Section 10. NONPOINT SOURCE NUTRIENT SIMULATION

Contents

Figures

Tables

SECTION 10. NONPOINT SOURCE NUTRIENT SIMULATION

10.1 Introduction to the Phase 5.3 Nonpoint Source Nutrient Simulation

10.1.1 Development of Nutrient Calibration Targets for Each Land Use

Edge-of-stream (EoS) target loads are used for calibrating nutrient loads from each land use as a start of the calibration process. Edge-of-stream nutrient target loads are used because relatively few literature values of nutrient loads from the land are reported as edge-of-field (EoF) loads. This is in contrast to the available estimates of sediment loads in Phase 5.3, in which most of the load estimates were made using techniques such as USLE, which are explicitly EoF calculations. The literature values of nutrients often come from watershed studies where loads from a single land use or mixed land uses are measured with in-stream concentrations. Consequently, a translation of the EoF nutrient loads simulated in a unit area of HSPF land to EoS nutrient loads usually reported in the literature was needed. Therefore, EoS nutrient target loads were used in Phase 5.3 for all land uses and literature values for nutrient exports were evaluated from that perspective.

Literature values are sometimes represented by the maximum, minimum, and median values, as well as the 25th and 75th percentiles (Beaulac and Reckhow 1982). Nutrient loads are usually skewed upward, particularly for high-loading land uses. The observed range from literature values is necessarily greater than the range of the Phase 5.3 Model estimated loads because the Phase 5.3 loads are based on the average land use condition for a land-segment (on the order of a county) and would therefore not have the potential for extreme soil, slope, and other conditions seen in some of the literature values (Beaulac and Reckhow 1982; Sweeney 2001).

The Phase 4.3 loads of an earlier Watershed Model application were also used as guidance because they were also based on literature values relevant to the Chesapeake region. A particular gap filled in by the use of the Phase 4.3 loads as guidance is time. The synthesis papers look at literature for land uses, such as cropland studies in the 1970s and 1980s (Beaulac and Reckhow 1982) to more recent periods (Sweeney 2001; Alexander et al. 2001). Agriculture is a dynamic industry with changing practices and standards, and the use of the Phase 4.3 target loads, which characterize Chesapeake Bay watershed land use loads from the years 1985 and 2000, provided additional characterization of nutrient loads and targets for land uses in the early years of the simulation.

Targets are given here as a single value, but they ranged in the calibration over an allowed degree of variation, usually driven by differences in hydrology or nutrient inputs.

Note: When referring to a specifically defined Phase 5.3 land use, the name is in italic. This is to avoid confusion with the general land use type. For example, *forest, woodlots, and wooded land* is a specifically defined Phase 5.3 land use distinct from the general land use of forest land.

10.1.2 Basic Approach

The basic approach was to take literature values as a median target for each land use, which was assumed to be the exported load from the median of the mass balance of inputs. To get the estimated EoS load targets for each land-segment based on the relative amounts of the input nutrients from fertilizers, manures, and atmospheric deposition for that segment/land use, a slope of the change in export load to nutrient load inputs was established. This approach allows increases in application loads to cause an increase in nutrient exports. For a particular land use, it allows a land-segment with high total input loads to get relatively high estimated nutrient targets and a land-segment with relatively low loads to get a lower target. The basic approach and key assumptions are explained below for wooded and crop land uses.

10.1.2.1 Basic Approach Used in *Forest, Woodlots, and Wooded* Land Use

The *forest, woodlots, and wooded* land use has only atmospheric deposition as an input load. The median forest total nitrogen load was assumed to be 3.1 lb/ac/yr, and that generally occurs with the average atmospheric deposition in the watershed of about 21 lb/ac/yr. It was also assumed that a response to an increase of atmospheric deposition would be linear, so that if the atmospheric deposition was twice that of the average deposition, the export of nitrogen would double (Equation 10.1). A linear response is consistent with the literature under moderate levels of nitrogen loading to forest. At high levels of atmospheric deposition, nitrogen saturation occurs and the rate of export increases faster than the rate of deposition increase (Aber et al. 1989; Aber et al. 2003; Goodale et al. 2002; Hunsacker et al. 1993; Stoddard 1994).

Equation 10.1:

$$\text{forest, woodlots, and wooded total nitrogen export target} = \text{3.1 lb/ac/yr median TN export} * \frac{\text{land-segment atmospheric deposition}}{\text{watershed average atmospheric deposition}}$$

The phosphorus loads from the *forest, woodlots, and wooded land use* have less variable atmospheric deposition inputs, so a constant median export target of 0.13 lb/ac/yr for total phosphorus is used everywhere.

10.1.2.2 Basic Approach used in Cropland Uses

In the case of cropland uses, the situation is made more complicated by the additional inputs of fertilizers and manures and the uptake of the majority of the applied nutrients by the harvested crop. Fortunately, Agricultural Census estimates of crop yields by county for the different crop types are available. They allow the opportunity to make adjustments in a nutrient balance for good and bad production years, as well as for land-segments that are inherently more or less productive.

Overall, the nutrient balance for a cropland use in a land-segment is defined as Equation 10.2.

Equation 10.2:

$$\text{cropland nutrient export target} = \text{inputs}_{\text{(fertilizer, manure, atmospheric deposition)}} - \text{crop uptake} - \text{incorporation into soil}$$

10.1.3 Nutrient Uptake Preference

The nutrient uptake preference is generally set at an 80 percent preference for nitrate and an associated 20 percent preference for ammonia in Phase 5.3. This means that in the HSPF simulation the dissolved inorganic nitrogen (DIN) nutrients will be taken up by plants in a nitrate-to-ammonia ratio of 4:1. With respect to the nutrients in soil available to be exported to rivers, the nitrate storage will be drawn down more by plant uptake than the ammonia storage and thus shift the balance toward the export of ammonia (all else being equal).

Nutrient uptake in plants is complex, and much remains to be understood; it is clear, however, that there are two separate and largely independent pathways of uptake for nitrate and ammonia (Tinker and Nye 2000). Using ^{15}N tracers to track nitrate and ammonia uptake in plants, other researchers have described terrestrial plant communities having a significantly higher uptake of nitrate in drier areas and a significantly higher ammonia uptake in wetter areas (Macko et al. 2005). Because of the potential variation in the general pattern of nutrient uptake, the uptake preference was allowed to vary slightly in the calibration if a better calibration could be achieved with a nutrient preference other than the standard 4:1 nitrate/ammonia preference.

10.1.4 Use of Crop Yield Data

To realistically simulate nutrient export from agricultural lands in the Chesapeake Bay watershed, several parameters need to be modeled effectively—particularly the amounts and rates of crop uptake because this is the fate of most of the nutrients applied to agricultural lands. Also important in simulating crop yields are soil nutrient balances, the mineralization rates of various nutrient species, the effect of droughts and heat, and nutrient wash-off from the surface, particularly immediately after nutrient applications.

Several analytical techniques were used to effectively simulate agricultural loads. Data from the Agricultural Census and NASS long-term yield data were compared. “Nutrient Management Scoping”—observing model responses to various theoretical loading rates—was also performed. Model sensitivity to variations in the timing of nutrient application was also tested. Also, data from the Agricultural Census, as well as soil survey data, were used to estimate the maximum yield potential. A new, more robust query application from NASS allows for annual crop yields by county as far back as 1896. This allows a comparison between modeled annual crop yields. It also enhances the ability to estimate recommended nutrient management rates by county and provides more than 9,000 new calibration points for the model.

10.2 Development of Edge-of-Stream Nutrient Targets

Models involve choices, and the choices in setting the nutrient targets are made from a collection of observed values described in the literature (which are always too few). Values given here as targets are an overall average, with a defined range of variance driven by the spatial variation in input nutrient loads. The nutrient export targets also informed by observed stream data during the calibration process and were in the end the nutrient targets that helped produce the best overall calibration. The targets suggested here are not represented as *correct*, but as *correctable* as more and better data and simulation approaches provide greater insight into land use loads under different nutrient input conditions.

More detailed local data for land use within each segment are always to be desired, though are rarely obtained. Because there are more than 300 land-segments and 24 land uses, the aim is to represent broad watershed characteristics of the export of nutrient loads from each simulated land use. Tables 10-1 and 10-2 list the mean and median nutrient export targets in pounds/acre-year for Phase 5.3 land uses in the Chesapeake watershed for total nitrogen and total phosphorus respectively.

Table 10-1. Mean and median of the edge-of-stream total nitrogen targets in pounds per acre for the Phase 5.3 calibration.

Edge-of-Stream Total Nitrogen (lb/a)	Mean	Median
forest, woodlots, and wooded	3.6	3.1
hay-unfertilized	6.8	6.2
nutrient management pasture	7.3	5.8
pasture	9.5	8.2
nutrient management hay	9.7	9.0
hay-fertilized	10.2	9.5
alfalfa	10.6	9.5
nutrient management alfalfa	10.9	11.6
high intensity impervious urban	11.8	9.9
low intensity impervious urban	11.9	10.4
high intensity pervious urban	12.8	10.9
low intensity pervious urban	13.2	11.2
extractive	14.0	13.1
harvested forest	24.3	21.4
bare-construction	29.5	26.4
nutrient management conservation till	37.1	39.6
conservation till receiving manures	38.1	39.6
nutrient management conventional till with manure	40.5	43.5
conventional till with manure	41.9	44.8
conventional till without manure	42.5	40.2
nutrient management conventional till without manure	42.9	42.4
degraded riparian pasture	52.4	45.9
nursery	286.7	253.8
animal feeding operations	1087.1	1045.7

Table 10-2. Mean and median of the edge-of-stream total phosphorus targets in pounds per acre for the Phase 5.3 calibration.

Edge-of-Stream Total Phosphorus (lb/a)	Mean	Median
hay-unfertilized	0.03	0.03
hay-fertilized	0.06	0.05
forest, woodlots, and wooded	0.14	0.13
nutrient management hay	0.16	0.15
nutrient management alfalfa	0.82	0.83
high intensity pervious urban	0.88	0.89
low intensity pervious urban	0.90	0.90
alfalfa	0.92	0.87
nutrient management pasture	0.94	0.83
pasture	0.99	0.92
harvested forest	1.14	1.02
nutrient management conservation till	1.88	1.56
conservation till with manures	2.00	1.73
nutrient management conventional till with manures	2.39	1.98
conventional till with manures	2.51	2.05
high intensity impervious urban	2.62	2.49
low intensity impervious urban	2.63	2.50
nutrient management high till without manures	3.07	2.92
conventional till without manures	3.09	3.08
extractive	4.83	4.42
bare-construction	9.67	8.81
degraded riparian pasture	11.77	10.97
animal feeding operations	59.97	56.45
nursery	118.51	111.98

10.2.1 *Forest, Woodlots, and Wooded Areas* EoS Nutrient Targets

The Phase 5.3 *forest, woodlots, and wooded areas* land use covers woodland, woodlots, and usually any wooded area of 30 meters by 30 meters remotely sensed by spectral analysis. The *forest, woodlots, and wooded areas* land use is the predominant land use in the Chesapeake Bay watershed. A good portion of what would normally be considered developed land falls in the *forest, woodlots, and wooded areas* category; examples are wooded urban parks and wooded low-density residential areas. Even heavily developed regions have considerable areas of *forest, woodlots, and wooded areas*. For example, within the city boundaries of the District of Columbia and Newport News, Virginia, there are 10 percent and 26 percent of the land respectively, is in *forest, woodlots, and wooded areas* in the Phase 5.3 Model.

Beaulac and Reckhow (1982) estimate the annual average forest and woodland total nitrogen export loads to be about 2.2 lb/ac-yr based on about eight studies. Beaulac and Reckhow report the range of total nitrogen export to be about 1.2 to 5.4 lb/ac-yr. Alexander et al. (2001) report SPARROW Model forest nitrogen load estimates of 1.6 to 10.0 lb/ac-yr and literature values of 0.1 to 9.6 lb/ac-yr. These are consistent with the Sweeney literature review (2001), which found a median total nitrogen export of 2.2 lb/ac-yr with a range of 0.1 to 9.1 lb/ac-yr from 17 publications of forest load estimates. Hunsaker et al. (1993), in their literature synthesis of forest nitrogen loads, had similar rates when highly loaded (18 to >80 lb/ac-yr) European forests were excluded from the data.

For Phase 5.3, a median value of 3.1 lb/ac-yr total nitrogen export was chosen to represent *forest, woodlots, and wooded land*; a value consistent with the literature (Campbell 1982: Langland et al. 1995; Castro et al. 1997; Ritter et al. 1984; Stevenson et al.1987; Nixon 1997; Riekerk et al. 1988; Clark et al. 2000; Goodale et al. 2002; Pan et al. 2005) and consistent with the relatively high portion of wooded land in the watershed associated with low density developed land. Total nitrogen loads from forest and woodlots decrease with decreasing nitrogen from atmospheric deposition and increase with increasing nitrogen deposition loads, particularly in the literature, after a high load threshold is reached (Figure 10-1). This is consistent with observations from Aber et al., (1989, 2003) and other studies of forests nitrogen loads and nutrient exports (e.g., Stoddard 1994).

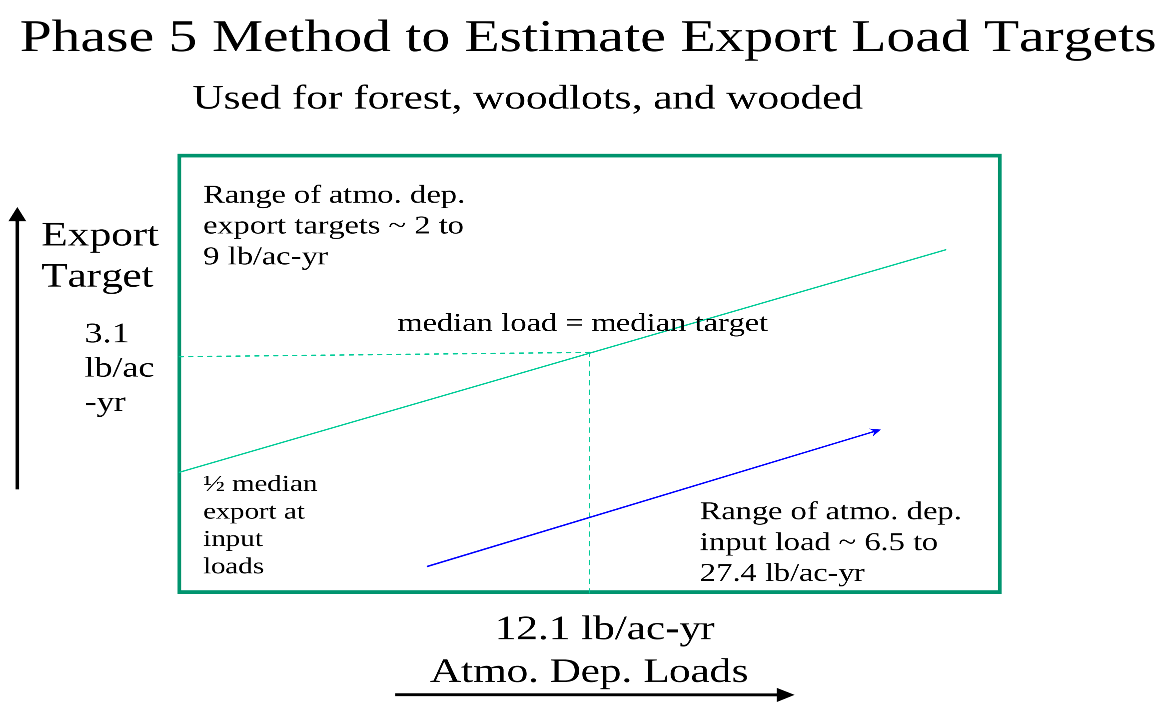


Figure 10-1. Method of adjusting *forest, woodlots, and wooded* land export targets to input loads.

