# 16 LANDSCAPE ECOLOGY AND LAND USE

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# THE CHESAPEAKE BAY WATERSHED: EFFECTS OF LAND USE AND GEOLOGY ON DISSOLVED NITROGEN CONCENTRATIONS

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***Abstract:*** We measured the concentrations of dissolved nitrogen fractions in the streams draining 153 subwatersheds of Chesapeake Bay. Six "clusters" of nested watersheds were sampled eight times during the period from the summer of 1992 to the summer of 1993. Clusters were located on the Coastal Plain (2) in Maryland, the Piedmont (1) in Maryland and Pennsylvania, and in the Appalachians (3) in Pennsylvania and New York. Clusters were selected to avoid urbanized areas, but included forest, cropland, pastureland, and residential areas. The sampling times included before, during, and after the record spring Susquehanna River freshet of 1993. Concentrations of nitrate and dissolved ammonium were measured in all samplings of all subwatersheds, while dissolved organic nitrogen was measured on a subset of sites.

Differences in concentrations of dissolved nitrogen fractions were related to the proportion of forest versus agriculture on the subwatersheds and to geological differences among clusters. Although little nitrate was discharged from any of the forested watersheds, high concentrations were discharged from agricultural watersheds. The concentrations of nitrate discharged from agricultural watersheds were highest in the Great Valley, followed by the Ridge and Valley, Piedmont, Coastal Plain, and Appalachian Plateau. Concentrations of dissolved ammonium were 20-fold lower than nitrate and there was less variation among sites, but the Conestoga River usually had the highest and Owego Creek the lowest concentrations. Concentrations of dissolved organic nitrogen were higher than ammonium concentrations but 10-fold lower than nitrate concentrations.

For primarily forested watersheds, nitrate concentrations were much lower. Coastal Plain forests had the lowest nitrate concentrations, but the highest ammonium concentrations. Dissolved organic nitrogen concentrations were much less variable among both forested and agricultural watersheds and thus constituted a larger proportion of total nitrogen in streams draining forested sites.

## INTRODUCTON

The watershed of the Chesapeake Bay is approximately 178,000 km$^2$ and includes the District of Columbia and Maryland and parts of New York, Pennsylvania, West Virginia, Delaware, and Virginia (Correll 1987, Seitz 1971). About 67% of the watershed is in the Appalachians, 15% in the Piedmont, and 18% in the Atlantic coastal plain physiographic provinces. The Appalachian province is composed of three geological subcategories, the Appalachian Plateau, the Ridge and Valley region, and the Great Valley. Of these, the Ridge and Valley is the most extensive in area. The coastal plain province may also be divided into inner and outer coastal plain regions. The outer coastal plain is very sandy and is relatively level, while the inner coastal plain has nutrient rich, well-developed fine sandy loam soils and typically has fairly steep slopes even though the overall relief is small. Major land uses include forest land (especially in the Appalachians), cropland, pastureland, and residential. The distribution or arrangement of land uses also differs characteristically among these physiographic provinces. For example, in the Ridge and Valley region, agriculture is localized in the valleys where the soils are productive and the ridges are usually forested. In the coastal plain, well-drained uplands are farmed, while the wetlands of the drainage divides and the riparian zones are usually forested (Hamilton et al. 1993).

Our goal is to assess the nutrient dynamics of this interesting but complex forested and agricultural landscape in order to better understand the current nonpoint sources of nutrients from the various nonurbanized parts of the watershed and the primary factors that control these nutrient sources. We believe that these pieces of information are needed to underpin better management of the watershed to reduce diffuse nutrient inputs to Chesapeake Bay. Much of our watershed research has been a long-term study of the Rhode River watershed, a coastal plain tributary to Chesapeake Bay (e.g., Correll 1977, 1981, Correll and Dixon 1980, Correll and Ford 1982, Correll et al. 1984, Correll et al. 1992, Jordan et al. 1991a, 1991b). Five years ago, we began to expand this research to other parts of the coastal plain (Correll 1991) and subsequently to the Piedmont regions of the Bay's watershed. Here, we report some of the results of a one-year exploratory study in which 153 streams were sampled in six geologically different regions of the Chesapeake Bay watershed. Our goal in this paper is to describe the major patterns of concentrations of dissolved nitrogen fractions in land discharges from the Chesapeake Bay watershed, especially their relationship to rural land uses and spatially prevalent geological formations.

## METHODS

### Sampling Sites

For each geological region or subregion, we selected the drainage basin of a moderate-sized watershed or in some cases several contiguous watersheds. Sampling stations always included the mainstem stream or streams draining this basin area, as well as stations on the larger tributaries or upstream reaches of this mainstem stream, and some headwaters of low-order tributaries. The combination of all sampling sites within the larger drainage basin was called a watershed "cluster." As much as possible, we selected the smaller subwatersheds we sampled to be representative of various combinations of agricultural and forested land use.

Six clusters of sites were selected (figure 1). Two were in the inner coastal plain. One consisted of 13 sites on the Rhode River watershed south of Annapolis, Maryland and the other included 23 sites on the German Branch of the Choptank River near Centreville, Maryland. One cluster was in the Piedmont and consisted of 21 sites on the Little Falls tributary of the Gunpowder River in Baltimore County, Masryland and York County, Pennsylvania Another cluster was in the Great Valley region of the Appalachians and included 36 sites on the Conestoga River near Lancaster, Pennsylvania. One cluster was in the Ridge and Valley region of the Appalachians and consisted of 24 sites on the Buffalo Creek White Deer Creek basins in northern Pennsylvania near Lewisburg. The sixth cluster was in the Appalachian Plateau region of the Appalachians near Owego, NewYork. It consisted of 36 sites on Owego, Catatonk, and Pipe Creeks.

**Figure 1. The Chesapeake Bay watershed in relation to the eastern United States. Circles mark the sites of our six watershed cluster sampling areas.** The cluster sites are labeled RR for Rhode River, GB for German Branch, LF for Little Falls, CR for Conestoga River, BC for Buffalo and White Deer Creeks, and OCP for Owego, Catatonk, and Pipe Creeks. The bottom figure shows a cross section through the physiographic provinces.

Percentages of the various watershed basins that were forested were estimated from the amount of green area on U.S. Geographical Survey (USGS) topographic quadrangle maps. In our initial field surveys, we found these maps to quite accurately delineate forested areas. For our cluster sites, we purposely avoided basins that contained large towns or cities and associated point sources of nutrients. We also avoided areas impacted by coal mining. For this analysis we assumed, as a first approximation, that areas not forested were agricultural. The proportion of each sampling site's drainage basin in forest and agriculture was classified within 20% ranges (0-20%, 20-40%, 40-60%, 60-80%, and 80-100%).

### Collection and Analysis of Samples

Each of the 153 stream stations were sampled eight times over a one year period. Samplings were taken (1) 15-18 July 1992, (2) 30 August - 2 September 1992, (3) 12-15 October 1992, (4)30 November -2 December 1992, (5) 1-4 March1993, (6) 5-7 April 1993,(7) 3-6 May1993, (8) 8-10 June 1993. Samples were taken in polyethylene bottles that were prerinsed in the stream several times. Samples were immediately filtered through Millipore HA filters (nominal 0.4 um pore size) that had been prewashed with distilled water, then immediately placed on ice until analysis, which was within two weeks.

Nitrate was analyzed by ion chromatography except when concentrations were below 1 umole $l^{-1}$, when it was analyzed by nonautomated colorimetry after copper amalgam reduction to nitrite American Public Health Administration 1976). Ammonium was analyzed by the hypochlorite oxidation technique ( American Public Health Administration 1976). Total Kjeldahl nitrogen (TKN) was digested according to Martin (1972), and the resultant ammonium was steam distilled and analyzed by Nesslerization (American Public Health Administration 1976). Organic nitrogen was calculated as TKN minus ammonium and total nitrogen was calculated as the sum of TKN and nitrate. Triplicate analyses were routinely performed on about 10% of the samples to assess analytical precision.

## RESULTS

### Comparisons among Mainstem Streams

Nitrate concentrations were always highest in the Conestoga River, followed by German Branch, Little Falls, Buffalo Creek, Owego Creek (below the confluence of the East and West Branches), and Muddy Creek (the main tributary of the Rhode River), respectively (figure 2a). There was little variation in nitrate concentration seasonally except that the Conestoga River had lower concentrations in the summer and early fall of 1992 than later, and both Little Falls and Owego Creek had somewhat higher nitrate concentrations in early March. Concentrations of dissolved ammonium were much lower (20-fold) than those for nitrate in all cases (figure 2b) and there was less variation among sites, but the Conestoga River usually had the highest concentrations and Owego Creek had the lowest concentrations most of the time. Concentrations of dissolved organic nitrogen (figure 2c) were higher than ammonium concentrations but much lower (10-fold) than nitrate concentrations. Owego

**Figure 2. Comparison of time series of nitrogen fraction concentrations among mainstrem stream stations for six watershed clusters.** Panel A is nitrate, Panel B is dissolved ammonium, and panel C is dissolved organic nitrogen. Solid round data points are for Rhode River, open squares are for German Branch, shaded squares are for Little Falls, open triangles are for Conestoga River, shaded triangles are for Buffalo Creek, and open hexagons are for Owego Creek.

**Table 1. Comparison of annual mean concentrations of nitrogen fractions in principal tributaries and mainstreams of watershed clusters.** All concentrations are given in umoles $l^{-1}$

Creek dissolved organic nitrogen concentrations were usually the lowest.

Mean annual concentrations of dissolved nitrogen fractions are summarized by watershed cluster, for the larger tributary subwatersheds in table 1. None of the listed subwatersheds is a subwatershed of any of the others in the list, and for each cluster these tributaries comprise over half of the mainstem stream's watershed. There was marked variability in annual mean nitrogen concentrations among these tributaries and sometimes the mean of the listed principal tributaries was quite different from the cluster's mainstem concentrations. For example, nitrate concentrations in German Branch tributaries ranged from 98 to 362 umole l$^{-1}$ with a mean among tributaries of 187 umole l$^{-1}$. German Branch itself had a mean concentration of 315 umole l$^{-1}$. Ammonium concentrations in German Branch tributaries ranged from 6.1 to 18.6 umole l$^{-1}$ and the mean among major tributaries was 9.4 umole l$^{-1}$. German Branch itself had a mean concentration of ammonium of 5.4 umole l$^{-1}$. In contrast, the Conestoga River means were similar to the means for its principal tributaries, even though both nitrate and ammonium varied widely among the tributaries (table 1). Of the total dissolved nitrogen for these cluster mainstem streams, organic nitrogen averaged 32, 9.0, 6.2, 4.2, 13, and 25%, respectively, for the Rhode River, German Branch, Little Falls, Conestoga River, Buffalo Creek, and the Owego cluster. Thus, organic nitrogen was a substantial fraction of the total nitrogen for the Rhode River and Owego clusters. Nitrate concentrations averaged the highest for the Conestoga River, followed by German Branch, Little Falls, Buffalo Creek, Owego Creek, and Rhode River (table 1). Nitrate concentrations for the Conestoga River averaged 22 times those for the Rhode River. Annual mean dissolved ammonium concentrations were highest for the Conestoga River, followed by Buffalo Creek, German Branch, Rhode River, Little Falls, and Owego Creek (table 1). Dissolved ammonium concentrations for the Conestoga River averaged 5.4 times those for Owego Creek.

**Comparisons among Forested Watersheds**

We compared time series of dissolved nitrogen concentrations for watersheds that were mostly forested (80-100%) and that we felt were the best example for a given watershed cluster (figure 3). There were no highly forested tributaries in the German Branch cluster region. Nitrate concentrations did not vary consistently with season and were usually about 10-fold lower for these forested subwatersheds than for cluster mainstem streams (figure 3a). Nitrate concentrations were always

**Figure 3. Comparison of time series of nitrogen fraction concentrations among selected primarily forested watershed streams for five watershed clusters.** Panel A is nitrate, panel B is dissolved ammonium, and panel C is dissolved organic notrogen. Solid round data points are for Rhode River mature forest control, open squares are for Little Falls, shaded squares are for Kettle Run in the Conestoga River basin, open triangles are for upper Deer Creek, and shadded triangles are for the northern tributary of the east branch of Willseyville Creek in the Catatonk Creek basin.

highest for the forested tributary of Little Falls. Dissolved ammonium concentrations were much lower than nitrate concentrations. Ammonium concentrations were generally lower in December and March and were usually the highest for Rhode River (figure 3b). Concentrations of dissolved organic nitrogen were also lower in December and March and were intermediate between those for nitrate and ammonium (figure 3c).

Annual mean concentrations for 18 primarily forested tributaries (table 2) demonstrate the variability among these sites, and the strong contrasts with the results from cluster mainstem streams (table 1). Nitrate concentrations were highest for the Conestoga River, followed by the Little Falls, Buffalo/White Deer Creek, Catatonk Creek, and Rhode River tributaries (table 2). Dissolved ammonium concentrations were highest for the Rhode River forested tributaries, followed by the Conestoga River, Buffalo/White Deer Creek, Little Falls, and Catatonk Creek tributaries. Dissolved organic nitrogen was a higher proportion of total nitrogen in these forested subwatersheds than found in the cluster mainstem streams, averaging 15, 11, 53, and 57%, respectively, for the Little Falls, Conestoga River, Buffalo/White Deer Creek, and Catatonk Creek forested tributaries (table 2).

