | 10.1 |
| Penn silt loam | Hunterdon | Delaware Twp. | 11.6 | 5.1 | 9.7 |

# APPENDICES

# APPENDIX A

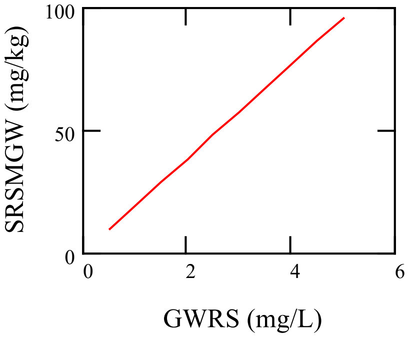
## Sensitivity of the Soil-Water Partition Equation to Modification of Component Parameters

The Department conducted a sensitivity analysis of the USEPA partition equation to determine the effects of modifying different equation parameters on the development of soil remediation standard. For this analysis, one variable was modified at a time, while the other chemical and environmental parameter values were set at default New Jersey values. Soil properties were varied within their normal ranges (USEPA, 1996a). The analysis was conducted in two phases. First, the sensitivity of Equation 1a and 1b was evaluated with respect to the organic carbon content, $K_{oc}$, Henry's law constant, ground water standard, the dilution-attenuation factor (*DAF*), soil moisture, soil air content, and soil bulk density. Second, the sensitivity of the *DAF* calculations (Equations 2 and 3) to the various parameters incorporated was evaluated. The examples below are for specific contaminants, but the observed sensitivities are the same for all contaminants.

1. Sensitivity of the soil remediation standard for the migration to ground water exposure pathway(SRS-MGW) to changes to the Ground Water Remediation Standard (GWRS). Results shown for xylene

| GWRS (mg/L) | SRSMGW (mg/kg) |
|---|---|
| 0.5 | 9.6 |
| 1 | 19.2 |
| 1.5 | 28.7 |
| 2 | 38.3 |
| 2.5 | 47.9 |
| 3 | 57.5 |
| 3.5 | 67.1 |
| 4 | 76.7 |
| 4.5 | 86.2 |
| 5 | 95.8 |

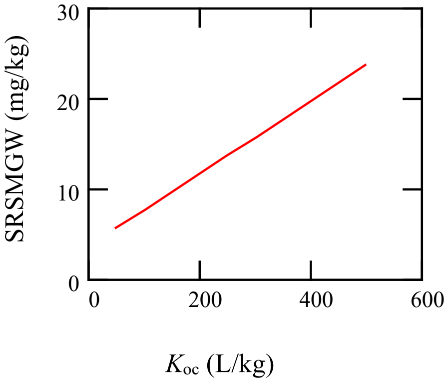
Sensitivity to GWRS is linear



2. Sensitivity of remediation standard (SRS-MGW) to changes to the ($K_{oc}$) soil organic carbon-water partition coefficient value. Results shown for xylene

| $K_{oc}$ (L/kg) | SRSMGW (mg/kg) |
|---|---|
| 50 | 5.7 |
| 100 | 7.7 |
| 150 | 9.7 |
| 200 | 11.7 |
| 250 | 13.7 |
| 300 | 15.7 |
| 350 | 17.7 |
| 400 | 19.7 |
| 450 | 21.7 |

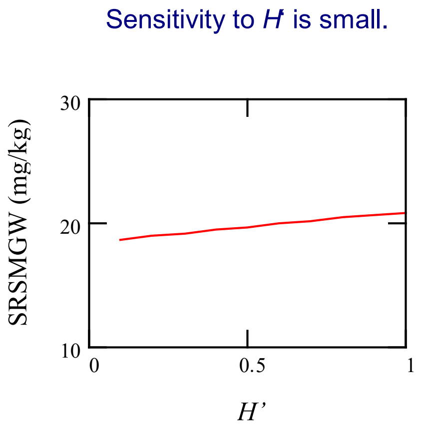
Sensitivity to $K_{oc}$ is linear.