Hunsaker et al. (1993) describes three studies in the Chesapeake Bay watershed of largely forested catchments of Stony Creek (Pennsylvania), Young Womans Creek (Pennsylvania), and Rhode River (Maryland). In these forested catchments, ammonia, nitrate, and organic nitrogen concentrations were measured and the annual average fluxes were estimated. Based on the average of these studies, the target splits of ammonia, nitrate, and organic nitrogen are 11 percent, 38 percent, and 51 percent, respectively. Based on the high portion of subsurface flow in wooded land uses, and with little literature as guidance, we arbitrarily assume that 90 percent of the nitrogen loads of each species are from subsurface export. We further assume that all exported organic nitrogen is dissolved and is split into 50 percent surface and 50 percent subsurface and, following Hunsaker et al., about 5 percent of the dissolved organic nitrogen is labile in the surface and subsurface export.

The labile and refractory organic nitrogen terms need more definition. In the Phase 5.3 Watershed Model, labile organic nitrogen is nitrogen that acts like the reactive organic nitrogen measured in a BOD_5. The use of this particular arbitrary split between labile and refractory organics has several distinct advantages. The first is that a consistent labile and refractory split can be used throughout the model for all nutrient loads from land, in rivers, or discharged by a pipe. The use of BOD_5 is important for point source loads, where BOD_5 measurements are common. Also, BOD_5 is perhaps the most common of the uncommon finds of organic nutrient measures in the literature. The five-day time of the BOD_5 measurement is consistent with the maximum time of travel in the simulated rivers of the Chesapeake Bay watershed, and within a five- day period the labile organic nitrogen may react within the model domain; if not, it simply acts as if it is refractory as the reaction time exceeds the average residence time of the model, at least for river waters. Using this definition limits the values labile organic nitrogen may take on in the targets. In the forests the reservoir of organic nitrogen is large, about two tons per acre (Hunsacker et al. 1993). Of this reservoir, about 2 to 7 percent is mineralized each year. Accordingly, in our forest simulation we assume that the bulk of the organic nitrogen exported is refractory (95 percent) and that only a portion (5 percent) is labile. Relatively more of the simulated forest nitrogen is considered refractory relative to other land uses due to observations that anthropogenic organic nitrogen is more bioavailable than forest-derived organic nitrogen (Seitzinger et al. 2002a; Wiegner et al. 2006).

Literature values for total phosphorus range from 0.01 to 0.9 lb/ac-yr and have a median value of about 0.10 lb/ac-yr, and this is the target load used in Phase 5.3. The median total phosphorus export from Phase 4.3 wooded lands is lower at 0.02 lb/ac-yr. A commonly used cellular ratio of nitrogen to phosphorus (by weight) is about 10:1, which would argue that almost all the observed total phosphorus export is organic. Following the organic nitrogen splits, one-half the organic phosphorus is assumed to be associated with the surface and one-half with the subsurface; it is also assumed that 5 percent of these surface and subsurface organic phosphorus pools are labile. A small portion, 5 percent of both the surface and subsurface phosphorus load exported, is further assumed to be dissolved inorganic phosphate. Forest total phosphorus exports will be largely unchanged by management actions because few BMPs are present to reduce forest phosphorus loads; silviculture BMPs are applied to the *harvested forest* land use.

The target loads of forest were varied in order to allow those forests with high atmospheric deposition loads to have a higher nitrogen export consistent with our understanding of forest nitrogen dynamics (Aber et al. 1989, 2003; Stoddard 1994). The relationship between atmospheric deposition of nitrogen and forest load export is shown in Figure 10.1. Target loads of phosphorus for *forests, woodlots, and wooded* have a median value of 0.13 lb/ac-yr due to a small constant concentration of phosphorus in atmospheric deposition as described in Section 5.

10.2.2 *Harvested Forest* EoS Nutrient Targets

Harvested forest has a higher nutrient loading rates due to a multitude of landscape changes. Surface and subsurface flows are increased in harvested forest areas due to the reduced evapotranspiration as well as the reduced interception storage (Wang et al. 2003; Arthur et al.

1999; Riekerk et al. 1988; Frick and Buell 1999). More flow is available for nutrient export, but microbial mineralization rates also increase as direct sunlight on the forest soils increases soil temperatures. The reservoir of nutrients is estimated to be on the order of about two tons of organic nitrogen in forest soils and an equivalent amount of organic phosphorus (Hunsaker et al. 1993). The higher flows and microbial rates increase the inorganic nutrient export from *harvested forest*. At the same time, erosion increases due to disturbed soils, decreased evapotranspiration, increased runoff volume, and decreased canopy cover, which increases the impact energy of raindrops (Wilson et al. 1999; Grace 2004; Hewlett, et al.1979; Keppeler et al. 2003; Perry 1998).

Observations of nutrient export vary widely (Arthur et al. 1998; Riekerk et al. 1988; Lebo and Herrmann 1998; Ensign and Mallin 2001; Goodale et al. 2002). Arthur et al. (1999) describe an order-of-magnitude increase in nitrate loads and a doubling of phosphate loads for harvested forest relative to undisturbed forest. This is substantiated by Riekerk et al. (1998), who observe that organic nutrient loads increase about six-fold and ammonia increases about two-fold over that of undisturbed forest. The Phase 5.3 nutrient export targets for *harvested forest* reflect these reported nutrient loads, with an estimated median total nitrogen annual export load of 21.4 lb/ac-yr and a median total phosphorus annual export load of 1.02 pound/ac-yr. The higher loads from harvested forest are primarily due to the disturbance of the forest soil and reservoir of soil organic nitrogen and are a single fixed target value for all *harvested forest*.

The period of time during which the disturbed forest exports this high nutrient load is another problem for the HSPF structure. The literature suggests that a return to the nutrient export rates of undisturbed forest occurs after about three to five years (Arthur et al. 1998; Castro et al. 1997). With only two wooded land uses of *forest/woodlot* and *harvested forest*, simulating the slow return of nutrient exports to the undisturbed forest rate is impractical and simplifying assumptions have to be made. Accordingly, the *harvested forest* nutrient export rates are applied in the simulation for the area of harvested forest for a single year, with an estimated forest harvest rate of 1 percent of forest annually.

10.2.3 Agricultural Land Uses Nutrient Targets

The EoF nutrient targets for agricultural land uses are scaled against literature values based on a calculated residual remaining after crop removal. The assumption is that the median literature values represent the median EoF value for the Chesapeake Bay watershed.

Edge-of-field estimates were calculated based on a mass balance of known inputs and outputs on the various land uses. These EoF balances were then compared to literature reviews of in-stream concentrations assigned “back-stream” to likely upland sources, to determine estimates of attenuation/delivery factors in lower-order streams (literally, between the EoF and the EoS since the Phase 5.3 model is more watershed scale than field scale). Not only does this assist in getting realistic values flowing into the reach portions of the model, ensuring a reasonable calibration of *modeled* in-stream attenuation, but it also addresses uncertainty in input/output estimates in the mass balance.

The mass balance on pervious surfaces was calculated based on the following data:

- Inputs: fertilizer (I_F), manure (I_M), N-fixation (I_L), atmospheric deposition (I_A)

- Outputs: crop removal/harvest (O_H), de-nitrification/volatilization (O_V), attenuation in lower-order streams (O_A)
- Residual (E): nitrogen (mobile) and phosphorus (mobile and immobile)
- The residual, E, is that which might reach the stream:

$$E = (I_A + I_F + I_M + I_L) - (O_H + O_V + O_A)$$

Because all the studies used in the literature review took values at a watershed outlet, it was hypothesized that these values would involve significant attenuation of nitrogen, and possible retention of phosphorus and sediment. CBP performed an analysis comparing the total annual loads at numerous monitoring points to the sum of all upland land use acres (from the CB watershed model) multiplied by the median EoS values for those land-uses, i.e., CB watershed model targets from the literature review, to estimate what the watershed model would predict if these edge-of-stream values were used as EoF values with no modeled in-stream attenuation. This comparison shows that the predicted loads were between 0.4 and 0.6 of the monitored loads for nitrogen, and near unity (0.9–1.1) for phosphorus and sediment. This indicated that roughly 40 to 60 percent attenuation of nitrogen was occurring in the basins from which the land-use-specific literature review values were derived. It also indicates only a small amount of retention of phosphorus and sediment in these studies. The attenuation values in this analysis are similar to those in a separate study by Rutgers University (Seitzinger et al. 2002b), which showed that between 37 and 76 percent attenuation of N would be predicted in various large rivers (both within and outside the Bay watershed) from a regression model (based on monitored values) of stream nitrogen concentration as a function of in-stream travel time and stream depth. Similarly, the annual mass balance in agricultural land uses in the watershed model inputs, which compares estimated annual application rates with reported crop removal, showed that the expected annual residual nitrogen (not removed by crop) would be approximately two to three times the EoS values found in the literature review, supporting the notion that considerable attenuation would occur between the EoF and watershed outlet. Seitzinger et al. (2002b) reported that approximately half of the attenuation of nitrogen should occur in first- to fourth-order streams; therefore, the EoF target for model river-segments was set to approximately two times the literature review value, given that the average river-segment is fourth order or less—in other words, effectively allowing for 50 percent of attenuation to occur within the modeled reaches that are greater than fourth order. It should be noted that the land module is effectively accounting for some nitrogen attenuation that happens in lower-order streams and that if the model resolution were to be increased (i.e., smaller land units), the EoF targets should be modified to decrease the attenuation because the stream network would be expected to attenuate more nitrogen.

Because the mass balances vary greatly by county, there is no one EoS value that would work for a particular land use. Instead, the range of calculated mass balances was scaled to meet the range of reported literature values. Given the spatial aggregation of the modeling scale, it was assumed that the most extreme values from the literature review were representative of anomalous field-driven or event-driven circumstances, and therefore only the upper 75 percent and lower 25 percent values from the literature review were considered to bound the model EoS targets.

Figure 10-2 shows a graphical depiction of the range of nutrient inputs of manures, fertilizer and atmospheric deposition for agricultural lands and how the target load is adjusted fro the range of nutrient inputs. Generally the lower nutrient inputs are for hay unfertilized agricultural land and the highest inputs are for row crops with conventional tillage and without nutrient management.

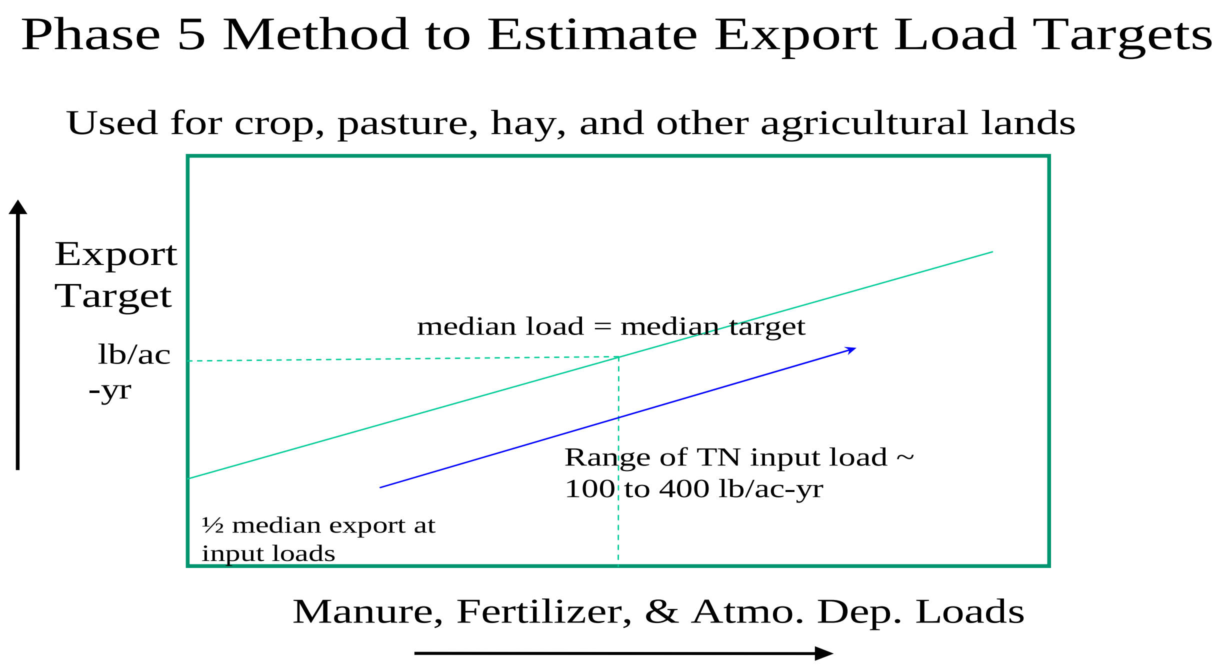


Figure 10-2. Method of adjusting agricultural land use export targets to input loads.

10.2.4 *Pasture* EoS Nutrient Targets

Consistent with literature values (Beaulac and Reckhow 1982; Alexander et al. 2001; Sweeney 2001), the *pasture* median target loading rates for total nitrogen are set at 8.2 lb/ac-yr with a range of 1.6 lb/ac-yr to 34.4 lb/ac-yr based on low to high pasture stocking rates. The range of the estimated stocking rates for the *pasture* is based on the estimated county segment area of *pasture* and the Agricultural Census estimate of pastured animals. The *pasture* target loading rates for total phosphorus is a median of 0.92 lb/ac-yr and with an estimated range of 0.14 lb/ac-yr to 2.69 lb/ac-yr based on estimated manure loads from the stocking rates of pastured animals.

10.2.5 *Degraded Riparian Pasture* EoS Nutrient Targets

The *degraded riparian pasture* land use represents unfenced riparian pasture with an associated stream degraded by livestock. This land use has high nutrient and sediment loads and is treated by riparian buffer BMPs. For each county segment, the median nutrient load target is set at about 5 times the *pasture* median target rate for nitrogen and about 11 times the *pasture* median target rate for total phosphorus. The estimated target nutrient load is higher in degraded riparian pasture because of direct defecation of livestock in streams and the increased time livestock spend in riparian areas. *Degraded riparian pasture* median target loading rates for total nitrogen are set at 45.9 lb/ac-yr with a range of 18.4 lb/ac-yr to 96.0 lb/ac-yr. The *degraded riparian pasture* target loading rates for total phosphorus is a median of 10.97 lb/ac-yr and with an estimated range of 3.99 lb/ac-yr to 21.19 lb/ac-yr.

10.2.6 *Nutrient Management Pasture* EoS Nutrient Targets

Nutrient management pasture is pasture that is included in a farm plan where crop nutrient management is practiced. *Nutrient management pasture* receives excess manures on a farm after all crop nutrient needs are satisfied. *Nutrient management pasture* is a land use originally calibrated as *pasture* and assumed to have the same estimated stacking rates as pasture, but it has additional nutrient loads applied as nitrogen and phosphorus from manures in excess of cropland need. *Nutrient management pasture* median target loading rates for total nitrogen are set at 5.8 lb/ac-yr with a range of 1.5 lb/ac-yr to 28.7 lb/ac-yr. The *nutrient management pasture* target loading rates for total phosphorus is a median of 0.83 lb/ac-yr and with an estimated range of 0.18 lb/ac-yr to 2.28 lb/ac-yr.

