

TITLE: Assessment of an Aquitard during a Ground Water Contamination Investigation

DATE: November, 2009.

SUBJECT: Guidance for evaluating whether an aquitard adequately protects underlying ground water when an overlying ground water zone is contaminated.

BACKGROUND:

When investigating contaminated ground water, aquitards typically need to be evaluated. Aquitards are low hydraulic conductivity geologic deposits that help protect ground water resources from contamination by impeding downward flow. Aquitards can decrease the susceptibility of underlying ground water to contamination by increasing both time of travel and the flow path distance from contaminated overlying ground water sources. However, no geologic formation or stratum is completely impermeable; some ground water flows across even the least permeable aquitards, given sufficient time. An assessment of the degree to which an aquitard can restrict the movement of ground water and contaminants is critical when evaluating the susceptibility of ground water underlying an aquitard.

Situations where evaluation of an aquitard may be needed include, but are not limited to:

- Evaluating areas of known or suspected soil and ground water contamination
- Siting new drinking water wells
- Designing a monitoring well network
- Designing or planning a ground water remediation system

GUIDANCE:

This guidance is designed for users who may need to monitor and/or investigate ground water zones beneath a contaminated ground water zone. It is designed for a reader who has an understanding of hydrogeology and contaminant transport. There is no “cookbook” or “one size fits all” approach for determining whether an aquitard provides adequate protection. The user should collect pertinent data and make a “weight-of-evidence” decision.

When evaluating the protective capabilities of an aquitard, the hydrogeologic characteristics and the contaminant characteristics should be considered. In general, some or all of the information listed below should be evaluated. Regardless of the conclusion regarding an aquitard’s protectiveness, ground water sampling from underlying ground water zones may be required. The user should contact the Ohio EPA division having regulatory authority.

1. **Material type and hydraulic conductivity.** The degree to which an aquitard protects underlying ground water resources depends on the vertical hydraulic conductivity, which is largely controlled by the type of material. The classification of

the material composing the aquitard (e.g., shales, clays, silty clays) can be determined by visual observation of geologic borings samples, tests pits, trenches or through laboratory analyses of soil or rock samples. Vertical hydraulic conductivity (K_v) is generally determined by laboratory analysis of undisturbed samples (i.e., Shelby tube) or by *in-situ* techniques, such pumping tests. Horizontal hydraulic conductivity may affect the K_v in an aquitard as it may influence the transport of water to fractures. The number of borings and laboratory samples needed depends on the geologic heterogeneity of the aquitard.

In general, clays and silty clays with low vertical hydraulic conductivity (e.g., $>1 \times 10^{-07}$ cm/sec) will more effectively protect underlying ground water than sands and gravels ($K_v \geq 10^{-04}$ cm/sec). Competent shale or some other relatively impermeable bedrock may also effectively isolate underlying ground water.

2. **Thickness.** The aquitard thickness is generally determined from geologic borings. Geophysical techniques, such as seismic refraction, resistivity, and borehole gamma-ray logs, also may be useful. In general, aquitards that are tens of feet thick protect better than thinner, low-permeability zones.

A 30-foot thick clay zone is often considered sufficient to protect underlying ground water. However, evaluating the degree of protection should not be based solely on thickness. Other criteria, such as material type, vertical and lateral discontinuities, presence/absence of hydraulically active fractures, and contaminant characteristics and concentrations should also be considered.

3. **Lateral continuity.** Aquitards that are laterally continuous generally provide better protection. Lateral continuity of an aquitard may be compromised by permeable zones formed by variations in material (e.g., silty clay with interbedded sand layers) or by structural discontinuities where the aquitard was not deposited or has been eroded, joint systems/fractures, or breaks caused by man-made structures such as water supply wells.

Lateral continuity is determined from a sufficient number of geologic borings. Geophysical methods may also be useful. The presence of discontinuous interbedded sands or permeable zones or fractures may provide conditions for contaminant migration. Regional hydrogeologic data and information from adjacent sites may provide helpful information as to the scale of the aquitard's lateral continuity.

4. **Secondary porosity features.** Secondary porosity features include, but are not limited to, fractures, tree roots, and animal burrows. An evaluation of these breaks is especially important when the ground water zone is shallow. However, such features can also be associated with ancient soil/weathering zones found deeper in the stratigraphic sequence. Fractures and other natural breaks can be identified through observation of soil/rock cores or excavations (tests pits and trenches).

