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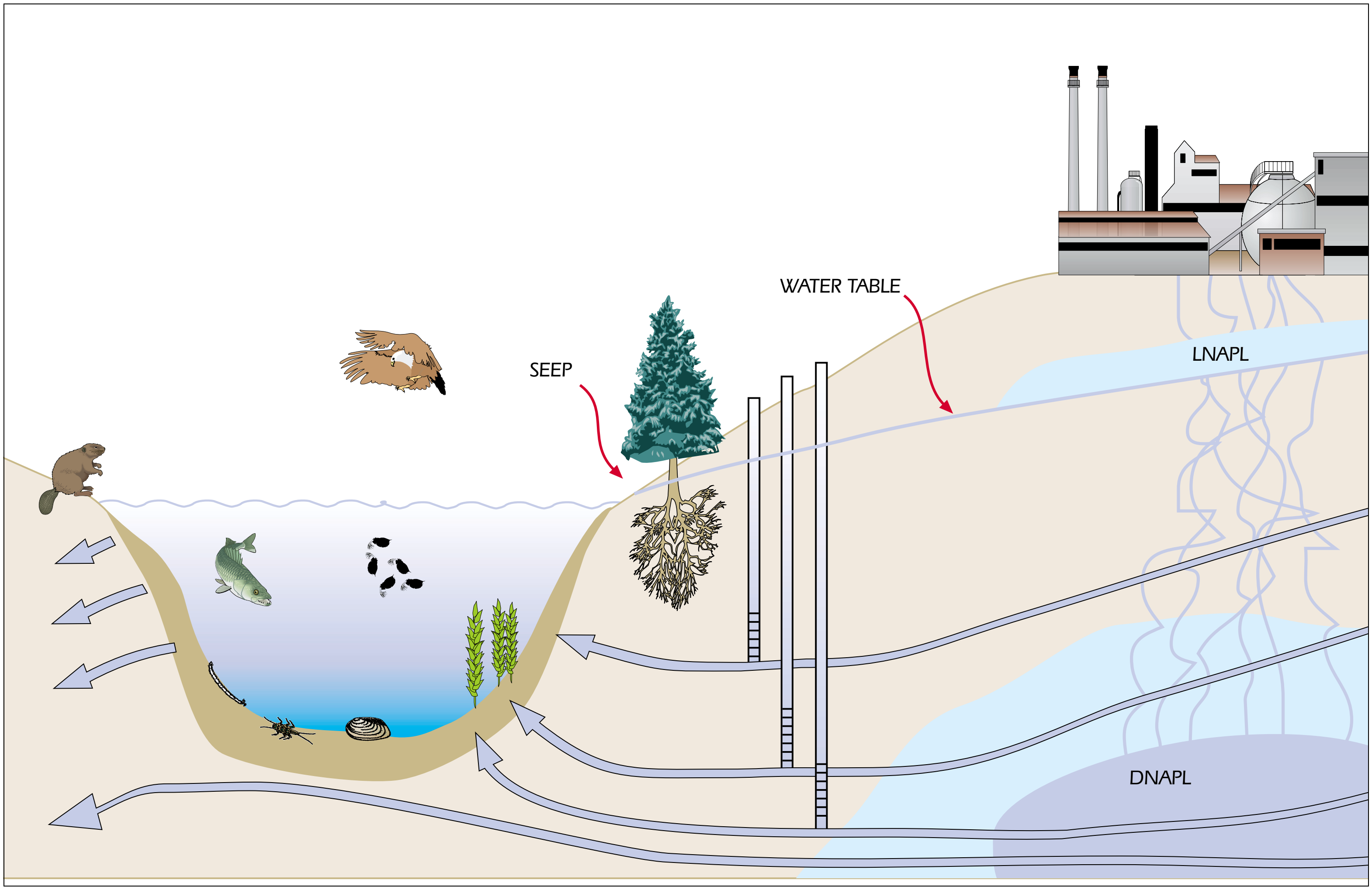
Proceedings of the Ground-Water/ Surface-Water Interactions Workshop



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INTRODUCTION

Although ground water and surface water are usually evaluated as separate water masses, they are connected by the ground-water/surface-water transition zone[1] in a hydrologic continuum. Understanding contaminant fate and transport in this zone is important to the U.S. Environmental Protection Agency's (EPA's) hazardous waste site cleanup programs across the nation because about 75% of RCRA and Superfund sites are located within a half mile of a surface water body, and almost half of all Superfund sites have impacted surface water. Investigations of ground water and surface water need to be integrated and incorporate recent advances in investigative techniques.

Ecological risk assessments for surface water bodies have all too often focused on the water column (where the ground-water contaminant plumes become extremely diluted), or on the sediments. Typically there has been little or no evaluation of contaminated ground-water discharges. Impacts from the discharge of contaminated ground water on the transition zone ecosystem have been ignored, even though this ecosystem provides important ecological services and is the most exposed to ground-water contaminants. Based on these considerations, the need to evaluate the transition zone is clear.

To address the technical concerns related to ecological impacts in the transition zone, the EPA's Office of Solid Waste and Emergency Response (OSWER) sponsored a workshop in January 1999, which was planned jointly by the Ecological Risk Assessment Forum and the Ground Water Forum.[2] The workshop was organized around answering two fundamental questions:

- How important is the transition zone ecologically?
- How can we measure hydrogeological, chemical, and biological conditions and changes in this zone?

There was a consensus among workshop participants that protecting this zone is important, and that there is a need for studies by interdisciplinary teams to ensure that valid data are obtained from the correct locations and at the right times so that valid conclusions are reached. Both forums plan to use the workshop information to submit research recommendations to EPA's Office of Research and Development, develop a list of suggested tools for investigating hydrogeological fate and transport and ecological effects at contaminated sites, develop Agency guidance, and conduct a pilot study using this methodology. The workshop and these proceedings provide a first step to understanding the fundamentals of evaluating the effects of contaminated ground water discharging through the transition zone.

WORKSHOP GOALS

The overall goal of the workshop was to provide an opportunity for individuals from various scientific and technical backgrounds to discuss the importance of the ground-water/surface-water transition zone and help regulators better understand environmental issues relating to the connections

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[1] In these proceedings, the authors may use terms other than "ground-water/surface-water transition zone" to indicate this zone of transition. These terms may be equivalent (e.g., ground-water/surface water interface) or more restrictive (e.g., hyporheic zone, which refers to the interface between ground water and lotic (moving) surface waters.)

[2] The Ecological Risk Assessment Forum and Ground Water Forum comprise ecological risk assessment and ground-water specialists, respectively, from EPA's Regional Offices, Headquarters, and Office of Research and Development. These forums help the EPA maintain consistency and develop national program guidance.

between ground water and surface water. Within this broad goal, the Ecological Risk Assessment Forum and Ground Water Forum had the following additional specific goals:

Ecological Risk Assessment Forum Goals:

- Develop a conceptual model for use in ecological risk assessment at sites where contaminated ground water discharges to surface water.
- Integrate structural, functional, and hydrogeological components and methods for evaluating changes to the ecosystem.

Ground Water Forum Goals:

- Increase awareness of new tools used to evaluate fate and transport within the transition zone.
- Identify and understand geological, hydrological, and chemical factors that might influence transition zone dynamics.

WORKSHOP DESIGN

A planning committee from the two forums designed the workshop to promote multidisciplinary interaction on a set of focus issues and questions. The workshop included invited platform speakers, a poster session, discussion groups, and an overall report-out from the groups and subsequent discussion. This approach worked well, resulting in fairly uniform agreement on concepts and recommendations regarding integration and use of investigatory tools.

Multidisciplinary Approach

Invited workshop participants included ecologists, geochemists, and hydrogeologists who work with the ground-water/surface-water transition zone (Appendix A).

Conceptual Model

A draft illustration of the conceptual model representing the forums' current understanding of ground-water/surface-water interactions for a river was presented and explained at the beginning of the workshop. The participants were asked to review the conceptual model and improve it as greater understanding was gained during the course of the workshop. Workshop participants also identified but did not address the need for research into other transition zone environments, such as those for lakes, estuaries, and wetlands.

Platform Speakers

The planning committee invited seven platform speakers to present topics representing a cross-section of information on ground-water/surface-water interactions; the presentations helped workshop participants address focus issues and questions in subsequent discussion groups. The abstracts of the speakers' presentations are included in this report:

- A Federal Statutory/Regulatory/Policy Perspective on Remedial Decision-making with Respect to Ground-Water/Surface Water Interaction (Guy Tomassoni, EPA's Office of Solid Waste)

- Interaction of Ground Water and Surface Water (Tom Winter, U.S. Geological Survey)

- Hydrogeology and Biogeochemistry of the Surface Water and Ground Water Interface of a Mountain Stream (Cliff Dahm, University of New Mexico)

- Ground-Water Plume Behavior Near the Ground-Water/Surface-Water Interface of a River (Brewster Conant, University of Waterloo)

- Assessment Approaches and Issues in Ecological Characterizations (Allen Burton, Wright State University),

- Delineation, Quantification, and Mitigation of Discharging Plumes (David Lee , AECL Chalk River, Ontario), and

- Field Technology and Ecological Characterization of the Hyporheic Zone (Dudley Williams, University of Toronto)

Poster Session

A poster session during the workshop allowed related papers to be presented outside of the formal discussion agenda. Abstracts of the posters are included in this report.

Discussion Groups

The topics of the three discussion groups were hydrogeology, chemistry, and biology as they relate to ground-water/surface-water interactions. Three of the platform speakers, Tom Winter, Allen Burton, and Cliff Dahm, and three members of EPA, Joseph Dlugosz, Ned Black, and Bruce Duncan, served as discussion group co-chairs to guide discussions along the focus issues listed in Appendix B. To focus the discussions further, participants were asked to consider first the scenario of ground water discharging to a river.

Each workshop participant was assigned to two of the three discussion groups, and each group was organized with a balance of hydrogeologists, geochemists, ecologists, and microbiologists to encourage dialogue among people with different academic backgrounds. When the groups rotated for the afternoon session, the co-chairs remained to provide continuity and briefly explain what the morning session had covered. Some of the focus group issues were not fully addressed due to lack of information or time, however. Discussion group summaries are included in this report.

Report Out and Overall Discussion

The information from the three discussion groups was summarized by the co-chairs and presented to all of the participants at the close of the workshop. This in turn led to a general group discussion of topics and future needs for research.

WORKSHOP RESULTS

The workshop brought together representatives from a variety of technical disciplines to focus on the ground-water/surface-water transition zone. Chemists, microbiologists, hydrogeologists, and ecologists from EPA, the U.S. Geological Survey (USGS), the National Oceanic and Atmospheric Administration (NOAA), state environmental agencies, other government agencies, academia, and industry discussed the hydrological, chemical, and biological processes that occur in this transition zone and how to measure and interpret changes in these processes. Discussions highlighted the need to revise the existing conceptual model for ecological risk assessment to evaluate the important structural and functional aspects of the transition zone. Information was provided about many tools used to evaluate the hydrological, chemical, and ecological aspects of this zone and the spatial and temporal scales at which measurements are needed.

The following is a summary of the key points drawn from the presentations by platform speakers, discussion group dialogues, and revisions to the conceptual model.

Platform Speakers

While providing a common multidisciplinary focus on transition zones, the speakers emphasized the following facets of transition zone hydrogeology, chemistry, and ecology:

- Physiography and climate affect the interaction of ground water and surface water across diverse landscapes. For example, movement of water through the transition zone is influenced by the position of surface water bodies within ground-water flow systems, small-scale geologic features beneath surface water, climate, and hyporheic exchange (the exchange of moving surface water with ground water). These seemingly diverse systems may be studied, analyzed, and managed under a unifying framework based on “hydrologic landscapes.” Transition zones are particularly important ecologically because they store and retain nutrients (and potentially contaminants), transform compounds biologically and chemically, provide refuge to benthic invertebrates, and are a base of the aquatic food web. Virtually no research has been conducted on the effects of contaminants on hyporheic communities. Research should evaluate indigenous microbial activity, organic matter/nutrient cycling, invertebrate community indices, tissue residues of dominant species, in situ toxicity, and in situ physicochemical profiles. Very site-specific research could include novel tools such as ecological food web modeling, semi-permeable membrane devices to evaluate bioaccumulation, toxicity identification evaluations to determine the classes of chemicals (e.g., metals or organic compounds) responsible for observed toxicity, and identification and evaluation of in situ stressors including physical stressors (e.g., flow or suspended solids). It also will be critical to establish appropriate uncontaminated reference sites for comparison with contaminated sites.

- The hydrogeology of the ground-water/surface-water transition zone strongly influences the spatial and temporal distribution of both aerobic and anaerobic microbial processes as well as the chemical form and concentration of nutrients, trace metals, and contaminants in surface and ground waters. Major hydrologic events such as spring snowmelt affect biochemical components. Studies that integrate hydrogeology, biogeochemistry, and aquatic ecology are needed to understand fully the dynamics and importance of the transition zone.

- Determining the location and magnitude of contaminant discharges to surface waters from ground-water plumes is a complex hydrogeological and biogeochemical problem. Although measurements of hydraulic gradient may be sufficient to delineate large discharge areas, numerous seepage studies have shown that areas of significant discharge can be small and easily missed. Even in relatively homogeneous terrain, flows may be highly focused at shorelines, and solute transport may be rapid. Geochemical conditions and contaminant concentrations may change drastically over intervals of a few centimeters. Closely spaced measurements can be used to determine contaminant concentrations in and flux from the streambed and to distinguish areas of high attenuation from areas of poor attenuation (e.g., sand stringers, interconnected zones of higher permeability, or other preferential flow paths).

- Physical and numerical model studies, like seepage studies, indicate surface-water head differences of a few centimeters between riffles, and pools in streams can produce surface-water exchange flows within permeable alluvial sediments despite net discharge of ground water to the stream. Modeling can be used to reveal interactions between surface water and ground water that are overlooked by larger scale models but have important chemical and biological consequences for the ground-water systems, the stream, and the biota.

Discussion Group Summaries

Each discussion group agreed on the importance of the ground-water/surface-water transition zone and emphasized the need for multidisciplinary approaches to evaluating fate, transport, and effects of contaminants in this zone. The main differences among the groups were in discussion of the tools used by each discipline.

Hydrogeology

The hydrogeology discussion group focused on using a tiered approach to determine the movement of ground water to surface water. The group recommended starting with a general reconnaissance of observable indicators of ground-water discharge and evolve to very detailed and focused sampling of hydraulics, chemistry, and biology. They recommended the following tiers:

- Use field methods that indicate ground-water discharge to surface water either indirectly (by observations of qualitative indicators or by chemical data) or directly (by using physical data to directly measure stage and calculate flow).

- Collect ground-water and surface-water samples over time and during different flow conditions.

- Adjust the field sampling strategy to account for different hydrologic landscapes.

The hydrogeology discussion group also suggested using a generic field design for investigating the ground-water/surface-water transition zone that includes use of piezometer nests, wells screened across the water table, and devices to measure or calculate the flow of water and chemicals through the transition zone. To address the interaction of ground water and surface water, the larger-scale (relative position of the surface water body within the ground-water flow system) hydrogeologic landscape processes and the smaller-scale (transition zone) processes should be evaluated.

The group recommended selecting field demonstration sites for research of ground-water/surface-water interaction in different geographic regimes that account for variation in hydrogeologic landscapes and climate. The design and effectiveness of site-characterization methods should be tested and evaluated, and based on the results, the conceptual model and tools for ground-water/surface-water transition zone characterization should be improved.

Chemistry

The chemistry discussion group emphasized that chemical information is used to evaluate contaminant chemistry and fate, biological processes, and flow paths. The group recommended the following:

- Develop initial estimates of actual or potential risks to receptors. Collect information on site geochemistry and contaminant flow paths—although this might be deferred until after an initial evaluation.

- Develop one or more standard conceptual models to identify important questions to ask and the data to collect at different types and scales of sites. Sampling efforts in the transition zone may be more costly than standard sampling of surface water or shallow ground water.

- Determine chemical variations in time and space. In the transition zone, chemical and biological processes occur over many different time scales such as daily cycles (e.g., temperature and transpiration), short-term weather events, invertebrate and fish life cycles, seasonal changes, and long-term climatic changes and events (such as extreme weather events). Characterizing the spatial extent of contaminant discharge to surface water is just as important as determining the concentration distribution in a ground-water plume. In a screening or predictive risk assessment, contaminant concentrations are used for comparisons to toxicity benchmarks. However, the mass flux or loading of contaminants is also important and influences both the impact of contaminants on habitats and the physical, chemical, and biological transformations of the contaminants at the transition zone. The flux of contaminants can change in magnitude and direction with changes in surface water temperature and stage.

Biology

The biology discussion group concluded that the transition zone is ecologically important. Some surface organisms have a life stage within this zone, and their productivity could be affected by contaminants in the zone. Less is known of the unique species that permanently inhabit the transition zone, and many have not been described. Transition zones often provide high quality habitats and are sites of contaminant reduction and nutrient and carbon cycling. Transition zones also can provide preferred habitat, refugia, sites of high biodiversity, habitat for the macrofaunal food base, microbial production, and energy transfer.

The group agreed that techniques and methods are available to evaluate the structure and function of the macrobiota and meiofauna. Methods also exist to sample organisms in the transition zone; however, many of these methods are neither standardized nor well-developed. In particular, there is no standard method to determine microbial community structure or activity/function. The group made the following recommendations:

- Use standard metrics, such as community composition, density, and species richness, to compare sample results regardless of the specific collection method. Evaluate functional feeding groups.

- Conduct bioaccumulation studies and stable isotope analyses to evaluate food chain relationships.

- Understand the basics of community structure and function at all levels before developing more methods to conduct toxicity testing.

- Coordinate ecologically related sampling in the transition zone with hydrogeological and chemical surveys at ground-water discharge sites. Use these surveys to help define the biological zones likely to be affected.

Conceptual Model

To produce the conceptual model shown in Figure 1, the workshop planning committee presented a draft model at the workshop and revised it from the comments received from participants. This model, drawn for a river, can be adapted to other sites (lake, tidal, estuaries, marshes, etc.). It combines ecological and hydrogeological concepts to focus on ecological processes in the transition zone and tools used to investigate fate, transport, and effects of contaminants in discharging ground water.

