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EFFECTS OF FOREST MANAGEMENT ON BIOGEOCHEMICAL FUNCTIONS IN SOUTHERN FORESTED WETLANDS

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Abstract: Southern forested wetlands perform two important biogeochemical functions on the landscape: 1) nutrient (N and P) removal from incident surface, subsurface, and ground waters, and 2) export of organic carbon and associated nutrients to aquatic ecosystems downstream. In addition to P sediment deposition, which can range from 1.6 to 36.0 kg ha^{-1} yr^{-1} P, denitrification of NO_3-N (0.5 to 350 kg ha^{-1} yr^{-1}) and P adsorption (130 to 199 kg ha^{-1} yr^{-1}) can be important mechanisms associated with N and P removal, respectively. Biological processes, uptake by plants (15.0 to 51.8 kg ha^{-1} yr^{-1} for N; 0.2 to 3.8 kg ha^{-1} yr^{-1} for P) and microorganism absorption (16.2 to 87.0 kg ha^{-1} yr^{-1} for N; 6.6 to 40.0 kg ha^{-1} yr^{-1} for P) are also important and are intimately associated with organic matter export. Clearcut harvests (ground-based or aerial), followed by natural regeneration, are the most common silvicultural techniques used in forested floodplains in the South. Ground-based methods have been shown to increase soil bulk density and decrease hydraulic conductivity and redox potential in wetter soils. In addition to the increases in soil temperature and soil wetness that frequently occur following forest harvesting, these added effects may be responsible for the reduced productivity and altered species composition observed following ground-based vs. aerial harvests. Changes in denitrification will be a function of the degree to which harvesting affects soil redox potential, substrate (C) availability, and nitrate production. In theory, denitrification rates should increase following harvesting, but low nitrate availability in acid soils may limit this effect. The effects of harvesting on P adsorption processes in forested wetland soils have not been studied. Reductions in plant uptake and litterfall and changes in species composition following harvesting could alter both nutrient retention/transformation and organic C export functions. On wetter sites, canopy removal may stimulate algal populations, providing a short-term mechanism for conserving geochemical exports. Clearcut harvest systems that minimize alterations in soil hydrology and promote rapid vegetation regrowth should have the least effect on biogeochemical functions in southern forested wetlands.

Key Words: bottomland hardwoods, clearcut harvesting, denitrification, floodplains, nitrogen, phosphorus, phosphorus adsorption, sediment deposition

INTRODUCTION

Palustrine forested wetlands represent about half (50.2%) of the estimated 40 million ha of wetlands in the conterminous U.S. (Wilen and Frayer 1990). Eighty-six percent of the total area of forested wetlands in the Southeast (VA, NC, SC, GA, and FL) occur in the Coastal Plain (Tansey and Cost 1990). The majority (67%) of these wetlands are found along small drainageways, with much smaller percentages in flatwoods, along major river floodplains, and in deepwater swamps, respectively (Table 1). In Caroline County, VA for example, non-tidal palustrine forested wetlands comprise 66% of the total wetland area of 11,372 ha; these are primarily small seasonally (4000 ha) and temporarily (1596 ha) flooded palustrine forested wetlands dominated by hardwoods (PFO1C's and PFO1A's, respectively) along stream drainages, averaging 5.4 and 2.4 ha in area, respectively (Walbridge and Struthers 1993). Eighty-six percent of the forested wetlands in the Southeast are bottomland (68%) or other (18%) hardwoods (Table 2).

We will focus our remarks on floodplain systems for two reasons: 1) within the category of forested wetlands, floodplain systems are the most important in terms of timber production and, therefore, are most

Table 1. Distribution of forested wetlands in the southeastern U.S. (VA, NC, SC, GA, and FL) (after Tansey and Cost 1990).

Wetland Location	% of Total Wetland Area
Narrow stream margins/small drainages	67
Flatwoods	15
Floodplains of major rivers	11
Deep swamps	8

Table 2. Distribution of forested wetlands in the southeastern U.S. (VA, NC, SC, GA, and FL)—by vegetation type (after Tansey and Cost 1990).

