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WATERSHED PROCESSES

Convenors: Dr. Russell Brinsfield and Dr. Harry Pionke

Toward a Sustainable Coastal Watershed:
The Chesapeake Experiment. Proceedings of a Conference
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COMPARISON OF NUTRIENT LOADINGS AMONG MAJOR TRIBUTARIES TO THE CHESAPEAKE BAY, MARYLAND AND VIRGINIA

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Abstract: In 1984, the U.S. Geological Survey, in cooperation with the Maryland Department of the Environment, began collecting data to estimate nutrient loads from four major tributaries to the Maryland part of Chesapeake Bay: the Susquehanna, Choptank, Patuxent, and Potomac Rivers. In 1988, a parallel program was begun in Virginia, in cooperation with the Virginia Department of Environmental Quality. The Virginia program included the five major tributaries to the Chesapeake Bay in Virginia: the James, Rappahannock, Appomattox, Pamunkey, and Mattaponi Rivers. Because loads of nutrients are greatest during periods of stormflow, the monitoring program included stormflow and base flow sampling. A regression model was developed that uses time, seasonality, and flow variables to estimate loads.

A preliminary comparison of loads among the nine stations shows that, in an average year, the proportion of total nitrogen load to Chesapeake Bay contributed by the Susquehanna, Potomac, and James Rivers was 66%, 24%, and 6%, respectively. The proportion of total phosphorus load was 38%, 26%, and 29%, respectively. The differences between nitrogen and phosphorus loads may be related to land use and/or regional geology; to whether the streams are point-source dominated or nonpoint-source dominated; or to differences in basin size, morphology, and streamflow. The relative proportion of nitrogen loads is similar to the relative percentage of discharge to Chesapeake Bay from these tributaries. The relative proportions of phosphorus loads do not appear to correlate with discharge.

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NITROGEN LOADS TO CHESAPEAKE BAY FROM GROUNDWATER

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Abstract: Groundwater enters Chesapeake Bay and its tidal tributaries by discharge to the nontidal tributaries and by direct discharge to tidal tributaries and the Bay from nearshore aquifers. Groundwater discharge to nontidal streams (base flow) accounts for about 40% of the total flow of the tributaries upstream from the Fall Line, and for 40% to 80% of the total flow of tributaries in the coastal plain downstream from the Fall Line. Some researchers have reported that direct discharge constitutes approximately 15% to 25% of all water entering the estuaries. Much of this groundwater, especially in the coastal plain, contains elevated concentrations of dissolved nitrogen species (> 1 milligram per liter), and groundwater is therefore a major source of nitrogen input into the Bay.

The quality of groundwater discharge is related to local variations in topography, geology, and land use. Base flow accounted for about 40% of the total nitrogen discharge in streamflow from the Delmarva Peninsula from October 1990 through September 1991.

Local-scale studies on the Delmarva Peninsula indicate that nitrogen-rich groundwater now discharging into streams is about 25 years old. Young groundwater that will discharge during the next 25 years contains higher concentrations of nitrogen than current discharge. This young water will continue to discharge to streams and estuaries even if nitrogen contributed by human activity is controlled. Knowledge of the groundwater nitrogen load to streams and the Bay is, therefore, necessary for evaluating the effectiveness of nitrogen-reduction measures.

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EDGE-OF-FIELD NUTRIENT LOSSES DURING THE 1993 SPRING FRESHET

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Abstract: Above average precipitation throughout the Chesapeake Bay drainage basin early in the spring of 1993 caused near-record levels of freshwater inputs to the Bay. Surface runoff discharge volume from a field-scale coastal plain agricultural watershed during March 1993 was the highest monthly total recorded during ten years of observation. However, nutrient and sediment concentrations associated with this discharge were near annual minimum levels, thereby countering the impact of the above-average surface runoff volume on long-term watershed nutrient losses. Groundwater recharge and discharge volumes also were above average during late winter/early spring 1993. Root-zone leaching losses of nitrate varied widely as a consequence of root-zone soil nitrate availability. The concentrated period of groundwater recharge during March 1993 increased hydraulic gradients in local groundwater flow systems, resulting in high rates of subsurface discharge into surface water bodies. The impact of this discharge on nitrogen transport into the Wye River during March 1993 was controlled by nitrate leaching losses that occurred years earlier in up-gradient agricultural systems. Nitrogen transport associated with groundwater discharge was the largest component of watershed nitrogen transport observed during March 1993, exceeding surface runoff nitrogen losses by nearly a factor of 20, and the highest observed rate of root-zone nitrogen leaching losses by a factor of 4.

INTRODUCTION

Above-average precipitation throughout the Chesapeake Bay drainage basin early in the spring of 1993 caused near-record levels of freshwater inputs to the Bay. In Chesapeake Bay sub-basins where March precipitation fell primarily as rain, or where snow melt was rapid, discharge rates were highest in March. For example, in the Choptank River watershed north of Greensboro, March discharge was more than double the long-term mean, while April discharge was approximately 60% above average (U. S. Geological Survey 1993). In other regions of the Bay watershed where large amounts of snow accumulated, peak discharge was delayed until April. In the Susquehanna River basin, March discharge was only 15 % above the 25 year mean, while April discharge was more than triple the average discharge rate.

These high freshwater discharge rates generated concern regarding nutrient inputs to Chesapeake Bay. The relationship between discharge volume and nutrient loads is apparent in long-term monitoring data from coastal plain watersheds (Maryland Department of the Environment 1990). The impact of annual variability of freshwater discharge on estuarine water quality and living resources also has been documented for coastal plain subestuaries of Chesapeake Bay (Stevenson et al. 1993). However, while intensive estuarine water column and fall line discharge monitoring have been conducted throughout the Chesapeake Bay watershed as part of the restoration effort, relatively little information has been collected regarding edge-of-field nutrient discharge patterns, which ultimately determine those observed at larger watershed scales.

Quantifying edge-of-field nutrient loads requires definition of system boundaries (figure 1). Although field boundaries are relatively apparent from a perspective of surface runoff nutrient transport, defining subsurface flow system boundaries, and quantifying nutrient transport across those boundaries, are more difficult. However, given the critical role of subsurface flow paths in water and nitrogen delivery to tidal waters in coastal plain regions of the Chesapeake Bay watershed (Bachman et al. 1994, Reay and Simmons 1992, Staver et al. 1989), subsurface flows must be included to accurately estimate total edge-of-field discharge.