10.2.7 *Hay-Unfertilized* EoS Nutrient Targets

Hay-unfertilized includes the land uses from the Agricultural Census of wild hay, idle cropland, and fallow land. These are considered to be areas in hay or in other ways herbaceous agricultural areas that do not receive fertilizer and are not harvested. Orchards are also included in this category.

The Phase 5.3 median target of exported nitrogen loads for *hay-unfertilized* is set to 6.2 lb/ac-yr, about twice that of the median load from *forest, woodlots, and wooded areas* and has a range of 2.0 to 13.1 lb/ac-yr . Because this land is assumed to be infrequently planted or harvested, the median target for the phosphorus exported load is set to 0.03 lb/ac-yr with a range of 0.01 to 0.05 lb/ac-yr.

10.2.8 *Hay-Fertilized* EoS Nutrient Targets

Hay-fertilized includes all tame and small grain hay except wild hay or alfalfa, which are included in other categories. These crops receive fertilizer and have a high degree of surface cover for most of the year. Failed cropland is also included in this category because failed cropland receives fertilizer but is not harvested, a pattern most similar to *hay-fertilized*.

Since the *hay-fertilized* land use receives fertilizer and has more extensive field operations of planting and harvest than the *hay-unfertilized* land use, the nutrient loads exported are assumed to be higher. Accordingly, the *hay-fertilized* nitrogen export median load target is set at 9.5 lb/ac-yr with a range of 1.9 to 41.5 lb/ac-yr and the phosphorus median load target set at 0.05 lb/ac-yr with a range of 0.01 to 0.42 lb/ac-yr. Nutrient management hay has a median target load of 9.0 and 0.15 lb/ac-yr for total nitrogen and total phosphorus respectively.

10.2.9 *Alfalfa* EoS Nutrient Targets

This land use category contains only *alfalfa* hay. *Alfalfa* is simulated similar to hay except that *alfalfa* is leguminous. The target loads of *alfalfa* are slightly more than those for *hay-fertilized* for nitrogen, but they are simulated with a different timing of application that reflects *alfalfa* nitrogen fixation. The nitrogen median export target load is 9.5 lb/ac-yr, and the phosphorus median export target load is similar to that of *hay-fertilized* at 0.87 lb/ac-year.

10.2.10 *Conventional Tillage Cropland Receiving Manures* EoS Nutrient Targets

The land use of *conventional tillage cropland receiving manures* in Phase 5.3 has a specific operational definition and is a collection of land uses defined in the Agricultural Census. The *conventional tillage cropland receiving manures* land use includes grain, corn, soybeans, and dry beans. Wheat, corn, and soybeans are the dominant crops in the Chesapeake Bay watershed, often planted in a two-year rotation on the same parcel of land. Crops in this land use receive nutrient inputs from manure application as well as fertilizer. This land use has conventional tillage operations in which less (often much less) than 35 percent cover remains on the ground at the time of planting.

Nutrient loads from cropland are highly variable, reflecting a variable and dynamic agricultural industry. In addition, these loads change over time, as in the two-decade period of the Phase 5.3 simulation (1985 to 2005); different agricultural practices such as nutrient management are tested, take hold, and then become widely implemented. This is seen in the changing estimates of fertilizer and manures that are applied to the cropland, as described in Section 5. Beaulac and Reckhow (1982) estimate that the range of total nitrogen load is 3 to 70 lb/ac-yr, with the 25 percent to 75 percent quartiles about 4 to 20 lb/ac-yr; the range of total phosphorus cropland load is about 1 to 17 lb/ac-yr, with the 25 percent to 75 percent quartiles about 4 to 6 lb/ac-yr. These values are consistent with the estimates of Langland et al. (1995) for cropland. Alexander et al. (2001) provide SPARROW estimates of total nitrogen loads ranging from 2 to 38 lb/ac-yr and literature values of cropland loads of 0.7 to 71 lb/ac-yr. The median target of Phase 5.3 *conventional cropland receiving manures* is set at 44.8 lb/ac-yr. The median phosphorus target is 2.05 lb/ac-yr, consistent with the range of literature values and the previous Phase 4.3 calibration.

10.2.11 *Conservation Tillage Cropland Receiving Manures* EoS Nutrient Targets

The land use *conservation tillage cropland receiving manures* is similar to the *conventional tillage cropland receiving manures* land use, but the tillage practice is changed in order to represent a cropland field operation that leaves at least 35 percent cover at the time of planting

The median target of Phase 5.3 *conservation cropland receiving manures* is set at 39.6 lb/ac-yr. This land use has the same base flow dissolved nitrogen calibration target as *conventional cropland receiving manures* but 18 percent less surface runoff and loads of particulate and dissolved nitrogen (Simpson 2007). The median phosphorus target is about 15 percent less than *conventional tillage cropland receiving manures*, or 1.73 lb/ac-yr (Simpson 2007).

10.2.12 *Conventional Tillage Cropland Without Manures* EoS Nutrient Targets

The *conventional tillage cropland without manures* land use in Phase 5.3 has a specific operational definition and is a collection of land uses defined in the Agricultural Census. The land use includes vegetables, tobacco, potatoes, peanuts, and cotton, all crops that do not receive manure fertilizer and are usually for direct human or animal consumption. Though not receiving manure nutrients, many of these crops, such as vegetables and tobacco, typically have high fertilizer inputs and extensive tillage operations. For these reasons, the *conventional tillage cropland without manures* nitrogen expert target is set to 40.2 lb/ac-yr and the phosphors export target is set to 3.08 lb/ac-yr.

10.2.13 Nutrient Management Hay, Pasture, Alfalfa, Conventional Tillage, Conventional Tillage Receiving Manures and Conservation Tillage Receiving Manures

This group of six nutrient management land uses have all the same characteristics as their original land use except that the BMP of nutrient management is applied, which reduces their export accordingly. The nutrient management BMP requires a separate land use for every type of land use to which it is applied because it is simulated as a reduction in fertilizer inputs; therefore, nutrient uptake and export are different from those of the original land use at every time step of the simulation. The same calibration parameters are used as in the original land use, and the export targets are not set *a priori* but are calculated by the Phase 5.3 simulation. Nutrient management land uses usually have less nutrient export than the original land use as a consequence of the reduced input loads. The calculated nutrient management median and mean loads are in tables 10-1 and 10-2 for total nitrogen and total phosphorus respectively.

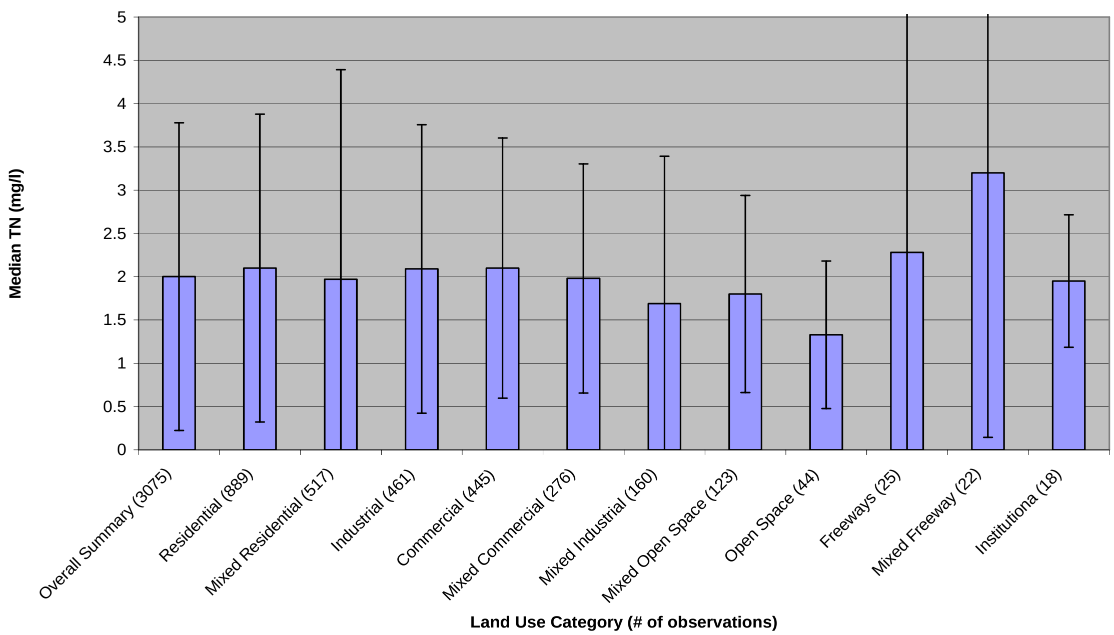
10.2.14 Developed Land EoS Nutrient Targets

There are five classifications of developed land*: low-intensity pervious, low-intensity impervious, high-intensity pervious, high-intensity impervious*, and *bare-construction*. The nutrient simulation of *bare-construction* land use is described in Section 10.2.1.18; all other developed land is described in Section 10.2.1.15. Key nutrient inputs to developed land include atmospheric deposition to pervious and impervious developed lands; urban fertilizer to pervious developed land only; and miscellaneous nutrient inputs from wildlife, pets, and curbside urban refuse to both. In the Phase 5.3 simulation of developed lands, a full mass balance approach is not used for all these inputs. Rather, the simulation of the magnitude of exported nutrient loads from developed lands is directly dependent on the volume of water discharged from *high-intensity developed* and *low-intensity developed* lands.

In the early stages of the Phase 5.3 Model development, when key land use decisions were being made, it was assumed that the NPDES Phase 1 stormwater data would be a resource for determining load differences among developed land uses through characteristics such as development intensity; hence, *high-* and *low-intensity developed* land uses were created for Phase 5.3. After the collection and analysis of the Phase 1 data, little predictive ability for differences in water quality concentrations has been found among different Chesapeake Bay watershed developed land uses, as shown in Figures 10-3 and 10-4 (Pitt et al. 2004; Maestre and Pitt 2005).

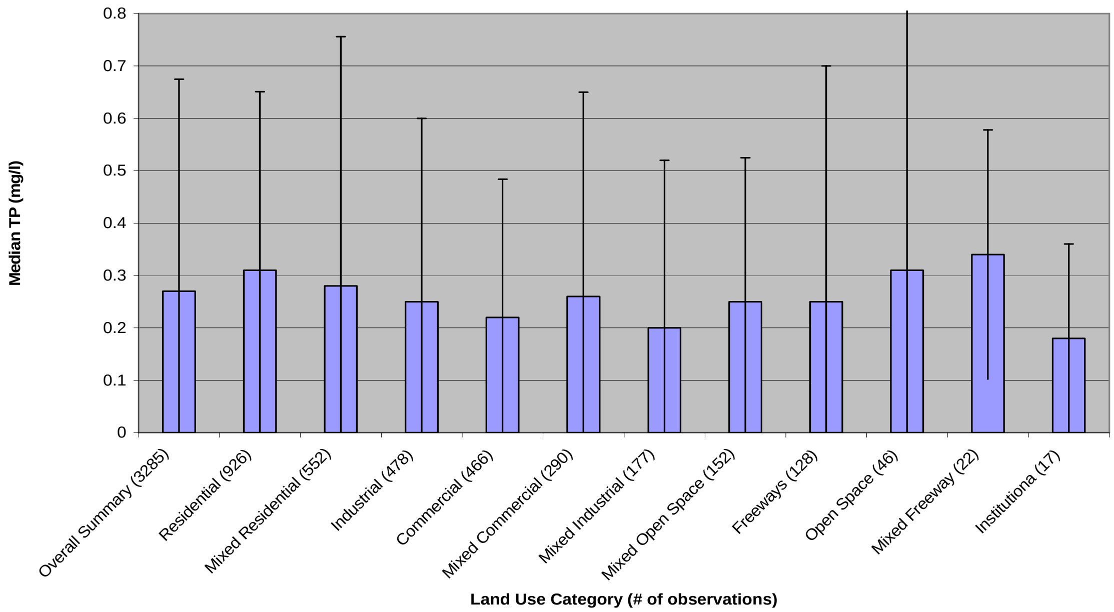
Figures 10-3 and 10-4 show the median concentrations of total nitrogen and total phosphorus for different land classifications and the number of observations in that classification. For the most part, the medians show remarkably little variability between land uses, especially when compared to the standard deviation.

Even though few differences have found in nutrient concentrations from different developed land uses, it is clear that the total load of nutrients from land uses with increased imperviousness has increased runoff or discharge volume, and the increased runoff combined with the constant concentration increases the nutrient loads from developed impervious land uses. The Phase 5.3 simulation uses this information to advantage through the fine resolution of imperviousness associated with each Phase 5.3 land-river-segment (Figure 4-4), as described below. For calculation of the developed land expected load, the overall median concentrations of 2.0 mg/l total nitrogen and 0.27 mg/l total phosphorus are used.



Note: Error bars are one standard deviation.

Figure 10-3. Median TN concentration in NPDES Phase I stormwater data using data from Pitt (undated).



Note: Error bars are one standard deviation.

Figure 10-4. Median TP concentration in NPDES Phase I stormwater data using data from Pitt (undated).

10.2.15 *Low-Intensity Pervious, Low-Intensity Impervious, High-Intensity Pervious, High-Intensity Impervious Developed* Land EoS Nutrient Targets

A standard practice for estimating nutrient loads from developed land is the simple method, in which the annual nutrient load is determined by the annual runoff multiplied by the median event mean concentration (EMC) (Schueler 1987; Pitt et al. 2004). The annual runoff is typically estimated from rainfall, detention storage, and the runoff coefficient or, in the case of the Phase 5.3 simulation, is directly simulated. The runoff estimates are taken directly from model output. The annual discharge of total surface water and groundwater from the Phase 5.3 model represents runoff, which is consistent with the Phase 1 observed data. Simply multiplying the annual discharge by the concentration gives an estimate of loading.

Although a single total phosphorus concentration of 0.27 mg/l is used for *high-intensity* and *low-intensity pervious developed* land, as well as for *high-intensity* and *low-intensity impervious developed* land, the annual discharge is unique for the calibration of developed land in every land-segment. Therefore, different land-segments have different total phosphorus loads depending on the precipitation to each particular land-segment and the resultant estimated discharge. The simulation is further spatially refined by the use of imperviousness estimated on every land-river-segment (Figure 4.4). A percent imperviousness is determined for both the *high-intensity* and *low-intensity developed* land use at the land-river-segment scale. The percent imperviousness provides an estimate of the mix of pervious and impervious area to be simulated on each Phase 5.3 land-river-segment. For example, if a land-river-segment was estimated to have 100 acres of *low-intensity developed* land and the percent imperviousness was 10 percent, then 10 acres of *impervious low- intensity developed* land and 90 acres of *pervious low-intensity* *developed* land would be simulated. The same would be done for *high-intensity developed* land, though typically with estimates of higher imperviousness.

While the Phase 5.3 simulation does not distinguish between the nutrient loads of the *high-intensity* and *low-intensity developed* land uses, this is tracked as unique areas for each at the land-river-segments and reported as model outputs of loads from *high-intensity* and *low-intensity developed* land use. Typically, the pervious and impervious land uses are combined to represent the total load from *high-intensity* and *low-intensity developed* lands.

For total nitrogen the approach is similar, but it also takes into account the varying nitrogen loads from atmospheric deposition. For nitrogen loads from developed lands, both atmospheric loading differences and discharge volume are tracked at the land-segment scale, as well as pervious-impervious differences at the land-river-segment scale, as described in the phosphorus simulation.