Vegetation Type	% of Total Wetland Area
Bottomland hardwoods	68
Other hardwoods	18
Pond pine	5
Other pines	9

often subjected to harvesting, and 2) floodplains play an important role in landscape processes (i.e., to a greater extent than some other types of forested wetlands such as basin systems) because they are biogeochemically linked to terrestrial and aquatic ecosystems upstream, upslope, and downstream.

Forested floodplains (i.e., bottomland hardwoods) currently comprise about 21% of the timberland in the Coastal Plain of the southern United States (McWilliams and Faulkner 1991). In 1977, it was estimated that the total area associated with these systems had decreased by 56%, primarily as a result of agricultural land-use conversion (Clark and Benforado 1981). Presently, losses continue at approximately 65,000 ha/yr (McWilliams and Faulkner 1991). As utilization pressures increase, public concern with regard to the maintenance of these wetlands remains high. The objectives of this paper are to review the important biogeochemical functions of southern forested wetlands and examine how these functions may be affected by forest harvesting.

BIOGEOCHEMICAL FUNCTIONS OF SOUTHERN FORESTED WETLANDS

Southern forested wetlands perform two important biogeochemical functions on the landscape: 1) sediment and nutrient (N and P) removal from incident surface, subsurface, and ground waters, and 2) transformation of inorganic nutrients to organic form, with subsequent release of organic C and associated N and P to downstream aquatic ecosystems (Mitsch and Gosselink 1993). Kuenzler (1989) reviewed N and P retention capacities of forested wetlands and found values ranging from 22 to 89 percent of total nonpoint source inputs for N and 20 to 81 percent for P. In this paper, we examine the mechanisms associated with N and P removal and/or retention in southern forested wetlands.

Sediment Deposition

It has long been known that sediment deposition, including both particulates and suspended solids, is a major mechanism associated with N and P removal in wetlands (Boto and Patrick 1979). For southern forested wetlands, fewer estimates exist for N (Table 3) than for P (Table 4). Phosphorus inputs via sediment deposition seem to increase both as a function of an increase in the importance of overbank flooding (cf. Brinson 1993) and increases in watershed P sources. Blackwater swamps and riparian forests along lower order streams have lower rates of sediment P deposition (1.6 to 3.0 kg ha^{-1} yr^{-1}) than forested wetlands found along higher order streams and rivers (13.6 to 36.0 kg ha^{-1} yr^{-1}).

Nitrogen Removal

As in most ecosystems, biologic processes dominate N retention and transformation in southern forested wetlands (Table 3). Denitrification has been suggested as the primary mechanism responsible for N removal in southern forested wetlands (e.g., Brinson et al. 1984, Peterjohn and Correll 1984). Using an anaerobic decomposition pathway, denitrifiers transform dissolved inorganic N (nitrate) to gaseous N_2O and/or N_2, releasing it to the atmosphere (a form of dissimilatory nitrate reduction). Nitrate can also be directly immobilized by soil microorganisms and reduced to ammonium internally (assimilatory nitrate reduction) or reduced to ammonium externally (a second form of dissimilatory nitrate reduction) (cf. Brinson et al. 1984, Bowden 1987).

Pathways of denitrification include both the direct denitrification of nitrate inputs as well as the denitrification of nitrate produced *in situ* by ammonium oxidation (nitrification). Ammonium can be produced *in situ* from the mineralization of organic N. Both ammonium and organic N can also enter the wetland in precipitation and via surface, subsurface, and groundwater flow. Processes affecting a wetland's ability to retain ammonium and/or organic N may ultimately be important in determining denitrification rates. Ammonium retention can be associated with microbial immobilization (16.2 to 87.0 kg ha^{-1} yr^{-1}), plant/algal uptake (15.0 to 51.8 kg ha^{-1} yr^{-1}), or adsorption by clay particles (0.8 to 64.2 kg ha^{-1} yr^{-1}) (Table 3). Since

Table 3. Mechanisms associated with nitrogen removal in floodplain and riparian forests in the southern United States.

