Water Quality Functions of Riparian Forest Buffers in Chesapeake Bay Watersheds

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ABSTRACT / Maryland, Virginia, and Pennsylvania, USA, have agreed to reduce nutrient loadings to Chesapeake Bay by 40% by the year 2000. This requires control of nonpoint sources of nutrients, much of which comes from agriculture. Riparian forest buffer systems (RFBS) provide effective control of nonpoint source (NPS) pollution in some types of agricultural watersheds. Control of NPS pollution is dependent on the type of pollutant and the hydrologic connection between pollution sources, the RFBS, and the stream. Water quality improvements are most likely in areas of where most of the excess precipitation moves across, in, or near the root zone of the RFBS. In areas such as the Inner Coastal Plain and Piedmont watersheds with thin soils, RFBS should retain 50%–90% of the total loading of nitrate in shallow groundwater, sediment in surface runoff, and total N in both surface runoff and groundwater. Retention of phosphorus is generally much less. In regions with deeper soils and/or greater regional groundwater recharge (such as parts of the Piedmont and the Valley and Ridge), RFBS water quality improvements are probably much less. The expected levels of pollutant control by RFBS are identified for each of nine physiographic provinces of the Chesapeake Bay Watershed. Issues related to of establishment, sustainability, and management are also discussed.

Research is sometimes applied to broad-scale environmental issues with inadequate knowledge or incomplete understanding. Public policies to encourage or require landscape management techniques such as riparian (streamside) management will often need to proceed with best professional judgment decisions based on incomplete understanding.

Riparian forest buffer systems (RFBS) are streamside ecosystems managed for the enhancement of water quality through control of nonpoint source pollution (NPS) and protection of the stream environment. The use of riparian management zones is relatively well established as a best management practice (BMP) for water quality improvement in forestry practices (Comer-

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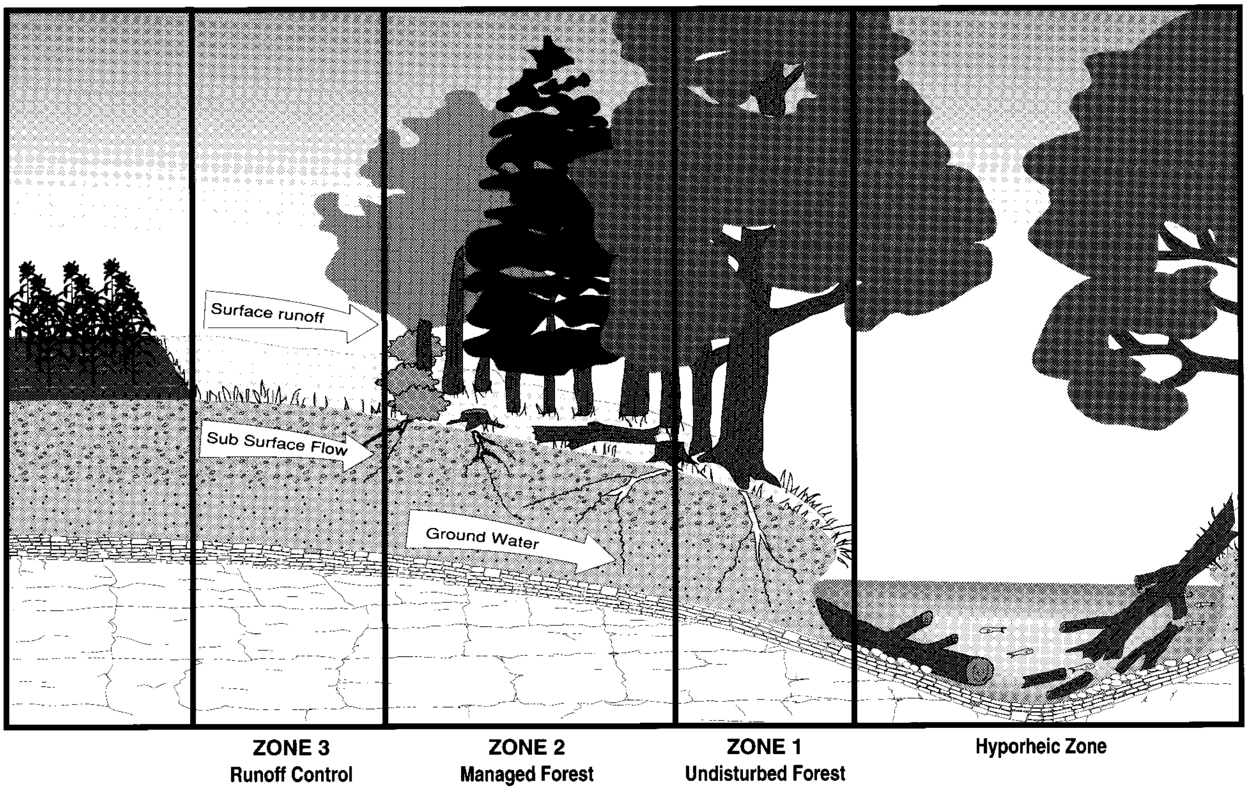


Figure 1. Schematic of the three-zone riparian forest buffer system.

ford and others 1992), but has been much less widely applied as a BMP in agricultural areas or in urban or suburban settings. Riparian ecosystems are especially important on small streams (first, second, and third order) which account for over three quarters of the total stream length in the United States (Leopold and others 1964). Riparian ecosystems are connected to aquatic ecosystems both by direct fluxes and, belowground, through the hyporheic zone (Triska 1993).

In 1991, the United States Department of Agriculture and other government and private agencies developed draft guidelines for riparian forest buffers. This effort resulted in a booklet entitled "Riparian Forest Buffers—Function and Design for Protection and Enhancement of Water Resources" (Welsch 1991), which specified a riparian buffer system consisting of three zones and the hyporheic zone (Figure 1). The hyporheic zone is the groundwater region where bidirectional flows between the stream and groundwater are common (Triska 1993). Zone 1 is permanent woody vegetation immediately adjacent to the stream bank. Zone 2 is managed forest occupying a strip upslope from zone 1. Zone 3 is a herbaceous filter strip upslope from zone 2. The specification applies to areas where cropland, grasslands, and/or pasture are adjacent to riparian areas on (1) permanent or intermittent streams, (2) margins of lakes and ponds, (3) margins of wetlands, or (4) margins of groundwater recharge areas such as sinkholes. The draft specification has been refined into a model state standard and general specifications by the USDA-Natural Resources Conservation Service (NRCS 1995).

Current understanding of the functions of the RFBS is based on studies that have been done in areas where riparian forests exist due to a combination of hydrology, soils, cultural practices, and economics. Knowledge of the functions of the three zones of the RFBS specification is derived from studies in existing riparian forests and on experimental and real-world grass buffer systems. Although results can be extrapolated from these existing systems to restored RFBS, most of the forest study sites are actually at some stage of restoration, following clearing within the last 20–80 years.

Compared to other NPS pollution control measures, RFBS can lead to longer-term changes in the structure and function of agricultural landscapes. To produce long-term improvements in water quality, RFBS must be designed with an understanding of: (1) the processes that remove or sequester pollutants entering the riparian buffer system; (2) the effects of riparian management practices on pollutants retention; (3) the effects of riparian forest buffers on aquatic ecosystems; (4) the

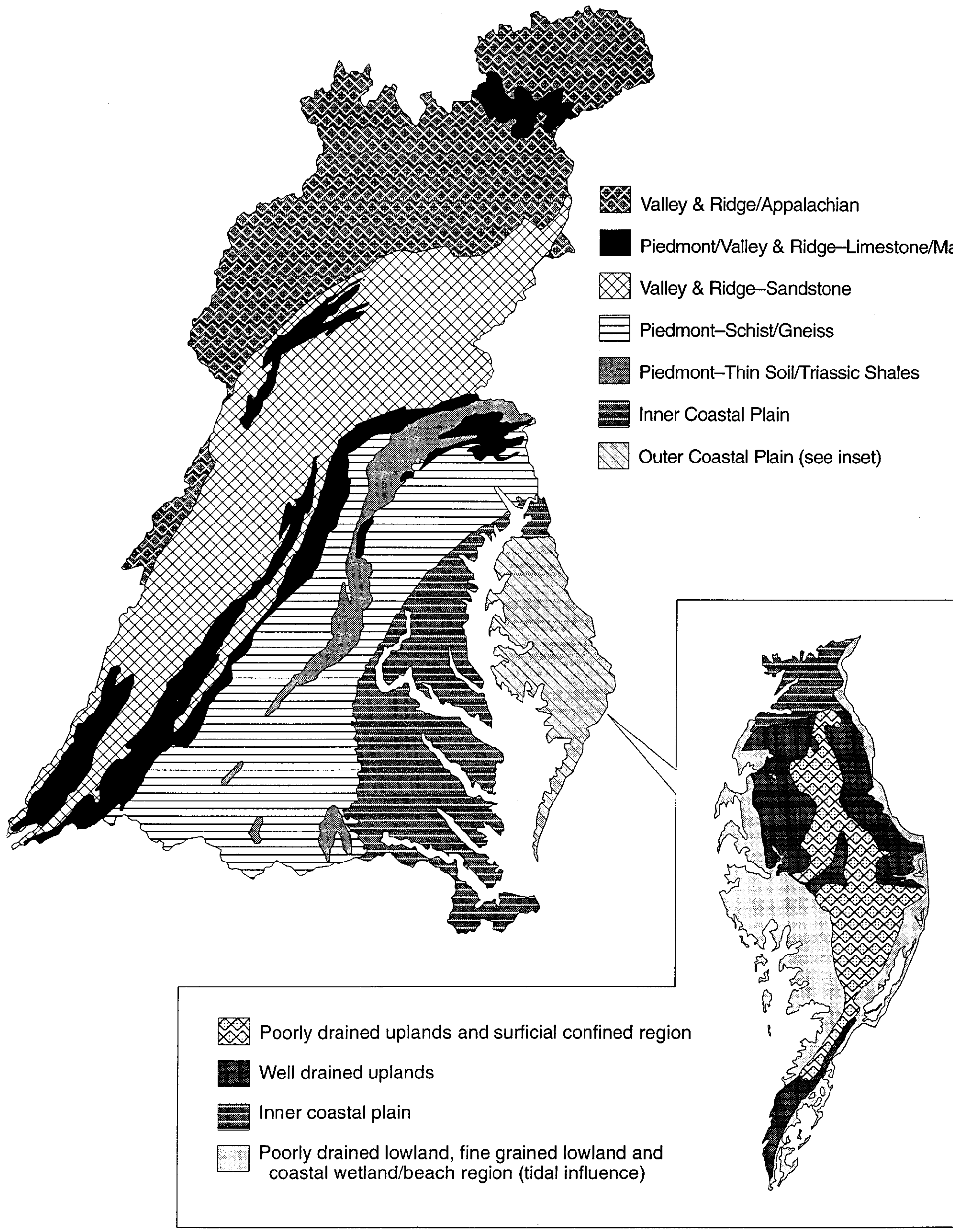


Figure 2. Physiographic regions of the Chesapeake Bay Watershed and hydrogeomorphic regions of the Delmarva Peninsula.

time to recovery after harvest of trees or reestablishment of riparian buffer systems; and (5) the effects of underlying soil and geologic materials on chemical, hydrological, and biological processes.

Recently, the Chesapeake Bay Program identified a need for guidance on the usefulness of riparian forest buffers for water quality improvement in the Chesapeake Bay Watersheds (CBW; Figure 2). These judgments were obtained by assembling a group of researchers with specialized knowledge to: (1) discuss the current state of knowledge of Riparian Forest Buffer Systems (RFBS); (2) determine how that knowledge related to the physiographic provinces of the CBW; and (3) reach consensus about the functions of RFBS in the CBW based on that current state of knowledge. Although there is seldom certainty about functions to be expected from riparian forest buffers in a given setting, we have reached general agreement about the validity of a set of consensus statements. In addition to the authors, a large number of reviewers were asked to examine the report and form their own judgements about the general conclusions. These reviewers, acknowl-

Table 1. Land use in physiographic regions of Chesapeake Bay Watershed (NCRI Chesapeake, 1982)

Physiographic region	Area (ha)					% of total
	Crop	Forest	Wetland	Other	Total	
Appalachian Plateau	659,700	2,611,100	181,400	658,800	4,111,000	28
Valley & Ridge	986,200	2,659,600	60,500	911,100	4,617,400	32
Piedmont	825,500	1,607,900	141,300	688,100	3,262,900	23
Coastal plain	768,600	1,020,400	509,300	119,800	2,418,000	17
Total	3,240,000	7,899,000	892,500	2,377,800	14,409,300	100
% of total	22	55	6	17	100	

edged below, generally agree with the consensus statements contained in this article.

