ABSTRACT

Title of Document: ALTERNATIVE SIMULATION OF SOIL PHOSPHORUS FOR AGRICULTURAL LAND USES IN THE CHESAPEAKE BAY WATERSHED MODEL

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Current restoration efforts for the Chesapeake Bay watershed mandate reducing nutrient and sediment loads to receiving waters. The Chesapeake Bay Watershed Model (WSM) estimates loading; however, some WSM routines have not been updated to reflect recent research. This study's objective was to improve the simulation of soil phosphorus dynamics by using an independent modeling tool (APLE) as an alternative to the current WSM approach. Identical assumptions of acreage, soil properties, nutrient management, and transport factors from the WSM were used as inputs to APLE. Outcomes represent revised estimates of phosphorus edge-of-field losses and estimates of change in soil labile phosphorus concentration. The modification resulted in a greater mean phosphorus loss estimate compared to the WSM, and a relationship between nutrient application, tillage, and soil phosphorus concentrations. Outcomes support APLE as an appropriate alternative model for simulating soil phosphorus dynamics, and for informing mitigation of soil phosphorus losses through best management practices.

ALTERNATIVE SIMULATION OF SOIL PHOSPHORUS FOR AGRICULTURAL LAND USES IN THE CHESAPEAKE BAY WATERSHED MODEL

By

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Abbreviations

APLE – Annual Phosphorus Loss Estimator

ASAE – American Society of Agronomic Engineering

BMP – Best Management Practices

CWA – Clean Water Act

DP – Dissolved Phosphorus

EOF – Edge of Field

EOS – Edge of Stream

HWM – High-till with Manure

K-S – Kolmogrov Smirnov Statistic

LWM – Low-till with Manure

N – Nitrogen

NHI – Nutrient Management High-till with Manure

NLO – Nutrient Management Low-till with Manure

NPS – Nonpoint Sources of Pollution

NRI – National Resources Inventory

P - Phosphorus

PAS - Pasture

PP – Particulate Phosphorus

TMDL – Total Maximum Daily Load

USEPA – United States Environmental Protection Agency

USDA – United States Department of Agriculture

USGS – United States Geological Survey

USLE – Universal Soil Loss Equation

WEP – Water Extractable Phosphorus

WSM – Chesapeake Bay Watershed Model

Chapter 1: Introduction

1.1 Chesapeake Bay Watershed

The Chesapeake Bay (Bay) has historically been recognized as an important natural, economic, and recreational resource (Boesch et al., 2001). The Bay watershed is comprised of approximately 166,000 km^2 of drainage in the Mid-Atlantic region, USA, 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) (Figure 1.1).More than 400 tributary streams and rivers drain to the Bay carrying nutrients and sediment loads. Several distinct physiographic regions – Appalachian Plateau, Appalachian Mountains, Ridge and Valley, Piedmont, and Coastal Plain - also uniquely contribute to highly diverse hydrologic transport pathways within the watershed (Figure 1.1). The complex physical characteristics of the watershed, exacerbated by a significantly large14:1 land to water ratio and combined with a wide variety of overlain land-uses and an expanding human population continue to place nutrient and sediment loading pressures on the Bay's ecosystems (Phillips, 2007).

1.2 Chesapeake Bay Water Quality

Efforts to improve the Bay's water quality and aquatic resources are longstanding. Most recently in May 2009, in recognition of the Bay as a "national treasure", President Obama signed an Executive Order to renew efforts for restoration intended to enhance partnerships, research, and fiscal commitments, and to create a means to quantify and track the Bay's restoration progress (Executive Order 13508). Accounting for the restoration progress is being implemented through the U.S. Environmental Protection Agency's (USEPA) authority under the Clean Water Act's (CWA) Total Maximum Daily Load (TMDL) process for identifying and addressing water quality impairments (Public Law 92-500, 1972). A primary objective of the CWA is to guide states to determine designated uses (e.g. public water supply) of its waters and to set and enforce standards to maintain those uses. Waters determined not to be meeting the standards for their designated use are categorized as "impaired" and a TMDL regulation must be developed by the state and approved by USEPA. The TMDL must specifically identify the pollutant(s) of concern and sources of pollutant(s), quantify targets for pollutant load reductions that would achieve designated use standards, allocate a numeric maximum load restriction among contributing pollutant sources, and outline an implementation plan to track progress (USEPA, 1999).

A comprehensive TMDL was developed in 2010 to address the Chesapeake Bay's eutrophication (USEPA, 2010a). Eutrophication results from excessive water nutrient concentrations that stimulate suspended algae and aquatic plants growth. This growth results in decreased water clarity and their decomposition depletes the available oxygen supply for other aquatic life (Cloern, 2001). Eutrophication has been a persistent concern to Bay health and the Bay TMDL imposes maximum loading limits for nitrogen (N), phosphorus (P) and sediment (USEPA, 2010a). Forty-four percent of the excessive P load has been attributed to agricultural nonpoint sources (NPS) (Figure 1.2).

Through the TMDL process, each Bay watershed state is allocated a portion of the maximum pollutant load relative to each state's contribution to the Bay watershed flow. Individual states retain the flexibility to determine appropriate methods for achieving the mandated load reduction requirements and, at present, many of the Bay states consider agricultural sources to be the most cost-effective option for pursuing reductions in nutrient loading (USEPA, 2010b).

Agricultural operations account for approximately 25% of the land acreage in the Bay watershed, producing greater than 50 commodities in the region (USEPA, 2010b). Central to most agricultural operations is the use of nutrient sources to provide N and Pfor crop growth. Producers are widely encouraged to use appropriate conservation measures to minimize losses of excess nutrients from the field to a receiving stream. For many years this mitigation of nonpoint sources from agricultural lands focused on N because of its application in larger quantities and greater mobility in soil compared to P. Moreover, because P is more readily adsorbed to soil particles, the increased use of conservation tillage techniques to reduce sediment erosion provided the co-benefit of reduced particulate-bound P losses. However, a growing body of research now indicates measurable concentrations of non-particulate, dissolved P are being delivered from fields to receiving streams by runoff and subsurface pathways (Soileau et al., 1994; Sharpley et al., 2001; He et al., 2006; Owens and Shipitalo, 2006), and P is considered the critical limiting nutrient to decrease eutrophication in fresh waters (Hecky and Kilham, 1988; Sharpley, 2000). Continuing research to address the sources and movement of P from agricultural landscapes remains crucial to understanding the connections between non-point source P pollution and water flow within the watershed (McDowell et al., 2001).

1.3 Research Objectives and Anticipated Outcomes

The Chesapeake Bay Watershed Model (WSM) is being used as a decision support tool in the Bay restoration efforts (USEPA, 2010c). The WSM is a primary component within the suite of integrated simulation models that estimate nutrient and sediment loads to the Bay over a defined duration, thus determining the appropriate loads required to attain water quality standards, as required by the TMDL regulation. While models like the WSM are commonly used to estimate nutrient and sediment loads across large landscapes, in many cases the estimation of P loads are derived from model routines that have not been updated with the pace of current research (Radcliffe et al., 2009).

The primary objective of this research was to evaluate, identify, and improve the mechanics and representation of soil P as simulated within the WSM. Priority areas of consideration addressed the simulation of biogeochemical P cycling as it relates to defined soil P pools and interactions among pools of soil P; movement and transport mechanisms of P within the soil profile and to the edge-of-field (EOF); and estimations of P losses to the EOF.

For WSM purposes, EOF P loss is defined as the expected P load from a defined land area (e.g. one acre) carried toward a receiving stream (USEPA, 2010c). The current WSM methodology does not estimate EOF P losses across the diverse landscape of the Bay watershed. Rather, for use in the WSM, EOF P load targets were defined for each land-use category based on literature sources and calibrated to in-stream P concentration data (USEPA, 2010c). The WSM was optimized to achieve these assigned EOF P-loading targets during simulations and the results were deemed “reasonable” for P loadings to a stream (USEPA, 2010c). Alternatively, the use of an independent P loss assessment tool for estimation of EOF P loading would provide a more reliable estimation of nonpoint source P transport from diverse landscapes.

The goal of this research was to offer improved modeling of soil P dynamics. This provides more reliable predictions of EOF P losses from agricultural landscapes and more accurate estimations of soil labile P concentrations over time. Revision of the WSM’s current simplistic soil P simulations to more accurately represent the complex interactions that exist among soil P forms and pools can provide further validity to the watershed modeling process for estimating non-point source P losses and projecting future off-field P loads. Given that the Chesapeake Bay TMDL and subsequent nutrient load allocations are currently defined by USEPA for each state, and states are in-progress to accomplish the mandated P load reductions, the implications of alternative P loss simulations may have great impact on the processes, effectiveness and achievability of the Chesapeake Bay watershed states’ plans for achieving mandated P loading reduction goals.

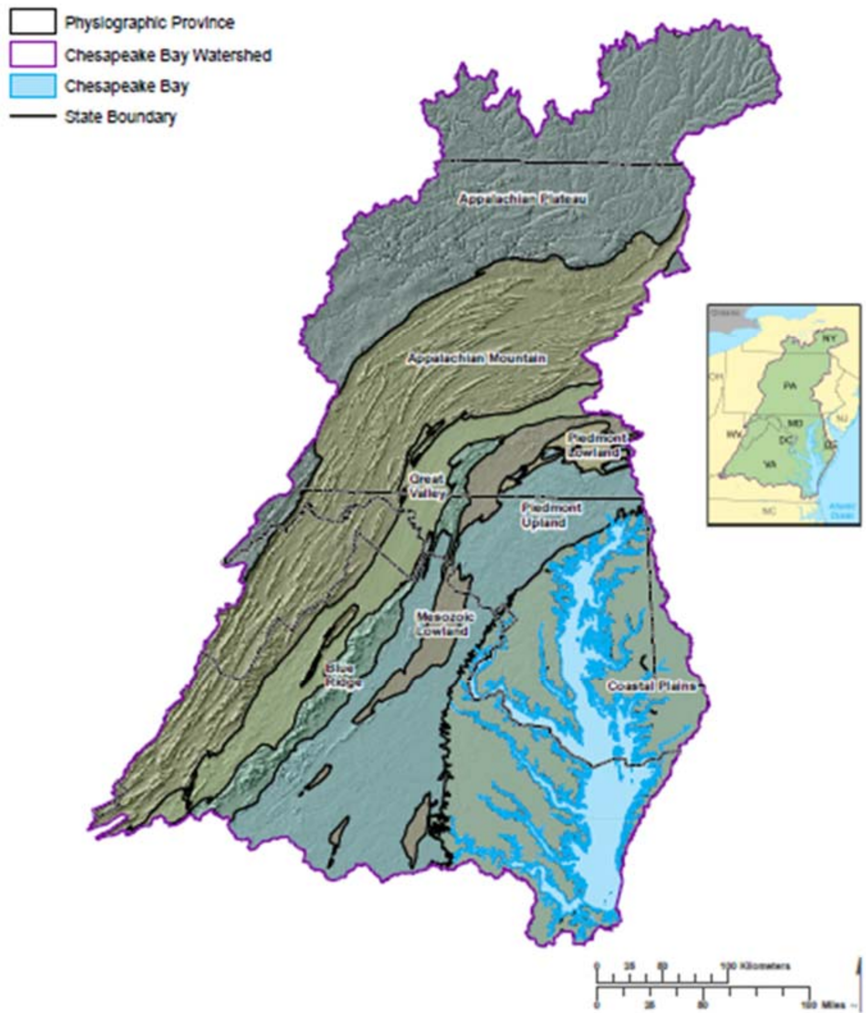
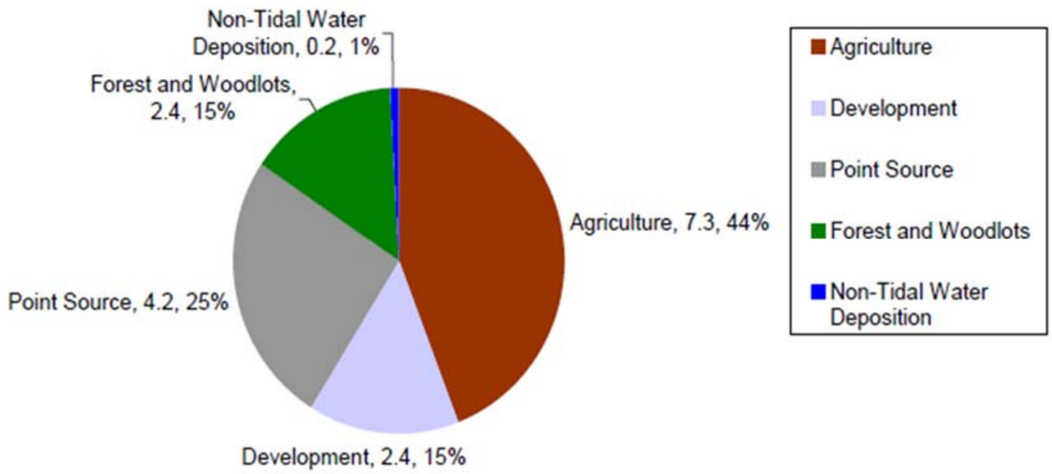


Figure 1.1: Physiographic regions of the Chesapeake Bay watershed (image courtesy of the Chesapeake Bay Program, www.chesapeakebay.net, accessed September 22, 2014)



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Chapter 2: Modeling phosphorus in the Chesapeake Bay watershed

2.1 The Chesapeake Bay Model Suite

2.1.1 Goals of the Chesapeake Bay Watershed Model

Across a given watershed, there is inherent variability in the land uses, agricultural practices, and hydrology. Often the spatial scale of a watershed can be prohibitive for directly measuring nutrient and sediment losses to a receiving stream. As a result, watershed managers increasingly employ hydrologic simulation models to estimate pollutant loads and their impacts to water quality (Williams et al., 1990; Arnold et al., 2012). Version Phase 5.3.2 of the Chesapeake Bay Watershed Model (WSM)[1]simulated pollutant loads from landscapes within the Bay watershed. This model was a lumped parameter model, segmented into land and river segments, overlaid to mimic nutrient and sediment delivery, interaction, and transport across hypothetical management scenarios (USEPA, 2010c). The WSM simulation occurred over the 1985-2005 time period, and results are calibrated to flow data and water quality monitoring data collected from the same time period (USEPA, 2010c).

The domain of the WSM expanded beyond the physical boundaries of the Bay watershed to cover the whole of the states of Virginia, Delaware, Maryland, and the District of Columbia, large portions of New York, Pennsylvania, and West Virginia, and small areas within North Carolina and Tennessee. The WSM used data inputs of estimated nutrient and sediment loads, combined with atmospheric deposition, meteorological data, and land use trends, to simulate likely hydrologic movement within

[1] All references throughout this document to the WSM denote Phase 5.3.2 unless otherwise noted.

the watershed and delivery to the Bay. Subsequently, loads were simulated in an Estuary Model to predict responses in water quality and aquatic resources. The integration of the WSM and the Estuary Model were the primary components of a suite of models for determining achievement of the TMDL goals (Figure 2.1).