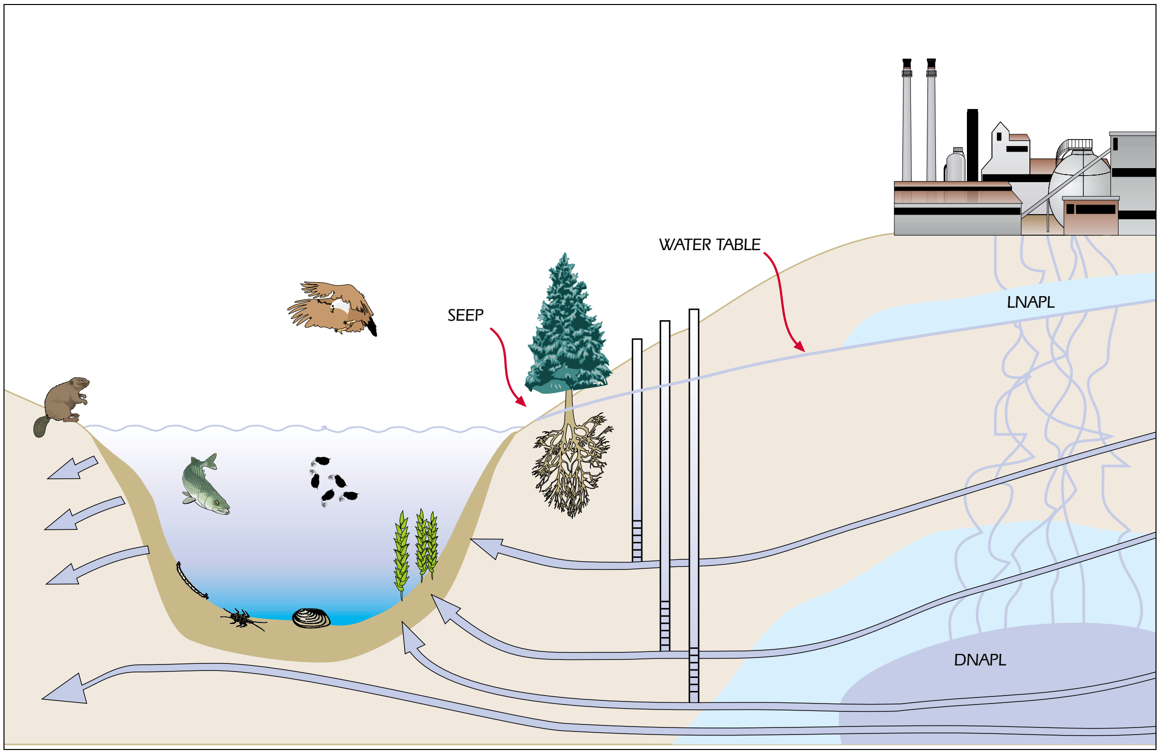

Figure 1. Conceptual model for contaminated ground water discharging to a river illustrating the need to look beyond surface water and benthic ecological receptors and hydrogeological fate and transport. Such a model should consider receptor exposure in the transition zone and account for finer-scale fate, transport, and effects from the discharge of contaminated ground water within this zone.

CONCLUSIONS

General consensus was reached that protecting the transition zone is important, and there is a need for interdisciplinary studies to understand and document the changes that occur in it. Conclusions related to the two fundamental organizing questions are discussed below.

How Important is the Transition Zone Ecologically?

The ground-water/surface-water transition zone is an ecological community with important ecosystem functions affecting several trophic levels from microbes to fish. As an ecotone (i.e., a transition from the ground-water ecosystem to the surface-water ecosystem), this zone provides key ecological services to the surface water ecosystem:

- Provides food for benthic macroinvertebrates. The microbial community serves as the food base to the small organisms within the zone that in turn are food for the benthic macroinvertebrates.
- Provides and maintains unique habitats or refugia, particularly in upwelling zones.
- Cycles nutrients and carbon in aquatic ecosystems.

The microbial and biological activity within this zone also may be important for natural attenuation, because large gradients can be created, which can result in subsurface conditions that change from anaerobic to aerobic over short distances. Biodegradation can cause organic contaminant concentrations to change over several orders of magnitude within this zone.

How Can We Measure Hydrogeological, Chemical, and Biological Conditions and Changes in this Zone?

Despite many unanswered questions (see next section) there are many tools from each of the disciplines that can be used to evaluate fate, transport, and effects in the transition zone. It was recognized that the types, locations, 0and times of measurements required to characterize this zone can vary depending on the questions being asked. Hydrogeologists and ecologists must work together to obtain information that is useful to both and to efficiently and properly evaluate this zone.

KEY RECOMMENDATIONS FOR RESEARCH

The recommendations presented below were identified during the various phases of the workshop, particularly within the discussion groups and during the report-out discussions on the final day.

Common Key Areas

The major recommendation common to all three discussion groups is that EPA should create a series of regional study areas of contaminated transition zone sites. Hydrogeologists, chemists, and biologists together should determine how, where, and what to sample and how to interpret the results. These scientists are obligated to integrate their objectives into a single conceptual model to evaluate transition zones.

Hydrogeology

EPA should encourage research in areas that increase the basic understanding of the influences of nearby surface-water bodies on contaminant plume migration. Delineation of plumes can be improved by more widespread application of the hydrologic landscape concepts in site characterization. Specifically, the following are needed: (1) improved techniques for measuring hydraulic heads, in stream and on-shore; (2) improved estimation methods of ground-water flow rates near the surface water boundary; and (3) improved methods for delineating plume concentrations near discharge zones. Increased use of tracers to help document and quantify the rate of ground-water discharges (or recharges) is needed. Better gradient quantitation methods are needed, especially in zones of rapidly fluctuating surface water stage. Also, there is a need for better assessment and evaluation of the heterogeneity of the ground-water zones adjacent to the surface-water bodies.

Chemistry

EPA should identify a number of regionally representative sites with contaminated transition zones—along with appropriate uncontaminated reference sites—to be studied by EPA’s regional and Office of Research and Development (ORD) laboratories and academic grantees. The sites should reflect the scales and contaminant problems typical of each region because the transition zone chemistry, biology, and hydrology of small mountain streams impacted by mines in Region 8, for example, may be very different from those of a zone where chlorinated solvent plumes discharge to one of the Great Lakes in Region 5. The study of ground-water discharge and transition zone flow in estuaries will be further complicated by tidal fluctuations. Members of the chemistry discussion group felt strongly that extrapolating data from small streams to large rivers and lakes is unacceptable. Also, some investigations techniques work well in small streams, but not in areas of high flow. As with any landscape approach, the chemical species and the dominant chemical and physical processes vary for different landscapes, but some basic processes may be common to some or all of these sites.

Biology

Biological investigations rely heavily on hydrogeological and chemical investigations, particularly for identifying discharge zones. The regional study sites recommended by the other two groups should be used to fulfill several biological research needs. The greatest need is for basic biological research, such as life histories, faunal surveys, and organism activity, so that the full importance of the transition zone can be determined and changes related to contaminants can be quantified. Sampling and evaluation tools for both contaminated and uncontaminated substrates need to be developed and standardized to determine contaminant effects on species richness, trophic structure, and organism growth for macrobiota, meiofauna, and microorganisms in the transition zone ecosystem. Quantitative links are needed between site-specific chemical, hydrogeological, and ecological factors and the valued functions of the transition zone (e.g., contaminant degradation, food base for benthic organisms, role as a refuge, and high quality habitat).

NEXT STEPS

This workshop was the first step in creating a multidisciplinary foundation for investigating, monitoring, and evaluating effects in the transition zone from the discharge of contaminated ground water. Future efforts building on this foundation should take many paths. For example, the conceptual model of the transition zone presented here is continually evolving. Conceptual models representing discharges to water bodies other than rivers need to be considered so that approaches and tools appropriate to wetlands, estuaries, and lakes—including those influenced by tides—can be identified and developed. Similarly, other pathways need to be identified and addressed, such as contaminated sediments as sources of contamination to ground water and to the transition zone where infiltration of surface water occurs.

Based on the workshop, the Ground Water Forum and the Ecological Risk Assessment Forum intend to:

- Submit research recommendations to ORD.
- Develop a list of suggested tools for investigating hydrogeological fate and transport and ecological effects at contaminated sites.
- Develop Agency guidance for incorporating the transition zone into risk assessments.
- Conduct a pilot study.

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PRESENTATION ABSTRACTS

A Federal Statutory/Regulatory/Policy Perspective on Remedial Decision-making with Respect to Ground-Water/Surface-Water Interaction

by Guy Tomassoni

The ground water/surface water interaction zone is important because 75% of Superfund and RCRA sites are located within a half mile of a surface water body. Forty-seven percent of Superfund sites have recorded impacts to surface water. Most RCRA sites are located adjacent to or near surface water (presumably for ease of transportation and manufacturing). Within the last 25 years, the Clean Water Act has succeeded in cleaning up point sources in the United States, and EPA now needs to consider non-point sources.

“Risk-based decision making” (RBDM) has received a bad reputation within EPA because it has been equated to “risk-based corrective action” (RBCA). A goal of this workshop is to provide the scientific basis to convince policy-makers to allow RBDM. EPA supports RBDM, but places more emphasis on site-specific evaluations based on sound science. RBDM generally requires a multidisciplinary approach, an understanding of requirements, and flexibility in applicable statutes, regulations, and policies.

There are many technical and policy issues regarding ground-water/surface-water interactions. Good policy is flexible, and good policy comes from good technical information. This workshop therefore may influence future policy. Superfund and RCRA remediation (“corrective action”) programs. These laws mandate protection of human health and environment. The Superfund National Contingency Plan offers greater detail; RCRA relies more on program guidance.

Highlights from “Rules of Thumb for Superfund Remedy Selection” (http://www.epa.gov/superfund/resources/rules/index.htm)

Superfund’s goal is to return usable ground water to beneficial uses (current and future) where practical. When this is not practical, Superfund strives to prevent further migration and exposure, and to evaluate opportunities for further risk reduction. Ground water generally is considered “potable” if it is so designated by the state, or considered so under federal drinking water guidelines. Preliminary remedial goals are set at levels that protect resources—including surface waters—that receive contaminated ground water, taking into account Clean Water Act requirements or state standards, if they are more stringent. Attaining drinking water standards in contaminated ground water is not always enough to protect sensitive ecological receptors. Final clean-up levels should be attained throughout the plume and beyond the edge of any wastes left in place. The “point of compliance” for a surface water body is where the release enters the surface water. Alternate concentration limits (ACLs) may be considered where contaminated ground-water discharges to surface water, where contaminated ground water does not lead to increased contaminants in surface water, where enforceable measures are available to prevent exposure to ground water, or where restoring ground water is “not practicable.” There are about 23 Superfund ACLs nationwide. EPA expects to use treatment to address “principal threats” posed by site where practical.

RCRA Setting, Based Upon the May 1, 1996, Advance Notice of Proposed Rulemaking (http://www.epa.gov/correctiveaction)

RCRA has similar requirements to Superfund with respect to: returning usable ground water to beneficial uses; points of compliance for ground water and surface water; protection of surface water from contaminated ground water; provisions for ACLs (but without an explicit link to “practicability”); and treatment of principal threats. If current human exposures are under control and no further migration of contaminated ground water is expected, primary near-term goals are established using two environmental indicators. Surface water becomes the boundary if the discharge of contaminated ground water is within “protective” limits.

The OSWER Policy Directive on Monitored Natural Attenuation (MNA) was issued in final form, and is pertinent to the ground water/surface water issue. It addresses dilution, dispersion, absorption, and degradation—all of which occur in ground water/surface water interaction. The directive requires controlling sources and monitoring; it stresses the need to look beyond obvious contaminants.

In summary, the majority of contaminated sites have serious potential to affect surface waters. The federal framework allows for RBDM with respect to ground water/surface water interaction, but we must still achieve the expectation of restoring ground water to beneficial use and ensure discharges of ground water to surface water are protective. Key policy issues to ponder—and to pass to senior managers—include:

- how to achieve short- and long-term protection;
- where, how, and how often to measure compliance;
- whether to restore ground water; even if it has no impact to surface water;
- the diversity of surface bodies;
- the relation of cleanup goals to the Clean Water Act’s National Pollutant Discharge Elimination System (NPDES) approach; and
- how to account for, track, and communicate total loads in watersheds.

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Interaction of Ground Water and Surface Water

By Thomas C. Winter

INTRODUCTION

Surface water bodies are hydraulically connected to ground water in most types of landscapes; as a result, surface-water bodies are integral parts of ground-water flow systems. Even if a surface water body is separated from the ground-water system by an unsaturated zone, seepage from the surface water may recharge ground water. Because of the interchange of water between these two components of the hydrologic system, development or contamination of one commonly affects the other. The movement of surface water and ground water is controlled to a large extent by the physiography (land-surface form and geology) of an area. In addition, climate, through the effects of precipitation and evapotranspiration, affects the distribution of water to—and removal from—landscapes. Therefore, it is necessary to understand the effects of physiography and climate on surface water runoff and ground-water flow systems in order to understand the interaction of ground water and surface water.

The purpose of this paper is to: present an overview of how physiography and climate affect the interaction of ground water and surface water and present the concept of hydrologic landscapes as a unifying framework for study, analysis, and management of seemingly diverse landscapes. Specifically discussed are the effects of the following factors on movement of water between ground water and surface water: (a) position of surface water bodies within ground-water flow systems; (b) small-scale geologic features in beds of surface water; (c) climate; and (d) hyporheic exchange.

GENERAL HYDROLOGICAL PROCESSES RELATED TO THE INTERACTION OF GROUND WATER AND SURFACE WATER

Position of Surface Water Bodies With Respect to Ground-Water Flow Systems

Ground water moves along flow paths of varying lengths from areas of recharge to areas of discharge. The source of water to the water table (ground-water recharge) is infiltration of precipitation through the unsaturated zone. Ground-water flow systems can be of greatly different sizes and depths, and they can overlie one another. Local flow systems are recharged at water-table highs and discharge to adjacent lowlands or surface water. Local flow systems are the most dynamic and the shallowest flow systems; therefore, they have the greatest interchange with surface water. Local flow systems can be underlain by intermediate and regional flow systems. Water in these deeper flow systems have longer flow paths, but they also eventually discharge to surface water. Surface water bodies that receive discharge from more than one flow system receive that water through different parts of their bed. Local flow systems discharge in the part nearest shore, and larger-magnitude flow systems discharge to surface water further offshore. Because of the different lengths and travel times of water within flow paths, the chemistry of water discharging into the surface water from different flow paths can be substantially different.

In some landscapes, surface water bodies lie at intermediate altitudes between major recharge and discharge areas. Surface water bodies in such settings commonly receive ground-water inflow on the upgradient side and have seepage to ground water on the downgradient side. Furthermore, depending on the distribution and magnitude of recharge in the uplands, the hinge line between ground-water inflow and outflow can move back and forth across part of the surface water bed.

The above characteristics of ground-water flow systems with respect to surface water apply in a general regional sense to most landscapes. However, the detailed distribution of seepage to and from surface water is controlled by: (a) the slope of the water table with respect to the slope of the surface water surface; (b) small-scale geologic features in the beds of surface water; and (c) climate.

Effect of Local Water-Table Configuration and Geologic Conditions on Seepage Distribution in Surface Water Beds

Upward breaks-in-slope of the water table result in upward components of ground-water flow beneath the area of lower slope and downward breaks-in-slope of the water table result in downward components of ground-water flow. These flow patterns apply to parts of many landscapes, but they are particularly relevant to the interaction of ground water with surface water because water tables generally have a steeper slope on both the inflow and outflow sides relative to the flat surface of surface water bodies. The ground-water flux through a surface water bed associated with these breaks-in-slope, whether the seepage is to or from the surface water, is not uniformly distributed areally. Where ground water moves to or from a surface water body underlain by isotropic and homogeneous porous media, the flux is greatest near the shoreline, and it decreases approximately exponentially away from the shoreline. Anisotropy of the porous media, which is a function of the orientation of sediment particles in the geologic materials, affects this pattern of seepage by causing the width of areas of equal flux to increase with increasing anisotropy. Yet the decreasing seepage away from the shoreline remains nonlinear.

Geologic heterogeneity of surface water beds also affects seepage patterns. Small-scale variations in sediment type can cause the locations and rates of seepage to vary substantially over small distances. For example, highly conductive sand beds within finer-grained porous media that intersect a surface water bed results in subaqueous springs. The horizontal and vertical hydraulic conductivity of the streambed can vary by several orders of magnitude because of the variability of streambed sediments. The complex distribution of seepage patterns caused by the heterogeneous geology of surface water beds has been documented by field studies in many settings.

Effect of Climate on Seepage Distribution in Surface Water Beds

The most dynamic boundary of most ground-water flow systems is the water table. The configuration of the water table changes continually in response to recharge to and discharge from the ground-water system. Changing meteorological conditions strongly affect seepage patterns in surface water beds, especially near the shoreline. The water table commonly intersects land surface at the shoreline, resulting in no unsaturated zone at this point. Infiltrating precipitation passes rapidly through a thin unsaturated zone adjacent to the shoreline, which causes water-table mounds to form quickly adjacent to the surface water. This process, termed “focused recharge,” can result in increased ground-water inflow to surface water bodies, or it can cause inflow to surface water bodies that normally have seepage to ground water. Each precipitation event has the potential to cause this highly transient flow condition near shorelines as well as at depressions in uplands.

Transpiration by near-shore plants has the opposite effect of focused recharge. Again, because the water table is near the land surface at edges of surface water bodies, plant roots can penetrate into the saturated zone, allowing the plants to transpire water directly from the ground-water system.

Transpiration of ground water commonly results in a drawdown of the water table much like the effect of a pumped well. This highly variable daily and seasonal transpiration of ground water may reduce ground-water discharge to a surface water body significantly or even cause movement of surface water into the subsurface. In many places, it is possible to measure diurnal changes in the direction of flow during seasons of active plant growth: that is, ground water moves into the surface water during the night, and surface water moves into shallow ground water during the day.

These periodic changes in the direction of flow also can take place on longer time scales. Focused recharge from precipitation predominates during wet periods, and drawdown by transpiration predominates during dry periods. As a result, the two processes—together with the geologic controls on seepage distribution—can cause flow conditions at the beds of surface water bodies to be extremely variable. These processes probably affect small surface water bodies more than large surface water bodies because the ratio of edge length to total volume is greater for small water bodies than it is for large ones.

A type of landscape that merits special attention are those areas underlain by limestone and dolomite. These landscapes, which are referred to as karst terrains, commonly have fractures and solution openings that become larger with time because of dissolution of the rocks. Ground-water recharge is very efficient in karst terrain because precipitation readily infiltrates through the rock openings that intersect the land surface. Water moves at greatly different rates through karst aquifers; it moves slowly through fine fractures and pores and rapidly through solution-enlarged fractures and conduits. The paths of water movement in karst terrain are especially unpredictable because of the many paths ground water takes through the maze of fractures and solution openings in the rock. Seeps and springs of all sizes are characteristic features of karst terrains. In addition, the location where the streams emerge can change, depending on the spatial distribution of ground-water recharge in relation to individual precipitation events. Large spring inflows to streams in karst terrain contrast sharply with the generally more-diffuse ground-water inflow characteristic of streams flowing across sand and gravel aquifers.