When nutrients are transported in surface runoff from an agricultural field, they immediately alter water quality in off-site surface waters. In contrast, when nutrients are leached downward through the soil profile to a level where they are unavailable for plant uptake, they have been lost from the agricultural system, but may not affect surface water quality for months or years. Thus, there are two relevant subsurface flow boundaries: the bottom of the crop rooting zone that defines the lower horizontal boundary of the agricultural system, and the discharge interface between the subsurface flow system and surface water. Located between these two boundaries are the vadose zone and unconfined aquifer (figure 1), both of which can store large quantities of water and solutes in coastal plain watersheds (Staver and Brinsfield 1991b). It is this potential for storage of large quantities of water relative to annual recharge/discharge volumes that causes the long residence times that have been documented in coastal plain subsurface flow systems (Dunkle et al. 1993). Thus, although rates of nutrient leaching from the root-zone control nutrient concentrations in the underlying aquifer, and the future potential for nutrient discharge across the aquifer/surface water interface, in the short-term they have little impact on nutrient discharge rates to surface waters. The objective of this paper is to assess the effects of the above-average precipitation in March 1993 on nutrient transport from field-scale coastal plain agricultural watersheds through both surface and subsurface flow paths.

Figure 1. Schematic overview of advective hydrologic transport system.

METHODS

This study includes data from precipitation, surface runoff, root-zone leaching, and groundwater discharge monitoring activities conducted during March 1993. All study sites are located in the Wye River drainage basin in Queen Anne’s County, Maryland (38½ 55' N, 76½ 09' W). The soil surface at the sites ranges from 2 to 6 m above sea level. An unconfined aquifer underlies the sites at a seasonably variable depth of less than 1 to 4.5 m below the soil surface. Groundwater discharges from all sites directly into tidal waters of the Wye River.

Surface runoff was monitored in a naturally defined 6 ha watershed that lies completely within a 14 ha agricultural field. The field has been planted continuously in corn for grain production since 1983, with an annual application of inorganic nitrogen fertilizer of approximately 158 kg/ha. Following corn harvest in 1992, a rye cover crop was planted in the watershed. Soils at this site belong to the Elkton and Mattapex Series, and are silty moderately well-drained, and nearly level (0-3% slopes). Edge-of-field surface runoff volume was measured using a calibrated flume instrumented with a flow meter connected to an automated sampler (Staver et al. 1988b). Discrete samples were collected at constant volume intervals. Generally, the elevation of the flume is above the water table. Consequently, surface runoff occurs only in close association with precipitation. Samples were transported from the field immediately after each event and frozen in polyethylene bottles until analysis.

Root-zone leachate was monitored in corn production systems with and without a rye winter cover crop using gravity lysimeters located 60 cm below the soil surface (Staver et al. 1988a). Soils at the leachate monitoring site, which was located approximately 500 m from the experimental watershed, belong to the Matapeake and Mattapex Series, and are more well-drained than the soils within the surface runoff monitoring site.

Samples were collected immediately after lysimeter flow ceased, which generally occurred within 24 hr of the end of the precipitation event. Lysimeter samples were filtered through a polycarbonate filter (nominal pore size 0.45 micron) and frozen until analysis.

Groundwater discharge was measured from an agricultural field located directly adjacent to the Wye River. Intensive hydraulic monitoring and groundwater sampling were used to quantify nitrate discharge from the unconfined aquifer into the Wye River (Staver and Brinsfield 1994). Soils at this site belong to the Sassafras Series, and are more well-drained than the soils at the other two sites. The site was planted with full-season soybeans in 1992, and remained fallow following harvest. Agricultural production at this site alternated between corn and soybeans prior to 1992. Groundwater samples were processed similarly to lysimeter samples.

Surface runoff samples were analyzed for total, and total dissolved (< 0.45 micron) nitrogen and phosphorus using a persulfate digestion (Valderamma 1981) followed by colorimetric analysis of phosphate content (Parsons et al. 1984) and nitrate analysis using high pressure liquid chromatography. Filtered groundwater and lysimeter samples also were analyzed for nitrate using high-pressure liquid chromatography.

DISCUSSION

Surface Runoff

Precipitation during March 1993 at the study site totaled 17.77 cm, nearly double the long-term average depth for March. High antecedent soil moisture levels at the beginning of the month, combined with the intensity of precipitation during the “blizzard of 1993” resulted in a high percentage of precipitation leaving the gauged watershed as surface runoff (figure 2). From 1985 through 1993, runoff volume from this watershed was approximately 10% of total precipitation volume. Thus, the March 1993 runoff/precipitation ratio of 0.66 was well above the long-term average value. The March runoff depth of 11.7 cm was approximately equal to the average annual runoff depth from that watershed for water years 1985 - 93, and exceeded total annual runoff depth in six of the eight previous water years.

Edge-of-field surface runoff nutrient losses are determined by runoff volume and nutrient concentrations. Both nitrogen (figure 3) and phosphorus (figure 4) concentrations in surface runoff from the experimental watershed have tended to be near annual minimum levels during late winter. This, combined with the potential for dilution during large runoff events, resulted in relatively low volume-averaged total nitrogen and phosphorus concentrations in surface runoff in March 1993. Nitrogen concentrations, which generally are highest during summer months following the application of nitrogen fertilizer, were particularly low during March 1993 (0.84 mg/l) relative to the nine-year average (3.57 mg/l). Consequently, surface runoff nitrogen and phosphorus losses from the experimental watershed during March

Figure 2. Long-term average and 1993 water year (WY) monthly precipitation depths, and 1993 water year monthly surface runoff depths from an agricultural watershed in the Wye River drainage basin.

Figure 3. Monthly volume-averaged nitrate-N, dissolved nitrogen, and total nitrogen concentrations in surface runoff from an agricultural watershed in continuous corn production.

Figure 4. Monthly volume-averaged phosphate-P, dissolved phosphorus, and total phosphorus concentrations in surface runoff from an agricultural watershed in continuous corn production.

Figure 5. Average water table elevation (relative to mean sea level) in an agricultural watershed in the Wye River drainage basin during the 1992 and 1993 water years.

1993 were much less extreme than runoff volumes. Total phosphorus transport in surface runoff during March was 0.39 kg/ha, which represented approximately 40% of the average annual total phosphorus loss from this watershed from 1985 through 1993. Because the average total nitrogen concentration in surface runoff during March 1993 was even lower than average annual concentrations (relative to phosphorus), the high surface runoff volumes were even less significant from a nitrogen perspective. Total nitrogen transport in surface runoff during March was 0.99 kg/ha, or approximately 24% of the average annual total nitrogen loss in surface runoff from this watershed from 1985 to 1993.

Leaching

The above-average precipitation in March 1993 also caused large volumes of water to cross the lower root-zone boundary. This produced a concentrated period of groundwater recharge, and an increase in water table elevations to near maximum levels within the experimental watershed (figure 5). At all the sites discussed in this study, the water table is located within several meters (or less) of the root zone. Because soil moisture levels in the intermediate vadose zone (between the root zone and the water table) are relatively constant in the long term, leachate volumes can be equated to recharge volumes. Below average temperatures, prolonged periods of precipitation, and snow or ice cover for much of March 1993 minimized evaporation losses. Thus, leachate (recharge) volume for the month (6 cm) was approximately equal to the difference between precipitation and surface runoff volume. Calculated recharge volumes for the experimental watershed during the previous five years, water averaged approximately 39 cm. Thus, the volume of groundwater recharge observed at this site in March 1993 was not nearly as extreme relative to average annual volumes as were surface runoff volumes.

