Coastal Plain Riparian Forests: Their Role In Filtering Agricultural Drainage

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INTRODUCTION

The Atlantic coastal plain of North America was almost entirely forested prior to European settlement in the 17th century. The European settlers lost very little time clearing much of the land for agriculture. In some areas they cleared the forest all of the way to the water's edge, but along small streams and in the rather hilly inner coastal plain it was often not worth the effort to clear the riparian areas. On the inner coastal plain of Maryland these areas were usually too steep or too water-logged in the spring to cultivate. Thus, a landscape developed in which the uplands were farmed and the lowland riparian zones were left as relict deciduous hardwood forest. These forests were usually logged but otherwise left unmanaged.

A number of studies have now found that conditions within coastal plain riparian forests are ideal for the removal of nitrate from agricultural drainage as it percolates through the soils of these forests. Although the soil surface in these forests is seldom covered with standing water, the groundwater table is near the surface, except during an extended drought. The oxidation/reduction potential of the soils beneath the water table is normally quite low (< 200 mv). The study sites include the Little Creek Watershed on the inner coastal plain of Georgia (Lowrance et al., 1984a), several middle coastal plain sites in North Carolina (Jacobs and Gilliam, 1985a,b), and the Rhode River watershed in the inner coastal plain in Maryland (Correll and Weller, 1989). These sites include first, second, and third order streams. Conceptually, it is important to remember that all of the water in a first order stream had to traverse its riparian zone before entering the channel. As stream order increases, the chances for physical/chemical interactions to occur are more and more restricted to flooding events; thus the name flood plain. Of course all streams receive some lateral inputs, but these become proportionally less important as stream order increases. At the Rhode River site, efficient removal of sediments and phosphorus (Peterjohn and Correll, 1984) from overland flows and removal of acidity from groundwater (Peterjohn and Correll, 1986) were also documented. In general, all of these functions of riparian forest were effective in both winter and summer.

Riparian forests have a number of other beneficial effects regardless of their geographical/geological setting. These might be termed habitat values and include shading of the stream channel, provision of leaf litter and woody debris, nesting sites and cover for birds and wildlife, and microclimatic effects on adjacent uplands (see reviews by Swanson et al., 1982; Naiman et al., 1988).

RESULTS

Surface Water Effects

Only a few studies have quantitated the effects of riparian forests on agricultural surface water drainages, perhaps due to the technical difficulties of making such measurements at most sites. At the Rhode River site in Maryland a 50-meter wide riparian forest removed 4.1 tonnes of sediment, 11 kg of particulate organic nitrogen, 3.0 kg of particulate phosphorus, 0.8 kg of ammonium-N, and 2.7 kg of nitrate-N per ha year during a one year study (Peterjohn and Correll, 1984). Most of these removals occurred within the first 19 meters (See Table 1). Of the amounts entering the riparian forest in overland flows 94% of the total particulates, 78% of exchangeable ammonium, 86% of particulate organic-N, 84% of total particulate-P, 74% of exchangeable orthophosphate-P, and 64% of particulate organic-C were removed in the forest. Precipitation that year was 100.4 cm, 7.6 cm below the long-term mean. Moreover, the year studied had relatively little overland flow. Watershed discharge that year was 23.2 cm, of which 7.1% occurred as overland flows. The long-term annual mean was 16.8% overland flow. Overland flows were most common during intense summer storms. Total summer discharges were 17.3% of annual discharges and overland flows in that summer were 20% of the total summer discharge.

The characteristics of the particulates entering the stream channel were significantly different from those leaving the fields. Due to selective removal of the coarser fraction, the average particulates entering the stream increased in organic-C from 1.5 to 8.2%, in organic-P from 0.064 to 0.14%, and in organic-N from 0.30 to 0.64%. The content of exchangeable ammonium and orthophosphate per weight of particulates also doubled.

In a study of a mid-coastal plain watershed in North Carolina, Cooper et al. (1986) measured the transport and deposition of soils marked with Cs-137, a radionuclide in bomb fallout in the 1950s and 1960s. They found that over 80% of the sediments discharged from farm fields into the riparian forest were deposited in the edges of the riparian forest before reaching the flood plain. About 50% of the sediments were deposited within 100 meters of their exit points from the fields.