The reviews of NPS pollution control by physiographic regions draw only upon studies done in the CBW or in similar physiographic regions in the eastern United States. For instance, information from the Piedmont and Coastal Plain of North Carolina and Georgia was used, but information from the glaciated regions of Iowa and Rhode Island was not used. In addition, we also attempted to use material from the refereed and peer-reviewed literature, to the extent possible. The exceptions to this occurred primarily when one of the authors felt that nonrefereed literature was both scientifically sound and necessary to understand riparian functions. To aid in understanding the hydrologic connections among riparian zones and adjacent land uses, a series of generalized hydrologic flow diagrams will be presented for the subdivisions of physiographic regions. The diagrams are not meant to be quantitative but provide a qualitative summary of important features of the flow paths affecting the potential for nonpoint source pollution control by RFBS. Based on these results, we provide a series of consensus statements and tables that summarize our best professional judgment of the potential effectiveness of RFBS for NPS pollution control in different parts of the CBW.

Atlantic Coastal Plain Province

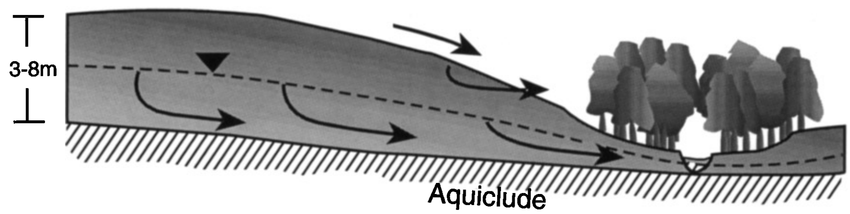
Land Use and Hydrology

The Coastal Plain (Figure 2) is an area of low topographic relief, relatively high moisture infiltration capacities, well-distributed rainfall throughout the year, and unconfined surficial aquifers. Streamflow is mainly derived from groundwater discharge from the surficial aquifer. The Coastal Plain has higher proportions of both cropland (32%) and wetland (21%) than any other physiographic province of the bay watershed (Table 1). Direct surface runoff in agricultural watersheds generally accounts for about 5%–15% of streamflow (Peterjohn and Correll 1984, Staver and others 1988). The remainder of the precipitation either infiltrates and is available for either groundwater recharge or evapotranspiration or goes directly into surface water as stream or detention storage. Although this general view of the Coastal Plain is useful, variations in soils, topography, subsurface stratigraphy, and land use within the Coastal Plain control the fate of NPS pollutants relative to RFBS.

The CBW Coastal Plain is often divided into Inner and Outer Coastal Plains. The Inner Coastal Plain (ICP) is mostly the western shore of Chesapeake Bay and the upper Eastern Shore. The Outer Coastal Plain (OCP) is primarily the lower Eastern Shore / Delmarva Peninsula. ICP areas have relatively high topographic relief compared to OCP systems and generally have finer textured, nutrient-rich soils compared to the nutrient-deficient, sandy soils of the OCP (Correll and others 1992). A more detailed classification of the Coastal Plain was developed by the US Geological Survey for the Delmarva Peninsula (Phillips and others 1993). This classification of hydrogeomorphic regions was based on analysis of geologic and geomorphic features, soils, drainage patterns, and land cover (Figure 2). The upland, nontidal area of the Delmarva was divided into ICP that closely correlates with the ICP of Correll and others (1992) and three OCP hydrogeomorphic regions: Well-Drained Upland (WDU), Poorly Drained Upland (PDU), and Surficial Confined (SC) region. Differences in the physical characteristics of these regions result in variations in the functions of RFBS within them. The following discussion presents the general hydrogeomorphic characteristics associated with each region.

Inner Coastal Plain. The ICP includes the portion of the Coastal Plain located on the western shore of Chesapeake Bay and the area immediately south of the fall line on the Delmarva Peninsula (Figure 2). Tidal sections of rivers extend far into the ICP, near the fall line in some cases. Watersheds in the ICP are characterized by well-drained soils on uplands with poorly drained

INNER COASTAL PLAIN



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	High, most water moves in or near root zone.	Bypass due to artificial subsurface drains. Organics in Zone 2.	Important on all streams. Rapid restoration of dentrification function. Ground cover in Zone 3.
Removal of sediment and sediment-borne pollutants	High/Medium	Convert concentrated flow to sheet flow.	Restore in all areas. Enhance existing forest with Zone 3 speaders.
Removal of dissolved phosphorus	Medium/Low	Control of dissolved P in surface runoff and groundwater is limited.	Restore in areas with major P load in surface runoff. Enhance existing forest with Zone 3.

Figure 3. Inner Coastal Plain—idealized flow system, expected level of RFBS function, constraints to achieving the function, and management factors critical to achieving the function.

soils limited to riparian zones. Land use is primarily agricultural on uplands and forested in riparian zones. Topography of this region is gently rolling with a high degree of stream incision.

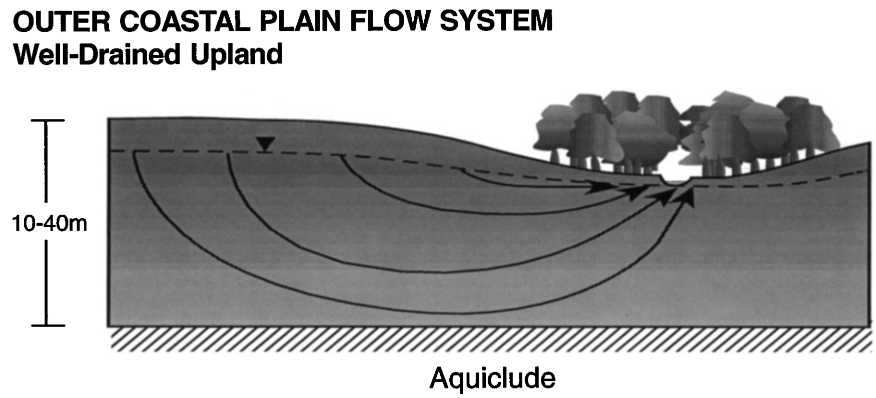
The ICP is a hydrologically complex region because sands and gravels that comprise the surficial aquifer are thin and overlie subcropping sands or finer-textured confining beds of older Coastal Plain aquifers. Stream valleys are commonly incised into the older units. As a result of this configuration, the surficial deposits do not form an extensive aquifer as they do in other parts of the Coastal Plain. Shallow groundwater flow systems in the surficial sediments commonly extend from topographic highs into the deeper aquifer, where they are close to the surface. If the surficial aquifer overlies a shallow confining bed, groundwater flow is restricted to shallow depths, where it comes into contact with riparian zone sediments and soils near aquifer discharge areas. The ICP flow system is represented by Figure 3.

The Rhode River Watershed along the western shore of Maryland is representative of the hydrologic conditions common to much of the ICP. This 2286-ha watershed is 62% forest, 23% croplands, 12% pasture, and 3% freshwater swamp (Jordan and others 1986). The watershed is underlain by a relatively impermeable clay layer that forms an effective aquiclude. Most groundwater flow to streams is in a shallow surficial aquifer (Correll 1983). The 160-yr average rainfall is 108 cm. For the Rhode River, slow streamflow (baseflow or groundwater discharge) averaged 29.6 cm of flow while quickflow (mostly stormflow or surface runoff from all contributing areas) accounted for 5.0 cm (Correll, unpublished, cited in Peterjohn and Correll 1984). Studies on Rhode River indicated that 86% of all watershed discharge comes from slow flow or groundwater discharge and 14% from direct surface runoff.

Well-Drained Upland. Watersheds in the WDU (Figure 2) are characterized by predominantly well-drained soils on uplands and poorly drained soils on floodplains in stream valleys. The topography is relatively flat to gently rolling with a high degree of stream incision (Phillips and others 1993). Most upland area is used for agricultural crop production with wooded areas generally confined to narrow riparian zones. Sediments of the surficial aquifers are primarily sand and gravel and range from about 6 to 12 m in the north to 24 to over 30 m thick in the south (Owens and Denny 1979). The aquifer is unconfined, and the depth to water ranges from 3 to 10 m beneath topographic highs, to land surface in surface-water discharge areas.

Groundwater flow paths range from about 1 km to several kilometers in length in the well-drained upland (Shedlock and others 1993). The longest, oldest flow paths originate at topographic highs, extend to the base of the aquifer, and discharge to second- and third-order streams through the hyporheic zone (beneath the stream channel). The water in these longer flow paths is generally less than 50 years old near aquifer discharge areas (Dunkle and others 1993). Shorter, younger flow paths originate in near-stream recharge areas and are the main source of baseflow to first-order streams. The well-drained upland flow system is represented in Figure 4.

Poorly Drained Upland/Surficial Confined. Watersheds in the PDU (Figure 2) are characterized by interspersion of poorly drained forests and moderately well-drained and well-drained agricultural land (Shedlock and others 1993). In the northern part, the region has



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Low, primarily removal from shorter flow paths.	Bypass flow due to deeper aquifers. Long flow paths surface in stream channels.	Concentration on headwater areas. Zone 1 important for nitrate removal.
Removal of sediment and sediment-borne pollutants	High/Medium	Concentrated flow must be converted to sheet flow.	On larger streams, focus on filtering eroded sediment. Enhance functions of Zones 2 & 3.
Removal of dissolved phosphorus	Medium/Low	Dissolved P control is limited. Focus on P load in surface runoff.	Increase vegetation uptake and accretion. Enhance existing forest and grass strips.

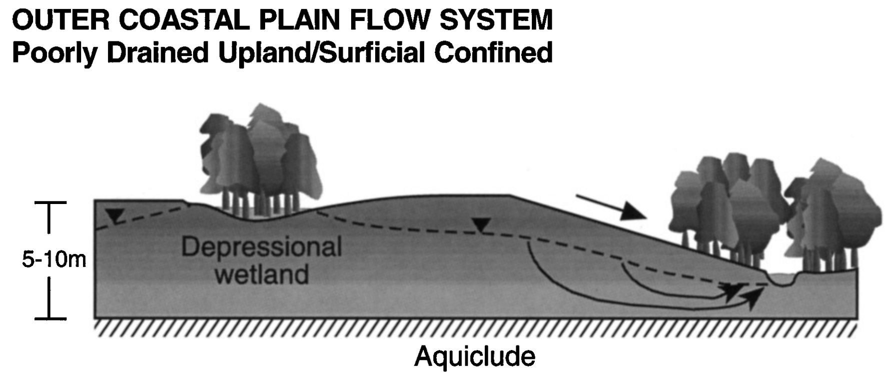
Figure 4. Outer Coastal Plain (Well-Drained Upland)—idealized flow system, expected level of RFBS function, constraints to achieving the function, and management factors critical to achieving the function.

hummocky topography and low relief with many seasonally ponded wooded depressions. In the southern part, topography is relatively flat, with broad poorly drained forested areas that are seasonally flooded. Streams are small and sluggish in the poorly drained upland and flow through shallowly incised valleys with low gradients (Phillips and others 1993). Riparian zones are usually forested and often contain wetlands. Some parts of the PDU have been ditched to promote drainage of agricultural fields.

Sediments that make up the surficial aquifer in the PDU are predominantly sands and gravels, similar to those in the well-drained upland. The sediments range in thickness from about 8 m in the north to more than 30 m in the south (Owens and Denny 1979). The water table is usually within 3 m of the land surface. This region is characterized as poorly drained because of the combination of regionally high water table and small degree of stream incision that results in groundwater gradients too low to effectively drain the region, rather than a low permeability substrate (Phillips and others 1993). Except for areas immediately adjacent to streams, groundwater flow paths in the PDU range from about 100 m to about 1 km in the northern part of the region where the aquifer is thin. In the southern part, where the aquifer is thick, flow paths are up to several kilometers in length and generally originate near the regional drainage divide. The generalized flow system for the poorly drained upland/surficial confined is shown in Figure 5.

