Revised Method and Outcomes for Estimating Soil Phosphorus Losses from Agricultural Land in the Chesapeake Bay Watershed Model

Alisha Spears Mulkey,* Frank J. Coale, Peter A. Vadas, Gary W. Shenk, and Gopal X. Bhatt

Abstract

Current restoration efforts for the Chesapeake Bay watershed mandate a timeline for reducing the load of nutrients and sediment into receiving waters. The Chesapeake Bay watershed model (WSM) has been used for two decades to simulate hydrology and nutrient and sediment transport; however, spatial limitations of the WSM preclude edge-of-field scale representation of phosphorus (P) losses. Rather, the WSM relies on literature-derived, county-scale rates of P loss (targets) for simulated land uses. An independent field-scale modeling tool, Annual Phosphorus Loss Estimator (APLE), was used as an alternative to the current WSM approach. Identical assumptions of county-level acreage, soil properties, nutrient management practices, and transport factors from the WSM were used as inputs to APLE. Incorporation of APLE P-loss estimates resulted in a greater estimated total P loss and a revised spatial pattern of P loss compared with the WSM’s original targets. Subsequently, APLE’s revised estimates for P loss were substituted into the WSM and resulted in improved WSM calibration performance at up to 79% of tributary monitoring stations. The incorporation of APLE into the WSM will improve its ability to assess P loss and the impact of field management on Chesapeake Bay water quality.

Core Ideas

• APLE estimated P losses were compared with the Chesapeake Bay Watershed Model’s (WSM) losses.
• Substituting the APLE estimated P loss into the WSM improved calibration performance.
• Findings suggest the importance of well-estimated transport factors in modeling P losses.

The Chesapeake Bay watershed is approximately 166,000 km² in the mid-Atlantic region of the United States, ranging from a northern boundary in central New York state (42° N, 75° W) and extending southward through the states of Pennsylvania, Maryland, Delaware, West Virginia, and the District of Columbia to a southern boundary in the state of Virginia (36° N, 76° W). More than 400 tributaries drain to the Bay, traversing several distinct physiographic regions characterized by highly diverse hydrologic transport pathways. The complex physical characteristics of the watershed are compounded by a large 14:1 land-to-water surface area ratio that exacerbates the nutrient and sediment loading pressures on the Bay’s ecosystems (Phillips, 2007).

Efforts to improve the water quality of the Bay are longstanding. A comprehensive total maximum daily load regulation was mandated in the watershed in 2010 to address excess nutrient and sediment loads (USEPA, 2010b). The Chesapeake Bay total maximum daily load created a 2025 implementation deadline for installing practices to reduce nutrient and sediment loads, including a 24% reduction in P loading (USEPA, 2010b). Reductions in agricultural nonpoint sources of P have been reported to be the most cost-effective option to achieve required reductions in P loading (USEPA, 2010a, Wäinberg et al., 2013).

Watershed managers have historically used the Chesapeake Bay watershed model (WSM, current version 5.3.2) to estimate P loading to the Bay from approximately 1000 simulated river segments across the watershed (USEPA, 2010c). The WSM is the primary landscape simulation model that estimates the nutrient and sediment loads to the Chesapeake Bay for determining attainment of the water quality standards required by the total maximum daily load. Though models like the WSM are commonly used to estimate nutrient and sediment loads across large landscapes, in many cases, the P losses were derived from model routines that have not been updated with the pace of current research (Radcliffe and Schoumans, 2009; Radcliffe et al., 2015). In addition, validating edge-of-field P losses at broader spatial scales challenges modelers due to limited field studies attributing P loss to specific land uses (USEPA, 2010c; Tetra Tech, Inc., 2015). Given these constraints,
Materials and Methods

The WSM primarily simulates agricultural practices at the county scale due to data availability from the USDA’s Census of Agriculture. Accordingly, a unique target P load is assumed in the WSM for each county segment and land use (n = 1175 target P loads for this study’s purposes). Likewise, APLE simulations were maintained at the county-based scale, and assumptions from the WSM were used as inputs to APLE (Table 1). Where the WSM input database did not include a required input element for APLE, input values derived from the academic literature were used. Assumptions of field and management conditions were considered representative of a given county land segment based on source data (Table 1).