2.1.2 Defining land and river segments

The mechanics of the WSM simulation relied on numerous land and river segments apportioned in the model design. The WSM tool has been available since the 1980's, with each subsequent version including a greater number of segments and longer simulation periods for a finer scale representation of the watershed (Donigian et al., 1994). The WSM included over 1000 river segments. Due largely to the availability of input data at the county level, land segments were generally defined at the county scale. A list of the counties include in the WSM and their corresponding land segment codes can be found in Appendix A. Separately, the river segments represented the land area draining to a defined section of a river or reservoir known as a "river reach". The WSM simulated river reaches that had average annual water flow greater than 100 $ft^3\ s^{-1}$ (2.83 $m^3\ s^{-1}$), thus smaller tributary streams were largely excluded from the WSM simulation (USEPA, 2010c). River segment boundaries were primarily defined to match those of the U.S. Geological Survey's (USGS) SPARROW model for calibration purposes (Martucci et al., 2006). Next, land and river segments were overlaid to distribute land segment loads to the appropriate river segment, that is to say, one watershed land segment could include one or more river segments to which it could contribute water flow. At pre-determined time intervals within the WSM, water flow and nutrient and sediment loads were calculated for each overlaid land segment to estimate their inflow to a river reach and their transport to the next river reach downstream.

2.1.3 Simulating delivered loads

The WSM was designed to primarily operate at the watershed scale rather than at the field scale. Accordingly, within the WSM, sediment and nutrient loads were simulated as delivery to the edge-of-stream (EOS) where they were calibrated to observed in-stream measurements. Conceptually, load deliveries to the EOS were the nutrient and sediment loads transported to the stream from erosion, runoff, or subsurface losses at the EOF. Sediment losses to the EOF were estimated for each land segment as an annual erosion rate (tons acre^{-1}or Mg ha^{-1}) based on assessments from the National Resources Inventory (NRI), Universal Soil Loss Equation (USLE), and literature review (USEPA, 2010c). Specifically, the NRI data provided an annual average erosion rate estimated by USLE or Revised USLE from the years 1982 and 1987 to derive a base rate for each land segment (USEPA, 2010c). Because the NRI data does not represent differences in tillage practices, the selection of years 1982 and 1987 were considered a conservative and appropriate representation of high-tillage practices likely occurring during that timeframe. Accordingly, erosion rates for high-tillage were set at 125% of the NRI average erosion rate and the erosion rate for conservation-tillage was set at 75% of the NRI average erosion rate to distinguish between field management practices within a land segment (USEPA, 2010c). To account for the dynamics occurring between the EOF and EOS that preserved sediment loads from reaching the EOS, a fractional Sediment

Delivery Factor that ranged in value from 0 to 1 was also applied to estimated eroded sediment loads (USEPA, 2010c).

To develop estimates of EOF nutrient losses in the WSM, an approach different from that used for sediment loss estimation was necessary due to the sparsity of field studies attributing nutrient losses to specific land uses (USEPA, 2010c). Instead, estimated P loads were established based on literature synthesis that provided a starting point for median P loss “targets” (Table 2.1). Edge-of-field P load targets were established as a base value for a given land use prior to P load reductions attributable to the implementation of conservation best management practices (BMPs) or physiographic variability (USEPA, 2010c). Phosphorus EOF losses were not directly simulated in the WSM, rather nutrient losses were simulated as EOS delivery to a river reach with flows greater than 100 ft^3 s^{-1}.Comparable to the Sediment Delivery Factor that was applied to the sediment load estimates, a fractional Regional Factor that ranged from 0.5 to 2 was applied to nutrient EOS loads to represent the processes that transport nutrients from EOF to stream and then stream to river (USEPA, 2010c).

The agricultural land use categories defined in the WSM were pasture, degraded riparian pasture, alfalfa hay production, unfertilized mixed grass hay, fertilized mixed grass hay, horticultural nursery stock production, conventional tillage row crop production with manure applications, conventional tillage row crop production without manure application, conservation tillage row crop production with manure application, and the facilities of animal feeding operations (USEPA, 2010c). Additionally several agricultural land uses were further subcategorized to account for the implementation of nutrient management planning. For example, “nutrient management pasture” would represent pasture acreage on which purposeful nutrient management planning was occurring (USEPA, 2010c).

Across the Bay watershed, land use assumptions were initiated with digital land coverage data followed by additional spatial refinement for agricultural lands. The U.S. Department of Agriculture’s (USDA) Census of Agriculture data was used for county-level detail of agricultural land acreage, crop type acreage, crop yield, animal numbers, and nutrient use. The Census of Agriculture is conducted every five years with data extrapolation occurring in intervening years for use in the WSM. Land uses at the county level were simulated as an average acre within each land segment, and crop types were grouped into composite crop sets based on common growing seasons, nutrient uptake, and nutrient input requirements (USEPA, 2012).

Management actions occurring on the land largely affect the potential nutrient or sediment loads delivered to a stream. To simulate alternative management scenarios, an annual database management tool known as Scenario Builder was used to track nutrients from generation to application, and will be discussed more completely later in this chapter.

2.1.4 Phosphorus inputs and uptake assumptions

The primary P input to agricultural lands occurs via managed nutrient applications because atmospheric deposition of P is widely considered to be a negligible contributor to the agricultural P load (USEPA, 2010c; Boynton, 1995). Animal manures, biosolids (municipal wastewater treatment residuals), and fertilizer inputs were estimated through

Scenario Builder (USEPA, 2012).Scenario Builder is an annual planning tool central to evaluating the predicted P loads derived from various land uses and management practices that has sufficient user-flexibility to evaluate multiple manure and fertilizer application scenarios. Scenario Builder users can adjust parameters for land use acreage, implemented BMPs and related load reduction efficiencies, estimated crop yield, and nutrient application rates. The application rate and timing of nutrient inputs, based on nutrient management recommendations, corresponded to regional growing seasons and crop needs to maximize crop uptake of applied nutrients. A fractional uptake for each month per year per county per crop was also calculated based on the recommended planting dates within the Bay watershed based on USDA plant hardiness zones and growing degree days. Application of fertilizer, biosolids, and manure to row crop and pasture land were based on definitions within Scenario Builder using a mass-balance approach that applied starter inorganic fertilizer, biosolids, manure, and additional inorganic fertilizer (if needed), in this order, to meet crop nutrient needs in a monthly per county per crop allocation. Manure directly excreted by grazing animals on pasture was not applied towards satisfying the pasture crop's nutrient needs. The mass-balance approach that was utilized implied that all manure nutrients generated annually in a county were applied, minus losses for N volatilization or losses of N or P incurred during storage and handling. Recommendations for nutrient application rates were based on N-based nutrient management planning strategies (USEPA, 2012). However, modeled nutrient application rates could exceed the agronomic recommended application rate where excess manure or biosolids existed in a county, per the annual mass-balance methodology. Application of P for meeting crop nutrient needs was also based on agronomic recommendations, but could also be exceeded since the typical P to N ratio of organic nutrient sources (i.e. manure and biosolids) is generally relatively higher than the P to N ratio required for crop growth (Ekholm et al., 2005). Using the assumptions outlined above, Scenario Builder provided input data to the WSM for annual P application rates to agricultural land and maximum expected crop uptake of applied nutrients over the simulation period for an average composite group of crops produced on each land segment.

2.1.5 Soil phosphorus cycling

For most agricultural land uses within the WSM, a subroutine known as AGCHEM was used for the soil P cycling simulation (USEPA, 2010c). Within AGCHEM, P may percolate through the soil profile in four model-defined soil layers: surface, upper, lower, and groundwater zones. Soil layers were assumed to be evenly mixed and at instant equilibrium upon nutrient application (Donigian et al., 1994).Mechanisms for the movement of P through the soil profile primarily occurred in two of the four defined layers, the surface and upper soil layers. Per the AGCHEM module definitions, organic P or sediment-bound P particles were subject to loss by surface soil erosion, and soil dissolved P was subject to loss via surface runoff or subsurface flow. Soil temperature, moisture, and sediment detachment factors influenced the potential for organic P, sediment-bound P and dissolved P transport.

Unique values for AGCHEM parameters can be used to represent the transformation and movement of soil P in its different simulated forms: dissolved phosphates, insoluble inorganic P, and organic P. However, although several parameters existed for the AGCHEM simulation, only four parameters were identified as sensitive for WSM calibration: 1) the rate of return of soil solution dissolved phosphate to organic P (P immobilization); 2) the organic P mineralization rate via microbial activity; 3) a Freundlich isotherm coefficient that controlled the P adsorption rate to soil solid phases; and 4) the capacity of the soil to permanently bind P (irreversible adsorption and precipitation) (USEPA, 2010c). These parameters were then manually adjusted within a specified range to optimize the EOF total P target loads as shown in Table 2.1 (personal communication, G. Shenk, February 19, 2012). These manual parameter adjustments, and the resulting distribution of soil P among simulated forms and soil pools attempted to describe the fate and transformations of soil dissolved P by simulating nutrient application, soil storage, and subsequent movement within the soil environment (Bicknell et al., 2001). To mimic solid-phase adsorption of soil solution P, a linear Freundlich isotherm was used under the assumptions of instant equilibrium between the soil solution and solid-phase soil particles, i.e., the isotherm dictated the exchange between P in the soil solution and P adsorbed to the soil solid phase. An additional condition of the soil P adsorption simulation was an imposed maximum soil P adsorption capacity that was established to maintain the target total P EOF export loads (personal communication, G. Shenk, February 19, 2012). Second, the complexity of organic P bioavailability was not simulated within the WSM. Rather, organic P concentrations were defined by a Redfield N:P ratio of 16:1 to match the in-stream ecological needs of the aquatic community whereby the Redfield ratio represented both the chemical makeup of phytoplankton as primary producers in the aquatic ecosystem, and the dissolved organic contribution to biological oxygen demand (USEPA, 2010c,personal communication, G. Shenk, February 19, 2012). Dissolved organic P is generally considered slowly bioavailable for aquatic primary production because it must first be decomposed into a labile (available) inorganic form prior to assimilation (McDowell et al., 2004).

Alternatively to AGCHEM, the PQUAL subroutine was also available within the WSM for nutrient simulation (USEPA, 2010c). As described above, AGCHEM used a nutrient mass balance approach to simulate soil P cycling. The alternative subroutine, PQUAL, used a simpler coefficient approach that associated soil P concentrations with a load of sediment-bound P loss or runoff P loss with the addition of a “potency factor”. Potency factors represent the relative strength of a pollutant in proportion to the load, with the assumption that dissolved phosphates are rapidly and strongly absorbed to soil particles (Donigian et al., 1994).

2.1.6 Watershed model calibration

Critical to all hydrologic simulation models is confidence in the parameter input values and the output. Therefore, throughout the WSM process, calibrations steps wereput in place to improve assurance and confidence in the results. As previously mentioned, AGCHEM parameters were manually adjusted to maintain the target total P EOF export loading rates. While target estimations for P loss were primarily developed based on literature reviews, EOF P loading is highly variable for any given agricultural practice (Beaulac and Reckhow, 1982; Harmel et al., 2006). To approximate the high spatial variability of P delivery to the EOS, the EOF loads were modified geographically with regional factors to better align estimated EOS loads with observed water quality data. After P was delivered in-stream, other calibration factors were assigned to account for fluvial processes that occurred during movement downstream. In most cases, P was considered “conservative” - deposited and held in-stream by sediments for relatively long durations and only moved downstream during a high-flow or scouring events (USEPA, 2010c).

Foremost in the calibration of EOF and EOS targets was the network of established USGS water quality monitoring stations to provide historical data for comparison of estimated loads against physical water quality monitoring data. The continued iteration and calibration of the WSM over the 1985-2005 simulation period attempted to reasonably match the WSM estimated loads with observed loads. The calibrated WSM was then used to project future P loads. Confidence in the accurate estimation of future P loads from agricultural land is essential to the TMDL process. Public agency managers in the Bay states are proceeding with implementation of agricultural best management practices (BMPs) to achieve aggressive P-loading reductions that are dependent on accurate local P loading estimates generated by the WSM. The accuracy of these P-loading projections have meaningful impacts on a state’s ability to meet the requirements of the TMDL allocation.

<u>2.2 Soil Phosphorus Dynamics</u>

2.2.1 Soil phosphorus pools

Soil P is a macronutrient critical to plant growth and is primarily taken up by plants as inorganic phosphates ($H_2PO_4^-$, HPO_4^{2-}) dissolved in the soil solution. While the dynamics of soil P include complex biogeochemical processes, for practical purposes of modeling soil P it is often described as different soil P pools to designate its form, availability, and transport characteristics. Phosphates dissolved in soil solution and the P compounds that can readily desorb or mineralize into soil solution for biological availability are known as the labile P pool (Stevenson and Cole, 1999). However, soil P is often considered mostly insoluble such that soluble P compounds added to the soil will readily adsorb to clay particles or precipitate with aluminum, iron, or calcium cations (Brady and Weil, 2002). Increased ionic strength decreases the rate of P desorption from solid soil surfaces into the soil solution, so the concentration of dissolved P in solution is generally low. To replenish the labile P pool, a dynamic equilibrium with an active and stable P pool occurs, where the active pool is considered to be in rapid equilibrium with the labile pool while the stable P pool is very slowly acting (Jones et al., 1984). The stable P pool is considered to be the less soluble and physically occluded portions that contain both organic and inorganic P compounds strongly resistant to desorption and mineralization. Exchanges between the labile, active, and stable pools occur through the processes of adsorption and desorption. Organic P is highly complex and greatly influenced by organic inputs to the soil and soil microorganism activity (Stevenson and Cole, 1999). The equilibrium between the labile pool and the organic pool occurs via immobilization and mineralization processes. Mineralization is the release of organic P into solution by soil microorganisms when their metabolic requirements for P are satisfied. Conversely, soil microorganisms may immobilize P to satisfy population growth demands. Other factors such as soil temperature and ratios of carbon, N, and P influence these microbial-mediated processes. The cycling between the soil P pools occurs to achieve equilibrium, but typically favors P adsorption and immobilization (Sharpley, 2000).