Watersheds in the SC region (Figure 2) are geomorphologically similar to the southern part of the PDU with low relief and shallow incision of stream valleys, features that contribute to the poor drainage in the region. Topographically, the area is a flat sandy plain with low ridges that rise a few meters above the surrounding landscape. The plain is dominated by poorly drained soils and the ridges are dominated by well-drained soils. Throughout the region, large tracts of forest are interspersed with agricultural fields on the plains; there are broad forested riparian zones and swamps around the major drainageways. With the exception of the sandy dune ridges, agricultural land is heavily ditched to promote soil drainage and would probably be forested wetlands in the absence of ditching (Phillips and others 1993).

The surficial aquifer is geologically heterogeneous in the region, although it is generally confined. A major sand unit 25–30 m thick is overlain by 0–13 m of complexly layered clay, slit, and peat, which is itself overlain by 1–6 m of wind-deposited sand with some peaty sand, silt, and clay lenses at the base (Owens and Denny 1979). The complex of fine-grained deposits acts as a confining unit between the sands of the upper and lower units, except some areas where it is absent or entirely composed of sand. The water table is generally less than 3 m below land surface and occurs in the upper sand unit. Local groundwater flow paths in the upper unit are relatively shallow, generally less than 300 m long, and extend from water-table highs in interfluves between ditches and streams into the ditches and streams. Regional groundwater flow paths in the lower



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Medium/High	Lower loadings. Lower rates of removal in head-water areas.	Restore first in headwaters then larger streams. Rapid restoration of dentrification function.
Removal of sediment and sediment-borne pollutants	High/Medium	Less surface runoff but similiar efficiencies as in other CP systems.	Enhance vegetation in broad existing areas. Restore in headwaters.
Removal of dissolved phosphorus	Medium/Low	Dissolved P control is limited. Focus on P load in surface runoff.	Increase vegetation uptake and accretion. Enhance existing forest and grass strips.

Figure 5. Outer Coastal Plain (Poorly Drained Upland / Surficial Confined)—idealized flow system, expected level of RFBS function, constraints to achieving the function, and management factors critical to achieving the function.

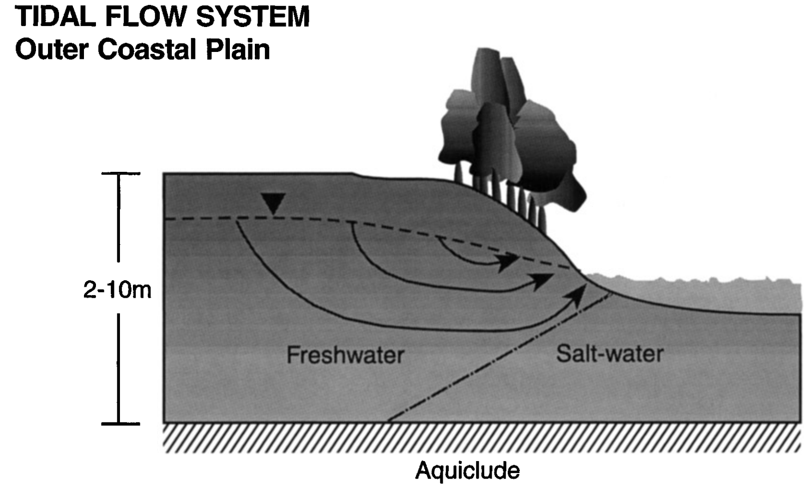
units are up to 10 km long and extend from the uplands near the regional drainage divide to major streams and rivers. Local and regional flow paths are separated in most areas by the confining layer, but local heads are higher than regional heads in most places, and shallow flow paths extend into the lower sand where confining beds are absent (Shedlock and others 1993). Residence time in the upper sand is 15 years or less; in the deeper unit, groundwater residence time is at least 40–50 years, except where there is hydraulic connection with the shallow unit (Dunkle and others 1993).

Tidal Areas. Tidally influenced areas of the Coastal Plain present unique situations for a number of reasons. First, water and pollutants moving through the terrestrial/aquatic interface move directly into the bay or tidal reaches of streams, providing a direct input of pollutants. Second, movements of groundwater in these tidal systems are affected by tidal movements of bay water that serve to restrict discharge from freshwater aquifers. Third, two main types of terrestrial–aquatic interfaces appear to exist, especially for groundwater fluxes. One case is a tidal stream, embayment, or main channel location where a marsh system forms a buffer at the terrestrial–aquatic interface. In areas with marsh, the nitrate removal function of the RFBS is less significant due to groundwater discharge through the marsh being stripped of nitrate in anaerobic marsh sediments. The second case is when the interface does not include the marsh system and discharge takes place from a sand aquifer directly into the bay or tidal stream. The flow system for the second case is the one that is shown in Figure 6.

Nutrient Budgets for Coastal Plain Riparian Forests

The most direct means of determining the NPS pollution control function of a riparian forest is to develop annual or longer-term mass balances. Developing nutrient or sediment budgets requires a watershed from which hydrologic measurements can be made that assure all watershed outputs are measured and sampled. If the riparian forest buffer is continuous around the entire stream system and groundwater discharging to streams moves through riparian soils and shallow sediments, the streamflow output can be treated as the output from the riparian forest system. The inputs to the riparian system must be estimated from sampling of groundwater and surface water inputs. The studies that have done this for Coastal Plain riparian forests are summarized in Table 2. Total N and total P retention have been estimated in studies of Watershed 109 (WS-109) of the Rhode River, Maryland, and the Heard Creek tributary of Little River, Georgia. Both of these Coastal Plain systems have effective aquicludes at depths that limit recharge to deep groundwater and that cause all or nearly all excess precipitation to move through riparian systems and exit the watershed as streamflow.

Estimates of N retention were 89% of input (Rhode River), and 66% of input (Little River). P retention in Rhode River was slightly less (80% of input) but much less in Little River (24% of input). Total N and P budgets for Little River did not include surface runoff inputs of N and P from the agricultural areas to the riparian forest but did include all streamflow outputs of N and P (Table 2). Streamflow includes surface runoff



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Low/Medium	Depth to water-tables. Bank erosion due to unstable soils.	Limit practice to areas without marsh wetlands down slope. Enhance vegetation uptake.
Removal of sediment and sediment-borne pollutants	High/Medium	Convert concentrated flow to sheet flow. Bank stability limits usefulness in some areas.	Restore/enhance in all areas. Limit to wider Zone 3 in some areas. Enhance Zone 3.
Removal of dissolved phosphorus	Medium/Low	Dissolved P control is limited. Focus on P load in surface runoff.	Increase vegetation uptake and accretion. Enhance existing forest and grass strips.

Figure 6. Coastal Plain (tidal areas)—idealized flow system, expected level of RFBS function, constraints to achieving the function and management factors critical to achieving the function.

Table 2. Total nitrogen, nitrate-nitrogen, and total phosphorus budgets for riparian forest ecosystems in the Coastal Plain

		Amount (kg/ha/y)			
Reference	Location	Input	Output	Retention[a]	Flux notes[b]
Total N					
Peterjohn and Correll (1984)	Rhode River, MD	83	9	74	NO_3, NH_4, Org-N in SRO, GW, P, PSF, PQF
Lowrance and others (1984)	Little River, GA	39	13	26	NO_3, NH_4, Org-N in GW, P, SF
Nitrate-N					
Correll and Weller (1989)	Rhode River, MD	45	6.4	38.6	NO_3 in GW, SF (baseflow only)
Lowrance and others (1984)	Little River, GA	22	2.1	19.9	NO_3 in GW, SF
Cooper and others (1986)	Beaverdam Creek, NC	35	5.1	29.9	NO_3 in GW, SRO, SF
Total-P					
Peterjohn and Correll (1984)	Rhode River, MD	3.6	0.7	2.9	Total P in SRO, GW, P, PSF, PQF
Lowrance and others (1984)	Little River, GA	5.1	3.9	1.2	Total P in GW, P, SF

[a]Retention = input − output.

[b]SRO = surface runoff input; GW = groundwater input; P = precipitation input; SF = streamflow output; PSF = partitioned slow flow; PQF = partitioned quick flow.

that moved through the riparian forest and contributed to stormflow. Therefore, the N and P retention (input − output) estimates for the Little River site are underestimates of the actual retention. Peterjohn and Correll (1984) included direct estimates of both surface runoff and groundwater inputs and outputs for Rhode River. Budget estimates were based on these direct measurements rather than streamflow outputs. Streamflow outputs for Rhode River were different than the riparian budget output for both total N and P. This difference has only a negligible effect on the total N budget, but has a large effect on the total P budget. If streamflow outputs are considered the output from the riparian forest for the Rhode River site, the total N retention is still 83% of inputs, but P retention is zero.

The Little River and Rhode River studies were both done in systems that are likely to maximize retention by natural riparian forests. Although the studies report

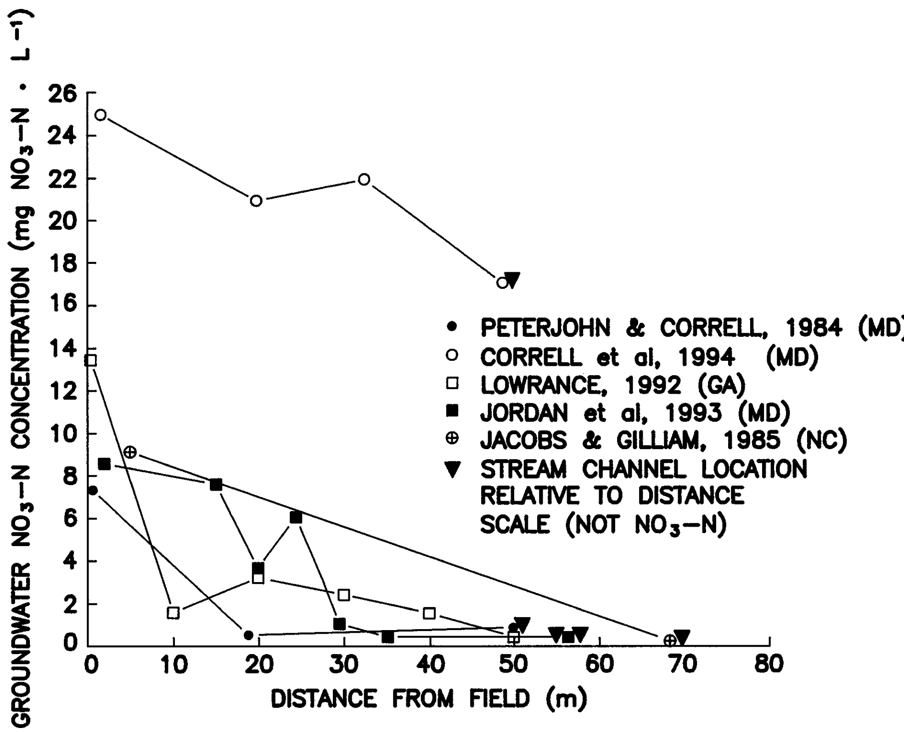


Figure 7. Nitrate concentrations in groundwater beneath riparian forests from five Coastal Plain sites.

different ranges of percent retention for N and P, retention of N was generally high. Both watersheds have percentages of agricultural land typical for the more agricultural portions of the Coastal Plain and are representative of potential inputs to riparian systems in the absence of animal confinement facilities and manure disposal systems. These natural riparian systems would appear to retain at least two thirds of the N inputs but perhaps as little as one third of the P input.