| | |
|---|---|
| 500 | 23.7 |

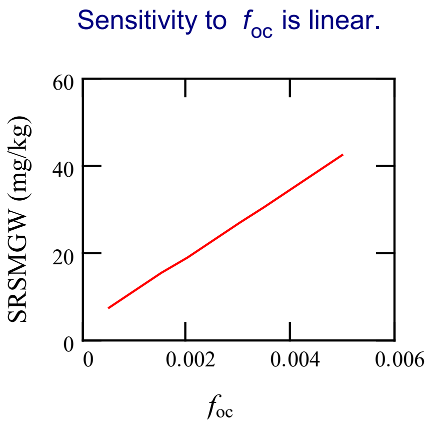
3. Sensitivity of remediation standard (SRS-MGW) to the Henry’s law constant (*H’*). Results shown for xylene

| *H'* | SRSMGW (mg/kg) |
|---|---|
| 0.1 | 18.7 |
| 0.2 | 19 |
| 0.3 | 19.2 |
| 0.4 | 19.5 |
| 0.5 | 17.9 |
| 0.6 | 20 |
| 0.7 | 20.2 |
| 0.8 | 20.4 |
| 0.9 | 20.7 |
| 1 | 20.9 |



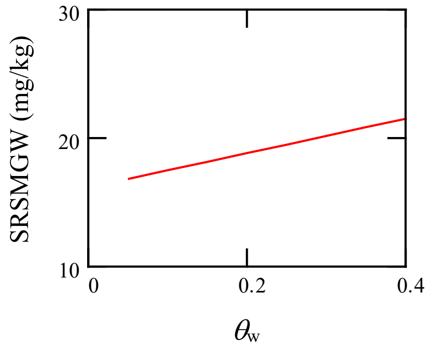
4. Sensitivity of remediation standard (SRS-MGW) to fraction organic carbon ($f_{oc}$). Results shown for xylene.

| $f_{oc}$ | SRSMGW (mg/kg) |
|---|---|
| 0.0005 | 7.6 |
| 0.001 | 11.4 |
| 0.0015 | 15.3 |
| 0.002 | 19.2 |
| 0.0025 | 23 |
| 0.003 | 26.9 |
| 0.0035 | 30.7 |
| 0.004 | 34.6 |
| 0.0045 | 38.5 |
| 0.005 | 42.3 |



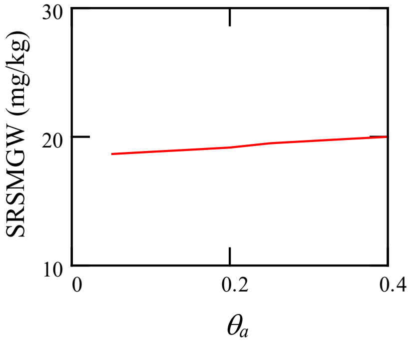
5. Sensitivity of remediation standard (SRS-MGW) to soil moisture ($\theta_w$)
Results shown for xylene.

| $\theta_w$ | SRSMGW (mg/kg) |
|---|---|
| 0.05 | 16.7 |
| 0.1 | 17.4 |
| 0.15 | 18.1 |
| 0.2 | 18.8 |
| 0.25 | 19.4 |
| 0.3 | 20.1 |
| 0.35 | 20.8 |
| 0.4 | 21.4 |


Sensitivity to $\theta_w$ is small.

