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Impacts of Rising Nitrogen Deposition on N Exports from Forests to Surface Waters in the Chesapeake Bay Watershed

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ABSTRACT / In this study, we applied a process-based forest ecosystem model, PnET-CN, to estimate inorganic N (nitrate) loading and retention under chronic increases of atmospheric N deposition in the Chesapeake Bay (CPB) watershed. The results indicated that the average N leaching loss from forested lands in the CPB watershed is 1.23 kg N ha^{-1} y^{-1} at current N deposition levels, suggesting approximately 88% of N is retained by forest ecosystems. Total dissolved inorganic N exported from the forested watersheds was 11,617 Mg N yr^{-1}. The predicted rates of the nitrate losses are well validated by the United States Geological Survey–National Water-Quality Assessment data measured from the gauged stations for forested drainages within the CPB watershed, and are also compatible with the field data of N loads associated with forests in the CPB watershed. If N deposition were twice current levels, the retention by forests would drop to 81%. Total N leaching loss to surface waters would then increase more than threefold. A nonlinear increase in N loads from forests under the extreme scenario of atmospheric N deposition shows the symptom of N saturation and an accelerated decline of forest functioning to retain atmospheric N deposition in the CPB watershed with rising levels of nitrogen deposition.

Human alternation of the natural N cycle has resulted in significant consequences in terrestrial, freshwater and marine ecosystems (Vitousek and others 1997). For the Chesapeake Bay (CPB); which is one of the largest and most productive estuaries in the world, excessive N loads to the Bay during the past decades have caused serious eutrophication and degradation of water quality (Castro and Driscoll 2002). It is estimated that total N inputs to CPB watershed are now six to eightfold greater than during precolonial times (Castro and Driscoll 2002). Nitrogen inputs to the CPB watershed originate from many sources. Atmospheric N deposition (i.e., nonpoint sources derived from emissions of nitrogen oxides from automobiles, and ammonia emissions from agriculture, urban areas, and industries) has drawn particular attention in recent years, because it may account for as much as 25 to 80% of the total N entering the bay (Sheeder and others 2002).

Forest ecosystems accumulate, store, and redistribute N within watersheds (Likens and Bormann 1995). Numerous studies indicate that forest ecosystems can function as a filter of atmospheric N deposition to stream water (Wickham and others 2002, Jones and others 2001). Forest covers approximately 56% of the CPB watershed (Gardner and others 1996). Research studies of the N cycle in forest ecosystems indicate that land-use history has prominent impacts on forest N cycling that could last for many decades (Aber and others 1997, Ollinger and others 2002a). Although most forests in North America remain N limited, several studies recognize that the forests in the Mid-Atlantic regions appear to have symptoms of N saturation because of chronic N input from atmospheric deposition (Fenn and others 1998). The forest saturation of N in the CPB watershed would exacerbate the existing problem of deteriorating water quality, eutrophication, and toxic effects on freshwater biota (Fenn and others 1998, Gardner and others 1996, Castro and Driscoll 2002).

Several watershed models were developed to assess N loading to the CPB from an atmospheric deposition perspective (Valigura and others 2000). Most models are statistical based and are limited in spatial-extrapolation capability and the ability to incorporate the impacts of dynamic changes of forests on nitrogen exports. Process-based biogeochemistry models can simulate general dynamics of nutrient cycles for forested landscapes and can be used to evaluate impacts of land-use change and processes of forest ecosystem func-

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tions on the N loading to the water system at a broad scale. Because many factors affect a forest's capacity to retain atmospheric N deposition—such as forest development stage, forest types, soil features, climate, and topographical characteristics—it is likely that N leaching loss patterns across a region will be spatially diverse. Biogeochemistry models, in contrast to statistical-based models, are advantageous not only because they simulate interactions among carbon (C), N, and water cycles in forest ecosystems, but also because they can simulate spatial variations of N cycling along with time dynamics (VEMAP 1995, Schimel and others 2000). Biogeochemistry models can be used to project potential changes in N cycling under the extreme scenarios of atmospheric N deposition and serve as a diagnostic tool for air pollution regulations because the models are developed based on the understanding of mechanistic processes of forest ecosystems (McGuire and others 1995, 1997, Aber and others 1997).

In this study, we apply the process-based ecosystem model PnET-CN (Aber and Driscoll 1997, Ollinger and others 2002b) to evaluate the impacts of increased atmospheric N deposition on N leaching losses and retention rates of forests in the CPB watershed. We also diagnose the responses of forest N leaching exports and retention rates under an extreme scenario of doubling current N deposition. The main objectives of this study are (1) to provide estimates at a regional scale on the N exports from forests of the CPB watershed using a process-based biogeochemistry model, which is independent from other estimates published in peer-reviewed literature; and (2) to predict the potential capacity of forest N retention with the extreme scenario of high N deposition (2X), and assess forest status of N saturation in the CPB watershed.

Methods and Data

The PnET-CN Model

A process-based forest ecosystem model, PnET-CN (Aber and Driscoll 1997), was adopted and modified to estimate N leaching losses from the forests in the CPB watershed. PnET-CN simulates carbon, nitrogen, and water cycles of forest ecosystems at a monthly-time- step. The model contains the features of historical land-use impacts and a complete N cycle. The ecosystem processes and mechanisms built in the PnET model were based on a large amount of research results and conclusions of ecosystem-scaled and long-term experiments (Likens and Bormann, 1995; Aber and Melillo, 2001). The parameters of the model were derived from field studies. Because the model is not calibrated, validation of results against empirical data is an important step to examine reliability and weakness in predictions. The original PnET-CN model was applied to site-level studies and was combined with statistical methods to estimate N cycling at a regional scale in New England (Aber and others 1997). To adjust the model for a regional geographic information system application that handled broad ranges of data variations, we revised the codes of the model and used localized parameters for regional simulations in the CPB watershed (Table 1).

The model assumes that atmospheric N deposition enters forest ecosystems and gathers in the soil N pools. The N available for tissue construction is determined by plant N uptake from the available N soil pools and the ratios of carbon to nitrogen in plant tissues. Nitrogen leaching loss from a forest stand is directly proportional to the nitrate remaining in the soil solution after plant uptake and to the drainage rate. Therefore, N leaching loss is indirectly related to several variables that affect N soil solution and drainage rate such as photosynthesis rate, available N pools, N uptake, and water-holding capacity.

Input Data and Land-Use History

Explicit geographically referenced data are required to run the model. The essential data layers include forest types, monthly minimum and maximum temperature, monthly precipitation, monthly solar radiation, and soil water-holding capacity. The spatial resolution of the model simulation was at 1 km, with forests in the CPB covering approximately 94,513 pixels at this resolution (Figure 1). To match the forest types derived from the Forest Service forest cover types (Zhu and Evans 1994) with the existing plant functional types used by the PnET model, we reclassified forests to northern hardwood, spruce-fir, pine, oak-hickory, and oak-pine using a mosaic approach (Figure 1). The oak-hickory forests are dominant in the CPB watershed and make up 55% of the forest cover. Details about the input data layers used in the model are described in Pan and others (2004).