Groundwater Effects

Most of the literature on coastal plain riparian forest effects on agricultural drainage focuses on removal of nitrate from shallow groundwater discharges from adjacent fields. The sites that have been studied all have confining soil layers near the surface that prevent deep infiltration of nitrate in the fields and force groundwater to percolate laterally through riparian forests before reaching drainage channels. A conceptual model of a cross section of this riparian forest wetland's soil (See Figure 1) summarizes the below-ground processes that dominate the nitrogen dynamics. Within that cross section, a series of processes occur in a defined order. When water first enters the forest, aerobic respiration is dominant until dissolved oxygen is consumed. After oxygen is consumed, conditions for denitrification are favorable with high concentrations of hydronium ion (a reactant), organic matter (the electron donor), and nitrate (the electron acceptor). After nitrate depletion, sulfate reduction is initiated as the oxidation/reduction potential drops. The width of the zones of oxygen, nitrate, and sulfate reduction varies seasonally with temperature and hydrologic factors.

Concentrations of solutes in groundwater may be increased due to evapotranspiration (ET). At the Rhode River site groundwater nitrate budgets were corrected for the effects of ET by a chloride budget technique (Peterjohn and Correll, 1986). Chloride, which acts as a passive tracer, increased in concentration in groundwater traversing the riparian forest due to the effects of ET. ET accounted for an average of 67% of the sum of precipitation and groundwater inputs to the riparian forest over a three year period (Correll and Weller, 1989). The high ET rates from the riparian forest are not surprising, since this wetland forest is seldom subjected to water stress. If the effects of ET were not corrected considerable error would result. At the same time that chloride concentrations in groundwater increased, nitrate concentrations decreased. After correcting for the concentrating effects of ET, the actual rates of nitrate removal were even larger.

While the nitrate concentrations were decreasing by three to five PPM, dissolved Kjeldahl-N (all reduced forms of N) only increased 0.27 PPM. Both annual and seasonal nitrate fluxes from the fields were highly variable among years. The variability may be related to differences in rainfall. Three years of continuous study spanned dry, average, and wet years (Correll and Weller, 1989; See Table 2). In all cases, even the extremely wet spring of 1984, most of the nitrate inputs from precipitation and cropland groundwater drainage was removed in the riparian forest. On average, 86% of the nitrate was removed annually, 81% in the winter, 90% in the spring, 83% in the summer, and 96% in the fall. Thus, nitrate removal was almost as efficient in the winter as in the other seasons. Although stream base-flow rates correlated with stream nitrate concentrations ($R^2 = 0.4$; Correll and Weller, 1989), seasonal nitrate removal efficiencies were not strongly related to precipitation volume (See Table 2). Studies at the Rhode River site have also consistently found that nitrate removal was not reflected in increased fluxes of dissolved reduced nitrogen fractions (Peterjohn and Correll, 1984, 1986; Correll and Weller, 1989). Thus, nitrate in groundwater was not merely converted to dissolved reduced forms of nitrogen. Nitrate removal from groundwater was about 40 kg nitrate-N/ha yr.

Another effect was the neutralization of most of the acidity in the groundwater. The cropland groundwater discharges had a mean annual pH of 4.5 (Peterjohn and Correll, 1986), while the groundwater entering the stream had a mean annual pH of 5.5. The acidity of the cropland groundwater was largely due to nitrification of reduced nitrogen fractions in the soil (Correll et al., 1987). The neutralization in the forest was probably due to denitrification in the soils (Peterjohn and Correll, 1986).

In middle coastal plain watersheds in North Carolina, 10 to 55 kg nitrate-N/ha yr were removed from cropland drainage by riparian forest (Jacobs and Gilliam, 1985 a,b). This was essentially all of the nitrate in these groundwaters. In other sites with heavy clay soils, not much groundwater movement occurred so only 6 to 12 kg nitrate-N/ha yr were removed. On the inner coastal plain of Georgia, efficient removal of nitrate from groundwater leaving farmlands was also reported (Lowrance et al., 1984a,b,c) with an annual average removal of 76%.

Flood Water Effects

Very few results have been reported for the quantitative effects of flood plains in the coastal plain on waters flooding out from the channel. Brinson et al. (1984) measured uptake of nutrients in a manipulative experiment on the flood plain of the Tar River in North Carolina. Plots were exposed to nutrient solutions weekly for 46 weeks and nutrient removals were recorded. Loading rates were one g/m^2 week of nitrate-N, ammonium-N, and phosphate-P, respectively. All three nutrients were essentially completely removed. The phosphate and ammonium accumulated in the sediments, while the nitrate was denitrified.