Nitrate concentrations in leachate collected at a depth of 60 cm were two orders of magnitude higher in corn/winter-fallow versus corn/rye cover treatments throughout March 1993 (figure 6). However, even in the corn/winter-fallow treatment, leachate nitrate concentrations during March 1993 were well below those reported for shallow groundwater under continuous corn production systems at this site (Brinsfield and Staver 1991), and in other coastal plain agricultural systems. This probably was attributable to the favorable weather conditions during the 1992 growing season that resulted in high levels of corn nitrogen utilization, and low levels of root-zone nitrate in the fall of 1992.

Nitrate transport out of the root zone (below a depth of 60 cm) during March 1993 was approximated by multiplying the volume of groundwater recharge (6 cm) determined for the gauged watershed with lysimeter nitrate concentrations. This approach yielded root-zone nitrate-N leaching losses during March 1993 of 4.8 kg/ha in corn/winter-fallow treatments and 0.02 kg/ha in corn/rye cover treatments. Because the soils within the leachate monitoring site were more conducive to infiltration than were soils in the gaged watershed, it is likely that this approach underestimated leachate volume and root-zone nitro-gen leaching losses within the leachate monitoring site.

Figure 6. Average nitrate-N concentrations in root zone leachate collected using gravity lysimeters from October 1992 through March 1993.

Groundwater Discharge

The large volume of groundwater recharge during March 1993 that raised water table elevations to near-maximum levels (figure 5) created the highest lateral hydraulic gradients recorded within the groundwater discharge zone between April 1992 and April 1994. As a result, maximum ten-day average groundwater discharge rates into the Wye River during the two year study were recorded during March 1993 (figure 7). March 1993 discharge totaled 16.6 m^3/m of shoreline, or approximately 20% of the total discharge for the 1993 water year. The length of the groundwater flow system at this site is approximately 140 m (Staver and Brinsfield 1994), yielding an area-normalized discharge volume of approximately 12 cm during March 1993. The surface soils at the groundwater discharge monitoring site are coarser textured relative to the soils at the surface runoff and leaching study sites. As a result, almost all precipitation at this site infiltrates, suggesting that groundwater recharge during March 1993 was nearly equivalent to the total monthly precipitation depth of 17.7 cm. Thus, it is not surprising that groundwater discharge volume at this site during March 1993 was approximately double the recharge volume (6 cm) at the surface runoff study site.

Figure 7. Daily and ten day average rates of groundwater discharge from an agricultural watershed into the Wye River from April 1992 through April 1994.

Nitrate-N concentrations in discharge at the groundwater discharge monitoring site averaged approximately 15 mg/l during March 1993, and remained relatively constant. Thus, the pattern of nitrate discharge was very similar to that for groundwater (figure 8). On an areal basis, the groundwater nitrate discharge rates into the intertidal zone of the Wye River translated into nitrogen losses from the agricultural system during March 1993 of approximately 19 kg/ha.

Figure 8. Daily and ten day average rates of groundwater nitrate-N discharge from an agricultural watershed into the Wye River from April 1992 through April 1994.

Flow System Comparisons

The volumes of edge-of-field discharge into the Wye River determined for the study sites during March 1993 (groundwater 12.2 cm, surface runoff 11.7 cm) were similar to larger-scale regional discharge volumes. March 1993 discharge for the 29268 ha region of the Choptank River watershed above Greensboro, which is located approximately 30 km east of the Wye study sites, was 13.2 cm (U.S. Geological Survey 1993). Because only negligible amounts of surface runoff occurred at the groundwater monitoring site, groundwater discharge closely approximated total discharge at this site. Because groundwater discharge was not considered at the surface runoff monitoring site, total discharge would have been somewhat greater than the surface runoff volume of 11.7 cm.

The variation in surface soil texture between the groundwater discharge and surface runoff monitoring sites dramatically altered partitioning of precipitation between surface and subsurface flow paths. Because phosphorus does not readily leach out of upper soil horizons in these soil types, phosphorus concentrations in shallow groundwater tend to be more than an order of magnitude lower than the phosphorus concentrations observed in surface runoff during March 1993 (Staver et al. 1988b). Thus, even though groundwater discharge (12.2 cm) and surface runoff (11.7 cm) volumes were similar at the two sites, the absence of surface runoff at the groundwater discharge monitoring site minimized phosphorus transport rates.

Because nitrogen can readily leach out of the root zone in the nitrate form, discharge flow path does not dictate the magnitude of nitrogen losses from a site. Losses are determined by relative concentrations of total nitrogen, which can vary widely seasonally, as well as owing to different cropping patterns. During March 1993, surface runoff losses from the corn/rye cover crop treatment (0.99 kg/ha) were minor relative to long-term average annual surface runoff losses (5.4 kg/ha), but were more than 20 times greater than March 1993 root-zone leaching losses (0.02 kg/ha) for the same cropping pattern. In the corn/winter-fallow treatments, March 1993 root-zone leaching losses were more than 200 times higher than where a cover crop was present, and about an order of magnitude higher than March 1993 surface runoff nitrogen losses where a cover crop was planted following corn harvest in the fall of 1992. Thus, the presence or absence of a cover crop determined whether nitrogen leaching losses exceeded surface runoff nitrogen transport from the agricultural system during March 1993, and also had a major effect on total nitrogen losses.

Nitrogen transport into the Wye River during March 1993 via groundwater discharge exceeded all other parameters of nitrogen movement calculated in this study. This occurred primarily as a result of the high concentration of nitrate-N (15 mg/l) in the unconfined aquifer at the study site, relative to nitrogen concentrations in surface runoff or root-zone leachate. The groundwater nitrate levels at this site were typical of those that have been reported for shallow groundwater underlying agricultural systems in the coastal plain region of the Chesapeake Bay watershed (Staver et al. 1988a, Magette et al. 1989, Weil et al. 1990), and indicate that previous up-gradient root zone leachate nitrate-N concentrations averaged approximately 15 mg/l. What is somewhat unusual about this study site is that groundwater discharges directly into intertidal sediments, with little potential for in-stream or riparian zone reductions in groundwater nitrate concentrations that have been documented in other coastal plain agricultural systems (Lowrance 1992, Jordan et al. 1993).