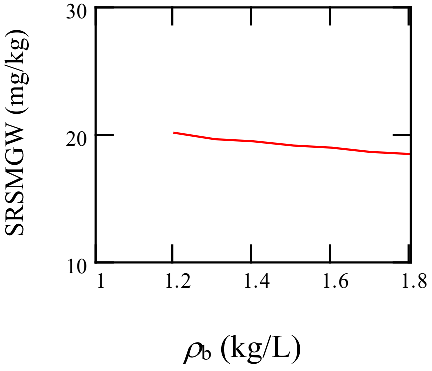
6. Sensitivity of remediation standard (SRS-MGW) to soil air content ($\theta_a$).
Results shown for xylene.

| $\theta_a$ | SRSMGW (mg/kg) |
|---|---|
| 0.05 | 18.7 |
| 0.1 | 18.9 |
| 0.15 | 19 |
| 0.2 | 19.2 |
| 0.25 | 19.4 |
| 0.3 | 19.6 |
| 0.35 | 19.8 |
| 0.4 | 20 |


Sensitivity to $\theta_a$ is small

7. Sensitivity of remediation standard (SRS-MGW) to soil bulk density ($\rho_b$)
Results shown for xylene.

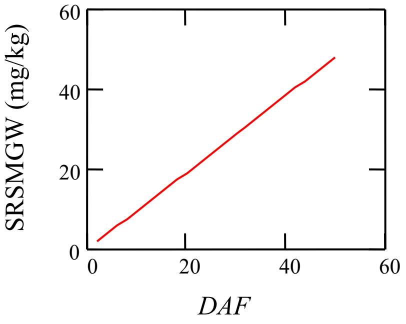
| $\rho_b$ (kg/L) | SRSMGW (mg/kg) |
|---|---|
| 1.2 | 20.1 |
| 1.3 | 19.7 |
| 1.4 | 19.4 |
| 1.5 | 19.2 |
| 1.6 | 18.9 |
| 1.7 | 18.7 |
| 1.8 | 18.5 |


Sensitivity to $\rho_b$ is small.

8. Sensitivity of remediation standard (SRS-MGW) to dilution-attenuation Factor (*DAF*). Results shown for xylene.

| *DAF* | SRSMGW (mg/kg) |
|---|---|
| 2 | 1.9 |
| 4 | 3.8 |
| 6 | 5.8 |
| 8 | 7.7 |
| 10 | 9.6 |
| 12 | 11.5 |
| 14 | 13.4 |
| 16 | 15.3 |
| 18 | 17.2 |
| 20 | 19.2 |
| 22 | 21.1 |
| 24 | 23 |
| 26 | 24.9 |
| 28 | 26.8 |
| 30 | 28.8 |
| 32 | 30.7 |
| 34 | 32.6 |
| 36 | 34.5 |
| 38 | 36.4 |
| 40 | 38.3 |
| 42 | 40.2 |
| 44 | 42.2 |
| 46 | 44 |
| 48 | 46 |
| 50 | 47.9 |

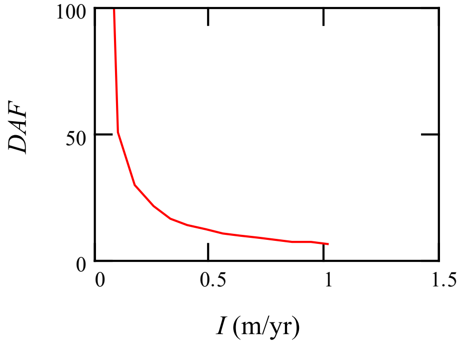
Sensitivity is linear with respect to *DAF*



9. Sensitivity of dilution-attenuation Factor (*DAF*) (and therefore the remediation standard (SRS-MGW)) to infiltration rate (*I*).

| *I* (m/yr) | *DAF* |
|---|---|
| 0.025 | 198 |
| 0.102 | 51 |
| 0.178 | 30 |
| 0.254 | 22 |
| 0.33 | 17 |
| 0.406 | 14 |
| 0.483 | 12 |
| 0.559 | 11 |
| 0.635 | 9.8 |
| 0.711 | 8.9 |
| 0.787 | 8.3 |
| 0.864 | 7.7 |
| 0.94 | 7.2 |
| 1.016 | 6.8 |