When nutrient applications are made to an agricultural field, the physical and chemical properties of the nutrient compound will affect the overall soil P response. In most cases, inorganic mineral fertilizer is added at agronomic rates to meet a crop's nutrient needs. Depending on management strategy, P applications are intended to address either immediate crop needs, crop needs for the duration of the growing season, or crop needs for multiple growing seasons. Therefore, P applications may be made more than once during a growing season, once per season, or a single application may be made to satisfy P needs for multiple growing seasons. While mineral fertilizers come in different forms, all are largely considered soluble and thus contribute to the labile P pool. In turn, fertilizer P additions disrupt the equilibrium between the P in the soil solution and the P on the solid phases (active, stable, and organic pools), triggering the adsorption/desorption and mineralization/immobilization processes that maintain soil solution P equilibrium. Organic nutrients, such as manures and biosolids, are also applied to fields to meet crop nutritional needs. However, unlike fertilizers that can be custom blended and precisely applied to accurately meet the crop's nutrition requirements, organic nutrient sources cannot readily be chemically separated prior to field application. Frequently, organic nutrient application rates are N-based and with typical N:P ratios of organic nutrients, higher relative proportions of P are added to the soil compared to N, relative to crop need. With repeated organic nutrient application, the excess applied P may accumulate creating P-enriched soils with measurably increased labile P concentrations. This scenario has been documented in the Bay watershed where the growth in animal production operations has created excess local supplies of manure P leading to on-field P applications in excess of agronomic nutrient needs and increasing the likelihood for soil P over-enrichment (Sharpley, 2000; Butler and Coale, 2005).

2.2.2 Measuring phosphorus

Phosphorus losses to the EOF or in-stream are often reported in concentrations of total P, which includes both the dissolved P (DP) in solution and solid phase particulate P (PP) adsorbed to soil particles. Total P concentration of water samples is often determined by chemical digestion methods. Typically, portions retained by a 0.45 µm filter are considered PP and digestion of unfiltered samples yield total P concentrations. Similarly, digestion of the portion of a water sample that passes through a 0.45 µm filter is considered DP. The difference between total P and DP can be calculated as a measure of PP (McDowell et al., 2004).

Within crop fields, agronomic soils tests are essential to nutrient planning. Traditionally, soil tests are used to assess the available nutrient concentration of the soil followed by a recommended nutrient application, if any, to enhance crop growth throughout the growing season. The goal of the soil test is to predict a crop’s response to P inputs; however, an increasing amount of research is evaluating the role of soil test P (STP) methods for their ability to assess the environmental risks of P loss from a field (Sharpley, 2000). Within the Bay watershed, common STP methods include Mehlich-1, Mehlich-3, and Bray-1 extractions (Gartley et al., 2002). The extractions are dilute chemical solutions that remove P from soils via dissolution and desorption to estimate crop response to soil P through the growing season (Sims, 1998). Correlations exist between these laboratory extraction methods that allow results to be interconverted among soil testing procedures (Sims, 1989). Other STPs are also considered environmental predictors of P loss, including water soluble P, potentially desorbable P, and degree of P saturation (DPS). Water soluble P is an extraction by water or dilute salt solution measuring the soil DP concentration at the time of sampling, though it is noted by Sims (1998) as a method easily influenced by recent P inputs. Stronger extractants (e.g. ion exchange resins or iron-oxide strips) provide a relatively un-saturable medium onto which P may readily adsorb and subsequently estimate the soil's labile P pool (Vadas et al., 2006; Vadas and White, 2010). Use of the DPS method relies on extracting soil with an ammonium oxalate solution to measure aluminum, iron, and P. It is commonly reported as a percentage expressing iron and aluminum bound P as a measure of the soil's relative P saturation capacity, and as an estimate of the soil's likelihood to release P to solution (Sims, 1998). Findings of STP research, whether for agronomic or environmental purposes, conclude strong linear relationships between soil test P measured by various STP methods and DP measured in field runoff (Vadas et al., 2005; Sharpley, 1995; Sharpley, 2000) allowing for improved assessment of risks for soil P loss.

Manure P content is strongly influenced by animal type and diet. Handbooks published regularly by the American Society of Agricultural Engineers (ASAE) act as a reference to calculate the mass of manure generated by an animal and the characteristics for the nutrient content of the manure (ASAE, 2003). References such as ASAE report "typical" manure characteristics rather than the greater variability in nutrient content based on diet and individual farm management (Pagliari and Laboski, 2012). However, this variability in manure nutrient content is not accounted for in the WSM, rather the ASAE reference values are assumed to be representative of the Bay watershed. Additionally, estimates of potential P loss from field-applied manures have been correlated to the variable P content of different manures. For example, Sharpley and Moyer (2000) studied the release of P under simulated rainfall on soil columns for swine, dairy, and poultry manures and concluded that the water extractable P (WEP) content of the manures provided the best estimate of P surface runoff losses. These findings were confirmed by further research from Kleinman, et al. (2002 and 2005), and by Vadas et al. (2005b).

2.2.3 Phosphorus transport mechanisms

Soil P is generally characterized by its insolubility and adsorption tendencies. Thus, concerns over P losses and off-site P transport focus primarily on DP loss by surface runoff and PP loss by sediment erosion from surface soil layers. The primary factors that determine the amount and mechanism for P loss are topography, soil P concentration, hydrology, and soil type (Sharpley, 2000).

First, research has demonstrated that long-term application of manure as a nutrient source is largely associated with an increased risk for P runoff, though the soil P concentration, characteristics of the manure source, incorporation methods and landscape position are also important contributing factors (Buda et al., 2009; Kleinman et al., 2002;

McDowell and Sharpley, 2002). Portions of the Coastal Plain of the Bay watershed are dissected by an extensive system of field drainage ditches designed to facilitate farming of poorly drained coastal soils. This subregion is dominated by agricultural land uses, particularly grain production and a high density of poultry production, creating a long history of field application of P-rich poultry litter in excess of crop needs. As a result, isolated regions of the landscape are characterized by P-enriched soils from which DP is readily available for transport via field drainage pathways to Bay tributaries (Kleinman et al., 2007). Long-term poultry litter waste applications on pastured terrain have been reported to increase soil Mehlich-3 P concentrations 85 times higher than unamended soils (Curtis et al., 2010). In the Bay watershed's central Pennsylvania Ridge and Valley region, soil studies pre and post application of swine manure found DP losses were most affected by flowpath length, clay content of the soil, and initial soil P concentration (McDowell and Sharpley, 2002).

Conversely, P applications from some biosolids sources have shown a lower risk for P runoff due to the wastewater treatment processes that enhance P retention. Treatment methods that add aluminum, iron, or lime consistently resulted in lower WEP and runoff DP concentrations when compared to soil application of equivalent total P loads from poultry litter (Penn and Sims, 2002; White et. al, 2010), dairy manure, and fertilizer (Whithers et al., 2001; Brandt et al., 2004; Elliott et al., 2005).

Another important P transport mechanism is sediment erosion with associated adsorbed PP in fine-grained soil particles susceptible to movement with overland flow. Sediment is generated on the landscape from the natural weathering of rock and soil, but can be accelerated by agricultural practices especially related to tillage and ground cover. Studies by the USGS attribute the largest quantities of sediment delivered to the Bay as originating in the three largest rivers within the watershed, the Susquehanna, Potomac, and James Rivers, and the greatest sediment deposition occurs near the physiographic fall line that separates the Piedmont and Coastal Plain regions (Ator et al., 2011). Other studies by McDowell et al. (2001) and McDowell and Sharpley (2003) evaluating P transport in a Central Pennsylvania subwatershed found PP was the largest proportion of measured total P in runoff, and that fractional uptake of P by fluvial sediments had a strong dependence on existing soil cations and adsorption rates increased proportional to soil clay content. The WSM's predictions of sediment losses, and associated P, are especially important given that USGS attributes a high percentage of the Bay watershed's annual sediment loads as originating from agricultural land uses and from stream bank erosion during large storm events (Langland, 2003).

2.3 Predicting and Simulating Phosphorus Transport

2.3.1 Role of model routines to simulate phosphorus transport

Though it is recognized that models are a simplified representation of the landscape and watershed processes, in many ways the use of watershed modeling allows a feasible quantification of pollutant loads over larger landscapes. Any watershed scale decision model designed to simulate the complex soil environment and nutrient cycling yet still account for spatial variability of the landscape is certain to have limitations. For example, many process-based models such as the WSM assume a complete mixing of a soil layer to estimate the quantity of nutrient available for loss. For P, this is problematic given P releases are controlled by chemical and biological dynamic timing rather than an immediate equilibrium (Maguire et al., 2000; Lookman et al., 1995). Accordingly, many models incorporate subroutines to better simulate multiple, distinct processes such as the WSM's nutrient cycling simulations available through AGCHEM and PQUAL. The WSM simulation for landscape P losses is calibrated to the ultimate delivery of total P at an EOS with flow greater than 100 $ft^3\ s^{-1}$. Therefore, true simulations of in-field geochemical P processes, EOF P losses, and P transport from the field to small order streams, and subsequent in-stream delivery are necessarily less sophisticated and incorporation of alternative methods for independent simulation of EOF P losses would enhance the WSM's overall P-loading estimations.

2.3.2 Estimating mineral and manure losses

As mentioned, losses of P are most commonly associated with surface soil layers as sources of runoff from soil P, manure, biosolids, and fertilizer. Improving the prediction and model representation of these sources is critical to the goals of watershed modeling. For example, research has shown that P releases to surface runoff from recent nutrient applications declines with time and rain events (Kleinman et al, 2002; Penn and Sims, 2002). Yet, these relationships are not appropriately simulated in most hydrologic models since many model routines assume a high level of soil incorporation, thus potentially underestimating runoff losses from unincorporated surface applications. To address this inconsistency, Vadas et al. (2005b; 2007; 2009) have developed a series of algorithms to specifically address surface applied nutrients. For example, a model routine was developed to simulate surface-applied manure decomposition for infiltration into soil P pools or losses to runoff. Methods were validated by field data from four states (Pennsylvania, Texas, Arkansas, and Georgia) and found the model to accurately simulate P dynamics in long term scenarios (Vadas et al., 2007).

The assumption of complete mixing of soil layers has also been applied to the application of mineral fertilizers. This could result in an underestimation of P runoff when applications are surface applied. Accordingly, Vadas et al. (2008) developed a model routine to simulate surface-applied fertilizer, and validated their results with published data from 11 runoff studies including a range of runoff and precipitation conditions, soil cover, and field sizes.

Additionally, outcomes from Vadas et al. (2004) empirical models have found results were sensitive to precipitation and runoff data. Relationships and model predictions of P losses from agricultural watersheds were shown to improve through the addition of a runoff to precipitation volume ratio and a P distribution factor that proportions the release of P to infiltration and runoff. These model routines also reflect the findings of McGrath et al. (2005) that demonstrated that as manure decomposes, manure available WEP changes.

2.3.3 Estimating soil phosphorus losses

One early method for estimating P losses was based on Beaulac and Reckhow's (1982) comprehensive literature review of field studies that culminated in an estimation of expected P loss as an export coefficient based largely on crop type or rotation and tillage practices. Using Beaulac and Reckhow's (1982) summations, model developers could associate a coefficient to given field conditions to predict P losses. Building on these concepts and an increasing amount of field research, a strong relationship has been established between the surface STP and the DP in surface runoff as a function of a soil's physical and chemical properties (Sharpley, 1995). As a result, surface runoff DP losses for modeling purposes can be calculated as the product of an STP extraction coefficient, the labile soil P in a depth of surface soil, and the overland flow volume (Sharpley et al. 2003). The extraction coefficient is generally the slope of the linear relationship of the measured runoff DP concentration versus the soil STP concentration. Variability in the regression slope has been attributed to soil depth (Andraski and Bundy, 2003), soil clay content (Cox and Hendricks, 2000), and presence of calcareous soils (Torbert et al., 2002). Additionally, Vadas et al. (2005) evaluated the extraction coefficients of 17 published studies that used different STP methods, and found results were not significantly different between Mehlich-3 P and Bray-1 P STP methods. A similar relationship has been derived to estimate PP loss by erosion processes but substitutes a P Enrichment Ratio (PER) parameter in place of an extraction coefficient. Given the affinity of fine clays for P adsorption, it was recognized that the P concentration in eroded soil, and thus the PER parameter, was far greater, or more highly enriched, than the source soil from which the eroded sediment originated (Sharpley et al., 2002).

2.4 Best Management Practices

2.4.1 Accounting for BMP load reductions

The increasing reliance on watershed models for their inherent time and cost effectiveness underscores the importance of updating model routines as new data is available or clarified. Soil P processes are dynamic and complex. Consequently, model estimations of P dynamics can be difficult, or even misleading, to apply across multiple diverse landscape settings in an effort to accurately estimate P losses from agricultural fields (Vadas et al., 2013). An additional factor that complicates the accurate prediction of P losses are the widely implemented agricultural nutrient management BMPs that are not simulated in most process models. Rather, BMP implementations are often tracked for post-processing as an "efficiency" or percentage reductions applied to the base load. In the case of the WSM, expert panels have been convened throughout the TMDL process to craft a methodology for BMP tracking and their associated efficiency credits as further support of nutrient load reductions (Palace et al., 1998). An account of the agricultural BMPs used in the WSM are included in Appendix B. However, using a methodology that applies a percentage reduction to nutrient loads without simulating the explicit effectiveness BMPs provide as implemented to reduce EOF P delivery creates a spatial discontinuity between the source of the P loading and the simulated EOF P loads derived from the WSM.

2.4.2 Cost-effectiveness of BMPs

In the absence of model simulation of BMPs, an alternative approach is to evaluate the cost effectiveness of BMPs per pound of nutrient reduction in order to prioritize and target BMP implementation. Recent efforts to derive the costs and cost-effectiveness for some BMPs included the report by Wieland et al. (2009) assessing 12 prevalent BMPs promoted in Maryland. Costs were defined as program costs, noting many states subsidize an operator's participation to further incentivize a practice, and constructed costs of the practice. Cost effectiveness was calculated as cost per pound per year of nutrient reduction using the nutrient efficiencies defined by the WSM methodology. Wieland et al. (2009) revealed the complexity in assigning a singular cost effectiveness to a particular BMP because of the many implementation scenarios that can exist on the landscape. They also suggested a payment scheme that better aligns subsidy payments with the nutrient reduction credit, rather than the fixed price payments generally used in current programs. The consequences of this shift would require greater assurances in the nutrient reduction credit (efficiency) assumed for each practice as determined by the WSM (Wieland et al., 2009).

A need for increased certainty in the efficiency credit for BMPs has been echoed by subsequent optimization studies (Chesapeake Bay Commission, 2012; Wainger et al., 2013) that evaluated costs for reaching the TMDL goals within the Bay watershed. When BMP options across the agricultural sector were compared to those in the urban stormwater sector and urban point source (e.g. waste water treatment plant) sector, both studies consistently found the least-cost options for achieving the TMDL goals resulted from reductions achieved by BMP implementation in the agricultural sector (Chesapeake Bay Commission, 2012; Wainger et al., 2013). Specifically, Wainger et al. (2013) developed an optimization framework for the Potomac River Basin to quantify trade-offs between BMPs for achieving the TMDL goals and co-benefits of ecosystem services, where co-benefits were defined as benefits beyond water quality improvement in the Bay. The primary finding was the conversion of agricultural production land to a re-vegetated native plant community condition resulted in the least-cost option across defined scenarios. Likewise, the Chesapeake Bay Commission's (2012) optimization study recognized the significant costs of achieving the TMDL would be borne primarily by tax payers within the Bay watershed. Their report advocated a nutrient trading framework across sectors and jurisdictions to mitigate tax payer burdens.