CONCLUSION

The results of this study suggest that phosphorus losses from Maryland coastal plain agricultural watersheds having surface cover adequate to minimize soil erosion were minor relative to total edge-of-field discharge volumes during March 1993. Large site-to-site differences were observed in edge-of-field phosphorus losses as a consequence of differences in soil infiltration capacity. Surface runoff nitrogen losses also were minor during March 1993 relative to discharge volume. Surface runoff nitrogen concentrations in continuous corn production systems tend to be near annual minimum levels during late winter, thereby minimizing the potential for surface runoff nitrogen transport, even in periods of above average discharge such as March 1993.

The high volume of leachate during March 1993 created the potential for nitrate transport from coastal plain agricultural systems. However, leachate nitrate concentrations were relatively low even in corn/winter-fallow treatments, probably owing to unusually low root zone nitrate levels following the 1992 growing season. Root-zone nitrogen leaching losses were highly dependent on root-zone nitrate availability, making it difficult to speculate regarding the applicability of results from this study to other coastal plain agricultural systems. However, this study does demonstrate how management of root zone nitrate availability during groundwater recharge periods can greatly reduce nitrate leaching losses.

This study suggests that the primary impact of the above-average precipitation during March 1993 on nutrient discharge rates from coastal plain agricultural watersheds was increased discharge of nitrate stored in subsurface flow systems. The addition of large volumes of recharge to subsurface flow systems during March 1993 forced large volumes of groundwater across the aquifer/surface water interface. At sites where previous leaching patterns resulted in high concentrations of nitrate in the unconfined aquifer up-gradient of the discharge interface, discharging groundwater transported large quantities of nitrate into surface waters. However, although the high groundwater discharge volume observed during March 1993 was a short-term consequence of above-average precipitation, the associated nitrogen discharge resulted from root zone nitrogen losses that occurred years earlier.

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REFERENCES

Bachman, L.J., P.J. Phillips, and J.M. Denver. 1994. Quality of baseflow in non-tidal tributaries to Chesapeake Bay, Delmarva Peninsula, Delaware and Maryland. Transactions American Geophysical Union, Abstract HB32-3.

Brinsfield, R.B., and K.W. Staver. 1991. The use of cereal grain cover crops for reducing ground water nitrate contamination in the Chesapeake Bay region. In: Hargrove W. L., ed. Cover crops for clean water. Ankeny, IA. Soil and Water Conservation Soc., 79-82.

Dunkle, S.A., L.N. Plummer, E. Busenburg, P.J. Phillips, J.M. Denver, P. A. Hamilton, R.L. Michel, and T.B. Coplen. 1993. Chlorofluorocarbons as dating tools and hydrologic tracers in shallow groundwater of the Delmarva Peninsula, Atlantic Coastal Plain, United States. Water Resour. Res., 29: 3837-3860.

Jordan, T.E., D.L. Correll, and D.E. Weller, 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. J. Environ. Qual. 22: 467-473.

Lowrance. R. 1992. Groundwater nitrate and denitrification in a Coastal Plain riparian forests. J. Environ. Qual. 21: 401-405.

Magette, W.L., R.A. Weismiller, J.S. Angle, and R.B. Brinsfield. 1989. A nitrate standard for the 1990 Farm Bill. J. Soil and Water Conserv. 44: 491-494.

Maryland Department of the Environment. 1990. Level I data summary report covering the period January 1, 1978 through December 31, 1989. Baltimore, 82 p.

Parsons, T.R., Y. Maita, and C.M. Lalli. 1984. A manual of chemical and biological methods for seawater analysis. Pergamon Press.

Reay, W. G., and G.M. Simmons. 1992. Groundwater discharge in coastal systems: Implications for Chesapeake Bay. In: Perspectives on Chesapeake Bay, 1992: Advances in estuarine sciences. Chesapeake Research Consortium. Solomons, MD: CRC Publ. No. 143, 17-44.

Staver, K.W., R. Brinsfield, and W. Magette. 1988a. Nitrogen export from Atlantic coastal plain soils. St. Joseph, MI: Amer. Soc. Agri. Eng., ASAE Paper No. 88-2040.

Staver, K. W., R. Brinsfield, and W. Magette.1988ab. Tillage effects on phosphorus transport from Atlantic coastal plain watersheds. American Society of Agricultural Engineers, St. Joseph, MI: ASAE Paper No. 88-2651.

Staver, K.W., R. Brinsfield, and J. Stevenson. 1989. The effect of best management practices on nitrogen transport into Chesapeake Bay. In: J. B., Summers and S.S. Anderson eds. Toxic substances in agricultural water supply and drainage. U.S. Committee on Irrigation and Drainage, Denver, CO: 163-180.

Staver, K.W., and R.B. Brinsfield. 1991b. Monitoring agrochemical transport into shallow unconfined aquifers. In: Agrochemical residue sampling design and techniques: Soil and groundwater. American Chemical Society Symposium Series No. 465, 264-278.

Staver, K.W., and R.B. Brinsfield. 1991a. Effect of cereal grain winter cover crops on surface water pollutant transport from Coastal Plain corn production systems. In: Hargrove W. L., ed. Cover crops for clean water. Soil and Water Conservation Soc., Ankeny, IA: 50-52.

Staver, K.W., and R.B. Brinsfield. 1994. Groundwater estuarine interactions in a coastal plain riparian agroecosystem. In: Riparian ecosystems in the humid U.S. Washington, DC: National Assoc. of Soil Conserv. Districts, 256-276.

Stevenson, J., L. Staver, and K. Staver. 1993. The survival of submersed aquatic vegetation along an estuarine gradient. Estuaries 16:347-359.

U.S. Geological Survey. 1993. Water resources data, Maryland and Delaware.

Valderamma, J.C. 1981. The simultaneous analysis of total nitrogen and total phosphorus in natural waters. Mar. Chem. 10: 109-122.

Weil, R.R., R.A. Weismiller, and R.S. Turner. 1990. Nitrate contamination of groundwater under irrigated coastal plain soils. J. Environ. Qual. 19: 441-448.

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EFFECTS OF SEDIMENT COMPOSITION AND LAND COVER ON NITRATE CONCENTRATIONS AND GROUNDWATER DISCHARGE TO ESTUARIES ON THE EASTERN SHORE, VIRGINIA

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Abstract: Studies on the Eastern Shore of Virginia show the relation of regional and local near-surface sediment composition to nitrate concentrations in groundwater that discharges to estuaries from cultivated fields. In fine-grained, organic-rich sediments, nitrate concentrations in groundwater beneath a cultivated field are low (near the detection limit) because low dissolved oxygen concentrations inhibit nitrification of ammonium to nitrate and promote denitrification of nitrate. In coarse sediments that contain little organic material, nitrate concentrations in groundwater are elevated and can exceed 10 milligrams per liter. Nitrate can denitrify near discharge areas where local or regional deposits have abundant organic material. This organic material is in the originally deposited sediments; little organic material is contributed by the overlying riparian vegetation.