Simulating developed lands with PQUAL requires the development of target for major species of nitrogen, including nitrate, ammonia, and organic nitrogen. Maestre and Pitt (2005) found a relationship between percent impervious and NO_3+NO_2 concentration with an intercept (completely pervious) of 0.756 and a slope of -0.009 mg/l per percent impervious. Following this estimate, one would calculate negative concentration for completely impervious urban land, which is an untenable Phase 5.3 calibration outcome. A way to deal with this problem, and one that is well within the range of error, is to assume that the concentration of NO_3+NO_2 is 0.8 for pervious land and 0 for impervious land. This is something of an extrapolation beyond the data, but since developed land loads by watershed are always a combination of pervious and impervious loads, this method provides reasonable results that match the slope found by Maestre and Pitt.

No relationship between total Kjeldahl nitrogen (TKN) and land use was found, so a constant value is used. Maintaining a weighted average concentration of 2.0 mg/l total nitrogen for all developed land and the above concentrations of NO_3+NO_2 gives a constant TKN of 1.4 mg/l, which is the median from Maestre and Pitt (2005) for all land use types. Multiplying by the runoff gives an median nitrogen load of about 10 lb/ac-year for pervious developed and about 11 lb/ac-year for the impervious developed median for the overall Phase 5.3 domain. Alexander et al. (2001) tabulate SPARROW estimates of urban, or developed, total nitrogen loads to be 3.2 to 156 lb/ac-yr and cite literature values of 1.4 to 34.3 1b/ac-yr.

The atmospheric loadings are estimated, and the output from developed lands, especially impervious developed, can be expected to be related to the amount of atmospheric deposition received. The area-weighted atmospheric deposition of nitrogen on urban land is 13.7 lb/ac-yr. The targets for impervious land are calculated by dividing the deposition for each segment by the average and using the result as a multiplier for the concentration. The same method is used for the pervious targets. The total nitrogen load input to pervious land is assumed to be only about half atmospheric deposition and the other half fertilizer and other sources, so the effect of the atmospheric deposition multiplier is halved.

For the pervious land use, the Phase 5.3 domain average concentration for NO_3+NO_2 is 0.8 mg/l and that for TKN is 1.45 mg/l, but this average concentration is adjusted by the degree of atmospheric deposition on each land-segment. For the impervious land use, the Phase 5.3 domain average TNK concentration is 1.45 ml/l only with no simulated nitrate loads as described above. For total nitrogen for both the pervious and impervious land uses, the weighted average concentration is 2.0 mg/l.

10.2.16 *Nursery* EoS Nutrient Targets

In the Phase 5.3 simulation, the *nursery* land use represents container nurseries, which typically have a high density of plants (10 to 100 plants per square meter) and high rates of nutrient applications. Annual fertilizer application rates are in the range of 350 lb to 1000 lb or more of nitrogen per acre, with equivalent application rates for phosphorus of 100 to 300 lb/ac with use efficiencies in the neighborhood of 10 to 20 percent.

The container nurseries feature a very high density of plants (10 to 100 plants per square meter are not uncommon) and therefore have very high rates of nutrient application. Although the plant density contributes much to these high rates, plant use efficiency is another big factor, with the nutrient use efficiencies of the plants ranging from 10 to 30 percent (Lea-Cox and Ristvey 2007). The phosphorus efficiencies are relatively lower than the nitrogen efficiencies because these operations use an organic growth medium that has little ability to trap anions such as phosphorus. Historically, it was not uncommon for growers to use 20-20-20 (nitrogen-phosphorus-potassium) as an all-purpose fertilizer, meaning that the phosphorus export rates could be even higher; however, it is thought that this practice has abated somewhat.

Nutrients unused for plant growth or lost by denitrification, volatilization, or sorption are discharged. Many but not all operations have a collection system for the purposes of either detaining or reusing the greenhouse runoff. Systems that have only detainment might have significant denitrification, but phosphorus export could still be high. Reuse systems are less common than detainment, but they do exist, and in at least one example a sophisticated collection and reuse system, in which water is recycled through the system at least five times, is used.

To simulate *nurseries*, nitrogen and phosphorus application rates are assumed as 600 and 200 lb, respectively. Furthermore, a 20 percent nitrogen and a 15 percent phosphorus utilization rate by the nursery plants is assumed, which removes by plant uptake 120 and 30 lb of nitrogen and phosphorus, respectively. Another 50 percent loss through volatilization, denitrification, or other losses through detainment or reuse systems are also assumed, resulting in a final estimated median target load of 254 lb/ac-yr for nitrogen and 112 lb/ac-yr for phosphorus.

10.2.17 *Bare-Construction* EoS Nutrient Targets

Although construction and extractive land uses have very small acreages within the watershed, they are important because of their high sediment and nutrient loading. Nutrient inputs are from atmospheric deposition only, and nutrient export is of this atmospheric load, largely unattenuated, and of mass wasting of the soil. In construction sites, large amounts of loose, disturbed soil is easily eroded as discussed in Section 9, resulting in high export loads of sediment and nutrients, particularly phosphorus.

There is little literature available regarding nutrient export rates from construction sites. Line et al. (2002) report total nitrogen and total phosphorus export rates of 7.4 and 2.6 lb/ac-yr, respectively, for the clearing and grading phase of residential construction, and 32.2 and 1.2 lb/ac-yr during house, road, and storm drain construction. Daniel et al. (1979) reports values ranging from 12.2 to 49.5 lb/ac-yr for total nitrogen and 6.7 to 17.9 lb/ac-yr for total phosphorus. For Phase 5.3, the median target values of 26.4 lb/ac-yr for total nitrogen and 8.81 lb/ac-yr for phosphorus were chosen. These target loads are what's estimated to be exported from the *bare-construction* land use without BMP controls, and the nutrient loads are decreased with the application of erosion and sediment control BMPs, as described in Section 6.

10.2.18 *Extractive* Land Use EoS Nutrient Targets

There is little literature available regarding nutrient export rates from active and abandoned mine sites. Brabets (1984) reports similar nutrient loading from the land areas as the reference sited above for the *bare-construction* land use. To represent the nutrient exports from the *extractive* land use, about half that of the *bare-construction* estimated nutrient export rate is used, translating into 13.1 and 4.42 lb/ac-yr for total nitrogen and total phosphorus, median target loads respectively.

10.2.19 Water Surfaces

Water surfaces also have nutrient input loads from deposition. The nutrient load input to the *open water* land use is assumed to be entirely from atmospheric deposition. The daily wet and dry deposition of dissolved inorganic nitrogen is obtained from regression and air quality models, as described in Section 5 and is estimated to have a mean value of 12.5 lb/ac-yr for water surfaces in the Chesapeake watershed over the 1985-2005 calibration period. Organic nitrogen disposition loads and aeolian phosphorus load deposition are also components of atmospheric deposition to open water and are described below.

The same estimated atmospheric organic nitrogen loads are used in both the watershed and tidal Bay models. Both models load atmospheric organic nitrogen to only water surfaces on the assumption that all organic nitrogen is derived from aeolian processes, which result in no net change in organic nitrogen on terrestrial surfaces but a net gain to water surfaces. Organic nitrogen is represented as wet fall only or dissolved organic nitrogen (DON). The magnitude of dry fall organic nitrogen is unknown.

In the 1992 versions of the watershed and tributary models, organic nitrogen is assumed to be about 670 ug/l (as N), based on data summarized by Smullen et al. (1982). The data showed considerable seasonal variability. The organic nitrogen load was constant in all watershed model segments. An equivalent annual load was used in the tributary model with application of the seasonal variability suggested by Smullen.

Organic nitrogen measurements from Bermuda are calculated at about 100 ug/l (as N) (Knap et al. 1986; Knap, Jickells et al. 1986). Moper and Zika (1887) reported an average DON concentration from the western Atlantic and Gulf of Mexico of about 100 ug/l (as N). This is consistent with the reported range from the North Sea and northeast Atlantic of between 90 ug/l and 120 ug/l (Scudlark and Church 1993). A recent study reports an annual volume-weighted average DON concentration in the mid-Atlantic coastal areas to be about 130 ug/l (as N) (Scudlark, Russel et al. 1996). Measurements in this study are consistent with the interannual variation (maximum in spring) reported by Smullen.

In the Phase 5.3 model, 130 ug/l (as N) is used as representative of an average annual wet deposition concentration to the watershed and tidal waters with the seasonal loading pattern suggested by Smullen (1982) and Scudlark et al.(1996). This will apply an average concentration of 98 ug/l from July to March rainfall and an average concentration of 224 ug/l from April to June. On average, this is an annual load of organic nitrogen of about 0.8 lb/ac-yr. Organic nitrogen deposition, which includes pollen, amino acids, and other relatively biologically available materials, is arbitrarily considered to be 30 percent labile organic nitrogen and 70 percent refractory.

10.3 Nutrient Calibration Decision Rules

The nutrient calibration was carried out separately for each land use in all land-segments. Since there are a large number of target loads to be reached, a manual calibration of nutrient export would be difficult. As a result, an automatic iterative method (AIM) was developed to carry out the land use nutrient calibration. The AIM iteratively adjusts model parameters based on a set of decision rules, which relate the sensitivity of model parameters to particular model outputs in attempting to match EoS loadings and county-level crop yield data. Calibration starts from the parameter set from manual calibration, and a group of computer programs control and implement the procedure.

The AIM is superior to a conventional trial-and-error approach in several respects. The calibration procedure is operated entirely automatically, offering the kind of efficiency required for a feasible calibration of the Phase 5.3 Model. The calibration methodology has clear rules and objectives, thus eliminating the potential subjective nature of individual model practitioners performing a calibration. The AIM ensures an equitable treatment of all land uses across the Phase 5.3 domain. Moreover, the procedure is repeatable, allowing an efficient model recalibration whenever new data are available and an update is needed.

Depending on the land uses, the Phase 5.3 model uses the HSPF AGCHEM or PQUAL module to simulate land nutrient processes. The AGCHEM module is a complete mass balance of nutrients and includes features like plant uptake and soil transformation of nutrient species (Bicknell et al. 2001). The PQUAL module is a supplier approach that simply associates a nutrient concentration with the load of sediment exported from a land use.

The AGCHEM module is applied to major land uses with nutrient application, such as cropland and other agricultural land uses, whereas the PQUAL module is used for either land uses receiving only atmospheric deposition or land uses with intensive animal waste where a detailed mass-balance approach would be difficult to apply, such as in the land use of animal feeding operations. Consequently, different decision rules are developed for different land uses, for both nitrogen and phosphorus, based on the module used. Within the AGCHEM module, the decision rules are further differentiated by the plant uptake methods and availability of field uptake data.

10.3.1 AGCHEM Nitrogen Calibration Rules

Nine land uses are simulated using the AGCHEM module, including three types of cropland, alfalfa, hay, pasture, two types of pervious urban, and wooded land. For each of these land uses, eight calibration targets are specified for nitrogen calibration, including NO_3, NH_3, labile organic nitrogen, and refractory organic nitrogen for both groundwater and surface layers (Figure 10.5). The calibration targets are the estimated average annual nutrient loads for these land uses, and all of the 1985 to 2005 simulation years were used to form the Phase 5.3 annual average, which was compared to the calibration targets.

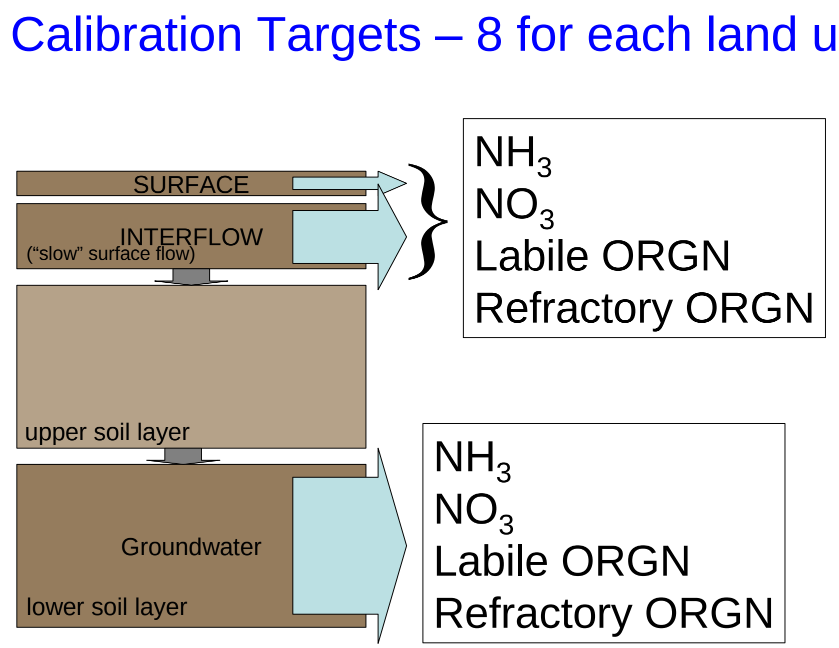


Figure 10-5. Eight calibration targets for nutrient species in both surface and base flow.

The nitrogen calibration objective is to reach each of the eight average annual target loads by adjusting relevant parameters. Nitrogen is simulated in the AGCHEM module in detailed soil processes, which involve about 100 parameters in four soil layers. Among them, however, eight are particularly sensitive and used for the calibration:

- KAM - organic N ammonification
- KDNI - denitrification rate of NO_3
- KNI - nitrification rate
- KIMAM - rate of return of ammonium to organic nitrogen
- KIMMI - rate of return of nitrate to organic nitrogen
- KLON - particulate/soluble partitioning coefficient for labile organic N
- KRON - particulate/soluble partitioning coefficient for refractory organic N
- WILTP - wilting point in yield-based uptake method.

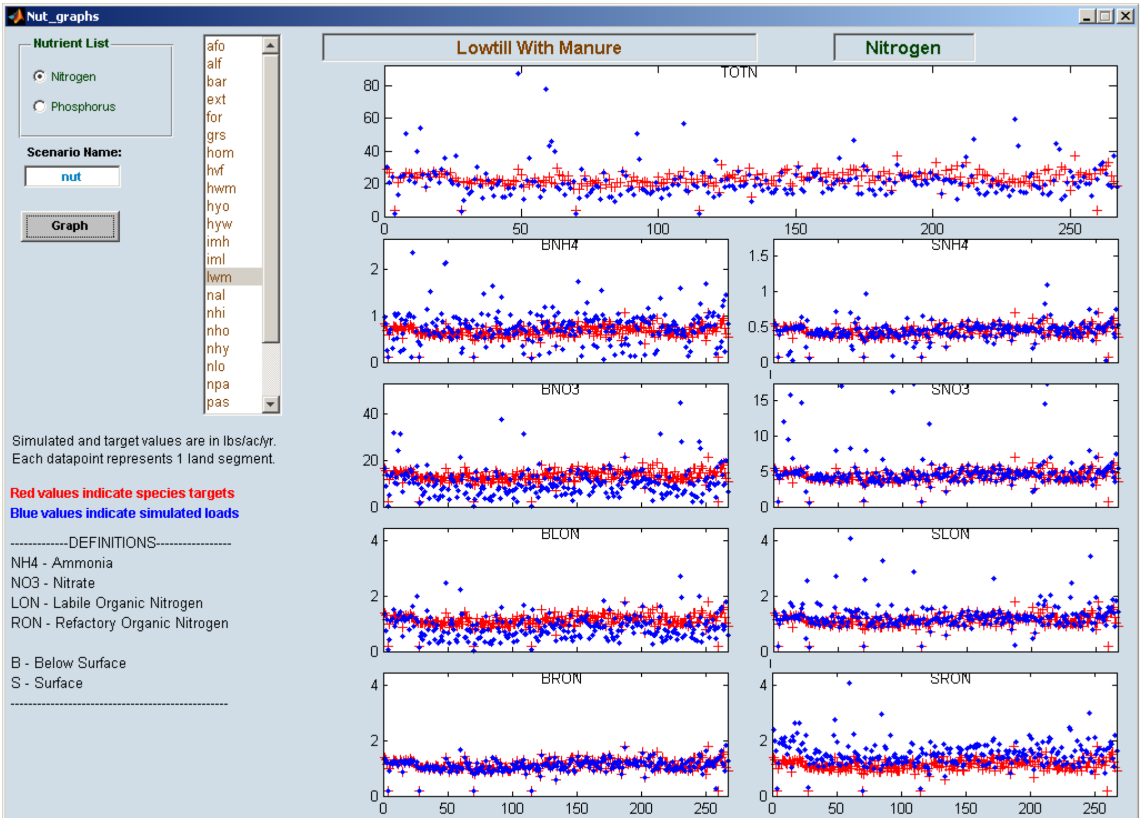
The KIMMI and KIMAM variables are unusual because they are unrepresentative of actual nutrient transformation pathways, but they are a way to represent the uptake of nitrate or ammonia by other plants in cropland such as weeds, or by soil bacteria, and they are a separate pathway to return inorganic nitrogen to organic nitrogen (other than that of the simulated crop).