DISCUSSION

It would seem that many coastal plain riparian forests are efficient filters for sediments and nutrients in cropland drainages. However, the rates of retention are presently rather poorly documented, especially for overland storm flows from the uplands and waters flooding out from channels onto flood plains. Most studies have not produced sufficient information to allow actual fluxes and mass balances to be calculated.

Nitrate removal from groundwaters has been studied more than the other effects. In cases where the groundwater is confined to shallow soil layers and conditions are adequate to maintain anoxia below the water table, nitrate removal is efficient. The actual fate of the nitrate removed from groundwater is uncertain, but several lines of evidence suggest that denitrification is very important. In one case, direct field measurements with chambers which cover the soil documented significant releases of nitrous oxide (Correll, 1991). At the North Carolina and Georgia sites, soil cores were incubated in the laboratory and confirmed the potential for high denitrification rates (Jacobs and Gilliam, 1985a,b; Lowrance et al., 1984a). Net assimilation into woody biomass of the forest was also evaluated as a nitrogen sink. At the Rhode River site nitrogen mass balances among cropland discharges, precipitation inputs, and wetland discharges indicated a net retention by the riparian forest wetland of 75 kg total-N per ha year. Of this net retention, nutrient assimilation and long-term storage in woody biomass accounted for 12 to 20 kg N per ha year (Correll and Weller, 1989). The lower value was for a stand of 8- to 9-year-old saplings, while the higher value was for a 30-year-old stand more representative of the overall forest. Sediment deposition within the forest was estimated to be 4 tonnes per ha year and this sediment contained 12 kg total-N (Peterjohn and Correll, 1984). Much of the assimilation of nitrogen by the trees is probably from existing soil nitrogen rather than from new nitrogen inputs. Also, any new nitrogen assimilation is probably from precipitation and from overland flows, since these inputs are exposed to the leaves and fine surface roots of the trees, where most nutrient assimilation occurs. Nitrate removal from groundwater usually occurs near the forest/cropland boundary and not uniformly throughout the riparian forest. All of these facts have led us to conclude that below ground denitrification is the chief mechanism for nitrate removal from the groundwater, rather than plant assimilation and storage in woody biomass.

The research group in Georgia (Lowrance et al., 1984a,b,c) suggest that much of the removal of nitrate from groundwater is due to plant assimilation, while the group in North Carolina (Cooper et al., 1986; Jacobs and Gilliam, 1985a,b) suggest that most of the removal is the result of denitrification. All attempts to directly quantify annual denitrification fluxes in these systems have been unsuccessful, due to the existence of extreme spatial and temporal variability, the problem of quantifying emissions of dinitrogen, and the possible artifacts inherent in laboratory incubations of soil cores. One study has demonstrated that the denitrifying microorganisms in these riparian soils have a broad pH optimum from 5.0 to 7.0 (Waring and Gilliam, 1983). Thus, the acidity of the cropland drainage does not preclude efficient denitrification.

A third mechanism of nitrogen removal, other than plant assimilation and denitrification, also needs to be considered. Nitrogen storage in the soils has not been quantified in any of the riparian forest studies cited. Since the volume of soil is large and most of the forest nitrogen is stored in the soil, considerable amounts of nitrogen could be stored through small increases in soil organic-N. Here we are concerned with changes in the composition of the existing soils, especially the subsoils, rather than the deposition of sediments eroded from the croplands as described. Changes in nutrient pools in soil profiles are extremely difficult to measure. The soil nutrients are spatially heterogeneous and analytical methods generally have a precision in the range of 5-10 percent. The riparian forest soils at the Rhode River site contain about 10 tonnes of N/ha in the top 1.3 m. About 25% of this nitrogen is concentrated in the top 10 cm. A change of one percent in this soil N pool is 100 kg N/ha, more than the net total-N retention of this forest soil per year. To address this possible nitrogen removal mechanism, we established permanent soil sampling plots in our study site and sampled them intensively in 1983. We plan to resample the same plots in 1993 to test for changes in organic matter and nitrogen content.

Clearly, these riparian forests are functioning as wetlands. The forest plants are producing organic carbon, which acts as an electron donor for below-ground processes. The shallow ground waters moving through these forests are maintained in an anoxic condition such that nitrate becomes a major electron acceptor resulting in high rates of denitrification. The riparian forests at the Rhode River are dominated by Sweet Gum (Liquidamber styraciflua) and Red Maple (Acer rubrum); (Peterjohn and Correll, 1984).