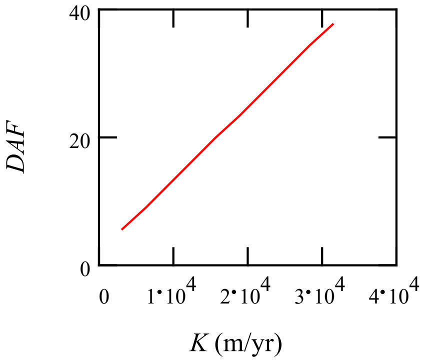
*DAF* (and SRSMGW) sensitivity is inversely proportional to infiltration rate, *I*, when mixing zone depth not constrained by aquifer thickness.



10. Sensitivity of dilution-attenuation factor (*DAF*) (and therefore the remediation standard (SRSMGW)) to hydraulic conductivity (*K*).

*DAF* (and SRSMGW) sensitivity is slightly less than linear with respect to conductivity, *K*. Mixing zone depth not constrained by aquifer thickness in this calculation.

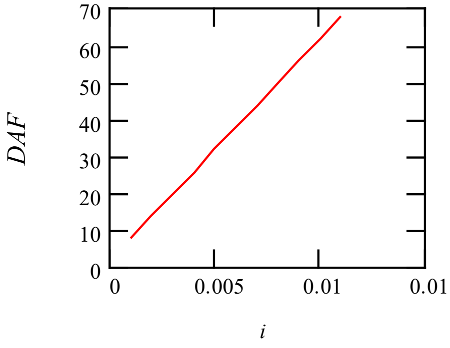
| *K* (m/yr) | *DAF* |
|---|---|
| 3155 | 6 |
| 6311 | 9 |
| 9467 | 13 |
| 12622 | 16 |
| 15778 | 20 |
| 18934 | 23 |
| 22089 | 27 |
| 25245 | 30 |
| 28401 | 34 |
| 31556 | 38 |



11. Sensitivity of dilution-attenuation factor (*DAF*) and therefore the remediation standard (SRSMGW) to gradient (*i*).

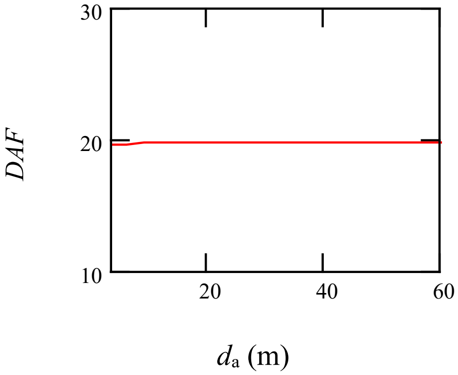
*DAF* (and SRSMGW) sensitivity is slightly less than linear with respect to gradient, *i*. Mixing zone depth not constrained by aquifer thickness in this calculation.

| *i* | *DAF* |
|---|---|
| 0.001 | 8 |
| 0.002 | 14 |
| 0.003 | 20 |
| 0.004 | 26 |
| 0.005 | 32 |
| 0.006 | 38 |
| 0.007 | 44 |
| 0.008 | 50 |
| 0.009 | 56 |
| 0.01 | 62 |
| 0.011 | 68 |



12. Sensitivity of dilution-attenuation factor (*DAF*) (and therefore the remediation standard for SRSMGW) to aquifer thickness ($d_a$).

When aquifer thickness is 3.4 m or greater, the aquifer thickness has no effect on *DAF* or the remediation standard, when using the default length for the Area of Concern parallel to the ground water flow (30.5 m)



13. Effect of size of area of concern on the remediation standard (SRSMGW).
    Results shown for xylene.

| *Remediation Standard for xylene as a function of the size of the area of concern (mg/kg)* | | | |
|---|---|---|---|
| | *Length of Site* | | |
| | *Parallel to GW flow (m)* | | |
| | *15.2* | *30.5* | *152* |
| Aquifer thickness = 3.5 m | 19 | 19 | 5 |
| Aquifer thickness = 15.2 m | 19 | 19 | 17 |

Under default conditions, a lower remediation standard results when the site length becomes large. However, this effect is reduced when the aquifer thickness increases.