Local sediment composition and land cover also can affect how and where groundwater discharges to estuaries. Fine-grained sediments as thick as 12 feet have been locally deposited near some estuaries over the relatively coarse sediments of regional formations. Consequently, ground-water discharge is probably focused in small areas where these fine sediments are eroded along the shore and discharge is diffused over large areas elsewhere. In areas where broad, low wetlands of riparian vegetation border upland fields, most of the groundwater can discharge by evapotranspiration. Nitrate is not transported to the estuary were groundwater is discharged by evapotranspiration.

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SITE ASSESSMENT FOR NPS POLLUTION ABATEMENT IN RIPARIAN ZONE SITES

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Abstract: In the humid East, groundwater passes through riparian ecosystems before discharging into streams. Characteristics of riparian ecosystems control the cycling and sequestering of nonpoint-source (NPS) pollutants within the near-stream environment. The degree to which these processes attenuate the concentration of groundwater contaminants depends on the extent of the riparian zone and flow patterns through it. If much agricultural drainage leaks to regional groundwater or enters the stream through the channel bottom, the possibility for groundwater renovation within the riparian zone decreases. Geophysical and hydrologic investigations help to define flow paths from upland sites to the stream. Seismic surveys reveal the depth to, and configuration of, bedrock interfaces, plus major discontinuities that might affect flow patterns. Slug tests give aquifer permeability and piezometric head distributions yield flow directions. Together, this information can be used to calculate the degree of interaction between discharging groundwater and riparian ecosystems. Seismic surveys, slug tests and piezometric head are used to assess flow patterns within riparian zone sites in the Valley and Ridge physiographic province of Pennsylvania. Flow paths are modeled for the range of measured properties and inferences drawn concerning the potential for groundwater renovation within the riparian zone.

INTRODUCTION

Preservation and restoration of riparian ecosystems have become an integral component in efforts to stabilize and improve the health of Chesapeake Bay and its tributary watersheds. Improved water quality that supports more vigorous and diverse commercial and recreational fisheries, and increased acreages of wild and scenic areas for habitat and aesthetics, are among the elements that describe a healthier Bay watershed. The Bay states have agreed to pursue water quality goals and have established numerical targets and deadlines. The goals are to be met and maintained during a time of increasing population with the concomitant agriculture, industry, and development required to sustain it.

Agriculture is a major source of nonpoint-source (NPS) contaminants to the Bay and its tributaries. Riparian ecosystems are the interface between upland agriculture and streams draining the Bay watershed. Processes occurring within riparian ecosystems can protect streams against agricultural contaminants. This suggests that maintaining and restoring riparian ecosystems is one way to reduce contaminants of agricultural origin entering the Bay. Expected masses of contaminants held from the stream by riparian processes are needed to include riparian zone management in farm plans. Additionally, riparian zone features promoting contaminant reduction and their distribution across the landscape must be known to assess their impact in the Bay watershed.

Riparian processes protecting streams from agricultural contaminants have been studied for less than 20 years. Only a few studies quantify contaminant reductions in terms useful for planning. Those studies were conducted in settings typical of only a small part of the Bay watershed (Correll and Weller 1989, and Lowrance et al. 1983, 1984). If landowners are to receive useful guidance in the appropriate management of riparian ecosystems, contaminant reduction within riparian ecosystems must be experimentally determined for a variety of common settings.

The Bay states have agreed to reduce the mass of contaminants reaching the Bay, and landowners will likely also be required to control the mass of contaminants leaving the farm. Consequently, research that measures the difference in mass entering and leaving the riparian zone provides directly useful information to land managers and planners. Mechanisms of contaminant reduction, their rates, and response to changing inputs are also important and can be discerned by augmenting mass balance information with concentration changes throughout the riparian zone.

The discussion to this point has been general without reference to particular contaminants or modes of transport. The remainder of this paper focuses on nitrogen, a major NPS contaminant of primarily agricultural origin. Most nitrogen transported from farm fields to streams is dissolved in groundwater. Nitrogen is removed from groundwater flowing through riparian ecosystems by chemical and microbial processes. However, the relative position of the stream, riparian ecosystem, and farm field (figure 1) within the hydrologic regime of the entire landscape controls the opportunity for nitrogen removal from agricultural drainage. Nearly all riparian ecosystems have the capacity to store or transform nitrogen. Riparian processes do not protect local streams, regardless of the biochemical potential to do so, when there is no interaction between riparian ecosystems and agricultural drainage (line A in figure 1).

Figure 1. A generalized subsurface flow system depicting local and regional flow paths. Flow paths labeled "A" bypass local streams and are unaffected by local, near-stream processes.

Site Selection Criteria

Physical characterization of the subsurface is the first step in determining the hydrologic regime that controls groundwater flow and associated chemical processes at a site. The characterization should define the flow paths of agricultural drainage from field to stream and determine how much of the water flowing in the stream interacts with the riparian zone. If the flow regime were simple and the subsurface material homogeneous, a few piezometers could be used to gather all the data necessary to calculate the mass flux of nitrogen into and out of a riparian ecosystem. Near-stream flow regimes are not always simple and the subsurface materials are never homogeneous. Hydrologic complexities include buried stream channels, with high lateral permeabilities, that divert flow parallel to the stream rather than through the riparian zone to the stream (e.g., Haycock and Burt 1993). High permeability materials consisting of organic matter or coarse deposits may cross the riparian zone and concentrate flow in relatively small areas (e.g., Cooper 1990). Each of these conditions requires different numbers and configurations of instruments to accurately measure the reduction in nitrogen delivery to a stream attributable to riparian processes. Instrumenting a site without regard for these and other complexities may lead to faulty interpretation of the collected data (Altman and Parizek in press).

A survey of geologic maps and literature of the area will situate the site within regional geology and may suggest complexities common to the area. This information, however, is not usually available at the scale of an experimental site. Information for specific sites under consideration needs to be determined in the field. Fortunately, a number of geophysical surveying techniques are quite useful for characterizing the subsurface.

Seismic refraction surveying is well suited to environmental and hydrogeologic studies, especially in the Piedmont and Valley and the Ridge portions of the Bay watershed with bedrock aquifer controls. There are many advantages to the use of refraction surveying in these projects that almost always concentrate on shallow subsurface processes. Detailed information about the subsurface soil and rock properties can be acquired through the use of seismic wave sources. A large body of data can be collected in a short time using relatively simple methods. The equipment required is portable, so it can be utilized in many locations inaccessible to truck-mounted equipment. Additionally, because the equipment is portable and requires no excavation, little or no damage to the site occurs. The data can be analyzed relatively simply and quickly, resulting in outputs such as soil thickness and bedrock structure beneath the site or even the depth of the water table. Urban and Pasquarell (1992) studied a shallow, fractured shale using seismic refraction, borehole core analysis and double-packer injection tests. They found a strong correlation between fracture frequency, seismic velocity, and hydraulic conductivity. Seismic refraction surveys then also provide a first-order estimate of the spatial distribution of hydraulic conductivity.