Information about the land-use history is required for the model simulation. Because of lack of precise information, we assumed, based on a general land-use history in the CPB watershed, that forests in the CPB watershed were established primarily from abandoned farmlands in the 18th century, and that the current forests are recovering from massive harvests of the secondary forests that occurred in the early 1930s. We ran the model for 200 years to fully incorporate the impacts of the cultivating and harvesting on forest ecosystems (Table 1).

Table 1. Initial conditions, inputs and parameters used to run PnET-CN in this study

Model runs	Preconditions of N deposition	CO_2	Run years
Run 1, control	No N input (0.0 g/m^2)	Fixed 280 ppmv	1800–2000
Run 2, scenario	Averaged N deposition (1991–2000) as the level of 2000, ramped from 1930 to 2000.	Fixed 280 ppmv	1800–2000
Run 3, scenario	Doubled N deposition level of 2000, ramped from 1930 to 2000	Fixed 280 ppmv	1800–2000

Vegetation		Parameters[a]						
Modeled types	USFS cover types	AmxA	AmxB	FolmsMx	GDDfolS	GDDfolE	GDDwoodS	GDDwoodE
N. hardwood	Maple–beech–birch	−46	71.5	300	100	900	900	1600
Spruce–fir	Spruce–fir	5.3	21.5	1000	300	1400	300	1400
Oak–hickory	Oak–hickory	−46	71.9	300	100	900	100	900
Pine	White–red–jack pine Slash pine Loblolly pine	5.3	21.5	800	900	1600	900	1600
Oak–pine	Oak–pine			Mosaics of oak (50%) and pine (50%)				

Land-use history		
Agriculture period: 1800–191900		Soil loss fraction: 0.10
Timber harvests:	Intensity	Biomass removed
1800	0.20	0.01
1926	0.80	0.90
1950	0.01	0.01

Geographic information system input data	*Sources*	*Resolution (km)*
Precipitation[b]	ZedX	1
Temperature[b]	ZedX	1
PAR[c]	Algorithm	1
Slope & aspect[d]	USGS	1
WHC[e]	STATSGO	1

[a]The description of the parameters and more parameters used in the model refer to Aber and Driscoll (1997) and Olinger et al. (2002b).
[b] Monthly precipitation, monthly minimum and maximum temperature data were generated by ZedX, Inc. using a mathematical algorithm to the 30-year (1971–2000) climatological station records, which were compiled as the TD3200/3210 datasets by the National Climatic Data Center.
[c]We calculated monthly potential solar radiation based on the trigonometric algorithm given by Swift (1976) that was a function of latitude, slope, and aspect. The ratios of measured to potential solar radiation were from measurement stations in the region (NCDC, www.ncdc.noaa.gov/oa/climate/surfaceinventories.html). Photosynthetically active radiation (PAR) was calculated as the solar radiation of daylight.
[d]The slope and aspect data we used for calculating solar radiation were derived from the DEM data (USGS 1987).
[e]Water-holding capacity (WHC) data at 50-cm soil depth were from the Natural Resources Conservation Service National Soil database (STATSGO http://water.usgs.gov/GIS/metadata/usgswrd/ussoil.html).

N Deposition Data and Scenarios

The wet nitrate (NO_3) and ammonium (NH_4) deposition scenarios were 10-years' averages from 1990 to 1999 at 1-km resolution, generated from wet deposition data (Sheeder and others 2002). The interpolation algorithms are based on concentration data collected at National Atmospheric Deposition Project/National Trends Network monitoring sites, and precipitation data from a denser network of National Atmospheric and Oceanic Administration Cooperative climatic sampling sites. We calculated dry deposition for NO_3-N and NH_4-N using wet/dry ratios that were reported for the watersheds (Valigura and others 2000). To dampen the order of magnitude variation in N deposition recorded in wet and dry years (Figure 2), we used 10-years' averages. For the CPB watershed, the average total N deposition, including the dry and wet deposition, was approximately 10.04 kg N ha^{-1}. Nitrogen deposition varied across the region because of complex landscape features. Higher N deposition generally occurred in the higher elevations and western highland areas of the CPB watershed (Figure 3).

For the scenario of increased N deposition, we assumed that N deposition before 1930 was approximately 20% of the current average level, which is approximately consistent with the rate of anthropogenic fixation of N in terrestrial ecosystems from preindustrial time (Galloway and others 1995), and linearly