In both the Rhode River and Little River studies, nitrate in subsurface flow made up the majority of total inputs to the riparian forest system. The input in groundwater for WS-109 of Rhode River in the year reported on in Peterjohn and Correll (1984) was 57 kg NO_3-N/ha/yr based on the area of riparian forest. This accounted for 69% of the total N input (Table 2). Based on two more years of data for WS-109 of Rhode River, the input averaged 45 kg NO_3-N/ha/yr (Correll and Weller 1989). Data from Little River showed that groundwater input was 22 kg NO_3-N/ha/yr, 56% of total N input. A third study of nitrate budgets (Cooper and others 1987) on a Coastal Plain watershed in North Carolina showed similar results to the Maryland and Georgia studies. Nitrate retention rates of 85%, 86%, and 90% for the three studies (North Carolina, Maryland, Georgia, respectively) reflect removal of nitrate through both denitrification and plant uptake. Plant uptake (and perhaps microbial immobilization) contribute to transformation of a predominantly nitrate input to the riparian zone into a predominantly organic N output in streamflow. Total N input to the riparian forest on Rhode River was 69% nitrate. Streamflow was 51% organic N (Correll and others 1992, Correll 1983). On the Little River, groundwater inputs to the riparian forest were 74% nitrate. Streamflow outputs were 18% nitrate and 80% organic N. A later study of the entire Little River watershed showed consistent trends of nitrate increase during stormflow, indicating that the nitrate removal/transformation capacity of riparian forests is partially by passed when water moves through more quickly during high flows (Lowrance and Leonard 1988).

Nitrate Transport in Shallow Groundwater

Although elemental, nutrient, chemical, and sediment budgets on a watershed scale are the most complete way to evaluate the functions of riparian forest buffers and offer the best information on potential load reductions, a number of studies have examined nitrate concentration changes in riparian forests (Figure 7). Studies in four separate Coastal Plain locations have shown that average annual edge-of-field nitrate levels of 7–14 mg NO_3-N/liter decreased to 1 mg/liter or less in shallow groundwater near streams (Peterjohn and Correll 1986, Jordan and others 1993, Correll and others 1994, Lowrance 1992). Decreases in chloride concentrations were generally small compared to nitrate decreases, indicating biological removal of nitrate. Most

Table 3. Aboveground woody vegetation uptake of N and P in Coastal Plain riparian forests

		Amount (kg/ha/yr)			
		Nitrogen		Phosphorus	
Reference	Location	Total uptake	Woody storage	Total uptake	Woody storage
Correll and Weller (1989)	Rhode River, MD	ND[a]	12–20	ND	3–5
Peterjohn and Correll (1984)	Rhode River, MD	77	12	10	1.7
Fail and others (1986, 1987) (mean)	Little River, GA	84	22	7.5	3.8

[a]ND = not determined.

studies of nitrate dynamics in riparian forests have shown that removal of nitrate from groundwater continued year-round. Mechanisms to explain this have not been elucidated, although it is likely that in some of the southeastern Coastal Plain areas, relatively warm soils and evergreen or tardily deciduous (broad-leaf trees that lose leaves in the spring) vegetation can provide biological removal of the nitrate. Winter root growth by deciduous or evergreen trees may also be a factor. Weil and others (1990) observed year-round reductions of groundwater nitrate in streamside forests on tributaries of the Choptank River on the Eastern Shore of Maryland. Groundwater under riparian forests always had less than 1 mg NO_3-N/liter while adjacent fields had concentrations of 15–40 mg NO_3N/liter. The decreases in chloride concentrations were much less than the nitrate decreases. Year-round nitrate removal has been observed, but not explained.

At least one study has shown that in situations with relatively high nitrate concentration entering from an adjacent field, substantial nitrate concentration reductions can occur but still leave high concentrations in shallow groundwater at the stream (Correll and others 1994) (Figure 7). This site, on a tributary of the Choptank River on the Delmarva Peninsula, is located in the Well-Drained Uplands. Nitrate concentration reductions were actually higher at this site than at two other Coastal Plain sites in Maryland (Peterjohn and Correll 1984, Jordan and others 1993), but groundwater concentrations near the stream were 12–18 mg NO_3N/liter. Similar results were inferred from a study of nitrate in regional groundwater and nitrate levels in streamflow for the WDU hydrogeomorphic region (Phillips and others 1993). In related work, Bohlke and Denver (1995) concluded that a riparian forest wetland next to a stream in the WDU had little effect on nitrate movement to the stream. Hydrologic data and groundwater flow modeling showed that groundwater discharges upward directly to the streambed from the aquifer system, effectively bypassing the riparian zone (Reilly and others 1994). Stream baseflow concentrations commonly exceeded 9 mg NO_3-N/liter.

Nitrate transport into tidal streams is often dominated by direct recharge through sediments in intertidal zones (Reay and others 1992, Simmons and others 1992, Staver and Brinsfield 1994). Approximately 80 kg/ha/yr of NO_3-N was discharged to a tidal creek in Maryland with apparently most groundwater moving at least 2 m below the ground surface in near-stream areas (Staver and Brinsfield 1994). These situations may allow little chance for nitrate removal, although the direct discharge to tidal streams makes riparian buffers desirable (Simmons and others 1992).

Vegetation Uptake and Denitrification

Direct estimates of N and P uptake by vegetation and storage of N and P in woody biomass have been made for riparian forests in Georgia and Maryland. Estimates from Watershed 109 of Rhode River (Peterjohn and Correll 1984, Correll and Weller 1989) indicated that total vegetation uptake of N and P was 77 and 10 kg/ha/yr, respectively. N and P storage in woody biomass was less than total uptake (Table 3). Extensive data on total N and P uptake and woody storage were reported by Fail and others (1986, 1987). Values for uptake and storage are similar for the Little River and Rhode River studies (Table 3). Fail and others (1986, 1987) reported mean storage of N in wood as 22 kg N/ha/yr (range was from about 15 to 42 kg N/ha/yr).

Denitrification has been shown to be an important N removal process in Coastal Plain riparian forests either: (1) through indirect measurement using the acetylene inhibition technique; (2) through measurement of environmental conditions that control denitrification (Eh, water-filled pore space, N and C availability) and verifying that proper environmental conditions exist; or (3) through measurement of denitrification potential (Ambus and Lowrance 1991, Lowrance and others 1984, Hendrickson 1981, Jacobs and Gilliam 1985, Correll and others 1994, Jordan and others 1993, Lowrance 1992). The general conclusion of all these studies was that denitrification occurred in most riparian forest soils, especially in the root zone, or that conditions were favorable for denitrification. Recent work by Bohlke and Denver (1995) indicated that denitrification can also occur in aquifer sediments below the influence of the riparian root zone where proper geochemical environments exist. Isotopic analy-

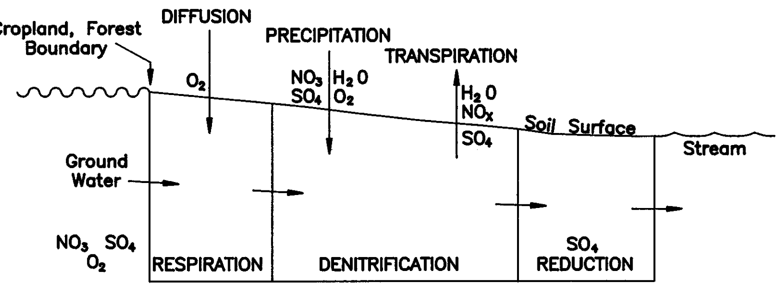


Figure 8. Conceptual model of belowground processes affecting groundwater nutrients in riparian forest.

sis of groundwater and streamflow at this site indicated that denitrification was not significantly reducing nitrate concentrations (Bohlke and Denver 1995).

Denitrification was measured in riparian forests of Little River, Georgia, in conjunction with water quality and hydrologic measurements (Hendrickson 1981). A total of 1114 soil cores (0–10 cm) were taken monthly for a year from six riparian forest sites on the Heard Creek tributary of Little River. Estimated denitrification rates ranged from 31 kg N/ha/yr for the top 50 cm of soil for the entire riparian zone of the watershed to 295 kg N/ha/yr under conditions of high N and C subsidy from a swine operation (Hendrickson 1981, Lowrance and others 1984). Lowest denitrification rates (1.4 kg N/ha/yr) were measured in a riparian zone adjacent to an old field that received no fertilizer application. Active cores (those producing N_2O above background levels) ranged from 11% to 66% of the cores taken, depending on the site. Soil cores taken to 50 cm in 10-cm increments showed that, except near the stream channel, denitrification activity below 20 cm depth was much lower than activity in the top 20 cm (Hendrickson 1981). Later studies from Little River have also shown that denitrification potentials at the top of the water table are measurable, but less than 1% of the rate in surface soils (Lowrance 1992). A restored RFBS had denitrification averaging 68 kg N/ha/yr due to a relict forested wetland soil and movement of high nitrate water in the root zone (Lowrance and others 1995).

The interaction of vegetation nitrogen uptake, organic carbon production via litterfall and root senescence, and microbial denitrification appear to be driving nitrate removal in most Coastal Plain riparian forests. Belowground processes affect nutrients largely by controlled oxidation–reduction conditions (Figure 8) (Correll and Weller 1989). Organic matter from decomposing litter and roots serves as an energy source, and oxygen is consumed through aerobic respiration, followed in sequence by nitrate reduction and then sulfate reduction when conditions become sufficiently reduced. In the presence of carbon-rich sediments or relict organic matter horizons, these processes could potentially proceed without forest vegetation. Stratified denitrification potential in riparian forests of Little River, Georgia, indicated that denitrification coincided with the stratification of N and C from litter and roots (Lowrance 1992). Nitrate removal in Coastal Plain RFBS seems to be dependent on interactions in the forest ecosystem rather than just a poorly drained soil adjacent to a stream. Nitrate removal in all Coastal Plain forest sites was due to these complex interactions of vegetation and the belowground environment. Groundwater containing nitrate must pass through or near the root zone for this mechanism to operate effectively. Although most of the Coastal Plain studies of nitrate removal were in areas with relatively flat wetland soils near the stream, removal often took place in areas immediately downslope from the fields on better drained soils.

Removal of Sediments and Nutrients in Surface Runoff

Coastal Plain studies have shown that removal of nutrients and sediment from surface runoff can take place in both zone 3 and zone 2. Sediment and nutrient deposition from surface runoff moving through a Coastal Plain riparian forest has been estimated from direct sampling of surface runoff in the Rhode River watershed (Peterjohn and Correll 1984). Estimates of sediment deposition have been made based on soil morphology and ^{137}Cs profiles in Little River, Georgia, and in Cypress Creek, North Carolina. Grass vegetated filter strips (GVFS) have been widely studied, with at least one detailed study of fescue buffers in the Coastal Plain of Maryland (Magette and others 1989).

The estimated range of sediment deposition rates in riparian forests is large and apparently somewhat dependent on estimation technique (Table 4). Although the different methods give widely divergent numbers, in all cases sediment deposition accounted for 80%–90% of gross erosion from the uplands. Relatively low overall deposition rates (4.2 Mg/ha/yr) reported from direct sampling were associated with 90% reductions in sediment concentration in 19 m of flow through a riparian forest (Peterjohn and Correll 1984). Sediment deposi-

Table 4. Sediment deposition in Coastal Plain riparian forests

Reference	Location	Sediment deposition (Mg/ha/yr)	Notes
Peterjohn and Correll (1984)	Rhode River, MD	4.2	Annual measurements, first order stream, runoff samples
Cooper and others (1987)	Cypress Creek, NC	105–315[a]	^{137}Cs measurements—forest edge
Cooper and others (1987)	Cypress Creek, NC	35–105[a]	^{137}Cs measurements—ephemeral and Intermittent streams
Cooper and others (1987)	Cypress Creek, NC	0–35[a]	^{137}Cs measurements—floodplain swamp
Lowrance and others (1986)	Little River, GA	35–52	Watershed based, long term, sediment delivery ratio, soil morphology
Lowrance and others (1988)	Little River, GA	256–262	Single field/forest system ^{137}Cs measurements

[a]Based on sediment depths in Cooper and others (1987) and assumed bulk density of 1.4 g/cm^3.

tion estimates need to be compared to the gross erosion rates from cropland with information on the contributing area–riparian area ratio. With a field–forest ratio of approximately 2:1, the riparian forest would attenuate cropland erosion rates of about 2.1 Mg/ha/yr (Peterjohn and Correll 1984). This is well below the tolerance value for the upland soils, and many fields would contribute higher sediment loads from erosion. In contrast, a sediment deposition rate of 35 Mg/ha/yr at a 2:1 field to forest ratio would attenuate erosion from cropland contributing up to 17 Mg/ha/yr. Very high sediment deposition rates (up to 315 Mg/ha/yr) reported from ^{137}Cs distribution studies (Table 4) were due to high deposition at field edge. This deposition was mostly coarse material and did not contain large amounts of adsorbed nutrients.

