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ABSTRACT

The Chesapeake Bay Agreement calls for a 40% reduction of controllable phosphorus and nitrogen to the tidal Bay by the year 2000. To accomplish this goal the Chesapeake Bay Program needs accurate estimates of nutrient loadings, including atmospheric deposition, from various land uses. The literature was reviewed on forest nitrogen pools and fluxes, and nitrogen data from research catchments in the Chesapeake Basin were identified. The structure of a nitrogen module for forests is recommended for the Chesapeake Bay Watershed Model along with the possible functional forms for fluxes.
Keywords: nitrogen loading, water quality modeling, forest nitrogen, Chesapeake Bay

INTRODUCTION

The Chesapeake Bay Watershed Model (Linker et al. 1993) is designed to simulate nutrient loads delivered to the estuary under different management scenarios. The nutrient loads are differentiated into anthropogenic loads amenable to management, and nonanthropogenic loads which are considered to be uncontrollable. The Chesapeake Bay Agreement requires a 40% reduction of controllable phosphorus and nitrogen to the tidal Bay by the year 2000. To determine the quantity of the controllable nutrient load and to evaluate the efficacy of nutrient control strategies, the entire 164,000 km^2 of the Chesapeake Bay drainage basin (Figure 1) have been simulated using the HSPF model (Hydrologic Simulation Program - Fortran). The HSPF model is a modular set of computer codes that simulate hydrology, nutrient and sediment export from pervious and impervious land uses, and the transport of these loads in rivers and reservoirs. Cropland is simulated with a detailed nonpoint-source load simulation module (AGCHEM) that includes application of fertilizer, manures, atmospheric deposition, crop uptake, soil binding, denitrification, and surface/subsurface export. Although 60% of the Chesapeake Bay drainage basin (hereafter called the Chesapeake Basin) is in forest, there is currently no module in HSPF to simulate forest nutrient outputs.

In order to select nutrient management options, it is necessary to have confidence in current estimates of contributions from different land uses and the ability of a model to provide acceptable estimates of contributions for possible future scenarios. From simulations in Phase II of the modeling activity it was known that the model was not adequately predicting nitrogen (N) for some basins (Donigian, et al. 1991). Usually N loads from forests would be considered an uncontrolled source; however, atmospheric N deposition to forests is expected to change with

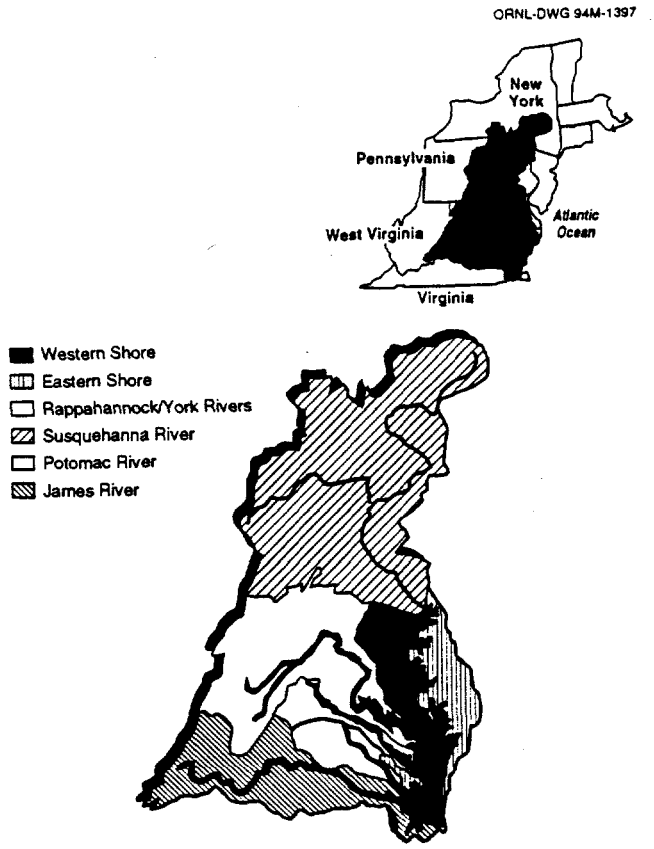

Figure 1—Major basins in the Chesapeake Bay drainage basin.

implementation of the 1990 Clean Air Act Amendments. Thus N loads from forests may change also. Total annual N deposition to the Chesapeake Basin ranges from 6 to 13 kg N/ha with higher loads in the northeastern portion (NAPAP 1991).