The inference of all such economic analyses is that achievement of the Bay TMDL goals will require difficult choices between BMP implementation and costs. A strategy to prioritize among these choices is critical and must be integrated into the TMDL process. Additionally, as new data relative to BMP cost estimates and BMP efficiency become available, BMP cost effectiveness should be re-evaluated with new economic analyses.

2.5Conclusion

The WSM simulates the transport of nutrients and sediment across the watershed and the subsequent delivery to the Bay. The use of the WSM is necessary to achieve a simulation of spatial and temporal trends across the vast watershed and to quantify water quality improvements mandated by the USEPA's Bay TMDL. However, simulating the diverse and dynamic landscape of the Bay watershed has often resulted in WSM parameter assumptions and subroutines that do not align with current soil P research. Rather WSM parameters were based on average, composite data sources that cannot adequately reflect unique P source(s) and transport mechanisms within an agricultural field. In the absence of a simulation at the edge-of-field scale, the WSM methodology assumed a target (base case) EOF P loss uniformly across the Bay watershed (Table 2.1). In contrast, EOF P losses are greatly influenced by topography, soil P concentrations, hydrology, and soil type. In addition, relative changes to the soil P concentration and cycling between the soil P pools is dependent on nutrient P application source and rate, soil properties, and mixing between soil layers. Such factors that influence EOF P losses and soil P concentrations are inherently occurring at scales smaller than the capability of the WSM. Thus a need exists for an alternative soil P simulation that is capable of better estimating EOF P losses and changes in soil P concentrations. With an improved simulation, a better understanding of management actions and appropriate, effective BMPs could be implemented.

Table 2.1: Edge-of-field (EOF) total P load targets (mean and median) used in the Chesapeake Bay Program Watershed Model (WSM) for five agricultural land uses. Targets were developed for all land uses and land segments to represent an estimated total P load loss from the EOF prior to application of load reductions attributable to implementation of best management practices (BMPs) (USEPA, 2010c).

Land-Use Category	Land-Use Code	WSM Mean EOF Target (kg P ha^{-1})	WSM Median EOF Target (kg P ha^{-1})
High-till with Manure	HWM	2.43	2.22
Low-till with Manure	LWM	2.40	2.21
Nutrient Management High-till with Manure	NHI	2.18	2.00
Nutrient Management Low-till with Manure	NLO	2.15	1.99
Pasture	PAS	0.922	0.788

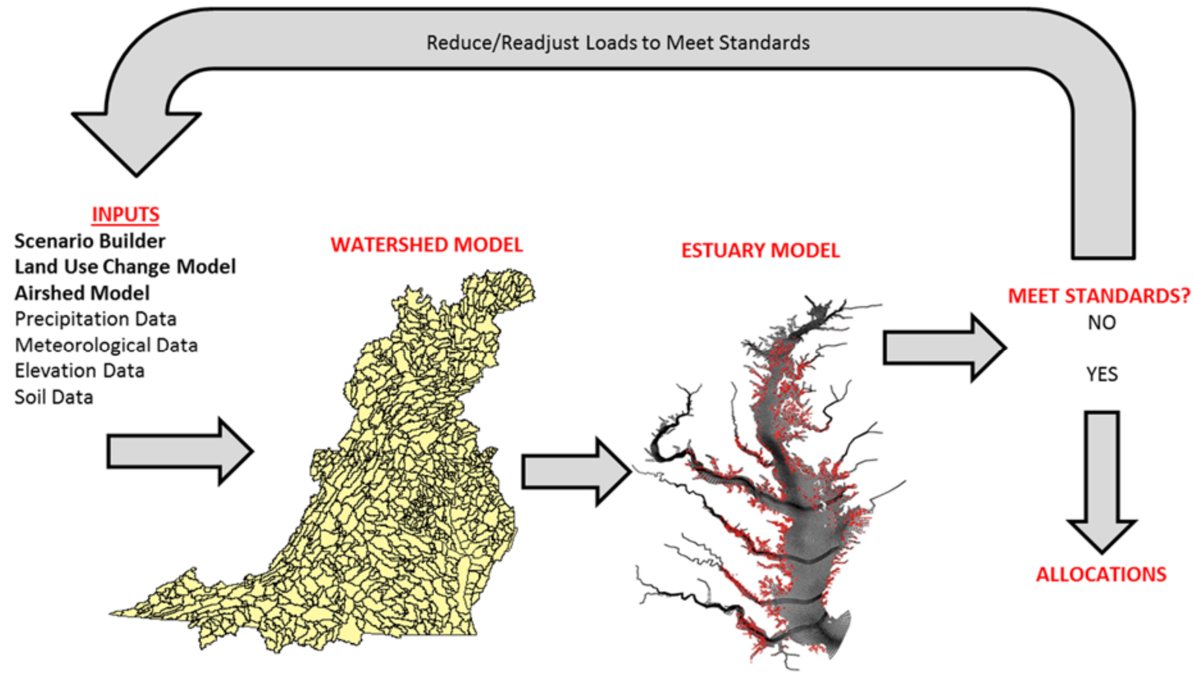

Figure 2.1: Conceptual flow diagram of the Chesapeake Bay Model Package process (redrawn from USEPA, courtesy of Chesapeake Bay Program, www.chesapeakebay.net, accessed*October 6, 2014)*

Chapter 3: Estimation of Phosphorus Losses Based on Land Uses

3.1 Abstract

Current restoration efforts for the Chesapeake Bay watershed mandate a timeline for reducing the load of nutrients and sediment to receiving waters. The Chesapeake Bay Watershed Model (WSM) did not have a simulation of edge-of-field (EOF) P losses, but rather relied on literature values of assumed target losses based on land use categories. Our objective was to simulate EOF P losses by using an independent modeling tool (APLE) 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. Outcomes represent revised estimates of EOF surface P losses. Findings from APLE exhibit a broader range and greater mean estimate of total phosphorus loss compared to the WSM, with estimates of sediment-bound P contributing the largest portion of Bay-wide total phosphorus losses (76% to 96% depending on land use). Additional analysis of the APLE-estimated EOF P losses through incorporation into the WSM resulted in improvements to the WSM performance.

3.2 Introduction

A comprehensive Total Maximum Daily Load (TMDL) was mandated in the Chesapeake Bay (Bay) watershed in 2010 to address excess nutrient and sediment loads that have resulted in eutrophication (USEPA, 2010a). The TMDL creates a 2025 implementation deadline for reducing nutrient and sediments loads, including a 24% reduction in P loading (USEPA, 2010a). Eutrophication has been a persistent concern to Bay health, and much of the excess nutrient loads are attributed to nonpoint sources from agriculture (USEPA, 2010c). At present, many states within the Bay watershed consider agricultural sources to be the most meaningful and cost-effective option to achieve required reductions in nutrient loading (USEPA, 2010b, Wainger et al., 2013).

Watershed managers use the Chesapeake Bay Watershed Model (WSM) to estimate P loads to the Bay (USEPA, 2010c). The WSM is a primary component of the larger suite of integrated simulation models that estimate the nutrient and sediment loads to the Bay over a defined duration. The function of the WSM is to quantify the maximum loading allowable from multiple sources while still attaining the water quality standards required by the TMDL. Though models like the WSM are commonly used to estimate nutrient and sediment loads across large landscapes, in many cases the P loads were derived from model routines that have not been updated with the pace of current research (Radcliffe et al., 2009, Radcliffe et al., 2014 (in review). In addition, validating EOF P loads at broader spatial scales challenges modelers due to limited field studies' attributing P losses to specific land uses (USEPA, 2010c). For modeling purposes, EOF is defined as the expected load loss from one simulated acre carried toward a receiving stream (USEPA, 2010c).

The WSM used literature values as the starting point for a P loss "target" to represent average export rates from agricultural land uses (USEPA, 2010c). The target loads were considered to be the base P load to the edge-of-field (EOF)from a given land use prior to any reductions attributable to implemented best management practices (BMPs) or loading differences attributed to physiographic regions within the Bay watershed. The EOF P loading targets represented the cumulative P loss associated with surface runoff, subsurface transport, and eroded sediment. Sediment losses, commonly considered a primary source of P transport to the EOF (McDowell et al., 2001), were estimated on a county basis as an annual erosion rate (kg ha^{-1}) based on assessments from the National Resources Inventory, Universal Soil Loss Equation, and literature review (USEPA, 2010c). While a portion of the sediment load was considered to be contributing to P loss, it was not defined as a distinct load type (e.g. sediment P), per se, because of WSM limitations.

Given the limits of the current WSM, the primary objective of our study was to evaluate alternative methods to simulate soil P as it relates to P transport mechanisms toward the EOF and P loss estimates to the EOF. Using an independent P loss assessment tool as an alternative to the current WSM subroutines for soil P simulation, may offer an improved modeling method for predicting EOF P losses from agricultural land uses.

3.3 Materials and Methods

3.3.1 Overview

The WSM simulates agricultural practices at the county scale due to the data availability from the USDA's Census of Agriculture. Accordingly, in this study, simulations were kept at a county-based scale. Data inputs of field and management conditions were also considered representative of a county segment based on source data (Table 3.1).

The alternative modeling tool investigated was the Annual Phosphorus Loss Estimator (APLE) developed by the USDA-ARS (Vadas, 2013). It is intended to be an agricultural field-scale simulation of EOF P losses from surface sources - sediment, soil, manure, and fertilizer – based on easily available user input data and transport factors

(Vadas, 2013). The APLE model runs on an annual time step and a period of 1992-2005 was simulated. Five agricultural land uses were simulated which parallel the land-use categories and definitions used in the WSM: 1) 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).

The WSM did not simulate EOF P losses, rather a target value for P loss was assigned to each land use based on literature review. Targets were further refined by additional factors such as physiographic region to account for EOS P delivery. Conversely, APLE predicts EOF P losses based on field conditions and management as defined by the user. Additionally, the WSM defined all P loss targets as a single cumulative total P load. Conversely, APLE can partition predicted total P losses among four distinct surface sources (sediment P, dissolved soil P, dissolved manure P, and dissolved fertilizer P) where the sum of the four sources is considered the total EOF P load. Some comparison of results between APLE and the WSM are limited by these differences. An example estimate of total P loss from APLE can be found in Appendix C.

3.3.2 Manure and fertilizer P loss equations

The following descriptions of APLE relationships are based on the Theoretical Documentation, Version 2.4 (Vadas, 2013). Manure applications in APLE were differentiated based on the solid content of the manure. For manures with less than 15% solids, it was assumed that 60% of the applied manure infiltrated into the soil and was unavailable for surface losses. The depth of incorporation by tillage, as defined by the user, resulted in decreased surface loss of manure P. Any manure remaining on the soil surface was considered to be available for runoff losses based on the available water extractable P (WEP) content of the manure. The WEP:total P ratio was a user- defined parameter specific to the manure source. The remaining non-WEP portion of the manure P mineralized to WEP during the annual cycle based on a user-defined rate of mineralization, which was set at 15% per year for this study.

Annual manure P runoff (kg ha^{-1}) was estimated by Eq. 3.1, where Manure WEP, expressed as kg P ha^{-1}, was the WEP on the field surface subject to runoff losses after application of reductions accounting for infiltration and tillage incorporation, as applicable.

$$\text{Manure P Runoff} = (\text{Manure WEP})\left(\text{Annual}\frac{\text{Runoff}}{\text{Precipitation}}\right)(\text{P Distribution Factor}) \quad [\text{Eq. 3.1}]$$

The WEP was modified by the ratio of annual runoff and precipitation, measured in mm, and the unitless P Distribution Factor that allocated released P between runoff and infiltration according to Eq. 3.2.

$$\text{P Distribution Factor} = \left(\text{Annual}\frac{\text{Runoff}}{\text{Precipitation}}\right)^{0.225} \quad [\text{Eq. 3.2}]$$

In pasture settings where manure was directly excreted, field coverage was assumed to be non-homogeneous, unlike with machine application, and therefore did not interact uniformly and consistently with surface runoff water following precipitation events. To account for this influence, a reduction factor was applied to directly excreted manures based on the dung coverage over the total pasture area from Eq. 3.3 and Eq. 3.4, where Manure Applied was the mass (kg) of manure.

$$\text{Dung Cover} = \left(\frac{\text{Manure Applied}}{0.25} * \frac{659}{100^4}\right)/\text{hectares [Eq. 3.3]}$$

$$\text{Dung Reduction Factor} = 1.2 * (250 * \text{Dung Cover})/[(250 * \text{Dung Cover}) + 73.1)] \quad \text{[Eq. 3.4]}$$

Estimates of fertilizer P runoff were similar to manure with the exception that all applications were considered to be highly soluble and available for runoff, following reductions attributed to subsurface placement or incorporation by tillage, according to Eq. 3.5 and Eq. 3.6, where Fertilizer P was the application rate (kg ha^{-1}).

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$$\text{Fertilizer P Runoff} = (\text{Fertilizer P})\left(\text{Annual}\frac{\text{Runoff}}{\text{Precipitation}}\right)(\text{Fertilizer P Distribution Factor}) \quad \text{[Eq. 3.5]}$$

$$\text{Fertilizer P Distribution Factor} = 0.034 \exp\left[(3.4)\left(\text{Annual}\frac{\text{Runoff}}{\text{Precipitation}}\right)\right] \quad \text{[Eq. 3.6]}$$

3.3.3 Sediment and soil dissolved P loss equations

APLE estimates sediment bound P loss and soil dissolved P loss. Sediment P was calculated by Eq. 3.7, where Eroded Sediment was annual loss (kg ha^{-1}) and Soil Total P (mg P kg^{-1}) was the concentration of P in the surface soil derived from the clay and organic matter content of the soil, and an initial Mehlich 3-P measure of soil labile P. The P Enrichment was the unitless ratio corresponding to the P content of the eroded soil to that of the source soil, which was inversely proportional to the eroded sediment load.

$$\text{Sediment P Loss} = (\text{Eroded Sediment})(\text{Soil Total P})(\text{P Enrichment})(10^{-6}) \quad \text{[Eq. 3.7]}$$

Soil dissolved P was calculated by Eq. 3.8, where Soil Labile P (mg P kg^{-1}) was one-half the concentration in the surface soil layer based on Mehlich 3-P extractions, and Annual Runoff was in L ha^{-1} modified by an extraction coefficient of 0.005.

$$\text{Dissolved Soil P} = (\text{Soil Labile P})(0.005)(\text{Annual Runoff})(10^{-6}) \quad \text{[Eq. 3.8]}$$

<u>3.4 Results and Discussion</u>

3.4.1 APLE results for the high-till with manure (HWM) land-use

The HWM scenario simulation by APLE resulted in an estimated mean annual total P loss of 10.1 kg ha^{-1}, compared to the mean target value of 2.44 kg ha^{-1} defined in the WSM (Table 3.2). The overall range of APLE estimates for annual total P loss that extended from a minimum of 0.012 to a maximum of 131 kg ha^{-1} , was substantially broader than the range of established WSM total P loss target values (1.70 to 7.51 kgha^{-1}) (Table 3.2). Other quantile results from the HWM land use included 33.3 kg ha^{-1}(95% quantile), 12.8 kg ha^{-1}(75% quantile), 5.98 kg ha^{-1}(50% quantile), 2.61 kg ha^{-1}(25% quantile), and 0.643 kg ha^{-1}(5% quantile) (Table 3.2). Additionally, median total P loss values from HWM land-use areas were depicted graphically by state-bounded sub-watersheds to demonstrate the narrow range of target EOF values used in the WSM compared to the wider range predicted by APLE (Figure 3.1). At the state level, the highest APLE-estimated median total P losses were attributed to Maryland followed by Pennsylvania, New York, Delaware, Virginia, and West Virginia.