When sediment loading is a concern, the RFBS would generally include a grass strip. Sediment removal by GVFS in the Coastal Plain is very effective for sheet flow in relatively short distances. If concentrated flow occurs across the GVFS, sediment removal is much less efficient. The GVFS also become less effective when multiple rainfall events take place in a few days or when sediment begins to accumulate and form berms, which can lead to channelized flow (Magette and others 1989). Field evaluations of GVFS indicated that they were more effective in Coastal Plain areas of Virginia than in steeper topography (Dillaha and others 1989b). Slopes in Coastal Plain areas were more uniform and field reconnaissance indicated that significant portions of stormwater runoff entered the GVFS as shallow uniform flow. These GVFS needed regular maintenance (sediment removal and possible revegetation every one to three years) because of the amounts of sediment deposition (Dillaha and others 1989b).

Nutrient removal from surface runoff has received very limited study. The 4.6-m filter strips used by Magette and others (1989), in the Maryland Coastal Plain generally did not remove total N from surface runoff and removed only 27% of the total P load. The 9.2-m filter strips had total N and P removals of nearly 50%. Peterjohn and Correll (1984) reported concentration reductions of 74% for total N and 70% for total P in flow through 19 m of mature riparian forest in Watershed 109 of Rhode River, Maryland.

Application of RFBS in Coastal Plain

The ICP (Figure 3) represents one end of the spectrum of riparian ecosystem function for removal of NPS pollutants. ICP areas typically have a high density of stream channels, well-developed ''natural'' riparian forests, and extensive connections between agricultural fields and riparian forest ecosystems. Because of the relatively large amount of scientific data collected from ICP type systems, primarily in Maryland, North Carolina, and Georgia, the scientific panel was able to make the most comprehensive consensus BPJ for these areas. These conclusions were:

1. Based on mass balances, established RFBS remove 85%–90% of nitrate input in shallow groundwater. For the systems studies, this was 20–39 kg NO_3N/ha/yr.
2. Based on mass balances, total N retention in established systems ranged from 67% to 89% of N input. For the systems studied, this was 26–74 kg N/ha/yr.
3. For the RFBS to be applicable in systems with artificial drainage near streams, the drainage system will have to be modified to work in conjunction with the RFBS.
4. Newly established systems are likely to have a substantial effect on subsurface nitrate loads in (at most) 5–10 years if anoxic sediments and high organic matter surface soils are already in place. By 15–20 years, reestablished RFBS should control groundwater nitrate loads in most ICP situations.

5. The nitrate concentration data from ICP systems indicates that higher nitrate loadings could be removed in the RFBS if it was exposed to higher loadings than represented in the mass balance studies. This is most likely to be true in systems with highest denitrification rates or potentials.
6. Based on the mass balances, net retention of phosphorus in established systems was 24%–81% of input. For the systems studied, this was 1.2–2.9 kg P/ha/yr. Retention of phosphorus in surface runoff appears to be mainly through retention of particulate phosphorus and infiltration in the RFBS. Retention of dissolved ortho-P appears to be considerably less effective for both surface runoff and subsurface flow.
7. For a contributing area–RFBS area ratio of about 2:1, the range of sediment and sediment-borne N and P reductions that could be expected under worst-case conditions is about 96% for sediment, 75% for total N, and 77% for total P. Most other cases—with a 2:1 area ratio and better upland conservation practices—would be expected to have lower concentrations leaving the RFBS. These numbers are based on the assumption of nonchannelized flow through the RFBS.

Because of the lack of quantitative information on RFBS functions in other hydrologic/physiographic/transpiration settings, the more detailed information from ICP settings will be used to guide quantitative estimates for the other settings. The consensus of the scientific panel was that the ICP data represented an upper limit on the functions of essentially unmanaged RFBS. Numerous management options and management factors discussed below could lead to increases in the effectiveness and sustainability of NPS pollution control functions, but in general practice, without depending on the management improvements, the effects of RFBS in the ICP would be representative of other systems in the CBW where essentially 100% of excess precipitation moves through an unmanaged RFBS. Where less than 100% of excess precipitation moves through the RFBS, the NPS pollution control effects would be proportionally less.

Because much of the groundwater flow reaches the stream channel through the hyporheic zone, interactions with biologically active soil layers appear to be limited in the Well-Drained Upland (Figure 4). The WDU represents the other end of the spectrum from the ICP. Processing of groundwater-borne pollutants, including nitrate, would be least in the WDU. Based on the present knowledge of these systems, RFBS in the WDU would remove some nitrate from groundwater moving in short, shallow flow paths. This removal function might be enhanced by vegetation management, especially in the zone 1 area, where tree roots could access groundwater discharge. Consensus decisions for the WDU were:

1. Where hydrologic connections between groundwater and biologically active soil layers are made, RFBS in the WDU should have about the same capacity for nitrate removal as in the ICP areas.
2. The zone 1 vegetation (adjacent to the stream channel) is very important because of potential access to water and pollutants in the hyporheic zone. Zone 1 vegetation should be managed for N uptake and for formation of high organic matter surface soils. Provision of leaf litter and other organic matter to the stream channels may increase denitrification in the channel and hyporheic zone.
3. RFBS in the WDU portion of the Coastal Plain would have about the same capacity to filter sediment and sediment-borne pollutants from surface runoff as RFBS in the ICP.
4. RFBS in the WDU may have a higher capacity for removing dissolved chemicals from surface runoff because of higher available storage for infiltrated surface runoff. This function is directly related to lower water tables in the RFBS.
5. Reestablishment of RFBS in the WDU should focus on headwater streams, many of which have been ditched. Enhancement of existing forests along both small and large streams should focus on control of surface runoff and surface-borne pollutants and on management of zone 1 to intercept nitrate enriched groundwater.

The Surficial Confined and Poorly Drained Upland (Figure 5) are intermediate between the WDU and the ICP. Specifically, the consensus BPJ on these regions were:

1. Potential for nitrate removal is intermediate between WDU and ICP. Generally lower regional groundwater concentrations of nitrate will lead to lower actual removal rates and to less important role for nitrate removal.
2. Agriculture in these regions is commonly associated with artificial drainage, which will need to be integrated with RFBS system.
3. Potential for control of sediment and sediment-borne chemicals should be similar to RFBS in the ICP, but actual removal is probably less because of lower loads of surface-borne pollutants.

4. Potential for control of dissolved chemicals in surface runoff may be less than in WDU because of higher water tables and generally less available storage.

RFBS in tidal areas (Figure 6) are dependent on two factors: depth to water table and bank stability. The interaction of water-table depth and nitrate removal via denitrification has been discussed extensively in previous sections. Bank stability is a major factor in tidally influenced areas because of wave action, boat wakes, storms, and rising sea level undermining trees at the water's edge. It is possible that in tidal areas with eroding shorelines, trees in a zone 1 position will contribute to localized erosion and destabilization.

The consensus BPJ on tidal areas of the Coastal Plain were:

1. In areas without a tidal marsh at the terrestrial–estuarine interface, nitrate removal should be significant if the water table is within or near the root zone of trees in zone 2. This removal would be both through direct vegetation uptake and through coupling of vegetation uptake/denitrification in surface soil. Where the water table is consistently below the root zone, significant nitrate reduction is unlikely to occur.
2. In areas where shoreline erosion is a problem or potential is high, zone 1 trees at the water's edge are likely to contribute to shoreline erosion due to undermining of trees and tree fall into tidal waters. If established in these situations, zone 1 trees need to be put in a position that is not likely to contribute to active erosion, cliff destabilization, or shading of marshes.
3. Functions of zone 3 for sediment and some nutrient removal should be similar to function in ICP systems.
4. Restoration of RFBS in tidal areas should concentrate on areas with shallow water tables, an absence of tidal wetlands, limited shoreline erosion problems, and in areas with substantial surface runoff into tidal waters from adjacent land uses.

Piedmont Province

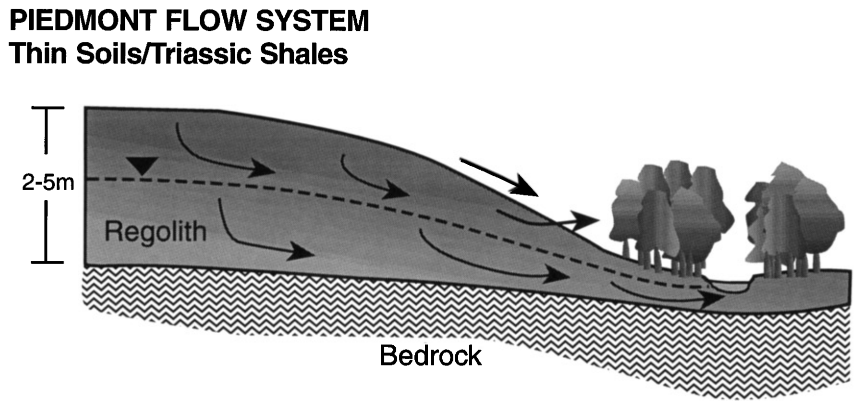
Land Use and Hydrology

The Piedmont Province is an upland region lying between the Coastal Plain and the Valley & Ridge provinces at elevations ranging from 30 to 300 m (Figure 2). The Piedmont accounts for 23% of the Chesapeake Drainage or 32,600 km^2 (Table 1) (NCRI Chesapeake 1982). Of this area, 49% is in woodland, 25% is used as cropland, 4% is wetland, and 21% is in other uses including pastures and suburban and urban land uses (NCRI Chesapeake 1982). Of the total cropland within the Chesapeake drainage, 25% lies within the Piedmont.

The Piedmont is underlain primarily by metamorphic Precambrian and early Paleozoic rocks subject to several episodes of folding. The majority of Piedmont basement materials are quartzites, gneisses, schists, and marbles. These rocks were metamorphosed from ancient sandstones, gabbros and granites, shales, and limestones, respectively. In Pennsylvania and Maryland, the marble belts form valleys; the gneiss, schist, quartzite, and granites from uplands (Hunt 1974). Pavich and others (1989) described the upland residual mantle (regolith-weathered rock, saprolite, subsoil, and soil) of Fairfax County, Virginia, as representative of the outer Piedmont Crystalline Province of Virginia and Maryland (Thornbury 1965). The area has a high drainage density with most of its perennial streams incised into unweathered bedrock.

Given the great age of the rocks, the high degree of weathering, and absence of quaternary glaciation, the regolith in the Piedmont can be quite deep. The maximum thickness of regolith is beneath flat upland hilltops. On schist, gneiss, and granite, it is typically 15–30 m deep. Rocks such as serpentine and quartzite, which weather slowly, have thin regolith (Pavich and others 1989). Throughout the outer Piedmont Crystalline Province, unweathered bedrock crops out in streams, and regolith is generally thin or absent in valleys of perennial streams (Pavich and others 1989). The contact between weathered and unweathered rock can be estimated on the side slopes of valleys by the location of heads of perennial streams at minor springs. Groundwater drains along the contact between weathered and unweathered rock and enters surface flow through springs (Pavich and others 1989). Most of the groundwater storage in the Piedmont is within the regolith above the unweathered bedrock (Pavich and others 1989). The saprolite acts as a relatively porous reservoir for groundwater. To a large extent, the depth of the regolith controls the hydrology of most Piedmont areas.

There are thought to be three general pathways for groundwater discharge in the Piedmont. In valleys underlain by weathered saprolite (often near headwaters), flow through the saprolite dominates baseflow. Water in the flow system is often oxic and may discharge nitrate directly to the stream channel. In valleys where streams have cut through the regolith to bedrock, springs begin in the valley flanks. Where streams have eroded to bedrock, discharge from fractures in the



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	High	Lower loadings than ICP. Valley shapes control local flow paths.	Select deeply rooted vegetation, restore small and large streams, seepage areas.
Removal of sediment and sediment-borne pollutants	High/Medium	Slope of non-floodplain areas. Volumes of surface runoff.	Restore in areas. Function dependent on Zone 3 in first few years. Enhance Zone 3.
Removal of dissolved phosphorus	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads. Increase infiltration.