METHODOLOGY

We focused on the identified need of the Chesapeake Bay Program to better simulate N outputs from the forested portions of the Bay drainage and a short-term desire of the U.S. Environmental Protection Agency (EPA) to be able to build off of the existing HSPF model, if possible, for implementation within one to two years. In addition, EPA was interested in a model that would be responsive to changes in atmospheric deposition. We undertook three activities to meet EPA's needs 1) reviewed the literature on forest N pools and fluxes, 2) reviewed N data from research catchments in the Chesapeake Basin and 3) convened a workshop of scientists knowledgeable about forest N literature, data, and models. The following tasks were performed.

- Evaluated the estimated N loads from forested watersheds using the current model.
- Evaluated the HSPF AGCHEM module structure with regard to possible modification and use for a new forest module.
- Identified available data for parameterization and verification of a new forest module.

Our recommendations are oriented towards the Chesapeake Bay Program's short-term model needs, and the recommended approach is not meant to undermine the need for process models of N cycling in forests as these models should be most responsive to changes in atmospheric deposition and disturbance events.

Currently the HSPF simulations for the Chesapeake Bay Drainage Basin predict that nitrate is the dominant form of hydrologic N output from forests. The model predicts mean annual nitrate concentrations from forests ranging from 0.292 mg N/L in the James River Drainage to 2.28 mg N/L in the Susquehanna River Drainage. Outputs of nitrate range from 0.86 to 7.09 kg/ha per year. The model predicts a much lower and narrower range of ammonium concentrations of 0.019 (Patuxent) to 0.036 mg N/L (James) and outputs of 0.04 to 0.10 kg N/ha per year. Similarly, the model predicts low concentrations and outputs of dissolved organic nitrogen (DON) of 0.01 (Susquehanna) to 0.05 mg N/L (Potomac) and 0.13 to 0.42 kg/ha per year, respectively.

In the Susquehanna and Potomac River Drainages the HSPF simulations underestimate the output of all forms of N measured at the fall line stations on these rivers . This would imply that there are additional sources of all forms of N in these drainages not accounted for by the model. In the James River Drainage, the HSPF simulation overpredicts the nitrate and ammonium output and underpredicts the DON output. In the Patuxent River Drainage the model and fall line measurements are in reasonably good agreement. Verification and evaluation of the HSPF simulations of N concentrations and outputs can best be accomplished by comparing the model simulation results with empirical data collected from completely forested catchments within each of the major drainage basins in the Chesapeake Bay Basin.

RESULTS

Comparison Between Measurements and HSPF Simulations. There are few data on nitrogen concentrations or outputs from forested catchments in the Chesapeake Basin . There are also more data on nitrate than on ammonium or dissolved organic N (DON) outputs. The data that we have been able to locate on annual average N concentration and output from forested catchments are compiled in Table I. The range in nitrate concentrations is 0.014 to 0.346 mg N/L and outputs range from 0.04 to 2.4 kg N/ha per year. There is some evidence that nitrate concentrations and outputs in the Susquehanna River Drainage and the northwestern portions of the Potomac River Drainage are somewhat higher than in the Virginia portions of the Chesapeake Basin. This may be the result of somewhat higher levels of atmospheric nitrate deposition in northern and western portions of the Basin, or a longer history of high N deposition rates. Nitrate concentrations from 0.5 to 1.0 mg N/L are now common in some streams of the Catskill Mountains of New York (Stoddard 1991) and streams at the Fernow Experimental Forest in West Virginia (Adams et al. 1993). However, all

Table I. Nitrate concentrations and fluxes from forested catchments in the Chesapeake Bay Drainage Basin. All concentrations are discharge-weighted (unless denoted by *) and expressed as μg N/L. All fluxes are expressed as kg N/ha/y.

Catchment	Physiographic province	Catchment area (ha)	Mean Conc. (μg N/L)			Mean Flux (kg N/ha/y)			Period	Ref.**
			NH_4	NO_3	DON	NH_4	NO_3	DON		
Susquehanna River Drainage:										
Stony Ck. (PA)	Valley and Ridge	5670	36	82	340	0.18	0.40	1.72	85-86	L. Reed
Young Womans Ck. (PA)	Appal. Plateaus	11970	16	310	214	0.07	1.44	1.00	85-89	L. Reed
Leading Ridge (PA)	Valley and Ridge	122	-	16	-	-	-	-	76-87	J. Lynch
Patuxent River Drainage:										
Rhode R. Trib (MD)	Coastal Plain	6.3	64	116	210	0.11	0.20	0.2	81-84	D. Weller
Potomac River Drainage:										
Hauver Br. (MD)	Blue Ridge	550	-	346	-	-	2.1	-	82-92	K. Rice
Bear Br. (MD)	Blue Ridge	98	-	330	-	-	2.4	-	91-92	K. Rice
Fishing Ck. Tr. (MD)	Blue Ridge	104	-	111	-	-	0.6	-	88-92	K. Rice
Mill R. (VA)	Valley and Ridge	303	-	20	-	-	0.07	-	83-92	A. O'Brien
Shelter R. (VA)	Valley and Ridge	36	-	14	-	-	0.04	-	83-92	A. O'Brien
Rappahanock River Basin:										
S. F. Brokenback R. (VA)	Blue Ridge	-	-	41	-	-	0.13	-	83-91	A. O'Brien
344 Western VA streams	Valley and Ridge, Blue Ridge	-	-	14 (80%<100)*	-	-	-	-	spr. 87	R. Webb