Over the entire Bay watershed, sediment P from the HWM land-use was overwhelmingly the largest contributor to APLE-predicted total P loads. The mean annual sediment P loss was 9.87 ± 0.2 kg ha^{-1}, and the median Bay-wide annual sediment P loss was estimated by APLE to be 5.77 kg ha^{-1}, or 96% of the APLE-estimated total P load. Upon evaluation of median APLE sediment P loss estimates from HWM land-use per land segment (i.e. county) for the 14 year simulation period, the following land segments demonstrated the highest predicted sediment P losses (kg ha^{-1}): Calvert, MD (45.6), Prince Georges, MD (44.4), Charles, MD (32.1); Northumberland, PA (22.2), Anne Arundel, MD (21.5), Montour, PA (20.7), Rockingham, VA (19.3), Baltimore, MD (19.0), Lancaster, PA (18.2), and Harford, MD (18.2).

APLE estimates of total dissolved P loss (manure dissolved P loss + fertilizer dissolved P loss + soil dissolved P loss) from the HWM land-use were very small relative to estimated sediment P losses. The Bay-wide mean annual total dissolved P loss was from HWM land-use category was 0.190 ± 0.004 kg ha^{-1}. The median annual total dissolved P loss load was 0.126 kg ha^{-1} and the maximum predicted annual total dissolved P loss was 2.52 kg ha^{-1}.

3.4.2APLE results for nutrient management high-till with manure (NHI) land use

The NHI scenarios estimated by APLE simulation resulted in a mean annual total P loss of 10.9 kg ha^{-1} as compared to the WSM target total P loss load value of 2.19 kg ha^{-1}(Table 3.2). The Bay-wide mean total P loads were similar for HWM and NHI land-uses. Also, similar to the HWM land use, APLE estimates of total P loss for NHI encompassed a much wider range (0.012 to 131 kg ha^{-1}) compared to the range of established WSM total P loss target values (1.53 to 6.76 kg ha^{-1}) (Table 3.2). Other quantile results from the NHI land use included 34.5 kg ha^{-1}(95% quantile), 13.8 kg ha^{-1}(75% quantile), 6.56 kg ha^{-1}(50% quantile), 2.96 kg ha^{-1}(25% quantile), and 0.738 kg ha^{-1}(5% quantile) (Table 3.2). Unexpectedly, the Bay-wide APLE total P loss loading estimates for the NHI land-use was slightly higher than the APLE estimates for the HWM land-use. Similar rates of soil erosion and surface runoff were assumed for each HWM and NHI land segment, but a lower assumed rate of crop P uptake on the NHI land segments resulted in slightly elevated total P loss estimates (average 21 kg ha^{-1} for the HWM land use compared to an average 20 kg ha^{-1} for the NHI land use).

At the state level, APLE simulations generated the highest median total P losses from the NHI land-use category in Maryland, followed by Pennsylvania, Virginia, New York, Delaware, and West Virginia (Figure 3.2). Again, as was the case with the HWM land-use, the largest contributor to total P loss was sediment P loads. APLE estimated the Bay-wide mean annual sediment P loss of 10.6 ± 0.3 kg ha^{-1} and a median annual loss of 6.29 kg ha^{-1}. Upon evaluation of median APLE sediment P loss estimates from NHI land-use per land segment (i.e. county) for the 14 year simulation period, the following land segments demonstrated the highest predicted sediment P losses (kg ha^{-1}): Calvert, MD (47.0), Prince Georges, MD (44.2), Charles, MD (32.0); Northumberland, PA (22.1), Anne Arundel, MD (21.9), Montour, PA (20.6), Dinwiddie, VA (20.6), Madison, VA (19.1), Baltimore, MD (18.9), and Rockingham, VA (18.6).

APLE estimates of total dissolved P losses from the NHI land-use were, again, very low with a Bay-wide mean annual loss of 0.253 ± 0.006 kg ha^{-1}and a median annual dissolved P loss of 0.154 kg ha^{-1}. The maximum APLE-predicted annual dissolved P loss was 5.41 kg ha^{-1}.

3.4.3APLE results for low-till with manure (LWM) land use

Under the LWM scenario APLE estimated mean annual total P losses of 4.34 kg ha^{-1}, compared to the WSM defined mean target value of 2.40 kg ha^{-1} (Table 3.2). Established WSM target ranges for total P loss from LWM land-uses were 1.65 to 7.48 kg ha^{-1}while APLE estimations of total P loss loads from LWM land-uses were, again, much more variable than WSM target values and ranged from 0.002 to 83.2 kg ha^{-1}(Table 3.2). Other quantile results from the LWM land use included 14.2 kg ha^{-1}(95% quantile), 5.43 kg ha^{-1}(75% quantile), 2.46 kg ha^{-1}(50% quantile), 1.02 kg ha^{-1}(25% quantile), and 0.214 kg ha^{-1}(5% quantile) (Table 3.2).

At the state level, the highest median total P losses estimated by APLE from the LWM land-use were attributed to Maryland followed by Pennsylvania, New York, Virginia, West Virginia, and Delaware (Figure 3.3).

APLE estimated that sediment P loss was again, as with the HWM and NHI land-uses, the largest contributor to total P loads from the LWM land-use, but the mean annual sediment P loss was decreased substantially to a Bay-wide mean of 4.12 ± 0.1 kg ha^{-1}with a median annual loss of 2.29 kg ha^{-1}. Evaluation of APLE total P loss estimates for the LWM land-use as the median loss per land segment over the 14 year simulation period resulting in the following rankings of land segments with the highest predicted losses (kg ha^{-1}): Prince Georges, MD (14.2), Rockingham, VA (10.6), Charles, MD (10.3), Calvert, MD (9.66),Montour, PA (8.36), Augusta, VA (8.18), Shenandoah, VA (7.87), Page, VA (7.61), Dinwiddie, VA (7.45), and Anne Arundel, MD (6.62).

For the LWM land-use, APLE predictions of total dissolved P loss had a Bay-wide mean annual loss of 0.220 ± 0.006 kg ha^{-1}and a median annual loss of 0.126 kg ha^{-1}. The maximum APLE-predicted annual dissolved P loss from the LWM land-use was 4.42 kg ha^{-1}.

3.4.4 APLE results for nutrient management low-till with manure (NLO) land use

APLE simulation of NLO land-use scenarios resulted in an estimated mean annual total P loss of 5.05 as compared to the established WSM mean target value of 2.16 kg ha^{-1}. Estimations of total P loss by APLE were similar between the NLO and LWM land uses (Table 3.2). Quantile results from the NLO land use included 83.4 kg ha^{-1} (maximum), 17.1 kg ha^{-1}(95% quantile), 6.24 kg ha^{-1}(75% quantile), 2.87 kg ha^{-1}(50% quantile), 1.21 kg ha^{-1}(25% quantile), 0.274 kg ha^{-1}(5% quantile), and 0.002 kg ha^{-1} (minimum) (Table 3.2). Again, the mean and median total P loss estimates were slightly higher for the NLO land-use as compared to non-nutrient management land-use comparison, LWM. Evaluation of the results confirm the same reasons, i.e., lower assumed rate of crop P uptake.

The highest median total P losses estimated by APLE were attributed to Maryland followed by Pennsylvania, Virginia, New York, Delaware, and West Virginia (Figure 3.4).

The APLE-estimated mean annual sediment P loss for the NLO land-use was 4.74 ± 0.1 kg ha^{-1} with a median annual loss of 2.65 kg ha^{-1}. Evaluating the results as the median loss per land segment (median of 14 years) results in the following land segments with the highest predicted losses (kg ha^{-1}):Prince Georges, MD (14.2), Madison, VA (13.7), Stafford, VA (11.5), Dinwiddie, VA (10.9), Charles, MD (10.4), Calvert, MD (10.1), Rockingham, VA (9.61), Montour, PA (8.20), Augusta, VA (8.08), and Shenandoah, VA (7.69).

APLE estimates of NLO total dissolved P loss revealed a mean annual dissolved P loss of 0.308 ± 0.010 kg $ha^{-1}$and a median annual dissolved P loss of 0.155 kg ha^{-1}. The maximum predicted annual dissolved P loss was 6.38 kg ha^{-1}.

3.4.5 APLE results for pasture (PAS) land use

The PAS land-use simulation in APLE included an additional category of manure to account for the direct deposition that occurs from grazing animals. This application rate was input as a separate parameter to better reflect the non-homogeneity of pasture manure deposition when compared to the more uniform applications of manure via machine application. This additional category of manure application was included in the APLE calculation of total P loss load for the PAS land-use.

APLE estimates for the PAS scenarios resulted in an estimated Bay-wide mean annual total P loss of 3.77 kg ha^{-1}, as compared to the WSM model target value of 0.925 kg ha^{-1} (Table 3.2). Predictions of total P loss from PAS are similar to both of the low-till land-uses (LWM and NLO*)* where residue or cover are assumed to be present (Table 3.2). Quantile results from the PAS land use included 80.4 kg ha^{-1} (maximum), 13.2 kg ha^{-1}(95% quantile), 4.28 kg ha^{-1}(75% quantile), 2.03 kg ha^{-1}(50% quantile), 0.845 kg ha^{-1}(25% quantile), 0.131 kg ha^{-1}(5% quantile), and 0.002 kg ha^{-1} (minimum) (Table 3.2).

For the PAS land-use, ranking among states for highest median total P losses occurred in a different order than was observed for the other land uses. The highest median total P loss was estimated for West Virginia, followed by Pennsylvania, Virginia, Maryland, New York, and Delaware (Figure 3.5)

The APLE estimates of Bay-wide mean annual sediment P loss were the lowest for PAS among the five agricultural land-uses, with a mean annual sediment P loss of 3.22 ± 0.1 kg ha^{-1} and a median annual sediment P loss of 1.54 kg ha^{-1}. Conversely, APLE estimates of total dissolved P losses were the highest for PAS among the five land-uses with a mean dissolved P loss of 0.549 ± 0.015 kg $ha^{-1}$and a median annual dissolved P loss of 0.274 kg ha^{-1}. The maximum predicted annual dissolved P loss for the PAS land-use was 12.4 kg ha^{-1}.

3.4.6 Discussion of APLE estimates of EOF P losses

When evaluated cumulatively, the results of the APLE simulation followed the expected trend for prediction of P losses to the EOF as related to tillage intensity. With increased soil disturbance associated with tillage practices, higher rates of sediment P loss and subsequently total P loss was predicted (Table 3.2). APLE's capability to segregate and evaluate loads by P source revealed that a markedly larger proportion of the total P load was generated from sediment P for high-tillage land-uses (HWM and NHI) as compared to low-tillage and pasture land-uses (LWM, NLO and PAS). Sediment P loss was the largest contributor to total P loss for all of the five land uses. Specifically, sediment P accounted for 96% of median total P loads from HWM, 93% of median total P loads from LWM, and 76% of median total P loads from PAS. Across all of the Bay land segments, APLE estimated sediment P losses were directly proportional ($R^2 = 0.86$) to the erosion rate provided as an input by the Chesapeake Bay Program. As a reminder, for this research, soil erosion rate per land segment was identical to the erosion rates used in the WSM based on 1982 and 1987 NRI data sets. All APLE simulation estimations were included in the output dataset to demonstrate the range of APLE estimations and to demonstrate the dependence of APLE estimated EOF P loads on the quality of the input data. Consequently, the erosion rate estimates provided by the Chesapeake Bay Program resulted in high APLE predictions of sediment P loss (Table 3.3). For example, APLE predicted a median sediment P loss of 45.6 kg ha^{-1} over the 14-year simulation for one HWM land segment, assuming a median erosion rate for the same land segment and time period of 44 Mg ha^{-1} y^{-1}(19.5 tons $acre^{-1}y^{-1}$). In contrast, in a review of 49 published field studies used to develop and validate the APLE tool, most sediment P losses were estimated at less than 12 kg ha^{-1}annually, and annual erosion rates were generally less than 11 Mg ha^{-1} y^{-1}(5 tons $acre^{-1}y^{-1}$)(Vadas et al., 2009). Likewise, high assumed erosion levels by the Chesapeake Bay Program contradict efforts since the 1960's by the USDA to limit sediment loss on cultivated fields to less than 5 tons $acre^{-1}year^{-1}$according to established soil loss tolerance guidance. Soil loss tolerance is defined as the maximum rate of annual soil erosion that will still permit a high level of crop productivity to be sustained (NRCS, 2006). Despite the high assumed rate of soil erosion by the WSM, the WSM uniformly simulated modest levels of total P loss across the Bay watershed with a maximum mean target value of 2.44 kg P ha^{-1} (HWM) and a maximum overall total P loss target of 7.51 kg P ha^{-1} (HWM) (Table 3.2). Accordingly, estimations of soil erosion rates in the WSM warrant additional review for future WSM improvements.