Hyporheic Exchange

Streambeds and banks are unique environments because they are where ground water that drains much of the subsurface of landscapes interacts with surface water that drains much of the surface of landscapes. “Hyporheic exchange” is the term given to the process of water and solute exchange in both directions across a streambed. The direction of seepage through the bed of streams commonly is related to abrupt changes in the slope of the streambed or to meanders in the stream channel. This process creates subsurface environments that have variable proportions of water from ground water and surface water. Depending on the type of sediment in the streambed and banks, the variability in slope of the streambed, and the hydraulic gradients in the adjacent ground-water system, the hyporheic zone can be as much as several feet in depth and hundreds of feet in width. The dimensions of the hyporheic zone generally increase with increasing width of the stream and permeability of streambed sediments. Because of this mixing between ground water and surface water in the hyporheic zone, the chemical and biological character of the hyporheic zone may differ markedly from adjacent surface water and ground water.

Although most work related to hyporheic-exchange processes has been done on streams, processes similar to hyporheic exchange also can take place in the beds of some lakes and wetlands because of the reversals in flow caused by focused recharge and transpiration from ground water near surface water, discussed above. Therefore, it is not enough to know only the relationship of surface water to ground-water flow systems and to small-scale seepage patterns in surface water beds, because hyporheic-exchange processes also can be important in some types of landscapes.

Hydrologic Landscapes as a Unifying Concept for Diverse Localities and Regions

As indicated above, many geologic and climatic factors affect the movement of water through a basin. The many different types of landforms, geologic settings, and climate variations that make up many regions of the Earth may make it seem that a unifying conceptual framework is impossible to achieve. Indeed, it is not unusual for scientists and water- and land managers to emphasize the uniqueness and complexity of a given locality rather than the similarities that it might have with other localities. However, with respect to the movement of water and chemicals, many seemingly diverse landscapes have some features in common, and it is these commonalities that need to be identified. Only by evaluating landscapes from a common conceptual framework can processes common to some or all landscapes be distinguished from processes unique to particular landscapes. A common conceptual framework also would lead to development of field designs of data collection programs that could be transferred to other landscapes having similar characteristics.

The concept of hydrologic landscapes is based on the idea that a single, simple physiographic feature is the basic building block of all landscapes. This feature is termed a “fundamental landscape unit,” and is defined as an upland adjacent to a lowland separated by a steeper break in slope. Water moves over the surface of a fundamental landscape unit depending upon the surface slope of the upland, lowland, and intervening steeper slope, and it moves through the subsurface depending upon the hydraulic characteristics of its internal geologic properties.

All landscapes can be conceived of as variations and multiples of fundamental landscape units. Variations and multiples of fundamental landscape units can be used to define a number of general landscape configurations; for example: (1) the width of the lowland, valley side or upland can range from narrow to wide; (2) the slopes of the three surfaces can vary; (3) the height of the valley side can range from small to large—that is, the upland can be only slightly higher than the lowland or it can be much higher; or (4) small fundamental landscape units can be superimposed on any or all of the surfaces of larger-scale fundamental landscape units.

General landscape configurations such as these can be used to define general landscape types that describe major physiographic features of the Earth. For example:

(1) A landscape consisting of narrow lowlands and uplands separated by high and steep valley sides is characteristic of mountainous terrain. This general configuration can be nested into multiples at different scales within mountainous terrain as one moves from high mountain basins to larger and larger valleys within a mountain range complex.

- A landscape consisting of very wide lowlands separated from much narrower uplands by steep valley sides is characteristic of basin and range physiography and basins of interior drainage. In this type of terrain, the uplands may range from being slightly higher to much higher than the lowlands.
- A landscape consisting of narrow lowlands separated from very broad uplands by valley sides of various slopes and heights is characteristic of plateaus and high plains.
- A landscape consisting of one or more small fundamental landscape units (terraces) nested within a larger lowland is characteristic of riverine valleys and coastal terrain. A landscape consisting of

numerous small fundamental landscape units superimposed on both the uplands and lowlands of larger fundamental landscape units is characteristic of hummocky glacial and dune terrain.

Common Hydrologic Characteristics of Generalized Hydrologic Landscapes

The movement of water over the surface and through the subsurface of generalized landscapes is controlled by common physical principles regardless of the geographic location of the landscapes. For example, if a landscape has low land slope and low-permeability soils, surface runoff will be slow and recharge to ground water will be limited. In contrast, if the soils are permeable in a region of low land slope, surface runoff may be limited but ground-water recharge will be high. In landscapes that have a shallow water table, transpiration directly from ground water may have a substantial effect on ground-water flow systems, and on the movement of ground water to and from surface water.

Landscapes characterized by multiples of fundamental landscape units can have complex ground-water flow systems because small-scale local flow systems associated with each topographic break in the landscape are superimposed on larger, more regional flow systems associated with larger fundamental landscape units. Two seemingly diverse landscapes, such as riverine and coastal terrain, have many of these types of physiographic characteristics in common, and presumably would have many hydrologic characteristics in common as well. Ground-water flow conditions in hummocky terrain are even more complex than riverine and coastal terrain because of the numerous small fundamental landscape units superimposed somewhat randomly on larger and larger fundamental landscape units. Indeed, in glacial and dune terrain, many multiples of fundamental-landscape-unit scale can be present. Furthermore, generally shallow water tables characteristic of coastal, riverine, and hummocky terrain result in the opportunity for highly transient local ground-water flow systems caused by focused recharge and transpiration directly from ground water.

Implications for management of water and remediation of contaminated localities

Management of water, and remediation of contaminated localities, requires sound understanding of hydrological processes. Contaminated ground water and surface water are common in all types of landscapes. Because of the cost of studies and of remediation, it is nearly impossible to devote adequate resources to the huge number of sites that need attention. Therefore, it is of great practical value to seek transferability of study design, study results, and remediation techniques. It is suggested that the concept of hydrologic landscapes can serve as a foundation for determining the commonalities of diverse localities, and sharpen the perspective of their differences. If this can be accomplished, the transfer value of study designs and remediation methods should substantially reduce the cost of site remediation.

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Hydrogeology and Biogeochemistry of the Surface Water and Ground Water Interface of a Mountain Stream

By Cliff Dahm

Our interdisciplinary research group has been studying the hydrogeology, biogeochemistry, and ecology of the surface water and ground water interface of the Rio Calaveras in the Jemez Mountains of northern New Mexico since 1991. Snowmelt is a prominent factor in the hydrogeology of both surface discharge and the alluvial ground water of the site. Strong interannual variability in the strength of the snowmelt signal affects both the biogeochemistry and ecology of the surface water and ground water. Water table variation in drought years is small, and upwelling and downwelling zones through the bed of the channel show a complex spatial pattern, with distinct losing and gaining sections of stream over a 150-meter reach throughout most of the year. Water table variation in wet years with good snow pack ranges between 40 to >100 centimeters in the alluvial flood plain, and most of the reach is gaining (upwelling) from March through May. Flow lines are directed towards the stream with both ground water and saturated overland flow contributing to increased stream discharge. Drought years are characterized by discharge increases as little as three times base flow while discharge increases during wet years exceed two orders of magnitude above base flow.

Biogeochemical characteristics of the surface water and ground water are strongly influenced by the hydrogeology. Snowmelt generates water that is rich in nitrate, dissolved organic carbon (DOC), and oxygen. Much of the increase in dissolved organic matter and nutrients is derived from the region of seasonal saturation (ROSS) that is inundated during snowmelt. Studies on the DOC leached from the ROSS have shown that half of this DOC is labile and metabolized within one month. Alluvial ground water shows strong vertical structure from the snowmelt inputs with peaks in oxygen, nitrate, DOC, and low molecular weight organic acids in the upper 50 centimeters in the first few weeks following snowmelt. As water table elevations drop, concentrations of oxygen, nitrate, sulfate, DOC, and organic acids decrease, while byproducts of anaerobic metabolism such as ferrous iron, manganous manganese, and methane increase. Surface water inputs of organic matter and nutrients also reflect the changing hydrology that occurs from snowmelt to base flow conditions. For example, nitrate and DOC levels are highest during the early stages of snowmelt and low during base flow conditions. Algal primary production shows a nitrogen limitation during low-flow conditions but not during times of increased stream discharge.

Interactions between surface waters and ground waters at this site also affect the biological communities of the stream benthos. High discharge during periods of snowmelt scours benthic algae and reduces chlorophyll concentrations and algal biomass throughout the reach. As snowmelt discharge decreases, a diatom-dominated benthic algal bloom commonly occurs over much of the stream bottom. As flows return to base flow conditions, a spatially heterogeneous pattern of algal community structure and biomass emerges. Persistent upwelling zones at base flow, where ground water discharges into surface water, are generally more productive reaches and composed of a complex mix of diatoms, green algae, and cyanobacteria. More focused benthic invertebrate activity appears to occur in these reaches. Persistent downwelling zones, where surface water recharges the ground water, commonly have lower rates of algal primary production and contain a higher proportion of cyanobacteria in the algal community. Hydrogeology, nutrient availability, and interactions between grazers and primary producers all play important roles in structuring the benthic algal community.

Integrative studies that combine hydrogeology, biogeochemistry, and aquatic ecology are needed to fully understand the dynamics and importance of the ground water/surface water interface. Research at Rio Calaveras in northern New Mexico has been designed to bring these disciplines together in a multidisciplinary study of a well-instrumented 150-meter reach of mountain stream. This research has shown the importance of major hydrologic events such as spring snowmelt in the overall hydrology, biogeochemistry, and ecology of this ecosystem. In addition, the distribution of aerobic and anaerobic microbial processes in the alluvial ground water system and the chemical form and concentration of nutrients and trace metals in the surface waters and ground waters are strongly affected by the hydrogeology of the ground water/surface water interface.

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Ground-water Plume Behavior Near The Ground-Water/Surface Water Interface of a River

By Brewster Conant, Jr.

INTRODUCTION

What happens to ground-water contaminant plumes as they discharge through river beds and the ground water/surface water interface (GWSI) is not well understood. Relatively few published studies address this issue, even though an estimated 51 percent of National Priority List sites are thought to impact surface water (U.S. EPA, 1991) and the most common route for the contaminants to migrate into the surface water was via ground-water transport (U.S. EPA, 1989). Understanding processes occurring beneath and near rivers becomes particularly relevant when making remediation decisions that are risk-based or involve natural attenuation. Such decisions could benefit greatly by identifying important plume transport and fate processes and by conducting detailed hydrogeological studies of plumes to characterize the spatial and temporal variations of contaminant discharges to rivers.

GROUND-WATER PLUME DEVELOPMENT

Many factors influence the transport and fate of contaminants in the subsurface prior to a ground-water plume discharging to the surface water of a river. To understand the significance of these factors, it is useful to consider the fundamentals of how dissolved-phase contaminant ground-water plumes are created. Several factors play important roles in plume development:

- Physical and chemical characteristics of the contaminants
- Geometry and temporal variations in the contaminant source zone
- Transport mechanisms (advection and dispersion)
- Reactions (destructive and non-destructive)

Many of these factors are just as applicable to contaminant behavior near and beneath rivers as they are away from the river. Knowing the behavior and concentration distribution of plumes, before they enter the complex conditions near and beneath a river, allows better assessment of what modifying effects near river processes have on the plume.

Contaminant Characteristics

A contaminant's physical and chemical characteristics play an important role in how the contaminant is transported and redistributed in the subsurface and the hazard it poses to aquatic life. Many types of contaminants are found in the subsurface including; synthetic organics, hydrocarbons, metals, other inorganics (e.g., nitrate), radionuclides, and pathogens (e.g., viruses and bacteria). Contaminants can be present as solids, liquids (e.g., non-aqueous phase liquids [NAPL]), dissolved in water, or present as gasses. Each contaminant has a different propensity to solubilize, sorb, bioconcentrate, volatilize, or react, and these characteristics affect both their mobility and toxicity. With respect to organic compounds, the strongly hydrophobic organic compounds (e.g., PCBs, pesticides, and PAHs) have higher bioconcentration factors and tend to be more toxic to aquatic life than less hydrophobic organics such as chlorinated volatile organic compounds (CVOCs). The strongly hydrophobic compounds generally have low aqueous solubilities and, when dissolved in water, move much more slowly than ground water (i.e., adsorb and are retarded), whereas, the CVOCs have higher solubilities and are less retarded. Consequently, many of the longer and higher concentration dissolved-phase organic plumes in ground water are dominated by the more mobile CVOCs, which are generally thought to be "less toxic" to aquatic life. However, aquatic biota located in the streambed and at the GWSI (i.e., not in the surface water) may still be adversely affected by CVOCs because they may be exposed to high concentrations in the discharging ground water prior to any dilution by surface water. If ground-water concentrations are higher than freshwater aquatic life standards or guidelines, the locations of these discharge zones may represent a hazard to both the benthic and hyporheic aquatic life in the streambed, regardless of how these contaminants might later attenuate in the surface waters of the open river channel.

Contaminant Source Zone

At many Superfund and RCRA sites, considerable effort is spent trying to delineate the source of contaminants impacting the ground water. These sites, particularly those involving CVOCs, typically involve so-called "point sources" of ground-water contamination resulting from spills or releases limited over relatively discrete release areas. This paper does not address "non-point" sources of contamination, such as nitrate and pesticide contamination from large-scale agricultural applications, even though such "source areas" cover more of the watershed area contributing water to the stream.

Each individual contaminant source zone has a particular distribution in the subsurface. The location, mass, and type of contaminants in the subsurface, along with characteristics of the subsurface geology and ground-water flow, will influence whether the source produces a ground-water plume with a continuous, variable, or a "slug" input. A source below the water table consisting of dense non-aqueous phase liquid (DNAPL) results in continuous dissolved-phase plumes that can persist for tens to hundreds of years if left to naturally dissolve (Feenstra, et al., 1996). Variable source plumes may be caused by variations in waste stream inputs, or by preferential dissolution and depletion of multi-component contaminant sources over time (Feenstra and Guiguer, 1996). Slug inputs are "instantaneous" or short duration releases that do not persist at the initial release location and move through the flow system as a localized mass. Of particular concern for impacts on surface water are the continuous and variable sources which represent long term sources of contaminants to a river. Continuous-source plumes may result in areas of the streambed being constantly exposed to high concentrations of contaminated ground water. Because contaminants enter streambed from the ground water below, the sediments become contaminated at ground water discharge locations. Even if those sediments are eroded away and transported down stream, the clean materials redeposited in their place will be subsequently contaminated by further ground water discharge.

TRANSPORT

Ground-Water Flow

The primary mechanism by which contaminants are transported away from source zones and toward ultimate points of discharge, such as rivers, is advection (i.e., dissolved phase contaminants moving with the ground water). Therefore, the ground-water flow system plays a fundamental role in determining where a dissolved phase plume from a contaminant source zone will go and whether a given surface water body may be affected. Many factors affect ground water flow including; climate (particularly precipitation recharge), watershed characteristics, geology, hydraulic conditions (water table slope and ground-water potential), and hydrogeologic boundary conditions (such as discharge or "no-flow" locations). Characterizing the ground-water flow system at a site can be more large scale or regional when initially conceptualizing potential contaminant plume flow paths. However, when investigating point-source plumes that reach rivers, the focus needs to be on smaller scale flow characteristics in order to accurately determine specific locations of contaminated ground water discharge to the river.

Several types of significant vertical ground-water flow behavior can occur both on a regional scale and on a more local scale in the vicinity of streams. Depending on the depth and location of a source zone, the plume may be transported through what Toth (1963) termed local, intermediate, regional, ground water flow systems. If the ground water plume develops in a shallow local flow system, it may discharge to the nearest surface water body. If the plume develops from a deeper source zone (e.g., DNAPL) or is located within a regional or intermediate flow system, it may travel beneath several lakes or streams before ultimately discharging to one of them. Winter (1999) shows some examples of vertical cross-sectional views of ground water interactions for streams, lakes, and wetlands. Different types of ground water/river interactions are also shown in Bear (1979, p. 52).

The lateral component ground water flow (i.e., in plan-view) near rivers exhibits a variety of behaviors. In a study of rivers in large alluvial aquifers by Larkin and Sharp (1992) showed that ground water flow could be base flow, under flow, or mixed flow, depending on the slope, sinuosity, and depth of penetration of the river in the aquifer. Base flow occurs when ground water flows essentially perpendicular to the river and discharges to it. Under flow occurs when ground-water flow near the river is parallel to the river and does not discharge to the river channel (at least not for some great distance). Mixed flow is a combination of base flow and under flow where ground water near the river flows at an angle to the river and discharges to it some distance downstream. Woessner (1998) presents some other variations in this behavior. One consequence of these possible behaviors is that plumes entering alluvial valleys may not necessarily travel straight across the flood plain toward the river but instead may travel down valley large distances before discharging through the stream bottom and into the stream. In such regimes, simply trying to determine the path of a plume near a river becomes a challenge and finding the exact areas of discharge may be very difficult.

Defining and Locating The Ground-Water/Surface-Water Interface

As ground water travels through the subsurface, it eventually reaches the GWSI near the stream or river. At the GWSI, a transition occurs between the hydraulic, biochemical, thermal, and ecological conditions of the surface water and those associated with the ground water. Because changes in these parameters may be gradational, defining the location of the GWSI is not simple. The location of the GWSI is not static and may change as a result of daily or seasonal fluctuations in river stage and ground water flow. The GWSI can be defined as the location where water having some portion of surface water is in contact with 100 percent ground water. This contact may occur right at the streambed-water column interface, or it may exist at some depth within the streambed or stream banks. The contact between the contrasting waters may be reasonably sharp or transitional. The primary reason that the GWSI may exist within the streambed materials, as opposed to the upper surface of them, is due to topographic variations in the streambed and changes in the slope of the river (i.e., hydraulic potential). Surface water may enter the sediments at downwelling zones and reenter the river at upwelling zones (Vaux, 1968, and Boulton,1993). Downwelling generally occurs at the head of riffles and upwelling (along with ground water discharge) occurs at the upstream edge and base of pools. Figure 1 is a schematic depicting downwelling and upwelling zones (in vertical cross section) and the effect on the location of the GWSI and a discharging ground water plume. The surface water can also leave the channel laterally and travel several meters or more into the streambanks and eventually reenter the channel down stream (Harvey and Bencala, 1993). Where surface water leaves the stream channel, ground water can not directly enter the channel; therefore, the GWSI and ground water plume will be pushed away from those locations and the plume may ultimately discharge elsewhere (Figure 1).