Five agricultural land uses were simulated in APLE, which parallel the land-use categories and definitions used in the WSM: high-till with manure, nutrient management high-till with manure, low-till with manure, nutrient management low-till with manure, and pasture. These land uses, excluding pasture, represent 90% of the cropland in the Chesapeake watershed’s dominant crops of wheat (*Triticum aestivum* L.), corn (*Zea mays* L.), and soybeans (*Glycine max* (L.) Merr.) (USEPA, 2010c). The four land uses used to represent cropland represent four states of management with regard to implementation of conservation tillage and nutrient management, two of the most common management practices in the watershed. Low-till with manure and nutrient management low-till with manure simulated conservation tillage, while nutrient management high-till with manure and nutrient management low-till with manure simulated nutrient management by reducing organic inputs to nitrogen-recommended levels. In some counties with nutrient management high-till with manure and nutrient management low-till with manure land uses, excess P applications occurred.

The alternative modeling tool, APLE, is intended to be an agricultural field-scale simulation of P losses based on easily available input data describing field conditions, management practices, and transport factors, as defined by the user (Vadas, 2013). Annual Phosphorus Loss Estimator estimates edge-of-field P losses from sediment, soil, manure, and fertilizer P sources through surface transport pathways. The APLE model calculates P losses on an annual time step, and for this study, a 14-yr time period of 1992 to 2005 was simulated.

The WSM’s P-loss targets are separated into inorganic and organic P forms through surface- and groundwater flow. Conversely, APLE can partition estimated total P losses among four distinct component sources (sediment P, dissolved soil P, dissolved manure P, and dissolved fertilizer P) within surface runoff pathways, where the sum of the individually estimated P losses equals the edge-of-field total P load.

### Table 1. Input data and data sources required for Annual Phosphorus Loss Estimator (APLE) simulations.

<table>
<thead>
<tr>
<th>Data parameter</th>
<th>Units</th>
<th>Data source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth of soil layers</td>
<td>mm</td>
<td>Variable based on land use</td>
</tr>
<tr>
<td>Mehlich-3 soil P</td>
<td>mg kg⁻¹</td>
<td>University soil testing laboratory historical summary data</td>
</tr>
<tr>
<td>Clay content</td>
<td>%</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Organic matter content</td>
<td>%</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Annual rain</td>
<td>mm</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Annual runoff</td>
<td>mm</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Annual erosion rate</td>
<td>kg ha⁻¹</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Field size†</td>
<td>ha</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Annual crop P uptake</td>
<td>kg ha⁻¹</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Manure application</td>
<td>kg ha⁻¹</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Manure solids</td>
<td>%</td>
<td>Scholarly literature</td>
</tr>
<tr>
<td>Manure water-extractable P/total P</td>
<td>%</td>
<td>Scholarly literature</td>
</tr>
<tr>
<td>Manure incorporation</td>
<td>% and mm</td>
<td>Variable based on land use</td>
</tr>
<tr>
<td>Fertilizer application</td>
<td>kg ha⁻¹</td>
<td>Chesapeake Bay Program</td>
</tr>
<tr>
<td>Fertilizer incorporation</td>
<td>% and mm</td>
<td>Variable based on land use</td>
</tr>
<tr>
<td>Degree soil mixing</td>
<td>%</td>
<td>Variable based on land use</td>
</tr>
</tbody>
</table>

† Chesapeake Bay Program Partnership, USEPA, state and agency partners (www.chesapeakebay.net).

‡ Pasture land use only.

The APLE model’s Relationships and Input Parameters

The APLE model estimated annual manure P runoff (kg P ha⁻¹) from manure water extractable P remaining on the field surface and subject to runoff losses after reductions for infiltration and tillage incorporation. In pasture settings, where manure was also directly excreted, field coverage was assumed to be non-homogeneous and unlikely to interact uniformly or consistently with surface runoff (Vadas, 2013).