Each of the above parameters requires four values, one for each soil layer. The parameters in the first two soil layers, surface flow and interflow, affect surface loading; the parameters in the third and fourth layers, consisting mainly of groundwater, affect subsurface loading.

The agricultural lands of *conventional tillage cropland without manures, conservation cropland receiving manures, conventional cropland receiving manures, alfalfa,* and *hay-fertilized* are all simulated using a yield-based uptake method based on the Agricultural Census crop yield data. For these land uses, four decision rules are developed to regulate parameters to meet nitrogen targets as well as match crop yield data:

- KAM is adjusted to balance organic nitrogen and inorganic nitrogen.
- KNI and KDNI (below surface only) are used to reach NH_4 and NO_3 targets.
- KLON/KRON is used to reach organic nitrogen targets.
- KIMAM/KIMNI and WILTP are used to reach uptake data.

Figure 10.6 is a graphic of the initial calibrated *conservation cropland receiving manures* land use. All the land-segments are displayed on the x-axis and are identified with a number designate, and the simulated annual export load is displayed in units of pound/acre-year on the y-axis. In this case, the initial total nitrogen target is a little over 20 lb/ac-yr and varies due to the relative amounts of fertilizer and manure loads and crop yields, as described in Section 5. Most of this nitrogen is in the form of nitrate, particularly subsurface nitrate. This initial target load was subsequently increased as described above based on guidance from the riverine calibration stations for total nitrogen.



TOTN = total nitrogen, BNH4 = subsurface ammonia, SNH4 = surface ammonia, BNO3 = subsurface nitrate, BLON = subsurface labile organic nitrogen, SLON = surface labile organic nitrogen, BRON = subsurface refractory organic nitrogen, SRON = surface refractory organic nitrogen.

Figure 10-6. A plot of the *conservation cropland receiving manures* calibration. All the land-segments are displayed on the x-axis with a number designate, and the simulated annual export load is displayed in units of pounds per acre per year on the y-axis.

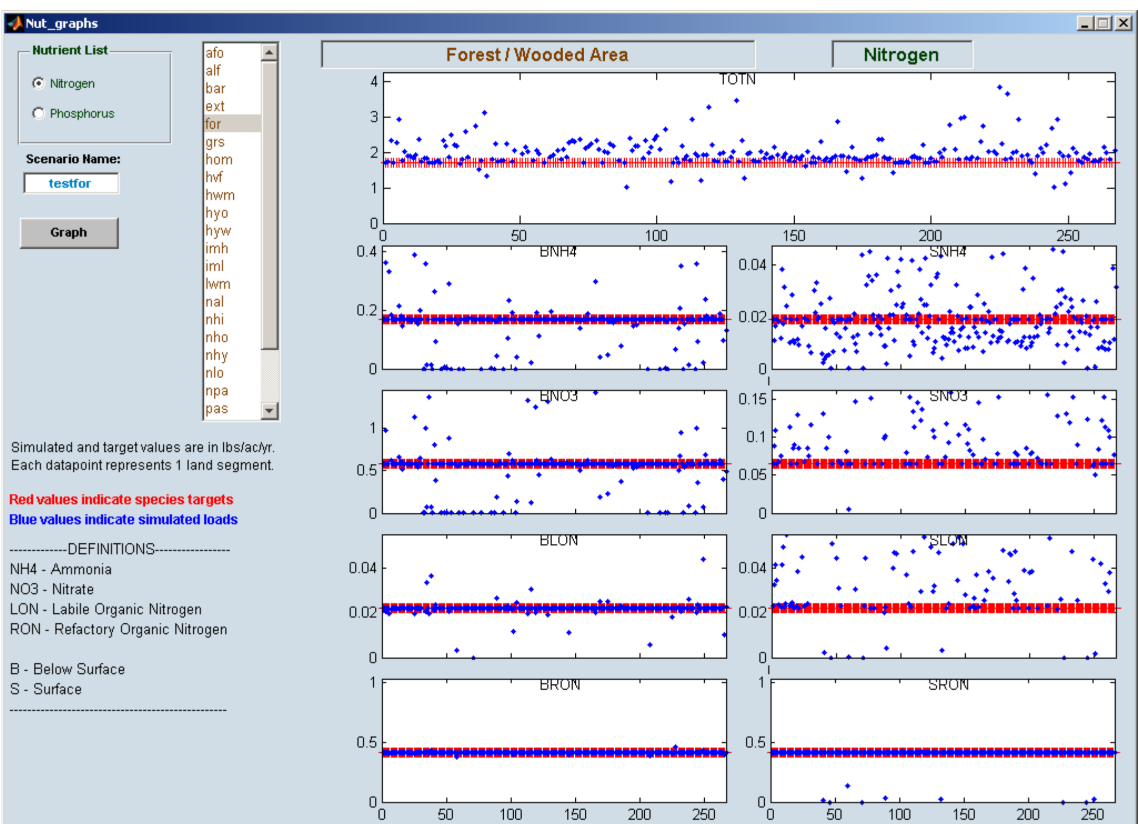
Hay-fertilized used the same decision rules as the above land uses for the same parameters, but it also has two additional rules. The first is to adjust the plant uptake rate to allow more uptake to remove excess manure, and the second is to optimize the ammonia volatilization rate by increasing it when surface DIN concentrations are 20 percent higher than the target DIN concentration.

Pasture is handled differently from the above land uses in two aspects. The plant uptake in *pasture* is simulated by a first-order method in HSPF. Consequently, wilting points are not applicable and only NH_4/NO_3 immobilization rate are adjusted to reach available uptake data. Also, the ammonia volatilization rate is optimized for pasture instead of remaining static as in other land uses, in order to get rid of NH_4 when surface DIN is 20 percent higher than targets. There are no yield data available for *pasture*, but we assumed that 70 percent of the inputs go to uptake and thereby estimated the uptake target from the input data.

The AGCHEM module does not reset the plant nitrogen storage every year to reflect plant harvest. To represent this in Phase 5.3, the daily first order return rate of plant biomass to soil organic nitrogen is set relatively high at about 0.02 after the crop harvest. This returns all the plant biomass to the soil by the next planting season. Of the returned plant nitrogen, 40 percent is returned to the soil as refractory organic nitrogen each year and the rest is returned as labile organic nitrogen. One effect this has is to increase slightly the refractory organic nitrogen export over time, but this increase is small, on the order of about 5 percent, and the overall approach is considered the best approach to achieving continuous simulations of soil nutrients given the limitations of HSPF.

Nitrogen calibration for the *forest, woodlots, and wooded* land use is focused on reaching EoS targets and also maintaining a stable storage over time. The model parameters are adjusted with the same decision rules used for croplands. In a manner similar to the cropland simulation, KIMMI and KIMAM are used to reach forest nitrogen uptake targets, which are estimated to be 60 lb/ac-yr (Hunsacker et al. 1993).

Maintaining a stable storage is the key to ensuring a reasonable simulation of the forest nitrogen cycle, and it is done by resetting the initial storage to the average storage of the last three years' simulation after each iteration of the AIM calibration. The reset ensures the initial storages are compatible with adjusted parameter values so that nitrogen storage in soil and trees will not be depleted over time. Figure 10.7 shows the results of the *forest*, *woodlots, and wooded* simulation for all the Phase 5.3 land-segments. This initial target load was subsequently increased as described above based on guidance from the riverine calibration stations for total nitrogen.



TOTN = total nitrogen, BNH4 = subsurface ammonia, SNH4 = surface ammonia, BNO3 = subsurface nitrate, BLON = subsurface labile organic nitrogen, SLON = surface labile organic nitrogen, BRON = subsurface refractory organic nitrogen, SRON = surface refractory organic nitrogen.

Figure 10-7. Results of the *forest, woodlots, and wooded* simulation for all the Phase 5.3 land-segments.

The calibration for the *high-intensity* and *low-intensity developed* pervious land uses and *hay-unfertilized* land use is done the same way as for the cropland for the most part, except that the KIMAM is used to reach a balance between organic N and inorganic N rather than matching uptake data as for croplands because there are no plant uptake data available for these land uses.

The initial storages for labile and refractory organic nitrogen are optimized as well. The optimization is done by resetting the initial storages after each iteration to the median storages of the entire simulation period. This approach is applied to all above land uses with the exception of forest. This intends to eliminate, for example, large spikes in the data for the first year due to a high initial storage, and it stabilizes the storage of nutrients over time.

10.3.2 AGCHEM Phosphorus Calibration Rules

Phosphorus calibration is relatively straightforward because the simulated phosphorus cycle in HSPF's AGCHEM module is less complex than the simulation of nitrogen in HSPF (Bicknell et al. 2001). The AGCHEM phosphorus simulation is focused on the fate and transformation of PO_4, while the simulation of organic P is largely left to a ratio of the simulated organic N; a Redfield stoichiometric ratio of 16:1 nitrogen to phosphorus is used.

The objective of the AGCHEM phosphorus calibration is to match PO_4 targets for both surface water and groundwater. Three parameters are sensitive in the phosphorus simulation and are used for the calibration. As with the nitrogen simulation, each parameter requires four values, one for each soil layer.

- KIMP - rate of return of phosphate to organic phosphorus
- KMP - organic P mineralization rate
- K1 - coefficient for the Freundlich adsorption/desorption equation.

The KIMP variable is like the KIMMI and KIMAM variables and represents the uptake of phosphate by other plants in the cropland land use such as weeds, or by soil bacteria, and is a separate pathway to return inorganic phosphate to organic phosphorus other than that of the simulated crop. Only four agricultural land uses—*conventional tillage cropland without manures, conservation cropland receiving manures, conventional cropland receiving manures*, and *alfalfa*— are simulated with the AGCHEM module. The rest of the land uses are simulated using the PQUAL module.

For these land uses, the phosphorus uptake is simulated by a first-order method and crop yield data are available. Three decision rules are developed to calibrate parameters to meet the phosphate targets as well as match crop yield data:

- Adjust KIMP to control transformation of PO_4.
- Adjust KMP to reach uptake targets.
- Adjust K1 to reach PO_4 targets.

The initial storages are not optimized in the phosphorus calibration and stay at the pre-specified values throughout the optimization process.

10.3.3 PQUAL Calibration Rules

PQUAL is used to simulate nitrogen and phosphorus loads for the land uses *bare-construction, extractive, harvested forest, degraded riparian pasture, animal feeding operations, nursery, low-intensity impervious,* and *high-intensity impervious*. These land uses generally have relatively smaller acreages and are assumed to receive only atmospheric deposition inputs or, in the case of animal feeding operations, only extremely high levels of manure inputs.

For wooded and urban land uses, PQUAL is used for the phosphorus simulation
For the *forest, woodlots, and wooded* land use, there is no manure or fertilizer source of phosphorus. The source of phosphorus in simulated woodland soil is assumed to be produced by natural weathering processes. If the AGCHEM simulation, which is based on mass balance, were used, it would be able to produce reasonable phosphorus export only by using a large initial storage of organic phosphorus as a stand-in for the natural weathering processes. The drawbacks in this approach are that the generation of phosphorus in the soil would be temperature-dependent and the export of phosphorus would decrease over time as the soil storage was depleted. To avoid these problems, the PQUAL simulation, which appropriately enough is sensitive to hydrologic and sediment processes only, was used.

In the case of the *high-intensity* and *low-intensity developed* pervious land uses, the phosphorus that is attributed to the land is thought to be primarily produced by scour of small streams downstream of the urban areas. Again in this case, attempting a mass balance on the land simulation would be inappropriate. Because the mechanism of phosphorus production is scouring, the sensitivities to hydrologic processes in the PQUAL simulation are appropriate.

10.4 Assessment of the Nutrient Calibration

As with the decision rules, the measurements used to assess the nutrient calibration also vary by land uses, depending on the modules by which they are simulated. For the land uses simulated with PQUAL, the only measurement used is the comparison of modeled exports to EoS targets because of the simple mechanism of the PQUAL simulation module. PQUAL calibration can quickly achieve exact EoS targets.

For the land uses simulated using the more complete nutrient simulation of AGCHEM, the assessment of nutrient calibration involves judicious weighting of five key calibration decision rules concerning different aspects of a mass-balance simulation. They are:

1. Modeled TN and TP loads are within 25 percent of targets
2. Model exports of nutrients are sensitive to climate variations and agricultural practices
3. Nutrient storage in the soil profile remains stable over the long term
4. Model parameters are within a reasonable range and in a fairly constrained distribution
5. Model simulated plant uptake is within the reasonable range of field data (where available)

10.4.1 Land Use Calibration to EoS Nutrient Targets

The assessment of the land use calibration to EoS targets is focused primarily on the TN and TP targets because these loads are the ones used to develop management policies. Also, the splits between nutrient species other than TN and TP are less well defined in the literature. The simulation of TN and TP loads within 25 percent of targets is widely accepted as a good calibration, according to the HSPF criteria for annual value (Donigian et al. 1984) and is used as a quantitative measurement of model performance. Figures 10-8 and 10-9 show the percent of land-segments that reached TN and TP targets for each land use.

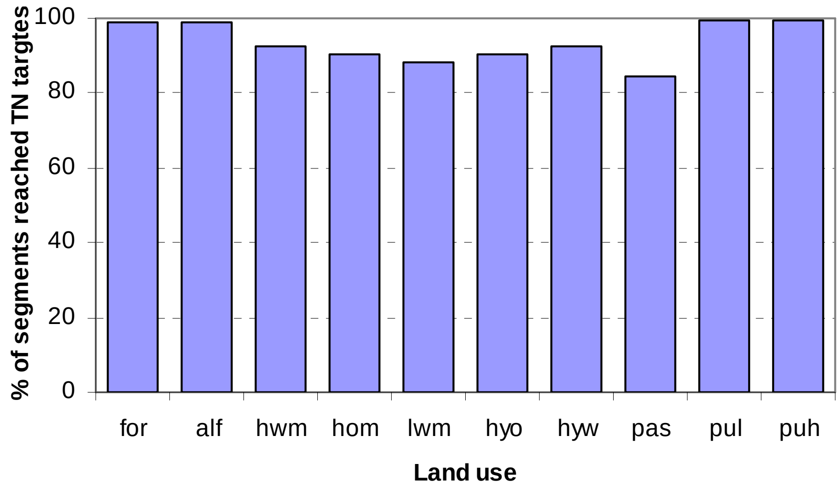


Figure 10-8. Percent of land-segments that reached TN targets for each land use.