### Comparisons among Highly Agricultural Watersheds

Time series of nitrate and dissolved ammonium concentrations were compared among subwatersheds of each cluster dominated by agriculture. For example, five highly agricultural Conestoga River subwatersheds (figure 4a) had nitrate concentrations that varied from about 300 to about 1300 umole $l^{-1}$. One that averaged about 800 to 900 umole $l^{-1}$ was selected for comparison purposes. Nitrate concentrations were highest for the Conestoga River agricultural tributaries, followed by Buffalo Creek, German Branch, Little

**Table 2. Comparisons of annual mean concentrations of nitrogen fractions among watersheds that were primarily forested. All concentrations are govin in umoles $1^{-1}$**

**Figure 4. Comparison of time series of nitrogen fraction concentrations among selected highly agricultural subwatersheds for six watershed clusters.** Panel A shows nitrate concentrations for five sites in the Conestoga River basin. Solid round data points are for the Conestoga River mainstrem above Lancaster, Pennsylvania; open squares are for Indian Run, shaded squares are for a small northeastern branch of Middle Creek, and shaded triangles are for upper Hammer Creek. Panel B (nitrate) and panel C (ammonium) are comparisons among representative agricultural subwatersheds of the six clusters. Solid round data points are for the Rhode River watershed, open squares are for Wildcat Branch of the German branch, shaded squares are for Bee Tree Run in the Little Falls basin, open triangles are for Indian Run in the Conestoga River basin, shaded triangles are for Beaver Run in the Buffalo Creek basin, and open hexagons are for a western tributary of the east branch of Owego Creek.

Falls, Rhode River, and Owego Creek (figure. 4b). Dissolved ammonium concentrations were much lower than nitrate concentrations (figure 4c) and tended to be highest for the Buffalo Creek tributaries, especially in the spring and summer.

Nitrogen concentration data for all stations within each cluster were grouped by the proportion of each drainage basin that was forested (table 3 and figure 5). Quite different patterns were found for nitrate (figure 5a) and dissolved ammonium concentrations (figure 5b). For all watershed clusters except the Rhode River and Owego Creek clusters, nitrate concentrations were strongly inversely related to the percentage of forest on the tributary watersheds. For the Rhode River, there was a clear inverse relationship, but the slope was much lower. The Conestoga River, Buffalo Creek, and Little Falls clusters had the steepest slopes (figure 5a). Ammonium concentrations had relatively little relationship to the proportion of forest on the tributary subwatershed (figure 5b), although both German Branch and Rhode River concentrations increased with increasing proportions of forest. Likewise, there was no clear pattern for dissolved organic nitrogen (table 3), although there were fewer data.

## DISCUSSION

Among our study watersheds, those with the highest nitrate concentrations were in the Appalachian province, which accounts for two-thirds of the entire Chesapeake Bay watershed. Susquehanna River discharge, which accounts for about 50% of the total watershed flow and originates almost entirely in the Appalachians, had a mean of 89 umolar nitrate in the spring from 1984 to 1988 (Jordan et al. 1991a).

In this study we measured the concentrations of nitrate discharged from three geologically and culturally different subregions of the Appalachians (table 1). In our Great Valley study cluster, we found quite high, 544 umolar, mean nitrate concentrations in the Conestoga River just above Lancaster, Pennsylvania. However, these concentrations were similar to those reported for 1985-1988 and 1988-1989 of 557 and 595 umolar, respectively, for the river downstream from Lancaster during base-flow. The high nitrate concentrations probably reflect the fact that the Conestoga River basin is highly agricultural with 52% cropland and 11% pasture in 1989. Much of the basin is within Lancaster County, which has some of the highest livestock densities in the United States. The

**Table 3. Annual mean concentrations of nitrogen fractions on streams draining watersheds with differing proportions of forest.** All concentrations are given in umoles $l^{-1}$.

livestock of Lancaster County produced about 4,500 metric tons of manure in 1982, and this production had been increasing rapidly since 1969 (Edwards and Seay 1987). Discharges from the Pequea Creek basin, which lies just south of the Conestoga River basin, averaged 300 umolar nitrate from 1977 to 1979 (Ward 1987). The Pequea Creek basin was also highly agricultural, with 54% rowcrops and 12% pasture.

Our Ridge and Valley study cluster (Buffalo Creek) had fairly intensive agriculture, but this was confined to a smaller proportion of the total basin than in the Conestoga River basin. Forested ridges comprise about half of the total Buffalo Creek basin. This land cover difference is reflected in the somewhat lower (257 umolar) nitrate concentrations we found in Buffalo Creek compared to the Conestoga River. In contrast, our Appalachian Plateau study site, the Owego/Catatonk/Pipe Creeks region, has substantial agricultural lands, but most of this land is used for low-density livestock grazing on the plateau. Relatively little land is in row crops, primarily in the narrow valleys. This difference in the intensity of agricultural land use may be the reason why nitrate concentrations averaged only 63 umolar in these drainage basins.

**Figure 5. Comparison of mean annual nitrate (panel A) and dissolved ammonium (panel B) for the subwatershed sites of each watershed cluster, grouped by percent of forest on the subwatershed.** Symbols for the cluster data points are the sames as for figure 2.

In the coastal plain (table 1), mean nitrate concentrations in German Branch discharges were 315 umolar, while in Rhode River discharges nitrate only averaged 25 umolar. We think that this is the result of two factors. First, the German Branch watershed is much more agricultural, with most of the area in row crop fields that are usually double cropped. The Rhode River watershed is only about 30% agricultural, with little or no double cropping. Second, Rhode River watershed streams are almost continuously lined with wide bands of riparian forest, while many sections of German Branch streams have thinner strips of forest or almost no riparian forest. These coastal plain riparian forests often are very effective at removing nitrate from both surface overland flows during storms (Peterjohn and Correll 1984), and from shallow groundwater moving from upland fields to stream channels (Correll 1991, Correll and Weller 1989, Jordan et al. 1993, Peterjohn and Correll 1986).

In our Piedmont cluster site on Little Falls, nitrate concentrations were also high. The annual mean for one tributary approached the levels found in the Conestoga River (table 1). The majority of the land on the watershed of Little Falls is in row crops and most of the stream discharge is groundwater. The high nitrate concentrations we measured in these streams may reflect high leaching from cropland fields and relatively inefficient removal of nitrate in the riparian zones.

Our measurements of nitrate concentrations in streams draining primarily forested basins in the Appalachians (table 2) can be compared to an 8-year mean of 1.1 umolar reported by Lynch and Corbett (1990) for forested basins in the Ridge and Valley region of central Pennsylvania. Our average nitrate concentrations were higher for primarily forested sites in all clusters, and were much higher on average for primarily forested sites on the Conestoga River basin (table 2). Many of our watersheds contained some houses or other disturbance. This is reflected in the variations among sites (table 2). For example, Kettle Run is the watershed with the least human disturbance in the Conestoga River cluster, and it had an annual mean nitrate concentration of only 22.5 umolar. Similarly, the headwaters of the main branch of Buffalo Creek and White Deer Creek had the least disturbance in our Buffalo Creek cluster, and they had annual mean nitrate concentrations of 2.4 and 6.6 umolar, respectively (table 2). The mature forest control watershed in the Rhode River cluster, which truly has no disturbance except atmospheric deposition, had an annual mean of 7.3 umolar nitrate in its discharges. This may be an indication that mature forests are less efficient at utilizing atmospheric inputs (Aber et al. 1989). Because few if any of our watersheds were truly entirely undisturbed forest, we summarized annual nutrient concentrations for all watersheds within 20% ranges of forest cover within each cluster (table 3 and figure 5). We plan to further refine the land use analyses, but it is already clear that forested watersheds discharged very low nitrate concentrations compared with highly agricultural watersheds, with the possible exception of the Appalachian Plateau cluster.

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# NITROGEN EXPORT FROM FOREST LAND IN THE CHESAPEAKE BAY REGION

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***Abstract:*** Because forestland occupies over half of the total area within the Chesapeake Bay drainage system and nitrogen (N) has been identified as a major cause of eutrophication in Chesapeake Bay, the export of N in streamflow from forested catchments is of great importance in the modeling and management of the Chesapeake Bay. Nitrogen export data for forested basins available from several sources in Chesapeake Bay region were compiled in order to determine the magnitude of N loads lost from forested basins and better understand those factors that cause N loads to vary. Although only a small fraction (< 15%) of atmospheric N deposition appears to be exported from most of these forested catchments, the N loads exported from different forested catchments in kg N $ha^{-1}$ $yr^{-1}$ varied by a factor of ten. Variations in N loads among years and basins were analyzed to determine factors controlling variability in N loads from forested basins. Preliminary analysis suggests that the condition of the forest, especially as it has been affected by repeated past insect defoliations and long-term atmospheric deposition, may be related to N load variations on some of these basins. Nitrogen export data from these forested basins are discussed in relation to current nitrogen saturation hypotheses.

## INTRODUCTION AND DISCUSSION

Nitrogen (N) loading of the Chesapeake Bay has been identified as a key element controlling productivity and eutrophication in the bay. Nitrogen export in streamflow from forestland is of major importance to understanding and managing water quality within Chesapeake Bay. Export of N from forestland is generally agreed to be the lowest per unit of land area of any land use; however, owing to the large forestland area, approximately 60% of the basin, even small changes in N export from forestland could represent very large changes in N loads to the bay. In Chesapeake Bay modeling efforts, N export of only 4.3 kg $ha^{-1}$ $yr^{-1}$ from forestland has been assumed (Linker et al. 1993) which is at least 100% lower than that used for other land use types. Forestland N export rates are also used to predict baseline or pre-development water quality conditions in the Chesapeake Bay basin.

The purpose of this paper is to summarize and discuss N export data in streamflow from forestland within the Chesapeake Bay region which includes the basin and adjacent areas. Nitrogen export data for forestland in the Chesapeake Bay region have not been previously compiled. The authors supplemented data from their own research with previously published data and unpublished data generously provided by other researchers in the region.

### Data Analysis

Studies summarized here are believed representative of forestland in the Chesapeake Bay region. Data from several research locations, which were included in the analysis, occurred just outside the boundaries for the Chesapeake Bay basin. Available data for dissimilar glaciated forested watersheds in the Catskills were too far from northern bounds of the region in New York to be included. All watersheds were essentially 100% forested. Although some of the larger watersheds no doubt contained scattered camps, residences, and roads, data were largely drawn from small undeveloped research watersheds. Only 12 data sources, including 25 streams, were found within the region. Eight of these data sources and 20 streams were located in the Chesapeake Bay basin.

Export refers to the flux, output, or load of N in streamflow leaving a watershed. Owing to a lack of information about each study, no attempt was made to standardize data for various methods of computing export, for the effects of varying "water years" used among studies, or for the varying numbers of samples used in these studies. Finally, all available data were compared regardless of the years represented. Similarities of computed export within broad geographic zones suggest that study methods did not appreciably affect the results. Means, standard deviations and ranges were reported for each data source where possible to indicate variability in annual export at a site over time. Export data from the various sources are expressed in terms of the mass of N per hectare per year.

**Nature of N Export in Forest Streams**

Nitrogen export in streamflow can occur as dissolved $NO_3$, $NH_4$, and organic N as well as undissolved particulate N. Dissolved $NO_3$ export data are most commonly reported for forest land and are emphasized in this paper. Dissolved $NH_4$ export from forested watersheds is generally very small relative to $NO_3$ export (table 1). Forest streams cited in table 1 show $NH_4$-N export averages 0.044 kg ha$^{-1}$ yr$^{-1}$, which is only about 6% of the respective average $NO_3$-N export of 0.70 kg ha$^{-1}$ yr$^{-1}$. Dissolved organic N export is seldom reported for forested watersheds, but available data shows organic N export can be very large relative to inorganic N export (table 1). High organic N export on Young Woman's Creek and Stoney Creek in Pennsylvania (Ott et al. 1991) may be related to sewage contamination from scattered hunting camps and homes on these large forested basins. Limited sampling on Stone, Roberts, and Benner Runs in Pennsylvania by the senior author showed that dissolved Kjeldahl N, which equals the sum of dissolved $NH_4$ and organic N, was generally below detection limits of 0.05 mg l$^{-1}$. Correll (1983) and Correll et al. (1994) reported relatively high total $NH_4$ and organic N export on Rhode River. no. 110, but most of this export may have been associated with particulates. Particulate organic N in forest streams may be relatively inert and thus may not directly affect water quality. Although $NO_3$ export will be emphasized in the remainder of this paper, the importance of other forms of N export from forest land, especially dissolved organic N, deserves further study.