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available data for forested catchments in the Chesapeake Basin suggest that nitrate outputs are still quite low, with concentrations generally in the range of 0.08 to 0.35 mg N/L in the Pennsylvania and Maryland portions of the Basin and in the range of 0.01 to 0.04 mg N/L in the Virginia portion of the Basin. Although we have identified data for only one forested catchment in the Coastal Plain and have no data for the Piedmont, we expect that nitrate outputs from forests in these physiographic provinces are similar to those for the highland regions.

Concentrations and outputs of ammonium from forested catchments are consistently low, much lower than for nitrate, and concentrations of nitrate in streams draining forested catchments in this region generally exhibit seasonal variation. Ammonium is both biologically and chemically very reactive and thus its mobility is restricted in forested catchments. Ammonium concentrations ranged from 0.016 to 0.064 mg N/L and outputs ranged from 0.07 to 0.18 kg N/ha per year in our compiled dataset (Table I). The somewhat higher values for the Coastal Plain catchment may reflect somewhat lower geochemical retention of ammonium in sandy soils compared to the clay-rich soils of the highlands. Somewhat higher nitrate levels occur during the non-growing season when forest vegetation is dormant. Stream discharge is also generally greater during the non-growing season, particularly in spring in regions that receive appreciable snowfall. The concentrations presented in Table I are primarily discharge-weighted concentrations; consequently, nitrate concentrations during the non-growing season in forested catchments are probably only slightly greater than the annual averages, whereas nitrate concentrations during the growing season may be considerably lower than the annual averages.

Comparisons of total inorganic N outputs (nitrate plus ammonium) and total N inputs (wet plus dry deposition) indicate that forested catchments in the Chesapeake Basin retain much of the N they receive via atmospheric deposition. Total annual outputs of inorganic N (ranging from 0.1 to 2.4 kg N/ha) are considerably lower than the annual N inputs (6 to 13 kg N/ha). Forests in the Chesapeake Basin may be leaching anywhere from 1% to 20-40% of their atmospherically deposited N. Low nitrate concentrations in stream water and low nitrate outputs relative to N inputs are indicative of stages 0 or 1 of watershed N saturation, as presented by Stoddard (1994).

Concentrations and outputs of DON are generally similar to those of nitrate, although we have fewer measures of DON (Table I). Concentrations of DON range from 0.21 to 0.34 mg N/L and outputs range from 0.2 to 1.7 kg N/ha/y. It is generally believed that a considerable portion of the DON output is biologically refractory and will not be remineralized to ammonium and nitrate very rapidly. However, there is considerable uncertainty as to exactly what fraction of the DON is indeed refractory and what fraction will be remineralized within streams and rivers and thus contribute to the N pool available to algae in the Chesapeake Bay.

Comparison between the empirical measurements of N concentration and output from forested catchments and the HSPF simulation predictions indicates that nitrate concentrations and outputs are overestimated by the model. The largest discrepancy between the model results and the empirical data is for the Susquehanna River Drainage. There is general agreement between the model and measured values for ammonium concentration and outputs. However, the model underpredicts the concentrations and outputs of DON by 5-10 times,

based on the measured values. Comparison between the DON in the model and the measured values must be treated with caution because the model does not really simulate all DON exported from forests. [DON output in the model is calculated as a small fraction (5.3%) of the output of biochemical oxygen demand and thus only represents the fraction of DON that is associated with readily oxidized organic material.]

Forest Nitrogen Pools and Fluxes Relevant to HSPF. The purpose of this section is to summarize information on nitrogen cycling in forests, with particular attention to "big-science" research that has resulted in a synthesis of data from numerous temperate forest ecosystems. Two examples of this type of research were the International Biological Program (1964-1974) that provided data on 116 forest research sites around the world (Reichle 1981), and the recently completed Integrated Forest Study (Johnson and Lindberg 1992) that included 17 forest research sites (16 in North America). This summary focuses on nitrogen pools and fluxes that are important to parameterizing the proposed module to the HSPF model for simulating nitrogen exports from forest watersheds (Figure 2). Estimates for default values for HSPF state variables and fluxes have been derived for use in lieu of site specific data on forest nitrogen cycling.