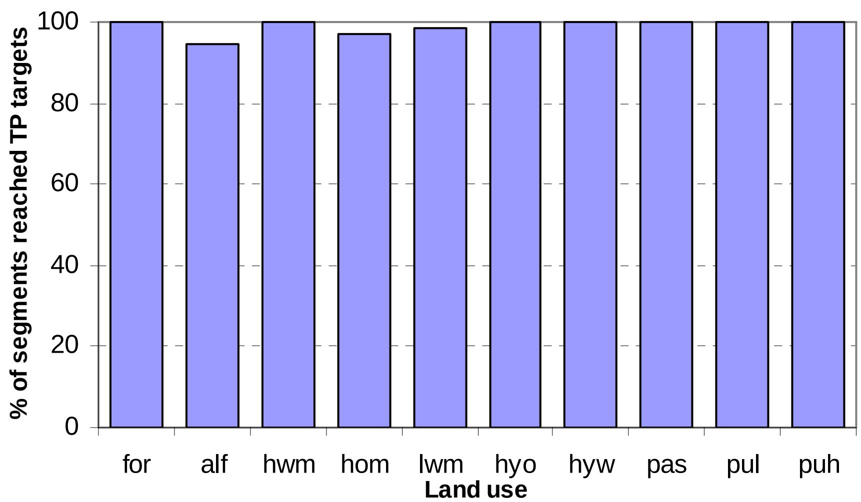


Figure 10-9. Percent of land-segments that reached TP targets for each land use.

10.4.2 Simulation of Plant Nutrient Uptake to Match Estimated Crop Yield Data

An accurate simulation of plant nutrient uptake provides the basis of a sound nutrient calibration. The fate of the majority of the atmospheric, manure, and fertilizer nutrients applied to cropland is primarily to be taken up by the crop. Simulated crop uptake is calibrated to be consistent with Agricultural Census county-based estimates of crop yield. Estimated crop yield data are available for the six agricultural land uses of *conventional cropland receiving manures, conventional tillage cropland without manures, conservation cropland receiving manures, alfalfa, hay-fertilized, and pasture*. Plots of the simulated plant uptake versus estimated crop yield data for each of the six land uses are shown in Figures 10-10 through to 10-115.

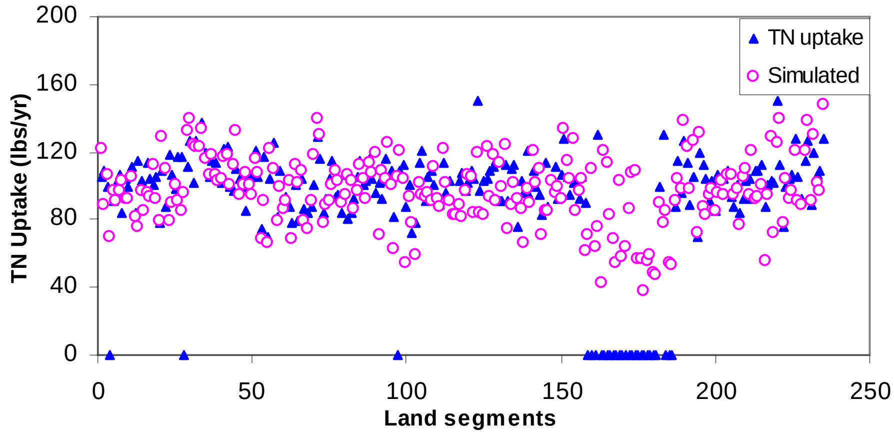


Figure 10-10. Simulated and estimated plant uptake for *conventional cropland receiving manures*.

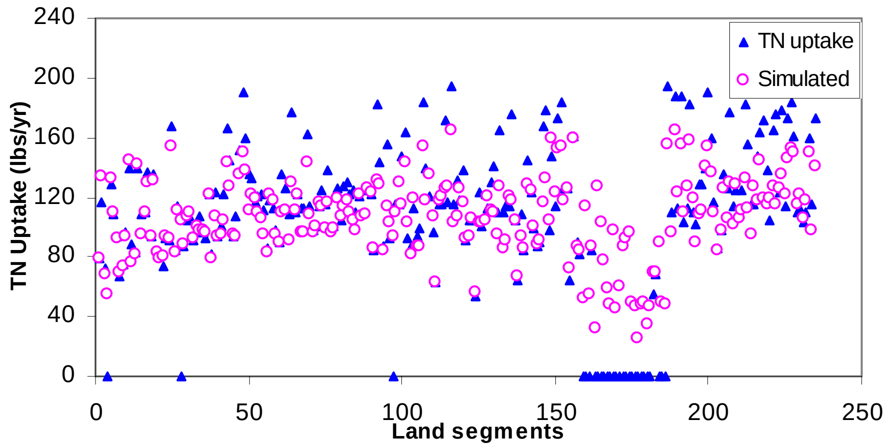


Figure 10-11. Simulated and estimated plant uptake for *conventional tillage cropland without manures*.

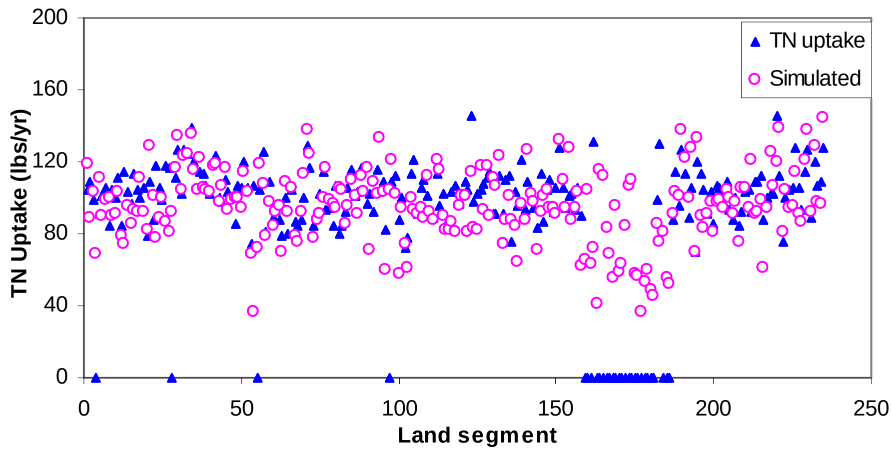


Figure 10-12. Simulated and estimated plant uptake for *conservation cropland receiving manures*.

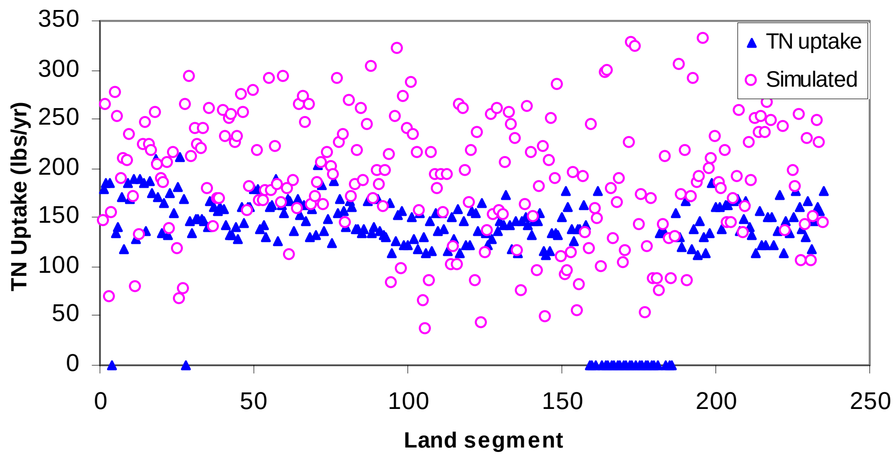


Figure 10-13. Simulated and estimated plant uptake for *alfalfa*.

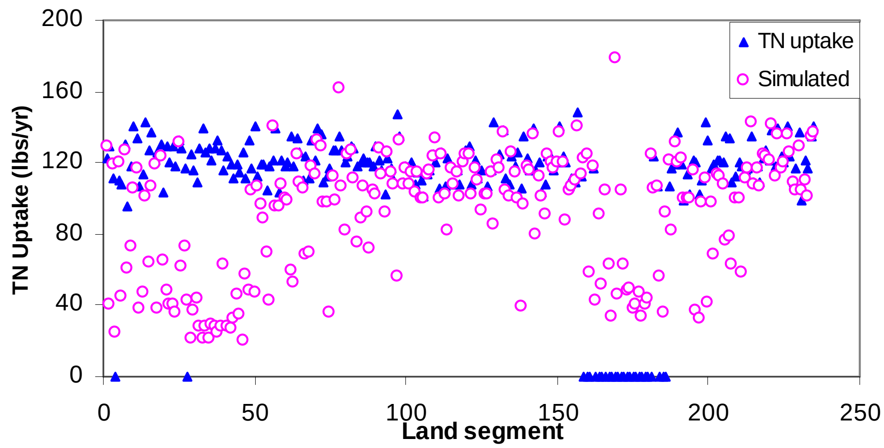


Figure 10-14. Simulated and estimated plant uptake for *hay-fertilized*.

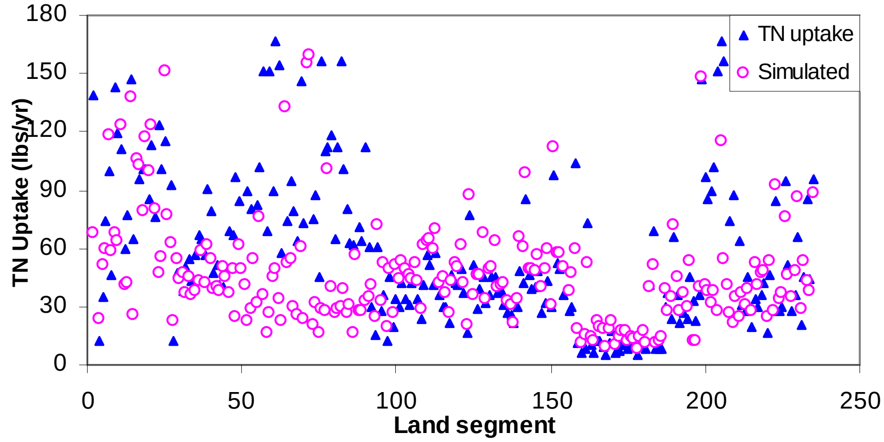


Figure 10-15. Simulated and estimated plant uptake for *pasture*.

10.4.3 Response to Nutrient Management Actions

The calibrated model will be used to develop management scenarios, so it is vital that the model responds reasonably to climate variations and agricultural practices. The sensitivity of model response is tested by comparing the model results for the regular crop with those for the nutrient management crop. The calibrated land parameters were used to run the counterpart of nutrient management land with reduced nutrient input. The model's capacity for simulating management actions should be reflected as reduced nutrient export in total nitrogen and total phosphorus.

However, during the Phase 5.3 nutrient calibration, an approach that led to the same nutrient application rates being applied to nutrient management crops as to regular crops was taken. As a result, the nutrient exports from nutrient management crops show no difference from regular crops, as shown in Figure 10-16 for *conventional cropland receiving manures*. This approach was subsequently found to underestimate the nutrient reduction benefits of nutrient management, and it will be corrected in the upcoming Phase 5.3.2 calibration.

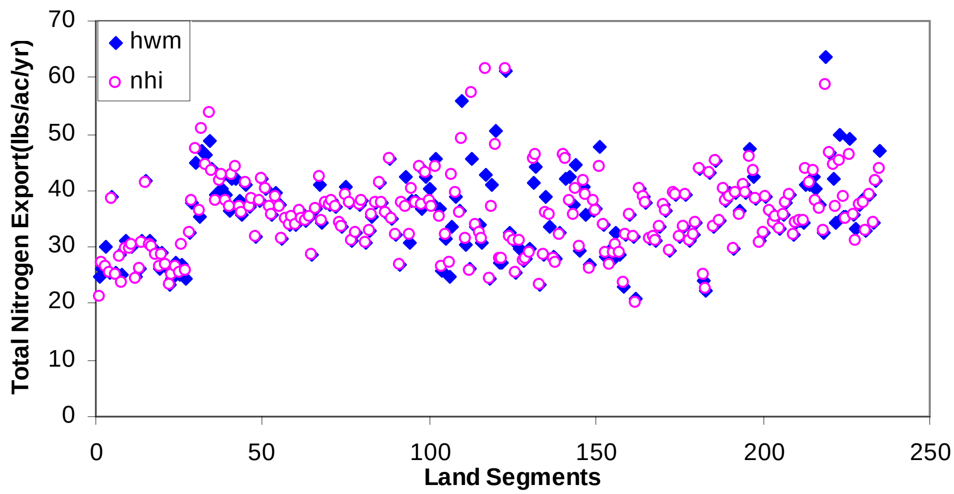


Figure 10-16. Total nitrogen export for *conventional cropland receiving manures* and its nutrient management land use.

10.4.4 Stable Storage Over Time

Nutrient storage in the soil profile should remain stable over the simulation period, particularly for the *forest, woodlots, and wooded* land use and cropland uses that have a large pool of soil organic nitrogen. Given the large numbers of land-segments in the watershed, it is difficult to show this aspect of model calibration in a concise way. As a result, the change of organic nitrogen storage over time for *forest, woodlots, and wooded* land in Prince Edward County, Virginia (land-segment A51147) is shown here as a representative example (Figure 10-17). The nitrogen storages for the other land-segment counties have a similar pattern and remain stable over the 1985 to 2005 simulation period.

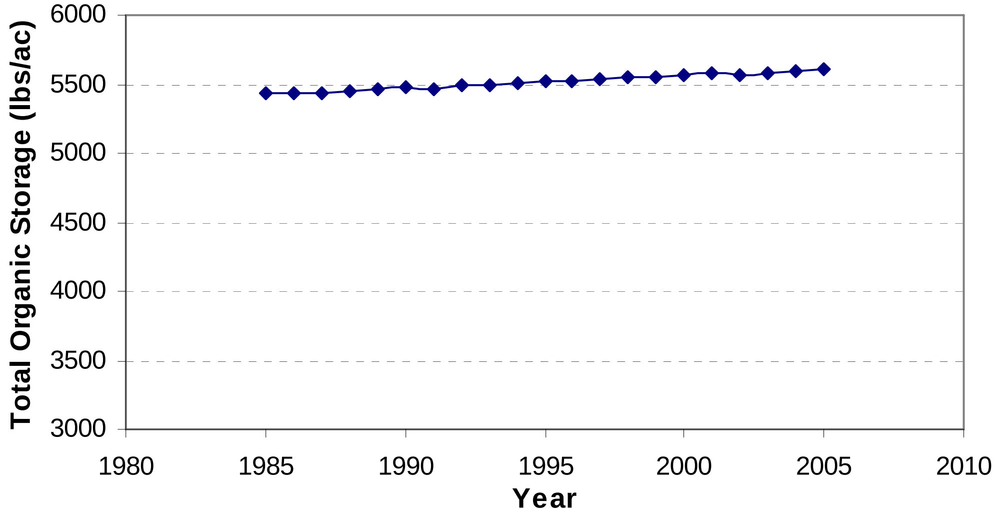


Figure 10-17. The change of organic nitrogen storage over time for *forest, woodlots, and wooded* land in Prince Edward County, Virginia.

10.4.5 Constrained Distribution of Model Parameters

The range of the calibrated model parameters was also evaluated to make sure their values are within a reasonable range and constrained distribution. Figures 10-18 through 10-24 show the percentile of calibrated nitrogen parameters on a log scale. For phosphorus, the majority of the land uses are simulated using PQUAL module, so this measurement is not applicable.