**Regional Variations in $NO_3$ Export**

Annual export of $NO_3$ from forestland within the Chesapeake Bay region (figure 1, table 2) may be viewed as falling generally within two zones with some anomalies. Within zone 1, comprised of the Ridge and Valley and northern Appalachian Plateau Provinces of Pennsylvania, $NO_3$ export is extremely low. Forestland studies in this northern zone by Lynch and Corbett (1991), Pionke (pres.comm.) and north-central Pennsylvania data from Dow and DeWalle (submitted) generally show $NO_3$-N export in streamflow to be < 1 kg ha$^{-1}$ yr$^{-1}$. In this region the Leading Ridge Watershed 1 in central Pennsylvania (Lynch and Corbett 1991) exhibited the lowest $NO_3$-N export of all basins studied; 0.025-0.06 kg ha$^{-1}$ yr$^{-1}$ . Export from Leading Ridge 3 at the same location never exceeded 0.12 kg ha$^{-1}$ yr$^{-1}$ even though this watershed was commercially clearcut during this time period. Within this region only data for Young Woman's Creek (Ott et al. 1991) show $NO_3$-N export greater than 1 kg ha$^{-1}$ yr$^{-1}$. The reason for such high export is not known, but pollution from scattered hunting camps and/or residences, known to exist on this the largest basin considered, is possible.

Farther to the south in zone 2, $NO_3$ export is much higher. Zone 2 is represented by sites in northwestern Maryland (Morgan and Eshleman, written communication), northcentral Maryland (Rice, written communication), southwestern Pennsylvania (Dow and DeWalle submitted), and northcentral West Virginia (Adams et al. 1993, Hicks et al. 1992). Within zone 2, $NO_3$-N export appears to range widely from about 1 to over 8 kg ha$^{-1}$ yr$^{-1}$ (table 2, figure 1). The two West Virginia sites representing five streams are located just outside the western boundary of the Chesapeake Bay Basin, but appear consistent with other sites in zone 2. Sites within zone 2 appear to be clustered, with the very highest $NO_3$ export occurring in northwestern MD (Morgan and Eshleman, pers. comm.) and nearby north-central West Virginia at the Fernow Experimental Forest (Adams et al. 1993).

The highest $NO_3$ export data for forestland in the Chesapeake Bay region were found at the Fernow Experimental Forest on watershed WS 4 (Adams et al. 1993), where $NO_3$-N export exceeded 8 kg ha$^{-1}$ yr$^{-1}$ during 1990. The 1990 period also produced the highest export within the western

Maryland sites (Morgan and Eshleman, written communication) and the north-cental Maryland site (Rice, pers. comm.), suggesting a common cause.

Data for White Oak Run and Deep Run watersheds in nearby northwestern Varginia (Cosby et al. 1991) and a tributary to the Rhode River (watershed No. 110) in central Maryland (Correll 1994) complicate the geographic pattern in zone 2. These watersheds showed low $NO_3$ export similar to that found in zone 1 despite being geographically distant from zone 1. Nitrate concentration data from other streams in western Virginia support the low N export finding for western Varginia. Webb et al. (1989) found a zero median $NO_3$ concentration in a one-time survey of 344 brook trout streams sampled in the Appalachian Mountains of Virginia.

**Nitrogen Saturation**

The large regional differences in $NO_3$ export from forestland within the Chesapeake Bay region can be characterized in relation to N saturation to better understand and predict the response of natural ecosystems to long-term N deposition from the atmosphere. Currently, response of forested watersheds/streams to N deposition is viewed as occurring in stages along a continuum (Aber et al. 1989, Stoddard in press):

Stage 0: Inputs of atmospheric N balanced by uptake into woody biomass and soil microbes with essentially no leaching stream losses.

Stage 1: Inputs of atmospheric N increasing with small leaching losses of N in the dormant season during episodes; uptake still dominant.

Stage 2: Leaching dominant over uptake, major N losses during dormant season events, plant and microbe uptake slows, some elevation of growing season baseflow N.

Stage 3: Watershed a net source of N, export exceeds deposition, high dormant and growing season stream N, acidification of soils and streams, eutrophication.

The export of N from forested watersheds in the Chesapeake Bay region can be roughly interpreted in terms of these stages, but quantitative definition of the stages is not possible.

Few watersheds in this region appear to be in stage 0, in that they all are losing some N in streamflow. Most watersheds in the Chesapeake Bay region appear to be at stage 1, showing small amounts of N loss. Stage 2 conditions may describe the Fernow and western Maryland sites. Annual export from these watersheds is approaching, and in some years exceeding, the typical total N wet deposition found in this region. For example, based upon NADP data for Parsons, West Virginia during the period 1982-93, inclusive, the mean annual wet deposition of N was 6.78 kg ha-1 yr-1, with a range from 5.34 to 8.21 kg $ha^{-1}$ $yr^{-1}$. Data from both Adams et al. (1993) and Morgan and Eshleman (pers. comm.) in table 2 show some annual $NO_3$-N export data within this range. No forested watersheds appear to be at the stage 3 condition; that is acting as net sources of N.

**Causes of Regional Variations in $NO_3$ Export**

Although most watersheds appear to be in a stage 1 condition, large differences in $NO_3$ export exist (table 2). In a very general sense, the export of nitrogen represents a balance between inputs from the atmosphere and uptake by vegetation and soil microbes. Assuming N gains by fixation and losses by denitrification and volatilization are relatively minor, some general observations can be made concerning regional variations in $NO_3$ export.

Wet N deposition as $NO_3$ and $NH_4$ certainly varies across the region, but probably not enough to account for the large differences in export between zones. Differences in wet N deposition do occur from year to year and probably explain some of the annual variation on each watershed. Note that the standard deviations of annual export for watersheds with multiple-year data in table 2 average 26% of the mean, with a range from 9% to 41%.

Dry deposition is much less precisely known and could also vary enough to cause some of the watershed and zonal differences observed. The fact that N export occurs in zones or clusters could result from subregional variations in air pollution and dry deposition.

Response of N export to increased atmospheric deposition is being studied at Fernow (Adams et al. 1993). Data for watershed WS 3 showed an immediate annual response to ammonium sulfate added to simulate enhanced atmospheric N deposition. In contrast, ammonium sulfate added for 4 years to WS 9, a watershed that had been previously farmed and replanted to trees, did not increase $NO_3$ export.

Watershed conditions that can affect N uptake such as forest age, forest disturbances, land use history, and site fertility could also affect variable export response. At Fernow, a mature-to-degrading forested watershed (WS 4) is found next to a

**Table 1. Comparison of the components of nitrogen export in streams on forestland in the Chesapeake Bay region.**

**Table 2. Annual dissolved nitrate-nitrogen export from forestland in the Chesapeake Bay region.**

watershed (WS 3) with actively growing young pole-sized trees regrown after clearcutting in 1969-70 (Adams et al. 1993). The $NO_3$-N export from WS 4 only exceeded that from WS 3 by about 0.5 kg $ha^{-1}$ $yr^{-1}$ for 1986-88, the period that preceded ammonium sulfate treatment. Thus, the differences in N uptake between a vigorously growing young forest and a mature forest at Fernow were not great enough to explain the high export at the WS 4 Fernow site compared to zone 1 sites. In addition, forests at many of the other sites outside Fernow are also at or approaching maturity and thus should also be exporting N at high rates.

Lynch and Corbett (1991) presented data for central Pennsylvania showing that clearcutting increased $NO_3$-N export by no more than 0.1 kg $ha^{-1}$ $yr^{-1}$ relative to uncut forestland. The $NO_3$ export differences between mature forest and recent clearcut were small relative to intraregional differences in export.

Export from WS 4, with mature forest at Fernow, can also be compared to export from WS 9, a historically farmed watershed planted to larch (Adams et al. 1993). Nitrate-N export from WS 4 averaged only about 1.4 kg $ha^{-1}$ $yr^{-1}$ greater than that for the planted, actively growing WS 9. Here again the difference between uptake by a rapidly aggrading forest site (WS 9) and a mature forest (WS 4) at Fernow is too small to explain large intraregional differences in N export.

Forest disturbance owings to insect defoliation can also alter the N cycle on watersheds. Forest on much of the upper portion of the Chesapeake Bay basin has experienced light to severe defoliation for the past 20-30 years, initially by oak leaf roller and more recently by gypsy moth larvae. Currently, defoliation of forests by gypsy moth larvae is occurring in the southern portions of the Chesapeake Bay basin. New, heavy defoliations are also again occurring in the northern portion of Pennsylvania, this time because of the elm spanworm. In years of major defoliation, increases in N export may occur. Later vigorous regrowth following tree mortality caused by repeated defoliations could lead to reduced N export. The overall impact of forest insect defoliation on N cycling has not been intensively studied, but could lead to significant changes in N export over large areas of the Chesapeake Bay basin.

Interpretation of differences in export between watersheds without knowledge of the changes in N stored in soil biomass accompanying vegetation or land use changes is obviously risky. Moreover, we have little knowledge of how N fixation (symbiotic and nonsymbiotic), as well as denitrification and N volatilization, vary among watersheds. Regardless, grouping of N export data into zones or clusters could suggest large differences in intraregional site fertility caused by climate and geology. Deep, fertile soils supporting undisturbed forest may simply have a larger N storage component in soil biomass, causing more $NO_3$ to be produced by nitrification and creating a higher leaching loss.

To illustrate the possible role of site fertility in controlling N export, data from DeWalle et al. (1988) were used to show large differences in soil water chemistry between Fernow WS 4 and a southwestern Pennsylvania site near Linn Run. Nitrate was the dominant anion in B-horizon soil water on Fernow WS 4 (72 µeq/l mean annual $NO_3$-N concentration), while sulfate was the dominant anion in soil water near Linn Run watershed with a mean annual $NO_3$-N of only 25 µeq/l. The ratio of these B-horizon soil water concentrations of 72/25=2.9 agrees reasonably well with differences in $NO_3$-N export between WS 4 and Linn Run given in table 1 (5.15/1.97=2.6). Ratios of B-horizon soil water for Ca (3.8) and Mg (2.7) also indicate greater fertility at Fernow relative to the Linn Run region. The relatively low site fertility near Linn Run is typical of most of the Appalachian Plateau in Pennsylvania. Thus, the high N export at Fernow relative to Linn Run may result from naturally higher soil fertility and biomass N. Fertility of soil at all sites should be compared for a more complete understanding of N dynamics.

Watershed hydrology cannot be overlooked as a possible source of variation in nitrate export within and among catchments. Conditions that lead to precipitation or melt water bypassing the soil matrix could reduce the opportunity for uptake by vegetation or soil biomass. Both overland flow, which may occur during extremely large rain and melt events, and macropore flow through the soil could allow water and the associated nitrogen to bypass the soil. For example, Dow et al. (in press) found that 0.26 to 0.53 kg $ha^{-1}$ of $NO_3$-N was lost from watersheds in north-central Pennsylvania (Baldwin, Stone, Roberts) during one major melt event after the blizzard of 1993. A loss of this magnitude exceeds the annual export reported in table 2 for these watersheds in a more typical year. Regional differences in hydrogeologic conditions that affect the fraction of precipitation that bypasses the soil via fracture and macropore flow could also influence the geographic pattern of N export.

## CONCLUSION

Two zones of $NO_3$ export from forest land were found within the Chesapeake Bay region. Zone 1 had low $NO_3$-N export, generally < 1 kg ha$^{-1}$ yr$^{-1}$. Zone 1 represented the northern Pennsylvania portion of the region and the geographically separated mountains of western Virginia and eastern Maryland coastal plain. Zone 2 represented much higher $NO_3$-N export ranging from about 1 to over 8 kg ha$^{-1}$ yr$^{-1}$. Zone 2 occurred primarily in the western portion of the Chesapeake Bay region in northwestern Maryland and northern West Virginia.

Based upon available data, the nitrogen export rate used to model forestland in the Chesapeake Bay basin is considerably higher than measured export rates from forested basins. Within the Chesapeake Bay region, dissolved $NO_3$-N export data for 25 forest streams averaged only 2.1 kg ha$^{-1}$ yr$^{-1}$, while dissolved $NH_4$-N export data for 13 forest streams only averaged 0.074 kg ha$^{-1}$ yr$^{-1}$. In comparison to the 4.3 kg ha$^{-1}$ yr$^{-1}$ total N export from forest land assumed in the Chesapeake Bay model, measured $NO_3$ and $NH_4$ export could only explain about half of the total.

Shortcomings of available N export data for forestland need to be recognized. Mean N export data given above are heavily biased toward 12 basins in the high-export zone 2, even though zone 2 appears to cover a relatively small portion of the Chesapeake Bay basin. Measured export does not account for export of dissolved or particulate organic N; a part of which could be in biologically available forms. Finally, about one-half of the basins surveyed are small, undisturbed experimental basins that do not contain human habitation and possible impacts from sewage. These shortcomings in available measured export data may be producing over- or underestimates of the true N export rate from forestland. More studies of total N export from forestland are needed, with careful attention to effects of sewage and other land use disturbances and the spatial pattern of N export within the Chesapeake Bay basin.