Figure 9. Piedmont (thin soils)—idealized flow system, expected level of RFBS function, constraints to achieving the function, and management factors critical to achieving the function.

bedrock also contribute to streamflow. Stream discharge from the bedrock groundwater system is bypassed by the shallower systems if the regolith is not entirely eroded away. Even where bedrock contributes, most of the water in streams originates in the regolith because the volume of water in storage is so much greater than in the fractured bedrock. Most groundwater recharge in the marble valleys occurs rather rapidly into fracture zones close to the land surface. The regolith, although variable in thickness, is usually thin, and discharge to streams is probably from discrete fracture zones (in springs or directly into stream channels). As a result, there is probably little interaction of the groundwater with the root zone of riparian systems in the marble valleys.

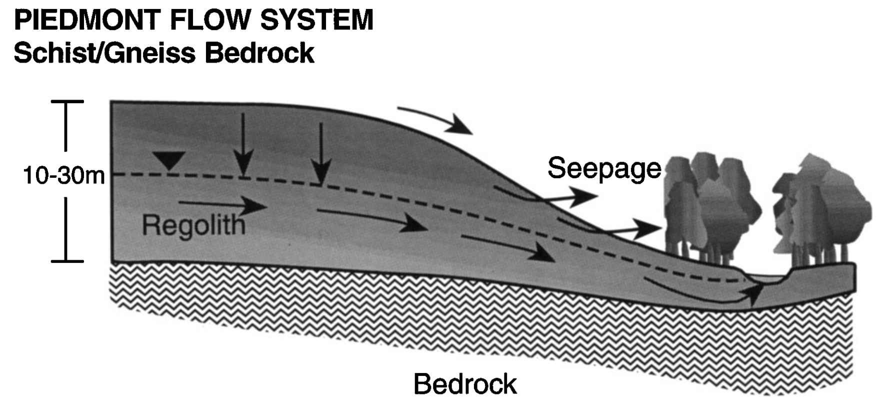
Based on hydrograph separations in the Piedmont of Chester County, Pennsylvania, Sloto (1994) found that baseflow ranges from 57% to 66% of watershed discharge, similar to estimates for the Virginia Piedmont of 60% (Pavich and others 1989). The remainder of streamflow occurs during and following storms, but the proportion that is surface runoff, as opposed to enhanced subsurface flows (e.g., through a near-stream rise in groundwater, drainage from soil layers, or rapid lateral transport through macropores), is difficult to determine. In forested watersheds, very little surface runoff occurs except from near-stream zones of high soil moisture. However, cultivated fields in the Piedmont generate greater surface runoff than fields in lower gradient Coastal Plains.

Three general hydrologic flow systems are represented for the Piedmont. The first system (Figure 9) represents areas of thin soils where much of the water moves in short flow paths to streams. These conditions are most likely in the Virginia Piedmont in the southern portions of the CBW. Areas with deeper soils and saprolite are divided into areas of schist and gneiss bedrock (Figure 10) and areas of marble bedrock (Figure 11).

Nitrate Transport in Shallow Groundwater

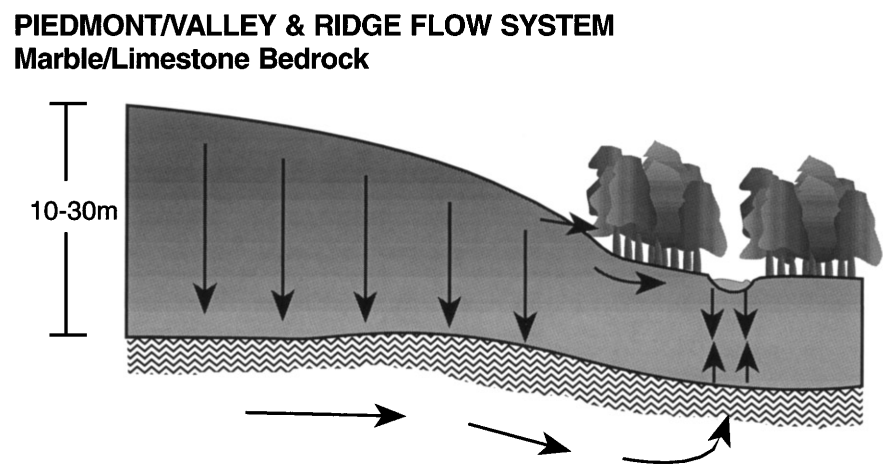
McFarland (1995) found that streams contained nitrate concentrations of 5–10 mg NO_3N/liter in the Maryland Piedmont. Most of the nitrate was attributed to discharge of water that was 0–5 years old from springs and from shallow flow systems in the regolith. Water in the bedrock was 20–30 years old with low or zero nitrate concentrations. Denitrification was suspected along older flow paths because of low dissolved oxygen and high iron concentrations in the water. This study indicated that riparian systems with deeply rooted vegetation could potentially reduce nitrate in streams by removing nitrate from spring flow and the shallow flow systems through the regolith.

Daniels and Gilliam (1996) examined spatial and temporal patterns of groundwater nitrate at three sites in the Piedmont of North Carolina. These areas are generally represented in Figure 9. Cultivated fields were separated from ephemeral or intermittent stream channels by 3–20 m of grass and naturally forested riparian



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Medium	More flow into regional aquifers, bypassing riparian zone.	Select deeply rooted vegetation. Restore in seepage areas.
Removal of sediment and sediment-borne pollutants	High/Medium	Slope of non-floodplain areas. Sediment loads in stream flow from valley sides.	Restore in areas with erosion impacting streams. Enhance existing forests with Zone 3.
Removal of dissolved phosphorus	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads. Increase infiltration.

Figure 10. Piedmont (schist/gneiss bedrock)—idealized flow system, expected level of RFBS function, constraints to achieving the function, and management factors critical to achieving the function.



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Low	Most flow into regional aquifers and into large rivers.	Dentrification focus. Select deeply rooted vegetation. Restore in seepage areas.
Removal of sediment and sediment-borne pollutants	High/Medium	Slope of non-floodplain areas. Sediment loads in stream flow from valley sides.	Restore in all areas with erosion impacting streams. Enhance existing forests with Zone 3.
Removal of dissolved phosphorus	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads. Increase infiltration and fine sediment filter.

Figure 11. Piedmont (marble bedrock)/Valley and Ridge (limestone bedrock)—idealized flow system, expected level of RFBS function, constraints to achieving the function, and management factors critical to achieving the function.

buffers. Nitrate concentrations in groundwater under the cultivated fields exceeded 10 mg/liter, but declined to lower levels in downslope wells. At one site, concentrations declined by as much as 30 mg/liter over a distance of 20 m from the field edge. The study did not include mass balance analyses of nitrogen losses, and interpretation is complicated by the fact that streamflow nitrate concentrations exceeded those in near-stream wells. Thus, the authors were unable to partition actual nitrogen removal within the riparian zone from mixing (or dilution) effects, although they speculated that both were involved.

The results of Daniels and Gilliam (1996) are consistent with findings from Coastal Plain studies showing that high rates of nitrogen removal occur in areas with high water table conditions and shallow groundwater movement near the root zone. This suggests that effective RFBS in the Piedmont depend strongly on the shallow flow paths of subsurface water in and near the riparian zone.

The likelihood that water reaches streams via shallow pathways depends both on the depth of the regolith in the vicinity of the stream and on the proportionate contributions to streamflow from the regolith and from the fractured zone (Sloto 1994). Olmstead and Healy (1962) concluded from analyses of temporal patterns in baseflow and water table elevations in the Brandywine Valley, Pennsylvania, that most streamflow originated from the regolith. Rose (1992, 1993) reached a similar conclusion from analyses of tritium variations in streamwater and groundwater in the Georgia Piedmont. If the regolith is beneath alluvial deposits near streams, much of the water reaching streams may pass through the riparian zone at substantial depths and nitrate would not be removed by the RFBS.

Hooper and others (1990) used end-member-mixing analysis of water chemistry to distinguish water sources in a Georgia Piedmont watershed. They concluded that hillslope drainage contributed a large portion of both baseflow and stormflow drainage during the wet winter months. Groundwater dominated the baseflow during the dry summer months with significant contributions from the organic horizon during storms. Rose (1992, 1993) inferred from analyses of tritium and inorganic analyses that while baseflow during dry summer months in the Georgia Piedmont originated from groundwater with an average residence time of 15–30 years, higher winter baseflows included a substantial component of water with a much shorter residence time (less than 10 years) and lacking the chemical signature of groundwater. The water from higher winter baseflows would be likely to come in contact with the biologically active riparian zone.

In the North Carolina Piedmont, Daniels and Gilliam (1996) noted that soil water in an alluvium overlying saprolite was chemically distinct and apparently isolated from the deeper groundwater in the saprolite. They attributed the isolation to low permeability of the Bt soil horizon (subsoil compacted by tillage) and inferred that water in the soil layers traveled laterally above the Bt horizon into the riparian zone. Kaplan and Newbold (1993) hypothesized extended periods of soil water drainage following storms to explain patterns of dissolved organic carbon concentrations in a Pennsylvania Piedmont stream. These shallow waters would also come in contact with riparian zone soils and vegetation.

Removal of Sediment and Nutrients in Surface Runoff

Daniels and Gilliam (1996) studied sediment and chemical reduction by GVFS and riparian areas for two years at six sites in the North Carolina Piedmont. They reported that the total sediment load reduction by the vegetated buffers during the study period ranged from 30% to 60%. However much of the sediment (mostly sand) passed through the vegetated buffers during one storm. When the results of that one storm were omitted from the calculations, the buffers removed approximately 80% of the sediment. Removals of silt plus clay averaged approximately 80% for the two-year study period. Total P removals in the filters ranged from 50% to 70%. Soluble orthophosphate removal was highly variable and usually was 50% or less. Removal of various forms of N was also variable and generally ranged from 40% to 60%. Most of the reductions were observed within 7 m of the field edge. The authors noted that the slope of the GVFS was less than that of the fields, so some of the sediment removal could be attributed to the change in slope alone. They further cautioned that the effectiveness of GVFS on steeper slopes might be limited. Where runoff from fields was directed as concentrated flow into riparian forest areas without complete vegetative cover, little or no reductions in either sediment or nutrients were observed. From these observations, Daniels and Gilliam (1996) recommended upslope dispersal of drainage water directed into forested areas. Preliminary data from another North Carolina Piedmont site show good control of sediment and sediment-associated P in surface runoff, but poor retention of dissolved nutrients (Parsons and others 1994a,b).

Application of RFBS in the Piedmont

The consensus BPJ was that RFBS in the Piedmont represented a range of conditions for NPS pollution control, depending on both the localized and watershed hydrology and the proportion of excess precipitation that moves through the RFBS. When hydrologic conditions lead to surface runoff to streams and movement of groundwater in or near the root zone of the RFBS, the degree of NPS pollution control should be similar to conditions measured in the North Carolina Piedmont and potentially as effective as the Inner Coastal Plain condition. When excess precipitation moves into deeper groundwater and into larger streams through the hyporheic zone, control of groundwater pollutants such as nitrate may be minimal. Baseflow/stormflow separations and examination of the timing and magnitude of baseflow for Piedmont watersheds should provide a conservative estimate of the quantity of water moving through RFBS.

In areas with thin soils, direct flow paths to streams and a large amount of water movement through surface runoff and seepage faces are common (Figure 9). The consensus BPJ were:

1. Nitrate removal would be approximately as effective as in ICP systems. Nitrate removal may be more dependent on vegetation processes because of potential for deeper rooting depth in more aerated soils and the potential for longer residence time for water in Piedmont RFBS.
2. Control of sediment and sediment-borne pollutants in surface runoff should be as effective as ICP and North Carolina Piedmont systems. Control of sediment in surface runoff is likely to be limited by development of concentrated flow channels, especially in steeper RFBS areas of the Piedmont. These areas may require an expanded zone 2.
3. Control of all sources of P should be represented well by ICP conditions and conditions from North Carolina studies. As in these systems, control should be more effective for sediment-borne P than for dissolved P in either surface runoff or groundwater.