# APPENDIX B

## Maps of Water Table Elevation and Hydraulic Conductivity Used in the Assessment of the Kirkwood-Cohansey Aquifer

Rancocas, Crosswicks, Assunpink, Blacks, and Crafts Creek Basins (43 gradient measurements and 1 aquifer test) (Watt et al. 2003)

Maurice and Cohansey River Basins (37 gradient measurements and 10 aquifer tests) (Charles et al., 2001)

Salem River and Raccoon, Oldman's, Alloway, and Stow Creek Basins (24 gradient measurements and 10 aquifer tests) (Johnson and Charles, 1997)

Toms River, Metedeconk River and Kettle Creek Basins (37 gradient measurements and 3 aquifer tests) (Watt et al. 1994)

Forked River and Cedar, Oyster, Mill, Westecunk, and Tuckerton Creek Basins and Adjacent Basins (11 gradient measurements and 2 aquifer tests) (Gordon, 2004).

Upper Maurice River Basin and Adjacent Areas (18 gradient measurements and 5 aquifer tests) (Lacombe and Rosman, 1995)

Mullica River Basin (47 gradient measurements and 6 aquifer tests) (Johnson and Watt, 1996).

Great Egg Harbor Basin (18 gradient measurements and 10 aquifer tests) (Watt and Johnson, 1992).

## References

Alexander, M. (1995). How Toxic are Chemicals in Soil? *Environ. Sci. Technol.* 29(11):2713-2717.

Bonazountas, M., Hetrick, D.M., Kostecki, P.T., and Calabrese, E.J. (1997). SESOIL in Environmental Fate and Risk Modeling. Bonazountas, M., Hetrick, D.M., Kostecki, P.T., and Calabrese, E.J. (eds.), Amherst Scientific Publishers, Amherst, MA.

Bonazountas, M. and Wagner, J.M. (1984). SESOIL: A Seasonal Soil Compartment Model, Draft Report. Office of Toxic Substances, U.S. Environmental Protection Agency: Washington, DC, PB86112406.

Brown, S.H., Alleman, D.W., Stewart, A., Andrew, L. and Hyde, B.G. (1996). The SPLP as an Alternative to TCLP Analysis for Metals: Connecticut Example Using Lead. In: Contaminated Soils, Vol. I, Calabrese, E.J., Kostecki, P.T. and Bonazountas, M., (eds.), Amherst Scientific Publishers, Amherst, MA, pp. 353-362.

Canace, R.J. and Sugarman, P.J. (2009). Kirkwood-Cohansey Water-table Aquifer, Information Circular, New Jersey Geological Survey, New Jersey Department of Environmental Protection, Trenton, NJ.

Carsel, R.F. and Parrish, R.S. (1988). Developing Joint Probability Distributions of Soil Water Retention Characteristics. *Water Resour. Res.* 24(5):755-769.

Carsel, R.F.; Parrish, R.S.; Jones, J.L.; Hansen, J.L and Lamb, R.L (1988). Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils.). *J. Contam. Hydrol.* 2: 111-124.

Charles, E.G., Behroozi, C., Schooley, J. and Hoffman, J.L. (1993). GSR-32: A Method for Evaluating Ground-Water-Recharge Areas in New Jersey. New Jersey Geological Survey, New Jersey Dept. of Environmental Protection: Trenton, New Jersey.

Charles, E.G., Storck, D.A., and Clawges, R.M. (2001). Hydrology of the unconfined aquifer system, Maurice River area: Maurice and Cohansey River basins, New Jersey, 1994-95: U.S. Geological Survey Water-Resources Investigations Report 01-4229, 5 sheets.