None of the forested watersheds in the Chesapeake Bay region appear to be at an advanced stage of nitrogen saturation according to the Aber/Stoddard classification. Most basins probably are at stage 1, where N uptake by vegetation and soil biomass is still dominant and only small amounts of N are leached. A few basins in extreme western Maryland and north-central West Virginia (Fernow) may be reaching stage 2, where $NO_3$ export is approaching N inputs from atmospheric wet deposition.

Possible causes of the large regional variations in annual $NO_3$ export, such as atmospheric wet and dry deposition, forest age, land use changes, forest insect defoliation, soil fertility, and basin hydrology have been discussed. Forest cutting and stand age effects do not appear to explain the large regional differences in $NO_3$ export. Regional differences in site fertility related to climate, geology, and soils are probably largely responsible for nitrogen export differences, but effects of regionally variable air pollution/dry deposition and watershed hydrology cannot be ruled out. Insect defoliations could markedly change nitrogen loads from forestland in the future. Also, we know little about variations in other N sources (fixation) or sinks (denitrification) on forested watersheds across the region.

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# NITRATE EXPORT FROM MANAGED AND UNMANAGED FORESTED WATERSHEDS IN THE CHESAPEAKE BAY WATERSHED

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***Abstract:*** Twenty years of streamflow and nitrate concentration data from the Leading Ridge Experimental Watersheds in central Pennsylvania were analyzed to determine the impacts of a 110-acre commercial forest harvest and a 90-acre clearcut-herbicide treatment on nitrate export. The commercial forest harvest employed best management practices (BMPs) to limit degradation of stream quality. Nitrate concentration and export data from both experiments were compared to an adjacent unmanaged, control watershed. Both the commercial clearcut and the clearcut-herbicide treatment resulted in significant ($p < 0.05$) increases in nitrate concentrations. The greatest increases occurred the first year after harvesting and remained elevated for approximately two years on the commercial harvest. The rapid recovery to near preharvest concentration levels was attributed to rapid regeneration on the cutover area. Elevated nitrate concentrations were more persistent on the clearcut-herbicide treatment because of suppressed regeneration of many of the hardwood species. Nevertheless, an influx of herbaceous vegetation, the second year rapidly reduced nitrate leaching to the stream. Despite the elevated nitrate concentrations, nitrate export from the commercial clearcut watershed never exceed 0.54 kg/ha/yr and has been within $\pm$0.05 kg/ha of the average annual preharvest export level of 0.34 kg/ha for the past 15 years. Similar, although somewhat higher, export values were recorded on the clearcut-herbicide treatment. In comparison, annual nitrate export from the unmanaged watershed ranged from 0.11 kg/ha to 0.26 kg/ha. Wet nitrate deposition to these watersheds averages approximately 23 kg/ha/yr. Export data from the control watershed indicates that retention of nitrate deposition on this watershed is very high, generally exceeding 99% of the annual wet deposition. The results of these experiments also indicate that timber harvesting on these and similar hardwood forests will not appreciable increase nitrate loading to Chesapeake Bay.

## INTRODUCTION

The Chesapeake Bay is one of the largest, most productive estuaries in the world. Unfortunately, the water quality of the Bay has been deteriorating since the time of the Industrial Revolution (Fisher et al. 1988). Nutrient enrichment has been identified as a major factor in the decline of Chesapeake Bay. Increased delivery of nitrates to the Bay is of particular concern because nitrogen limits primary productivity in many estuarine environments and increased loading may lead to eutrophication and deterioration of water quality. Major sources of nitrogen to surface waters within the Bay watershed include agricultural runoff, discharges from sewage treatment plants and industrial sources, atmospheric deposition, and runoff from managed and unmanaged forestlands. Forestland runoff is of particular interest because over 60% of the watershed is classified as forestland. Consequently, any silvicultural activities within the Bay's watershed that might stimulate increased leaching of nitrogen to surface waters is of concern.

Changes in stream chemistry following timber harvesting were not considered a problem until the late 1960s following publication of results of astudy on the Hubbard Brook Experimental Forest in New Hampshire (Bormann et al. 1968, Likens et al. 1970). Since then a number of studies have shown that changes in the concentration of a number of nutrients, including nitrogen, do occur following timber harvesting (Aubertin and Patric 1972, 1974, Corbett et al. 1978, Douglass and Swank 1975, Federer et al. 1989, Hornbeck et al. 1987, Kochenderfer and Aubertin 1974, Lynch and Corbett 1990, Lynch et al. 1985, Lynch et al. 1975, Martin et al. 1984, Martin and Pierce 1980, Reinhart 1973, Swank and Douglass 1975). Increases in nutrient concentrations following harvesting have been attributed to accelerated nutrient leaching owing to exposure of the site to greater-than-normal amounts of heat and moisture, acceleration of the nitrification process, and increased leaching of nutrients owing to the loss of uptake following plant removal (Likens et al. 1970). These studies have also shown that the changes are highly variable, very site specific, and generally related to the severity of the cut and the rate of revegetation of the cutover area. The rate and degree of revegetation appear to control the longevity of increased nutrient concentrations.

Two things most forest-harvesting and nutrient-cycling studies have in common are that most have been restricted largely to nutrient concentrations and that most have been relatively short-term studies, lasting 3 to 5 years.Because timber harvesting, particularly clearcutting, substantially increases stream discharge (Hibbert 1967, Lynch et al. 1972, Hornbeck et al. 1993), evaluation of its impact on nutrient cycling must take this into consideration. For example, increases in nitrate concentrations that generally follow forest harvesting are actually greater when increased flow effects are considered. Consequently, the most effective way to evaluate the effects of forest management activities on nutrient cycling is to express the changes in both concentration (mg/l) and export units (kg/ha), the latter of which considers both concentration and discharge interactions.

The purpose of this paper is to report on short- and long-term changes in nitrate concentrations and export from a 44.5ha commercial clearcut, a 43ha clearcut-herbicide experiment, and an undisturbed (unmanaged) forested watershed, all of which are located in central Pennsylvania. The results of this study are interpreted in light of potential nitrate contributions to Chesapeake Bay from forestlands. These results are also compared to relative nitrate contributions to Chesapeake Bay from other nonpoint sources.

**Site Description**

This study was carried out on the Leading Ridge Experimental Watersheds located in the Ridge and Valley province of central Pennsylvania. The three adjacent watersheds are 123 (LR1), 104 (LR3), and 43 (LR2) ha in area. These watersheds were selected to be representative of approximately 4 million ha of forestland, much of which lies within the Chesapeake Bay watershed.

Watersheds within this research unit are equipped with modified broad-crested Trenton weirs with a sharp-crested, 90-degree, V- notch in the center to measure stream discharge. Streamflow is monitored using FW-1 water level recorders and a seven-day chart cycle. All watersheds have southeastern aspects and range in elevation from 244 to 442 m. Mean slopes on the watersheds vary from 12% to 17%, with maximum slopes approaching 50%. Soils on the lower slopes are primarily silt loams and stony loams that are well drained and have high moisture holding capacity. The middle and upper slopes include well-drained cobbly loams and stony loams with high moisture holding capacity. The ridge top is composed of cobbly and sandy loams. The average depth of the soil mantle is <2 m. Eight soil series are found on the watersheds. They include four Ultisols—Andover, Buchanan, Clymer, and Laidig—and four Inceptisols—Berks, Dekalb, Hazleton, and Weikert.

Repeated timber harvesting plus the effects of fire and disease during the 1800s have resulted in the present-day, uneven-aged forest. Changes occur in the vegetative community from the valley to the ridge top. The ridge top and upper slope consists of a community of red oak (*Quercus rubra Ashe.*), chestnut oak (*Quercus prinus L.*), black oak (*Quercus velutina Lamb.*) and pitch pine (*Pinus rigida Mill.*). The lower slope and bottomland species include white pine (Pinus strobus L.), eastern hemlock (*Tsuga canadensis (L.) Carr.*), black birch (*Betula lenta L.*), red, black, and white oak (*Quercus alba L.*), and tulip poplar (*Liriodendron tulipfera L.*). The understory dominants include black gum (*Nyssa sylvatica Marsh.*), red maple (*Acer rubrum L.*), flowering dogwood (*Cornus florida L.*), and witch hazel (*Hammamelis virginiana L.*).

The streams draining each basin develop from two perennial, first-order channels and several intermittent channels. Runoff takes place primarily as subsurface flow, rather than surface runoff. Monthly stream discharge, reported as a percentage of the total annual streamflow, increases steadily from October to April, after which it declines throughout the growing season. Historically, about 40% of the annual precipitation occurs as streamflow.

## Research Approach

The paired-watershed method was used to evaluate changes in stream discharge and nitrate concentrations and export following a commercial harvest and a clearcut-herbicide experiment on these central Pennsylvania watersheds. Leading Ridge watershed 1 (LR1) was used as the undisturbed control watershed. Leading Ridge Watershed3 (LR3) was selected for the commercial clearcut, while Leading Ridge watershed 2 (LR2) was clearcut in three stages (lower slope, middle slope, and upper slope-ridge top) and sprayed with herbicides following both the middle slope and upper slope clearcuts to control regrowth. Changes in water quantity were based on linear regression analyses using preharvest stream discharge measurements from treated and control watersheds. The annual and seasonal regression equations and measured stream discharge from the control watershed were used to predict stream discharge from the treated watersheds, assuming their vegetative cover had not been altered. The difference between the predicted discharge and the measured discharge on the commercial clearcut watershed and the clearcut-herbicide treatment represent the change in streamflow as a response to treatment. Statistical significance of differences between predicted and measured values was determined using standard t-test techniques.

Changes in stream nitrate concentrations on the commercial clearcut watershed were based on chemical analyses of stream water samples collected during a 3-year calibration period (October 1973 to September 1976) and an 11-year postharvest period. The statistical significance of differences between the control and harvested watersheds before and after harvesting was determined using analysis of variance techniques. No pretreatment nitrate concentration data were available for the clearcut-herbicide treatment. Routine stream quality samples were collected weekly with some sampling during stormflow periods to ensure that all discharge classes were represented. Although only nitrate fluctuations are discussed in this paper, all samples were analyzed for pH, sulfate, calcium, magnesium, potassium, sodium, and alkalinity ($CaCO_3$) as well. These analyses were conducted in the water quality laboratory of the Environmental Resources Research Institute at the Pennsylvania State University. All laboratory analyses followed methodology recommended by the U.S. Environmental Protection Agency (1983). All concentration (mg/l) and export (kg/ha) units are given in mg $NO_3$/l and kg $NO_3$/ha.

Using the time-stage hydrographs, each water sample and corresponding nitrate concentration taken over the study period was matched with the stage at the time of sampling. Predictive relationships for nitrate were developed by tabulating the mean concentrations from weekly sample observations falling into 0.3-foot intervals of stage height for each watershed. Ionic export was then calculated for the digitized stage values (time increments for stage readings varied as necessary to record details in stage hydrographs) by converting stage to area inches of water and multiplying this volume by the predicted ionic nitrate concentration for the observed stage based on the above statistical relationships. These incremental export estimates were then totaled into seasonal and annual export estimates from each watershed. Statistical comparisons of pre- versus postharvest concentration and stream discharge data using standard analysis of variance techniques were used to evaluate the significance of the commercial harvest on nitrate export from these experimental watersheds.

The commercial clearcut harvesting, which was conducted following established Pennsylvania Bureau of Forestry policies, commenced on 1 October 1976 and ended on 3 May 1977; 44.5 ha of the 104-ha watershed were harvested. In order to control nonpoint pollution during and following logging, best management practices (BMPs) were developed and employed on this commercial harvesting. For a detailed description of this study and the BMPs that were used, see Lynch and Corbett (1990).

The clearcut-herbicide treatment was initiated in 1966 with the harvesting of 8.5 ha on the lower slope of LR2. The clearcut was expanded upslope by an additional 10.9 ha harvest in 1972. The final forest harvest on this watershed was conducted in 1976 with the removal of all commercial timber on 17 ha on the upper slope and ridge top. In 1974 the entire cutover area (19.4 ha) on the lower and middle slopes was sprayed with herbicides to eliminate regrowth of all woody and herbaceous species. The application of herbicides was repeated in 1977 and included the upper slope and ridge top portion as well. It should be emphasized here that the herbicide applications to this watershed were not intended to represent normal forest management operations in the region, but were used as an experimental method of preventing vegetation regrowth to test for maximum water yield increases.

## RESULTS

### Stream Discharge

Water yield increases resulting from the commercial clearcut are given in Table 1. The first-year increase was 13.7 area-cm, all of which occurred during the growing season. The dormant season showed a non-significant decrease of 2.8 area-cm. The first year-increase is equivalent to 32 cm on an area-cut basis, in that only 43% of the 104ha watershed was clearcut. The annual yield increase was sharply lower the second year following harvesting, amounting to only 3.4 cm. The second-year decrease was greater than generally reported for similar studies and was attributed to the rapid regrowth on the cut-over area that increased evapotranspirational losses and an unequal distribution of rainfall during the summer and early fall months. Rainfall was below normal for four of the six growing season months even though growing season precipitation was above normal. The magnitude of water yield increases is influenced strongly by the amount and distribution of both seasonal and annual precipitation (Lynch et al. 1972). The decreasing trend in water yield increases continued through the third growing season. By the fourth growing season, water yield increases were statistically unaffected by harvesting. Very similar results have been reported as a result of similar experiment conducted throughout the northeastern United States (Hornbeck et al. 1993).