Piedmont areas with deeper soils and saprolite are likely to have longer flow paths and more water entering the stream channel directly from these longer flow paths and the hyporheic zone. These types of Piedmont systems are represented both by areas with primarily gneiss/schist bedrock and with primarily marble bedrock (Figures 10 and 11). Areas with primarily schist bedrock will have substantial seepage and will be subject to treatment in RFBS.

For Piedmont areas represented in Figures 10 and 11, the consensus BPJ were:

1. Nitrate removal would be medium in the Piedmont areas with schist/gneiss bedrock and should be used to control movement of water in both shallow water table conditions and in seepage areas near streams. Nitrate removal should be least important in Piedmont areas underlain by marble because of movement of groundwater and associated nitrate into regional aquifer systems that will recharge larger streams. This component of groundwater flow is likely to bypass riparian systems. In both systems, nitrate removal will likely be enhanced by deeply rooted vegetation.
2. Control of sediment and sediment-borne chemicals will depend on management of zone 3 to reduce the effects of concentrated flow and to protect reestablished forests. Steeper slopes in riparian areas may limit both the sediment filtering capacity and the retention time of water. These conditions may require expanded zone 3 and/or zone 2.
3. Control of all sources of phosphorus will be limited by ability to remove dissolved P in surface runoff. Areas with high sediment-borne surface runoff P loads should be restored on a priority basis because of potential for controlling these P types.

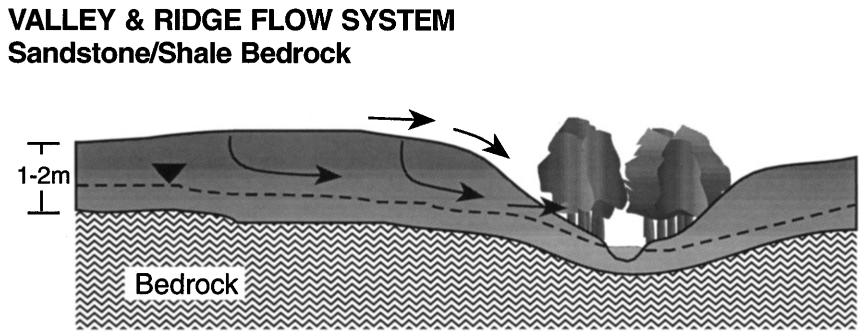
Valley & Ridge/Appalachian Provinces

Land Use and Hydrology

The Valley & Ridge Province (Figure 2) is the area in which structures due to folding dominate the topography. The Valley & Ridge and Appalachian Plateau make up about 60% of the Chesapeake Bay Watershed (Table 1). Geomorphologically, the Valley & Ridge Province is one of folded mountains in which resistant strata form ridges and weaker rocks are worn down to lowlands. Valleys within this province are underlain by limestone or shale, and the ridges are capped by the more resistant rocks (well-cemented siliceous sandstone and conglomerate).

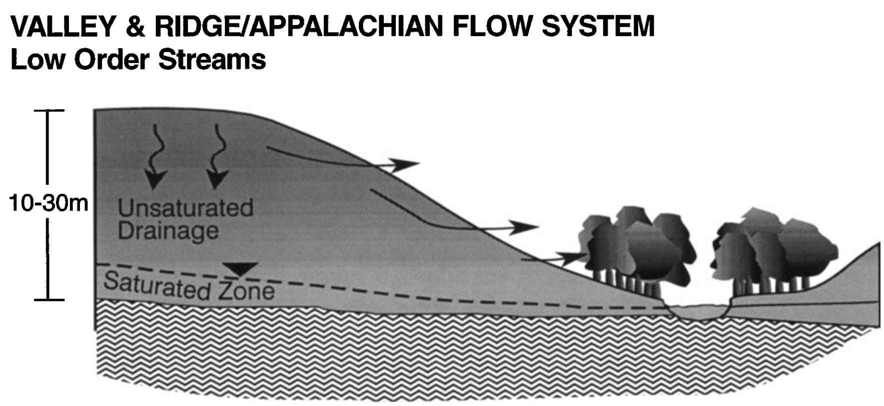
The physical characteristics of the Valley & Ridge are intimately connected with its streams, which are the primary causes of the present topography. Streams develop mostly on belts of soft rock, crossing hard rock ridges infrequently and usually at right angles. The bay watershed encompasses the middle section of the Valley & Ridge. Distinctive features of this section are conspicuous trellised drainage patterns and a comparative absence of ridges on its southeastern one quarter to one third, the Great Valley (Fenneman 1938).

Heath (1984) placed the Valley & Ridge in the Central Nonglaciated groundwater region. The region is characterized by thin regolith underlain by fractured sedimentary bedrock. The principal water-bearing openings in the bedrock are fractures that develop both along bedding planes and across them at steep angles. Openings developed along the fractures are usually less than 1 mm wide. The principal exception to this is in limestone, where water moving through the original fractures has enlarged them to form, at the extreme, extensive cavernous systems capable of transmitting large amounts of subsurface flow. Recharge of groundwater in this region generally occurs in outcrop areas of the bedrock aquifers in the uplands between streams. Discharge from the groundwater system is by springs,



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Medium/High	Presence of seeps and floodplains. Valley configurations.	Select for vegetation uptake especially early in growing season. Deeply rooted.
Removal of sediment and sediment-borne pollutants	High/Medium	Sediment loads in stream flow from valley walls. Slopes of non-floodplains.	Restore in all areas with stream erosion. Enhance Zone 3 to control sediment.
Removal of dissolved phosphorus	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads. Increase infiltration.

Figure 12. Valley and Ridge (sandstone / shale bedrock)—idealized flow system, expected level of RFBS function, constraints to achieving the function and management factors critical to achieving the function.



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Medium/High	Residence time of water. Presence of seeps and floodplains.	Select deeply rooted vegetation for uptake. Zone 1 is important for removal.
Removal of sediment and sediment-borne pollutants	High/Medium	Sediment loads in stream flow from valley walls. Slopes of non-floodplains.	Restore in all areas with stream erosion. Enhance Zone 3 to control sediment.
Removal of dissolved phosphorus	Medium/Low	Control of dissolved P in surface runoff.	Restore in areas with large surface runoff P loads. Increase infiltration.

Figure 13. Valley and Ridge / Appalachian (low-order streams)—idealized flow system, expected level of RFBS function, constraints to achieving the function and management factors critical to achieving the function.

seepage areas, and direct inflow to the streambed. Flow characteristics are most complicated within the limestone aquifers and connections between lower-order streams and regional groundwater are quite variable in time and space. Flow systems in the Valley & Ridge regions are based on large valleys with either limestone bedrock (Figure 11), or sandstone / shale bedrock (Figure 12) and low-order streams draining ridge tops (Figure 13, Appalachian and Valley & Ridge).

Although 60% of the Chesapeake Bay watershed is located within the Valley & Ridge and Appalachian Plateau, little research has been conducted to evaluate RFBS. The lack of riparian ecosystem research in these regions may be explained by the complex hydrology and the small amount of wetlands in this province. The most intensive agricultural NPS pollution occurs in the limestone valley areas with complex hydrology for the scale of riparian zone studies. Many of the conditions that promote improvements in the chemical composition of waters discharging through riparian ecosystems (small land surface slopes, high water tables, and low aeration status) are commonly associated with wetlands. Only 1% of the Valley & Ridge located in the Chesapeake Bay watershed is classified as wetlands, which constitutes 7% of the wetlands in the Chesapeake Bay watershed (Table 1). This contrasts with the 57% of CBW wetlands on the Coastal Plain comprising 21% of the Coastal Plain within the CBW.

Nitrate Transport in Shallow Groundwater

The Mahantango Creek Watershed is located within the Susquehanna River Basin approximately 40 km north of Harrisburg, Pennsylvania. Topography, geology, and land use are typical of upland watersheds in the Valley & Ridge with relatively steep land-surface slopes and minimal floodplain development or alluvium. Most stream reaches expose bedrock over all or part of their length. Land use within the Mahantango Creek Watershed is approximately 57% cropland, 35% forest and woodlots, and 8% permanent pasture.

Groundwater provides most of the 60%–80% of streamflow estimated to be subsurface return flow (Gburek and others 1986). Because ridge-top soils are highly permeable, nearly all rainfall infiltrates. The finer-textured poorly drained soils adjacent to the stream often function as groundwater discharge zones during the dormant season. A shallow, approximately 10- to 15-m layer of weathered fractured bedrock overlays the entire watershed and has higher transmissivity than the deeper, less-fractured bedrock (Gburek and Urban 1990). There is a general pattern of higher nitrate concentration in shallow groundwater and lower concentration in deep groundwater. In the experimental area, all aquifer waters, both shallow and deep, discharge to the surface streams.

Although it is a very small portion of the watershed area, the near-stream zone exerts major controls on stream flow chemistry and hydrology. Because it is hydrologically dynamic, particularly as related to seep zone formation, the near-stream zone can control the amount and timing of surface runoff and thus downstream flooding. The water table response to storms strongly influences or controls subsurface discharge, the nature and extent of riparian vegetation, streambank stability, and the nature of the chemical and biological systems to which chemicals in the discharge are exposed. Surface–groundwater interactions are more frequent and longer lasting on lower portions of the watershed and the nitrate content of groundwater is less likely to be altered in the riparian zone in the upper portions of headwater streams than downstream (Pionke and others 1988, Pionke and Urban 1984).

A study of Mahantango Creek focused on the role of existing riparian zones (Schnabel 1986). On one side of the stream was a relatively flat, well-maintained grass strip, a steeper strip of woods, and then cropland. On the other side, a steep strip of woods (20–60 m) separated the stream from cropland. Nitrate-N concentrations in shallow groundwater under the grass strip were reduced by 25%–50% between 9 m and 6 m from the stream during the growing season. There were generally small differences in nitrate-N concentrations in shallow groundwater 3 m from the stream and baseflow in the stream. The water table was frequently deeper than 1 m, particularly on the wooded side of the stream. The wooded side was much steeper and did not develop seepage zones as frequently as on the less steep, grassed side of the stream (Schnabel 1986). Gburek and others (1986) estimated that nitrate reduction within the riparian zone of the Mahantango Creek Watershed was equivalent to 4% of the mineral N exported from the watershed during the year. The limited impact of riparian processes on total N export resulted from the small area near the stream thought to support denitrification at optimum rates, combined with the fact that the area generally expands after soil temperatures begin to decrease, presumably limiting denitrification and plant uptake rates. From simulation with a mixing model that viewed baseflow as a mixture of discharge from the shallow fractured part of the aquifer and the deeper, less fractured portion of the aquifer, Schnabel and others (1993) concluded that the riparian zone was not the major control on temporal variation in nitrate concentration at the outlet of Mahantango Creek.

A study designed to examine groundwater nitrate dynamics was conducted in the western portion of the Valley & Ridge Province on Bald Eagle Creek (Altman and Parizek 1995). Nitrate levels in groundwater decreased from 5 to 8 mg NO_3-N/liter beneath the field to less than 0.5 mg NO_3-N/liter in the riparian zone. Without further hydrologic data, this reduction might be attributed to riparian processes. Based on flow-net analysis, water sampled in the riparian zone apparently did not originate from the field area with elevated nitrate levels. The groundwater flow direction did not follow the surface topography but instead followed the local bedrock topography. Groundwater was actually flowing toward the larger creek (Bald Eagle Creek), which the tributary stream was feeding. Their report did not address the fate of the nitrate-enriched water as it moved through the riparian system associated with Bald Eagle Creek, but does demonstrate the difficulty of determining RFBS function in areas of complex hydrogeology such as the Valley & Ridge.

Walker Branch Watershed in eastern Tennessee is an area similar to the Valley & Ridge region of the Chesapeake Bay. The study area was a 38.4-ha forested watershed with deep, highly weathered soils, a network of ephemeral stream channels, and a spring-fed perennial stream that flowed over dolomite bedrock in the lower portion of the watershed (Mulholland and others 1990). The watershed has broad ridges that slope steeply to narrow valleys. Soils have very high hydraulic conductivities in the surface. Reduced hydraulic conductivities at depth in the soil are associated with increasing clay. The weathered zone ranges in depth from a meter near the stream to about 30 m at the basin divides. Water held above the shallow restrictive layer flowed through the rhizosphere and was virtually depleted of nitrate (Mulholland and others 1990). However, water passing through the restrictive layer had higher nitrate concentrations. Groundwater transferred between catchments or leaked to deeper groundwater and discharged near the watershed outlet bypassed the riparian zone closest to the source of NPS pollutants. Where these transfers occurred, nitrate was less likely to be removed from groundwater by riparian zone processes.