Evaluating whether or not hydraulically active fractures provide contamination migration conduits is important. Information on the distribution of fractures in Ohio

can be found in Brockman and Szabo (2000) and Allred (2000). In general, a lower probability of fractures exists when (Cherry et al. 2006, and Bradbury et al. 2006):

- The aquitard is 50 to 100s of feet thick.
- There are few or no visible fractures in cores, boreholes, and outcrops.
- The aquitard matrix is highly plastic.
- Field and laboratory hydraulic conductivity are about the same.
- There is little response across the aquitard to pumping or recharge.
- Vertical gradient across the aquitard is high (>1)
- Penetration of tracers is less than a few feet.

Additional information pertaining to evaluating for fractures can be found in TGM Chapter 3 (Ohio EPA, 2006a), Cherry et al. (2006), and Bradbury et al. (2006).

5. **Man-made conduits.** Aquitard effectiveness may also be reduced through conduits created by man-made structures including, but not limited to, inadequately sealed (grouted) or improperly decommissioned water supply wells, monitoring wells, or oil/gas wells; water supply wells that are screened across multiple ground water zones; dry wells; building or bridge pilings; or hydraulic elevator cylinders. Conduits can also be created by excavation and/or construction activities.

Water supply well records (logs) are available from [ODNR-Division of Water](#) and local health departments; these should be carefully reviewed and field-verified. In addition, man-made conduits may be identified during site walkovers or through reviewing historic records, e.g., Sanborn Fire Insurance Maps, building plans, ODNR Division of Mineral Resources Management (oil and gas well) records, etc.

6. **Hydraulic gradients.** Vertical hydraulic gradient affects how a contaminant moves in the subsurface. It is determined by evaluating differences in static water elevations (hydraulic heads) in well/piezometer clusters. Upward gradients can limit the migration of some contaminants. However, contaminants still can move downward by molecular diffusion in lower permeability zones even when the hydraulic gradient is upward. Also, dense non-aqueous liquids (DNAPLs) can move downward (particularly through fractures) when the hydraulic gradient is upward.

A downward gradient does not itself prove that an aquitard is ineffective at protecting underlying ground water. Interconnection between zones above and below an aquitard could be assessed by pumping tests. However, prior to conducting pumping tests the potential to draw contaminants downward into an underlying clean ground water should be considered.

7. **Attenuation capacity.** Attenuation and retardation within the aquitard can occur as a contaminants move through it. These processes are determined by the geologic material and contaminant types and characteristics.

Characteristics of the aquitard that are useful for evaluating attenuation and retardation include: bulk density, soil pH, mineral content, fraction of organic carbon,

and cation exchange capacity. These parameters are described in more detail in TGM Chapters 3 and 14 (Ohio EPA, 2006a and 2007a). Characteristics of contaminants are described below.

8. **Physical/chemical characteristics of contaminants.** The mobility of contaminants through an aquitard depends on whether the contaminant is free phase or dissolved in the pore water, the degree to which it can be sorbed onto particles or dissolved in the pore water, and its duration/persistence in the subsurface.

Relative solubility controls whether a contaminant exists in ground water primarily as a dissolved (soluble) or free liquid phase (insoluble). Movement of the dissolved phase is generally in the direction of ground water flow and is governed primarily by advection-dispersion and biological/chemical attenuation.

The free phase is referred to as a non-aqueous phase liquid (NAPL). The movement of NAPL is influenced by the fluid density. Fluid density is defined as the mass of fluid per unit volume (g/cm^3). If a contaminant is more dense than ground water ($> 1\text{cm}^3/\text{gm}$), it tends to sink and may accumulate as a dense non-aqueous phase liquid (DNAPL). Conversely, a contaminant less dense tends to remain in the upper portions of the first saturated zone as a light non-aqueous phase liquid (LNAPL). Most LNAPLs in ground water are petroleum hydrocarbons (e.g., oils and fuels). Most DNAPLs are chlorinated compounds (e.g., carbon tetrachloride, tetrachloroethene, polychlorinated biphenyls), or creosote (U.S. EPA, 1993; Cherry et al., 2000, and Bradbury et al. 2006). The first ground water zone is susceptible to both LNAPL and DNAPL contamination. Deeper ground water zones are also susceptible to DNAPL contamination because DNAPLs sink even when the flow is generally horizontal.