The APLE model’s estimates of fertilizer P runoff were similar to manure P, with the exception that all field applications were considered to be highly soluble and available for runoff, following reductions attributed to tillage incorporation (Vadas, 2013).
Sediment-bound P losses were estimated by APLE from soil erosion rates, soil P concentrations, and the ratio of P in the eroded soil to that in the source soil (Vadas, 2013). Assumed soil erosion rates (Mg ha\(^{-1}\)) were provided by the USEPA Chesapeake Bay Program at a county scale based on assessments from the National Resources Inventory (NRI), the Revised Universal Soil Loss Equation (RUSLE), and literature review (USEPA, 2010c). Soil erosion estimates were based on annual average erosion rates in the years 1982 and 1987. The available data did not represent differences in tillage practices, so the WSM assumed that soil erosion rates for high-tillage land uses were 125% of the NRI county average and that the erosion rates for conservation-tillage land uses were 75% of the NRI county average erosion rate (USEPA, 2010c). Soil-dissolved P losses were estimated from labile soil P concentrations and estimates of surface water runoff volumes (Vadas, 2013). Surface runoff volume was provided by the USEPA Chesapeake Bay Program as county-scale average annual runoff based on calibration to riverine water quality monitoring stations.

Changes in APLE-simulated soil P pool concentrations, resulting from nutrient additions and the effecting estimated P losses, were influenced by soil layer depth(s), soil organic matter and clay content, and the depth of tillage incorporation. Soil layer depth and tillage incorporation were assumed to be a single homogenous soil layer with a depth of 0 to 178 mm for the high-till with manure and nutrient management high-till with manure land uses to represent complete soil mixing resulting from conventional tillage practices. Two soil layers with depths of 0 to 25 mm and 25 to 178 mm were established for the low-till with manure, nutrient management low-till with manure, and pasture land uses. The shallow soil layer defined for low-till with manure, nutrient management low-till with manure, and pasture land uses allowed for simulation of P stratification in surface soil layers as a result of reduced or nonexistent tillage. Tillage incorporation was assumed as 102 mm for low-till with manure and nutrient management low-till with manure land uses and as 13 mm for pasture land uses to represent incorporation from natural processes. Assumed soil organic matter and soil clay contents were provided by the USEPA Chesapeake Bay Program at a county scale from the NRCS SSURGO database (USEPA, 2010c).

**Estimating Agronomic Soil-Test Phosphorus**

Agronomic soil-test P concentration was a required input for the APLE simulation; however, this parameter was not previously used within the WSM. The APLE model required an initial soil-test P concentration (Mehlich-3 P) for each county. Given the expanse and diversity of the Chesapeake Bay watershed’s agricultural landscape, acquiring representative soil-test P data was a challenge. Ultimately, historical university soil testing laboratory data was acquired. To account for the variety of soil-test P extraction methods utilized by different university soil testing laboratories, the soil-test P datasets were converted to a single uniform fertility index value scale and subsequently converted to equivalent Mehlich-3 P concentration, as described by Coale (2001). For some states, quantitative records of county level soil-test P data were unavailable. Instead, the data consisted of the number of soil samples analyzed per county and the number of those analyses that were ranked in each of the qualitative soil fertility interpretation categories (i.e., “low,” “medium,” “high,” etc.). Under these circumstances, when a fertility index value could not be directly calculated, it was assumed that all soil analyses that were grouped within the “low” interpretive category had a fertility index value of 25; soils with “medium” P fertility status were assigned a fertility index value of 38; soils categorized as “high” were assigned a fertility index value of 76; and soils with P fertility status of “very high” or “excessive” were assumed to have a fertility index value of 200. The result was a weighted average soil-test P concentration, Mehlich-3 P equivalent, based on the distribution of samples in each of the qualitative categories. A single soil-test P concentration was estimated for each county in the Chesapeake Bay watershed. Exceptions to the method outlined above include the use of different assumed fertility index values for soil analyses from the state of Virginia to ensure consistent soil fertility interpretive categories across states.

**Calibration Testing of APLE Output Results in the WSM**

Following the APLE simulation of the agricultural land uses, the revised annual estimates of P loss for each county were substituted into the WSM in place of the WSM’s original surface P-loss targets to evaluate if the spatial distribution of APLE-estimated P losses resulted in an improved calibration performance. However, calibration testing required additional data management steps to align the APLE edge-of-field estimates with the WSM targets, since the WSM did not explicitly simulate edge-of-field. Rather, the WSM combines all field-level and hillslope processes within the P-loss targets into an edge-of-stream load, excluding the effects of best management practices and physiographic differences (USEPA, 2010c). Scaling of APLE-estimated P loads for calibration testing was performed such that the net P load for each of the agricultural land uses was unchanged at the watershed scale, and the spatial distribution of P losses, as estimated by APLE, was maintained. Therefore, the scaling allowed for the appropriate conversion of APLE P loads from edge-of-field to edge-of-stream scale without affecting the spatial intercounty variability provided by the APLE estimated loads. Furthermore, it also preserved the difference in loads between land uses within a county.