The GWSI is not synonymous with the term “hyporheic zone.” The hyporheic zone is an ecological term that generally refers to an ecotone where both ground water and surface water are present in a streambed along with a specific set of biota (i.e., the hyporheos). Hyporheic zones occur as a result of flowing waters (e.g., streams) and so the term is not applicable to quiet waters (e.g., lakes) even though they have GWSIs too. A broader definition of the hyporheic zone has been proposed by White (1993) that includes any area impacted by channel (i.e., surface) water, but one set of specific criteria defining this zone has not yet been agreed upon. Delineating both the GWSI and the hyporheic zone is important when considering ecotoxicological impacts because a unique set of benthic and hyporheic aquatic life have adapted to the stream environment. The hyporheic zone may represent an ecological resource needing protection. Other work suggests the GWSI may also be an important natural attenuation zone for contaminated ground water discharge.

Dispersion

Dispersion of contaminants in ground water refers to a process by which dissolved phase concentrations are reduced by the spreading out of the plume and hydrodynamic mixing of the water with cleaner surrounding ground water. Reductions in plume concentrations by dispersion in ground water flowing in aquifer sands and gravels is a very, very weak process compared to the turbulent mixing processes that occur in the open channel flow of rivers. Because of low lateral dispersion, plumes emanating from discrete source zones (e.g., DNAPL) are generally long thin “snake” like plumes (Rivett, et al., 1994) rather than wide “fan” shaped plumes. One important implication of low dispersion is that high concentration “cores” of ground water plumes (Cherry 1996), measured a short distance downgradient of the source, may not diminish much before reaching the river. Therefore, it is possible for very high concentration portions of the plume to reach discharge areas unless other reactions (e.g., biodegradation) occur along those flow paths to reduce the concentrations.

In locations where surface water enters the streambed, a hyporheic zone “mixing” of surface water with ground water may occur. This mixing process will result in what may appear to be quite substantial reductions in plume concentrations. The mechanisms causing this type of mixing are not well understood and result in “apparent” dispersion. Some of the uncertainty may stem from the fact that the hyporheic zone represents primarily a “surface water” flow path as opposed to a “ground water” flow path. For instance, the “mixing” that supposedly occurs in the hyporheic zone may actually be the result of ground water mixing with surface water at the base of the water column which then reenters the subsurface at a nearby downwelling zone (see the downstream downwelling zone in Figure 1).

Reactions

Two types of reactions can occur in the subsurface, destructive and non-destructive. Destructive reactions destroy or irreversibly transform the contaminant into other compounds. These reactions include biodegradation, abiotic

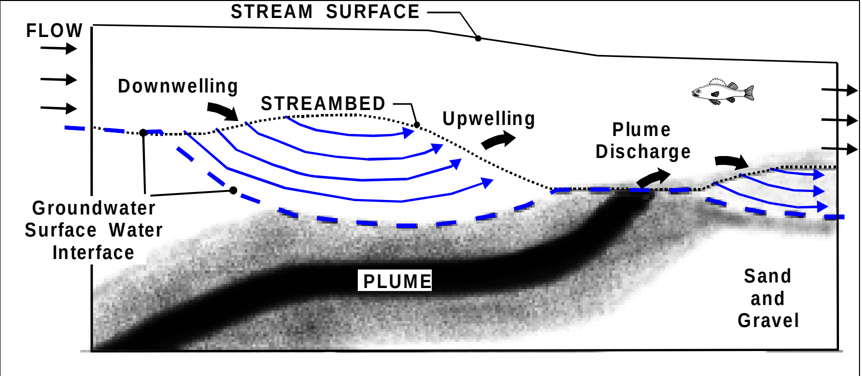

Figure 1. Groundwater/Surface-Water Interface (GWSI) of a Stream and Influence on Plume Discharge

reactions, and radioactive decay. Non-destructive reactions are reversible processes that may result in changes in contaminant concentrations in the ground water but do not destroy or transform the compound. These reactions include such things as adsorption, precipitation and dissolution, and ion exchange. A good discussion of these types of reactions as they apply to natural attenuation of chlorinated solvents can be found in U.S. EPA (1998). The relative importance of these reactions may be different in the immediate vicinity of the river than in the rest of the aquifer. In the streambed, high organic carbon content deposits contribute to higher adsorption than is typical for the surrounding aquifer. Adsorption of contaminants slows down (retards the movement of the contaminants relative to ground water flow) and sequesters them for later release. Adsorption results in contaminant loading of the sediments and in delayed breakthrough of contaminants flowing into the stream channel. Moreover, the high organic carbon and nutrient cycling also sustains a microbiological community that contributes to a greater potential for biodegradation. Biodegradation may greatly reduce contaminant concentrations. In some cases these reactions may be beneficial but in others the transformation products may be more toxic than the parent compound. In some instances (particularly petroleum product plumes), reactions that transform organic contaminants may also consume all the dissolved phase oxygen in the ground water and cause the ground water plume to become anaerobic. The adverse effect of this anaerobic water on the hyporheic and benthic aquatic life (that require oxygen to live) may be even greater than the toxic effects of the contaminants.

A TETRACHLOROETHYLENE (PCE) GROUND-WATER PLUME DISCHARGING TO A RIVER

To illustrate the importance of some of the above factors, results of investigations are presented for a site located in Angus Ontario, where a dissolved-phase PCE ground water plume from a dry cleaning facility discharges into the nearby Pine River. Previous subsurface investigations at this site using the Waterloo Profiler (Pitkin, 1994; Writt 1996) and recent work (Conant, unpublished data) have delineated a dissolved phase ground water plume that emanates from a PCE DNAPL source area. The plume travels 205 m laterally through a shallow but locally confined aquifer before discharging upward through a silt and peat semi-confining unit and then the sandy streambed deposits underlying the Pine River. The plume is approximately 50 m wide and has a vertical thickness of 4 to 6 m. Water quality data collected with the Waterloo Profiler show that the peak PCE concentrations in the plume at the bank of the river (<5 m from the river) are about 8000 µg/ℓ. Virtually no PCE degradation products were detected in the aquifer beneath the stream bank. Drivepoint piezometers screened in the aquifer at the river's edge show that there is a strong upward hydraulic gradient at the river. These piezometers have water levels approximately 1 m higher than the river stage. Water quality testing beneath the opposite bank of the river shows that the plume does not pass beyond the opposite bank.

Periodic sampling of the river water where the ground water plume discharges has detected no contamination, or very low PCE concentrations, generally less than 2 µg/ℓ. No PCE degradation products have been detected in the surface water. The river is about 14 m wide and during most of the year is generally less than 0.75 m deep and flows at approximately 1.5 to 2.9 cubic meters per second. The estimated total flux of dissolved PCE contamination traveling within the aquifer ground water toward the river each year (expressed as equivalent pure phase PCE) is approximately 15 to 40 liters (Writt 1996). In the River channel massive dilution of the discharging PCE ground water plume by the surface water occurs and the plume does not appear to significantly impact the surface water quality. However, high concentrations of contaminants within the streambed itself represent locations where adverse ecological impacts may be occurring. At some locations, concentrations in water samples collected from within the streambed were much higher than EPA's Freshwater Aquatic Life Chronic Toxicity Standard for PCE of 840 µg/ℓ and the Canadian Water Quality Guideline of 110 µg/ℓ for the protection of aquatic life.

Plume water traveling through the streambed deposits is subject to a wide range of hydrological and geochemical (redox) conditions which are spatially variable on a scale of centimeters to meters. Streambed temperature surveys have identified areas of the streambed dominated by ground water discharge. Hundreds of water samples have been collected to characterize discharge zones and locate the plume. The Waterloo Profiler, the newly developed "Mini-Waterloo Profiler," and "driveable multilevel samplers" have been used to collect interstitial water samples from the streambed and underlying shallow aquifer. These samples have been analyzed for both inorganic and organic parameters. Soil coring, ground penetrating radar surveys, and slug testing of streambed mini-piezometers have also been used to help develop a conceptual model of the subsurface system.

Four different types of flow conditions have been observed beneath the river at the site and are been associated with varying geochemical conditions. The four types of ground water flow in the streambed include: no flow, short circuit, high flow, and low to moderate flow (see Figure 2). In no flow locations, no ground water is discharging to the stream as a result of geological barriers or hydraulic barriers like downwelling. Consequently, at those locations the interstitial water in the streambed is geochemically quite similar to surface water and is not contaminated. The "short circuit" condition refers to discharge at springs and seeps where PCE contaminated ground water flows rapidly up through very localized gaps in the semi-confining unit and undergoes little or no attenuation or modification. In high flow areas, more permeable deposits result in areas of higher ground water flux. These areas are reflected in strong temperature anomalies at the streambed surface. More rapid flow and shorter residence times in the streambed deposits results in the discharge of contaminated ground water that has been only briefly exposed to reducing conditions (i.e., anaerobic and nitrate reduction). Consequently, PCE contaminated ground water has undergone very little degradation and attenuation. In the low to moderate ground-water discharge zones, contaminated ground water flows up through moderately permeable geological deposits where sulfate reducing and methanogenic conditions occur and substantial reductive dehalogenation of PCE is indicated by the presence of relatively high concentrations of degradation products (i.e., 100s to 1,000s of µg/ℓ of trichloroethylene, *cis*-1,2-dichloroethylene, vinyl chloride, ethene, and ethane). PCE concentrations at one location dropped from about 3700 µg/ℓ to less than 50 µg/ℓ within a vertical distance of 15 cm and there was a corresponding increase in the concentrations of degradation products which was primarily *cis*-1,2-dichloroethylene (see Figure 3). In low to moderate ground-water discharge areas PCE concentrations in the streambed are reduced to below the EPA's Freshwater Aquatic Life Chronic Toxicity Standard. At some of those locations, however, 100s up to a 1800 µg/ℓ of vinyl chloride (a human carcinogen) has been created. The potential hazard posed by vinyl chloride is unknown because it does not have an aquatic life water quality standard or guideline. In the short circuit and high ground water discharge zones the concentrations in the streambed were observed to be higher than the EPA standard for PCE. At this site the potential impact of the plume is clearly quite spatially variable.

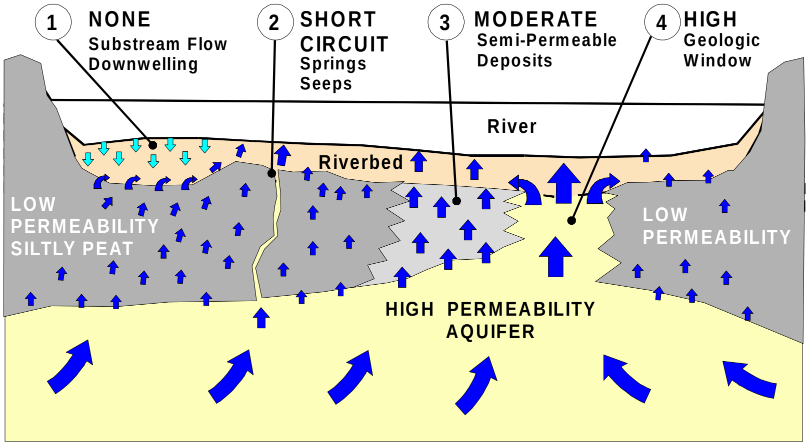

Figure 2. Types of Groundwater Discharge

In terms of the overall plume behavior, it is important to note that the only place where substantial degradation and transformation of PCE is observed is in the last 3 m of the plume's flow path from the source area. Some portions of the plume that have traveled 200 m laterally through the aquifer and arrive at the streambed as PCE, may end up transforming completely and discharging to the surface water column as vinyl chloride or *cis*-1,2-dichloroethylene instead. At this site, water quality monitoring in the aquifer upgradient and immediately adjacent to the river does not fully characterize the type or concentration of contaminants that ultimately enter the surface water.

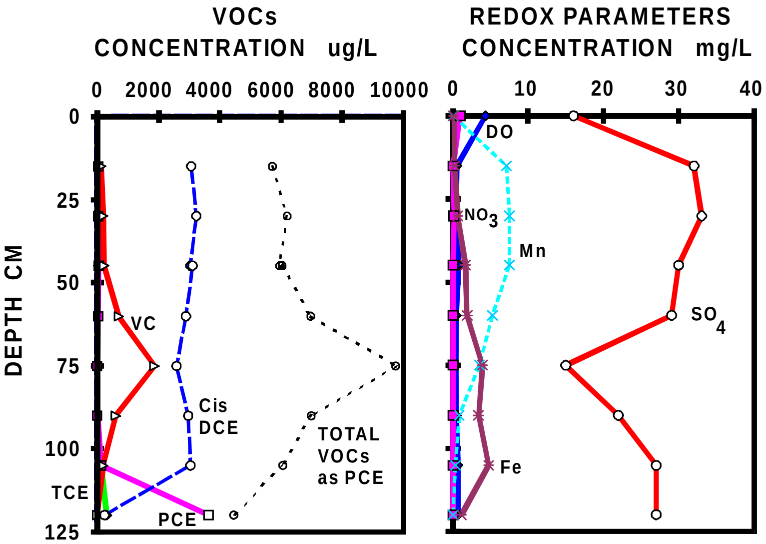

Figure 3. Groundwater Concentrations at a Low to Moderate Discharge Zone

SUMMARY

Determining the location and magnitude of contaminant discharges to rivers from ground-water plumes is a complex hydrogeological and biogeochemical problem. Determining specific ground-water flow paths near a stream and its GWSI is not an easy task. Moreover, the effect of transport and fate processes on the plume near the GWSI and within streambed deposits may be quite different from those observed in the aquifer further away from the stream. Large changes in geochemical conditions and plume concentrations may occur in the streambed over intervals of only centimeters, both vertically and horizontally. Measurements of ground water plume concentrations made adjacent to the stream or in the aquifer underlying the stream banks may not accurately reflect either the concentrations of contaminants in the streambed or the contaminant flux that ultimately reaches the surface water. The Angus study shows that a range of different plume discharge behaviors can occur at a single site and that closely spaced vertical and horizontal water quality sampling is necessary to detect these behaviors. In some places, reactions in the streambed transformed contaminants to daughter products and reduced the overall concentration of contaminants discharging to the river. In other places no attenuation of contaminants occurred and aquatic life in the streambed at these discharge zones had the greatest exposure to the parent compound. Aquatic life in the surface water column is typically less at risk from ground water contamination than benthic organisms because of dilution with clean surface water. The current challenge for hydrogeologists is to assist ecologists in identifying potential problem discharge zones so the toxicological impacts on benthic and hyporheic aquatic life can be assessed.

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Assessment Approaches and Issues in Ecological Characterizations

By G. Allen Burton, Jr. and Marc S. Greenberg

Ecosystems are extremely complex; consisting of a multitude of species that vary widely in the sensitivity to contaminants and who are dependent on each other to varying degrees. Ecosystems are routinely impacted by natural disturbances (e.g., high or low flows, habitat alteration, food availability), some of which can be quite severe and cover over large areas (e.g., hurricanes, flooding, drought, anoxia, temperature shock, invasive species, disease). These natural disturbance events must be considered when trying to ascertain the role of human (anthropogenic) disturbances. Ecosystems are also dynamic and vary through space (spatially) and time (temporally). These variations can be important at the millimeter scale where microenvironments determine nutrient and contaminant availability. However, distances of kilometers may be more significant for biogeographical issues such as forest fragmentation, foraging, and migration. Practical time scale issues vary in importance from minutes to decades. So, when we try and discern whether or not ecosystems are impacted by anthropogenic disturbances, we must do so in the context of these ecosystem complexity issues. The importance of an anthropogenic disturbance, such as exposure to chemicals, follows these natural spatial and temporal processes to a large extent. In other words, the significance of chemical exposure to an organism, population, or community may vary in importance over distances of mm to km and time periods of minutes to years, depending on the organism’s behavior and the chemical’s fate. However, these somber realities of complexity are not insurmountable. The following discussion will show effective ways of determining whether ecosystems are significantly impacted and which stressors are causing the primary problems.

Traditional water quality assessments typically focus on water quality standards, which assume if a single chemical criteria is exceeded then impairment to the receiving water or its beneficial use designation may exist. A limited number of states, such as Ohio and North Carolina, have also developed biocriteria, which rank indigenous fish and benthic macroinvertebrate communities into classifications ranging from poor to excellent. Toxicity testing of surrogate species, such as the fathead minnow (*Pimephales promelas*) and water flea (*Ceriodaphnia dubia*), have been incorporated into the National Pollutant Discharge Elimination System (NPDES) permit program for wastewater effluents. Toxicity testing requirements are occasionally incorporated into a permit and require testing of upstream water, effluent, and near- and far-field receiving water samples. More recently, sediment toxicity test methods have been developed by the U.S. Environmental Protection Agency (U.S. EPA) (U.S. EPA 1994); however, these have not been incorporated into NPDES permits and have been used to only a limited extent in assessments of aquatic ecosystem contamination. Each of these approaches has associated strengths and weaknesses, describing one aspect of contaminant effects under a certain set of exposure assumptions, which may or may not be realistic. These approaches can be used with confidence in situations where gross contamination exists. However, most of our current environmental concerns are more complex and often of a chronic toxicity nature. Often in remediation projects one must decide to what point or level clean-up should extend. In complex watersheds, there often is a need to decipher to what degree each potential source of pollution is contributing to impairment. It is now well accepted by those in the field of ecotoxicology that an integrated approach that combines several traditional assessment approaches, plus other non-standardized methods is necessary to reduce the uncertainty of whether *significant* ecosystem contamination exists (e.g., Burton

1999, Chapman, et al., 1992). This integrated approach is described below in the context of its application to ground water and surface water transition zones.

All ecosystems and their resident species are stressed at one time or another. We tend to focus on that subset of ecosystems where anthropogenic stressors are at issue. Since natural and anthropogenic stressors can be physical, chemical, or biological, the assessment process must consider all of them. Ecosystems, their interacting components, and the stressors which affect them are dynamic and not in equilibrium. So the assessment process must also consider organism exposures to stressors from a magnitude, frequency, and duration perspective. These realities dictate that an integrated assessment contain the components listed in Table 1.

Table 1. Elements of an Integrated Assessment of Aquatic Ecosystems

Component	*Sampling Media*	*Frequency*
Habitat	Drainage area, riparian zone, waterway	Seasonal
Chemistry	Drainage area soil, water, sediment, pollutant sources, and tissues of key receptors	Low and high flow
Biota	Benthos, fish, and fish-eating wildlife	Seasonal
Toxicity	Surface water, pore water and sediment (laboratory and *in situ*)	Low and high flow

Within the four general components of habitat, chemistry, indigenous biota, and toxicity the primary stressors and receptors can be identified with the proper sampling and test design. This approach can follow the ecological risk assessment paradigm whereby there is a problem formulation step, followed by field and laboratory assessments of exposure and effects and finally a risk characterization via a weight-of-evidence approach.