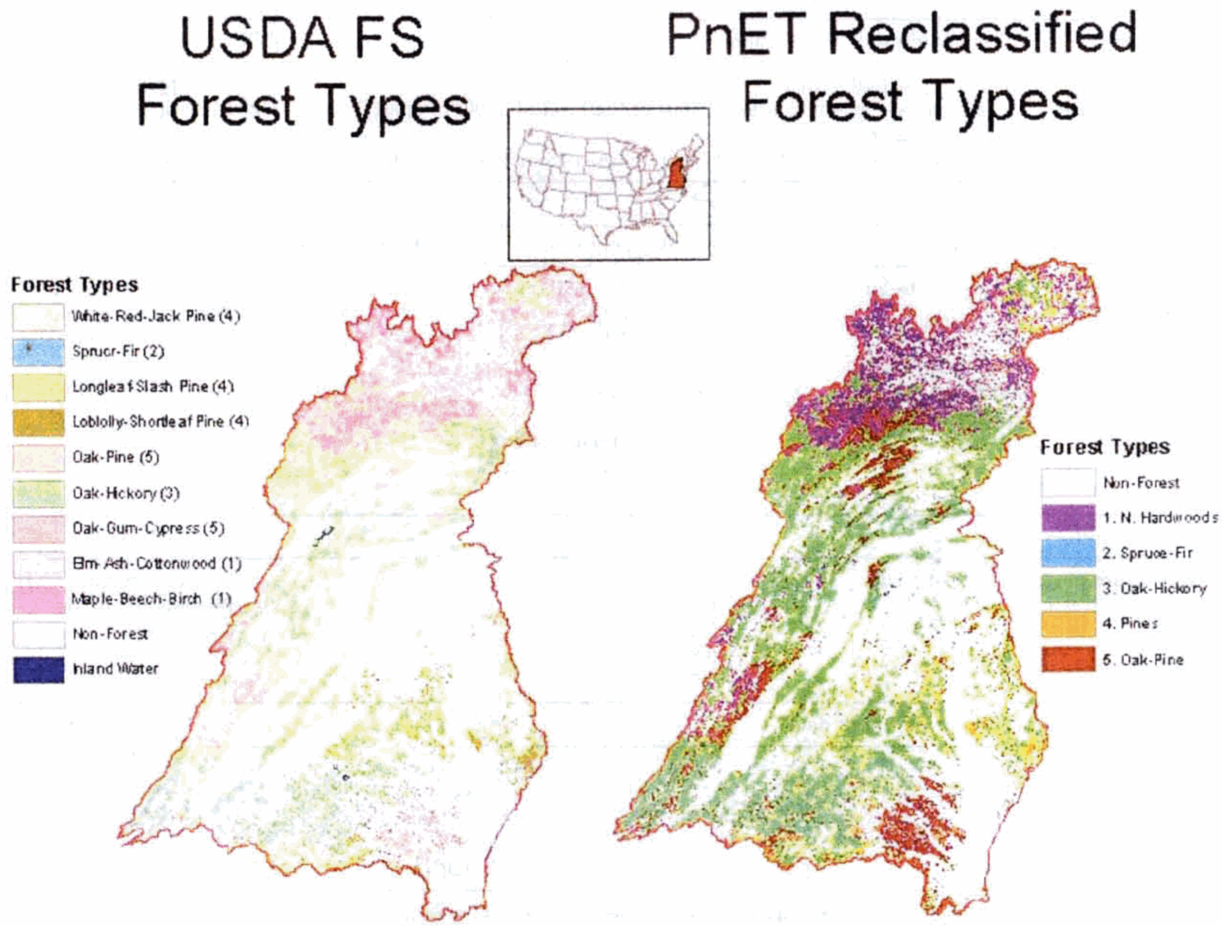

Figure 1. The reclassification of the USDA Forest Service Forest Types to match the PnET-CN functional types.

increased as a constant ramp function. This linear relationship was interpolated for each pixel based on the specific value at the pixel. The same assumption was applied to create the scenario of the doubled N deposition. We designed 3 model experiments to examine the impacts of increasing N deposition on forest ecosystems and watersheds of the CPB: the control (i.e., with no N deposition), the current level of N deposition, and the doubled level of N deposition as inputs for the model simulations (Table 1).

Results and Discussion

Current Forest N Leaching Losses and Retention

Under the control condition (no extra N input from atmosphere), the N leaching losses are near zero (Figure 3). This result indicates a closed and tight N cycle in forest ecosystems. Slight N leaching losses from some areas of northern pine and spruce-fir forests (Figure 4a) reflect that coniferous forests grown in high elevations are less N-conserving (Figure 3a).

The current N deposition scenario starts with low inputs of N deposition to forests (~2 kg N ha^{-1}) before 1930 and linearly rises to the average of current N deposition levels. This N increase coincided with the secondary recovery process of the forests in the area. The current N deposition rate ranges from 5.13 to 16.11 kg N ha^{-1} y^{-1} in the CPB watershed. N leaching losses increased greatly with the extra chronic N inputs to the forests, especially in the upper CPB region characterized by mountains and lower water holding capacity (Figure 3a, 3b, Figure 4b). In the upper CPB, the drainage rate is high, and the area also receives more precipitation and higher N deposition. The nitrogen-leaching rate ranges from 0.18 to 10.59 kg N ha^{-1} y^{-1} with a regional mean of 1.23 kg N ha^{-1} y^{-1} (Table 2). The forest N retention rate for the region is 88%, and total N discharged from forested lands to surface waters

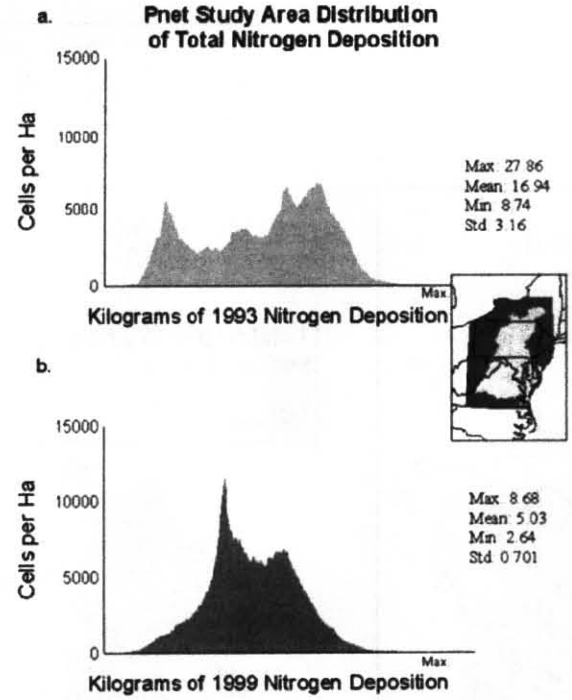

Figure 2. Total wet N deposition (NO_3 $+NH_4$ kg N ha^{-1}) in wet year (1993) (a) and dry year (1999) (b) in the Chesapeake Bay watershed and surrounding area.

in the CPB watershed is estimated at 11,617 Mg N per year (Table 2).

The capacity of forest sites to retain N inputs varies. In general, deciduous forests have higher N retention than coniferous forests, which is consistent with other studies that consider deciduous forests to have tighter N cycling (Waring and Schlesinger 1985, Ollinger and others 2002a, Aber and others 2003). The spruce-fir forests at high elevation have the lowest N retention rate (78%), and oak-hickory forests have the highest (90%). Lower retention rates in coniferous forests may also be related to site conditions. Spruce-fir forests grow mostly in high mountains. Pine and oak-pine forests are found in sandy coastal plain. These forests are characterized by high drainage and N leaching rates. The N retention rate of the northern hardwoods is lower than the oak-hickory forests because the growth in the northern areas may be constrained by lower temperature, and therefore lower N uptake and increased N leaching off.

N Leaching Losses Under the Doubled N Deposition Scenario

With doubled N deposition, N leaching losses increased remarkably across the region (Figure 4c). The regional N retention rate in forests dropped, compared the current level of N deposition, from 88% to 81%. The total N export from the forested lands would be 35,737 Mg N yr^{-1}, which is more than 3 times of the current N loading to the CPB (Table 2).

Different forests responded to the doubled N deposition at different rates. Spruce-fir forests decreased by nearly 22% in N retention rate, which shows the symptoms of N saturation. This result is consistent with experimental studies that indicate particular severe N saturation symptoms in high-altitude spruce-fir ecosystems in the Appalachian Mountains (Fenn and others 1998). The retention rate in pine forests decreased by 12%, oak-pine forest 9%, northern hardwood forests 6%, and oak-hickory forests 5% (Table 2). Coniferous forests apparently leach more N than deciduous forests because they are less N demanding and less sensitive to additional N inputs, indicated by N fertilization experiments (Reich and Schoettle 1988, Magill and others 2000).

The N exports from forests to surface waters could increase greatly under the extreme scenario of doubled N deposition, but the increase is not linear. This result implies that N leaching losses to groundwater or surface runoff could be greater as forest ecosystems approach a saturated status with rising levels of nitrogen deposition. Several studies recognize the function of forests alleviating N nutrient loads to water systems (Wickham and others 2002, Jones and others 2001). Our study shows that the capacity of forests to retain N could quickly decline if N deposition continues to rise to a higher level. The results from the chronic N addition experiments in the coniferous and mixed hardwood forests conducted at the Harvard Forest show that the tripled N additions (15 g m^{-2} yr^{-1} vs. 5 g m^{-1} yr^{-1}) caused four- to sixfold higher N leaching losses in the stands after 9 years of continuous N inputs even though the coniferous and hardwood forests had totally different response patterns of N leaching loss over time (Magill and others 2000). Although the spatial and timing scales and the magnitude of N additions in the fertilization experiment are very different from our modeled N deposition experiment for the CPB watershed, both consistently illustrate a nonlinear increase in nitrate leaching loss after a long-term excess N addition to forests.