Simple models that are used to summarize information on forest nitrogen cycling usually consist of no less than 3 or 4 state variables: organic-N in mineral soil, nitrogen in forest floor organic matter, available nitrogen in soil and forest floor layers, and total plant N (see for example, Cole and Rapp 1981, Aber et al. 1983, Nadelhoffer et al. 1985). The primary inputs to forests are atmospheric nitrogen deposition and nitrogen-fixation. The principal exports are leaching losses from soil and denitrification. Nitrogen inputs in deposition, retention of ammonium-N by soils, the release of organically bound nitrogen through mineralization, and uptake of available nitrogen by plant roots and microorganisms each play some role in determining the amount of nitrate leaching from forest soils.

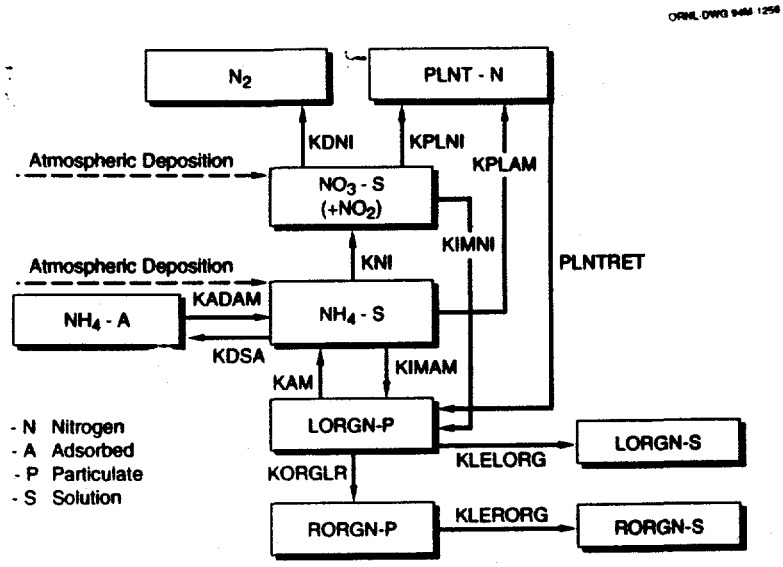

Figure 2—Proposed module of forest nitrogen transformations for HSPF model.

Nitrogen deficiency is a common condition in forests, and nitrogen deficient forests tend to retain and conserve nitrogen entering the ecosystem via fertilizer applications or atmospheric nitrogen deposition. However, with continuous long-term nitrogen inputs, forests may progress through several stages to a condition of "nitrogen saturation" (Stoddard 1994). Nitrogen saturation occurs when nitrogen sources in the ecosystem exceed nitrogen sinks, i.e., when the combined inputs from nitrogen mineralization and atmospheric deposition exceed the nitrogen uptake capacity by plants and soil microorganisms (Stoddard 1994).

As a minimum requirement, we would expect HSPF simulations of nitrogen outputs from forests to mimic changes in streamwater nitrate concentrations that accompany the four stages of nitrogen saturation described by Stoddard (1994). In the first stage (stage 0), streamwater nitrate concentrations are very low during the growing season (less than those measured in deposition), and may exhibit transient spring maximums that are attributable to the flushing of nitrogen stored in soils or snowpacks during the dormant season. The second stage of nitrogen saturation (stage 1) is similar to stage 0 in that demands for nitrogen by biota still exceed available supplies; consequently there is little or no nitrate leaching during the growing season. However, in stage 1 the onset of nitrogen limitation is delayed during the spring giving rise to an amplification in the seasonal pattern of stream water nitrate concentrations (i.e., concentrations during the spring are elevated and exceed concentrations typically measured in atmospheric deposition). During stage 2, there is a further reduction in nitrogen limitations to forest productivity during the growing season and nitrogen sources begin to exceed nitrogen sinks. Net nitrification is an important component of net soil nitrogen mineralization in stage 2 forests. The seasonal pattern in streamwater nitrate concentrations, which is typical of stage 0 and stage 1 forests, begins to disappear in stage 2. Stage 2 forests are characterized by elevated concentrations of nitrate in streamwater at baseflow and in groundwater. In the final stage of nitrogen saturation (stage 3), nitrogen demands by biota are exceeded at all times and biological controls on nitrogen losses from the ecosystem cease to function. Nitrogen sources (atmospheric N deposition and mineralization) exceed nitrogen sinks (biological demands) even during the growing season. At this stage, streamwater nitrate concentrations are chronically elevated above those observed in atmospheric N deposition and hydrologic losses of nitrogen exceed nitrogen inputs to the forest. Various estimates suggest that long-term deposition rates exceeding 5 to 20 kg N/ha per year may result in nitrogen saturation and nitrate leaching from forests (Stoddard 1994).