For assessing potential contamination in ground water/surface water transition zones it is critical to team hydrologists, hydrogeologists, aquatic biologists/toxicologists, and environmental chemists in the assessment process. A tiered assessment approach is the most cost effective way to conduct an integrated assessment, eliminating the collection of data which may not be necessary (Table 2). The specific measurement methods that are used in these approaches should be optimized for each study, depending on the problem and questions being asked. For example, in freshwater systems this means optimizing the indicator species used for toxicity testing and response endpoints (e.g., sublethal biomarkers, growth, reproduction, tissue residues, mortality), selecting the appropriate exposure *in situ* (e.g., surficial vs. deep sediments, small mesh to reduce suspended solids, UV blockers to prevent photo-induced toxicity from polycyclic aromatic hydrocarbons), or selecting the appropriate data analysis methods for the benthic invertebrates (e.g., metrics like Invertebrate Community Index, orthogonal comparisons).

Assessing the ecological significance of ground water/surface water transition zones will present some new challenges. Virtually no contaminant effects research has been conducted on biological communities which inhabit the hyporheic zones. It will be critical to establish good reference sites as a point of comparison. These transition zones are particularly important in the storage and retention of nutrients (and possibly contaminants), biological and chemical transformations, as a refugia for invertebrates, and a base of the aquatic food web. Therefore, the measurement endpoints should be focused on determining effects on these traits. Appropriate measurement endpoints could include: indigenous microbial activity, organic matter/nutrient cycling (for more advanced studies), invertebrate

Table 2. Tiered Assessment Approach for Characterizing Ground water/Surface Water Transition Zone Contamination*

Tier 1a:Hydrological characterization of transition zone locations, upwelling vs. downwelling, rates, surface water dynamics.

Tier 1b:Characterization of benthic invertebrates (sediment surface and hyporheous, grabs, colonization, transplants) and habitat quality.

Tier 2: Toxicity testing of indicator species (sediment (laboratory); surface water (high and low flow), surficial sediment and pore water (*in situ*)). Tissue residue analysis of *Lumbriculus variegatas* (*in situ* exposure) and dominant indigenous species.

Tier 3: Site-specific studies to separate physical and chemical stressors with associated chemical analyses, if needed.

* Assumes initial problem formulation process has identified contamination of ground water or surface water with potential transfer to the other.

community indices (meiofaunal and macrofaunal–grab and colonization), tissue residues of dominant species, *in situ* toxicity, and *in situ* physicochemical profiles (e.g., via peepers, datasondes).

If Tiers 1and 2 indicate that the surface or ground waters are toxic and/or are impacting the indigenous community then Tier 3 may be necessary to tease out which stressors dominate at the site. These are very site-specific based designs, but can include novel, yet proven, tools such as ecological food web modeling, semi-permeable membrane devices (SPMDs) to look at bioaccumulation potential, toxicity identification evaluations (TIEs) which fractionate chemical classes for toxicity testing, and stressor identification evaluations (SIEs) which are *in situ* based TIEs but incorporate other physical stressor determinations (Burton et al 1996 and 1998; Greenberg et al 1998), and more detailed characterizations of community effects and exposure dynamics.

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Delineation, Quantification, and Mitigation of Discharging Plumes

By David R. Lee

INTRODUCTION

Methods have been developed for locating and sampling ground water and solute discharge areas on the beds of surface waters. In many settings, these can aid in the assessment of natural attenuation or in estimating the direct flux of ground water contaminants to surface waters. Where plumes are not sufficiently attenuated by natural mechanisms before reaching surface waters, passive subsurface treatment methods, as exemplified by the Chalk River wall and curtain, are now demonstrated at full scale. The purpose of this presentation was to highlight the author’s approach to these problems.

The concept of monitored, natural attenuation sounds good, but putting it into practice will take careful work. If it has been difficult to monitor natural attenuation in relatively simple, well-characterized hydrogeological settings, then it will be even more difficult to perform such monitoring near the beds of surface water bodies. Transients in flow and changes in water levels are only a part of the difficulty.

Another unappreciated difficulty, is the profound influence of geologic heterogeneity on contaminant migration. Heterogeneity can result in orders of magnitude variations in flow within a relatively small volume of earth. Many people believe that hydraulic conductivities at a site vary by factors of 1.2 to 1.5. However, in actual fact at most sites, hydraulic conductivities vary by factors of 10 to 300! Since one of the controls on attenuation is ground-water residence time, attenuation may vary widely across most sites. Therefore, the technical information on which to base an evaluation of attenuation at real sites depends upon the determination of spatial distributions in flow, particularly on finding the faster flow areas at each site.

Measurements of hydraulic gradient can indicate large discharge areas. However, the results of numerous seepage studies have shown that areas of rapid discharge can be small and easily missed. If not located, zones of contaminant entry will not be assessed. In other words, if flow is focused, as it often is, the impacts of the discharge and the processes or evidence for attenuation may have to be monitored within the relatively small, fast-flow areas, which have the greatest potential for poor attenuation and transport of contaminants to surface. If flow rates exceed the required reaction times, the potential for subsurface attenuation may not be realized. High flow areas occur where there are preferential flow paths, such as sand stringers or interconnected zones of higher permeability. The areas where these flow paths intersect surface waters may be overlooked without thorough field work. Even in relatively homogeneous terrain, flow may be highly focused at the shorelines and transport may be rapid.

IS ATTENUATION WISHFUL THINKING OR REALITY?

While it is reasonable to expect some attenuation for many contaminants at most sites, those who seek to monitor attenuation or to measure impact face many pitfalls. Sampling must include the faster flow areas in order for measurements of flow and contaminant concentrations to be representative. If the act of sampling dilutes the ground-water concentrations, and this is easy to do near the sediment/water interface, the sample and the resulting chemical analyses may be inappropriate for contaminant flux calculations. Thus conclusions may be biased and non-conservative as a result of incomplete or improper sampling. It may be easier to find evidence for attenuation than to establish sufficient attenuation.

DEVELOPMENT OF METHODS TO LOCATE AREAS OF SIGNIFICANT DISCHARGE

There is a growing awareness that the application of existing technologies is key to valid monitoring of natural attenuation. One promising method is the sediment probe, a specifically designed for the detection of ground water upwelling (Lee ,1985; Lee and Beattie, 1991). Towed behind a moving boat, the sediment probe is in contact with sediments, and it measures sediment properties. Once areas of ground water discharge have been found and delineated, they may be assessed using traditional, quantitative methods (Lee and Dal Bianco, 1994; Harvey, et al., 1997; Lee, et al., 1999). Traditional methods such as piezometers (e.g., Lee and Harvey 1996; Geist, et al., 1998) may be used for pore water collection and measurement of hydraulic head and conductivity. Under some conditions, seepage meters (e.g., Lee and Cherry, 1978; Lee. 1977; Lee and Hynes 1978) may be appropriate for measuring the flux of ground water across the sediment/water interface.

The sediment probe has been used to find and confirm discharge areas on the cobble sediments and in the 2m/s currents of the Columbia River (Lee, et al. 1999). In that work, quantitative samplers showed that, without exception, probe "hot spots" were areas of ground water inflow and some of these inflows bore contaminants.

The sediment probe has also been used to locate ground water discharge into the shallow ocean (Vanek and Lee, 1991). Other methods have been developed to aid in demonstrating attenuation near the interface (e.g., Lee, 1988; Winters and Lee 1987).

Having been proven in a variety of settings, the sediment-probe method is now ready for use in identifying areas where it may be necessary to monitor attenuation, or the lack thereof. This is essentially a reconnaissance method, a targeting tool. It requires a slight contrast in dissolved solids concentrations between the ground water of interest and the overlying surface water. Where the plume itself is different in dissolved solids, it can tell us, “No, the contaminant is not here,” or “Yes, it is, and, the signal keeps getting larger as we move in this direction.” By applying such methods, it is possible to design a monitoring system for contaminant attenuation and to provide a basis for deciding whether to rely on the process of natural attenuation. Clearly, in order to show that attenuation is sufficient, it must be known where discharge occurs, particularly where it is most rapid, and evidence of acceptable flux of solutes must be obtained.

There is potential for incorporating additional sensors on the sediment probe to make it sensitive to conditions other than electrical conductance.

DEVELOPMENT OF METHODS FOR PLUME MITIGATION

In settings where attenuation is found to be insufficient, subsurface treatment systems, like those first described by McMurty and Elton (1985), can be constructed to enhance natural attenuation mechanisms and therefore minimize impacts on surface waters. An example of such a treatment system is the wall and curtain at the Atomic Energy of Canada, Ltd.’s (AECL) Chalk River Laboratories.

The wall-and-curtain treatment system was installed in 1998 to mitigate the discharge of a strontium-90 plume. In this system, contaminated ground water is directed through a subsurface, permeable, granular curtain of a natural, ion exchange mineral, called clinoptilolite. Based on the results of *in situ* testing, clinoptilolite was highly absorbent for strontium. A bed of clinoptilolite 2 m thick was installed underground. It is predicted to retain the strontium-90 for at least 60 years, during which time its concentration will decay to one-fourth (or less) of the input concentrations. It is expected that this subsurface facility will operate passively at low cost with no maintenance except for the required effluent monitoring. Unlike other methods for subsurface treatment, the wall-and-curtain provides an adjustable capture zone and a single point of flow for checking regulatory compliance (Lee, et al., 1998).

FINAL COMMENT

In the process of exercising these methods at major contaminant sites, I have concluded that two factors have combined to create a vicious circle. The factors are 1.general lack of understanding of ground water-contaminant seepage to surface water and 2. self interest among plume owners The vicious circle is as follows: if there is little proof of a problem and little public understanding, there is little regulatory demand for better information and little funding for developing and applying methods.

Many contaminant plumes have been mapped to the margin of a river, lake, wetland or estuary. But, there is little advantage for a plume owner to map it further unless this is required. Piped effluents must meet or exceed drinking water standards, but there is little enforcement of the same water-quality standards where it is a ground water contaminant plume, not a pipe-flow, that is entering surface waters. Without measurements, there is little understanding and no violations. Or, if measurements at 2 or 3 points looked OK, then the discharge was deemed OK. We humans tend not to seek what we fear we might find. When things are out of sight, they are out of mind.

CONCLUSION

Methods have been developed, applied successfully and have shown the movement of ground water contaminants to surface waters. It is hoped that this workshop will result in broader application of the methods highlighted here and other, equally appropriate, methods that have not been mentioned (my apologies). Hopefully with the issuance of these workshop proceedings, the EPA will begin the task of requiring site-specific evidence where natural attenuation is claimed to be a remedy, but is not monitored, and will require mitigation where attenuation is not sufficient.

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Field Technology and Ecological Characterization of the Hyporheic Zone

By D. Dudley Williams

The hyporheic zone is a 3-dimensional aquatic interstitial ecotone formed within the mixed substrate particles that comprise the bed of a natural, running water channel (Figure 1). It is a middle zone bordered by the surface water of the stream or river above, and by the true ground water below. Although it receives water from both of these sources, the relative strengths of input depend on the configuration of the bed materials and interstitial flow paths, and on the prevailing hydraulic heads. These heads vary spatially and seasonally to alter hyporheic habitat volume and to produce ragged-edged boundaries to the zone (Williams, 1993). Water that flows across these boundaries is subject to changes brought about by distinctive, local chemical and physical properties, microbial processes, and metazoan community dynamics.

Hyporheic research has been progressing at varying rates over the past 30 years, although, recently, progress has been more sustained and intense. Undoubtedly, one of the major factors that limited progress in the 1970s and 1980s was the perception that it is very difficult to sample the hyporheic

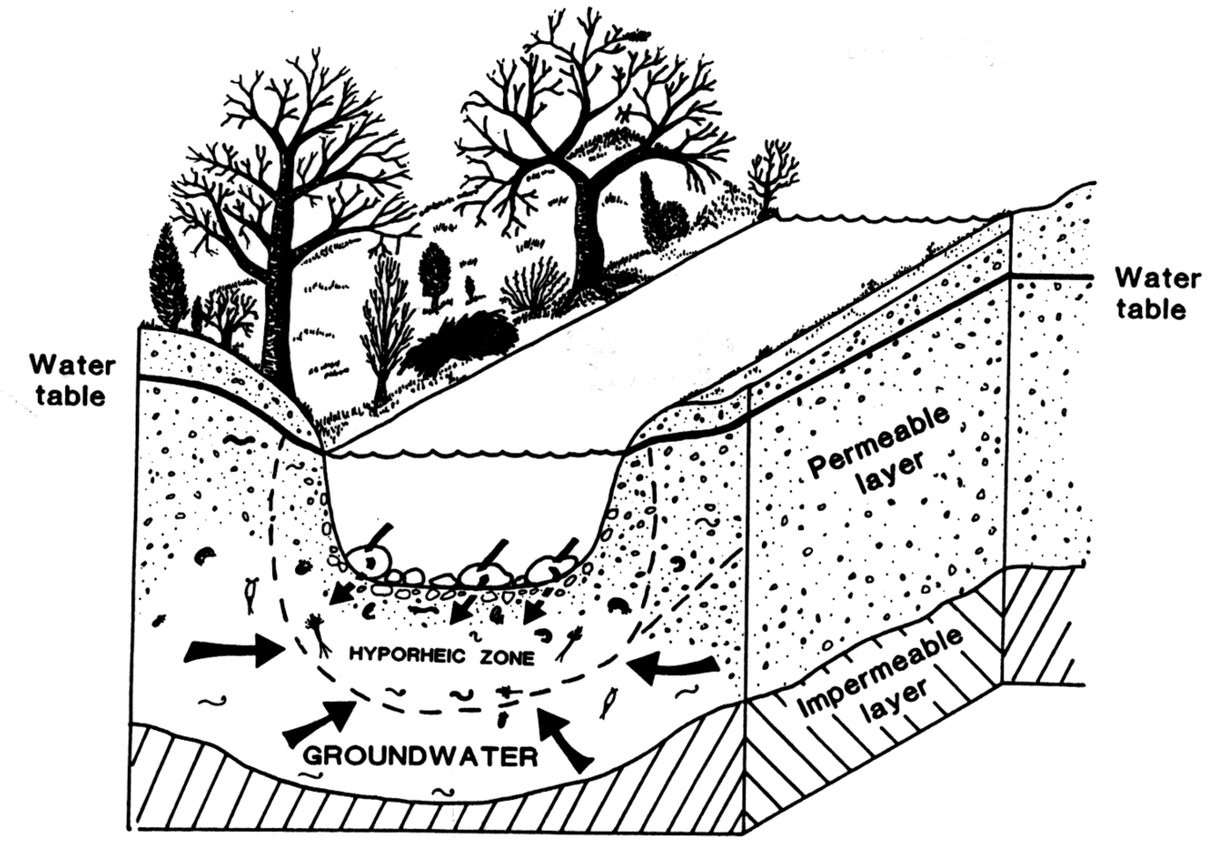

Figure 1. Diagrammatic section through a stream channel showing the approximate position of the hyporheic zone during winter, low flow conditions

zone in any meaningfully quantitative manner. True, extracting largely soft-bodied invertebrates from the interstices among highly heterogeneous and hard lotic bed materials is difficult. However, a sufficient number of techniques now exists (some of them old, but with recent modifications) that makes ecological characterization of this zone possible.

Many running water invertebrates can be collected from hyporheic sediments. Typically, maximum densities may occur around 10 to 40 cm below the streambed surface, but densities of 700 invertebrates per 1 liter of sediment at 100 cm depths are not uncommon (Williams and Hynes, 1974). The hyporheic fauna itself has two main components (Table 1). Differences in the spatial and temporal residence profiles of these two components suggest different functional roles for the two groups within the zone. Although not conclusively proven, there is evidence to suggest that the hyporheic zone may act as a refuge from extreme conditions on the streambed surface (Williams and Hynes, 1977). For example, spates are known to wash benthic organisms downstream as surface substrates are scoured, and droughts and toxic pollutant plumes kill surface-dwelling animals (Hynes, et al. 1974; Williams, 1987). The rapidity with which certain taxa recolonize these denuded substrates has been shown to be due, at least in part, to vertical migration from the hyporheic zone (Dole-Olivier, et al., 1997). Again, the discovery of diapausing nymphs of the cool water-adapted winter stonefly *Allocapnia vivipara* in the hyporheic zone during the summer warm-water phase of temperate streams is further evidence of a refugium (Harper and Hynes, 1970).

Table 1. The two primary components of the hyporheos (after Williams and Hynes 1974).

(**1**) Species derived from hypogean environments such as ground water, subterranean water bodies, and waterlogged soil. These have been dubbed “permanent” members of the hyporheos as they complete their entire life cycles in the interstices. The permanent hyporheos includes rotifers, nematode worms, oligochaetes, mites, copepods, ostracods, cladocerans, tardigrades, and syncarid and peracarid crustaceans.

(**2**) Species derived from the streambed benthos—particularly the early-instar larvae of aquatic insects. These spend only part of their life cycles in the hyporheic zone, having to return to the stream surface in order to metamorphose into a terrestrial, adult stage. These have been dubbed “occasional” members of the hyporheos, although “transient” members may be a better term.

While the hyporheic zone is a fascinating system for the furthering of purely academic enquiry, it also is emerging as an important site for the transformation and storage of nutrients (Triska, et al., 1994). For example, nitrification, a major chemolithotrophic process, occurs in the hyporheic zone, converting the predominant form of inorganic nitrogen in incoming waters from ammonium to nitrate. Although the amount and rate of production of biomass contributed to the lotic food web by nitrifying bacteria are typically lower than those generated by heterotrophs, in streams receiving high levels of nitrogen from riparian agriculture production through nitrification could be quite significant. Similarly, bacterial alkaline phosphatase activity is known to occur in the hyporheic zone, and release of phosphorus from organic P may supply this important nutrient to surface (benthic) and hyporheic biota.

In addition, there is some evidence that lithological and geochemical processes in the hyporheic zone may mediate the availability of N and P (Storey, et al., 1999). For example, substrate particles that have a high cation exchange capacity, as a consequence of their chemical composition and size, will tend to sorb inorganic P and ammonium. In the latter case, hyporheic sediments have the capacity to function as a transient storage pool for dissolved inorganic nitrogen. In these respects, the hyporheic zone should be of interest to water managers and conservationists, as custodians of national water resources.

Hyporheic sampling techniques roughly fall into four categories (Table 2). Unfortunately, virtually all of these samplers have limitations. For example, well digging cannot be used in mid-stream and is not very quantitative; freeze cores may drive organisms away as they form; mechanical corers may have depth or substrate particle size limitations; and artificial substrates may fail to re-establish natural sediment profiles and/or detrital components. Further, many of these samplers have neither been evaluated in more than one location, nor evaluated against each other.

Table 2. The four main categories of hyporheic samplers.