A Brief Description of the Seismic Refraction Technique

If a seismic source is generated at the surface, the resulting wave front will propagate in all directions in the subsurface. The shallowest layer is usually soil. When the P-wave traveling in the soil reaches a layer with a higher P-wave velocity it is refracted according to Snell's law. Waves traveling at a critical angle, determined by the P-wave velocities in each layer, are refracted along the boundary between layers at the P-wave velocity of the faster layer. Energy from this refracted wave is continually radiated back to the surface at the critical angle (figure 2). If an array of geophones is placed on the surface to record these data, the first arrivals near the seismic source will be direct P-wave arrivals traveling in the upper layer. At and beyond some critical distance, the refracted P-waves will reach the geophones first, even though the path they travel is longer than the direct path. This is because the refracted wave travels faster along the subsurface boundary than in the upper layer and overtakes the slower- traveling direct P-wave. The plot of first arrival times against distance from the seismic source can then be used to create a depth — a seismic velocity profile. Seismic velocities are directly related to the density of the material through which they travel. Soils or alluvium and bedrock have very different seismic velocities. Stress relief fractures generated from erosion of overburden and the subsequent releases of residual stress in the rock are common in much of the Bay watershed. Stress relief fracturing increases the secondary porosity and permeability of shallow bedrock units, decreases the density of fractured layers, and consequently decreases the seismic velocity.

Three assumptions are made when using seismic refractive methods:

1. A general increase in P-wave velocity with depth is assumed. Seismic refractive techniques cannot detect lower-velocity layers underlying higher-velocity layers. When this assumption is violated, the deeper low-velocity layer is interpreted to be part of the shallower high-velocity layer. Fortunately, this assumption is generally valid for shallow subsurface work.
2. All contacts between layers are assumed to be planar
3. The velocity in each layer is assumed to be constant everywhere.

Assumptions 2 and 3 are also generally valid. However, variations in alluvium composition, bedrock lithology, or degree of fracturing can occur over the length of a seismic transect, resulting in significant lateral variations in seismic velocity. The standard seismic technique can be modified to detect and correct for deviations from the assumptions of planar layers of constant velocity.

A completed seismic refraction survey gives depth to bedrock, orientation, and thickness of subsurface layers and a qualitative distribution of hydraulic conductivity in the riparian zone. These

Figure 2. Schematic of refractive seismic data collection. Time for seismic waves to reach the sensors is recorded. Qualitative interpretations are given in the figure.

site characteristics are needed to position hydrologic instruments such as piezometers or monitoring wells so that calculated fluxes will accurately reflect fluxes in the field. The boundaries and dimensions of the subsurface materials at a site can also be obtained from a grid of numerous borehole cores. Borehole drilling is slower, more expensive, and disturbs the site more than seismic refraction techniques. If the seismic survey of the site reveals subsurface anomalies or complexities that will create large uncertainties in data interpretation, the site can be abandoned without a large investment in time and instrumentation.

A Brief Description of Slug Tests

If the site is acceptable, hydraulic conductivity of the layers identified with the seismic survey must be determined. Piezometers are installed in each identified layer (figure 3) to determine hydraulic conductivities and to measure energy gradients through the riparian zone. These data, along with nitrogen concentrations are required to calculate mass fluxes of nitrogen into and out of a riparian zone. Slug tests are commonly performed to reliably measure hydraulic conductivity. Slug tests can be done in a variety of ways to match project goals (Domenico 1990). Typically, they are performed by adding a known quantity of water to the piezometer and measuring the drop in water level over time (figure 4). Hydraulic conductivity is then determined by history matching the output of an appropriate equation to the collected data. Piezometers are also commonly used to sample groundwater for chemical analysis. Consequently, adding water to the piezometer may introduce error to chemical analyses of samples collected soon after slug testing. An alternative to adding water and recording the rate of drop of water in the piezometer is to remove a slug of water from the piezometer and record the rate of rise. The rate at which water rises in the piezometer is related to permeability by the same equations used in the injection method.

All drilling procedures disturb the aquifer materials around the piezometer point. Reliable slug test results depend on minimizing the disturbance to ensure good undisturbed contact between water in the piezometer and the aquifer. There is a series of solutions to the general slug test equation for confined and unconfined aquifers, for predominantly horizontal or vertical flow and other properties. Slug test results can be significantly influenced by the choice of equations used to history match the test data (Hinsby et al. 1992).

Figure 3. Schematic of a piezometer nest. The relative level of water in the piezometers indicates the force moving water within an aquifer. Hydraulic conductivity can be determined by performing slug tests on the piezometers.

Figure 4. Schematic of a slug test performed in a piezometer.

Subsurface boundaries and dimensions, from the seismic survey, and the distribution of hydraulic conductivities, from slug tests, can then be used to simulate flow directions and velocities (e.g., MODFLOW ([McDonald and Harbaugh 1988, FLOTRANS ([Guiguer et al. 1993]). Results of the simulation may suggest refinements in the number or position of sampling devices to ensure reliable estimates of mass fluxes into and out of the riparian zone. The mass of nitrogen kept from the stream by processes occurring in the riparian zone is then calculated from difference in mass flux across riparian zone boundaries.

Attributing the mass of nitrogen entering a riparian zone but not leaving it to biochemical processes within the riparian zone, and therefore sustainable over the long term, assumes quasi steady state transport conditions. A slowly advancing body of contaminated water could give a similar distribution of nitrogen within the riparian zone and similar mass fluxes into and out of the riparian zone. The data collected under such conditions would reveal nothing about nitrogen removal from discharging water by riparian processes. These conditions are most likely to exist where recharge rates are low and subsurface materials have a large water storage capacity. The steady state assumption can be tested by combining ages of the water entering and leaving the riparian zone with information about the history of agricultural practices.

Case Study - East Mahantango Creek

Site Description

A site characterization similar to that described above was conducted at a site located along the East Mahantango Creek, approximately 40 km north of Harrisburg in central Pennsylvania (figure 5). The site was characterized with a seismic refraction survey, slug tests, contours of piezometric head, and simulated flow paths. Climate in the Mahantango valley is humid and temperate. Annual precipitation and stream flow average approximately 1,000 mm and 580 mm, respectively. Sixty to eighty percent of stream flow is predominantly groundwater runoff (Gburek et al. 1986).

East Mahantango Creek is a second-order stream at the study site. The soil at the site is mapped as a medium-textured, well to somewhat poorly drained alluvial soil, Basher silt loam (Soil Conservation Service, 1982). The soil is approximately 20-40 in deep. The parent material is a transported, yellowish-brown, medium to fine grained sand, typical of alluvial deposits in the area.