Mechanism	Study	Location	N Removal Rate (kg ha^{-1} yr^{-1})
Sediment/particulate deposition	Peterjohn and Correll (1984)	Maryland	11.0
Denitrification (potential estimates)	Engler and Patrick (1974)	Bayou Sorrel, LA	350.0[1]
	Lowrance et al. (1984)	Little River, GA	31.5
	Ambus and Lowrance (1991)	Coastal Plain, GA	2–224
Denitrification (mass balance estimates)	Brinson et al. (1984)	Tar River, NC	130.0[2]
	Peterjohn and Correll (1984)	Maryland	47.7[3]
	Jacobs and Gilliam (1985)	Coastal Plain, NC	0.5–2.5
NH_4^+ adsorption	Brinson et al. (1984)	Tar River, NC	64.2[2]
	Peterjohn and Correll (1984)	Maryland	0.8[4]
Microbial immobilization[5]	Brinson et al. (1984)	Tar River, NC	16.2[2]
	Qualls (1984)	Coastal Plain, NC	87.0
Plant uptake	Brinson et al. (1984)	Tar River, NC	15.5[2]
	Lowrance et al. (1984)	Little River, GA	51.8
	Peterjohn and Correll (1984)	Maryland	15.0

[1] Potential estimate under conditions favoring denitrification (cf. Bowden 1987).
[2] Estimate over a 10 month period under an addition rate of 10 kg ha^{-1} mo^{-1}.
[3] Nitrate removal from both surface (2.7) and subsurface (45.0) flow; at least two thirds attributed to denitrification.
[4] Estimate based on NH_4^+ removal.
[5] Associated with leaf litter.

mineralization and nitrification are favored during periods of drydown, while denitrification is stimulated under flooded conditions, seasonal dynamics linked to flooding cycles can play an important role in determining annual rates of N retention and denitrification in southern forested wetlands (Brinson et al. 1984).

Denitrification estimates in Table 3 range from 0.5 to 350 kg ha^{-1} yr^{-1}, varying over 4 orders of magnitude. Rates are either estimates inferred from mass balance analyses (including one estimate based on the disappearance of added nitrate) or potential estimates based on nitrous oxide production from soil cores or slurries using the acetylene block technique. Bowden (1987) suggested that potential estimates for undisturbed systems in excess of 10 kg ha^{-1} yr^{-1} could exceed actual rates by 40–1000 x. However, Table 3 indicates similar ranges for both mass balance (0.5 to 130 kg ha^{-1} yr^{-1}) and potential (2 to 350 kg ha^{-1} yr^{-1}) estimates. Ranges of literature estimates for annual N removal via denitrification in southern forested wetlands are also roughly equivalent to those for other mechanisms of N retention, which may reflect a coupling of these other mechanisms with denitrification (cf. Brinson et al. 1984). Further studies, including both direct estimates of denitrification rates and ecosystem-level analyses comparable to Hemond's (1983) study of Thoreau's Bog, would greatly improve our understanding of the relative importance of the various pathways associated with N removal and retention in southern forested wetlands.

Phosphorus Retention

In addition to sediment deposition, plant uptake (0.2 to 3.8 kg ha^{-1} yr^{-1}), microbial immobilization (6.6 to 40.0 kg ha^{-1} yr^{-1}), and adsorption to clays (130 to 199 kg ha^{-1} yr^{-1}) have been shown to be important processes associated with phosphorus uptake and storage in southern forested wetlands (Table 4). Geochemical processes play a larger role in P vs. N retention and are thought to represent the primary mechanism controlling long-term P storage in most wetlands (Brinson et al. 1984, Richardson and Marshall 1986). Since P has no stable gaseous form, net retention is related to an increase in P storage within the wetland rather than a loss to the atmosphere.

The potential for phosphate removal from incident surface, subsurface, and/or ground waters has been shown to vary among acidic wetland soils, consistent with variations in their noncrystalline (oxalate-extractable) Al and Fe content (Richardson 1985). P sorption capacities have since been found to correlate with noncrystalline Al and Fe concentrations across a wide range of both wetland and upland soils (Richardson 1985, Richardson et al. 1988, Walbridge and Birk 1988, Walbridge et al. 1991). Only a few studies have examined P sorption potentials in southern forested wetland soils. Richardson et al. (1988) found that NC Coastal Plain swamps had relatively high P sorption capacities, similar to those of more northern swamps and fens described by Richardson (1985) (Walbridge and Struthers

Table 4. Mechanisms associated with phosphorus removal in floodplain and riparian forests in the southern United States.