Removal of Sediment and Nutrients in Surface Runoff

Studies by Dillaha and others (1988, 1989a,b) have shown the potential efficacy and limitations of grassed filter strips for controlling NPS pollution in the Valley & Ridge. Orchardgrass (*Dactylis glomerata*) filter strips were used to control potential sediment and nutrient losses from feedlots (Dillaha and others 1988). With shallow, uniform surface flow, 81% and 91% of the sediment and soluble solids were removed by 4.6 and 9.2 m GVFS, respectively. Where concentrated flow was allowed to occur, removal was much less. The GVFS were ineffective for controlling dissolved nutrients and nutrients associated with fine sediment. Concentrations of soluble N and P in effluents from GVFS were found to be high enough to cause eutrophication in receiving waters.

In a similar study of orchardgrass filter strips below fertilized cropland, Dillaha and others (1989b) obtained comparable results to the feedlot experiment. The sediment was initially trapped at the top of the GVFS. However, the GVFS became ineffective as it gradually became inundated with sediment. In surveys of farms that employed GVFS along streams in Virginia, Dillaha and others (1989a,b) found that in Valley & Ridge areas, the GVFS tended to be less effective than in flatter Coastal Plain sites. Except for localized erosion control along the stream bank, GVFS did little to mitigate NPS pollution from the upland in the Valley & Ridge because surface runoff usually became concentrated within the fields in natural drainageways before entering the GVFS. In general, the GVFS were most effective below smaller fields where water could enter the GVFS before it had an opportunity to concentrate.

Even where the GVFS had potential for sediment trapping, in many cases inadequate maintenance had rendered them ineffective (Dillaha et al. 1989a). Lack of mowing sometimes allowed taller weeds to shade out low ground cover, thereby reducing the capability of the GVFS to trap sediment. Erosion across the GVFS had caused severe gully problems in some cases. Heavy traffic had sometimes damaged the sod and created ruts. Sediment buildup on some sites had caused the upper margin of the GVFS to be higher than the adjacent field, or sometimes ditches from moldboard plowing were created parallel to the upper edge of the GVFS. In either of these two latter situations, water ran parallel to the edge of the GVFS until it could get across it in concentrated flow.

Application of RFBS in the Valley & Ridge/Appalachian Plateau

The Valley & Ridge is represented by larger-order streams draining the main valleys with either limestone bedrock (Figure 11) or shale/sandstone bedrock (Figure 12) and by smaller order streams draining the ridges (Figure 13). The situation for sediment and P sources is thought to be similar to the Piedmont hydrologic settings. Nitrate removal will probably show the most variability among the different valley types and with different valley configurations and floodplain extent. Consensus BPJ for larger order streams in the Valley & Ridge for nitrate removal functions were:

1. Valley & Ridge areas with limestone bedrock (Figure 11) will have the least potential for nitrate removal due to most flow going into regional aquifers that are intercepted primarily by major rivers. Seepage areas, springs, and floodplains will have the most potential for nitrate removal. Deep-rooted vegetation should be used to control nitrate in these areas.
2. Valley & Ridge with sandstone/shale bedrock (Figure 12) will have more potential for nitrate removal due to less movement of groundwater and nitrate into regional aquifers and the importance and prevalence of seepage areas moving nitrate into biologically active soil horizons. The processing of nitrate is controlled by the presence and size of the floodplain and by the presence of seepage areas

and springs. As in other Piedmont and Valley & Ridge settings, deep-rooted vegetation should be used to maximize the potential for N uptake.

3. Nitrate removal from low-order streams in both Valley & Ridge and Appalachian Plateau (Figure 13) settings will depend on residence time of water and the presence of seeps and floodplains. In these cases, as in other situations without extensive wetlands, the use of deeply rooted vegetation should enhance nitrate uptake. Because of the limited extent of riparian systems in areas of high relief, zone 1 will be important for nitrate removal in these smaller streams.

Factors Affecting Nonpoint Source Pollution Control

Loading Rates

As a nonpoint pollution control practice, RFBS represent a long-term investment that can change landscape structure. As a long-term management option, it is quite likely that RFBS will be exposed to a wide range of pollutant loadings due to both interannual variation and changes in management practices in source areas. Information on how mature RFBS respond to changing pollutant loads is essential to understanding long-term sustainability of RFBS.

Higher rates of nitrate removal would be possible under higher loadings of nitrate, especially where denitrification is the primary means of nitrate removal. Given the range in nutrient uptake observed both among different plant species and within the same plant species, it is likely that vegetation uptake will increase with increasing loads, if there is significant hydrologic interaction with vegetation.

Increasing loads of P are likely to be less effectively controlled than increasing loads of N, because of the lack of an analogous microbial process to remove or sequester P in the RFBS. If increasing P loads are to be controlled, it will require both effective management of zones 3 and 2 for sediment removal and management of zone 2 for infiltration. If dissolved or particulate P can be retained in the root zone, it will be available for both biological and chemical removal processes. If RFBS have some absolute removal potential for P, reducing input loads should increase the efficiency of removal.

Management to control increasing loads of sediment and sediment-borne chemicals will require specific management of zones 3 and 2 for sediment retention. Most of the mass of sediment will be deposited in zone 3, and most of the sediment-borne nutrients will be deposited in zone 2. Increased sediment loadings to zone 3 will require increased management to eliminate concentrated flows, remove accumulated sediment (especially in berms), and restore the herbaceous vegetation. Increased sediment and sediment-borne chemicals to zone 2 should lead to higher amounts of chemical deposition in surface litter. As with other dissolved P in surface runoff, the ability of zone 2 to retain P may be limited, especially under high loadings of dissolved P.

Loading rate–buffer width relationships are only poorly defined, especially for dissolved pollutants. In published studies with water clearly in contact with surface litter or the biologically active root zone, buffers of about 33 m have been effective for at least sediment and nitrate removal. One of the difficulties in describing these relationships is that increasing pollutant loads may also be accompanied by increasing water volumes in surface runoff, groundwater, or both. In the presence of increased water movement, denitrification for nitrate removal should be enhanced and sedimentation and infiltration may be decreased. Increased surface runoff and loading of sediment and sediment-borne chemicals can be accommodated by management of zones 3 and 2 to increase roughness and control channelized flow. Although mass balance approaches may be extrapolated to higher loading rates, they provide only an estimate and may not predict real-world responses.

Stream Order/Size

Regardless of the size of the stream or the hydrologic setting, water moving across the surface or through the root zone of a RFBS should show reduction in either nitrate (groundwater) or sediment and sediment-borne chemical loads reaching the stream (surface runoff). On lower-order streams there is greatest potential for interactions between water and riparian areas. As streams increase in size, the effects of immediately adjacent riparian ecosystems should decrease relative to the overall water quality of the stream. For NPS pollution control, the change in impact of RFBS as stream order increases can be estimated based on hydrologic contributions from upstream and from the riparian ecosystem. For first-order streams, the potential impact of the RFBS on chemical load or flow-weighted concentration is directly related to the proportion of the excess precipitation from the contributing area that moves through or near the root zone or surface of the RFBS. For all streams above first order, the contributing area is only one source of pollutants, with upstream reaches providing the other source. For second-order and above, the pollution control by RFBS is based on both the proportion of water from the contributing area that moves through the riparian system and the relative sizes of the two potential pollutant loads—upstream sources or adjacent land uses. Clearly, the larger the stream, the less impact a RFBS along a particular stream reach can have on reduction in overall load within that reach.

On a watershed basis, the higher the proportion of total streamflow originating from relatively short flow paths to small streams, the larger the potential impact of RFBS. In comparing the potential effectiveness of RFBS among watersheds, drainage density (length of channel per unit area of watershed) should provide a useful starting point. Higher drainage density implies greater potential importance for RFBS in NPS pollution control.

Sustainability and Establishment

The three-zone RFBS specification is based on studies of naturally occurring riparian forests along low-order (first- to fourth-order) streams and experimental-scale grass filter strips. Under natural conditions, riparian forest ecosystems formed a dynamic yet stable buffering system along most shorelines, rivers, and streams in the CBW. Although few studies have documented the specific changes in water quality during the establishment period of a riparian forest, established RFBS are expected to sustain water quality over the long term in a manner similar to the natural system.

The effect of upstream activities that modify hydrology or pollutant loads, loading rates, or the changes due to management of the RFBS, such as timber harvest, add uncertainty and risk to predicting changes in water quality over time. However, existing research, knowledge of riparian ecology, and experience with related hydrologic systems can form the basis for recommendations on the sustainability of RFBS.

RFBS should be used as part of an integrated land management or conservation system that consists of watershed scale management, NPS pollution control, and active management of the RFBS. In this way, RFBS become part of conservation; stormwater, nutrient, and farm management; timber harvest; and other land management planning efforts.

Watershed management is essential to reduce overall pollutant loadings and integrate the riparian area as part of a landscape influenced by upstream hydrology. In a landscape context, RFBS that mimic the natural ecosystems of the area will increase the likelihood of long-term sustainability. Consideration of existing riparian forests and linkage of RFBS as continuous stream corridors is desirable. Source management and land conservation measures are important in conserving natural resources, reducing overall pollution, and limiting stress on the RFBS. These measures, along with maintenance of buffer plantings, are especially important during the establishment phase and in preventing excessive runoff or sediment and nutrient loading beyond the capacity of the buffer. RFBS management, such as periodic harvesting, runoff control maintenance, control of invasive plants, etc., is desirable to maximize performance and ensure long-term effectiveness. Continued runoff control and protection of zone 1 functions are essential to maintaining optimum performance in RFBS.

RFBS are subject to failure, as are any best management practices. Acute failure, such as runoff inputs that exceed the design of the RFBS and cut gullies or channels, or chronic problems, such as a gradual decrease in phosphorus retention, should be addressed as part of a watershed management plan. Where gullies have formed into or through riparian forests, measures other than flow-spreading in zone 3 will be necessary to control channelized flow. Because of the commitment of land required for RFBS establishment, the approaches used for establishment and subsequent management should contribute to a RFBS that is sustainable for decades.

Summary and Conclusions

Riparian forest buffer systems are generally effective for control of sediment and sediment-borne pollutants carried in surface runoff. Properly managed RFBS should provide a high level of control of sediment and sediment-borne chemicals regardless of physiographic region. Natural riparian forest studies indicate that forests are particularly effective in filtering fine sediments and promoting codeposition of sediment as water infiltrates. The slope of the RFBS is the main factor limiting the effectiveness of the sediment removal function. In all physiographic settings, it is important to convert concentrated flow to sheet flow in order to optimize RFBS function. Conversion to sheet flow and deposition of coarse sediment that could damage young vegetation are the primary functions of zone 3—the grass vegetated filter strip.

The next most general function of RFBS is to control nitrate in shallow groundwater moving toward streams. When groundwater moves in short, shallow flow paths, such as in the Inner Coastal Plain, 90% of the nitrate input may be removed. In contrast, nitrate removal may be minimal in areas where water moves to regional groundwater such as in Piedmont and Valley & Ridge areas with marble or limestone bedrock, respectively. In these and some Outer Coastal Plain regions, high nitrate groundwater may emerge in stream channels and bypass most of the RFBS. In the areas where this occurs or where high nitrate water moves out in seepage faces, deeply rooted trees in zone 1 or in seepage areas may help remove nitrate. The degree to which nitrate (or other groundwater pollutants) will be removed in the RFBS depends on the proportion of groundwater moving in or near the biologically active root zone and on the residence time of the groundwater in these biologically active areas.

The least general function of RFBS appears to be control of dissolved phosphorus in surface runoff or shallow groundwater. Control of sediment-borne P is generally effective. In certain situations, dissolved P can contribute a substantial amount of total P load. Most of the soluble P is bioavailable, so the potential impact of a unit of dissolved P on aquatic ecosystems is greater. It appears that natural riparian forests have very low net dissolved P retention. In managing for increased P retention, effective fine-sediment control should be coupled with use of vegetation that can increase P uptake into plant tissue.

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