Despite the rather impressive buffering effects of riparian forests, significant amounts of nutrients and sediments still enter the streams and often cause overenrichment problems in receiving waters (e.g., Jordan et al., 1986; 1991 a,b). At the Rhode River site, the cropland/riparian forest watershed discharged 2.5 kg of nitrate-N and 1.3 kg total-P per ha during one year (Peterjohn and Correll, 1984). This nitrate discharge was 17% of the total nitrate inputs from atmospheric deposition and farm chemicals. Nitrate accounted for 51% of the annual total-N discharge. Discharges were highly seasonal with 61% of the nitrate being discharged in the spring and 72% of the total-P being discharged in the summer. Thus, there is a clear need for better farm field management to keep more sediments and nutrients on the fields as well as improved management of the riparian zone to improve the interception of nutrients and sediments discharged from the fields.

For better management of these coastal plain riparian forests, we need to attain a much better understanding of the factors controlling nutrient and sediment retention. Our present understanding of these factors is far from adequate. For example, the dependence of nutrient retention on forest species composition, age structure, width, or continuity are not known. Tree harvesting may help maintain high vegetative uptake and associated nutrient retention (Lowrance et al., 1984a; 1985), but denitrification may be more important than vegetative uptake in some systems.

We also need more information on how commonly shallow impermeable layers force agricultural groundwater discharges into riparian zones. Forests without such layers would still filter surface discharges and have diverse habitat values, but might not have significant impacts on groundwater quality. Optimal agricultural and riparian management schemes may well be different in landscapes with and without shallow confining layers.

We also need to be concerned about the impact of these systems on the atmosphere. The products of denitrification include both dinitrogen and nitrous oxide. If the major product of the denitrification in these wetlands is nitrous oxide, then the protection afforded to aquatic systems is at the cost of increased atmospheric pollution. Ideally, we would manage riparian zones to produce mostly dinitrogen, since nitrous oxide is both a greenhouse gas and reacts with ozone in the stratosphere, depleting the ozone layer that we rely on for protection from solar ultraviolet radiation (Liu et al., 1977). Nitrous oxide has been increasing in the stratosphere at a rate of 0.2% per year since 1975 (Rasmussen and Khalil, 1986). Bolin et al. (1983) estimated that the increase in nitrous oxide over the period 1973-1983 lead to a 1.5% reduction in stratospheric ozone concentration. More information is needed on emissions of nitrous oxide from forests, especially from riparian forests receiving high nitrate fluxes. Mellilo et al. (1983) concluded that forests may be significant sources of nitrous oxide to the atmosphere. Therefore, we need to determine not only how to more effectively remove nitrate from crop drainages in riparian forests, but also how to maintain appropriate conditions for the complete transformation of the nitrate to harmless dinitrogen gas.

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Table 1. Changes in seasonal and yearly mean nutrient concentrations in mg/L for surface runoff from cropland on the Rhode River watershed as it moves through a riparian forest along a first order stream (Peterjohn and Correll, 1984).