The Mahantango valley is underlain by two geologic formations, Trimmers Rock (Late Devonian) and Catskill (Late Devonian - Early Mississippian). The Trimmers Rock formation is primarily shale in the study area. The overlying Catskill formation consists of interbedded sandstones, siltstones, and shales, and becomes increasingly coarse-grained to the north. Previous analysis of well yields (Cline 1968) indicated that rock fracture patterns are as important to formation permeability as rock type, and based on specific capacity data (Urban 1977) the two formations are hydrologically similar.

Figure 5. Location of the study site on East Mahantango Creek. The dashed line represents East Mahantango Creek. Contour lines are at 20 ft intervals.

Seismic Survey

A seismic refraction survey was conducted at the site to estimate the depth of alluvium and fractured bedrock throughout the site in order to produce a map of bedrock elevation. Survey results were used to position piezometers and to test a concern that a buried channel may cut across the meander.

Refraction data for this study were collected using a BISON GEOPRO 8024 24 channel digital seismograph with signal enhancement capability. All 24 geophones were spaced 5, 10, or 15 ft apart with constant spacing. The seismic signal was generated by striking an 8 lb sledgehammer on a 6 in square metal alloy plate lying on the ground at the desired source location. A mercury switch attached to the sledge hammer handle initiated the recording of data by the seismograph. Typically, 10 to 20 hammer blows were needed in this study to achieve good signal-to-noise ratios. For transects longer than the geophone array, a second array was begun by placing the second array's first geophone at the location of the first array's reverse profile shot point. The forward profile shot point for the second array is usually placed at the location of the last geophone of the first array. This procedure was repeated as required to cover the desired profile. The fastest sampling rate possible, 0.2 millisecond, was used to give the highest resolution of first arrival times.

Data analysis was carried out using GeoLogic's PC-based REFRACT program version 2.29. The program has subroutines to (1) select first-arrival times, (2) compute time-distance relationships, and (3) generate layer profiles showing layer boundaries and seismic velocities First arrivals are chosen for each geophone. REFRACT then produces a time-distance plot from the first-arrival times, geophone spacing, source location, and elevations of the geophones and sources. If a reasonable time-distance plot is produced, REFRACT generates a layered depth-velocity profile. The profile includes the thickness of each layer, the dip of the contact between layers and true seismic velocities for each layer.

Uncertainties in first-arrival picks, lateral variations in velocity, and local near-receiver effects can introduce uncertainties in the interpretation of the depth-velocity profile. Considering fits to the data and consistency along and among profiles, differences in shallow bedrock velocity of a few hundred ft/sec can be resolved, and differences in alluvium thickness of about a foot can be detected.

Slug Tests

Piezometers constructed of 1-in diameter PVC pipe, with a 10 in section of well screening, capped at the bottom, were installed at depths of 5, 7.5, 10, and 16-18 ft at locations indicated in figure 6. Slug tests were performed by withdrawing a quantity of water sufficient to drop the water level in the piezometer approximately 3 ft. The water elevation within the piezometer was determined with a meter that detected a closed circuit when the sensor touched water. The water levels were measured every 15 sec at the beginning of the recovery period. Sampling intervals were increased as the response slowed.

Slug test data were analyzed with AQTESOLV (Aquifer Test Solver) program, version 1.1. AQTESOLV uses the Bouwer and Rice (1976) solution for unconfined aquifers. The Bouwer and Rice (1976) solution for hydraulic conductivity is:

Flow net determination

Flow nets for the site were produced from a finite element solution to the equation governing two-dimensional, steady state water flow. FLOTRANS (Guiguer et al. 1993) utilizes a Galerkin finite element approach to determine water potential and stream functions. The program generated equipotentials, streamlines, and velocities throughout the flow field. Flow field dimensions were measured at the site, layer boundaries were taken from the seismic survey, and hydraulic conductivities were calculated from slug tests of piezometers installed at the site as described above.

Figure 6. Positions of piezometers at the study site. Closed circles mark original positions. Open circles mark positions of piezometers added later.

DISCUSSION

Seismic Survey

The initial seismic survey at the site was conducted in the fall of 1991. Most of the time-distance plots from the site showed three or four layers with different velocities. The upper layer with a seismic velocity of approximately 1,000 ft/sec is typical of dry soil. Seismic velocities increased to approximately 5,000 ft/sec, indicating the position of the water table 3-5 ft below land surface. The next recognizable layer had a seismic velocity of approximately 10,000 ft/sec, typical of highly fractured bedrock in the area (Urban and Pasquarell 1992). The deepest layer began about 15-40 ft below land surface, and had seismic velocity typical of unfractured bedrock (16,000 - 25,000 ft/sec). An archetype geologic layer profile for the site is shown in figure 7. The geologic layer profile is a guide for depths at which instrumentation should be placed. Geologic materials with seismic velocities in the range of layer 4 have very low permeabilities and are considered flow restrictive. Consequently, it is unnecessary to instrument the site deeper than the upper boundary of layer 4 (15-40 ft). From the seismic velocities in layers 2 and 3, we anticipate differences in their hydraulic conductivity that will affect how water flows through the riparian zone. Hydrologic properties of each of these layers must be investigated further.

A contour map of depth to bedrock (figure 8), seismic velocity < 16,000 ft/sec, was generated from geologic layer profiles for the entire site. A bedrock high exists near the center of the point bar, and just west of the point bar. We were concerned that the stream may have cut across the meander in the past because of the low relief and abrupt change in stream direction. The contours of bedrock depth provide no evidence of buried stream channels that might divert flow across the point bar. If a buried

Figure 7. Typical profile of seismic velocities at the study site.

channel had been discovered, the site would have been moved. The bedrock high reduces the cross-section for flow and may cause rather constant groundwater levels throughout the year. The point bar was chosen for further investigation, anticipating generalized flow from the bedrock high towards the stream.

Piezometric water potential

Piezometric water potentials were measured once a week for a month during site characterization. A contour map of water potentials following a rain storm (figure 9) shows a groundwater mound over the bedrock high and potential gradients toward the stream in all directions.

Figure 8. Contour map of depth to bedrock at the study site.

Figure 9. Equipotential lines generated from piezometer data following a heavy rain, spring 1993.

A few weeks after the storm, the mound had dissipated and flow was more generally from the field to the stream along the transect of piezometers. Additional piezometers were installed to the east and west of the existing transect to better define the flow field and to provide replication for groundwater chemistry samples and slug testing.

Slug Tests

Slug tests were conducted on two transects of piezometers (figure 10). Many of the piezometers at the 5 ft depth were not tested for lack of a sufficient depth of water. Results of these tests show great variability in hydraulic conductivity both with depth and position (table 1). Hydraulic conductivity ranges over four orders of magnitude, from 0.0034 ft/day at 10 ft depth near the stream to 56.16 ft/day at 7.5 ft depth farthest from the stream. Hydraulic conductivity ranged over three orders of magnitude with depth at position C and with lateral distance at a depth of 7.5 ft along the transect SB to C.