Mechanism	Study	Location	P Removal Rate (kg ha^{-1} yr^{-1})
Sediment/particulate deposition	Brown (1978)	Prairie Creek, FL	32.5
	Mitsch et al. (1979a)	Cache River, IL	36.0
	Mitsch et al. (1979b)	Kankakee River, IL	13.6
	Yarbro (1983)	Creeping Swamp, NC	1.6–1.7
	Peterjohn and Correll (1984)	Maryland	3.0
Adsorption	Brinson et al. (1984)	Tar River, NC	199.0[1]
	Richardson et al. (1988)	Clarkton, NC	130.0[2]
Microbial immobilization	Brinson et al. (1984)	Tar River, NC	6.6[1]
	Richardson et al. (1988)	Clarkton, NC	35.0–40.0[2,3]
Woody plants	Schlesinger (1978)	Okefenokee, GA	0.2
	Mitsch et al. (1979a)	Cache River, IL	1.0
	Yarbro (1983)	Creeping Swamp, NC	0.6–1.2
	Brinson et al. (1984)	Tar River, NC	1.3[1]
	Lowrance et al. (1984)	Little River, GA	3.8
	Peterjohn and Correll (1984)	Maryland	1.6
Non-woody plants	Mitsch et al. (1979a)	Cache River, IL	3.3
	Yarbro (1983)	Creeping Swamp, NC	0.4

[1] Estimate over a 10 month period under an addition rate of 10 kg ha^{-1} mo^{-1}.
[2] Sewage-enriched system.
[3] Based on increase in $CHCl_3$-labile (microbial) P.

1993). Soil P sorption capacities reported for Coastal Plain floodplain forests in Georgia and Texas are lower than those reported for NC swamps but are much higher than the P sorption capacities of surrounding upland soils, suggesting that these forested wetlands may represent sites of comparatively high phosphate retention on the landscape (Walbridge et al. 1992, Walbridge and Struthers 1993). The comparatively high concentrations of amorphous Al and Fe often found in forested wetland soils could be due to the tendency for these sites to act as local areas of clay deposition on the landscape (Lowrance et al. 1988). High noncrystalline Al and Fe concentrations in southern forested wetland soils could also be due to the tendency for crystalline Al and Fe compounds to dissolve (i.e., transform to noncrystalline form) under flooded conditions (Kuo and Baker 1982, Sah and Mikkelsen 1986) and/or the ability of organic matter, often present in relatively high concentrations in wetland soils due to reduced decomposition, to inhibit Al and Fe crystallization (Schwertmann 1966, Kodama and Schnitzer 1977, 1980, Hsu 1989, Schwertmann and Taylor 1989).

Transformation of Inorganic Nutrients to Organic Form

Southern forested wetlands can also transform inorganic N and P to organic forms that are subsequently exported downstream. Elder (1985) found that forested riparian wetlands along the Apalachicola River in northern Florida were net importers of NH_4^+, NO_3^-, and PO_4^{3-}, and net exporters of both dissolved and particulate organic N and P, with minimal net retention of either nutrient. Approximately 1.1 kg/ha of N and 0.03 kg/ha of P were transformed to organic form annually. Similar transformations of inorganic N and P have been observed in other southern forested wetlands (Yarbro 1983, Kemp and Day 1984). Yarbro (1983) found considerable variation in annual P transformation rates during a 2-year study (0.015 to 2.4 kg/ha). Yarbro's lower estimate is comparable to Elder's (1985) estimate for northern Florida; her higher estimate was observed under conditions of increased flooding during a particularly wet year.

The association of inorganic N and P with organic carbon involves biological processes such as microbial immobilization, plant and/or algal uptake, and litterfall production. Southern forested wetlands commonly act as sources of organic C, with annual carbon exports ranging from 104 to 998 kg/ha (Day et al. 1977, Mulholland and Kuenzler 1979, Cuffney 1988). The quality of organic carbon exports can vary as a function of wetland type, discharge rate, and distance downstream (Leff and Meyer 1991).