Devaull, G; Ettinger, R and Gustafson, J. (2002). Chemical Vapor Intrusion from Soil or Ground water to Indoor Air: Significance of Unsaturated Zone Biodegradation of Aromatic Hydrocarbons. *Soil and Sediment Contamination.* 11(4):625-641.

Eagleson, P.S. (1978). Climate, Soil and Vegetation, 1-7. *Water Resour. Res.* 14(5):705-776.

Fendorf, S., La Force, M.J. and Guangchao, L. (2004). Temporal Changes in Soil Partitioning and Bioaccessibility of Arsenic, Chromium, and Lead. *J. Environ. Qual.* 33:2049-2055.

Foth, H.D. (1984). Fundamentals of Soil Science, 7th ed. John Wiley and Sons: New York.

Gordon, A.D. (2004). Hydrology of the Unconfined Kirkwood-Cohansey Aquifer System, Forked River and Cedar, Oyster, Mill, Westecunk, and Tuckerton Creek Basins and Adjacent Basins in the Southern Ocean County Area, New Jersey 1998-99. Water-Resources Investigations Report 03-4337, U.S. Department of the Interior, U.S. Geological Survey, Washington, DC.

Hoffman, J.L. (1999). MS Excel Workbook Implementing the NJGS Ground-Water-Recharge Methodology. NJ Geological Survey Digital Geodata Series 99-2, revised 2009, Trenton, N.J., www.njgeology.org.

Howard, P.H., Boethling, R.S., Jarvis, W.F., Meylan, W.M. and Michalenko, E.M. (1991). Handbook of Environmental Degradation Rates. Lewis Publishers, Chelsea, Michigan

Johnson, M.L. and Charles, E.G. (1997). Hydrology of the Unconfined Aquifer System, Salem River Area: Salem River and Raccoon, Oldmans, Alloway and Stow Creek Basins, New Jersey, 1993-94. Water-Resources Investigations Report 96-4195, U.S. Geological Survey, West Trenton, NJ.

Johnson, M.L. and Watt, M.K. (1996). Hydrology of the Unconfined Aquifer System, Mullica River Basin, New Jersey, 1991-92. Water-Resources Investigations Report 94-4234, U.S. Geological Survey, West Trenton, NJ.

Lackovic, J.A., Nikolaidis, N.P., Chheda, P., Carley, R.J. and Patton, E. (1997). Evaluation of Batch Leaching Procedures for Estimating Metal Mobility in Glaciated Soils. *Ground Water Monitoring Review*. Summer, 1997. 231-240.

Lacombe, P.J. and Rosman, R. (1995). Hydrology of the Unconfined Aquifer System in the Upper Maurice River Basin and Adjacent Areas in Gloucester County, New Jersey, 1986-87. Water-Resources Investigations Report 92-4128, U.S. Geological Survey, West Trenton, NJ.

Lee, S.Z., Allen, H.E., Huang, C.P., Sparks, D.L., Sanders, P.F. and Peijnenburg, W.J.G.M.(1996). Predicting Soil-Water Partition Coefficients for Cadmium. *Environ. Sci. Technol.* 30(12): 3418-3424.

Lock, K. and Janssen, C.R. (2003). Influence of Aging on Metal Availability in Soils. *Rev. Environ. Contam. Toxicol.* 178:1-21.

Loehr, R.C. and Webster, M.T. (1996). Behavior of Fresh vs. Aged Chemicals in Soil. *J. Soil Contamination*. 5(4): 361-383.

Martin, M (1990). Ground-water flow in the New Jersey Coastal Plain: OpenFile Report 87-528, U.S. Geological Survey, Washington, DC.

Melancon, S.M., Pollard, J.E. and Hern, S.C. (1986). Evaluation of SESOIL, PRZM and PESTAN in a Laboratory Column Leaching Experiment. *Environ. Toxicol. Chem.* 5*:*865-878.