Nitrogen outputs from forests are not a simple linear function of atmospheric nitrogen inputs. There is a large variation in estimated nitrogen outputs from forests where nitrogen inputs are estimated to be $\leq$10 kg N/ha per year. The absence of a simple relationship between nitrogen input and nitrogen output is supported by results from the Integrated Forest Study where only 11% of the variation in nitrate leaching from 16 sites could be explained by a simple linear regression against atmospheric nitrogen input (Van Miegroet et al. 1992). However, Van Miegroet et al. (1992) found that 67% of the variation in nitrate leaching across the sites was explained by a combination of atmospheric nitrogen inputs, nitrogen uptake by overstory trees, and net nitrogen mineralization in the top 10 cm of mineral soil.

Amounts of nitrogen in aboveground (wood and leaves) and belowground (root) tree biomass were reviewed for temperate deciduous and coniferous forests. Nitrogen sequestered

in belowground tree roots was estimated for forest stands during the Integrated Forest Study but not during the International Biological Program. Based on 17 sites where data on both aboveground and belowground tree nitrogen are available, the mean (± SD) ratio of belowground nitrogen to aboveground nitrogen in forest trees is 0.27 ± 0.13. For sites where no estimate of belowground nitrogen in tree roots is available, the amounts of belowground tree nitrogen were predicted using this ratio. This estimation method resulted in reasonable agreement between the predicted total amounts of nitrogen in trees and reported total amounts of nitrogen in trees for sites that were part of the Integrated Forest Study. Temperate coniferous and deciduous forests do not differ markedly in the amount of nitrogen in aboveground tree biomass (Cole and Rapp 1981), therefore no distinction has been made here between the two forest types. We estimated the mean (± SD) amount of nitrogen in aboveground tree biomass at 456 ± 199 kg/ha. The determination of nitrogen amounts in belowground root biomass is difficult and subject to large uncertainties. The mean (± SD) predicted amount of nitrogen in tree roots based on the data reviewed is 123 ± 54 kg/ha. Based on the literature, we predict the mean (± SD) amount of nitrogen in total tree biomass in temperate forests is 579 (± 253) kg/ha.

Most of the nitrogen in forest ecosystems is sequestered as organic-N in soil. The mean (± SD) amount of nitrogen in the forest floor of 41 forests was 735 ± 643 kg/ha, an amount that was on average 11% of the total nitrogen in the forest floor and mineral soil combined (6484 ± 2782 kg N/ha). The mean turnover time of forest floor nitrogen is 5.5 and 17.9 years, respectively, in temperate deciduous and temperate coniferous forests (Cole and Rapp 1981). Part of the total soil nitrogen is included in soil microbes. Holmes and Zak (1994) determined that ≈170 kg N/ha was tied up in soil microorganisms in temperate northern hardwood forests. This nitrogen pool was relatively constant over the growing season and did not change with seasonal changes in rates of net nitrogen mineralization.

Currently, soil nitrogen transformations and plant uptake are simulated for each of the four soil layers in the HSPF model (surface, upper soil, lower soil, and groundwater). The layers are connected by hydrologic flows. For the forest module, it has been recommended that the surface and upper soil layer be combined into a single, biologically active soil layer which corresponds to the O + A or O + E soil horizons. The lower soil layer corresponds to soil below the A or E horizon. Nitrogen transformations in soil and plant uptake by forests will be modeled only for the upper soil layer in the HSPF forest module. The distribution of nitrogen between surface soil layers (O + A or O + E horizons) and lower soil layers (below A or E horizon) at low-elevation study sites in the Integrated Forest Study (Johnson and Lindberg 1992) were reviewed. The mean (± SD) amount of nitrogen in the surface soil layers of 10 forests was 2055 ± 927 kg/ha or ≈35% of the total soil nitrogen (5797 kg/ha) to depths ≥45 cm. Based on detailed soils data from the Integrated Forest Study and estimates of total nitrogen in the forest floor and mineral soil the nitrogen content of surface forest soils is estimated to be on the order of 2200 kg/ha.