The distribution of sediment losses assumed by the WSM Bay-wide included 75% of erosion rates less than 18 Mg ha^{-1} y^{-1} (8 tons $acre^{-1}$ y^{-1}) with a mean annual rate of 14 Mg ha^{-1} (6.19 tons $acre^{-1}$) and a median annual rate of 8.5 Mg ha^{-1} (3.78 tons $acre^{-1}$) for the HWM and NHI land uses. For low-till land uses (LWM and NLO), where less sediment disturbance was assumed due to reduced tillage, the mean annual rate of erosion decreased to 8.3 Mg ha^{-1} (3.71 tons $acre^{-1}$) and a median of 4.5 Mg ha^{-1} (1.99 tons $acre^{-1}$). Likewise, for the PAS land-use, the WSM input mean annual rate of erosion was further reduced to 3.7 Mg ha^{-1} (1.64 tons $acre^{-1}$) with a median erosion rate of 1.2 Mg ha^{-1} (0.52 tons $acre^{-1}$). As input rates of erosion decreased, APLE estimates of sediment P losses decreased and the total dissolved P losses became a greater contributor to the predicted total P EOF loss load. However, estimations of total dissolved P loss were markedly small compared to sediment P loss. Specifically, all land segments had an average annual total dissolved P loss of less than 1 kg DP ha^{-1} for all five land uses (Table 3.3). This is in contrast to published field studies reviewed by Vadas et al. (2009) that found measured dissolved P losses often ranged from 5 to 10 kg DP ha^{-1} $year^{-1}$. Reduced APLE estimates of total dissolved P loss may also be attributable to estimates of surface runoff assumed in the WSM. For example, when considered as a percentage of annual precipitation, surface runoff assumed by the WSM was only 7% of annual precipitation for the HWM and NHI land uses and 5% of annual precipitation for the LWM, NLO, and PAS land uses. These modest runoff percentages would not represent medium-high surface runoff events that could readily transport dissolved P to the EOF, and as a consequence the APLE-predicted total dissolved P losses may be underestimated. Additionally, results from APLE represent only surface P losses, as these are commonly the greatest contributing transport mechanisms. However, in regions of the Coastal Plain dominated by artificial drainage systems that provide direct subsurface connectivity to the Bay, it is recognized that APLE may also underestimate dissolved P losses because APLE does not account for subsurface P losses (Kleinman et al., 2007).

3.4.7 Calibration testing of the APLE results in the WSM

Subsequent to APLE estimations of EOF P losses, APLE EOF P loads were incorporated for calibration testing into the WSM as substitute values for the defined WSM EOF P loss targets. Results from running the WSM with incorporation of the original P loss targets and, alternatively, running the WSM with APLE-generated EOF P losses were compared against observed in-stream monitoring data used by the Chesapeake Bay Program for model calibration performance (USEPA, 2010c).

Incorporating APLE results into the WSM for the calibration testing required additional steps because, 1) APLE estimates represented surface total P, while the WSM also required a subsurface total P component, 2) APLE results contain an organic P portion of total P loss, while the WSM estimates organic P loss independent of inorganic P loss, and 3) APLE estimates represented EOF losses while the WSM relied on losses delivered to the EOS at streams with flow greater than 100 $ft^3\ s^{-1}$. Accordingly, to align the APLE EOF estimates with the WSM total P loss estimates, WSM surface total P loads were calculated as the sum of the WSM's targets for inorganic surface P and surface organic P for each land segment and land use. The WSM's surface total P loss was multiplied by the area extent of each land use (hectares) for each land segment and land use for years 1992, 1997, and 2002 to derive the WSM estimate of the total mass of surface P losses. Similarly, the APLE total P loss outputs, averaged for 1992 to 2005 per land segment, were multiplied by the same area extent of each land use to calculate APLE's estimate of total P mass loss for each land use. The sum of all the WSM surface total P mass loss estimates was divided by the sum of all the APLE surface total P mass loss estimates to calculate a single relative scaling factor of 0.221 The scaling factor represented the necessary modifications to the APLE EOF estimates to align to the WSM's EOS estimates, thus all averaged APLE EOF estimates were multiplied by the scaling factor of 0.221. Last, the scaled surface total P estimates from APLE were assumed a fixed ratio of surface to subsurface total P, including the ratio for organic P, to be able to test WSM performance for predicting P losses against observed losses.

Analysis was conducted using three calibration scenarios: 1) the WSM base case which utilized the original defined WSM P loss targets and did not include any APLE estimates of EOF P losses; 2) the WSM with incorporation of APLE EOF P loss estimates, scaled as defined above, substituted in place of the original defined P loss targets, and modified by regional factors to represent the spatial variability of EOS delivery across the Bay watershed ; and 3) the WSM with incorporation of scaled APLE EOF P loss estimates without modification with the regional EOS delivery factors. Cumulative frequency distribution curves of observed in-stream monitoring data across low, normal, and high flow regimes were evaluated based on a Kolmogrov-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 (Boes et al., 1974). The results were evaluated against 210 individual stream monitoring stations for total P, representing 22 sub-basins at the approximately 12 digit HUC scale within the Bay watershed. The median K-S statistic for the 210 individual water quality monitoring stations showed minor change (range of 0.32-0.33) between the observed in-stream monitoring data and the base case scenario and scaled APLE EOF P loss estimates scenarios with and without the modification by regional delivery factors (Figure 3.6). When the K-S statistics were evaluated by sub-basins, the median K-S statistic decreased at 59% of the 22 sub-basins when comparing the WSM base case scenario to the scaled APLE estimates modified by the regional delivery factor, and at 50% of the 22 sub-basins when comparing the WSM base case to scaled APLE P loss estimates without modification by the regional delivery factors.

While results from the calibration testing that incorporated APLE EOF estimates into the full WSM simulation resulted in enhanced model performance as demonstrated by a lower K-S statistic at some sub-basins, it had been expected that APLE, a validated, soil P focused simulation, would produce greater change in the WSM performance. However, one must also consider the differences in purpose and scale between APLE and the WSM when assessing the results of the calibration analysis. For example, the WSM is a process-based hydrology model developed to simulate sediment and nutrient loads at broader geographic scales than APLE, and the WSM uses observed in-stream data collected from water quality monitoring stations located at higher-order streams (flow greater than 100 $ft^3 s^{-1}$) than those likely adjacent to an APLE-simulated land segment. Thus while APLE estimated higher mean EOF P losses than the original WSM targets, the analysis still included other processes of the WSM, e.g. EOF to EOS transport and in-stream transport to higher order streams that may have affected the analyses outcome. Additionally, the APLE-estimated P losses were simulated for only five land uses while the WSM contains twenty-six total land uses. The remaining land uses not simulated by APLE were assumed to have no change in P loss or WSM methodology. Thus, the take away message remains that the WSM was limited to literature-derived targets of total P losses delivered to the EOF while APLE results represent annual EOF estimates for surface transport mechanisms and forms of P loss by land segment. Though changes in the K-S statistic from the calibration analysis were small, results demonstrate improvement to the WSM performance in estimating total P loss loads through substitution of APLE EOF P loss estimates in place of the literature-defined total P loss targets currently in use in the WSM. By utilizing APLE to estimate spatial and temporal variability of EOF total P losses, the WSM methodology is incorporating improved science-based simulations of land segment characteristics.

3.5 Conclusion

The APLE sources of P loss – sediment P, soil P, manure P, and fertilizer P– allowed additional analysis and insight into the reasonableness of results. Results for all simulated land uses exhibited greater variability and a higher mean estimate of total P loss compared to the WSM targets. This outcome demonstrates the broader spatial variability in soil P processes and affirms APLE's ability to simulate EOF losses based on unique land segment conditions as opposed to assumed uniform losses in the WSM. Additionally, APLE outcomes indicated that sediment P loads were significantly higher for all simulated land uses compared to dissolved P losses. High estimates of sediment loss assumed as an input to APLE may warrant future research or review of the WSM's methodology for deriving land segment erosion rates. Specifically, sediment loss rates may not correspond directly to observed losses in the field especially given the conservative timeframe selected for model calibration (1982 and 1987) and may not reflect the intentional better management of cultivated lands for reduced erosion. Second, as expected, reduced rates of erosion and soil incorporation assumed for lower tillage systems or pasture land uses resulted in increased dissolved surface runoff losses relative to sediment erosion losses. Additionally, future research and appropriate simulation of subsurface transport processes are also needed to improve the simulation of Coastal Plain regions more commonly dominated by subsurface pathways.

The performance of APLE in the broader WSM calibration context provides an area of future research into other WSM parameters that contribute to total P delivery to the Bay. While APLE represents an important contribution to validated, science-based soil P simulations, it is limited to providing a single component to the larger WSM suite of routines that would need to be considered holistically to better simulate biogeochemical processes, fate, and transport in the WSM.

Table 3.1: List of user inputs and data sources for APLE simulations

Category	Data Parameter	Units	Data Source
Soil Properties	Depth of soil layers	inches	Variable based on land use
	Mehlich-3 soil P	mg kg^{-1}	University labs *(see Chapter 4)*
	Clay content	percent	Chesapeake Bay Program[1]
	Organic matter content	percent	Chesapeake Bay Program
Transport Mechanisms	Annual rain	inches	Chesapeake Bay Program
	Annual runoff	inches	Chesapeake Bay Program
	Annual erosion rate	tons $acre^{-1}$	Chesapeake Bay Program
Field Properties	Field size *(pasture land use only)*	acres	Chesapeake Bay Program
	Annual crop P uptake	pounds $acre^{-1}$	Chesapeake Bay Program
	Manure application	kg ha^{-1}	Chesapeake Bay Program
	Manure solids	percent	Scholarly literature
	Manure WEP/TP	percent	Scholarly literature
	Manure incorporation	percent and inches	Variable based on land use
	Fertilizer application	kg ha^{-1}	Chesapeake Bay Program
	Fertilizer incorporation	percent and inches	Variable based on land use
	Degree soil mixing	percent	Variable based on land use

1. Chesapeake Bay Program Partnership, USEPA, state and agency partners. www.chesapeakebay.net

Table 3.2: Descriptive statistics for Bay-wide annual total P loss from APLE estimates and the defined WSM annual total P loss targets for five agricultural land-use categories used in the Chesapeake Bay Watershed model (WSM). Land-use categories are: HWM = high till with manure, NHI = nutrient management high-till with manure, LWM=low-till with manure, NLO=nutrient management low-till with manure and PAS=pasture. N = 2644 to 2882 land segments depending on land use.

	Annual Total P loss (kg P ha^{-1})				
Quantile	HWM	NHI	LWM	NLO	PAS
APLE 95% quantile	33.3	34.5	14.2	17.1	13.2
APLE 75% quantile	12.8	13.8	5.43	6.24	4.28
APLE 50% quantile	5.98	6.56	2.46	2.87	2.03
APLE 25% quantile	2.61	2.96	1.02	1.21	0.845
APLE 5% quantile	0.642	0.738	0.214	0.274	0.131
APLE minimum	0.012	0.012	0.002	0.002	0.003
APLE mean	10.1 (0.2)	10.9 (0.3)	4.34 (0.1)	5.05 (0.1)	3.77 (0.1)
APLE maximum	131	131	83.2	83.4	80.4
WSM 95% quantile	3.61	3.25	3.50	3.15	1.74
WSM 75% quantile	2.46	2.22	2.43	2.19	1.02
WSM 50% quantile	2.22	2.00	2.21	1.99	0.788
WSM 25% quantile	2.12	1.91	2.10	1.89	0.673
WSM 5% quantile	1.99	1.79	1.97	1.77	0.551
WSM minimum	1.70	1.53	1.65	1.49	0.00
WSM mean	2.44 (0.05)	2.19 (0.04)	2.40 (0.04)	2.16 (0.04)	0.925 (0.03)
WSM maximum	7.51	6.76	7.48	6.73	5.81

Table 3.3: Descriptive statistics for Bay-wide annual sediment P loss and total dissolved P loss from APLE estimates for five agricultural land-use categories used in the Chesapeake Bay Watershed model (WSM). Land-use categories are: HWM = high till with manure, NHI = nutrient management high-till with manure, LWM=low-till with manure, NLO=nutrient management low-till with manure and PAS=pasture. N = 2644 to 2882 land segments depending on land use.

	Annual Sediment P loss (kg sediment P ha^{-1})				
Quantile	HWM	NHI	LWM	NLO	PAS
APLE 95% quantile	33.0	34.3	13.7	16.3	12.1
APLE 75% quantile	12.6	13.5	5.11	5.91	3.50
APLE 50% quantile	5.77	6.29	2.29	2.65	1.54
APLE 25% quantile	2.53	2.78	0.922	1.07	0.582
APLE 5% quantile	0.560	0.654	0.166	0.218	0.093
APLE minimum	0.011	0.011	0.001	0.989 $^{E-3}$	0.835 $^{E-3}$
APLE mean	9.87	10.6	4.12	4.74	3.22
APLE maximum	131	131	83.1	83.2	79.9
	Annual Total Dissolved P loss (kg DP ha^{-1})				
APLE 95% quantile	0.575	0.807	0.700	1.17	2.03
APLE 75% quantile	0.243	0.307	0.265	0.337	0.667
APLE 50% quantile	0.126	0.154	0.126	0.154	0.274
APLE 25% quantile	0.057	0.069	0.050	0.060	0.099
APLE 5% quantile	0.015	0.018	0.009	0.011	0.017
APLE minimum	0.627 $^{E-3}$	0.598^{E-3}	0.726^{E-3}	0.786^{E-3}	0.998^{E-3}
APLE mean	0.190	0.253	0.220	0.307	0.549
APLE maximum	2.52	5.41	4.42	6.38	12.4

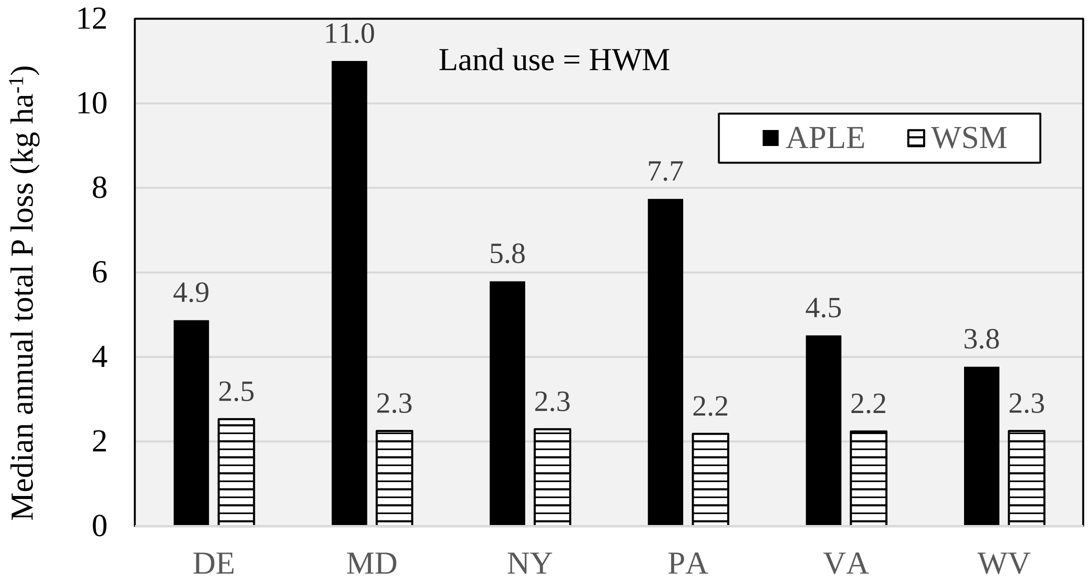

Figure 3.1: Median annual total P loss (kg ha^{-1}) estimates from APLE (black bar) compared to Chesapeake Bay Program Watershed Model (WSM) targets (striped bar) for the high-till with manure (HWM) land use for each state in the Chesapeake Bay watershed.