Validation and Comparison with Other Estimates

One important objective in modeling research is to use ground survey data or data of other estimates to validate or compare with the model results. The validation is a way to evaluate the uncertainty in models and provides a rigorous testing of model accuracy (VEMAP

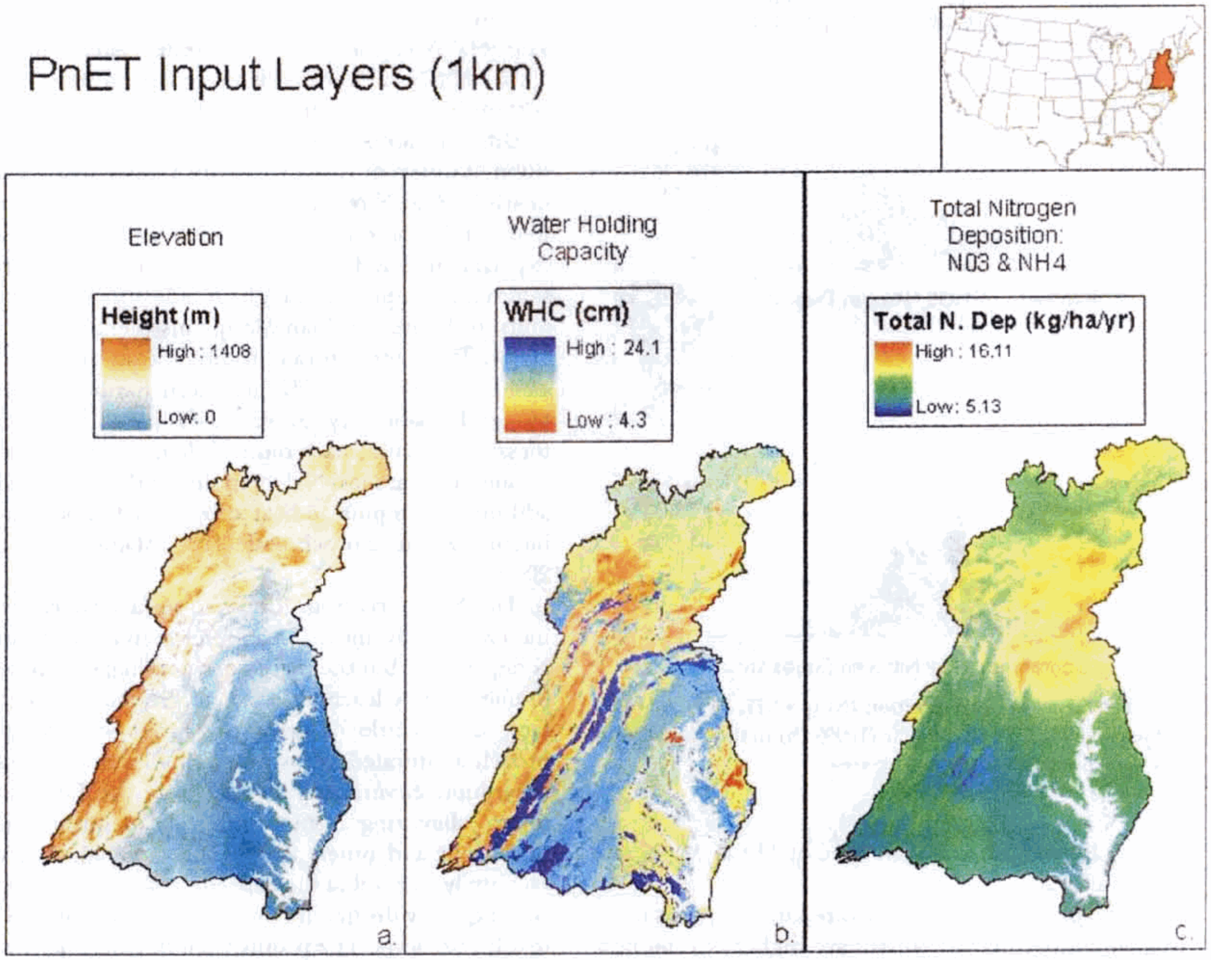

Figure 3. The maps of elevation (a), water-holding capacity (WHC) (b), and total atmospheric N deposition (c) of the Chesapeake Bay watershed. The resolution is 1 km.

1995, Aber and others 1995). For this study, we used the database of the U.S. Geological Survey (USGS) and data from the literature to validate/compare our model results. These data are either from measurements for the whole basin or for small watersheds in or near the CPB, or from estimates made by other models. Although the comparisons may not be entirely rigorous, it provides general information about the model performance.

Validation using the USGS-NAQWA gage station data. We derived the data from the USGS National Water-Quality Assessment (NAQWA) program (http://water.usgs.gov/naqwa/), which are nitrate fluxes measured from gauge stations for 118 drainages within the CPB watershed. We excluded gauge stations from non-forest areas within the CPB. The measured data were averaged for the recording years. We converted the measured nitrate fluxes to the mean N loss rates based on the mean annual stream-flow and drainage areas. Meanwhile, we aggregated the modeled values of nitrate losses for the corresponding drainages and compared them with the USGS data (Figure 5). The modeled values are generally well validated by the measured data, but have a narrower range of variations. Even though these drainages are mostly located in forest areas, they also include certain portions of other land-use types such as agricultural lands. It is not surprising to have greater variations in the gage station measurements that may reflect effects of N loads from other land-cover types (non-forests). The model slightly overestimated N losses in most of the drainages, likely because the model predictions were for late 1990s, whereas the NAQWA data were mostly measured between 1970s and 1980s. The increasing atmospheric