NJDEP (2012). Ground Water Technical Guidance: Site Investigation, Remedial Investigation, Remedial Action Performance Monitoring. New Jersey Department of Environmental Protection, Site Remediation Program, Trenton, NJ. Available at http://www.nj.gov/dep/srp/guidance/

Pavlostathis, S.G. and Mathavan, G.N. (1992). Desorption Kinetics of Selected Volatile Organic Compounds from Field Contaminated Soils. *Environ. Sci.Technol.* 26(3): 532-538.

Riley, R.G., Szecsody, J.E., Sklarew, D.S., Mitroshkov, A.V., Gent, P.M., Brown, C.F. and Thompson, C.J. (2010). Desorption Behavior of Carbon Tetrachloride and Chloroform in Contaminated Low Organic Carbon Aquifer Sediments. *Chemosphere* 79:807-813.

Sander, M., Lu, Y. and Pignatello, J.J. (2006). Conditioning-Annealing Studies of Natural Organic Matter Solids Linking Irreversible Sorption to Irreversible Structural Expansion. *Environ. Sci. Technol.* 40:170-178.

Sanders, P.F. (1995). Calculation of Soil Cleanup Criteria for Volatile Organic Compounds as Controlled by the Soil-to-Ground Water Pathway: Comparison of Four Unsaturated Zone Leaching Models. *J. Soil Contamination.* 4(1): 1-24.

Sanders, P.F. and Talimcioglu, N.M. (1997). Soil-to-Indoor Air Exposure Models for Volatile Organic Compounds: The Effect of Soil Moisture. *Environ. Toxicol. Chem.* 16 (12): 2597–2604.

Tedrow, J.F. (1986). Soils of New Jersey. R.E.Krieger: Malabar, FL.

USEPA (2002). Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites. US Environmental Protection Agency, Office of Solid Waste and Emergency Response: Washington, DC, OSWER 9355.4-24

USEPA (1996a). Soil Screening Guidance: Technical Background Document, May 1996. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response: Washington, DC, EPA/540/R-95/128

USEPA (1996b). Soil Screening Guidance: User's Guide, July 1996. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response: Washington, DC, EPA/540/R-96/018.

Volwinkel, E.F. and Foster, W.K. (1981). Hydrogeologic conditions in the Coastal Plain of New Jersey. Open-File Report 81-405, U.S. Department of the Interior, Geological Survey, Trenton, NJ

Watt, M.K. and Johnson, M.L. (1992). Water Resources of the Unconfined Aquifer System of the Great Egg Harbor River Basin, New Jersey, 1989-90. Water-Resources Investigations Report 91-4126, U.S. Geological Survey, West Trenton, NJ.

Watt, M.K., Johnson, M.L. and Lacombe, P.J. (1994). Hydrology of the Unconfined Aquifer System, Toms River, Metedeconk River, and Kettle Creek Basins, New Jersey, 1987-90. Water Resources Investigations Report 93-4110, U.S. Geological Survey, West Trenton, NJ.

Watt, M.K., Kane, A.C., Charles, E.G. and Storck, D.A. (2003). Hydrology of the Unconfined Aquifer System, Rancocas Creek Area: Crosswicks, Assunpink, Blacks, and Crafts Creek Basins, New Jersey, 1996. Water-Resources Investigations Report 02-4280, U.S. Department of the Interior, U.S. Geological Survey, Washington, DC.

Yeh, G.T. (1981). AT123D: Analytical Transient One-, Two-, and Three-Dimensional Simulation of Waste Transport in the Aquifer System. Publication No. 1439, Environmental Sciences Division, Oak Ridge National Laboratory: Oak Ridge, TN, ORNL-5602.

Yin, Y., Allen, H.E., Li, Y., Huang, C.P. and Sanders, P. (1996). Adsorption of Mercury(II) by Soil: Effects of pH, Chloride and Organic Matter. *J. Environ. Qual.* 25(4): 837-844.