(1) **digging of small wells** in the exposed (above water) areas of gravel bars and stream margins to reach the water table, and then straining the interstitial water so exposed through a fine-mesh net;	Karaman-Chappuis technique, see Schwocrbel (1970) Sassuchin (1930)
(2) **freeze cores** that use chemicals such as liquid nitrogen, liquid carbon dioxide, or a mixture of “dry ice” (crushed solid carbon dioxide) and acetone or alcohol to freeze the substrate around a standpipe driven into the bed;	Efford (1960) Stocker and Williams (1972) Danielopol, et al. (1980) Bretschko and Klemens (1986)
(3) **mechanical corers** that, when driven into the bed, either isolate a sample of the surrounding substratum and its fauna for subsequent removal, or suck up interstitial water and organisms from a desired depth;	Bou and Rouch (1967) Husmann (1971) Mundie (1971) Williams and Hynes (1974)
(4) **artificial substrate samplers** that involve placing a sterilized portion of natural stream bed into perforated containers that are sunk into the bed and then removed after a desired period of colonization.	Moon (1935) Coleman and Hynes (1970) Hynes (1974) Panek (1991) Fraser, et al. (1996)

Recently, we compared the field performance of four hyporheic samplers at a single riffle on the Speed River, Ontario (Fraser and Williams, 1997). These samplers were: the standpipe corer, the freeze corer, a pump sampler, and the colonization corer. Each sampler was assessed, at different sediment depths, for accuracy and precision in terms of total invertebrate density, taxon richness, and invertebrate size distribution.

Since previous studies have concluded that the standpipe corer and the freeze corer, following *in situ* electropositioning, provide good estimates of hyporheic density (Williams, 1981; Bretschko and Klemens, 1986), the *a priori* assumption was made to accept their data as the accuracy standard. Sampler precision was calculated as the coefficient of variation (CV), which is the standard deviation expressed as the percentage of the mean.

In terms of faunal density (Figure 2), the colonization corer estimates were significantly less than those obtained by the other three corers all of which produced very similar results.

In terms of overall taxon richness, there were no detectable differences among the samplers. Further, all of the samplers captured individuals representing greater than 90% of the available taxon pool. For example, all of the samplers captured nematodes, molluscs, ostracods, copepods, mites, mayflies, stoneflies, caddisflies, beetles, and dipterans. However, tardigrades were captured only by the freeze and pump samplers; cladocerans were not captured by the freeze corer; and amphipods were not captured by the pump sampler. In terms of the percent insect larvae captured (another measure of taxon bias), the pump sampler collected the fewest, although this was significantly so only at a depth of 20 cm.

In terms of invertebrate size, as measured by chironomid larval length, there was a decrease with increasing depth for all of the samplers. The only difference detected among the samplers was that, at 20 cm, the pump sampler captured slightly smaller larvae than the other three.

For all four samplers tested and all of the measures compared (density, richness, and size), the level of precision was generally between 20 and 40%, but increased with depth. No sampler yielded a consistently higher level of precision than any other.

The conclusions that may be drawn from this comparative study are:

(1) All four samplers would suffice for collecting purely qualitative data.
(2) In terms of removing an exact, representative portion of habitat (to obtain absolute measures), only the freeze corer qualified. However, and in support of the *a priori* assumption, no statistical differences were detected between this sampler and the standpipe corer for any of the measured variables, at any depth.
(3) The colonization corer consistently underestimated total invertebrate density.
(4) The pump sampler was capture selective both in terms of invertebrate type and size - the bias towards non-insects and smaller insects probably reflecting a filtering effect of the interstices.

As to recommendations for possible standardization of hyporheic sampling are concerned, pragmatically the goals should determine the means. Some examples are given in Table 3. Regrettably, the holy grail of a perfect hyporheic sampler still seems to evade us and, indeed, may never be attainable. Nevertheless, samplers do exist that allow acceptable levels of sediment description, water sampling, and faunal characterization to be made—although perhaps not through one apparatus alone. Such techniques have the potential, either singly or in combination, to help researchers answer some of the sophisticated questions that 30 years of hyporheic study is now demanding.

Table 3. Examples of hyporheic samplers suited to specific information goals

(1) If **survey information** is required, relatively quickly, on invertebrate densities and types at a variety of depths, then the standpipe corer would be suitable. This corer has been shown to produce a mean error density estimate of around 19%, and captures virtually all of the common taxa found in the hyporheic zone (Williams, 1981). Both Cummins (1975) and Elliott (1977) have suggested that this level of accuracy is acceptable in estimating benthic densities, and so perhaps the same should be applied to the hyporheos.

(2) If a **larger sample volume, together** with a description of invertebrates and the undisturbed sediments in which they live, is required, then the freeze corer (preceded by electropositioning) would be the choice.

(3) If **periodic assessment** (with moderate precision) of the hyporheos is required from a particular site, with minimal, long-term habitat disturbance, then the colonization corer would be appropriate - especially if routine hydrogeological and chemical data are needed also.

(4) The colonization corer also would be the most suited to **manipulative studies** of hyporheic dynamics—as it allows different combinations of hyporheic sediments (e.g., particle size and/or organic content) to be presented for colonization.

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DISCUSSION GROUP SUMMARIES

Hydrogeology Discussion Group Summary

By Thomas C. Winter and Joseph Dlugosz

INTRODUCTION

The great variety of sediment types in the beds of most surface water bodies results in substantial variability in the location and rates of seepage across the bed. The exchange of water between ground water and surface water ranges from slow, diffuse seepage to rapid, concentrated flow at specific localities. Determining the location, rate, volume, and chemistry of water moving between these two components of the hydrologic system is difficult, expensive, and highly uncertain. Nevertheless, the need for understanding the hydrologic processes and measuring the interaction of water and dissolved chemicals between ground water and surface water is fundamental to environmental management. To address these challenges and needs, the hydrogeology discussion group focused on the hydrogeologic aspects of understanding and measuring the interaction of water and dissolved chemicals between ground water and surface water at sites where ground water has been contaminated.

To focus the discussion on the interface between ground water and surface water, the group made several presumptions: (1) the hydrogeologic framework of a site has been defined; (2) the source area of the contaminant is known; (3) the flow pathways and plume configuration are reasonably well defined; (4) the chemical characteristics and decomposition products of the contaminants are known; and (5) the contaminant is a potential threat to the environment. Given this information, it was suggested that the actual determination of the movement of ground water to the surface water body could be accomplished through a tiered approach: A sequence of actions could be followed that begins with a general reconnaissance of observable indicators of ground-water discharge and evolves to very detailed and focused sampling of hydraulic head, chemistry, and biology.

This summary of the discussion group presents:

(1) Field methods that can be used for (a) reconnaissance of observable qualitative indicators of ground-water discharge to surface water, (b) direct measurement and calculated flow of water between ground water and surface water using physical data, and (c) indicators of flow between ground water and surface water using chemical data;

(2) Considerations for temporal sampling of water flow and chemistry; and

(3) Variations of field sampling strategies that may be needed in different hydrologic landscapes.

The material presented here is considered to be a supplement to another EPA report (U.S. EPA, 1991) that presented a review of methods for assessing non-point source contributions of contaminants to surface water. Some of the information presented briefly in this summary is discussed in much more detail in the EPA report.

FIELD METHODS FOR DETERMINING THE INTERACTION OF GROUND WATER AND SURFACE WATER

Observable Qualitative Indicators of Ground-Water Discharge to Surface Water

Many indicators of ground-water discharge to surface water can be used to determine specific localities where a contaminant plume may be entering a surface-water body. The most common indicators are seeps and springs; infrared mapping; aquatic plants; phreatophytes; unique sediment zones such as mineral precipitates; water color; odor from contaminants; and mapping of lineaments in fractured-rock settings. It was suggested that a field reconnaissance of these easily observable characteristics would identify specific localities where detailed measurements and sampling could be focused. If the skills of biologists are available, benthic organisms also can be useful indicators of ground-water discharge.

Observation of seeps and springs is relatively straightforward if the flow rates are high. In fractured-rock landscapes, mapping of lineaments can be useful if the fractures are open. Ground-water flow concentrated in the fractures enter surface-water bodies as springs. In settings where seepage rates are low, it is easier to observe seeps during colder times of year when ground water and air temperatures are considerably different, because the water vapor above seeps is visible. Furthermore, in climates where surface water freezes or snow is on the ground, areas of appreciable ground-water inflow remain open. The difference in temperature between ground water and surface water also makes infrared mapping a useful reconnaissance tool, especially in mid-summer when the difference in temperatures of ground water and surface water are at a maximum.

Some chemical constituents dissolved in anoxic ground water precipitate upon contacting oxygenated surface water. For example, iron and manganese oxides are common indicators of seep areas. Contaminated ground water commonly has color and odor. Water color and odor from contaminants can be used as an indicator of ground-water inflow, especially if the inflow consists of the contaminated water.

Aquatic plants can be indicators of ground-water discharge. The following are a few examples: (1) Swanson, et al. (1984) indicated that cattails are indicators of fresh ground-water input to saline prairie lakes in North Dakota, (2) Rosenberry, et al. (in review) indicated that Marsh Marigold was an indicator of springs in Minnesota, (3) Lodge, et al. (1989) indicated that submerged aquatic plant biomass was greater where ground-water inflow velocity was greater, and (4) Klijn and Witte (1999) discussed the relationship of plants to ground-water flow systems. In addition to aquatic plants, upland phreatophytic plants near a surface-water body are indicators of the presence of ground water at shallow depths.

Benthic organisms can be indicators of ground-water discharge to surface water. Numerous examples of the relationship of organisms to water flow and chemistry are provided by studies of the hyporheic zone beneath streams. With respect to lakes and wetlands as well as streams, ostracods are especially useful because they have specific tolerances to water temperature and chemistry. An additional benefit to using ostracods is that some of the chemical constituents and isotopes that are present in the water while the organisms are alive are incorporated into their shells. Therefore, study of ostracod shells in sediments can provide a valuable record of past ground water and surface water relationships.

Direct Measurement and Calculated Flow of Water Between Ground Water and Surface Water Using Physical Data

The reconnaissance methods discussed above may be useful for identifying locations of ground-water inflow to surface water, but they do not indicate the quantities of water that move across the interface. Measurement of water quantity can be done by (1) using instruments that directly measure the water flux, or a physical or chemical property from which flux can be calculated, at the specific locality of the instrument (herein referred to as direct measurements); or (2) calculating the flux over a broader area of surface-water bed using streamflow data or ground-water flow nets. A drawback of direct measurements is that they sample a point in space, and, because of the great variation in sediment types in most surface water beds, measurements need to be taken at many places in the bed. Furthermore, most measurements are taken at a point in time because the devices generally are not equipped with recorders. For these reasons, it also is desirable to calculate the flux through broader areas of surface-water beds to obtain independent estimates of flux. This approach averages out the spatial variability of flux and it provides a check on values determined by direct measurements.

Direct measurements: Methods for directly measuring the flux of water between ground water and surface water include the use of seepage meters, mini-piezometers, temperature profiles in the sediments, heat-flow meters, hydraulic properties of sediments determined from cores, and direct-contact resistivity probes. Although these were considered by the discussion group to be methods for direct measurements, only seepage meters can be used for direct measurements of water flux. The other methods use devices that make direct measurements of hydraulic head, hydraulic conductivity, temperature, or electrical conductance, and the water flux then needs to be calculated from these data.

Seepage meters are chambers (commonly, cut-off 55-gallon drums) that are set on the bed of a surface water body (Lee, 1977). After the chamber is pushed into and allowed to settle into the sediment, a tube is inserted into an opening in the top or side of the chamber. The tube has a small bag attached at the end and a valve positioned between the chamber and the bag. The bag can be attached empty if ground water is known to be seeping in, or filled with a known volume of water if the direction of seepage is unknown or if it is known that surface water is seeping out. To measure the flux, the valve is opened and the change in water volume in the bag over a given period of time is a measure of flux per that period of time. Seepage meters are perhaps the most commonly used devices for measuring water flux between ground water and surface water, and different sizes and types of chambers other than 55-gallon drums have been used. A number of studies have evaluated the uncertainties in using the seepage-meter method for determining flux through surface-water beds (Shaw and Prepas, 1990; Belanger and Montgomery, 1992). Seepage meters have been used largely to make discrete measurements at a point in time, but a recording seepage meter was developed recently by Paulsen, et al. (unpublished manuscript) using ultrasonic flow technology.

Mini-piezometers are used to determine the hydraulic gradient between a surface-water body and the ground water beneath it. A small diameter well is inserted into the surface-water bed, and, in the most common design, a flexible tubing is attached from the well to a manometer board. Another piece of tubing is attached to the other side of the manometer and the other end is placed in the surface water. Both ground water and surface water are drawn into the manometer using a hand pump. After air is bled back into the manometer and the water levels in each tube stabilized, the difference in head can be measured directly (Lee and Cherry, 1978; Winter, et al., 1988). The difference in head between ground water and surface water can also be determined simply by measuring the level of ground water in the well and the level of surface water outside the well. Mini-piezometers provide data only on hydraulic gradient. To determine water flux, hydraulic conductivity of the sediments need to be determined as well as the cross-sectional area of the flux.

The transport of heat by flowing water has been used to determine the interaction of ground water and surface water. By measuring the temperature of surface water and the temperature at shallow depths in sediments, Silliman and Booth (1993) mapped gaining and losing reaches of a stream in Indiana. Sediment temperatures had little diurnal variability in areas of ground-water inflow because of the stability of ground-water temperatures. Sediment temperatures had much more variability in areas of surface water flow to ground water because they reflected the large diurnal variability of the surface water. This approach is useful for determining flow direction. Lapham (1989) used sediment-temperature data to determine flow rates and hydraulic conductivity of the sediments based on fundamental properties of heat transport. Heat-flow meters, consisting of a heating element and a ring of temperature sensors, placed at a distance from the heater, have been used to measure the rate and direction of water movement through sediments. A pulse of heat is applied to a heating device and the rate and direction of water movement is determined by measuring the time it takes for the heat pulse to be sensed by the thermistors in the direction of flow.

Hydraulic properties of sediments can be determined by laboratory studies of sediment cores. These data can then be used to calculate ground-water flux if the hydraulic gradient and area of surface-water bed through which the water flux is taking place is known. Probes that measure electrical resistivity have been used to locate contaminant plumes entering surface water. These probes are most effective if the conductance of the contaminant is substantially different than the conductance of the ambient ground water.

Calculated from streamflow data and flow nets: The quantity of water moving between ground water and surface water over scales larger than can be determined by direct measurement using individual sensors generally is determined by stream discharge data or by ground-water flow nets. The most direct method for determining ground-water inflow or stream losses to ground water is to make stream discharge measurements at different locations along a stream. The difference in discharge between two localities is the quantity of gain or loss of water for the reach of stream between the measurement sites. The accuracy of the values is related almost entirely to the accuracy of the discharge measurements.

The flow-net approach is probably the most common method used for determining the interaction of ground water and surface water. The term flow net is used broadly herein as any calculation of ground-water flux, including simulation models, that makes use of a network of wells for determining hydraulic gradients, estimates of hydraulic conductivity of the geologic units and sediments, and cross-sectional area of the interface of ground water and surface water. The accuracy of the values is related to the quantity and quality of the hydrogeologic data, and the grid spacing that is justified by these data.

Indicators of Flow Between Ground Water and Surface Water Using Chemical Data

The chemistries of ground water and surface water commonly are different enough—especially at contaminated sites— that some chemical constituents or isotopic properties of water can be used to determine the interaction of ground water and surface water. Devices for collection of water samples for determination of the chemical characteristics of water passing through sediments consist of two basic types: (1) collection at the sediment-water interface; and (2) collection at various depths in the sediment by inserting a device into the sediments.

Constituents: Nearly all chemical constituents have the potential to be useful in determining the contribution of ground water to surface water. By calculating mass balances of the constituents, the flux of water can be quantified. Isotopes of some elements, such as nitrogen and radon, are particularly useful because in some cases a specific contaminant source can be identified. Isotopes of water are among the most useful because they are part of the water molecule itself and are not subject to modification by chemical reactions. The age of ground water can be determined by analyzing for tritium and chlorofluorocarbons, which are useful for identifying ground-water flow paths.

Sampling at the sediment-water interface: Devices that have been developed for sampling water at the sediment-water interface include drag probes, seepage meters, diffusion bags, bubble collectors, and biosensors. Of these devices, seepage meters are the only ones that actually collect a water sample large enough to be analyzed in the laboratory for many constituents. Furthermore, by knowing the water flux, the flux of a constituent or isotope can be calculated. Drag probes, such as used for measurement of temperature, specific conductance, and radioactivity, are used primarily to locate areas of inflow. Vapor diffusion samplers are placed in the sediments and can collect certain contaminants that diffuse into the bag, and they also can measure microbiological activity through the production of hydrogen. Devices that collect gas bubbles are used to determine the chemical constituents in the bubbles, which are an indication of the gases being produced in the sediments.

Sampling at depth in sediments: Devices that have been developed for sampling or measuring water chemistry at depth in sediments consist of (1) multi-level samplers that are driven into the sediments; and (2) probes through which individual samples can be drawn from any depth—or a constituent measured—but can then be driven deeper to collect samples at other specific depths. Examples of the first are pore-water peepers, gel samplers, and multi-level samplers. Pore-water peepers are blocks of plastic that have chambers machined into them at specified intervals (Hesslein, 1976). A porous membrane is placed over the chambers and held in place by another cover of plastic that has holes machined at the same intervals. The chambers are filled with deionized water, and the device is driven into the sediments. The device is left in place for a period of time for the chemicals to diffuse across the membrane and equilibrate with the ambient pore water (usually weeks). The device is then removed and the water in the chamber is extracted and analyzed. Gel samplers are similar, but the collection device is a thin film of polyacrylamide gel that is placed on a flat Perspex probe, covered with a porous membrane, and held in place by a thin plate that has a window cut the full length of the probe (Krom, et al., 1994). The device also is driven into the sediments and left to equilibrate (usually only minutes to a day). After equilibration, the device is removed and the gel sectioned at any desired interval to obtain the samples.

Multi-level samplers are rigid tubes that have ports machined into them at specified intervals. Flexible tubing is attached to each port and brought to the surface. Water samples can then be drawn from individual ports using a pump at the surface. Squeezing or centrifuging pore water from segments of sediment cores can also be considered multi-level sampling. Of these methods, only rigid-tube multi-level samplers can be used for repeated sampling of precisely the same location and depth because the device can be left in place.

Examples of probes through which water samples can be withdrawn, or a measurement made, from a single depth and then pushed deeper to collect other individual samples include mini-piezometers and Geoprobes.