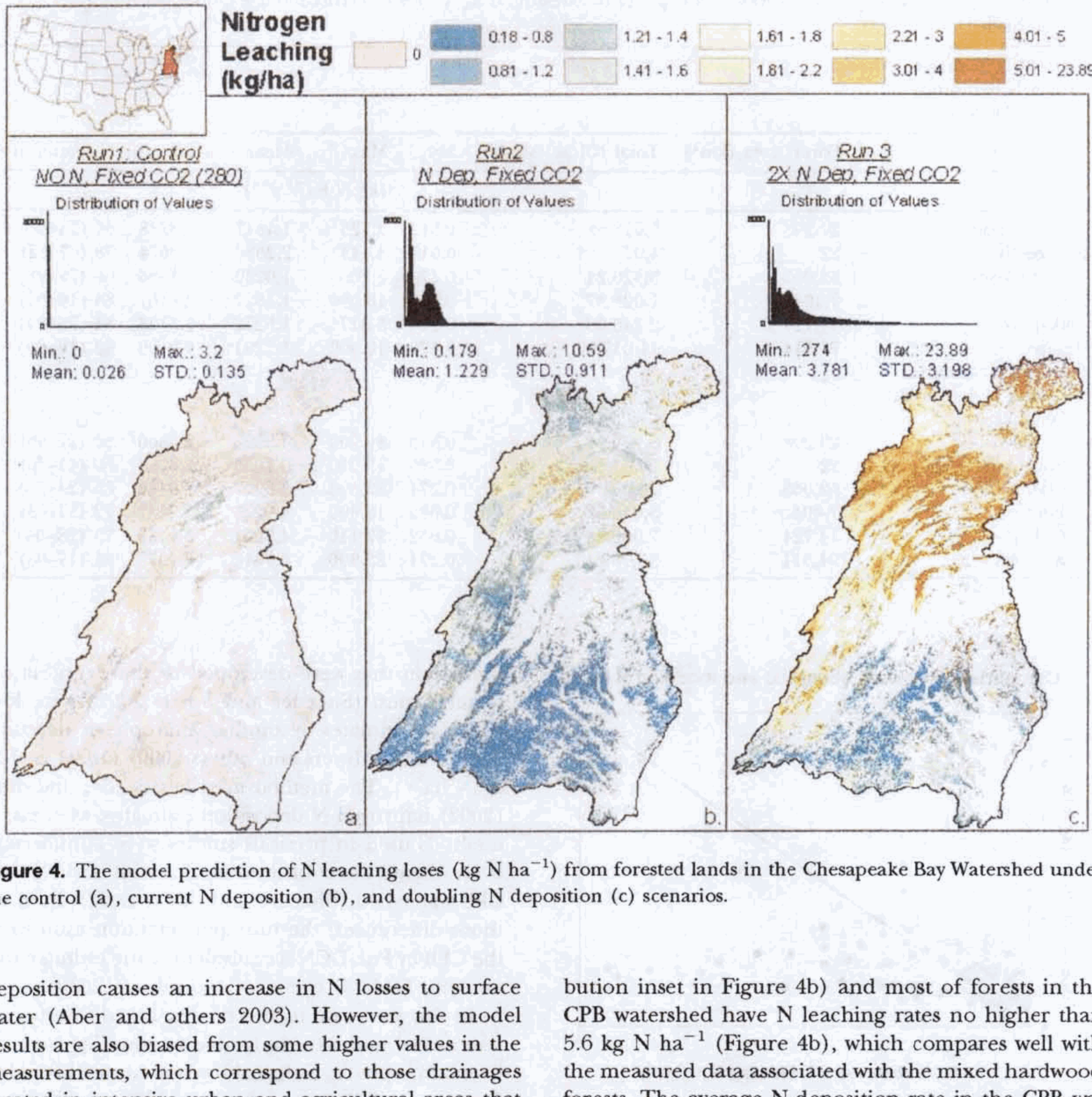

Figure 4. The model prediction of N leaching loses (kg N ha^{-1}) from forested lands in the Chesapeake Bay Watershed under the control (a), current N deposition (b), and doubling N deposition (c) scenarios.

deposition causes an increase in N losses to surface water (Aber and others 2003). However, the model results are also biased from some higher values in the measurements, which correspond to those drainages located in intensive urban and agricultural areas that generally have lower N retention capacity than forested lands (Figure 5, Table 3).

Comparison with the measured data in forested lands. For forests in the CPB watershed, the predicted N leaching rates range from 0.21 to 10.59 kg N ha^{-1} y^{-1} for coniferous forests; and 0.18 to 5.58 kg N ha^{-1} y^{-1} for hardwood/mixed forests, compared to the measured losses of 0.26–5.15 kg N ha^{-1} associated with mixed hardwood forests in or near the CPB watersheds (Table 4). PnET-CN predicted some pine forests to have much higher N leaching rates (Table 2), which are likely associated with edaphic conditions of sites such as sandy soils. However, these sites are few (see the distribution inset in Figure 4b) and most of forests in the CPB watershed have N leaching rates no higher than 5.6 kg N ha^{-1} (Figure 4b), which compares well with the measured data associated with the mixed hardwood forests. The average N deposition rate in the CPB watershed in our study is lower than the estimate used by the EAP and NOAA program (Valigura and others 2000, Stacey and others 2000) (10.04 vs. 12.91 kg N ha^{-1}). The measured depositions rates in the mixed hardwood forests (Gardner and others 1996, see Table 4), ranging from 6.52 to 16.0 kg N ha^{-1}, are comparable to our data in forested lands, ranging from 5.13 to 16.11 kg N ha^{-1} (Table 4). The predicted forest retention rates are between 18 to 97% for coniferous forests and 50 to 98% for hardwood/mixed forests (Table 2), compared to the estimates for the mixed hardwood forests that range from 23 to 98% (Table 4).

Comparison with the data-based estimates for the CPB

Table 2. The predictions of the forest N exports to streams and N retention rates in the Chesapeake Basin watershed

Tree groups	Forest area (km^2)	Total N loss (Mg N)	Min	Max	Mean	SD	N retention (%)
			(kg N ha^{-1} y^{-1})				
Current N Scenario (Mean N deposition = 10.04 kg N ha^{-1} y^{-1}							
N. hardwood	20,298	3,013.88	0.313	2.725	1.4847	0.3288	86 (78–96)
Spruce–fir	22	4.97	0.617	4.444	2.2580	0.8574	78 (67–92)
Oak–hickory	52,065	5,326.24	0.179	2.766	1.0230	0.5064	90 (78–98)
Pine	7,404	1,023.37	0.207	10.590	1.3822	2.0161	84 (18–97)
Oak–pine	14,724	2,248.66	0.224	5.817	1.5272	1.3723	84 (58–97)
Region	94,514	11,617.00	0.179	10.590	1.2291	0.9109	88 (18–98)
Doubled N scenario (Mean N deposition = 20.07 kg N ha^{-1} y^{-1})							
N. hardwood	20,298	8,916.97	0.553	20.900	4.3928	2.4800	80 (27–96)
Spruce–fir	22	20.03	3.580	15.930	9.1053	2.8084	56 (41–76)
Oak–hickory	52,065	16,091.65	0.274	23.890	3.0907	2.9424	85 (24–98)
Pine	7,404	3,648.52	0.933	18.990	4.9278	3.9481	72 (17–93)
Oak–pine	14,724	7,060.28	0.632	22.110	4.7951	3.8428	75 (28–95)
Region	94,514	35,737.46	0.274	23.890	3.7812	3.1978	81 (17–98)

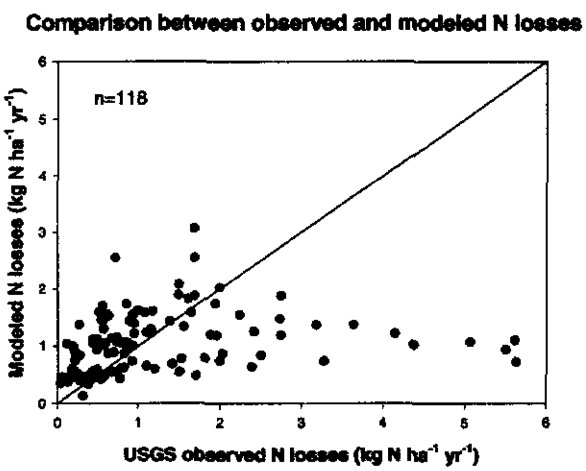

Figure 5. Comparison between the modeled and the United States Geological Survey (USGS) observed nitrate losses for the gauged drainages (n = 118) in the Chesapeake Bay watershed.