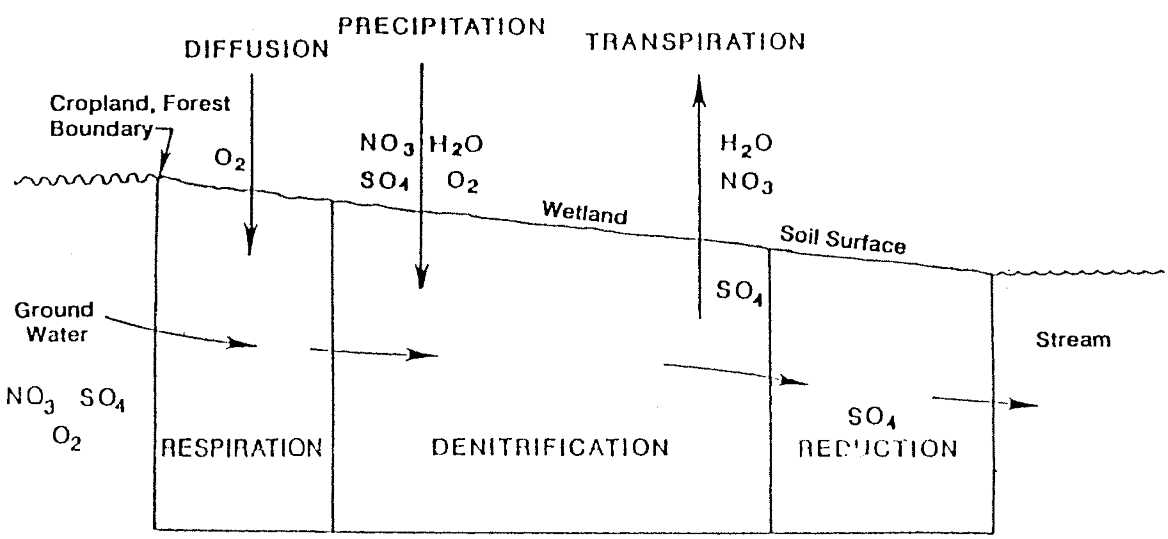
Position	Season	Total Sus. Part.	Nitrate N	Ammonium-N Exch. Part.	Ammonium-N Diss.	Organic-N Part.	Organic-N Diss.	Total-P Part.	Total-P Diss.
Entering riparian forest	Spring	8,840	3.73	0.73	3.63	27.7	1.47	3.22	0.26
	Summer	11,500	10.5	0.52	1.17	32.1	2.72	11.9	0.13
	Fall	3,830	1.57	0.30	0.90	16.8	0.78	3.29	0.13
	Winter	1,760	1.99	0.05	0.25	1.32	2.04	0.86	0.32
	Year	6,480	4.45	0.40	1.49	19.5	1.75	4.82	0.21
19 m into riparian forest	Spring	1,380	2.60	0.22	1.23	6.47	1.18	2.31	0.08
	Summer	966	1.93	0.12	0.41	5.06	1.44	2.09	0.09
	Fall	122	0.34	0.04	0.07	2.61	0.53	0.60	0.39
	Winter	176	2.18	0.04	0.16	0.37	1.33	0.06	0.38
	Year	661	1.76	0.10	0.47	3.63	1.12	1.27	0.24
Leaving riparian forest after 50 m	Spring	372	0.74	0.08	0.40	2.54	1.18	0.45	0.25
	Summer	524	1.03	0.11	0.18	3.46	0.71	1.04	0.18
	Fall*	----	----	----	----	----	----	----	----
	Winter	360	1.05	0.08	0.65	2.02	0.08	----	----
Year		419	0.94	0.09	0.41	2.67	0.66	0.74	0.22

* No overland flow reached these samplers in the fall.

Table 2. Fluxes of water and nitrate entering a first order stream riparian forest on Rhode River watershed via precipitation and cropland groundwater drainage and leaving forest via stream base flow and evapotranspiration (ET). From Correll and Weller, 1989. Watershed is composed of 10.4 ha of rowcrops and 5.9 ha riparian forest.

	Water Volume, cm				Nitrate, kg N/season or year		
Time period	Precip- itation	Crop drainage	ET	Stream baseflow	input	output	Input- output
Spring, 1981	25.4	18.9	26.9	17.3	64.4	8.0	56.4
Summer	29.7	10.4	31.3	8.8	45.9	5.5	40.4
Fall	18.8	1.1	18.2	1.7	10.8	0.6	10.2
Winter	26.5	38.3	33.1	31.7	209.6	28.3	181.3
Year one	100.4	68.7	109.6	59.5	330.7	42.1	288.6
Fall, 1983	28.4	4.3	28.9	3.8	21.0	0.7	20.3
Winter	34.0	40.4	38.6	35.8	96.7	29.5	67.2
Spring, 1984	50.8	47.2	55.6	42.4	191.0	14.6	176.4
Summer	22.2	18.1	25.8	14.6	56.6	12.7	43.9
Year two	135.5	110.0	148.9	96.6	365.3	57.4	307.9
Fall, 1984	31.0	1.0	30.5	1.5	13.4	0.1	13.3
Winter	34.4	13.6	33.5	14.4	46.3	8.3	38.0
Spring, 1985	18.8	9.1	20.8	7.1	26.5	5.0	21.5
Summer	20.2	0.9	20.4	0.7	6.7	0.6	6.1
Year three	104.3	24.6	105.2	23.7	93.0	14.0	79.0
Mean Fluxes for three years							
Winter	31.6	30.8	35.1	27.3	117.5	22.0	95.5
Spring	31.7	25.0	34.4	22.3	93.9	9.1	84.8
Summer	24.0	9.8	25.8	8.0	36.4	6.2	30.2
Fall	26.1	2.1	25.9	2.3	15.1	0.4	14.7
Year	113.4	67.8	121.2	59.9	262.9	37.9	225.0

Figure 1. Conceptual model of a cross section through the soils of a riparian forest.



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