Whether a contaminant tends to be sorbed onto soil is largely dependent on its Soil Organic Carbon-Water Partitioning Coefficient (K_{oc}) and/or Distribution Coefficient (K_d).

The K_{oc} is the ratio of the mass of a chemical that is adsorbed in the soil per unit mass of organic carbon in the soil per the equilibrium chemical concentration in solution. K_{oc} values are useful in predicting the mobility of organic soil contaminants; higher K_{oc} generally correlates to less mobile chemicals, while lower K_{oc} correlates to greater mobility. Organic compounds with $K_{oc} > 2000 \text{ ml}/\text{gm}$ are not very mobile (e.g., polynuclear aromatic hydrocarbons). K_{oc} values for a given constituent may vary greatly in the literature. The K_d is the ratio of a chemical's sorbed concentration (mg/kg) to the dissolved concentration (mg/L) at equilibrium. For organic compounds, a K_d may be calculated by multiplying the K_{oc} by the fraction of organic carbon (f_{oc}):

$$K_d = K_{oc} \times f_{oc}$$

Depending on the regulatory program, Ohio EPA will generally accept the values for K_{oc} and K_d listed in Appendix A, Table A-5 of the [Division of Hazardous Waste Management, Closure Review Guidance](#) (Ohio EPA, 2009), or the Division of Emergency and Remedial Response, Voluntary Action Program,

[Support Document for Development of Generic Numeric Standards and Risk Assessment \(Ohio EPA, 2008\)](#). However, the coefficients presented are not applicable where mobilization is from enhanced solvation - the condition under which solubilities of a contaminant is increased due to the presence of organic solvents. The K_d values presented for the metals assume relatively dilute solution conditions, a narrow range of soil moisture content is applicable, and consistent range of soil organic matter is present. If these basic assumptions are not met, site-specific determination of the leaching of inorganic substances is warranted.

Acceptable values for K_d for some metals can be found in Table 3 of Ohio EPA Division of Hazardous Waste Management, [Vadose Zone Modeling for RCRA Closure](#) (Ohio EPA, 2005).

9. **Presence of conditions that may change the mobility of contaminants.** If conditions exist (both natural and anthropogenic) that may cause contaminants to become more mobile, then leaching may become more probable. For example, natural outcrops of coal or pyritic shale may lower the pH of the environment and mobilize some metals. Similarly, sites impacted by acid spills may also exhibit lower pH conditions that enhance the mobility of certain metals and other contaminants.
10. **Persistence.** Degradation of contaminants in the environment can be biotic (biologically mediated) or abiotic (chemical reaction). Degradation can account for the loss of a pollutant and the formation of daughter products. Whether degradation is occurring (or not) needs to be evaluated. If an assumption that degradation is occurring is not justified and supported by field data, concentrations of the parent compound may be underestimated. Likewise, if degradation is occurring, but not accounted for, daughter products may not be properly addressed. A particularly insidious biodegradation problem is the formation of vinyl chloride from tetrachloroethene→trichloroethene→dichloroethene. Vinyl chloride is both more toxic and more mobile than its parent compounds.

Literature values for biodegradation rates vary greatly and often are based on laboratory testing or under field conditions where the factors affecting biodegradation can be controlled. The *Committee on In Situ Bioremediation*¹ recommended that the effectiveness of intrinsic bioremediation should be continually monitored by analyzing the fate of the contaminants and other reactants and products indicative of bioremediation. This monitoring includes three types of information: documented loss of contaminants, laboratory assays showing that the microorganisms have the potential to transform contaminants under the expected site conditions, and confirming evidence that the biodegradation potential is actually attained in the field. Additional information can be found in ASTM E1943-98 (2004), US EPA (1998), Interstate Technology Research Council (ITRC, 1999), and Natural Resource Council (NRC, 2000). When determining a biodegradation rate, the age of the contaminant

¹ The *Committee on In Situ Bioremediation* was established in 1992 with the task of developing guidelines for evaluating in-situ bioremediation projects and determining whether they are meeting or will meet clean-up goals. It represents the span of groups involved in bioremediation: buyers of bioremediation services, bioremediation contractors, environmental regulators, and academic researchers.