watershed. We compared our results with the tabular data of the estimates based on measurement data for the CPB by Stacey and others (2000) (Table 4). The model predicted a lower N loss rate averaging 12%. The recorded N export in the Chesapeake Bay watershed related to atmospheric N sources is 1.39 kg N ha^{-1} y^{-1} and the modeled mean estimate is 1.23 kg N ha^{-1} y^{-1}. However, the N deposition data used in our model simulation that were developed by the Penn State research group (Sheeder and others 2002) were lower than the estimates by another atmospheric deposition work group (Meyers and others 2000) (10.04 vs. 12.91 kg N ha^{-1}). The method used by Sheeder and others (2002) improved N deposition estimates, whereas, the methods used in previous studies were considered to overestimate the contribution of atmospheric N to the CPB watershed (Sheeder and others 2002). Even with those differences, the nitrogen retention estimate for the CPB by PnET-CN is equivalent to the estimate (88% vs. 89%, Table 4) by Stacey and others (2000).

In this study, we used a process-based forest ecosystem model and were only able to consider N sources from atmospheric N deposition and nitrate losses from forests to surface water. This is very different from most empirical watershed models that include all land-use types in watersheds and may also cover all different sources of N loads in streams (Alexander and others 2000, Castro and others 2003). In addition, the differences in the N deposition data generated by different models and used in making estimates of N exports make it even harder for this study to have rigorous validation and comparison.

For example, agricultural and urban lands are considered to have lower N retention rates than forests. Given the average retention rate for agricultural lands of 85%, and urban lands of 40% (Castro and others 2000, 2003), and land cover of 33.3% and 8.4% of the

Table 3. The drainages with higher nitrate fluxes

Gauge stations of drainages	Latitude	Longitude	Drainage area (miles^2)	Runoff (cm)	N loss (kg N/ha/y)
Applemans Run below Light Street, PA	41.0320	−76.4272	1.99	45.32	3.63
Applemans Run above Light Street, PA	41.0314	−76.4199	1.72	39.60	3.17
Juniata River at Mapleton Depot, PA	40.3923	−77.9350	2030.00	43.10	5.61
Little Conestoga Creek near Churchtown, PA	40.1448	−75.9886	5.82	44.89	5.50
Dunning Creek at Belden, PA	40.0717	−78.4925	172.00	46.75	4.13
Conestoga River at Lancaster, PA	40.0501	−76.2772	324.00	42.55	5.06
Pequea Creek at Martic Forge, PA	39.9059	−76.3283	148.00	56.34	5.62
Pequea Creek Tributary near Mt. Nebo, PA	39.8909	−76.3033	0.20	56.22	3.26
Swatara Creek at Harrisburg Airport at Middletown	39.8184	−77.1069	0.38	54.64	4.35

Table 4. Atmospheric inputs, stream-water N losses and retention rates measured or estimated for forests or entire Chesapeake Bay (CPB) watershed

Watershed/pixels	Mean stream output (kg N ha^{-1}y^{-1})	N deposition (kg N ha^{-1}y^{-1})	N retention (%)
Rhode River, MD[a]	1.77	11.8	85
Baldwin Ck, PA[a]	2.19	15.5	87
Benner Run, PA[a]	0.73	15.6	95
Rober Run, PA[a]	0.64	15.2	96
Stone Run, PA[a]	0.26	15.3	98
Miller Run, MD[a]	3.51	6.52	46
Upper Big Run, MD[a]	2.79	6.52	57
Monroe Run, MD[a]	5.02	6.52	23
Peapatch Ridge, MD[a]	4.70	6.52	28
Lower Big Run, MD[a]	4.27	6.52	35
Fernow #4, WV[a]	5.15	16.0	68
PnET-CN (conifer)[b]	0.21–10.59	6.43–13.72	18–97
PnET-CN (hardwood/mixed)[b]	0.18–5.82	5.13–16.11	58–98
CPB[c]	1.39	12.91	89
CPB[d]	2.28	12.91	82
CPB[e]	1.90	10.00	81
PnET-CN (CPB)[f]	1.23	10.04	88
PnET-CN (CPB)[g]	1.46	10.04	85

[a]Measured data for forests in the CPB watershed (Gardner and others 1996).
[b]Ranges of the values predicted by PnET-CN for forests in the CPB watershed.
[c]Estimates for the CPB watershed based on measured data (Stacey and others 2000).
[d]Estimates for the CPB watershed by the SPARROW model (Alexander and others 2000).
[e]Estimates for forested lands in the CPB watershed by the CBP model (Linker and others 1999).
[f]Estimates for forested lands in the CPB watershed by the PnET-CN model.
[g]Estimates for the CPB watershed based on the result in f and adjustments to other land use types.

total CPB watershed respectively, we calculated the mean N loading in the CPB watershed to be 1.46 kg N ha^{-1} y^{-1} and total N loss 23,504 Mg N per year from atmospheric N deposition, slightly higher than the estimate by Stacey and others (1.39 kg N ha^{-1} y^{-1}, see Table 4), which also estimated total N loss of 22,410 Mg N per year from the CPB watershed. However, the N deposition data we used are lower than the data in Stacey and others (2000), which may imply that either we overestimate N exports related to the atmospheric N sources, or that the N deposition in Stacey and others (2000) is overestimated.

Comparison with other model estimates for the CPB watershed. At the scale of the CBP watershed, the nitrogen loading rates estimated by the SPARROW model (Smith and others 1997) for the CBP (Alexander and others 2000) are higher than the estimates based on the measured data (Stacey and others 2000) and by the PnET-CN model, which resulted in lower N retention estimates (Table 4). The national SPARROW model was designed as an empirical function that manipulates impacts of landscape and stream channel characteristics on transportation of nitrogen mass from different sources in watershed. The parameters in the model were estimated using a nonlinear regression approach (Alexander and others 2000). The input sources of N in the model include five major classes, of which atmospheric deposition is one. The model was calibrated using the USGS stream monitoring records of total nitrogen at 374 sites in the conterminous United States (Alexander and others 1998). The percentage contribution of the atmospheric nitrogen sources to surface waters in SPARROW is much higher than in Hydrological Simulation Program Fortran (Bicknell and others 1997), a dynamic hydrology model for the CPB watershed (32% vs. 20%). If the value of 20% were used to calculate the atmospheric N contribution to stream export in SPARROW, it would proportionally lower SPARROW's estimate of N export to 1.43 kg N ha^{-1}, closer to the estimates of Stacey and others (2000) and the PnET-CN model (1.39 and 1.46 kg N ha^{-1} y^{-1}, respectively) (Table 4).