First, the WSM surface total P loads were calculated as the sum of the WSM’s defined targets for surface inorganic P and surface organic P loads for each county segment and land use. The WSM edge-of-stream surface total P mass loss was multiplied by the area extent of each land use within the county for selected years 1992, 1997, and 2002—based on the availability of agricultural census data—to derive the WSM’s estimate of the surface total P mass loss. Concurrently, the APLE total P-loss outputs, averaged for 1992 to 2005 per county and land use, were multiplied by the same area extent of each land use to calculate APLE’s estimate of surface total P mass loss. Second, the WSM’s edge-of-stream surface P loads and the APLE edge-of-field surface P loads were each summed to calculate two estimates of total P loss for the watershed. Next, the sum of the WSM’s surface total P mass loss was divided by the sum of the APLE surface total P mass loss to calculate a single relative scaling factor. Lastly, this scaling factor was applied to the initial APLE edge-of-field estimates, and the WSM was recalibrated against 209 riverine monitoring stations.

Calibration was performed using the same methods as in the original WSM, as described in Section 11 of USEPA (2010c). The automated calibration routine uses the differences in the cumulative frequency distribution of simulated and observed concentration data for iteratively adjusting the model. A system of rules for updating riverine simulation parameters were
developed by estimating sensitivity to bias in five quintiles of the cumulative frequency distributions (USEPA 2010c).

Calibration analysis was conducted using three separate land-use loading scenarios: (i) the WSM base case calibration, which utilized the original defined WSM P-loss targets without any APLE estimates of edge-of-field P losses (baseline); (ii) the WSM calibration, with incorporation of scaled APLE edge-of-field P-loss estimates substituted for the original defined P-loss targets, along with WSM regional factors (APLE with regional factors); and (iii) the WSM calibration, with incorporation of scaled APLE edge-of-field P-loss estimates without WSM regional factors (APLE without regional factors). Regional factors are subbasin-scale multiplicative factors that assist in resolving unexplained regional differences in delivered loads to the Chesapeake Bay (USEPA, 2010c). Consequentially, the calibration scenario APLE with regional factors would help evaluate how APLE-estimated loads impacted the WSM performance, whereas APLE without regional factors assessed the relative uncertainty imbedded within the regional factors.

Calibration output was evaluated using cumulative frequency distribution curves of observed riverine monitoring data across low, normal, and high flow regimes. Calibration performance was evaluated by a Kolmogorov–Smirnov (K–S) statistic that quantified the goodness of fit between two curves, where a K–S value of zero indicated a perfect fit between the estimated and observed curves (Massey, 1951). The three WSM scenarios were calibrated against 209 individual stream monitoring stations for total P, representing 19 subbasins within the Chesapeake Bay watershed.

### Results

#### High-Till Land Uses

The high-till with manure land-use simulation by APLE resulted in an estimated mean annual total P loss of 7.96 kg ha⁻¹ for all simulated counties of the Chesapeake Bay watershed, compared with the mean target value of 2.15 kg ha⁻¹ defined in the WSM (Table 2). The APLE model's estimates for annual total P loss for the high-till with manure land use ranged from 0.01 to 96.10 kg ha⁻¹, compared with a much narrower range for WSM total P-loss target values (1.29 to 9.61 kg ha⁻¹) (Table 2). Additionally, APLE estimates of median edge-of-field total P loss at the state scale revealed a much wider range of annual median total P loss as compared with the narrow range of target P losses assumed in the WSM at similar spatial scales (Fig. 1).