(or date of release) should be considered. An extremely short half life is inconsistent with a very old plume. The user should consult with the appropriate Ohio EPA regulatory program to determine whether literature values of degradation are acceptable and, if so, how they can be applied.¹

11. **Estimation of ground water travel time and contaminant migration.** Time of travel through an aquitard can affect the vulnerability of underlying ground water. Ground water travel time is calculated based on the average linear velocity, which is determined by:

$$v = \frac{K}{n_e} \times \frac{\partial h}{\partial l} \text{ where:}$$

- v = average linear velocity (length/time)
- K = hydraulic conductivity (length/time)
- n_e = effective porosity (volume/volume)
- ∂h/∂l = hydraulic gradient (length/length)

This equation and the parameters are discussed in Chapter 3 of the TGM (Ohio EPA 2006a). For vertical flow through an aquitard, the hydraulic gradient is often assumed to be one (1).

Contaminant migration rate is often estimated using the above velocity equation. However, this assumes that the dissolved solute travels at the average linear ground water velocity. The rate of advancement of a dissolved contaminant can be substantially different, as it is affected by adsorption/desorption, precipitation, oxidation, and biodegradation. Mobility can also be affected by the ratio of the size of the contaminant molecule to the pore channel size or the proportion of clays present. Hydrodynamic dispersion can even cause a contaminant to move faster than the average linear ground water velocity. Dispersion affects all solutes, whereas adsorption, chemical reactions, and biodegradation affect specific constituents at different rates². Therefore, a release from a contaminant source that contains multiple constituents can result in individual constituents moving at different rates.

12. **Contaminant release.** Historical information concerning the initial contaminant source, if known, needs to be evaluated. Contaminant migration can be influenced by whether the soil contamination was caused by a release of free product that may float or sink or by disposal of contaminated solid material or waste. The age of the release may also be important. For example, a weight-of-evidence demonstration

¹ For RCRA Closures, the Division of Hazardous Waste Management will not accept literature values for biodegradation of organic chemicals. If biodegradation rates are included in a model, site-specific data, including the methods used, number of samples, and laboratory data reports must be supplied to verify these inputs. For DERR/VAP properties, see [Technical Decision Guidance \(TDG\)](#) document VA [30007.09.021](#) (Ohio EPA, 2009).

² See TGM Chapter 5 (Ohio EPA, 2007b) for additional explanation on how these parameters influence ground water flow paths.

that deeper ground water zones are protected may be possible if there is evidence that soil contamination has been present for 50 years, is totally sequestered in the shallow soils, there has not been recent releases to the subsurface, and the underlying ground water is not contaminated, then that may provide a weight-of-evidence argument regarding impact to deep ground water.

Concentrations of contaminants within an overlying ground water zone are important when evaluating potential impact to an underlying ground water zone. If the concentrations are low, then simple analytical equations may be sufficient to show that ground water underlying the aquitard would likely be protected.

Sampling of soil and underlying ground water may indicate that the contaminants are sequestered at a shallow depth and concentrations at depth are significantly less. However, this by itself is not sufficient to indicate that the contaminants would not migrate to the deeper ground water. Multiple sampling events over time and/or evidence of when the initial release occurred would also need to be evaluated.

13. **Ground water use.** While a high likelihood of extensive current and/or potential ground water use below the contaminated zone does not by itself indicate there is a potential for contamination to reach the ground water zone, it may dictate that a higher degree of conservatism is needed when evaluating whether an aquitard adequately protects underlying ground water. Situations warranting a higher level of protection include locations over a drinking water source water protection area, a Sole Source Aquifer, or a 100 gallons per minute (gpm) unconsolidated aquifer, or when the water is being used within ½ mile. Information on Sole Source Aquifers and drinking water source water protection areas may be obtained from Ohio EPA, [Division of Drinking and Ground Waters, Ohio's Source Water Protection Program](#)).

Ground water use can be evaluated by obtaining well logs from [Ohio Department of Natural Resources, Division of Water](#) and local health departments. Also, the Voluntary Action Program [Technical Decision Guidance \(TDG\)](#) document [VA3001.09.005](#) provides further guidance on evaluating ground water use (Ohio EPA, 2009).

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