### Nitrate Concentrations

Mean annual streamwater nitrate data for the commercial clearcut and control watersheds prior to and following harvesting are presented in table 2. Mean monthly nitrate concentrations from 1974 through 1988 for both the control and commercial harvest are shown in figure 1. A comparison of nitrate concentrations from the commercial harvest with the clearcut-herbicide treatment is shown in figure 2. Nitrate concentrations during the 3-year calibration period were very similar on both the control and commercial clearcut watersheds. Following harvesting, nitrate concentrations increased significantly ($p < 0.05$). However, these increases were quite small and relatively short lived. By the end of the third year following harvesting, nitrate concentrations had returned to near preharvesting levels as had earlier increases in stream discharge. Increases in nitrate concentrations were attributed to an increase in the decomposition of logging debris and a reduction in the uptake of nitrogen owing to harvesting and a corresponding increase in leaching rates. As is evident in figure 1, increased nitrate concentrations were most evident during the growing seasons. The rapid decline in nitrate concentrations on LR3 was attributed to rapid regeneration of the cut-over area. Such a response is ideal when nitrate losses through leaching are to be kept to a minimum.

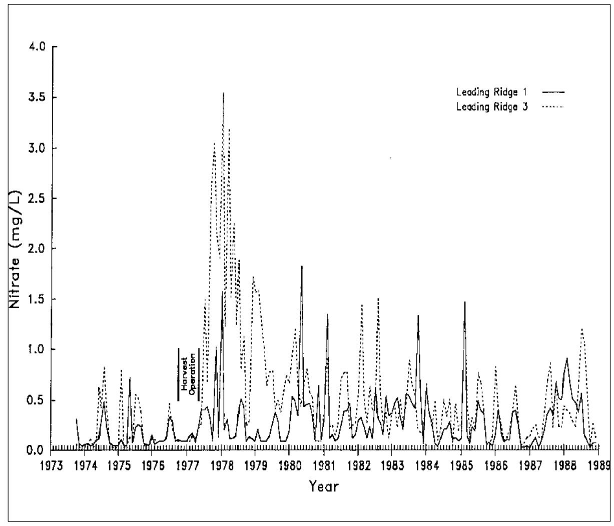


**Figure 1. Trends in stream water nitrate concentrations (mg/l) for the undisturbed (LR1) and commercial clearcut (LR3) watersheds before (1973-76), during (October 1976-May 1977), and after harvesting.**

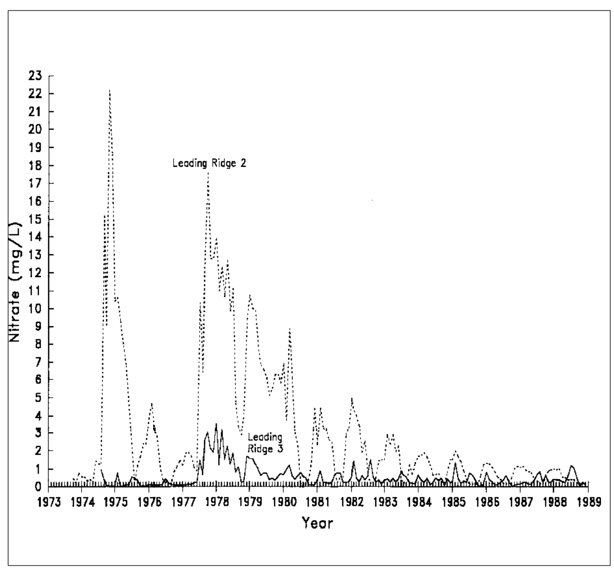


**Figure 2. A comparison of trends in stream water nitrate concentrations (mg/l) for the commercial clearcut (LR3) and clearcut-herbicided (LR2) watersheds.**

**Table 1. Annual and seasonal water yield increases following the commercial clearcutting of 44.5 ha on Leading Ridge watershed 3.**

| | Water Yield Change | | |
|---|---|---|---|
| Water Year (May-April) | Growing Season[1] | Dormant Season[2] | Annual |
| 1977 | +14.55* | -2.85 | +13.75 |
| 1978 | +2.74* | +0.43 | +3.43* |
| 1979 | +1.40* | -1.04* | +1.78* |
| 1980 | +0.13 | +6.04* | +5.08* |
| 1981 | -3.31 | -1.12 | -3.66 |
| 1982 | +0.06 | -2.02 | -2.65 |
| 1983 | +0.79 | +0.13 | +1.44 |
| 1984 | +0.22 | +0.62 | +0.80 |
| 1985 | +0.30 | +0.72 | +2.23 |

[1] May through October
[2] November through April
[3]$p > 0.05$

**Table 2. Mean annual nitrate concentrations (mg/l) and annual nitrate export (kg/ha) from commercial clearcut (LR3) and undisturbed (LR1) watershed before and after harvesting.**

| | Watershed | | Watershed | |
|---|---|---|---|---|
| Period | LR1 mg/l | LR3 mg/l | LR1 Kg/ha | LR3 Kg/ha |
| Preharvest | 0.03 | 0.04 | 0.18 | 0.34 |
| Postharvest | | | | |
| Year 1 | 0.11 | 0.04[1] | 0.22 | 0.50[1] |
| Year 2 | 0.05 | 0.28[1] | 0.21 | 0.41 |
| Year 3 | 0.05 | 0.14[1] | 0.26 | 0.54[1] |
| Year 5 | 0.05 | 0.12 | 0.11 | 0.24 |
| Year 7 | 0.10 | 0.10 | 0.18 | 0.34 |
| Year 9 | 0.05 | 0.08 | 0.15 | 0.31 |
| Year 11 | 0.11 | 0.09 | 0.14 | 0.25 |

[1]$p < 0.05$

The importance of rapid regeneration is also evident when nitrate concentrations from the clearcut-herbicided watershed are compared with the commercial clearcut. Herbicides were applied in the summer of 1974 and again in 1977. With the elimination of virtually all vegetation on this watershed, nitrate concentrations increased rapidly and were seven to eight times greater than measured on the commercial clearcut watershed. Despite the repeated use of herbicides on this watershed, an invasion of herbaceous plants rapidly reduced nitrate losses to near prespraying levels within 5 years. Since 1985, nitrate concentrations on the clearcut-herbicided watershed have generally remained below 1.0 mg/l and only slightly higher than the control watershed. Although this study was not designed to simulate normal silvicultural practices in the mid-Atlantic region, the results illustrate nicely the resiliency of forest ecosystems to overcome severe disturbance with minimal loss of nitrogen.

**Nitrate Export**

Nitrate export from the commercial clearcut was quite small over the 11-year postharvesting period, ranging from 0.24 kg/ha to 0.54 kg/ha. On the unmanaged watershed nitrate export ranged from 0.11 kg/ha to 0.26 kg/ha during the same period (table 2). The combined effects of increasing stream discharge and nitrate concentrations on the commercial clearcut resulted in slightly higher, statistically significant ($p < 0.05$) increases in nitrate export the first and third year after harvesting. As would be expected, the majority of the increased export occurred during the growing season (figure 3), owing primarily to an increase in streamflow as a result of reduced evapotranspiration. Nitrate export from LR3 in the second year following harvesting was also elevated, however, not significantly so (table 2). With revegetation of the cut-over area and a rapid decline in stream discharge increases and a subsequent decline in nitrate leaching, nitrate export returned to precutting levels by the end of the third year and has remained at these levels for the duration of the study. Although slight increases in nitrate concentrations have been observed during the growing seasons (figure 1), these increases have been insufficient to significantly increase annual or seasonal export of nitrogen owing to the low flow conditions under which they occur.

**DISCUSSION**

The observed changes in stream water nitrate concentrations and export do not represent a signification degradation of stream quality, nor do they indicate a significant deterioration of site quality. Rapid recovery to preharvest levels and subsequent minimal loss of nitrate was attributable largely to rapid revegetation of the cutover area. At no time during the course of this study did nitrate concentrations exceed drinking water standards. This was also true for the severely disturbed clearcut-herbicided watershed. It appears that the "Best Management Practices" developed by the Pennsylvania Bureau of Forestry were very effective in controlling nitrate loss from this silvicultural operation and that even severely disturbed watersheds in the mid-Atlantic region have the capacity to retain nitrates as long as a vegetative cover is maintained.

In the context of total nitrate loading to Chesapeake Bay, it would appear from the results from

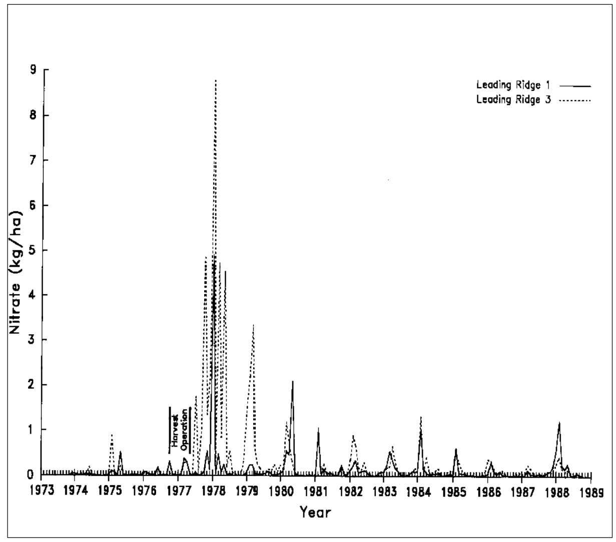


**Figure 3. Trends in nitrate export (kg/ha) from the undisturbed (LR1) and commercial clearcut (LR3) watersheds before (1973-76), during (October 1976-May 1977), and after harvesting.**

this study that forestlands do not represent a major nonpoint source of nitrogen to the Bay compared to other land uses. It also appears that commercial forest harvesting in the Chesapeake Bay watershed is unlikely to significantly increase nitrogen export to the Bay, especially when rapid revegetation of the site is achieved. Total point and nonpoint nitrate loading to Chesapeake Bay has been estimated to range from $12.4 \times 10^7$ to $35.0 \times 10^7$ kg (Tyler 1988). It has also been estimated that approximately $10.0 \times 10^6$ ha of forestland are found within the Bay's watershed (Tyler 1988). If it is assumed that nitrate losses from the undisturbed LR1 watershed are representative of all unmanaged forestland in the Bay's watershed, annual nitrate export to the Bay would range from $1.1 \times 10^6$ kg/yr to $2.6 \times 10^6$ kg/yr. The average nitrate export from LR1 was 0.18 kg/ha/yr over the 16-year study period. This would equate to an average export of $1.8 \times 10^6$ kg/yr from all forestland in the Chesapeake basin over the 16-year period. In comparison to the total nitrate load to the Bay, unmanaged forestlands would contribute on average 0.5% to 1.5% of the total nitrate load depending upon what estimate is used. Assuming nitrate losses from LR3 are representative of intensive silvicultural activities on forestlands in the Bay, nitrate export to the Bay would not exceed $5.4 \times 10^6$ kg annually if all forestlands were managed. This represents only 1.5% to 4.5% of the estimated total nitrate load to the Bay.

Nitrate export from forestlands in the Bay is also smaller than that estimated to result from direct deposition of atmospheric nitrates to surface waters within the Bay's watershed. It has been estimated that Chesapeake Bay proper plus its tributaries account for approximately 6.3% (1.15 x $10^6$ ha) of the total surface area of the basin (Tyler 1988). It has also been estimated that approximately 30.6 x $10^6$ kg of nitrate are deposited annually to these surface waters by atmospheric depositional processes (Tyler 1988). Using the highest measured nitrate export from the Leading Ridge Experimental Watersheds (0.54 kg/ha for managed forestlands and 0.26 kg/ha for unmanaged forestlands, table 2), atmospheric contributions directly to surface waters are almost 6 times larger than that coming from managed forestlands and 12 times greater than that from unmanaged forestlands in the basin.

It may be argued that nitrate losses from the Leading Ridge watersheds are not representative of all forestlands in the Chesapeake Bay watershed. Indeed, studies on the Fernow Experimental Watersheds in West Virginia (Aubertin and Patric, 1974) and the Hubbard Brook Experimental Forests in New Hampshire (Martin et al. 1984, Hornbeck et al. 1987) have shown larger increases in nitrate concentrations and export following commercial harvesting. Dow and DeWalle (submitted) estimated nitrate export in 1988 from three unmanaged forested watersheds (Benner, Roberts, and Stone Runs) in north-central Pennsylvania to be 0.9 kg/ha, 2.56, and 3.0 kg/ha, respectively. Nitrate export from these watersheds during a four-week period following the blizzard of 1993 were 2.3 kg/ha, 1.4 kg/ha, and 1.1 kg/ha, respectively. Export from these watersheds during the spring snowmelt period was estimated to account for 22% to 44% of the total nitrate deposition on these watersheds from the nearly 1 m snowfall. DeWalle and Pionke (1995) also summarized nitrate-nitrogen data from other watershed studies within and near the basin. The results indicate considerable intrawatershed variability ranging from < 0.1 kg/ha/yr to over 5 kg/ha/yr. However, they urge caution in interpretation of these data owing to differences in the length of record, methods used to calculate export, number of observations, past land-use history, etc.