The APLE model estimated that bay-wide mean total P losses were similar for high-till with manure and nutrient management high-till with manure land uses (Table 2). Land uses defined as “nutrient management” in the WSM were intended to simulate regulated acres within the watershed subject to agronomic recommendations for crop nutrient needs. The APLE model estimated a mean annual total P loss of 9.24 kg ha⁻¹ on the nutrient management high-till with manure land use as compared with the WSM target total P-loss load value of 2.96 kg ha⁻¹ (Table 2). The APLE estimates of total P loss for nutrient management high-till with manure encompassed a similarly wide range as the high-till with manure land use (Table 2). Unexpectedly, the bay-wide APLE total P-loss loading estimates for the nutrient management high-till with manure land use were greater than the APLE estimates for the high-till with manure land use, despite similar assumptions of soil erosion and surface runoff for each high-till with manure and nutrient management high-till with manure county segment. Results demonstrate that, on the nutrient management land uses, organic sources of biosolids and manure were typically applied at an appropriate rate for nitrogen during the simulation period; however, the WSM preferentially applied biosolids in addition to the other organic sources. As a result of such assumptions, the rate of biosolids application on nutrient management land uses is approximately 30 times greater than the non-nutrient-managed lands and creates an additional nutrient source contributing to estimated total P losses.

Over the entire Chesapeake Bay watershed, sediment P from the high-till with manure and nutrient management high-till with manure land uses was overwhelmingly the largest contributor to APLE-estimated total P loads. The high-till with manure

<table>
<thead>
<tr>
<th></th>
<th>HWM</th>
<th>NHI</th>
<th>LWM</th>
<th>NLO</th>
<th>PAS</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>APLE</strong> mean†</td>
<td>7.96 (0.2)</td>
<td>9.24 (0.2)</td>
<td>5.75 (0.1)</td>
<td>6.29 (0.2)</td>
<td>3.48 (0.1)</td>
</tr>
<tr>
<td><strong>APLE</strong> maximum</td>
<td>96.10</td>
<td>102.97</td>
<td>83.93</td>
<td>78.47</td>
<td>75.75</td>
</tr>
<tr>
<td><strong>WSM</strong> mean†</td>
<td>7.96 (0.2)</td>
<td>9.24 (0.2)</td>
<td>5.75 (0.1)</td>
<td>6.29 (0.2)</td>
<td>3.48 (0.1)</td>
</tr>
<tr>
<td><strong>WSM</strong> maximum</td>
<td>96.10</td>
<td>102.97</td>
<td>83.93</td>
<td>78.47</td>
<td>75.75</td>
</tr>
</tbody>
</table>

† Values in parentheses represent standard error mean.

![Fig. 1. Median annual total P loss (kg ha⁻¹) estimates from the Annual Phosphorus Loss Estimator (APLE, black bar) compared with Chesapeake Bay Program watershed model (WSM) targets (striped bar) for the high-till with manure land use for each state in the Chesapeake Bay watershed.](image)
mean annual sediment P loss was 7.78 kg ha\(^{-1}\) The high-till with manure median annual sediment P loss was 4.59 kg ha\(^{-1}\) or 96% of the APLE-estimated median total P loss for the bay watershed (Tables 2 and 3). The nutrient management high-till with manure mean annual sediment P loss was 8.98 kg ha\(^{-1}\) (Table 3).

The APLE estimates of annual total dissolved P loss (manure-disolved P loss + fertilizer-dissolved P loss + soil-dissolved P loss) from the high-till with manure and nutrient management high-till with manure land uses were low, relative to estimated sediment P losses (Table 3). The bay-wide mean annual dissolved P loss from the high-till with manure land use was 0.18 kg ha\(^{-1}\). The bay-wide high-till with manure median annual dissolved P loss load was 0.11 kg ha\(^{-1}\), and the maximum estimated annual dissolved P loss was 2.64 kg ha\(^{-1}\) (Table 3). The bay-wide mean annual dissolved P loss from the nutrient management high-till with manure land use was 0.26 kg ha\(^{-1}\), with a median annual dissolved P loss of 0.15 kg ha\(^{-1}\). The maximum APLE-predicted annual dissolved P loss for nutrient management high-till with manure was 5.22 kg ha\(^{-1}\) (Table 3).

### Low-Till Land Uses

Estimations of annual total P loss by APLE were similar between the low-till with manure and nutrient management low-till with manure land uses (Table 2). On the low-till with manure land use, APLE estimated a bay-wide mean annual total P loss of 5.75 kg ha\(^{-1}\), compared with the WSM defined mean target value of 1.67 kg ha\(^{-1}\) (Table 2). Established WSM target ranges for total P loss from low-till with manure land uses were 0.78 to 5.64 kg ha\(^{-1}\), while APLE estimated a bay-wide mean annual total P loss of 0.22 kg ha\(^{-1}\) (Table 2). As anticipated, however, the APLE-estimated mean annual sediment P loss on low-tillage land uses was decreased noticeably relative to the tillage-intensive scenarios for high-till with manure and nutrient management high-till with manure land uses (Table 3).