IMPACTS OF HARVESTING

Examination of the above mechanisms related to the nutrient retention and transformation functions of southern forested wetlands suggests that these functions are unique to the forested wetland environment and are strongly linked to both seasonal dynamics and landscape-level processes. Seasonal dynamics are intimately associated with flooding regime and hydroperiod. Landscape-level processes are related to the type and magnitude of inputs from upstream/upslope and the effects of wetland exports on downstream ecosystems. The unique biogeochemical functions of southern forested wetlands are driven primarily by hydrology, soil chemistry, and biotic processes. Theoretically, the magnitude to which forest harvesting will affect the biogeochemical functions of southern forested wetlands will be determined by the effects of harvesting on these underlying driving variables.

Alterations in the biogeochemical functions of wetlands due to harvesting will vary as a function of 1) the harvest and regeneration method employed, 2) road construction, and 3) the type of floodplain ecosystem harvested. The timing of harvesting (i.e., season of harvest, stand age) could also be important, although harvesting usually occurs during the summer, when sites are driest and is generally limited to older (mature) stands. As knowledge concerning the effects of harvesting on the biogeochemical functions of southern forested wetlands develops, it might also become important to consider landscape-level linkages between potential upstream/upslope sources of N and P and harvesting effects on N and P retention (cf., Costanza et al. 1990, Walbridge 1993).

Harvest and Regeneration Method

The most common silvicultural scenario currently employed in floodplain hardwood management is clearcut harvesting followed by natural regeneration. Clearcut harvest methods may be broadly classed as ground-based vs. aerial and differ markedly in the degree to which soil and site processes are influenced. Alterations in biogeochemical functions are likely to be manifested through harvest-induced changes in short-term hydrologic relationships, microclimate, and revegetation dynamics. Particular aspects of biogeochemical cycling that are sensitive to harvest disturbances include 1) decomposition/mineralization, 2) assimilation of nutrients in vegetation, and 3) nutrient retention/transformation processes.

Given the inherent variability associated with forested floodplain ecosystems, management-induced changes in biogeochemical functions can be quite subtle and difficult to detect. Some silvicultural effects on biogeochemical functions are short-term in nature, while others cause measurable changes to occur for longer periods of time. While the majority (98%) of wetland harvesting uses ground-based methodologies (Bryce J. Stokes, USFS Forest Engineering Project Leader–Southern Forest Experiment Station, personal commmunication), there is increasing interest in the use of aerial systems.

When used under conditions of excessive soil wetness, ground-based methods, consisting of manual or mechanized felling combined with skidder removal of logs, have been shown to increase soil bulk density while decreasing hydraulic conductivity and oxidation-reduction potential, thus impeding gas exchange and ion movement via mass flow (Mader et al. 1989, Aust et al. 1990). The direct effects of these changes on N and P retention have not been examined. Lower redox potentials should favor increased rates of denitrification, given sufficient amounts of labile carbon and available nitrate. Increased soil bulk density and reduced hydraulic conductivity might reduce phosphate adsorption by soil minerals by reducing infiltration, although adsorption efficiency could increase due to increased contact time between water and sediments.

Combined with their higher potential for direct damage to advance regeneration and residual stumps, ground-based harvests have been shown to reduce natural regeneration productivity and alter species composition relative to levels observed following aerial techniques (Table 5, Lloyd et al. 1992). Slower vegetative re-occupancy following harvesting signals reduced nutrient assimilation by the vegetation; stabilization and tightness of geochemical cycling will be impaired until site occupancy is re-established. Stone et al. (1978) suggest that these alterations in geochemical cycling are temporary and will dissipate during the first decade of stand development (Stone et al. 1978), although re-establishment of geochemical equilibrium may occur at a different level.

In the short term, lower vegetation productivity will prolong increased soil temperatures, which could either slow or increase decomposition rates depending on soil moisture availability (Table 6, Griffin et al. 1992). In the long term, altered species composition, with lower net primary productivity (NPP), would reduce litterfall quantities and quality, increasing forest floor turnover time and reducing nutrient circulation. The magnitude of these changes is uncertain, however, and would depend on the degree of change in species composition, which is, successively, a function of the magnitude of soil and root impacts.