The importance of land-use, both past and present, is extremely important and must be considered in evaluating nitrate export from forestlands. The authors have observed significantly higher nitrate concentrations on three south-central Pennsylvania watersheds that were severely defoliated by gypsy moths in 1984-85. Nitrate concentrations on Bear Gap, Wildcat Run, and Sweetroot, all located on the Buchanan State Forest in Bedford County, averaged 0.82 mg/l, 0.52 mg/l, and 3.17 mg/l, respectively (Unpublished file report). Following a severe forest fire that burnt more than 3,650 ha on three watersheds in central Pennsylvania in 1990, the authors observed nitrate concentrations in stream water that were significantly higher than an adjacent unburnt control watershed. Nitrate concentration on Jews Run, the most severely (nearly 100%) burnt watershed, averaged 2.95 mg/l for the 3 years following the fire compared to 0.52 mg/l on the unburnt watershed. Nitrate concentrations on two additional watersheds that had approximately 50% of their forest cover destroyed averaged 1.47 and 1.42 mg/l during the3-year postfire period. Although stream discharge was not measured on any of the streams, nitrate export was estimated using a simple water balance approach for the region. In central Pennsylvania, approximately 40% of the annual precipitation occurs are runoff. Given an annual precipitation of 102 cm and adjusting for seasonal changes in nitrate concentrations and discharge, nitrate export for Jews Run was estimated to be 4.2 kg/ha/yr compared to 0.7 kg/ha/yr for the control watershed. Approximate 2 kg/ha/yr of nitrate was estimated to be exported from the other two watersheds as a result of the fire.

It is obvious that considerable variability exists in the amount of nitrogen exported from forestlands and that current and past landuse can be an important determining factor. It is also evident, that forestlands retain a very large percentage of nitrogen deposited on them by atmospheric processes and that, in comparison to other land uses in the region, represent a significant nitrogen sink. Nevertheless, if accurate estimates of nitrate export to the Bay from forestlands are to be undertaken, additional monitoring of streams draining both managed and unmanaged forests will be necessary.

## ACKNOWLEDGMENTS

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1-3 June 1994. Norfolk, VA
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# GROUND WATER CONTROLS ON HYDROLOGY AND WATER QUALITY WITHIN RURAL UPLAND WATERSHEDS OF THE CHESAPEAKE BAY BASIN

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***Abstract:*** Streamflow from rural watersheds tributary to the Chesapeake Bay is derived primarily from groundwater. Upland watersheds within the nonglaciated portion of the Bay basin are commonly underlain by bedrock that is severely fractured at shallow depths as a result of stress-relief fracturing. The highly conductive shallow fracture layer is a water transfer zone of major importance, controlling groundwater flow and its quality both within and from the watersheds. The shallow fracture zone interacts with the regional aquifer, providing it with recharge in the uplands and receiving its discharge in the near-stream zone. It also supports a lateral saturated flow component that influences the dynamics of upland springs and seepage areas along with flow through the riparian zone. The Agricultural Research Service of the U.S. Department of Agriculture at University Park, Pennsylvania, has devoted a major portion of its recent research effort toward characterizing this fractured hydrogeology and quantifying its role in the hydrology and water quality of rural upland watersheds with their mix of forest, cropland, pasture, and suburban land use. Here we summarize this research. Characterization of the fractured aquifer geometry and its hydraulic properties are first presented. Modeling of groundwater flow and flowpaths is illustrated, and effects of land use distribution on patterns of nitrate contamination within the groundwater and its transport to the stream are shown. Finally, insight is provided into upland groundwater controls on the near-stream environment in context of seepage face development as a source of storm runoff production, and riparian zone flow dynamics for potential reduction of nitrate concentrations generated upgradient within the watershed.

## INTRODUCTION AND DISCUSSION

The Susquehanna River provides about one-half of the freshwater input to Chesapeake Bay. The Appalachian Valley and Ridge physiographic province is a major landform within the Susquehanna basin, and as such is a potentially important control on water quantity and quality inputs to the Bay. Perennial streams from Valley and Ridge upland watersheds have much of their total flow derived from subsurface sources; these same upland areas are also typically underlain by bedrock that is severely fractured and weathered at shallow depths as the result of stress-relief fracturing (Wyrick and Borchers 1981). Percolate escaping the root zone drains directly to the highly conductive shallow layer of fracturing, making it a water transfer zone of major importance influencing both hydrology and water quality. The shallow fracture layer supports a lateral saturated flow component, which contributes directly to upland springs, seeps, and streamflow, and it also interacts with the regional aquifer, providing it with recharge in the uplands and receiving its discharge in near-stream zone.

To evaluate in-basin groundwater, riparian zone, and downstream water quality impacts of land management practices within this setting, we must understand the controls exerted by ground water flow on contaminant transport. The Pasture Systems and Watershed Management Research Unit, of the Agricultural Research Service of the U.S. Department of Agriculture,(USDA-ARS) at University Park, Pennsylvania has devoted a major portion of its recent research program toward characterizing hydrogeology within upland watersheds of the Valley and Ridge Province and assessing its effects on ground water, streamflow, and associated water quality. Here we describe the physical setting and aquifer geometry and hydraulic properties, and develop simulations of subsurface flowpaths and patterns of nitrate quality within the groundwater of an upland agricultural watershed typical of the nonglaciated, folded, and faulted Appalachian Valley and Ridge physiographic province. Delivery of groundwater from over the watershed to the riparian zone is also considered. Finally, implications for water quality management within rural upland watersheds are developed.

## Study Area

WE-38 is a 7.2km$^2$ upland agricultural watershed within the Chesapeake Bay drainage ( figure. 1). It is located immediately to the east of the Susquehanna River approximately 40 km north of Harrisburg, Pennsylvania .Climate is temperate and humid, with average annual precipitation of approximately 1,100 mm and streamflow of about 450 mm, roughly 40% of precipitation. Mature forest covers the dominant ridge forming the northern watershed boundary, while cropland, pasture, and small woodlots dominate the rolling hills of the watershed interior. Farming activities are primarily dairy, poultry, and cash cropping, and recommended conservation practices and fertilizer and pesticide application rates are usually followed. The typical crop rotation is corn-wheat-hay-meadow, with about 35% of the watershed in corn in any one year. Shallow residual soils less than 1.5 m deep, mostly silt loams, cover the watershed. The underlying bedrock consists of two formations dipping to the north. The Trimmers Rock formation (Late Devonian) is primarily shale and outcrops at the watershed outlet at the south in a near-horizontal position. The overlying Catskill formation (Late Devonian-Early Mississippian) consists of interbedded shales, siltstones, and sandstones, becoming increasingly coarse grained to the north. Dip of the Catskill strata increases to about 30$^{o}$ where a relatively pure quartz-sandstone-conglomerate outcrops to form the northern ridge. Rainfall, climate, soil water, groundwater, and streamflow data have been routinely collected on WE-38 since 1967. Water quality analyses of soil water, groundwater, and streamflow are also available over most of this same time period.

### *Groundwater*

Findings of past research into the role of ground -water in upland watersheds of the Valley and Ridge province are generalized as follows: groundwater return flow can account for up to 60-80% of annual streamflow; water tables exhibit steep gradients directly related to surface topography; major groundwater divides are coincident with topographic divides; rock fracturing is more significant than rock type in controlling groundwater flow in the regional aquifer; and well yields decrease substantially below about 90 m depth (Cline 1968, Urban 1977). Recent research efforts are integrating the results of rock coring, seismic testing, and hydraulic testing of wells and coreholes (Urban and Pasquarell 1993), instru-

**Figure 1. Watershed WE-38 location, topography, and generalized geology.**

mented cross section investigations (Gburek and Urban 1990), and groundwater modeling (Gburek et al. in press) to characterize watershed-scale geometry and hydraulic properties of the layered fractured bedrock and its controls on the ground water flow system. Results of all studies are summarized in table 1 as generalized depths, seismic signatures, fracture frequencies, and hydraulic properties of the characteristic fracture layers. Our assumption is that these all are watershed-scale values.

***Nitrates***

Nitrate ($NO_3$) is considered to contribute to the eutrophication of Chesapeake Bay (Environmental Protection Agency 1983). Further, it is generally accepted that a major impact of agriculture on water quality is nonpoint-source contamination by $NO_3$. Agricultural land use has been shown to affect the quality of watershed runoff (e.g. ,Alberts et al. 1978, McDowell et al. 1984), as well as the quality of groundwater (e.g., Spalding and Exner 1980, Edmunds et al. 1982). The literature relating land use and / or fertilizer applications to runoff quality generally implies that when assessing nutrient input to streamflow and the potential for its control, there is an immediate and predictable relationship between land use and streamflow quality. However, the literature relating land use to groundwater quality rarely considers the interrelationships between land use practices, groundwater quality, and the quality of streamflow leaving the watershed. If groundwater is a major contributor to total streamflow, as within the Bay basin, its quality dynamics should dominate the chemical loads discharged by upland watersheds. Thus, water quality modeling efforts related to Chesapeake Bay should include the factors controlling nitrate loads generated from these upland watershed areas, that is, water and $NO_3$ recharge to groundwater, subsurface flow dynamics, resultant flow and $NO_3$ inputs to the streams draining the watersheds, and downstream effects.

Land use distribution controls nitrate input to the groundwater of WE-38; management varies by farm and crop, but this variation is not significant enough to override the generalization that corn production is the prime source of $NO_3$. Surface runoff exports little $NO_3$ or organic N from WE-38 (Pionke et al. 1988). Consequently, total $NO_3$ export from WE-38 is controlled by groundwater discharge, both during and between storm events. Groundwater discharge accounts for 70%-80% of the water exported, and its $NO_3$ concentrations are at least 10 times higher than those of surface runoff (Schnabel 1986, Pionke et al. 1988). Total nitrate-nitrogen ($NO_3$-N) export from WE-38 was 27, 50 and 39 kg $ha^{-1}$ in 1970, 1983, and 1984, respectively (Gburek 1977, Gburek et al. 1986); corresponding annual average $NO_3$-N concentrations in the stream were 5.4, 8.1, 5.9 mg $l^{-1}$.

Inputs of $NO_3$ to groundwater originate within the root zone as fertilizer additions or decomposition of manure, plant residue, and soil organic matter. Nitrate losses from the root zone have been found to approximate those in streamflow, suggesting that the below-root-zone flow system is mainly a zone of transmission (Pionke and Urban, 1985). Nitrate-nitrogen in wells representing the aquifer of WE-38 between 10 m and 60 m depth was found to be lowest beneath forest, averaging less than 1 mg $l^{-1}$, and highest under cropland in the midwatershed position, approximately 5 mg $l^{-1}$; concentrations in deeper groundwater near the watershed outlet were about 1 mg $l^{-1}$ (Pionke and Urban, 1985). Nitrate-nitrogen concentrations within the shallower portions of the fracture layer (< 15 m) at the near-stream cross-section sites relate directly to overlying land use, 7 to 22 mg $l^{-1}$ under a corn-based strip-crop rotation, and less than 1 mg $l^{-1}$ under meadow (Gburek and Urban 1990). Concentrations of $NO_3$-N in baseflow at the WE-38 outlet are generally about an order of magnitude higher than those in surface runoff, over 5 mg $l^{-1}$ as compared to less than 1 mg $l^{-1}$, respectively (Pionke et al. 1988).

***Groundwater Modeling***

MODFLOW (McDonald and Harbaugh 1984) has been used in both areal and cross-section formats to successfully model groundwater flow within WE-38 for determination of hydraulic parameters (Gburek et al. in press). In the areal simulations, the aquifer was represented by a 25 x 27 grid of cells, each 122 m square, in the horizontal, and six cells in the vertical representing the three layers detailed in table 1; all layer boundaries were assumed to parallel the land surface. Lateral and bottom watershed boundaries were defined as impermeable, and stream nodes were positioned to represent the channel system. Using the hydraulic parameter values in table 1, observed water table topography was successfully simulated under average annual recharge rates, as was time-variable watershed drainage representing an extended baseflow recession.