For the low-till with manure land use, APLE estimations of total dissolved P loss had a bay-wide mean annual loss of 0.22 kg ha\(^{-1}\) and a median annual dissolved P loss of 0.13 kg ha\(^{-1}\) (Table 3). The APLE estimates of nutrient management low-till with manure dissolved P loss revealed a mean annual dissolved P loss of 0.30 kg ha\(^{-1}\) and a median annual dissolved P loss of 0.16 kg ha\(^{-1}\) (Table 3). The maximum APLE-estimated annual dissolved P loss from the low-till with manure land use was 4.55 kg ha\(^{-1}\), which was nearly twice the annual loss estimated for the high-till with manure land use (Table 3). The observed increase in dissolved P loss coincides with reduced tillage practices for the low-till with manure land use. A link has been well established between the long-term application of P nutrient sources that increase surface-soil P concentration and consequently increase risk of dissolved P losses when coupled with reduced tillage (Kleinman et al., 2002, 2009; Buda et al., 2009; Abdi et al., 2014).

### Pasture Land Use

In addition to the machine application of fertilizer and manure assumed for the four cropland land uses, the pasture land-use simulation in APLE included a category of manure to account for the direct deposition of manure from grazing animals.

The APLE estimates for the pasture land use resulted in an estimated Chesapeake Bay-wide mean annual total P loss of 3.48 kg ha\(^{-1}\), as compared with the relatively modest WSM target value of 0.90 kg ha\(^{-1}\) (Table 2). For the pasture land use at the state level, the highest estimated median total P losses were West Virginia, followed by Virginia, Pennsylvania, Maryland, New York, and Delaware.

As expected from minimized erosion on pasture land uses, the APLE estimates of mean annual sediment P loss were the lowest for pasture among the five simulated agricultural land uses. The mean annual sediment P loss for pasture was 2.99 kg ha\(^{-1}\), with a median annual sediment P loss of 1.42 kg ha\(^{-1}\) (Table 3). Conversely, APLE estimates of total dissolved P loss for the pasture land use were the greatest among the land uses, with a mean dissolved P loss of 0.49 kg ha\(^{-1}\) and a median annual dissolved P loss of 0.25 kg ha\(^{-1}\) (Table 3). The maximum estimated annual

### Table 3. Descriptive statistics for Chesapeake Bay-wide annual sediment P loss and total dissolved P loss from Annual P Loss Estimator (APLE) estimates for five agricultural land-use categories used in the Chesapeake Bay watershed model (WSM). Land-use categories are high-till with manure (HWM), nutrient management high-till with manure (NHI), low-till with manure (LWM), nutrient management low-till with manure (NLO), and pasture (PAS). N = 2644 to 2882 land segments, depending on land use.

<table>
<thead>
<tr>
<th>Land Use</th>
<th>Annual Sediment P Loss</th>
<th>Annual Total Dissolved P Loss</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>HWM</td>
<td>NHI</td>
</tr>
<tr>
<td>APLE Quantiles</td>
<td></td>
<td></td>
</tr>
<tr>
<td>95%</td>
<td>25.62</td>
<td>28.67</td>
</tr>
<tr>
<td>75%</td>
<td>9.98</td>
<td>11.57</td>
</tr>
<tr>
<td>50%</td>
<td>4.59</td>
<td>5.34</td>
</tr>
<tr>
<td>25%</td>
<td>2.01</td>
<td>2.42</td>
</tr>
<tr>
<td>5%</td>
<td>0.46</td>
<td>0.57</td>
</tr>
<tr>
<td>APLE Minimum</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>APLE Mean†</td>
<td>7.78 (0.2)</td>
<td>8.98 (0.2)</td>
</tr>
<tr>
<td>APLE Maximum</td>
<td>95.86</td>
<td>102.70</td>
</tr>
</tbody>
</table>

† Values in parentheses represent standard error mean.
dissolved P loss for the pasture land use was 13.50 kg ha\(^{-1}\). The increasing contribution of dissolved P loss relative to the total P loss affirms the link between the surface applications of manure without incorporation and an increased risk for dissolved P loss.

