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MEMORANDUM

TO: Carl Will

FROM: Kirby Olson *KO*

RE: **Cleanup levels for PNM's Person Station**

DATE: April 7, 2000

In reference to your question about whether an EPA SSL level would be different for regions which have different soil PRGs (Region 9 and Region 6), I've attached some excerpts from the EPA SSL technical background document. For both organic and inorganic constituents, the target levels for groundwater are calculated from an MCL or MCLG for drinking water, so they should differ only if the drinking water standards differ. You can see that the DAF factor is also a term in this equation, so if the two regions show the same level to each other for a DAF of one; they will show the same level to each other for a DAF of 20.

Cc:

thus

$$M_a = C_w H' \theta_a V_{sp} \quad (19)$$

Substituting into Equation 13:

$$C_t = \frac{C_s \rho_b + C_w \theta_w + C_w H' \theta_a}{\rho_b} \quad (20)$$

or

$$C_s = C_t - C_w \left(\frac{\theta_w + \theta_a H'}{\rho_b} \right) \quad (21)$$

Substituting into Equation 12 and rearranging:

Soil-Water Partition Equation for Migration to Ground Water Pathway: Inorganic Contaminants

$$C_t = C_w \left(K_d + \frac{\theta_w + \theta_a H'}{\rho_b} \right) \quad (22)$$

Parameter/Definition (units)	Default	Source
C_t /screening level in soil (mg/kg)	-	-
C_w /target soil leachate concentration (mg/L)	(nonzero MCLG, MCL, or HBL) \times 20 DAF	Table 1 (nonzero MCLG, MCL); Section 2.5.6 (DAF for 0.5-acre source)
K_d /soil-water partition coefficient (L/kg)	chemical-specific	see Part 5
θ_w /water-filled soil porosity (L _{water} /L _{soil})	0.3 (30%)	U.S. EPA/ORD
θ_a /air-filled soil porosity (L _{air} /L _{soil})	0.13	$n - \theta_w$
n /total soil porosity (L _{pore} /L _{soil})	0.43	$1 - \rho_b/\rho_s$
ρ_b /dry soil bulk density (kg/L)	1.5	U.S. EPA, 1991b
ρ_s /soil particle density (kg/L)	2.65	U.S. EPA, 1991b
H' /dimensionless Henry's law constant	$H \times 41$, where 41 is a conversion factor	U.S. EPA, 1991b
H /Henry's law constant (atm-m ³ /mol)	chemical-specific	see Part 5

Equation 22 is used to calculate SSLs (total soil concentrations, C_t) corresponding to soil leachate concentrations (C_w) equal to the target contaminant soil leachate concentration. The equation assumes that soil water, solids, and gas are conserved during sampling. If soil gas is lost during sampling, θ_a should be assumed to be zero. Likewise, for inorganic contaminants except mercury, there is no significant vapor pressure and H' may be assumed to be zero.

The User's Guide (U.S. EPA, 1996) describes how to develop site-specific estimates of the soil parameters needed to calculate SSLs. Default soil parameter values for the partition equation are the same as those used for the VF equation (see Section 2.4.2) except for average water-filled soil porosity (θ_w). A conservative value (0.15) was used in the VF equation because the model is most sensitive to this parameter. Because migration to ground water SSLs are not particularly sensitive to soil water content (see Section 2.5.7), a value that is more typical of subsurface conditions (0.30) was used. This value is between the mean field capacity (0.20) of Class B soils (Carsel et al., 1988) and the saturated volumetric water content for loam (0.43).

K_d varies by chemical and soil type. Because of different influences on K_d values, derivations of K_d values for organic compounds and metals were treated separately in the SSL methodology.

2.5.2 Organic Compounds—Partition Theory. Past research has demonstrated that, for hydrophobic organic chemicals, soil organic matter is the dominant sorbing component in soil and that K_d is linear with respect to soil organic carbon content (OC) as long as OC is above a critical level (Dragun, 1988). Thus, K_d can be normalized with respect to soil organic carbon to K_{oc} , a chemical-specific partitioning coefficient that is independent of soil type, as follows:

$$K_d = K_{oc} f_{oc} \quad (23)$$

where

K_{oc} = organic carbon partition coefficient (L/kg)
 f_{oc} = fraction of organic carbon in soil (mg/mg)

Substituting into Equation 22:

Soil-Water Partition Equation for Migration to Ground Water Pathway: Organic Contaminants

$$C_t = C_w \left((K_{oc} f_{oc}) + \frac{\theta_w + \theta_a H'}{\rho_b} \right) \quad (24)$$

Parameter/Definition (units)	Default	Source
C_t /screening level in soil (mg/kg)	—	—
C_w /target leachate concentration (mg/L)	(nonzero MCLG, MCL, or HBL) \times 20 DAF	Table 1 (MCL, nonzero MCLG); Section 2.5.6 (DAF for a 0.5-acre source)
K_{oc} /soil organic carbon-water partition coefficient (L/kg)	chemical-specific	see Part 5
f_{oc} /organic carbon content of soil (kg/kg)	0.002 (0.2%)	Carsel et al., 1988
θ_w /water-filled soil porosity (L_{water}/L_{soil})	0.3 (30%)	U.S. EPA/ORD
θ_a /air-filled soil porosity (L_{air}/L_{soil})	0.13	$n - \theta_w$
n /total soil porosity (L_{pore}/L_{soil})	0.43	$1 - \rho_b/\rho_s$
ρ_b /dry soil bulk density (kg/L)	1.5	U.S. EPA, 1991b
ρ_s /soil particle density (kg/L)	2.65	U.S. EPA, 1991b
H' /dimensionless Henry's law constant	$H \times 41$, where 41 is a conversion factor	U.S. EPA, 1991b
H /Henry's law constant ($atm \cdot m^3/mol$)	chemical-specific	see Part 5

Part 5 of this document provides K_{oc} values for organic chemicals and describes their development.