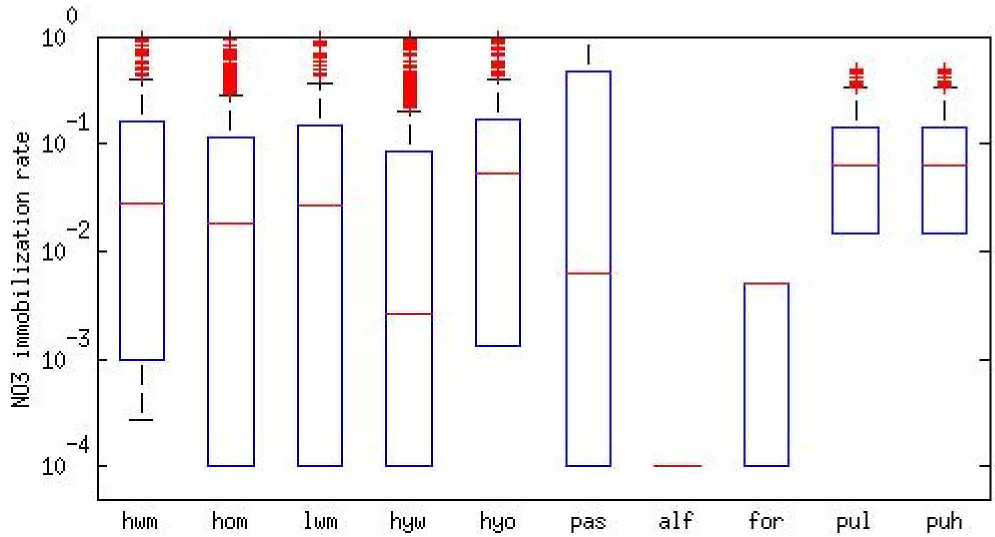


Figure 10-18. Percentile of calibrated NO_3 immobilization rate on a log scale.

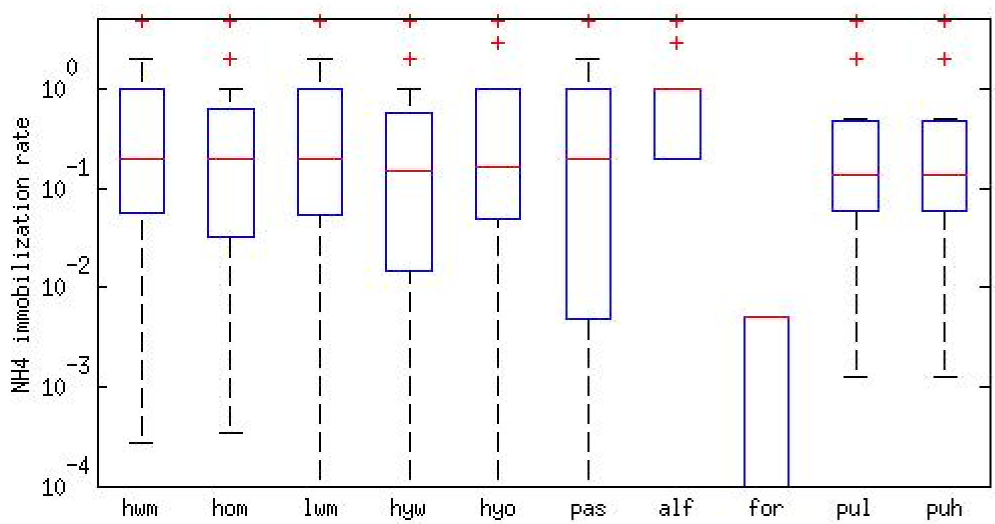


Figure 10-19. Percentile of calibrated NH_4 immobilization rate on a log scale.

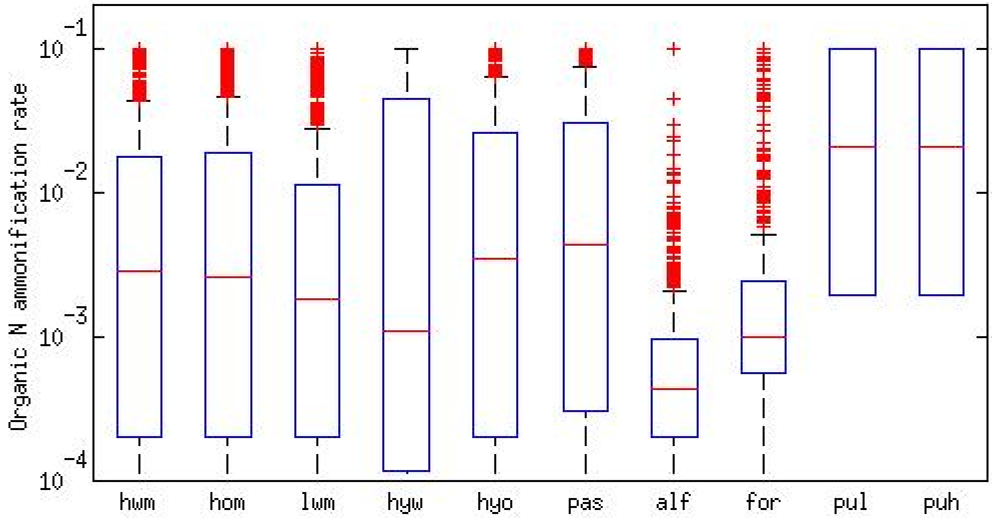


Figure 10-20. Percentile of calibrated organic N ammonification rate on a log scale.

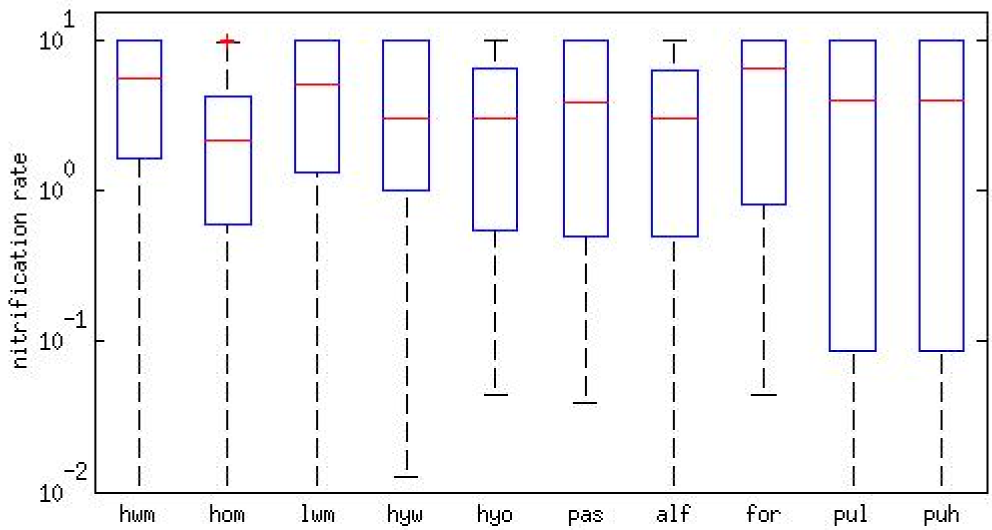


Figure 10-21. Percentile of calibrated nitrification rate on a log scale.

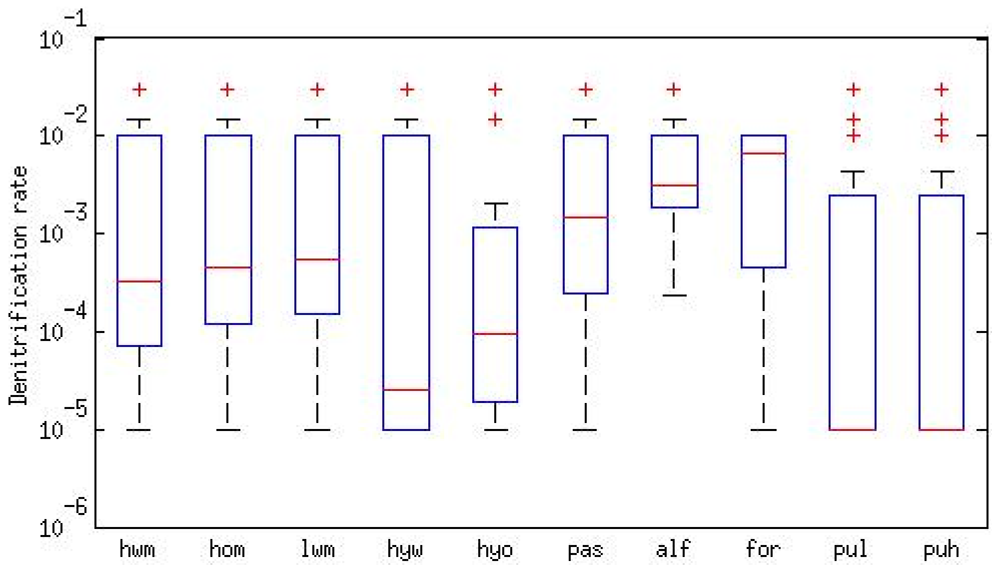


Figure 10-22. Percentile of calibrated denitrification rate on a log scale.

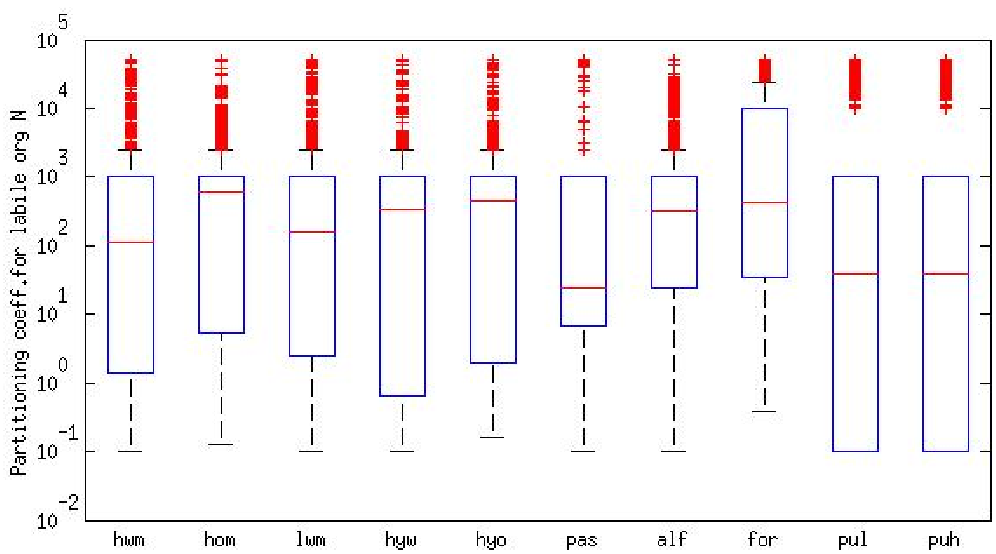


Figure 10-23. Percentile of calibrated partitioning coefficient for labile organic N on a log scale.

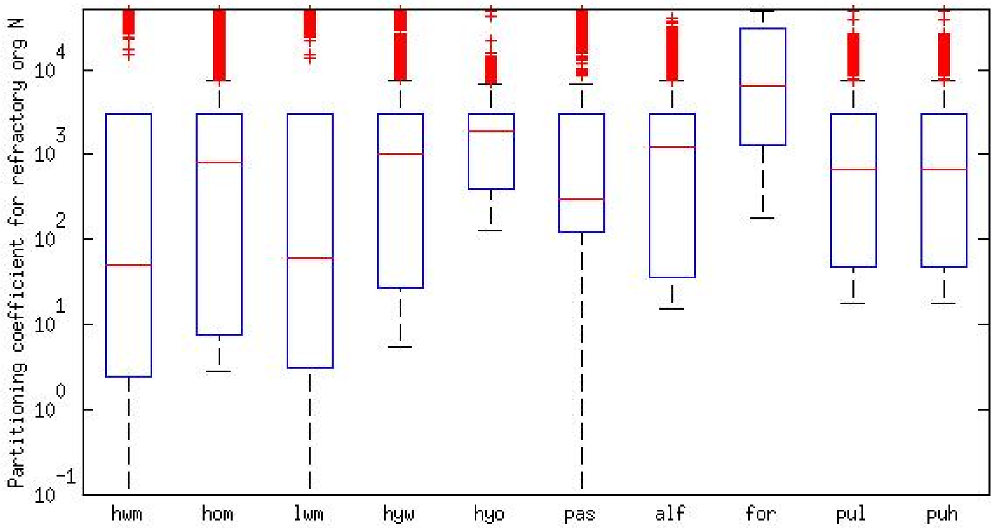


Figure 10-24. Percentile of calibrated partitioning coefficient for refractory organic N on a log scale.

10.5 Regional Nutrient Transport Factors

The Phase 5.3 simulation relies on county-level assessments of land use and animal numbers. This approach sets key model inputs, such as the land use and the manure and fertilizer inputs, at the county scale, which corresponds to about 300 different spatial units over the Phase 5.3 Model domain. Many of these key inputs for agricultural land uses are based on an analysis of the available manure, fertilizer, and atmospheric deposition inputs and the record of crop production, and hence nutrient uptake, in the county-level data. The riverine simulation is at a finer scale and has more than a thousand spatial riverine segments. To deal with the uncertainties in distributing county-level data down to the smaller river-segment units, as well as to adjust for any geographic differences in nutrient fate and transport, regional nutrient transport factors are used.

The application of regional nutrient transport factors is needed because of uncertainties in inputs. For example, the fertilizer application data for a particular land-segment are based on best professional judgment and are related to fertilizer sales only at the scale of the entire model domain. Finer-scale data are unavailable due to the unknown factor of the degree of fertilizer transportation between the sites of sale and application. There's also uncertainty in agricultural practices at spatial scales less than the county level. In addition, regional geomorphic characteristics such as soil qualities and slopes can have a large effect on exports. Other influences include the spatial range of average rainfall, between 37 inches and 56 inches per year. This difference can create a large difference in exports, particularly on the windward and leeward sides of mountains. Finally, the export to a 100-cfs or greater simulated river is dependent on the characteristics of the local stream network that might contribute or reduce nutrients.

The regional nutrient transport factors are used in calibrating the river simulation to local observed data, and it becomes necessary to adjust the local export values to meet water quality concentrations in the stream. The regional nutrient transport factors are determined by riverine calibration. The loads from the calibrated land uses are run in the riverine simulation using the Phase 5.3 decision rules and compared to the observed loads from the more than 100 calibration stations for nutrients. The automated iterative method of calibration for the river at the conclusion of the calibration run examines the difference between the simulated and the observed load and provides a separate TN and TP regional nutrient factor that is applied to only the nonpoint source loads specifically associated with that monitoring station. The regional factor may be positive or negative and is separate for both TN and TP because each has separate processes that affect their fate and transport. Typically, two to six iterations of calibration are used to set the final regional nutrient factors. The regional nutrient transport factors for total nitrogen and total phosphorus are illustrated in Figure 10-25.

The process of setting the regional transport factor is done from the top down in the river network. That is, the headwater segments are first examined for each group of segments associated with the first major downstream gage. For this group of segments, the regional nutrient factor is set as described above. Then the next major downstream gage and associated model segments are examined. With the segments above the first gage already having a set regional nutrient factor, the regional factor is applied to only to the nonpoint source loads in the segment controlled by the second gage. This process is then repeated until the regional nutrient factors have been set in the entire riverine portion of the watershed.