Road Construction

Depending on their design, construction of haul-roads associated with clearcut ground-based systems may alter hydrologic relationships due to the potential for floodplain constriction and subsequent changes in floodwater velocity caused by road beds and associated

Table 5. First year stump survival and net changes in densities on regeneration plots by species compared by harvesting system in blackwater floodplains, south Alabama.

Species	Percent Survival of Stumps		Percent Change in Density on Regeneration Plots	
	Aerial	Ground	Aerial	Ground
Acer rubrum L.	93.6	60.0	−80.2	−89.3
Cliftonia monophylla Britt	92.7	41.2	+11.6	−87.4
Cyrilla racemiflora L.	94.7	52.5	−61.4	−83.4
Ilex opaca Aiton	95.8	74.2	−68.5	−76.0
Liriodendron tulipifera L.	85.7	60.9	−21.7	−10.8
Magnolia virginiana L.	80.8	77.4	−41.4	−22.6
Nyssa sylvatica var. *biflora* (Walter) Sargent	59.3	48.2	−27.6	+128.9
All species	86.3	60.3	−46.5	−68.5

Table 6. Depth (below soil surface) of oxidized zones following harvests in blackwater floodplains, south Alabama.

Treatment	Depth (cm)
Aerial	16.8 a*
Ground	15.0 ab
Undisturbed	12.3 b
Microsite	**Depth (cm)**
Concave	5.8 a
Flat	16.5 b
Convex	23.0 c

* Means in columns followed by the same letter are not statistically different at the 0.05 probability level.

culverts (Gosselink et al. 1990). Such effects have the potential to alter the energy signature of the system (Lugo et al. 1990) and, similarly, create alterations in species composition and spatial distribution. The linkage between flood- and ground-water flow and NPP in wetland systems is well documented (Conner and Day 1976, Brown et al. 1979) and can be expected to relate strongly to a number of biogeochemical functions. The development of forest communities with lower productivity would change nutrient uptake rates, litter quality, and subsequent decomposition rates. As a result, nutrient retention/transformation and organic C export functions would undergo alterations as biogeochemical input/output relationships are re-established.

Type of Floodplain Ecosystem

Reductions in quantities of nutrients circulating and the possibility of subsequent decreases in N and P availability stemming from decomposition would have more significant implications for NPP in nutrient-poor systems, such as those associated with blackwater streams, as opposed to nutrient-rich redwater systems. In the former, P deficiencies are particularly likely to be exacerbated and, therefore, could influence species distribution as well as productivity. Conversely, nutrient-rich systems may temporarily act as net sources of nutrients via floodwaters rather than sinks or transformation zones.

Changes in gaseous transformations are difficult to predict due to the lack of disturbance-related data, but in the case of denitrification, would be a function of the degree to which harvesting stimulates anaerobic conditions, available C sources, nitrate availability, and populations of anaerobic microorganisms. In theory, these conditions should exist following a harvest, which causes a short-term increase in site wetness. However, limitations imposed by nitrate availability under acid soil conditions can result in no measurable increase in denitrification activity in some cases (Thornton et al. 1992).

There are indications that on moister floodplains, canopy removal may stimulate algal populations via enchancement of available sunlight (Brown et al. 1992). This observation, in combination with the nutrient assimilation capacity of these microflora in a wetland environment (Atchue et al. 1983), suggests that stimulation of algal populations may provide a short-term mechanism for conservation of geochemical exports following system disturbances.

In summary, those clearcut harvest systems that minimize alterations in soil hydrology and promote rapid revegetation can be expected to have the least effect on biogeochemical functions in floodplain forest systems. Aerial and/or carefully designed ground-based systems may meet these requirements, but this is partially dependent on the nutrient status of the particular forested wetland. Partial cuts (e.g., group and single-tree selection) would probably cause little measurable change associated with timber cutting *per se.* Those facets of harvesting with the greatest potential for impact (e.g., logging road design and skidder operations under wet soil conditions) need further evaluation to ensure that the unique biogeochemical functions of southern forested wetlands are re-established as rapidly as possible following harvesting.

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