Hydraulic conductivity does, however, generally decrease with dept, and many of the lowest conductivities were measured at positions nearest the stream. With this distribution, much of the flow through the riparian zone is expected to occur in the upper part of the saturated media. The low conductivities bordering the stream may force more water near the surface as the less permeable soils act as a dam.

Figure 10. Position of piezometer nests used to measure hydraulic conductivity.

Flow net

A flow net consisting of equipotentials and streamlines was constructed from a two-dimensional, steady state simulation of water flow along the transect SB to C extended into the field 500 ft (figure 11). An average recharge rate typical of the Mahantango Creek watershed was used as the upper boundary condition and average conductivities for each depth (table 2) were used in the FLOTRANS (Guiguer et al. 1993) program to produce a flow net of the area. The difference in

Table 1. Hydraulic conductivity from slug tests along transects SB to C and SBW to E on figure 10.

hydraulic conductivity between 5 and 7.5 ft and between 7.5 and 10 ft was great enough to defract the flow lines, resulting in the greatest lateral component to flow at a depth of approximately 7.5 ft. Because recharge is uniformly distributed across the surface of the aquifer, the area between each pair of streamlines represents approximately 5% of discharge into the stream. Relatively little (< 15%) of the groundwater discharging into the stream moved through the deeper part of the aquifer. With riparian vegetation 60-100 ft from the stream, most drainage from adjacent agricultural fields moves through the riparian zone in the higher conductivity layer approximately 7.5 ft below the soil surface. Agricultural drainage then discharges into the stream through a narrow band near the stream (figure 11). Riparian processes in areas with flow regimes like that depicted in figure 11 are not expected to remove as much nitrate from agricultural drainage as where there is a greater lateral component to flow from the field through the riparian zone at a shallower depth. The design of chemical and biological data collection programs should make use of the geohydrologic data. However, the program should also be broad enough to test the truth of these interpretations. The flow regime depicted in figure 11 assumes homogeneous layers and reduces a three-dimensional process to two dimensions. Flow nets for the study area generated from three-dimensional simulation models incorporating more of the media heterogeneity may lead to different conclusions about nitrogen removal from agricultural drainage in the riparian zone.

Figure 11. Flow net for transect C to SB of figure 10. Vertical lines are equipotential lines. Horizontal lines are boundaries between layers with differing hydraulic conductivities. Curved lines are streamlines.

SUMMARY

The establishment of riparian zone research sites and subsequent data collection and interpretation consume much time and money. A procedure is presented to relatively quickly explore the hydrogeology of a site to assess its suitability for investigating NPS pollution abatement. A rapid and relatively inexpensive assessment makes it more likely that unsuitable sites will be rejected before so much has been invested in a site that the research question is changed to accommodate the site, rather than changing sites to answer the intended question. Boundaries and dimensions of subsurface materials and qualitative permeabilities are determined with seismic refractive techniques. Water potential gradients are obtained from a network of piezometers. Slug

Table 2. Average hydraulic conductivity from slug tests.

Depth (ft)	Average Hydraulic Conductivity (ft/day)
5	2.5
7.5	12.0
10	1.3
15-18	0.6

tests on the piezometers provide hydraulic conductivities. These data are used to generate prototypical flow nets. Together, these tests should expose hydrological complexities at the site that must then be factored into a data collection program. Flow path information such as contained in figure 11, combined with microbial activity and rooting depths for different geomorphic settings, can be used to qualitatively compare the likelihood that riparian zone processes will buffer streams from NPS pollutants before reaching the stream.

REFERENCES

Altman, S. J., and R.R. Parizek. In press. Evaluation of nitrate removal from groundwater in the riparian zone. In: Riparian ecosystems in the humid East.

Bouwer, H., and R.C. Rice. 1976. A slug test method for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resources Research 12(3):423-428.

Cline, G.D. 1986 Geologic factors influencing well yields in a folded sandstone-siltstone-shale terrain within the East Mahantango Creek watershed, M.S. thesis, Pennsylvania State Univ.

Cooper, A.B. 1990. Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. Hydrobiologia 202:13-26

Correll. D.L., and Weller, D.E. 1989. Factors limiting processes in freshwater wetlands: An agricultural primary stream riparian forest. Edgewater, MD: Smithsonian Environmental Research Center, DOE Symposium Series, R.R.

Domenico, P.A., and F.W. Schwartz. 1990. Physical and chemical hydrology. New York: John Wiley and Sons.

Gburek, W.J., J.B. Urban, and R.R. Schnabel. 1986. Nitrate contamination of groundwater in an upland Pennsylvania watershed. In: Proceedings of agricul tural impacts on groundwater A conference. Dublin, OH: National Water Well Association.

Guiguer, N, J., Molson, T. Franz and E. Frind. 1993. FLOTRANS, two-dimensional steady-state flownet and advective-dispersive contaminant transport model. User guide version 2.1. Waterloo, Ontario: Waterloo Hydrogeologic Software, Waterloo Centre for Groundwater Research.

Haycock, N.E., and T.P. Burt. 1993. Role of flood plain sediments in reducing the nitrate concen tration of subsurface runoff: A case study in the Cotwolds, UK. Hydrological Processes 7:287-295.

Hinsby, K., P.L. Bjerg, L.J. Andersen, B. Skov, and E.V. Clausen. 1992. A mini slug test method for determination of a local hydraulic conductivity of an unconfined sandy aquifer. J. of Hydrology 136:87-106.

Lowrance, R.R., R.L. Todd, and L.E. Amussen. 1983. Waterborne nutrient budgets for the riparian zone of an agricultural watershed. Agriculture, Ecosystems and the Environment 10:371-384.

Lowrance, R.R., R.L. Todd, and L.E. Amussen. 1984. Nutrient cycling in an agricultural water shed: I. Phreatic movement. JEQ 13:22-27.

McDonald, M.G. ,and A.W. Harbaugh. 1988. A modular three-dimensional finite-difference groundwater flow model. In: Techniques of water-resources investigations of the United States Geological Survey, Chap A1, Book 6.

Soil Conservation Survice. 1982. Soil Survey of Schuykill County, Pa. US Dept. of Agriculture.

Urban, J.B. 1977. The Mahantango Creek watershed evaluating the shallow groundwater regime. In: Correll, D.L. ed. Watershed research in Eastern North America. Washington DC: Smithsonian Institute.

Urban, J.B., and G.C. Pasquarell. 1992. Combining well packer tests and seismic refraction surveys for hydrologic characterization of fractured rock. Paper presented at The Sixth National Outdoor Action Conference on Aquifer Restoration, Groundwater Monitoring and Geophysical Methods.