The critical organic carbon content, f_{oc}^* , represents OC below which sorption to mineral surfaces begins to be significant. This level is likely to be variable and to depend on both the properties of the soil and of the chemical sorbate (Curtis et al., 1986). Attempts to quantitatively relate f_{oc}^* to such properties have been made (see McCarty et al., 1981), but at this time there is no reliable method for estimating f_{oc}^* for specific chemicals and soils. Nevertheless, research has demonstrated that, for volatile halogenated hydrocarbons, f_{oc}^* is about 0.001, or 0.1 percent OC, for many low-carbon soils and aquifer materials (Piwoni and Banerjee, 1989; Schwarzenbach and Westall, 1981).

If soil OC is below this critical level, Equation 24 should be used with caution. This is especially true if soils contain significant quantities of fine-grained minerals with high sorptive properties (e.g., clays). If sorption to minerals is significant, Equation 24 will underpredict sorption and overpredict contaminant concentrations in soil pore water. However, this f_{oc}^* level is by no means the case for all soils; Abdul et al. (1987) found that, for certain organic compounds and aquifer materials, sorption was linear and could be adequately modeled down to $f_{oc} = 0.0003$ by considering K_{oc} alone.

For soils with significant inorganic and organic sorption (i.e., soils with $f_{oc} < 0.001$), the following equation has been developed (McCarty et al., 1981; Karickhoff, 1984):

$$K_d = (K_{oc} f_{oc}) + (K_{io} f_{io}) \quad (25)$$

where

$$\begin{aligned} K_{io} &= \text{soil inorganic partition coefficient} \\ f_{io} &= \text{fraction of inorganic material} \\ f_{io} + f_{oc} &= 1. \end{aligned}$$

4. **Adsorption is reversible.** The methodology assumes that desorption processes operate in the same way as adsorption processes, since most of the K_{oc} values are measured by adsorption experiments rather than by desorption experiments. In actuality, desorption is slower to some degree than adsorption and, in some cases, organics can be irreversibly bound to the soil matrix. In general, the significance of this effect increases with K_{ow} .

This assumption is **conservative**. Slower desorption rates and irreversible sorption will result in lower pore-water concentrations than that predicted by the methodology. Again, the level of knowledge on desorption processes is not sufficient to consider desorption kinetics and degree of reversibility for all of the subject chemicals.

2.5.5 Dilution/Attenuation Factor Development. As contaminants in soil leachate move through soil and ground water, they are subjected to physical, chemical, and biological processes that tend to reduce the eventual contaminant concentration at the receptor point (i.e., drinking water well). These processes include adsorption onto soil and aquifer media, chemical transformation (e.g., hydrolysis, precipitation), biological degradation, and dilution due to mixing of the leachate with ambient ground water. The reduction in concentration can be expressed succinctly by a DAF, which is defined as the ratio of contaminant concentration in soil leachate to the concentration in ground water at the receptor point. When calculating SSLs, a DAF is used to backcalculate the target soil leachate concentration from an acceptable ground water concentration (e.g., MCLG). For example, if the acceptable ground water concentration is 0.05 mg/L and the DAF is 10, the target leachate concentration would be 0.5 mg/L.

The SSL methodology addresses only one of these dilution-attenuation processes: contaminant dilution in ground water. A simple equation derived from a geohydrologic water-balance relationship has been developed for the methodology, as described in the following subsection. The ratio factor calculated by this equation is referred to as a dilution factor rather than a DAF because it does not consider processes that attenuate contaminants in the subsurface (i.e., adsorption and degradation processes). This simplifying assumption was necessary for several reasons.

First, the infinite source assumption results in all subsurface adsorption sites being eventually filled and no longer available to attenuate contaminants. Second, soil contamination extends to the water table, eliminating attenuation processes in the unsaturated zone. Additionally, the receptor well is assumed to be at the edge of the source, minimizing the opportunity for attenuation in the aquifer. Finally, chemical-specific biological and chemical degradation rates are not known for many of the SSL chemicals; where they are available they are usually based on laboratory studies under simplified, controlled conditions. Because natural subsurface conditions such as pH, redox conditions, soil mineralogy, and available nutrients have been shown to markedly affect natural chemical and biological degradation rates, and because the national variability in these properties is significant and has not been characterized, EPA does not believe that it is possible at this time to incorporate these degradation processes into the simple site-specific methodology for national application.

If adsorption or degradation processes are expected to significantly attenuate contaminant concentrations at a site (e.g., for sites with deep water tables or soil conditions that will attenuate contaminants), the site manager is encouraged to consider the option of using more sophisticated fate and transport models. Many of these models can consider adsorption and degradation processes and can model transient conditions necessary to consider a finite source size. Part 3 of this document presents information on the selection and use of such models for SSL application.