Although the situation is complex, recent work by Stump and Binkley (1993) supports the following three assumptions about controls on net nitrogen mineralization in forest soils: (1) net nitrogen mineralization in the forest floor is correlated with that in total soil (forest floor plus mineral soil), (2) litter chemistry controls net nitrogen mineralization rates, and (3) the

lignin:nitrogen ratio controls the litter decomposition rate. Based on data for low-elevation forests from the Integrated Forest Study, ≈1 to 7% of the organic nitrogen in soil is mineralized each year (Van Miegroet et al. 1992). For the literature reviewed, the mean (± SD) for net nitrogen mineralization in forest soils is 49 ± 48 kg N/ha per year. Based on an estimated total soil nitrogen content of 2200 kg/ha in the surface soil layers (O + A horizon), ≈2 to 7% of the organic-N in forest soil is mineralized each year.

In HSPF, soil nitrogen mineralization is modeled as two processes: ammonification (KAM: conversion of organic-N to ammonium) and nitrification (KNI: conversion of ammonium to nitrate). Relative nitrification (i.e., the fractional contribution of nitrification to total nitrogen mineralization) is high in forest soils that are losing nitrogen via nitrate leaching. Otherwise, relative nitrification in forest soils is highly variable and difficult to predict. The current paradigm, which is probably oversimplified, is that nitrification rates are controlled by competition between soil microorganisms and plant roots for available soil ammonium (Riha et al. 1986, Johnson 1992). In any case, nitrification is expected to be a small fraction of total soil nitrogen mineralization in nitrogen deficient forest ecosystems. Relative nitrification can be adjusted in HSPF model simulations, within the limits set by estimates of net soil nitrogen mineralization, to yield realistic values of soil solution nitrate concentrations.

In both the Integrated Forest Study and the International Biological Program, nitrogen uptake was determined only for aboveground parts of forest biomass (i.e., nitrogen assimilated for leaf and wood production). The mean (± SD) nitrogen uptake allocated to aboveground forest biomass in temperate forest ecosystems is 58 ± 29 kg/ha per year. There are fewer studies specific to nitrogen uptake for the production of tree roots, but those studies indicate that the mean allocation of nitrogen uptake to root production (64 kg N/ha per year) is comparable to that for aboveground biomass. Where nitrogen uptake allocated belowground is unknown, the total nitrogen uptake is estimated as twice the nitrogen allocated aboveground. Based on this rough method of estimation, the mean (± SD) total nitrogen uptake by forest biomass is 116 ± 58 kg/ha per year. Although the fraction of nitrogen uptake allocated belowground is not constant across temperate forest ecosystems (Joslin and Henderson 1987), the assumption that nitrogen uptake is equally apportioned to aboveground and belowground biomass is supported by studies of 9 forest stands in Wisconsin where apparent total nitrogen uptake by vegetation ranged from 47 to 143 kg/ha per year (Nadelhoffer et al. 1985).

In HSPF, the uptake of nitrogen by plants from soil is modeled separately for available soil ammonium and available soil nitrate. It is difficult to ascertain the relative importance of ammonium and nitrate as nitrogen sources for the production of forest biomass because plants vary widely in their utilization of different forms of inorganic soil nitrogen. For forests where soil nitrification is a small fraction of total net nitrogen mineralization (i.e., forests in stage 0 of nitrogen saturation), it can be assumed that most of the total nitrogen uptake originates from the available soil ammonium pool (NH4-S). Even so, the root uptake of soil ammonium may be less than that for nitrate under conditions of low soil moisture (i.e., drought) - even when plants exhibit a preference for soil ammonium (Gijsman 1991). The contribution of soil nitrate to total nitrogen uptake is expected to progressively increase for forests at stage 1, 2, or 3 of nitrogen saturation. Studies by Nadelhoffer et al. (1984) indicate that for nine forests in southern Wisconsin (where measurable net nitrification occurred), soil nitrate supplied between

70 and 100% of the annual nitrogen uptake by forest biomass. In the absence of a better alternative, it is logical to model the fractional contribution of available soil ammonium and soil nitrate to total nitrogen uptake as a function of the stage of nitrogen saturation.

The return of plant nitrogen to soil is modeled as a single flux (PLNTRET) in HSPF although the flux is comprised of the following three separate processes: (1) aboveground litterfall (leaf fall, woody litterfall, and reproductive parts), (2) belowground nitrogen returns (root mortality and decomposition), and (3) leaching of canopy nitrogen by precipitation. For the literature reviewed, the mean (± SD) return of nitrogen in aboveground litterfall is 35 ± 18 kg/ha per year. Fewer data are available on nitrogen losses from tree roots, but studies indicate a mean (± SD) belowground nitrogen return of 50 ± 19 kg/ha per year. When aboveground and belowground fluxes have been determined in the same study, the average ratio of belowground:aboveground nitrogen return is ≈1.6. Using a factor of 1.6, the mean (±SD) total return of plant nitrogen to soil via litterfall and losses from roots in temperate forests was predicted to be 92 ± 46 kg/ha per year. This estimated return was somewhat less than our estimated total nitrogen uptake by forests of 116 ± 58 kg N/ha per year. Leaching of plant nitrogen is negligible relative to other aboveground and belowground nitrogen fluxes to soil.