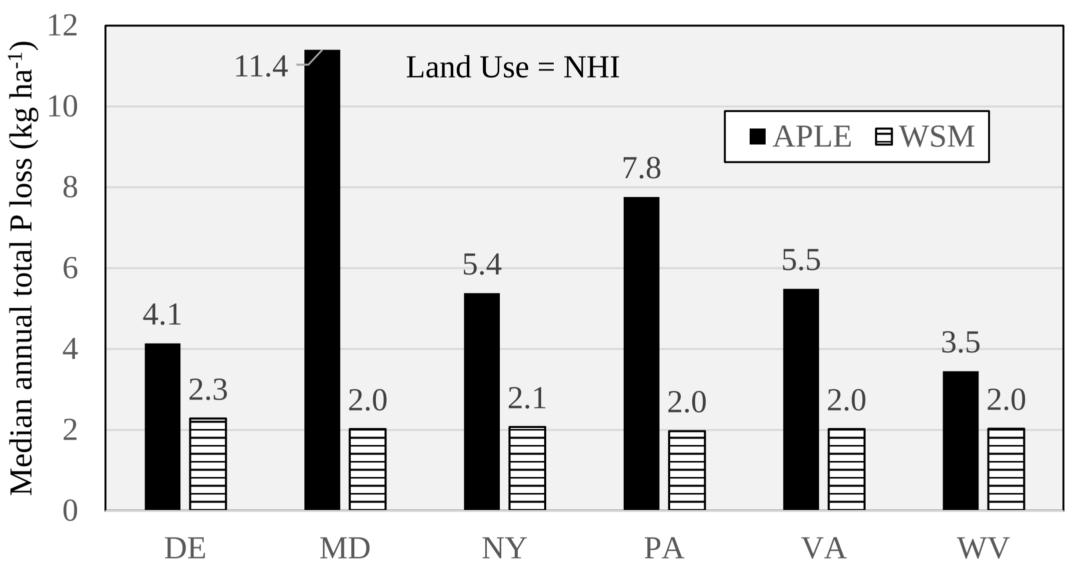

Figure 3.2: Median annual total P loss (kg ha^{-1}) estimates from APLE (black bar) compared to Chesapeake Bay Program Watershed Model (WSM) targets (striped bar) for the nutrient management high-till with manure (NHI) land use for each state in the Chesapeake Bay watershed.

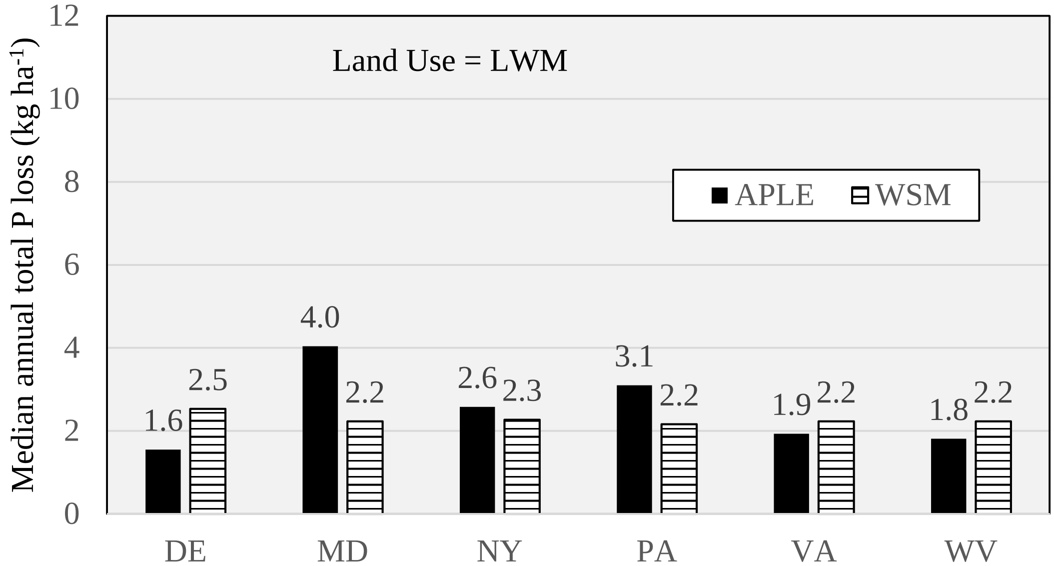

Figure 3.3: Median annual total P loss (kg ha^{-1}) estimates from APLE (black bar) compared to Chesapeake Bay Program Watershed Model (WSM) targets (striped bar) for the low-till with manure (LWM) land use for each state in the Chesapeake Bay watershed.

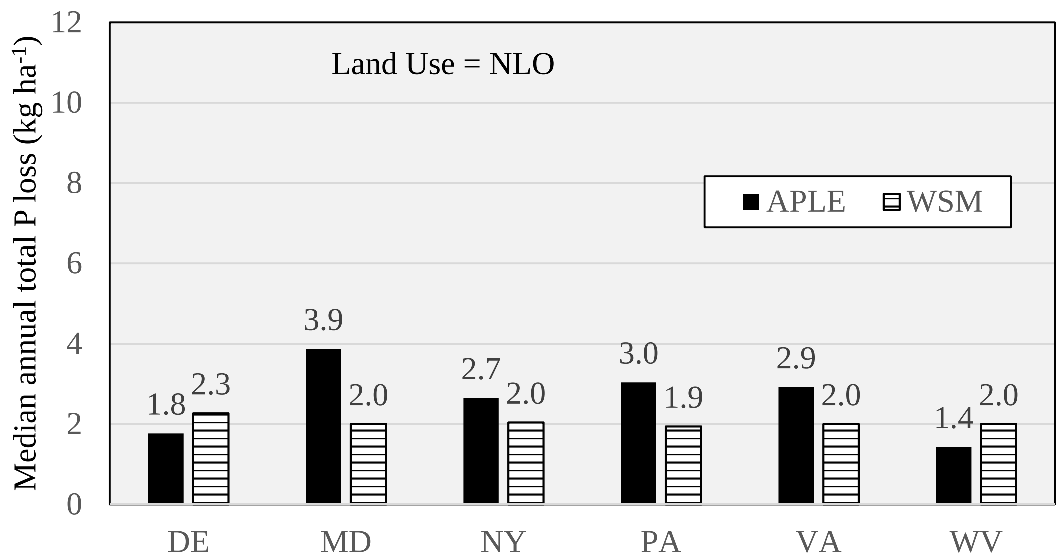

Figure 3.4: Median annual total P loss (kg ha^{-1}) estimates from APLE (black bar) compared to Chesapeake Bay Program Watershed Model (WSM) targets (striped bar) for the nutrient management low-till with manure (NLO) land use for each state in the Chesapeake Bay watershed.

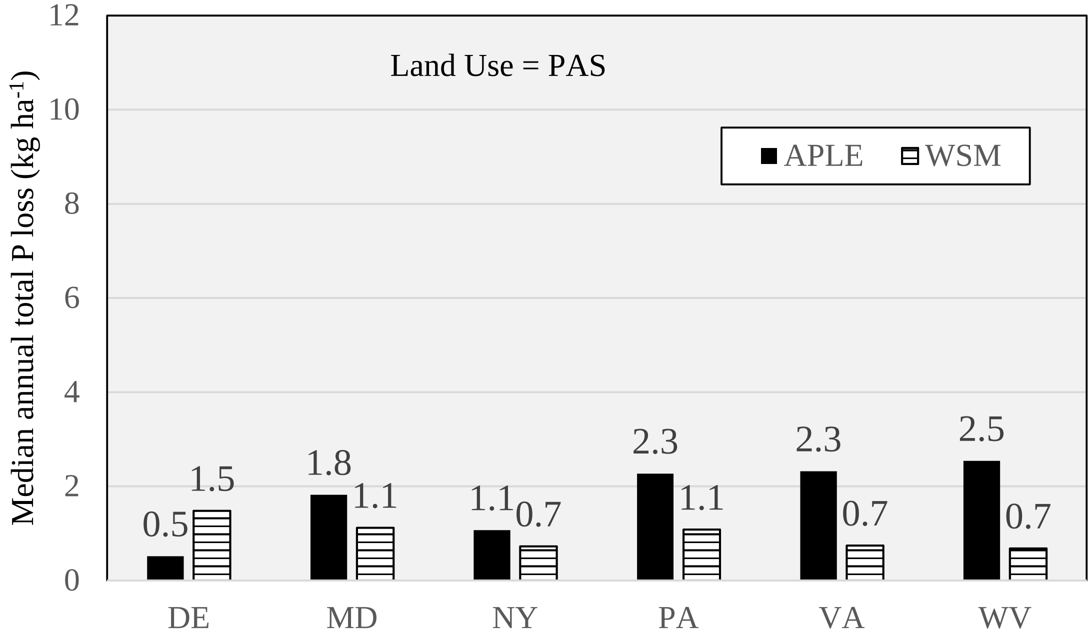

Figure 3.5: Median annual total P loss (kg ha^{-1}) estimates from APLE (black bar) compared to Chesapeake Bay Program Watershed Model (WSM) targets (striped bar) for the pasture (PAS) land use for each state in the Chesapeake Bay watershed.

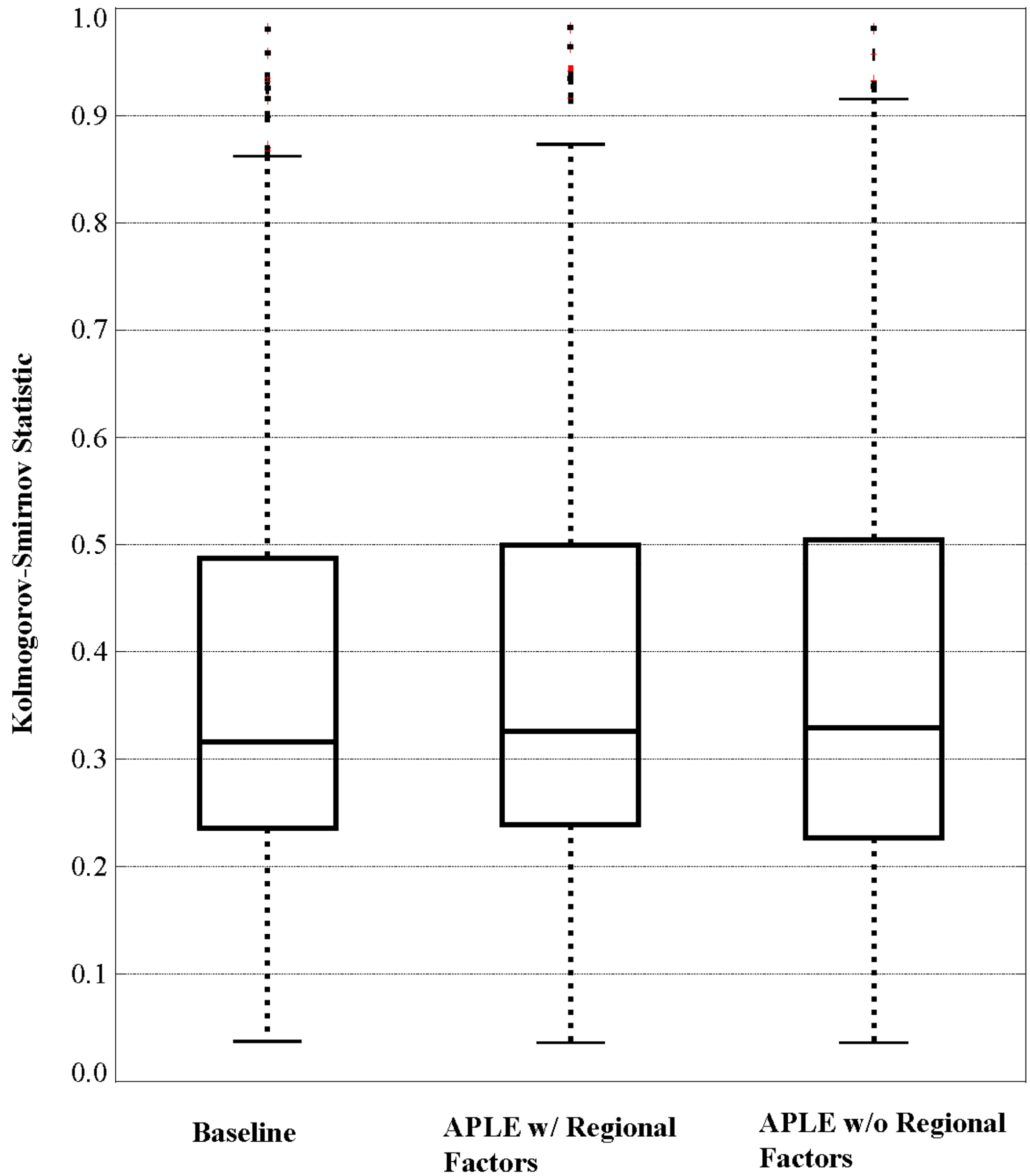

Figure 3.6: Kolmogrov-Smirnov (K-S) statistic from cumulative frequency distribution curves (CFDC) of APLE estimated total P loads from the Chesapeake Bay Program Watershed Model (WSM) compared to observed data from 210 water quality monitoring stations within the Chesapeake Bay watershed. A K-S statistic of zero indicates a perfect fit between the estimated and observed data's CFDC. Sensitivity scenarios included a base case (Baseline), scaled APLE results with regional factors (APLE w/ Regional Factors), and scaled APLE results without regional factors (APLE w/o Regional Factors).

Chapter 4: Tillage and Nutrient Application Effects on Soil Phosphorus Concentrations

4.1 Abstract

The Chesapeake Bay Watershed Model (WSM) estimates loads of nutrients and sediment being delivered from the landscape to receiving waters. However, the model routine within the WSM that simulates soil P dynamics has not been updated to reflect current research findings on factors contributing to soil P cycling. Our objective was to improve the simulation of soil P cycling by using an independent modeling tool (APLE) 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. Conditions were simulated from1992-2005 to estimate changes in soil labile P concentrations over time. APLE-estimated results revealed a highly significant ($p < 0.0001$) relationship between each category of applied manure P and soil labile P concentrations for all land uses simulated. Additionally, changes to soil labile P concentrations were most strongly affected by the degree of soil mixing between the simulated soil layers whereby reduced soil mixing in low-till and pasture land uses resulted in higher percent changes to labile P concentrations of the surface soil layer as compared to high-till land uses.