The estimates of N loading rates either derived from the measurements (Stacey and others 2000) or by the national SPARROW model (Alexander and others 2000) are based on total lands in the CPB watershed including forests, agriculture, and urban areas. However, the N loss rates originally estimated by PnET-CN are for forested lands. The only local model that separates land-use types is the Chesapeake Bay Program Model, a continuous and deterministic watershed model (Linker and others 1999). In addition, the N deposition rate used in the Chesapeake Bay Program Model is close to that in the PnET-CN simulation (Table 4). However, the CBP model estimated an N loading rate from forests of 1.9 kg N ha^{-1}, and forest retention rate of 81%, which are much higher N loading and lower retention than estimated by PnET-CN (Table 4).

Conclusion

Our simulations, based on the ramped N increase compared with the condition with no N deposition in the past 70 years, indicate that at current N deposition levels, the N leaching rate from forested lands of the CPB watershed is 1.23 kg N ha^{-1}, and the N retention rate by forests is approximately 88%. The total nitrate discharged annually from forests to streams was estimated at more than 11,600 Mg N y^{-1}. If atmospheric N deposition were twice current values, N retention in forests would drop to 81%. Total nitrate leaching loss to streams would increase more than threefold and be 35,737 Mg N y^{-1}. The remarkable increase in N leaching loss predicted by the model suggests a nonlinear increase in N losses from forested lands and an aggravating decline of forest retention in the CPB watershed as forests approach N saturation with rising levels of atmospheric N deposition.

The predicted rates of N losses are well validated by the USGS-NAQWA nitrate flux data from forested gauge stations, except for a few drainages located in intensive agricultural or urban areas, where the measured data represent integrated effects of different land types. In addition, the predicted N loading and retention of forests in the CPB watershed are also compatible to the measured data associated with the mixed hardwood forests in or near the CPB watershed both for N deposition and exports.

At the scale of watershed, the modeled N leaching losses and N retention rates for forests in the CPB watershed compare well with the estimates based on measurements for the entire basin by Stacey and others (2000). If agricultural and urban land types and their retention capacities for atmospheric N deposition are considered, the PnET-CN model predicted 5% higher N losses than Stacey and others (2000) and may imply either an overestimate in N loads by PnET-CN, or an overestimate in N deposition in Stacey and others (2000) PnET-CN model predicted lower N exports than the national SPARROW model for the CPB watershed, but the prediction by national SPARROW could be jeopardized by the uncertainty of the ratios used in the model that separate the atmospheric nitrogen sources from other N sources in surface waters. If an appropriate ratio was used in the SPARROW model, it could result in a very similar N loading rate as predicted by PnET-CN after the latter was adjusted to other land cover types besides forests. Regarding the prediction of N loading forforested lands in the CPB watershed, it is unclear why the Chesapeake Bay Program Model estimated a much higher rate than PnET-CN.

Our analysis presents a solid validation and comparison between the PnET-CN predictions and measured data for forests and forested drainages in the CPB watershed. It also demonstrates compatible results with other estimates either based on measurements or models for the CPB watershed after a few factors were considered and adjusted. The PnET-CN model appears to be a reliable model that can predict N leaching losses from forest ecosystems in the Chesapeake Region with reasonable accuracy. Our results indicated that the function of forests for alleviating N nutrient loads to surface waters could be diminished quickly as forest ecosystems approach N saturation status, a likely situation in the CPB watershed. It is important to develop regulation policies that reduce atmospheric N deposition and help to retard N losses from forests in the most fragile areas (Castro and Driscoll 2002). The process-based model PnET-CN, different from statistic-based models, has strength to diagnose potential changes of N cycling in watersheds under different levels of atmospheric N deposition. Future studies will simulate the regulation scenarios for control of N deposition and effects on stream N exports, to provide information to guide management decisions.

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References

Aber, J. D., S. V. Ollinger, C. A. Federer, P. B. Reich, M. L. Goulden, D. W. Kicklighter, J. M. Melillo, and R. G. Lathrop Jr. 1995. Predicting the effects of climate change on water yield and forest production in the Northeastern U.S. *Climate Research* 5:207–222.

Aber, J. D., and C. T. Driscoll. 1997. Effects of land use, climate variation and N deposition on N cycling and C storage in northern hardwood forests. *Global Biogeochemical Cycles* 11:639–648.

Aber, J. D., S. V. Ollinger, and C. T. Driscoll. 1997. Modeling nitrogen saturation in forest ecosystems in response to land use and atmospheric deposition. *Ecological Modeling* 101:61–78.

Aber, J. D., and J. M. Melillo. 2001. Terrestrial ecosystems. 2nd ed. Academic Press, San Diego, California 556.

Aber, J. D., C. L. Goodale, S. V. Ollinger, M-L. Smith, A. H. Magill, M. E. Martin, R. A. Hallett, and J. L. Stoddard. 2003. Is nitrogen deposition altering the nitrogen status of northern forests?. *Bioscience* 53:375–389.

Alexander, R. B., J. R. Slack, A. S. Ludtke, K. K. Fitzgerald, and T. L. Schertz. 1998. Data from selected U.S. Geological Survey national stream water quality monitoring networks. *Water Resources Research* 34:2401–2405.

Alexander, R. B., R. A. Smith, G. E. Schwartz, S. D. Preston, J. W. Brakebill, R. Srinivasan, and P. A. Pacheco. 2000. Atmospheric nitrogen flux from the watersheds of major estuaries of the United States: An application of the SPARROW watershed model. Pages 119–170. *in* R. A. Valigura, R. B. Alensander, M. S. Castro, T. P. Meyers, H. W. Paerl, P. E. Stacey, and R. E. Turner. Eds, Nitrogen loading in coastal water bodies: an atmospheric perspective. The American Geophysical Union, Washington, DC.

Bicknell, B. R., J. C. Imhoff, J. L. Kittle, A. S. Donigian Jr., and R. C. Johanson. 1997. Hydrological simulation program: Fortran user's manual for release 11. EPA/600/R-97/080. U.S. Environmental Protection Agency, Environmental Research Laboratory, Althens, Georgia 766.

Castro, M. S., C. T. Driscoll, T. E. Jordan, W. G. Reay, W. R. Boynton, S. P. Seitzinger, R. V. Styles, and J. E. Cable. 2000. Contribution of atmospheric deposition to the total nitrogen loads to thirty-four estuaries on the Atlantic and Gulf coasts of the United States. Pages 77–106. *in* R. A. Valigura, R. B. Alensander, M. S. Castro, T. P. Meyers, H. W. Paerl, P. E. Stacey, and R. E. Turner. Eds, Nitrogen loading in coastal water bodies: an atmospheric perspective. The American Geophysical Union, Washington, DC.

Castro, M. S., and C. T. Driscoll. 2002. Atmospheric nitrogen deposition to estuaries in the Mid-Atlantic and Northeastern United States. 2002. *Environmental Science Technology* 2002:3242–3249.

Castro, M. S., C. T. Driscoll, T. E. Jordan, W. G. Reay, and W. R. Boynton. 2003. Sources of nitrogen to estuaries in the United States. *Estuaries* 28(3):803–814.