Nitrogen fixation, denitrification, and foliar uptake of atmospheric nitrogen deposition are not expected to be important in forests of the Chesapeake Basin. Nitrogen fixation is not included as a process in the current version of HSPF. Denitrification is modeled as a single flux (KDNI) from the surface soil in HSPF. Omission of this flux in HSPF simulations of nitrogen losses from forests is not expected to contribute to large errors in hydrologic nitrogen outputs [Goodroad and Keeney (1984) reported 0.2 to 2.1 kg N/ha per year]. Lovett and Lindberg (1993) conclude that in most cases canopy uptake of atmospheric nitrogen deposition is small (1 to 12 kg N/ha per year in the Integrated Forest Study).

We cannot forecast the type or extent of forest disturbance that may impact nitrogen losses from watersheds in the Chesapeake Basin. However, we can outline the theoretical framework for forest controls on nitrogen losses and thereby contribute to defining various scenarios for model simulations with the forest module of HSPF. Devegetation by clearcutting, fire, or hurricane affects evapotranspiration, hydrologic fluxes, and soil temperatures in ways that generally increase nitrogen export from forest ecosystems. In undisturbed forest watersheds, the annual amount of soil nitrogen mineralized, or assimilated by vegetation, or returned from vegetation to the soil far exceeds annual ecosystem nitrogen losses (Swank 1986). By comparison, many studies demonstrate that forest disturbance can measurably increase ecosystem losses of nitrate-nitrogen (Likens et al. 1970; Vitousek and Melillo 1979; Swank 1984; Swank 1986).

Vitousek and Melillo (1979) reviewed studies on the effect of various disturbances (mostly clearcutting) on nitrate export from forests. In Vitousek and Melillo's review, the range of nitrate-N concentrations in drainage water from 23 disturbed forests was 0.01 to 25 mg/L as compared to 0.001 to 1.7 mg/L in controls. Vitousek and Melillo's summary of data from 11 studies also indicates that nitrogen exports range between 0.6 to 124 kg nitrate-N/ha per year from disturbed forests versus exports of 0.03 to 4.5 kg nitrate-N/ha per year from control forests. Nitrogen losses as a result of disturbance are highly variable because processes like

nitrogen uptake by regrowth vegetation, nitrogen immobilization, lags in nitrification, and a lack of water can act independently or in combination prevent hydrologic losses of nitrate from disturbed forests (Vitousek and Melillo 1979; Vitousek et al. 1982). Disturbances that reduce plant uptake (like clearcutting or defoliation by herbivorous insects) remove biological controls on the forest nitrogen cycle. Studies at Coweeta, North Carolina, demonstrate that partial defoliation of forests by herbivorous insects is followed by increased stream export of nitrate nitrogen (Swank 1986). Consequently, increased nitrogen exports from forests following gypsy moth infestations is expected. Disturbances that decrease the uptake of soil nitrogen by vegetation can, in effect, induce a temporary condition of nitrogen saturation where inputs from atmospheric deposition and soil nitrogen mineralization exceed forest demands (Vitousek and Melillo 1979). Soil nitrogen mineralized in excess of forest demands is then at risk of loss to streamwater or groundwater. Although forest disturbance by be temporary, elevated streamwater nitrate concentrations may persist for many years after disturbance (Swank 1984).

DISCUSSION AND CONCLUSIONS

We conclude that the current AGCHEM module for N cycling and loss from agricultural systems in HSPF can be modified to allow simulation of N fluxes within and outputs from forested ecosystems (Figure 2). The major modifications to the AGCHEM module are:

- Subdivision of the ORG-N compartment into four soil organic N compartments, two consisting of particulate organic N pools (labile = LORGN-P, refractory =RORGN-P) and two soluble organic N pools (LORGN-S, RORGN-S) resulting from leaching of the particulate pools, and
- Addition of a flux (PLNTRET) from the plant N compartment (PLNT-N) to the soil labile organic N compartment (LORGN-P).

The PLNTRET flux consists of both aboveground and belowground return of N to the soil. Although the modified AGCHEM module for forests is a gross simplification of the complex cycling of N in forested ecosystems, we believe that it will allow a reasonable simulation of current N exports from forests to drainage waters in the Chesapeake Basin given the data available to parameterize it and current state of knowledge.