CONSIDERATIONS FOR TEMPORAL SAMPLING OF WATER FLOW AND CHEMISTRY

The time interval for sampling water flow and chemistry depends on the phase of the program, physical characteristics or chemical constituents of interest, climatic setting, and hydrogeologic setting. In general, more sampling is needed in the initial phases of a program when the extent of a problem is being determined, and less sampling is needed for long-term monitoring. For example, it commonly is desirable to continuously monitor water flow and hydraulic head in the initial phases of a study to characterize the variability on daily, seasonal, and annual scales. At the same time, sampling for chemical constituents also needs to be done more frequently at this time to relate the concentrations and mass transport of constituents to flow regime and to climate. Once the relationship of mass transport to flow and climate is reasonably well understood, the frequency of sampling can be reduced.

Hydrogeologic setting comes into play in sampling frequency because some settings are inherently more simple, thus easier to characterize and monitor flow and chemistry, than others. Similarly, the climate that drives the hydrologic system is much less variable, thus easier to characterize and monitor, in some regions than in others. If a sampling program includes biological factors, sampling frequency may need to include considerations related to the life cycles of the organisms.

An important climate consideration in both initial site characterization and long-term monitoring is the effect of extreme climatic events. Extreme climatic events, such as droughts and deluges of precipitation, can have a greater effect on a site than many years of more normal conditions. These effects include rearrangement of bed sediments, changes in water flow paths, mass-transport of chemicals, and biological conditions of a surface water bed. One catastrophic event can greatly alter the perception of how well a hydrologic system is understood, and how it should be managed or mitigated. Although difficult to anticipate, a plan for sampling during catastrophic events should be in place.

VARIATIONS IN FIELD MONITORING AND SAMPLING STRATEGIES FOR DIFFERENT HYDROLOGIC LANDSCAPES

A generic field design for determining the interaction of surface water with ground water includes the use of piezometer nests, water-table wells, and devices to measure or calculate the flow of water and chemicals across the surface-water bed. The conceptual model in Figure 1 of the Executive Summary shows ground-water seepage inflow on one side of the surface-water body and surface-water seepage out on the other. Actual conditions could be as indicated, have ground-water inflow on both sides, or have surface-water seepage out on both sides. The important point of the diagram is to stress that the interaction of ground water and surface water can be reasonably well understood only by addressing the larger-scale processes related to the position of the surface-water body within ground-water flow systems as well as the smaller-scale processes related to geology of the surface-water bed and climate.

The advantage of having permanent installations, such as wells and piezometers, in the upland is that they can be easily equipped to obtain continuous records. The disadvantage of having these installations is that they do not indicate the precise location or chemistry of seepage across the sediment-water interface. The advantage of the devices used within the surface-water body is that they can be used to pinpoint the location, rates, and chemistry of seepage water. The disadvantage of using these devices is that few can be used to obtain continuous records. Furthermore, few devices used within the surface-water body can be left in place for long periods of time because of floods, currents, ice, and water safety.

Although the generic field design may be applicable to many actual field settings, it is conceivable that the design would need to be altered somewhat for different hydrologic landscapes. For example, some landscapes, such as riverine and coastal, have wetlands at the base of terraces in the uplands. If a source of ground-water contamination was located on the terrace, the contaminant plume could conceivably discharge to the wetlands at the base of the terrace. In this case it would be desirable to place an additional piezometer nest in the wetland. Other modifications to the field design might be related to the geologic complexity of the site. For example, if the geologic framework has a series of aquifers and aquitards or lateral geologic discontinuities, it might be necessary to place piezometers in the different geologic units in order to better understand the ground-water flow paths.

Frequency of sampling for chemical constituents also would depend on hydrologic setting. For example, in coastal areas affected by tides, the water flow and chemical transport paths could be greatly affected by the tidal exchange and storm surges. In northern and mountainous areas, runoff and ground-water recharge from snow melt can have a substantial effect on ground-water flow paths and chemical transport.

Because of the variety of hydrologic landscapes and the variability of climate, a need exists for development of type localities that would become benchmarks for the various landscape types. At these type localities, design of field installations, effectiveness of various sensors and devices, sampling frequency, and study and site characterization approaches could be tested and evaluated. Such knowledge could lead to efficient and cost effective approaches to dealing with contaminated sites in the hydrologic landscapes represented by a given type locality.

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Chemistry Discussion Group Summary

By Allen Burton and Ned Black

INTRODUCTION

The chemistry discussion group agreed to adopt the broad term ground-water/surface-water transition zone," unless it was specifically addressing the classical stream hyporheic zone. In this summary, individual topics that were discussed frequently over the course of the day are summarized under single headings. The group's discussions sometimes veered into issues belonging to the biology discussion group, such as the importance of establishing clear reasons for adding the transition zone habitat to the risk assessments performed at contaminated sites. Some group members expressed concern that project managers should establish the justification for sampling a transition zone site (e.g., complete pathways to receptors) prior to extensive use of the sampling and analytic techniques we discussed.

An obvious—but important—point to remember is that the contaminants in question are the same ones (e.g., dissolved or NAPL chlorinated solvents and petroleum hydrocarbons, pesticides, dissolved or particle-bound metals) that we encounter in contaminated ground waters and surface waters. Thus, we need to collect information on the same parameters we use to predict the geochemical fate of these contaminants in both ground water and surface-water bodies. We also need to collect the chemical and physical information commonly used in ecological risk assessments and natural attenuation assessments to determine the dominant biological processes and the potential confounding factors in bioassays. Finally, we need to collect chemical information which helps locate zones where a ground-water plume or hyporheic flow is entering a surface-water body. There is overlap among these parameters, but we should remember the three different uses of chemical information:

1. Contaminant chemistry and fate
2. Biological processes
3. Identification of flow paths

The transport of dissolved contaminants from surface water into the subsurface through hyporheic flow or ground-water recharge from a losing stream was included in our discussion of the transition zone. With regard to flow paths and sources of contamination, the deposition of contaminated sediments was excluded from our discussion. Other groups within EPA are addressing the issue of contaminated sediments.

It is possible to list many chemical and physical parameters (see below) to measure in order to satisfy the three information needs listed above. As for any ecological risk assessment, a screening process will determine what level of site chemistry characterization should be performed. In other words, it is not necessary to collect the same information at all sites. In order to justify extensive work on a site, a screen must demonstrate the presence of contaminants at levels sufficient to present risk to actual or potential receptors. For the chemistry discussion group, screening information also included parameters for determining site geochemistry and contaminant flow paths, although collection of this information might be deferred until after a screen.

One or more standard conceptual model should be developed to identify the important questions to ask and data to collect at different types and scales of sites. Sampling efforts in the transition zone may be more costly than standard sampling of surface water or shallow ground water. At the very least, project managers and responsible parties familiar with only surface waters or only ground water will have to be taught to use different tools.

LIST OF PARAMETERS AND TOOLS

Screening Tools

- Semi-permeable membrane devices (SPMDs)—
  Widely accepted as a presence/absence screening tool. Requires extensive calibration (e.g., of equilibration times) and sensitivity analysis to determine exact concentrations. EPA researchers, in cooperation with other government or academic scientists, should perform sensitivity experiments to determine if there are situations where SPMDs can be easily used to measure concentrations.

- Drag probes for temperature, conductivity, and gamma anomalies—
  Useful in lakes, estuaries, and large rivers to determine zones of ground-water discharge.

- Piezometers and mini piezometers—
  Multiple piezometers with low-flow sampling can provide adequate samples of transition zone interstitial water and, of course, ground water. In order to sample just the transition zone, extreme care is required in depth placement of the screens. Piezometers can be placed both on land and in stream or lake beds.

- Freeze sampling techniques—
  Typically used to obtain biological samples, but could also be used to sample water and substratum for chemical analysis.

- Colonization corers—
  Also a biological sampler, but can incorporate nested piezometers.

- Bead pipes (ceramic beads).

- Dye tracers of ground-water and stream flow.

- Walk river bed with a hand auger.

- During low flow, note odor and visual observations.

- Photoionization detector (PID).

- Passive diffusion samplers.

- Analyze bubbles of gas (marsh or lake setting).

- Multi-level samplers.

- Seepage meters.

- Cores (solids analysis and visual).

- Laser-induced fluorescence (LIF), qualitatively determine VOC presence, BTEX, SVOCs, dense non-aqueous phase liquids (DNAPLs).

- Cores of trees (For instance, in a mangrove swamp. However, the contaminant may actually be metabolized in roots so false negatives are possible.).

- Field chemistry with a HACH spectrometer (nitrate, ammonia).

- Chemetrics for sulfides.

- Differential global positioning system (GPS).

- Velocity meter.

- Tidal stage.

Post-Screening Tools

- Multi-level wells.

- Everything on screening tools list.

TIME SCALES

Hyporheic and transition zone chemical and biological processes follow several different time scales. At a minimum, these can be described as daily cycles (e.g., temperature and river stage), normal weather changes, invertebrate and fish life cycles, seasonal changes and long-term climatic changes and events (such as extreme weather events). The difficulties of meshing the natural time scales of the environment with our schedules for sampling contaminated sites are shared with risk assessments and cleanups at all outdoor sites. Clearly, an environment such as the transition zone with strong diurnal and seasonal controls on biology and chemistry requires multiple sampling events if we desire great confidence that all pertinent processes are understood. And just as clearly, constraints on sampling budgets and the desire of regulators to respond to contaminated sites with an appropriate level of effort make limited sampling schedules the overwhelming norm. The most protective option may be to plan our sampling to coincide with the expected worst-case time of day and season. For the transition zone in a variety of habitats, the worst case sampling time may not be known. Thus, one of the mandates of the Regional study areas recommended below will be to determine the worst (i.e., the best) times to sample. For some transition zone habitats, recognized international experts will be able to offer suggestions for sampling schedules.

SPATIAL CONSIDERATIONS

As with a ground-water plume, the spatial extent of contaminants is important information. For sites with a contaminant plume flowing from the subsurface into a water body, the effect in the transition zone may be limited to a discrete discharge zone. Also, the discharge zone for a contaminant plume may occur some distance from shore. An effective way to locate a discharge zone is to sample along a series of transects in the ground water. For a stream, it is also important to sample the bank opposing the discharge area. It must be remembered that a ground-water plume can flow entirely under a stream without any discharge. For classic hyporheic transport parallel to the flow of a stream, discharge can occur anywhere in the bed. For a site with a hard substratum, the impact of the contaminants will be in the open water column. Although contaminants so discharged are an environmental problem, the impact on the transition zone, or the exact nature of the transition zone itself, may be hard to define. In lakes, zones of discharge from and recharge to ground water can occur in complex patterns.

CONCENTRATION AND FLUX

In a screening or predictive risk assessment, contaminant concentrations are used for comparisons to toxicity benchmarks. However, the flux, or loading, of contaminants is also important information that bears on both the impact of the contaminants on the habitat and on the physical, chemical, and biological transformations of the contaminants at the transition zone. The flux of contaminants can change in magnitude and direction with changes in surface water temperature and flow stage.

DETECTION LIMITS

The issue of detection limits for transition zone sampling is the same as for all other sites subject to risk assessments. Before a sampling and analysis plan is developed, the exact values of the toxicity benchmarks to be used for screening purposes must be determined. Otherwise, the sampling budget may be used to collect information of no use to the risk assessors.

RECOMMENDATIONS

EPA should create a series of Regional study areas of contaminated transition zone sites, with appropriate uncontaminated reference sites. These would be studied by EPA Regional and ORD laboratories and academic grantees. The sites should be scaled appropriately to the typical sites for the Region. For instance, the hyporheic chemistry, biology, and hydrology of small mountain streams impacted by mines could be very different than a zone of chlorinated solvent-contaminated ground-water discharge in one of the Great Lakes. Ground-water discharge and hyporheic flow in estuaries will have the further complicating factor of tides. Sites of all sizes will be encountered by the Agency. Members of the chemistry discussion group felt strongly that extrapolating from small streams to large rivers and lakes is unacceptable. Also, some methods work in small streams, but not in areas of high flow. As with any landscape approach, the species and the dominant chemical and physical processes of the environment change with different landscapes.

Biological Discussion Group Summary

By Cliff Dahm and Bruce Duncan

This session opened with the following question: “Is the hyporheic zone considered an ecological habitat to be protected or a ‘treatment opportunity’ zone for restoration of contaminated ground-water discharges to surface water?”

The group agreed early in the discussion to define the zone of interest (the ground-water/surface-water transition zone) as the “transition zone” rather than use the term “hyporheic zone,” which has a more restricted meaning where surface waters and ground waters are actively mixing. Mixing in this zone is very important, and in a stream, surface water moving into this zone can return back to surface water within a short distance and be “processed” through the transition zone multiple times.

An early question raised by the participants was how the zone can be defined biologically in order to focus on and demonstrate exposure of organisms. This requires more than a hydrological definition. There also is a need to link the transition zone to valued resources, such as fish. If there is an impact on the meiofaunal community, does that affect trout? This characterization of food web links, which is needed to demonstrate risk and answer the question “who cares?,” led to two important points: (1) What are the important services that this zone performs? and (2) if these services are impaired, how can we make that determination? Superfund managers now accept the importance of benthic macroinvertebrates to stream ecosystems; there is not the same recognition for organisms such as meiofauna or microbes in the transition zone.

Scale was another concern. There is a need to look at the spatial extent of impact to assess whether the contaminant discharge results in a risk to critical habitat such that action is warranted. Some hydrogeologists expressed frustration that they already know there is contamination in upwelling areas, but biologists countered that: (1) we do not know what the “pristine” state should be; and (2) even if the contamination is not cleaned up, there are other communities in other parts of the stream. So would analysis of the transition zone really matter? One attitude was: If someone is discharging without a permit, then they are in violation. “Who cares” is not an issue. Often, “no action” is what happens because an adverse impact cannot be demonstrated over a realistic scale.

A concern was raised about the reluctance of managers to invest in studies of transition zones. Given that we are not successful in getting biological measurements in ground water or surface water, how can we convince managers to do biological measurements at the interface? How do you convince someone that the transition zone is important when there are competing resources requiring protection? The solution is to demonstrate the functions that occur in the transition zone and what happens when those functions are lost.

The *Guidelines for Ecological Risk Assessment* [1] should be used to evaluate the transition zone:

- Who is present or affected? What do stakeholders care about in the system? What are the management goals (some are predefined such as no net loss of wetlands, or meeting Ambient Water Quality Criteria)?

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[1] Risk Assessment Forum. EPA/630-R-95/002F, April 1998. 171 p. http://www.epa.gov/ncea/ecorisk.html.

- Identify the assessment endpoints (i.e., some biological entity or function that you care about), the exposures, the measurements to be made, and then the effects. The ecoological risk paradigm should cover everything and help maintain a big picture perspective.

The link between contaminated sediments and contaminated ground water in the transition zone was another issue. How is the issue of contaminated ground water different from the issue of contaminated sediment? The biological definition of the transition zone does not cover this; change in chemical conditions and rates are needed as well.

There are examples where removal of contaminants occurs within the transition zone with no removal in ground water. Ground-water wells cannot reveal the full story. More thought should be put into field sampling of mobile contaminants. You cannot just sample sediment. For example, you might have sand that appears very clean, but has contaminated ground water moving through it. Sediment and water are part of a system and need to be dealt with together, not separately nor sequentially. Also, there is a need to consider the contribution from contaminated sediments (top down) into the ground water. Sources need to be distinguished because of the polluter's perspective.

During the presentation session on the first day of the workshop, the following questions predominated:

- Why should we be interested in biology?
- Why should the public care or be interested?
- What are the services and processes that the transition zone provides?
- Why is the transition zone important ecologically?
- What biogeochemical measures would be ideal?

Participants were interested in contaminant migration and fate; others were interested in the effects on biological resources (macrobiota, communities, microbial processes) in the transition zone. When considering applicable biological measures, the biological discussion group had difficulty identifying microbial measurements with broad applicability. There is good success with macroinvertebrate indicators, less so with microbiota and meiofauna. A multidisciplinary approach is needed to provide synergy.

The discussion followed three aspects of the transition zone: (1) Why is the zone important ecologically? (2) What are the methods that can be used to assess ecological structure and function? and (3) What research is needed to better determine the ecological importance of this transition zone and to develop needed tools for sampling this zone?

WHY IS THE TRANSITION ZONE IMPORTANT ECOLOGICALLY? WHAT ECOLOGICAL SERVICES ARE PROVIDED?

These issues led to additional questions: Do all transition zones need to be protected, especially if you see no impact to the surface water? Is there intrinsic value to the transition zone itself, apart from the surface water? Historically, people study "ecological entities." The recent trend is to look at transition zones or ecotones. We do not know much about ecotones as an ecosystem entities. The hyporheic zone is one important ecotone. Some surface organisms have a phase in the hyporheic zone, which implies that productivity could be affected. The hyporheic zone also serves as a "nursery" for secondary producers. Less is known of the permanent hyporheic zone species—they often can be distinct, undescribed species.

The group discussed the importance of transition-zone function and compared it to wetlands. Regulations require restoration of wetlands if they are destroyed. This concept could be applied to the transition zone; the goal could be to restore function rather than restore appearance (no net loss). It was pointed out that we need both function and structure (species).

Another question was “why should the public care about the important function of microbiology?” or “what would the environmental effect be from the loss of that function?” Several structural and functional elements are extremely important in this system. Transition zones often provide high quality habitat and are sites of contaminant reduction and nutrient and carbon cycling. A good example was made for fish. Three major biological services are tied to fish: refugia, food sources, and reproductive zones. The links from microbes to macrobiota to fish are essential to the aquatic food web. Trout are known to seek out transition zones. When a river is contaminated, refugia can sustain the fish. The table below summarizes functional values identified for microbiota and macrobiota/fish.

Transition Zone Functional Values	Microbiota	Macrobiota/Fish
1. Food source	√	
2. Preferred habitat for some species (upwelling area)		√
3. Refugia for macro (predator avoidance)(biodiversity)		√
4. Microbially active zone	√	
5. Habitat for food base	√	√
6. Cleaning zone (filters), vegetation, aquatic and riparian	√	
7. Energy transfer	√	√
8. Discharge areas may have high biodiversity		√

(1) High quality habitats/refugia

Discharge zones can provide thermal refugia for anadromous fish both for resting and for spawning. Upwelling areas may be important by providing chemical/olfactory signals to anadromous and migratory fish. The zone provides a microbial food supply to the fish and the upwelling areas can act as incubators. Salmon need high quality water including cool water refugia in otherwise warm stream reaches. Conversely, ground-water discharge environments may be the only areas where it is warm enough to survive in very cold areas. Snow dimples have been used for years as surface manifestations of ground-water discharges. Also, small areas in a lake could provide a large percentage of the trout population with support. These can be unique habitats and important energy sources. Certain fish seek out upwelling areas and shellfish may also live in these zones. Macrophytes (e.g., shallow eelgrass beds) may also benefit. Macrophytes may establish preferentially in beds related to discharging ground water. Sometimes ground-water discharges into marine areas are the only areas where emergent vegetation can grow. Another question is whether some macroinvertebrates and fish avoid contaminated ground-water upwelling areas. Trout have good olfactory sense and will avoid metals at concentrations well below toxic levels.