**Table 1. Summary of aquifer geometry and properties.**

| Layer<br>Depth range, M | Highly Fractured<br>3-9 m | Moderately Fractured<br>9-24 m | Regional aquifer<br>24-90 m |
|---|---|---|---|
| Seismis velocity, m sec-[1] | 1,100 - 2,300 | 3,000 - 4,900 | 4,3000 - 6,100 |
| Fractured frequency, m-[1] | 36 | 20 | < 5 |
| K, m day-[1] | | | |
| Regional aquifer testing (Cline 1986) | n/a | n/a | 0.1 |
| Packer testing (Urban and Pasquarell 1993) | 3.8 | 0.7 | ~0 |
| Near-stream study[1] (Gburek and Urban, 1990) | 2.2e 0.6w | 0.64e 0.3w | n/a |
| Areal modeling (Gburek et al. in press) | 15. | 0.15 | 0.015 |
| Near-stream modeling (Gburek et al. in press) | 3.0 (9.)[b] | 0.15 | 0.015 |
| $S_y$ | | | |
| Regional aquifer testing | n/a | n/a | 0.0001 |
| Packer yesting | n/a | n/a | n/a |
| Near-stream study | n/a | n/a | n/a |
| Areal modeling | 0.005 | 0.001 | 0.0001 |
| Near-stream modeling | n/a | n/a | n/a |

[1]e and w indicate east and west cross section
[2]parenthetical value is K used for near-stream zone

Simulations in the cross-section format were designed in context of the same layered fractured geometry. These were carried out in complete divide-to-stream profiles representing both near-stream cross-section study sites (Gburek and Urban 1990). Fine grid spacing was used to characterize detailed flow geometry in the shallower parts of the profile and in the near-stream zone. Deeper depths and positions further removed from the stream were represented with a more coarse grid. Using the hydraulic parameters indicated in table 1, steady state simulations representing both high- and low-recharge rates successfully matched equipotential distributions in the near-stream environment observed under similar conditions in the watershed.

### *Integration and Objectives*

While groundwater quality in general is the major influence on long-term quality of streamflow leaving the watershed, knowledge of the pathways of groundwater flow within the watershed is needed to tie this influence to land use and management at the surface of the watershed. Groundwater investigations and observed $NO_3$-N concentrations within WE-38 indicate that patterns of $NO_3$ in groundwater and streamflow are controlled by the layered fractured bedrock and a relatively local scale of landuse distribution. Thus, the critical question is: Can our current knowledge of the hydraulics of the groundwater flow system aid in understanding these patterns? Knowing aquifer geometry and hydraulic parameters, we can model typical groundwater flow systems at the watershed scale and analyze resultant patterns of flowpaths to infer the fate of contaminants introduced to groundwater. Typical cross sections at both the watershed and local scale can then be modeled for groundwater flow, flowpaths, and associated travel times, and patterns of $NO_3$-N concentration within groundwater can be simulated based on observed and hypothetical distributions of land use. Delivery of groundwater from the watershed to the riparian zone can be examined within these same flow fields. Based on the simulations, implications for water quality management of rural upland agricultural watersheds within the Chesapeake Bay watershed can be addressed.

### Watershed-Scale Areal Modeling

The geometry and hydraulic parameters of table 1 were input to MODFLOW to simulate the steady state groundwater flow system in a three-dimensional areal format under an annual average recharge of 1.0 mm day$^{-1}$ inferred by annual average baseflow these are the same conditions discussed abovein the section groundwater modeling. The solution provides the basis for the flowpath modeling examined here. To minimize possible effects of bottom geometry on flowpath patterns, the impermeable boundary was assumed to be a plane underlying the entire watershed beginning approximately 90 m below the watershed outlet and sloping upward parallel to the generalized land surface slope toward the northern watershed divide. This change in geometry from the impermeable bottom uniformly 90 m below the land surface used in the original simulations (Gburek et al. in press) was found to have minimal effect on flows and head values.

***Flowpaths***

MODPATH and its companion, MODPATH-PLOT (Pollock 1989), simulate and display flowpaths by tracking particle movement within steady state groundwater flow simulations from MODFLOW. Figure 2 shows flowpaths resulting from particles being introduced at the surface of the water table in the center of each 122-m$^2$ cell used for modeling the three-dimensional flow field within WE-38. MODFLOW cells are the squares on the figure, flowpaths are the finer lines, and the channel network is bold. The pattern of flowpaths indicates that there should be only local-scale effects of land use on groundwater and the receiving streams. In most cases, the flowpaths originating at the water table move approximately downgradient following land surface slope and emerge to the channel network either directly at the nearest stream or, at most, into the downslope stream of the next highest order. Exceptions are a few flowpaths beginning along the major divide that bypass the first-order streams and enter higher order streams approximately one-third of the way down the watershed. No major groups of flowpaths pass under one channel to emerge at another further downgradient. Thus, the generalization can be made that when the area of concern is at the scale of a second-order stream or greater, land use affecting the groundwater and groundwater-driven streamflow is only that within the subwatershed.

Also of note is the observation that many of the flowpaths, once in the vicinity of the channel, turn and move parallel to the channel some distance before they emerge to the stream. The subsurface topography used in the modeling is such that all layer boundaries except the impermeable bottom are defined as parallel to the land surface. This assumption produces a trough of high conductivity beneath all channels that results in dominant subsurface flow components under and parallel to the streams; this phenomenon was noted by Gburek and Urban (1990) when examining equipotential patterns under the two experimental cross sections. However, the minimal distances groundwater travels along these flowpaths does not alter the previous conclusion regarding the scale at which watershed areas influence groundwater and stream quality.

**Figure 2. Flowpaths initiated at the water table in the center of all MODFLOW cells; annual average recharge.**

**Watershed-Scale Cross-Section Modeling**

Patterns of flowpaths, travel times, and concentration distributions can be visualized more easily within a cross-section format. The watershed-scale section designated as A-A on figure. 2 is examined here, assuming that it is a self-contained flow cross section,( i.e., all groundwater flow to the channels crossing section A-A comes only from within the section itself). Figure 2 supports this assumption in that the four channels crossing the section are roughly orthogonal to the section and flowpaths within the section are generally orthogonal to these streams. MODFLOW was applied to the cross-section geometry shown in figure. 3 using the same 122 m grid spacing in the horizontal and the same layer geometry, hydraulic properties, and node spacing in the vertical as was used for the three-dimensional modeling. Lateral boundaries were assumed impermeable, and the plane, sloped, impermeable bottom boundary can be seen clearly. Flow was assumed to exit the section only at the four crossing streams. The average annual recharge of 1.0 mm day$^{-1}$ was used to simulate the steady state flow field within the cross section, and water table topography from the simulation compared favorably to observed well levels within or near the section. Thus, the groundwater flow field simulated within this watershed-scale cross section is assumed to be a valid two-dimensional representation of the three-dimensional flow field at the watershed scale.

Figure 3 shows selected flowpaths within the watershed-scale cross section; the bottom boundary of the zone of fracturing at 24 m depth is shown for reference. All flowpaths portrayed begin at or near topographic divides, the most critical positions for defining the subsurface flow field in strongly layered systems (Gburek et al. 1991). As in the areal simulation, no nested subsurface flow systems appear within the cross section, (i.e., no flowpaths pass under one stream to emerge at another lower in elevation and further downgradient along the section). Basically, the entire cross section can be viewed as consisting of eight relatively independent divide-to-stream groundwater flow systems. Flowpaths shown range from those beginning at the ridge top divide and dipping deep into the profile, in some cases approaching very near to the impermeable bottom, then returning to the stream (i.e., the deepest flowpath within the section from 0 to 1,000 m from the watershed divide), to the other extreme of those remaining entirely within the fractured portion of the aquifer (i.e., the shallowest flowpath shown within the flow system from about 1,300 to 1,700 m from the divide). Recall, though, that these are only flowpaths selected that begin in the vicinity of topographic divides. Shorter flowpaths that begin in positions farther downslope from those shown are omitted, but these will all be in shallower positions than those shown. This simulation indicates that land use in any position on the watershed will affect only the local-scale groundwater flow system beneath it and, consequently, only the stream receiving discharge from the same groundwater system.

Travel times along the flowpaths are also indicated; the dots on the flowpaths are spaced at 10-day increments of travel. These times are developed by MODPATH using effective porosity; in our case, values of specific yield ($S_y$) were used. Because there is virtually no matrix conductivity or porosity in the fractured bedrock of WE-38, this can be considered a valid first approximation. Also note that these travel times reflect flow only through the saturated flow system; times associated with movement through the unsaturated system are not considered here. Maximum travel times along the flowpaths at the north of the section (the flow system spanning 0 to 1,000 m along the x-axis) are over 90 days from watershed divide to stream. Travel times are approximately 150 days along the deepest flowpath in the flow system spanning 1,300 to 1,700 m. Travel times along the other deeper flowpaths from local divides to streams range from approximately 40 to 80 days, but there was no attempt to portray the longest and deepest of all possible flowpaths. (Travel times are further addressed when examining the local-scale cross section.)

**Figure 3. Selected flowpaths and travel times within a watershed-scale cross section.**

## Local-Scale Cross-Section Modeling

That part of the watershed-scale cross section from 1,200 m to 2,700 m in figure 3 was isolated and examined independently to portray the groundwater flow system in more detail. A local-scale cross section like this can be extracted from the watershed-scale flow field while still maintaining all its important characteristics because the larger section was shown to consist only of a series of isolated flow fields; approximate boundaries of symmetry occur at all groundwater divides and points of groundwater discharge. This local-scale cross section (figure 4) has two crossing streams and three interstream areas of contributing land use, each approximately 500 m in length. Previous geomorphic analysis of WE-38 indicated that the typical divide-to-stream distance was on the order of 500 m, so the scale of groundwater patterns within this local-scale cross section should be indicative of those at the watershed scale. Flow within the local-scale cross section was modeled with MODFLOW under the annual average rate of recharge using the same assumptions of internal and external geometry as were used in the previous modeling. However, horizontal grid spacing was reduced to 30 m for better resolution of the flow field and evaluation of effects of land use on patterns of groundwater contamination.

### *Flowpaths and Travel Times*

As was done in the watershed-scale cross section, selected flowpaths beginning near all water table divides are shown in figure 4a. They range from those dipping very close to the impermeable bottom to those remaining entirely within the shallowest zone of fracturing. Still other flowpaths begin at the water table farther downslope and are both shallower and shorter than those shown. While this figure supports the earlier conclusion — land use affects only the local-scale groundwater and the stream draining that part of the cross section — it more obviously defines the local groundwater flow systems as a function of local groundwater divides rather than those of surface topography. For example, this simulation shows that topographic and groundwater divides near 1,700 m are separated by over 100 m.

Maximum times of travel range from 110-plus days for the bottommost flowpath in the south to 160-plus days for that in the north; this exceeds times along the deepest flowpaths of the watershed-scale cross section. Travel times along the shallow flowpaths range from 10-plus days in the north and south to 60-plus days in the middle section flowing from the divide to the south. A few short and fast flowpaths appear within the small flow system from the middle divide to the northern stream; these represent a very compact but complete subsurface flow system formed by the watershed topography. Basically though, the earlier conclusions remain unchanged. Once contaminants are introduced into the groundwater system, even if near the divide, their effects may be felt throughout the groundwater body and in the stream within a time scale of months, rather than that of years commonly associated with subsurface flow systems having higher specific yields. Further, flowpaths originating closer to the stream may transport contaminants through the groundwater flow system within periods of days or weeks because they are moving only through the high-conductivity low-porosity zone of fracturing.

### *Nitrate Concentration Distributions*

Patterns of $NO_3$ concentration within the groundwater resulting from spatially variable land use over the watershed can be simulated by using root zone and contaminant transport models in conjunction with the groundwater flow system developed. EPIC (Williams et al. 1984), a physically based one-dimensional root zone model, has previously been run for both continuous corn and meadow (the latter assumed equivalent to woods or other unmanaged land uses) to represent extremes in potential nitrate input to groundwater on WE-38 (Gburek et al. 1991). These runs used parameters characterizing Meckesville soil (typic fragiudult), a deep, well-drained, channery loam typical of soils over the interior of WE-38, and nutrient inputs (fertilizers and manure) appropriate to each land use as determined by farm surveys on the watershed. Percolate quantity simulated by EPIC compared favorably to that derived from field data, monthly mean $NO_3$-N concentrations in percolate from the pasture simulations compared favorably to concentrations found in shallow groundwater under meadow, and the range of concentrations simulated for percolate from continuous corn compared well to concentrations observed in both deeper and shallower aquifers under cropland (Pionke and Urban 1985, Gburek and Urban 1990). Annual average $NO_3$-N concentration simulated in percolate (assumed equalto recharge to ground water) was 16 mg $l^{-1}$ from corn and 0.2 mg $l^{-1}$ from meadow.

A simple mixing cell-based contaminant transport model was developed to simulate nitrate transport (assumed dissolved and conservative) through the groundwater flow system output by MODFLOW simulations (Gburek et al. 1991). Distribution of land use over the aquifer is represented by nitrate concentrations from EPIC simulations being input to the contaminant transport model as areally variable concentrations in recharge. The resulting pattern of groundwater quality is then represented by the within-cell concentration distribution from the transport model.