Fenn, M. E., M. A. Poth, J. D. Aber, J. S. Baron, B. T. Bormann, D. W. Johnson, A. D. Lemly, S. G. McNuty, D. F. Ryan, and R. Stottlemyer. 1998. Nitrogen excess in North American ecosystems: Predisposing factors, ecosystem responses, and management strategies. *Ecological Applications* 8:706–733.

Galloway, J. N., W. H. Schlesinger, H. Levy II, A. Michaels, and J. L. Schnoor. 1995. Nitrogen fixation: atmospheric enhancement- environmental response. *Global Biogeochemical Cycle* 9:235–252.

Gardner, R. H., M. S. Castro, R. P. Morgan, and S. W. Seagle. 1996. Perspectives on Chesapeake Bay: nitrogen dynamics in forested lands of the Chesapeake Basin. Scientific Technical Advisor Committee, Chesapeake Bay Program, USA 36.

Jones, K. B., A. C. Neale, M. S. Nash, R. D. Van Remortel, J. D. Wickham, K. H. Riitters, and R. V. O'Neill. 2001. Predicting nutrient and sediment loadings to streams from landscape metrix: A multiple watershed study from the United States Mid-Atlantic region. *Landscape Ecology* 16:301–312.

Likens, G. E., and F. H. Bormann. 1995. Biogeochemistry of forest ecosystem. 2nd ed. Springer-Verlag, New York.

Linker, L. C., G. W. Shenk, R. L. Dennis, and J. L. Sweeney. 1999. *Cross-media models for the chesapeake Bay watershed and airshed.* November, 1999. Chesapeake Bay Program Office, Annapolis, MD. *http://www.chesapeakebay.net/modsc.htm &ftp://ftp.chesapeakebay.net/Modeling/Change1985to2000.xls*

Magill, A. H., J. D. Aber, G. M. Berntson, W. H. McDowell, K. J. Nadelhoffer, J. M. Melillo, and P. Steudler. 2000. Long-term nitrogen additions and nitrogen saturation in two temperate forests. *Ecosystems* 3:238–253.

McGuire, A. D., J. M. Melillo, and L. A. Joyce. 1995. The role of nitrogen in the response of forest net primary production to elevated atmospheric carbon dioxide. *Annual Reviews of Ecology and Systematics* 26:473–503.

McGuire, A. D., J. M. Melillo, D. W. Kicklighter, Y. Pan, X. Xiao, J. Helfrich, B. Moore III, C. J. Vorosmarty, and A. L. Schloss. 1997. Equilibrium responses of global net primary production and carbon storage to doubled atmospheric carbon dioxide: sensitivity to changes in vegetation nitrogen concentration. *Global Biogeochemistry Cycles* 11:173–189.

Meyers, T., J. Sickles, R. Dennis, K. Russel, J. Galloway, and T.

Church. 2000. Atmospheric nitrogen deposition in coastal estuaries and their watersheds. Pages 53–76. *in* R. A. Valigura, R. B. Alensander, M. S. Castro, T. P. Meyers, H. W. Paerl, P. E. Stacey, and R. E. Turner. Eds, Nitrogen loading in coastal water bodies: an atmospheric perspective. The American Geophysical Union, Washington, DC.

Ollinger, S. V., M. L. Smith, M. E. Martin, R. A. Hallet, C. L. Goodale, and J. D. Aber. 2002a. Regional variation in foliage chemistry and N cycling among forests of diverse history and composition. *Ecology* 83:339–355.

Ollinger, S. V., J. D. Aber, P. B. Reich, and R. J. Freuder. 2002b. Interactive effects of nitrogen deposition, tropospheric ozone, elevated CO_2 land use history on the carbon dynamics of northern hardwood forests. *Global Change Biology* 8:545–562.

Pan Y., J. Hom, J. Jenkins, and R. Birdsey. 2004. Importance of foliar nitrogen concentration to predict forest productivity in the Mid-Atlantic region. Forest Science 50(2) (in press)

Reich, P. B., and A. W. Schoettle. 1988. Role of phosphorus and nitrogen in photosynthetic and whole plant carbon gain and nutrient use efficiency in eastern white pine. *Oecologia* 77:25–33.

Schimel, D., J. Melillo, H. Tian, A. D. McGuire, D. Kicklighter, T. Kittel, N. Rosenbloom, S. Running, P. Thornton, D. Ojima, W. Parton, R. Kelly, M. Sykes, R. Neilson, and B. Rizzo. 2000. Contribution of increasing CO_2 and climate to carbon storage by ecosystems in the United States. *Science* 287:2004–2006.

Sheeder, S. A., J. A. Lynch, and J. Crimm. 2002. Modeling atmospheric nitrogen deposition and transport in the Chesapeake Bay watershed. *Journal of Environmental Quality* 31:1194–1206.

Smith, R. A., G. E. Schwarz, and R. B. Alexander. 1997. Regional interpretation of water-quality monitoring data. *Water Resources Research* 33:2781–2798.

Stacey, P. E., H. S. Greening, J. N. Kremer, D. Peterson, and D. A. Tomasko. 2000. Contribution of atmospheric nitrogen deposition to U.S. estuaries: Summary and conclusion. Pages 187–226. *in* R. A. Valigura, R. B. Alensander, M. S. Castro, T. P. Meyers, H. W. Paerl, P. E. Stacey, and R. E. Turner. Eds, Nitrogen loading in coastal water bodies: an atmospheric perspective. The American Geophysical Union, Washington, DC.

Swift Jr, L. W. 1976. Algorithm for solar radiation on mountain slopes. *Water Resources Research* 12:108–112.

United States Geological Survey. 1987. Digital elevation models: National Mapping Program technical instruction: data users guide 5. U.S. Department of the Interior, Geological Survey, Reston, VA 38.

Valigura, R. A., R. B. Alensander, M. S. Castro, T. P. Meyers, H. W. Paerl, P. E. Stacey, and R. E. Turner. 2000. Nitrogen loading in coastal water bodies: an atmospheric perspective. The American Geophysical Union, Washington, DC 254.

VEMAP Members 1995. Vegetation/ecosystem modeling and analysis project: Comparing biogeography and biogeochemistry models in a continental-scale study of terrestrial ecosystem responses to climate change and CO_2 doubling. *Global Biogeochemistry Cycles* 4:407–437.

Vitousek, P. M., H. A. Mooney, J. Lubachenco, and J. M. Melillo. 1997. Human domination of earth's ecosystems. *Science* 277:494–499.

Waring, R. H., and W. H. Schlesinger. 1985. Forest ecosystems: concept and management. Academic Press, Orlando, Florida 340.

Wickham, J. D., R. V. O'Neil, K. H. Riitters, E. R. Smith, T. G. Wade, and K. B. Jones. 2002. Geographic targeting of increases in nutrient export due to future urbanization. *Ecological Application* 12:93–106.

Zhu, Z., and D. L. Evans. 1994. U.S. forest types and predicted percent forest cover from AVHRR data. *Photogrammetric Engineering and Remote Sensing* 60:525–531.