These zones also may limit benthic invertebrate exposure to low oxygen and contaminants by creating oxygenated, clean zones. These zones may also be areas of plant and animal biodiversity. They can be areas of high water quality in alluvial aquifers. Some European countries are interested in identifying high quality ground-water discharge zones (good quality refugia) in the midst of contaminated rivers to preserve as critical habitat.

(2) Contaminant attenuation/removal.

The transition zone is important for chemical and biochemical reactions that influence the quality of the ground water discharged into the surface water. Metals, halogenated organic solvents, polycyclic aromatic hydrocarbons, volatile organic compounds (VOCs), and nutrients can be degraded or removed from ground water within the transition zone.

Volatile organic compounds (VOCs) were discussed in particular. The issue was whether there are concentration thresholds of VOCs above which they poison biological communities. Where there is a large VOC plume, there could also be bioaccumulating contaminants. If the VOCs were then degraded, but the bioaccumulative contaminants (e.g., PCBs, creosote) were not, then bioaccumulation of toxicants still would occur. This has implications for remedial decisions, especially if contaminants are brought in through ground water. Some participants expressed the opinion that VOCs are ignored generally because their toxicity thresholds are much greater than those for heavier contaminants, and therefore they seem to show no risk in the water column. However, risk thresholds based on continuous exposure to a hazard such as VOCs are different than those used in water quality criteria.

(3) Cycling of nutrients and carbon

Nutrients and carbon cycle very actively in this zone. Strong redox gradients enhance biogeochemical activity and microbial processes. Both aerobic and anaerobic processes often occur within close proximity of each other. Microbial biomass can serve as the base of a detrital food chain that can be important to overall ecoystem productivity.

(4) Food base for benthic organisms

Microbes and fungi can provide food for other transition zone organisms that are more intimately involved in the benthic food web of the surface water body. Many macroinvertebrates use the transition zone extensively, and they are food for other organisms. If the zone is contaminated, the result for invertebrates could be mortality, biomagnification and/or bioaccumulation.

WHAT METHODS CAN BE USED TO ASSESS THIS TRANSITION ZONE ECOSYSTEM?

Current methods for studying transition zones generally are not standardized and sometimes not well developed. For example, scales may be mismatched (wells are too big to sample over decimeter or centimeter gradients). Regardless of these difficulties, it is very important that ecologically related sampling in the transition zone be coordinated with hydrogeological and chemical surveys at ground water discharge sites. It was useful in the discussion to distinguish two groups of organisms, microbiota and meiofauna/macrobiota. It was noted that it is difficult (but important) to show contaminant effects on these groups.

(1) Microbiota

a. *Community structure*. There is no standard method to determine microbial community structure. Some methods in use include culturing, metabolic profiling, fatty acid fingerprinting, molecular probes, or nucleic acid characterization. These methods are either limited or time-consuming. Other methods involve 1) collection using ceramic beads or other artificial substrates that collect a sample population in the transition zone; 2) artificial cores with natural materials; and 3) artificial habitats/substrates. Procedures, methods, and equipment are usually designed to answer the specific questions at hand. A method to evaluate drinking water called UDI (Under Direct

Influence) was mentioned. The suggestion was made to focus on the algal community as a surrogate. The algal community and the benthic interface has diagnostic value because there is a rich literature of algae as bioindicators. The comment was made that including diatoms would be time consuming and not too practical. A suggestion was made to develop tools to measure activity first, then measure structure.

b. *Microbial activity/function*. Again, there is no single ideal method. Methods in use or proposed include bioassays (such as the Microtox bioassay), determination of metabolic rates and pathways, describing the dominant terminal electron accepting process (methanogenesis, sulfate reduction, iron reduction, manganese reduction, denitrification, or aerobic respiration), measuring molecular hydrogen and testing for metal tolerance.

It may be difficult to generate interest in microbial function—microbes in septic tanks that provide organic degradation are a familiar example. The transition zone is important for carbon cycling, nutrient cycling, and a detrital-based food chain. Contamination should not interfere with these processes and the decomposer community. So, what would be the appropriate method to evaluate decomposition? Is the desired method to identify the amount of carbon no longer available (tied up in ligands or refractory) or metabolized?

Another suggestion was to evaluate biological oxygen demand (BOD) and/or chemical oxygen demand (COD). For example, the presence of soluble reduced metals will result in high COD and affect interface chemistry. If ground water has high BOD/COD and dissolved oxygen (DO) is present, that observation is important. However, all agreed that BOD and COD are presently impossible to resolve across small scales, although fine-scale characterization of DO is possible.

2. Macrobiota/Meiofauna

a. *Community structure*. Several methods exist for sampling organisms in the transition zone (see D. Williams' abstract on page 39 of this report) and various standard metrics can be computed (community composition, density, species richness). Benthic and ground-water taxa can be distinguished.

b. *Function*. The following were suggested: bioaccumulation studies and stable isotope analyses (e.g. $^{15}N/^{14}N$, $^{13}C/^{12}C$, and $^{34}S/^{32}S$) for food chain relationships. Functional feeding groups can be evaluated.

It was suggested that these basics (community structure and function at all levels) be understood first before developing more methods to conduct toxicity testing.

WHAT RESEARCH IS NEEDED TO BETTER UNDERSTAND THE ECOLOGICAL IMPORTANCE OF AND ASSESS EFFECTS ON THE TRANSITION ZONE?

(1) Basic biological research

Most recommendations centered around basic science needs regarding the transition zone (e.g., life histories, faunal surveys, activity measurements) and sampling/evaluation tools. Life history characteristics of transition zone organisms are generally lacking. Food chain relationships that describe the linkages among microbial, meiofaunal and macrofaunal organisms also are lacking. A suggestion was made to develop methods to conduct a subsurface biomass study. Because no large reference databases exist (compared with surface water data), faunal surveys should be done for major riverine ecosystems using a hydroclimatic landscape approach (see T. Winter's abstract on page 46 of this report). These surveys would be used to develop reference conditions in a national database. If there are differences between geographic areas, it may be best to look at functional differences rather than community differences so data can be compared across broad regions. The Chemistry Group also suggested establishing regional hyporheic study sites.

(2) Macrobiota

Species richness and growth could be evaluated. The physiology of transition zone invertebrates is poorly known (e.g., O_2 uptake rates and mechanisms are often unknown). Respiration studies are needed as well as information on trophic structure. Stable isotopes of nitrogen might be an effective way to determine food chain relationships. Dissolved oxygen availability should be accurately measured. Good biological indicators are as yet uncertain and likely vary for differing flow paths or discharge zones. One should look at biological impacts but use chemical and hydrologic conditions to help define sampling zones.

(3) Indicators of ground-water discharge zones

The Chemistry Group discussed the scenario of a plume entering a stream and how to detect effects in the subsurface. They suggested looking in four dimensions: vertically, horizontally, temporally, and downstream. In general, a point source will be easier to detect than a diffuse plume. You will need several transects across the river. What biological components should be measured? Potential electron acceptors and dissolved hydrogen are good biogeochemically informative constituents to measure. You can characterize the microbial community in many ways. Culturing methods normally select for small subsets of the total microbial community. Molecular techniques also can be used, but presently none of these methods are easily and routinely applied.

Indicator choices depend on the question to be answered. Which attributes are you protecting? Microbial assays need to be used, even if these assays are not yet perfected. Promising techniques are currently under development. Morphological measurements in the system are easier to make than biological measurements. Intensive sampling near the point of discharge plus additional transects would be useful. Sampling should include "vertical distributions" through the food chain.

(4) Biological indicators of GW discharge zones

Are there any biological attributes that help define ground-water discharge zones? For example, can you look for benthic algal blooms? Are fish numbers and distributions in context with other indicators a useful means to locate discharge zones where high quality aerobic ground water is present. Some species may tend to remain in an area even if contaminated. The mechanisms by which fish and other species avoid contaminants is very complicated. Distribution of fish does not necessary follow water quality parameters. Are ostracods good indicators? The consensus was no. It was suggested that midge larvae might be better indicators for ground-water discharge zones. One documented indicator is the presence of high biomass benthic algal mats, but this is limited to zones with enhanced nutrient discharge. Some discharge zones are dead zones, especially where anaerobic, metal-rich ground waters are discharging. There is an important research need to try and correlate between bottom type and patchiness with ground-water discharge. In lake ecosystems, these zones may be linked to aggregations of zooplankton. Acoustic techniques that detect these aggregations may be able to locate ground-water discharge points in lakes

(5) Chemical/physical indicators of ground-water discharge zones

Temperature and conductivity probes are simple, easily-used, and rugged tools for determining ground water discharge locations. These methods could be routinely used to guide sampling in many aquatic ecosystems. Bottom drags with temperature and conductivity probes also can be considered if site conditions warrant. Protocols are needed to allow better comparisons among sediment samples and data from temperature and conductivity probes. Although DO probes are somewhat unstable in the field, investigators could use combined temperature, conductivity, Eh, and DO measuring instrumentation to look for discharge zones. Oregon State University has a suite of fiber optic sensors/ probes that are commercially available and potentially useful in these transition zones. Redox measurements in the field are a problem because of a lack of equilibrium in many samples, and redox potential is often dominated by iron biogeochemistry. Tools needed for improved sampling of ground water discharge zones include:

- Sampling devices to collect organisms effectively and quantitatively along transition zones;
- Dependable and cost-effective geophysical and tracer tools to delineate transition zones and guide biological sampling; and
- Routine survey tools to better characterize microbial community structure and activity and assess water quality and condition.

(6) Scale

Strong gradients in physical and chemical parameters commonly exist in the transition zone. For example, the distribution of redox sensitive solutes can be very steep. Sampling often must be at the centimeter scale or finer resolution. All participants agreed that we need better methods to sample gradients and narrow transition zones. Microcosms or fine-scale bioassays may be approaches to consider.

(7) Hydrology

Knowledge of hydrologic characteristics of the transition zone is crucial. For example, transpiration rates may be very important to the hydrology of these interface zones, but there are large regional differences. Chemical and isotopic tracers may be the best methods to determine the effect of the transition zone on overall stream quality. Some tracers also are sensitive to in-stream processes. Other participants pointed out the need for subsurface measures in addition to surface water sampling. Unresolved questions include:

- What techniques are available for measuring the volume of water entrained into the hyporheic zone?
- What are biological consequences of remediation (pump and treat) that reverse flows in the transition zone?

Injecting oxygenated water could change the redox chemistry within the hyporheic zone. Highly regulated rivers (dammed) affect the hydrology of this interface as well.

(8) Signal-to-noise and partitioning sources

Some practical sampling questions were raised about characterizing the transition zone. How is it different from a place without ground-water recharge or discharge? The responsible party will need to prove that the background contamination “noise” is greater than their contribution. How can you compensate for variability (from the regulator's perspective)? We need screening tools (inexpensive) to identify the problem and focus the sampling. What methods can distinguish ground water from sediment sources? Ground-water discharge may become contaminated as it flows through the hyporheic zone, becoming a "fingerprinting" challenge.

(9) Temporal variability

Temporal variability is important: hourly variability in the hydrology, chemistry, and biology of the transition zone has been noted. When are the best-case and worst-case times for sampling; which season or seasons should be sampled? Different life stages have different susceptibilities and exposures. Ecologists and hydrogeologists need to collaborate. Ecologists can specify time of year and depths of concern; hydrogeologists can determine the hydrologic regime and geochemistry.

(10) Remote sensing

Field studies combined with remote sensing now can be used to better understand the heterogeneity and landscape characteristics of transition zones. Hydrology and food resources for important species are not homogeneously distributed but often highly localized. Remote sensing provides a tool for assessing landscape-scale patterns of hydrology and biotic distributions. Certain patterns on the landscape (e.g., localized plankton blooms) may be surficial indicators of processes occurring in the transition zone. Researchers and managers need to combine extensive and intensive analyses.

(11) Toxicity testing

Are there any non-lethal endpoints or tools that could be used to determine or screen for toxicity on transition zone organisms? Growth studies are generally more sensitive than mortality or fecundity studies. Are there any ground-water toxicity tests or ground-water bioassays? One suggestion was that Elmid beetle larvae in the hyporheic zone may be suitable test species.

(12) Nutrients

The role of transition zones in overall nutrient cycling is still poorly known. Nutrient effects need to be related to species effects, such as effects on sea grasses or corals. The management goal would be to protect "normal" nutrient cycling. Most people live near coasts, and impacts on transition zones that affect riverine delivery of nutrients or ground-water discharge of nutrients in estuarine or coastal waters are critical processes that need to be better understood and monitored. In general, we do not know the trends in nutrient delivery from these transition zones for rivers, estuaries, or coastal waters. In addition, the rates and locations for nutrient transformations by microbial organisms in transition zones in coastal regions deserves further study. There have been relatively few attempts at quantifying these processes.

Similarly, nutrient cycling processes in the hyporheic zone should be better studied. Hyporheic zones receive dissolved oxygen when surface water recharges ground waters. Oxygen participates in important biogeochemical processes such as aerobic respiration, nitrification, metal oxidation, sulfur oxidation, and methane oxidation. For example, if ammonium levels are high in ground waters, nitrification rates can increase and lead to higher concentrations of dissolved nitrate. Where these processes occur and the seasonality of such processes can affect both surface water and ground water quality. Can the portion of nutrient loading in a surface water body that is derived from ground water be distinguished from that derived from surface sources? The U.S. Geological Survey (USGS) has done some related work on this topic in Massachusetts. The contribution from ground water in polluted areas is at least as great as the contribution from rivers in many coastal areas. Tools exist for toxicity testing, but comparable tools do not exist for assessing impacts on nutrient cycling. What methods exist to test whether nitrate is being removed or if that function is impaired? Researchers are working on these methods, but they are not yet regularly employed in monitoring programs.

(13) Dissolved oxygen

The availability of dissolved oxygen plays a major role in the characteristics of ground waters in transition zones. Not all oxygen depauperate discharge zones are caused by pollution; some are naturally low in DO due to hydrologic flow paths (residence time) and rates of microbial metabolism on sediment organic matter. Anaerobic ground waters may contain increased concentrations of dissolved metals, sulfur, and methane. Dissolved oxygen is a master variable in processes and chemical characteristics of transition zones.

(14) Reference comparisons

A disturbed zone needs to be compared to a “normal” reference. How do you identify conditions for comparison? How can you identify effects of the contaminants? How can biological conditions be used as a reference? “Acceptable” conditions need to be defined. Some biotic species (e.g., caddisflies and mayflies) can be used to define reference conditions. Paleontology tools can be used to determine prior conditions. Either reference or gradient comparisons can be used to evaluate changes. The group recommended assessments that allow cross-comparison after remediation (monitoring). The group considered how to define reference conditions in ground water for a superfund site. One approach would be to evaluate current approaches for macroinvertebrates. It would be crucial to locate samples in ground water outside the area of influence. Defining what is meant by reference or reference condition always is challenging. The area should have the same ground-water characteristics in terms of hydrology and chemistry, but without the contamination. This is difficult, because the plume may be a small part of the total ground-water discharge and dispersed contamination may be widespread at a site. It may be easy to find nearby discharge locations that apparently are not contaminated, but it will be critical to carefully assess if these aquifer sediments and ground waters are actually not contaminated.

(15) Correlations between hydrology, sediment, and biology

There have been some correlations described between hydraulic conductivity and ground-water discharge, but not further linked with the biology. Differences in biota occur between upwelling and downwelling areas. Silty or clayey soils (sediments) can inhibit the ground-water flux. Most freshwater macroorganisms do not like turbid water. There may be a juxtaposition of preferred soil type and discharge zones. Adequate characterization of soil structure, porosity and organic matter content are necessary. Clogging, percent organics, amount of DO, and other variables need to be measured. Organisms often preferentially select substrate, so standard artificial substrates sometimes can be used as a surrogate for enhanced comparability between sites.

(16) Bioaccumulation

Diffuse flows and low concentrations of contaminants are hard to measure. Measuring biota that receive contaminants from multiple sources will increase the problem of documenting that a problem exists only from a single ground water source. Bioaccumulation is not always a problem. Lipid bags may not be a very good method for assessing bioaccumulation, because one of the main biological components is accumulation through the food chain. Semi-permeable membrane devices (SPMD) “fat bags” might be a better method. Another possible method would be to look at higher trophic levels such as fish. Nitrogen isotope signatures change over time and are dependent on the trophic level of the animal. Therefore, fish $^{15}N/^{14}N$ ratios and bioaccumulation analyses can be used in combination to deduce an impact from contaminant delivery through ground waters to surface waters.

WHAT BIOLOGICAL MEASUREMENTS DO YOU WISH YOU HAD AT YOUR SITES?

During this discussion, participants identified key measurements that biologists, chemists, and hydrologists would have liked to have had in studies of transition zones:

- Botanical analysis indicative of natural acidic stream condition for studies considering anthropogenic acidification. Sediment probes and piezometers have been used, but no biological data have been collected.
- Sediment and interstitial water toxicity data on Daphnia. Toxicity testing in general would be valuable as we usually get only chemical information. Would the results from those methods be any different than from existing bioassays?
- A test where you can measure impacts on nutrient cycling.
- How many replicates can be processed to account for patchiness? How patchy can it get?
- Toxicity tests for biota in the hyporheic zone following their reaction to exposure or accumulation over time. The tests should be analogous to fish indicators (e.g., hiccuping) or integrative tests such as bee pollen sampling of contamination over a certain radius.

There was general agreement that it would it be useful to develop a suite of toxicity tests for microbes and invertebrates. Microtox is the only commonly used test (luminescence is the endpoint), usually for screening. Certain contaminants lower luminescence and many microbes thrive on contaminants. Microbial toxicity tests therefore may not show anything. There is a lot of natural variability spatially and temporally in electron accepting process. Results depend on the location and timing of sampling. Microtox is usually used for sediment toxicity. One needs to design and interpret the test based on the endpoint of concern.

OTHER QUESTIONS/SUGGESTIONS

- Is organic carbon available to the food-base (labile organic carbon content) a sensitive indicator of microbial activity?
- Can microbiota in the transition zone be thought of as sources of primary productivity like microbial communities in estuarine sediments?
- In Europe, invertebrate organisms are sometimes used as indicators of ground-water quality. Transition zone organisms in the U.S. also could be evaluated for their potential as indicators.
- How can adverse ecological impacts in the transition zone be recognized? Would an indication be when you do not have the anticipated biodiversity?
- What scale should be used to define adverse impacts? The scale depends on the site’s risk management goal.
- Encourage thinking about the need to better integrate biology, hydrology, and biogeochemistry.
- The workshop report should include references to available methods for microbial, epifauna and meiofauna sampling. There are methods available for many species.