This suite of submodels was used to simulate nitrate patterns in the local-scale cross section based on three patterns of land use over the cross section, the actual and two hypothetical. For the actual configuration (figure 4b), land use was resolved as either forest or a strip-crop rotation of approximately 35% corn and 65% unfertilized crops. Nitrate input to groundwater from the strip-crop land use was characterized by weighting inputs from the two croplands in appropriate proportions, giving a $NO_3$-N concentration of approximately 6.0 mg $l^{-1}$. The strip-crop and forest land uses indicated by the icons on figure 4b represent the actual land use distribution over the section within resolution of the grid used in modeling. Figure 4b also shows the simulated distribution of $NO_3$-N in groundwater resulting from this pattern of land use. To develop the $NO_3$-N contours, cell-by-cell concentrations from the transport simulation were contoured using commercial gridding and contouring software.

The forestland use at the southern end (right) of the section produces $NO_3$-N concentrations < 1.0 mg $l^{-1}$ at all aquifer depths to below the stream at lateral positions from approximately 2,200 to

**Figure 4. Selected flowpaths and travel times within a local-scale cross section (a), simulated patterns of $NO_3$-N concentrations under actual land use (b), and two variations in land use position (c and d).**

2,600 m. Similarly, groundwater quality in the northern end (left) of the section (1,200-1,600 m) is dominated by the strip-crop land use over that area. The center of the section shows more dramatic differences. Here, effects of the forestland use bordering the south side of the northernmost stream (at 1,600-1,800 m) extend more obviously to the south within the groundwater body (1,700-2,000 m). This wooded land use is almost entirely to the north of the topographic divide, but the recharge it produces affects groundwater only under the topographically defined watershed to the south, providing a tongue of concentrations < 1.0 mg $l^{-1}$ to a position nearly under the stream at 2,100 m.

Figures 4c and d illustrate effects of landuse positioning. Because about 35% of WE-38 is in corn, this same percentage of each interstream zone in figures 4c and d is assumed to be "planted" in corn as one continuous field, with a $NO_3$-N concentration of 16.0 mg $l^{-1}$ in its percolate. The other 65% of the land is assumed to be unfertilized, with a percolate $NO_3$-N concentration of 0.2 mg $l^{-1}$. The concentration patterns of figure 4c result from cornfields being positioned as shown by the icons: at the southern divide, immediately adjacent to the northern stream, and directly over the middle ridge. Viewing this cross section as three independent flow fields, the following observations are made. Positioning the corn land use at the divide (south) contaminates the entire depth of the aquifer below with a $NO_3$-N concentration of 16.0 mg $l^{-1}$. Downslope of the corn at shallower depths however, concentrations within the fracture layer exhibit progressive dilution of the 16.0 mg $l^{-1}$ by recharge from the unfertilized land use. In the north, where corn is placed adjacent to the stream, it affects only the shallowest portions of the flow system and then only immediately below the corn field. Concentrations of substantially less than 16.0 mg $l^{-1}$ occur since this level of input is diluted by large amounts of low-concentration groundwater originally recharged from the unfertilized land use upslope. The middle interstream portion of the cross section is also worth reconsidering. Even though the corn land use is spread equally over both sides of the topographic divide, it affects only the groundwater and stream to the south. As mentioned previously, the groundwater divide is over 100 m to the north of the topographic divide, so any land use within that distance of the topographic divide will only affect groundwater to the south. For this land use positioning simulated in figure 4c, there is no effect on the groundwater flow system and stream to the north.

Figure 4d shows changes to these patterns resulting from moving the cornfields slightly. Simply moving the corn land use one node (30 m) from the southern divide substantially reduces nitrate concentrations within the regional aquifer, generally to less than 10 mg $l^{-1}$ $NO_3$-N, and moderately reduces nitrate concentrations in all downslope portions within the fracture layer. This move illustrates the very sensitive nature of the recharge zone to repositioning of contaminant inputs over this strongly layered hydrogeology — moving the most intense land use only tens of meters off the divide can greatly reduce contamination of the regional aquifer. Also, the corn previously adjacent to the stream was moved to a position approximately midway upslope. Results of this move are obvious but not substantial. There is still no major contamination of the deeper aquifer from this positioning. Only the zone of fracturing continues to be impacted, and even this is minor; maximum concentrations are only about 8 mg $l^{-1}$, and these occur only near the water table. When the middle field of corn is moved entirely to the north side of the topographic divide, it still impacts the groundwater to the south of the divide, further illustrating that groundwater and topographic divides are offset from one another in this area.

Not immediately obvious from figures 4c and d is that $NO_3$-N concentrations of the two streams (i.e., contours directly below the streams) are virtually identical under both land use configurations. This occurs because these are steady state simulations with the same aggregate input of nitrate and percolate to each section in each case. This leads directly to the conclusion that for closed systems and nonreactive contaminants, positioning of land use over the section does not affect the long-term average concentrations or loads of contaminants leaving the watershed under baseflow conditions as long as the relative percentages of land uses remain constant and the disparity between topographic and water table divides is considered.

## Riparian Zone Considerations

Elevated levels of $NO_3$ are expected in recharge from agricultural land use, but the resultant high concentrations of $NO_3$ in groundwater may be attenuated enroute to becoming streamflow if the ground water flows through riparian ecosystems (e.g., Lowrance et al. 1984, Correll and Weller, 1989). Cooper (1990) concluded that catchment hydrology, particularly flow paths through the riparian zone, determine the control of near-stream environments on pollutant flux. Under humid-climate conditions, as in the Chesapeake drainage, groundwater convergence to the stream is the prime cause of the high-moisture, high-water-table, riparian conditions effective for nitrate reduction, but as shown previously, subsurface geometry and its controls on the ground water flow system at the watershed scale also influence these conditions.

Whether riparian zone processes can substantially reduce nitrogen discharge into a stream depends on both the areal extent of the riparian zone (RZ) and the hydrologic linkages between upland nitrate sources (i.e., farm fields) and the RZ. A variety of linkages with varying degrees of complexity are possible on a watershed, ranging from landscape positions with short, shallow, direct paths between cropped areas and riparian zones, to those with deep flow paths through limited riparian zones. When flow paths are direct and shallow, nearly all agricultural drainage passes laterally through the RZ before discharging to the stream. In contrast, when flow paths are less constricted, more of the groundwater discharge can bypass some or all of the RZ, either as deeper flow before converging to the channel, or as loss to a deeper groundwater system. Several factors are important in defining the total interactions between flow paths and the RZ, but results of the modeling described here can help develop inferences on operation of the RZ from the hydrologic viewpoint. As emphasized, flow path definition is critical because flowpaths describe delivery of recharge from the agricultural land use to the RZ. Of prime importance is the origin of those flowpaths passing through the RZ, because only land uses from which these flowpaths originate have the potential to be affected by RZ processes. We illustrate this by reconsidering the local-scale cross-section modeling.

### *Local-Scale Cross Section*

Figures 5a and b show flowpaths within the groundwater flow system spanning the distance from approximately 1,200 to 1,700 m in figure. 4. These simulations were produced based on recharge rates representing spring-time high water table conditions (3 mm day$^{-1}$) and late-summer dry conditions (0.1 mm day $^{-1}$). Flowpaths shown indicate proportions of total groundwater moving above the flowpath. This can be done by portraying flowpaths that begin at the water table at distances from the stream representing the proportion of flow desired. Because recharge is uniform over the land surface, the proportion of flow moving to, and consequently through, the groundwater flow system is that proportion of land surface captured between any two adjacent flowpaths. Flowpaths are shown that encompass 98%, 95%, and 90% of total groundwater flow, with additional flowpaths being reduced sequentially by 10% increments until definition becomes impossible.

Figure 5 shows that recharge rate affects flowpath position, and thus partially determines the percentage of total groundwater flow potentially affected by RZ processes. The high-recharge case shows that approximately 90% of the total groundwater flow remains within the shallow fracture layer, thus approaching the stream in a very shallow position. The low recharge simulation shows that only 80% of the flow remains entirely within the shallow fracture layer; the 90% and greater flowpaths approach the stream from significantly deeper positions within the subsurface flow system. Consequently, there is more of a potential for these flowpaths to bypass RZ processes. While there is minimal difference between the two cases, there is nonetheless an indication of the potential for effects of recharge rate and the larger-scale

**Figure 5. Flowpaths for riparian zone considerations; percentage of total groundwater flow in cross section above flowpaths indicated.**

hydrogeology to contribute to the effectiveness of the riparian zone for $NO_3$ removal from groundwater.

### *Near-stream Conditions*

Figures 6a and b focus on the near-stream portion of figures 5a and b. A "riparian zone" 20 m wide and 3 m deep is shown for reference; these literature-based dimensions represent an area potentially active in removal of nitrates from groundwater (e.g., Jacobs and Gilliam 1985, Lowrance 1992). It is easier to evaluate potential for the RZ to remove nitrates from upgradient land uses in this portrayal in that entry of specific flowpaths into the RZ is clearly defined. Figure 6a, the high-recharge configuration, shows that 95% of the groundwater flow, and consequently, 95% of the land surface area, is potentially affected by RZ processes. Figure 6b shows that under low-flow conditions, recharge and groundwater from only about 80% of the land surface can be fully influenced by the riparian zone. However, the hydrologic status of the watershed associated with the two figures must also be considered. The high-recharge configuration, allowing more of the groundwater to pass fully through the RZ, represents spring-time conditions, when denitrification processes are less effective because of cooler temperatures and residence times within the RZ are shorter because of higher flowrates. Conversely, when the potential for denitrification is greater under higher temperatures later in the year and longer residence times, the flow regime shows that more of the flowpaths tend to bypass the R2. Thus, effectiveness of RZ processes alone must be integrated with the hydrologic conditions indicated by flowpath simulation to fully evaluate the potential for a landscape to attenuate $NO_3$ inputs to the groundwater from upgradient land uses.

**Figure 6. Flowpaths for riparian zone consider ations; focus on near-stream zone.**

### Implications for Land Management

Rural upland watersheds of the Chesapeake drainage are of mixed land use — cropped and forested land uses interspersed with small urban or suburban areas all contribute to the same flow systems. Contaminants introduced into the flow systems from the agricultural land uses are generally "nonpoint" in nature, and their effects are manifested at several time and space scales and in more than one flow component. Over the long term, the upland watershed's hydrologic system is fixed in time and space. There are seasonal variations in flow components, as well as event-generated variations, but basic hydrologic conditions remain relatively constant from year to year. There is little opportunity to manage the hydrology of these groundwater-dominated systems at a scale large enough to affect the overall flowpaths; thus, water quality management strategies will likely be nonhydrologic in nature: assessment of land use within the watershed; control of agricultural chemicals on specific land uses; and control of land use distribution over the watershed. Control and management of chemicals applied is the best overall strategy, but this strategy should still be implemented in context of the flow system to be most effective. Different land uses and / or levels of management on different parts of the watershed may be necessary to achieve target chemical concentrations within the groundwater body or target concentrations and / or loads in the stream.

When concerned with nitrate loads to and / or concentrations within the stream, the findings imply that one can inventory land use down to the scale of perhaps the second-order subwatershed, aggregate land use inputs of nitrate to groundwater within these areas, and transfer them directly to loads and / or average concentrations in the stream. A minor complication that must be considered is that groundwater divides are not necessarily coincident with topographic divides at this small scale. Land use within one small topographically defined watershed can possibly contribute to an adjacent stream. Potential for losses of $NO_3$ within the RZ must also be considered when using this approach. In general, though, altering the particular mix of land uses and / or levels of management on the subwatershed can be used to achieve target concentrations and / or loads.

As importantly, groundwater quality within these upland watersheds is affected locally and differentially by land use position. Recharge to the regional aquifer occurs only from the uppermost positions of the landscape, so control of nitrogen inputs in these areas is critical to limiting contamination of the regional aquifer and its long-term, low-level contributions to the stream. Control of inputs can be realized either by managing chemical inputs on the land use, or by restricting the types of land use allowed. Recharge resulting from land uses over the remainder of the watershed affects the deeper aquifer less because it tends to be short-circuited by the shallow fracture layer, flowing laterally to the stream. Management of land uses and / or associated chemical inputs in these lower landscape positions is critical only to the quality of the shallow groundwater downslope, but because of their relatively short travel times to the stream, these land uses may control the short-term cyclic entry of $NO_3$-N to the stream. Finally, contaminants introduced to the groundwater from these watershed positions nearer to the stream are also those more likely to be affected by the R2processes because their flowpaths approach the stream in more shallow positions of the cross section.

## SUMMARY

Where all components of the hydrologic cycle are intimately connected, as within upland watersheds of the Chesapeake Bay basin, land management and planning schemes must consider the influence of the groundwater flow system. Water quality objectives may not be achieved most effectively by chemical and / or land use management applied categorically over these watersheds. Rather, strategies should be developed in context of the interactions between land use, groundwater movement, and contaminant transport, and the component of streamflow produced by groundwater discharge. Subsurface watershed boundaries, effective depths, and areal extents of the bedrock layering, hydraulic conductivity, and porosity distributions, and locations of the dominant groundwater units of recharge, lateral flow, and discharge are needed to define these interactions.

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