**Calibration Testing of APLE Output Results in the WSM**

Subsequent to generating APLE estimations of edge-of-field P losses, APLE P loads were incorporated for calibration testing into the WSM as substitute values for the original WSM P-loss targets. Three scenarios were run in the WSM and compared with observed water quality data at 209 riverine monitoring stations with flow rates that average at least 2.83 m\(^3\) s\(^{-1}\) (100 ft\(^3\) s\(^{-1}\)).

When evaluated at the 209 individual water quality monitoring stations, 140 of 209 stations were improved (\(p < 0.001\), one-tailed test) for the APLE with regional factors scenario versus the baseline scenario, and the median K–S statistic for the APLE with regional factors calibration scenario showed a 14% improvement versus the baseline scenario. Likewise, 143 of 209 stations were improved (\(p < 0.001\)) for the APLE without regional factors scenario versus the baseline scenario, and the median K–S statistic for the APLE without regional factors calibration scenario showed an 11% improvement versus the baseline scenario. When the K–S statistics were further evaluated at the scale of 19 individual subbasins within the Chesapeake Bay watershed, the median K–S statistic for the APLE with regional factors scenario was improved in 14 out of 19 (74%, \(p < 0.05\)) of subbasins. Likewise, when the APLE without regional factors scenario was compared with the WSM baseline scenario, 15 out of 19 (79%, \(p < 0.001\)) of Chesapeake Bay watershed subbasins showed improved calibration performance.

**Discussion**

**APLE Estimates of Edge-of-Field P Losses**

When evaluated cumulatively, APLE’s estimation of P loss demonstrated the expected outcome that higher rates of sediment P loss and, subsequently, total P loss were associated with increased soil disturbance resulting from tillage practices (Tables 2 and 3). Sediment P loss was the largest contributor to total P loss for all of the simulated land uses. Sediment P accounted for 96% of median total P loss from high-till with manure, 95% of median total P loss from low-till with manure, and 76% of median total P loss from pasture land uses. Significant sediment P losses were influenced by some atypically high soil erosion rate input values per county segment, provided by the USEPA Chesapeake Bay Program. The mean annual erosion rate assumed within the WSM for the high-till with manure land use was 13.9 Mg ha\(^{-1}\) yr\(^{-1}\) (6.19 tons ha\(^{-1}\) yr\(^{-1}\)), but included a range of 0.003 to 163 Mg ha\(^{-1}\) yr\(^{-1}\). Conversely, in an analysis used to evaluate the APLE model, the maximum value of annual sediment erosion was 40 Mg ha\(^{-1}\) yr\(^{-1}\) (18 tons ac\(^{-1}\) yr\(^{-1}\)) with rates generally less than 11 Mg ha\(^{-1}\) yr\(^{-1}\) (5 tons ac\(^{-1}\) yr\(^{-1}\)) (Bolster and Vadas, 2013). The dispersion of soil erosion rates assumed by the WSM suggests sediment P contributions to total P losses were not fully considered in the WSM’s P-loss target calculations. Rather, the WSM assumed the spatial variability of total P loss across the Chesapeake Bay watershed was related to the balance of inputs and uptake. In the future, county-scale estimates of soil erosion rates utilized in the WSM should be reevaluated due to the impact that soil erosion input data can necessarily have on estimating total P loss.

As expected, when input rates for soil erosion decreased, APLE estimates of sediment P losses decreased and the dissolved P losses became a greater contributor to the estimated total P edge-of-field load. However, APLE estimations of dissolved P loss across the watershed were consistently smaller compared with sediment P losses. All county segments had an average annual dissolved P loss of less than 0.5 kg ha\(^{-1}\) for all land uses (Table 3), while published field studies concluded that average annual edge-of-field dissolved P loads were often 1 to 2 kg ha\(^{-1}\) yr\(^{-1}\) (Staver, 2004; Vadas et al., 2009).