The functional formulation of the fluxes in the modified AGCHEM module for forests are somewhat different from the agricultural version. Because plant and microbial uptake of inorganic N from forest soils is extremely efficient but can be saturated at high N concentrations, a saturation kinetics formulation of these N uptake fluxes (KPLNI, KPLAM, KIMNI, KIMAM) is preferred. The maximum plant uptake rate (U_{max}) should be a function of season (uptake is limited to the growing season) and overall growth rate of forest vegetation (function of forest maturity, insect attack, etc.). Most of the total uptake of N by plants is returned to the soil each year. Although a portion of this plant N return (PLNTRET) occurs over the entire year (particularly the belowground portion), it can be approximated by an annual pulse to the labile soil organic N pool in October. Mineralization (KAM) and nitrification (KNI) can be approximated as first-order rates as in the agricultural AGCHEM module. Denitrification in well-drained forest soils is generally very low and can be set to zero for most of the upland forests to be considered. Ammonium partitioning and organic N

leaching fluxes can be approximated with partitioning functions based on total compartment sizes. Conversion of labile organic N to refractory organic N is a slow process that can be approximated with first-order kinetics.

For many of the compartment sizes and fluxes of the modified AGCHEM module for forests there are only very approximate and widely ranging estimates for parameterization that can be taken from the literature. Storage of N in forest vegetation (PLNT-N) is probably on the order of 400-800 kg N/ha. Annual total plant uptake is probably 80-120 kg/ha, and the return of plant N to the soil is about 80-90% of total plant uptake. Maximum plant uptake rates (U_{max}) and return could be up to twice these values. For the purposes of simulating N loss from forest catchments to surface waters, the most important compartments to parameterize correctly in the modified AGCHEM module are the pools of soluble nitrate (NO_3-N), ammonium (NH_4-N), and organic N (LORGN-S, RORGN-P). Fortunately, it is these compartments for which there is at least some data in the form of discharge-weighted mean annual concentrations in streams draining forested catchments (Table I). Thus, the process of parameterizing the modified AGCHEM module for forests in the Chesapeake should be directed toward achieving realistic concentrations of stream nitrate, ammonium, and DON. The primary features in the dataset to be simulated correctly are the seasonal variation and annual flux of N species in stream water. However, while this parameterization process will allow simulation of current N outputs from forests in the Chesapeake Basin, it will be of limited value for predicting long-term N outputs because we as yet do not understand the controls of N loss from forests and their relationship to N deposition, forest maturation, and disturbance. More mechanistic, predictive models of N cycling and output from forests are currently being developed at the University of Virginia and the University of New Hampshire under funding from EPA. However, these models are not yet available.

HSPF is a model that contains a high degree of specificity for instream processes but a low degree of specificity for terrestrial landscape processes. Hydrologic models often tradeoff temporal and spatial specificity and/or amount of detail between different processes being simulated. HSPF can be run with short time steps and routing to capture hydrologic patterns of stream flow and concentration that are driven by precipitation events. Data needs and computing time for HSPF are extensive. HSPF is a continuous simulation program and requires continuous data to drive the simulations—rainfall is required with evapotranspiration, temperature, and solar intensity desirable. However, often the possible detail of short time steps is not utilized for management decisions where seasonal or annual averages are adequate. HSPF models terrestrial landscape processes very coarsely as it uses large subbasins (63 model segments with an average area of 260,300 ha for the Chesapeake) and ignores spatial pattern of land cover on the subbasin. The implementation of spatially explicit nonpoint-source watershed models is advancing rapidly with the development of Geographic Information Systems and more powerful computers. A spatially distributed model may best serve the Chesapeake Bay Program for its long-term needs of local watershed management.

The current implementation of HSPF for the Chesapeake Bay drainage is to evaluate management options with regard to the mandated 40% reduction of controllable phosphorus and nitrogen to the tidal Bay by the year 2000. These reductions will have to come primarily from improved management of land use, e.g., best management practices for agriculture and

silviculture, installation of buffer strips, and reclamation or installation of wetlands and riparian zones. Landscape ecology research has shown that the position of pattern of land use on a watershed is important for modeling nonpoint-source pollutants (Peterjohn and Correll 1984, Osborne and Wiley 1988, Hunsaker et al. 1992, Weller et al. 1993). Thus for the long-term the Chesapeake Bay Program should consider a spatially explicit modeling approach. At least two such modeling activities are ongoing in the Chesapeake Basin (CEES 1993, Weller 1993).

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