For the regions close to tidal waters, such as the Coastal Plain and the Western Shore, and unassociated with river-segments, another approach was used. Few of the segments in the Coastal Plain and Western Shore are monitored; for the few that are, however, the regional transport factors can be calculated. An assessment showed that the Coastal Plain segments had a different regional transport factor, which was generally less in both TN and TP than the Western Shore. The relatively low regional transport factors for the Coastal Plain are consistent with the low gradients, relatively high water tables, and high potential for denitrification described in the literature. The complete table of total nitrogen and total phosphorus regional transport factors by land-river segment can be found in the Chesapeake Community Model Program data library for Phase 5.3 in the Model Input section:
http://ches.communitymodeling.org/models/CBPhase5/datalibrary/model-input.php

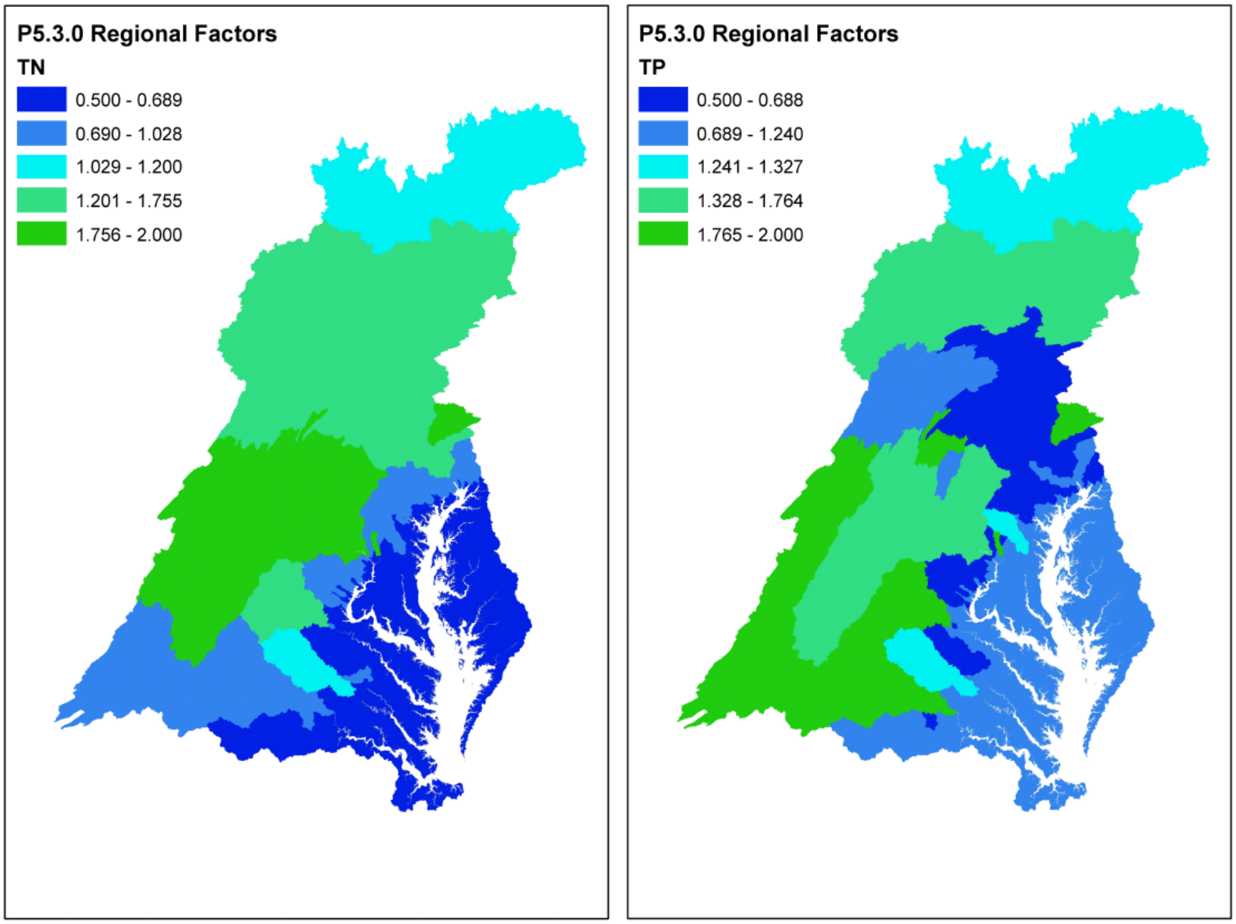


Figure 10-25. Total nitrogen and total phosphorus regional nutrient transport factors.

References

Aber, J.D., K.J. Nadelhoffer, P. Steudler, and J.M. Melillo. 1989. Nitrogen saturation in northern forest ecosystems *Bioscience* 39(6):378–386.

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(4):375–389.

Alexander, R.B., R.A. Smith, G.E. Schwarz, S.D. Preston, J.W. Brakebill, R. Srinivasan, and P.A. Pacheco. 2001. Atmospheric nitrogen flux from the watersheds of major estuaries of the Unites States: An application of the SPARROW watershed model. In *Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective*. American Geophysical Union. Pp.119–170. Washington, DC.

Arthur, M.A., G.B. Coltharp, and D.L. Brown. 1998. Effects of best management practices on forest streamwater quality in Eastern Kentucky *Journal of the American Water Resources Association* 34(3):481–495.

Brabets, T.P. 1984. *Runoff and Water Quality Characteristics of Surface Mined Lands in Illinois*. Water-Resources Investigations Report 83-4265. U.S. Geological Survey, Reston, VA.

Beaulac, M.N., and K.W. Reckhow. 1982. An examination of land use-nutrient export relationships. *Water Resources Bulletin* 16(6):1013–1022.

Bicknell, B.R., J.C. Imhoff, J.L. Kittle, Jr., A.S. Donigian, Jr., and R. C. Johnson. 1997. *Hydrological Simulation Program - Fortran (HSPF): User's Manual for Release 11*. U.S. Environmental Protection Agency. EPA/600/R-97/080, 755 p. Athens, GA.

Bicknell, B.R., J.C. Imhoff, J.L. Kittle, Jr., T.H. Jobes, and A.S. Donigian, Jr. 2001. *Hydrologic Simulation Program—Fortran. HSPF Version 12 Users Manual*. U.S. Environmental Protection Agency, Office of Research and Development, Athens, GA, and U.S. Geological Survey, Hydrologic Analysis Software Support Program, Reston, VA.

Campbell, K.L. 1982. Nutrient transport from North Florida agricultural fields and watersheds. Ed. L.B. Baldwin and A.B. Bottcher. IFAS Conference on Nonpoint Pollution Control Technology, Gainsville, Florida, March 9–10, 1982.

Castro, M. S., K.N. Eshleman, R.P. Morgan II, S.W. Seagle, R.H. Gardner, and L.F. Pitelka. 1997. Nitrogen dynamics in forested watersheds of the Chesapeake Bay. STAC Report Number 97-3. Scientific and Technical Advisory Committee, Frostburg, MD. June.

Clark, G.M., D.K. Mueller, and M.A. Mast. 2000. Nutrient concentrations and yields in undeveloped stream basins of the United States. *Journal of the American Water Resources Association* 36(4): 849–860.

Daniel, T.C., P.E. McGuire, D. Stoffel, and B. Miller. 1979. Sediment and nutrient yield from residential construction sites. *Journal of Environmental Quality* 8(3):304–308.

Donigian, A.S., Jr., J.C. Imhoff, B.R. Bicknell, and J.L. Kittle, Jr. 1984. *Application Guide for Hydrological Simulation Program – Fortran (HSPF).* U.S. Environmental Protection Agency, Environmental Research Laboratory, Office of Research and Development, Athens, GA.

Ensign, S.H., and M.A. Mallin. 2001. Stream water quality changes following timber harvest in a coastal plain swamp forest. *Water Research* 35:3381–3390.

Flynn, K.M., Hummel, P.R., Lumb, A.M., and Kittle, J.L., Jr., 1995. *User's Manual for ANNIE, Version 2, A Computer Program for Interactive Hydrologic Data Management*: U.S. Geological Survey Water-Resources Investigations Report 95-4085, 211 p. Reston, VA.

Frick, E.A., and G.R. Buell. 1999. Relation of land use to nutrient and suspended sediment concentrations, loads, and yields in the upper Chattahoochee River basin, Georgia, 1993–98. Ed. K.J. Hatcher. 1999 Georgia Water Resources Conference, University of Georgia, Athens, GA.

Goodale, C.L., K. Lajtha, K.J. Nadelhoffer, E.W. Boyer, and N.A. Jaworski. 2002. Forest nitrogen sinks in large eastern U.S. watersheds: Estimates from forest inventory and an ecosystem model. *Biogeochemistry* 57(58):239–266.

Hunsacker, C.T., C.T. Garten, and P.J. Mulholland. 1993. *Nitrogen Outputs from Forested Watershed in the Chesapeake Bay Drainage Basin.* Oak Ridge National Laboratory, Environmental Sciences Division, Oak Ridge, TN.

Johanson, R.C., Imhoff, J.D., and Davis, H.H., Jr., 1980. *Users Manual for Hydrological Simulation Program - Fortran (HSPF)*: Environmental Research Laboratory, EPA-600/9-80-015, April 1980. Athens, GA

Knap, A., T. Jickells, et al. 1986. Significance of atmospheric-derived fixed nitrogen on productivity of the Sargasso Sea. *Nature* 320(March 13):158–160.

Langland, M.J., P.I. Lietman, and S. Hoffman. 1995. *Synthesis of Nutrient and Sediment Data for Watersheds within the Chesapeake Bay Drainage Basin*. Water Resources Investigations Report 95-4233. U.S. Geological Survey in cooperation with U.S. Environmental Protection Agency, Lemoyne, PA.

Lea-Cox, J., and A. Ristvey. 2007. Personal communication between Robert Burgholzer and Drs. John Lea-Cox and Andrew Ristvey during a March 23, 2007, meeting on University of Maryland College Park research and outreach with the nursery industry in Maryland.

Lobo, M.E., and R.B. Herrmann. 1998. Harvest impacts on forest outflow in coastal North Carolina. *Journal of Environmental Quality* 27:1382–1395.

Line, D.E., N.M. White, D.L. Osmond, Gregory D. Jennings, and C.B. Mojonnier. 2002. Pollutant export from various and uses in the Upper Neuse River Basin. *Water Environment Research* 74(1):100-108.

Macko, S., L. Wang, and P. D'Odorico. 2005. Nitrogen uptake preferences by plants in arid and semiarid ecosystems. *Eos Transactions* AGU 86(52), Fall Meeting Supplement, Abstract B43D-0300.

Maestre, A., and R. Pitt. 2005. Identification of significant factors affecting stormwater quality using the NSQD. Unpublished monograph. <http://unix.eng.ua.edu/~rpitt/Research/ms4/Paper/CHI%20monograph%20(maestre%20pitt%202005)%20draft.pdf>. Accessed July 10, 2008.

Nixon, S.W. 1997. Prehistoric nutrient inputs and productivity in Narragansett Bay. *Estuaries* 20(2):253–261.

Pan, Y., R. Birdsey, J. Hom, and K. McCullough. 2005. *Forest Productivity and Effects of Nitrogen Deposition on Water Quality*. U.S. Department of Agriculture, Forest Service, Northwestern Area, Global Change Research. Newtown Square, PA.

Perry, D.A., 1998. The scientific basis of forestry. *Annual Review of Ecological Systems* 29(1998):435–466.

Pitt, R., A. Maestre, and R. Morquecho. 2004. *The National Stormwater Quality Database* (NSQD, version 1.1) Dept. of Civil and Environmental Engineering, University of Alabama, Tuscaloosa, AL. February 16. <http://unix.eng.ua.edu/~rpitt/Research/ms4/NSQD%20Summary%20Table%20Ver%201_1%20043005.doc>. Accessed July 10, 2008.

Riekerk, H., D.G. Neary, and W.T. Swank. 1988. The magnitude of upland silviculture nonpoint source pollution in the South. Ed. D.D. Hook and R. Lea. Conference on the Forested Wetlands of the Southern United States, Ashevile, NC. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. Pp. 8–18.

Ritter, W.F., and A.E.M Chirnside. 1984. Impact of land use on groundwater quality in Southern Delaware. *Groundwater* 22(1):38–47.

Scudlark, J. R., K. M. Russel, et al. 1996. Dissolved Organic Nitrogen in Precipitation: Collection, Analysis, and Atmospheric Flux. Report to Maryland Department of Natural Resources, Annapolis, MD.

Scudlark, J. R., and T. M. Church, 1993. Atmospheric input of inorganic nitrogen to Delaware Bay. *Estuaries* 16(4):747–754.

Seitzinger, S. P., R. W. Sanders, and R. Styles. 2002a. Bioavailability of DON from natural and anthropogenic sources to estuarine plankton. *Limnology and Oceanography* 47: 353–366.

Seitzinger, S.P., R.V. Styles, E.W. Boyer, R.B. Alexander, G. Billen, R.W. Howarth, B. Mayer, and V.B. Nico. 2002b. Nitrogen retention in rivers: Model development and application to watersheds in the northeastern U.S.A.. *Biogeochemistry* 57/58**:**199–237.

Schueler, T. 1987. Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban Best Management Practices. Metropolitan Washington Council of Governments, Washington, DC.

Simpson, T. 2007. Water Quality Steering Committee conference call materials. August 27, 2007, Agricultural, Forest, and Wetland BMP Final Recommendations.

<http://www.chesapeakebay.net/pubs/calendar/WQSC_08-27-07_Handout_3_8824.pdf >. Accessed September 13, 2007.

Smullen, J.T., J.L. Taft, and J. Macknis. 1982. Nutrient and sediment loads to the tidal Chesapeake Bay system. In *U.S. EPA Chesapeake Bay Program Technical Studies: A Synthesis.* Chesapeake Bay Program Office, Annapolis, MD.

Stevenson, J.C., R. Brinsfield, and K. Staver. 1987. Surface runoff and groundwater impacts from agricultural activities in the Chesapeake region. In *Proceedings of the U.S. Committee on Irrigation and Drainage, Regional Meeting, Washington, DC*, ed. L. Stevens.

Stoddard, J.L., ed. 1994. Long-term changes in watershed retention of nitrogen: Its causes and aquatic consequences. In *Environmental Chemistry of Lakes and Reservoirs,* ed. L.A. Baker. ACS Advances in Chemistry Series No. 237. American Chemical Society, Washington, DC. 83 pp.

Sweeney, J. 2001. Personal communication of a compilation of nutrient export studies done by the Chesapeake Bay Program Office.

Tinker, P.B., and P.H. Nye. 2000. Solute movement in the rhizosphere. In *Topics in Sustainable Agronomy* (series), ed. R. La, P. Sanchez, M. Summer, M.E. Swisher, P.B. Tinker, and R.E. White. Oxford University Press, New York. 444 pp. ISBN 0-19-512492-8.

Wang, P., L. Linker, and K. Eshleman. 2003. Dynamic parameterization to simulate DIN export due to gypsy moth defoliation. In P.M.A. Sloot et al., eds. Springer-Verlag. *Lecture Notes in Computer Science* 2659:30–38.

Wiegner, T.N., S.P. Seitzinger, P.M. Glibert, and D.A. Bronk. 2006. Bioavailability of dissolved organic nitrogen and carbon from nine rivers in the eastern United States. *Aquatic Microbial Ecology* 43:277–287.

Cite as:
USEPA (U.S. Environmental Protection Agency). 2010. *Chesapeake Bay Phase 5.3 Community Watershed Model*. EPA 903S10002 - CBP/TRS-303-10. U.S. Environmental Protection Agency, Chesapeake Bay Program Office, Annapolis MD. December 2010.

Errata Revision June 1, 2011