A comparison of P-loss field studies and the WSM’s assumptions for erosion and surface runoff highlights the important connections between P source and P transport. An adequate consideration of P transport mechanisms is considered integral to estimating P losses (McDowell et al., 2001; Gentry et al., 2007; Staver et al., 2014). However, APLE simulations indicated an apparent decoupling of transport factors and P export rates within the WSM. The WSM’s P-loss targets were developed from literature values, and higher P-loss targets were assigned to counties that were expected to contribute greater nutrient inputs (e.g., counties with higher counts of animal operations). Despite the variability in landform, hydrology, and management inputs across the Chesapeake Bay watershed, the WSM assumed uniform and modest edge-of-stream P losses across the watershed (Fig. 1). Although APLE estimations were limited to edge-of-field P losses, APLE assessments disclosed a substantially greater average annual P loss and a much greater range in edge-of-field P losses. When WSM processes were applied to convert edge-of-field P loss estimates to edge-of-stream loads, the magnitudes of the watershed average edge-of-field P loss and the geographic variability in APLE-estimated losses were not likely to be reduced and compressed, respectively, to conform to the edge-of-stream targets assumed by the WSM (Fig. 2). Further, a comparison of the spatial distribution of APLE’s estimated median P loss and the WSM’s P-loss targets demonstrated that APLE’s largest estimations of P loss logically coincided with county segments with the highest rates of soil erosion. The APLE model better captured the influence of transport mechanisms on estimated P loss, as opposed to the WSM’s assumption that P losses were directly related to animal manure availability in a county. An enhanced capacity to represent the spatial variability in subsin P loss is an important step toward watershed modeling’s ability to attribute P losses to finer scales or management conditions.

For now, APLE does not estimate field-scale P loss through subsurface pathways, which limits its ability to improve estimates of P loss for the WSM. While surface pathways often dominate field-scale P loss, subsurface P loss can be substantial in poorly drained, low-relief regions dominated by artificial drainage systems, which are common in the Coastal Plain region of the Chesapeake Bay watershed (Kleinman et al., 2007).

**Calibration Testing of APLE Output Results in the WSM**

For WSM calibration testing evaluation at the 209 individual water quality monitoring stations, 67% of stations were improved for the APLE with regional factors scenario versus the baseline scenario, and 68% of stations were improved for the APLE without regional factors scenario versus the baseline scenario. When evaluated at the subbasin scale using APLE-estimated P losses, model performance improved for 74 and 79%, respectively, of the riverine monitoring stations. The improved WSM calibration performance, when scaled APLE P losses were incorporated, suggests that APLE may have
better simulated the inherent spatial variability of the watershed, as opposed to the WSM's uniform assumptions of P loss (Fig. 2). The subbasins that did not exhibit improved calibration results were the York River (Virginia), Lower Rappahannock River (Virginia), Choptank River (Maryland and Delaware), and the Lower Eastern Shore (Maryland and Delaware). One commonality between the latter three subbasins that may have adversely affected the calibration results is a small number of riverine monitoring stations (1–3 stations in each subbasin) with little available monitoring data. Additionally, differences in purpose and scale between APLE and the WSM were expected to affect calibration performance. The WSM is a watershed-scale model developed to simulate hydrology, sediment, and nutrient loads at broader geographic scales, while APLE is a field-based model with empirically-based calculations. Nonetheless, improvements seen in model calibration performance using APLE indicate that a future WSM could benefit from reevaluating transport factors and from coupling process-based estimations of edge-of-field targets beyond nutrient inputs alone.

Conclusion

The APLE-determined sources of P loss (sediment P, soil P, manure P, and fertilizer P) for agricultural land uses exhibited a wider range and larger estimated mean total P loss compared with the original WSM P-loss targets. This observation is consistent with the broad spatial variability expected for soil P source dynamics and transport processes and affirms APLE's ability to simulate edge-of-field losses based on unique land-segment conditions, as opposed to the relatively uniform P losses calculated by the current WSM. The upper range of soil erosion rates used in the WSM was too broad for appropriately estimating sediment P losses. Future WSM improvements should better characterize the links between soil P sources and transport mechanisms. Overall, WSM estimates of Chesapeake Bay watershed-wide total P loss were improved by the substitution of APLE edge-of-field P-loss estimates in place of the literature-defined total P-loss targets currently in use for agricultural land. Future refinement of the assumptions utilized by the WSM will improve estimates of edge-of-field P losses. Also, future research and appropriate simulation of P loss through sub-surface transport processes are needed.

References


Terra Tech, Inc. 2015. Agricultural and forest land use loading rate literature review—Summary and results. Special report prepared for Chesapeake Bay Program Office. Terra Tech Inc., Fairfax, VA.


USEPA. 2010c. Phase 5.3 watershed model. EPA 903S10004 Bin 305. Chesapeake Bay Program Office, USEPA, Annapolis, MD.