4.2 Introduction

Because the dynamics of soil P include complex biogeochemical processes, for practical purposes of modeling soil P, it is often described as different soil P pools to designate the form, availability, and transport characteristics. Phosphates dissolved in soil solution and the P compounds that can readily desorb or mineralize into soil solution for biological availability are known as the labile P pool (Stevenson and Cole, 1999).However, soil P is often considered mostly insoluble such that soluble P added to the soil will readily adsorb to or precipitate with aluminum, iron, or calcium cations depending on soil mineralogy and soil pH (Brady and Weil, 2002). Increased soil solution ionic strength decreases the rate of desorption from solid soil surfaces into the soil solution, so the amount of dissolved P in solution is relatively small and constant. To replenish the labile pool, a dynamic equilibrium with active and stable P pools occur, where the active pool is considered to be in rapid equilibrium with the labile pool while the stable pool is very slowly acting (Jones et al., 1984). The stable P pool is considered the less soluble and physically occluded portions that contain both organic and inorganic compounds strongly resistant to desorption and mineralization. While organic P often accounts for the largest percentage of the total soil P, it is highly complex and greatly influenced by organic inputs to the soil and soil microorganism activity (Stevenson and Cole, 1999). The equilibrium between the labile P pool and the organic P pool is controlled by immobilization and mineralization processes that are dependent on soil temperature, moisture, and ratios of carbon, N, and P in the soil. The cycling between the soil P pools occurs to achieve equilibrium following addition of P to the system, but typically favors P adsorption and immobilization (Sharpley, 2000).

When nutrient applications are made to a field, the physical and chemical properties of the nutrient source will affect the overall soil P partitioning response. In most cases, inorganic mineral fertilizer P is added at agronomic rates to meet a crop’s nutrient needs. Depending on management strategy, P applications are intended to address either immediate crop needs, crop needs for the duration of the growing season, or crop needs for multiple growing seasons. Therefore, P applications may be made more than once during a growing season, once per season, or a single application may be made to satisfy P needs for multiple growing seasons. While mineral fertilizers come in different forms, all are largely considered soluble and thus contribute to the labile P pool. In turn, fertilizer P additions disrupt the equilibrium between the P in the soil solution and the P on the solid phases (active, stable, and organic pools), triggering the adsorption/desorption and mineralization/immobilization processes that maintain P equilibrium. Organic nutrients, such as manures and biosolids (municipal wastewater treatment residuals), are also applied to fields to meet crop nutrient needs. However, unlike fertilizer that can be custom blended to specifically meet the crop needs, the nutrient elements within organic sources cannot be easily separated by physical or chemical means. When organic nutrient application rates are based on the N-supplying capability of the applied nutrient source, then, frequently, a higher proportion of P is added to the soil compared to N, relative to crop need. With repeated organic nutrient application, the relative excess applied P may accumulate creating P-enriched soils. When P applications exceed the soil's P retention capacity, labile P concentrations will become elevated. This scenario has been documented in portions of the Bay watershed where the growth in animal production operations has created localized excesses of manure, increasing the likelihood for soil P over-enrichment (Sharpley, 2000; Butler and Coale, 2005).

For purposes of the Chesapeake Bay Watershed Model (WSM) simulation, P inputs were estimated through Scenario Builder (USEPA, 2012). Scenario Builder is an annual planning tool that allows alternative land management scenarios to be designed. For example, Scenario Builder input parameters can be adjusted to represent land use acreage, BMPs and related load reduction efficiencies, estimated crop yield, and nutrient applications. The outcome simulates the effects of management changes on Bay water quality. Initial assumptions to Scenario Builder for crop acreage and manure volume are based on regular USDA Census of Agriculture reports which is joined with recommendations for nutrient application rates and crop yield goals provided by respective states (USEPA, 2012). Scenario Builder uses a mass-balance approach to apply appropriate nutrient rates for meeting crop needs, with the exception that manure directly excreted on pasture is not applied towards the calculation of satisfying crop nutrient needs. The mass-balance approach implies that all manure generated in the watershed annually will be land-applied annually, less losses for N volatilization or losses incurred during storage and handling. Of important note is that modeled P application rates can exceed the agronomic recommended application rate because most nutrient planning scenarios are N-based (USEPA, 2012).

The output of Scenario Builder represents a menu of nutrient inputs and crop nutrient uptake based on a user-designed set of field conditions. This menu is then simulated within the WSM to represent nutrient cycling and availability between soil P pools.

The translation of soil P dynamics to representative pools for model simulations within the WSM was largely built on the work of Jones et al. (1984) and Sharpley et al. (1984). The WSM includes four model-defined soil layers - surface, upper, lower, and groundwater zones – occurring in the soil profile and each defined layer is assumed to be uniformly mixed and at instant equilibrium upon P application to the soil system (Donigian et al., 1994). Within the WSM, exchanges of P among the defined soil pools are controlled by manually imposed parameters that are adjusted in order to maintain a constant concentration of labile P. Consequently, the WSM cannot simulate the effects of tillage or nutrient application on changing soil P concentrations. Additionally, the WSM assumes that nutrient inputs to the soil system react similarly among the soil pools, and thus the WSM makes no distinction between the P forms and P solubility of inorganic versus organic nutrient sources.

Given the limitations of the current WSM methodology, the primary objective of our study was to evaluate alternative methods for simulating soil P as it relates to P portioning and cycling among the soil P pools and changes in soil P concentrations over time based on nutrient inputs and management. Use of an independent P loss assessment tool as an alternative to the current WSM soil P simulation may be an improved modeling methodology for soil P nutrient cycling in the Chesapeake Bay watershed.

4.3 Materials and Methods

4.3.1 Overview

Agricultural practices simulated in the WSM are largely defined at the county scale due to the availability of data from the USDA Census of Agriculture. Accordingly, simulations were kept at a consistent county-based scale utilizing the same land segments in this study. Data inputs of field and management conditions were also considered representative of a county segment based on source data (Table 4.1). To generate alternative estimates of P losses from agricultural production fields within the Chesapeake Bay watershed, the Annual Phosphorus Loss Estimator (APLE), developed by the USDA-ARS (Vadas, 2013), was used to simulate edge-of-field (EOF) P loss loads for the 14-year period of 1992-2005. APLE was designed to be an agricultural field-scale simulation of EOF losses from surface sources - sediment, soil, manure, and fertilizer – based on easily available user input data and transport factors (Vadas, 2013). The APLE model ran on an annual time step. Five Chesapeake Bay watershed land-use categories currently evaluated in the WSM were simulated with APLE: 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).

4.3.2 Soil test P concentrations

To initialize the soil P pools simulated in APLE, it was necessary to input four soil properties – soil layer depth, soil P concentration, soil organic matter content, and soil clay content (Vadas, 2013). Soil layer depth was assumed to be a single soil layer with a depth of 178 mm (7 inches) for HWM and NHI, and two soil layers with depths of 0 to 25 mm (1 inch) and 25 to 178 mm (1 to 6 inches), for LWM, NLO and PAS land uses to represent soil mixing by typical tillage practices. Additionally, the shallow soil layers assumed for LWM, NLO and PAS land uses allowed for appropriate simulation of P stratification in surface layers as a result of reduced or non-existent tillage. Assumed soil organic matter and soil clay contents were provided by the Chesapeake Bay Program at a county-scale segment from the Natural Resources Conservation Service's SSURGO database (USEPA, 2010c). Soil P concentration was not previously used as an input to the WSM, but was a required input for APLE simulations. Additionally, given the expanse and diversity of the Bay watershed's agricultural landscape, acquiring representative soil test P records was a challenge, but, ultimately, university soil testing laboratory historical records were acquired and used. APLE required an input of soil test P concentrations based on Mehlich-3 P soil-test extractions (mg P kg^{-1}) for year 1 (1992) of the 14-year simulation period. However, during that timeframe, most university soil testing laboratories in the Bay watershed were using a different soil-test P extraction methods. The diverse soil-test P output datasets were converted to a single unitless Fertility Index Value (FIV) scale (Coale, 2001). In turn, the FIV scale was subdivided into qualitative interpretive categories reflecting the soil P fertility status and the associated nutrient application recommendations. Historical soil test P records were requested from each Bay watershed university soil testing laboratory at the county scale for year 1 (1992). The availability of historical soil test P data varied among the Bay states and in some cases, records of county level soil test P quantitative data were unavailable. In lieu of quantitative laboratory output, the data from some states 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). When FIV could not be directly calculated, it was assumed that all soil analyses that were grouped within the "low" interpretive category had FIV=25;

soils with “medium” P fertility status had an assigned FIV=38; soils categorized as “high” had FIV=76; and soils with P fertility status of “very high” or “excessive” were assumed to have FIV=200.

$$FIV \ = (1.11 * Mehlich - 3\ P) + 7 \quad [Eq. 4.1]$$

Using these assumptions for FIV, Eq. 4.1 (Coale, 2001) was used to solve for the soil P concentration, as Mehlich-3 P (mg kg^{-1}). A weighted average soil P concentration, Mehlich-3 P equivalent, was calculated for each county in the Bay watershed. Exceptions to the method outlined above include use of different assumed FIVs for soil analyses from the state of Virginia due to state-unique qualitative soil fertility interpretive categories, and weighted average soil-test P data from West Virginia was proportioned according to crop acreage per county. A complete accounting of county-average soil-test P concentration assumptions are presented in Appendix D.

4.3.3 Simulation of soil P pools

The APLE model simulates three inorganic soil P pools – labile, active, and stable – and one organic P pool (Vadas, 2013). Labile P concentrations were initialized in year 1 (1992) based on Mehlich-3 P soil-test equivalent concentrations with the assumption that the APLE initial labile pool concentrations were half of the Mehlich-3 P concentration.

Active P is initialized from the labile P concentration by Eq. 4.2,

$$Active\ P = \frac{(Labile\ P)(1 - PSC)}{PSC} \quad [Eq. 4.2]$$

where PSC is the P sorption coefficient that represents the portion of inorganic P added to soil that remains labile upon reaching equilibrium, defined by Eq. 4.3.

$$PSC = -0.053(\ln(\%\ clay)) + 0.001(Labile\ P) - 0.029(\%\ organic\ C) + 0.42 \quad [Eq. 4.3]$$

The soil clay and organic matter content were the same used by the Chesapeake Bay Program in order to be consistent with the WSM (Table 4.1). The soil organic C content was assumed to be 58% of the soil organic matter percent. The possible range for PSC was 0.05 to 0.90. Stable P is initialized by Eq. 4.4:

$$Stable\ P = 4 * Active\ P \quad [Eq. 4.4]$$

Lastly, the organic P pool is initialized by

$$Organic\ P = \frac{\frac{\%\ organic\ C * 10000}{8}}{14} * 1.2 * depth * 0.25 \quad [Eq. 4.5]$$

based on the ratio of C:N of 14:1 and N:P of 8:1. These ratios are maintained throughout APLE as organic additions occur via mineralization or manure application.

The sum of the four pools – labile P, active P, stable P, and organic P - is Soil Total P.

When an addition of fertilizer, manure, or biosolids is made to the soil, APLE distributes the applied P based on the tillage, incorporation, and soil depth defined by the user (Vadas, 2013). Fertilizer P was distributed as 100% to the inorganic P pools while organic nutrient sources were assumed to be distributed as 5% to the organic P pool and 95% to the inorganic pools. Additions to the inorganic P pools are initially added to the Labile P pool triggering a change in the equilibrium status between the pools and requiring fractions of P moved between the stable, active, and labile pools based on the equations:

$$Fraction\ added\ to\ Active\ P\ pool = 0.1\ per\ day \quad [Eq. 4.6]$$

$$Fraction\ added\ to\ Stable\ P\ pool = (-0.187 * PSC) + 0.189 \quad [Eq. 4.7]$$

$$Fraction\ removed\ from\ Labile\ P\ pool = 0.41 * PSC^2 + 0.54 * PSC + 0.005 \quad [Eq. 4.8].$$

Soil mixing between the two layers was defined by the user based on tillage or natural mixing practices. Mixing occurs to reduce soil P in one layer and increase soil P in the second layer proportional to the degree of mixing.

4.4 Results and Discussion

4.4.1 Nutrient application rates and land uses

The mean Bay-wide nutrient application rate reported by the Chesapeake Bay Program for solid manures was 6.10 kg TP ha^{-1} yr^{-1} for HWM and LWM land uses, and an average reduced application rate of 4.46 kg TP ha^{-1} $yr^{-1}$for nutrient management land uses NHI and NLO. For the APLE simulations P application rates were kept the same as they were established for the WSM for the two manure-receiving land uses, HWM and LWM and the two nutrient management land uses, NHI and NLO to compare effects of tillage practices on changes to soil P concentrations. The Chesapeake Bay Program reported amean application rate of 3.66 kg TP ha^{-1} $yr^{-1}$for the PAS land use. Lower mean P application rates were assumed for liquid manures applied to HWM and LWM land uses (3.20 kg TP ha^{-1} yr^{-1}) and NHI and NLO land uses (2.69 kg TP ha^{-1} yr^{-1}). The P application rate for PAS land uses increased significantly for liquid manure sources to a mean of 19.4 kg TP ha^{-1} yr^{-1}. In addition, APLE assumed a mean application of 18.4 kg TP ha^{-1} yr^{-1} for the PAS land use due to direct excretion from grazing animals. Rates of manure P application to the PAS land use were generally higher than other land uses due to the WSM's assumptions that 1) directly excreted manure did not contribute to crop nutrient need, and 2) land segments with excess annual manure often used the PAS land use for disposal application (USEPA, 2012).

A second source of organic nutrients, biosolids, was also assumed for application in some counties of Virginia during the simulation period. Results included a minimal average nutrient application for HWM, LWM, and PAS(less than 0.5 kg TP ha^{-1} yr^{-1}) but a significantly increased mean rate of application on the nutrient management lands uses NHI and NLO (15.5 kg TP ha^{-1} yr^{-1}).

Inorganic fertilizer was also simulated for annual application with a mean of 16.1 kg TP ha^{-1} yr^{-1}, for the HWM and LWM land uses and 14.1 kg TP ha^{-1} yr^{-1} for the NHI and NLO land uses. The mean application rate of fertilizer P to PAS land uses was 9.46 kg TP ha^{-1} yr^{-1}. When considered as an annual average at the state level, P application rates were more consistent with the manure sources commonly associated with a state. For example, average annual applications of solid manure were greatest in Delaware, Maryland, and followed by West Virginia, which are states commonly associated with dry litter poultry production operations. Conversely, New York, Pennsylvania, and Virginia are more commonly associated with liquid manure from dairy and swine operations which was reflected by higher mean annual application rates of liquid manure. Little variability in fertilizer application rates occurred among states.

APLE analyses were conducted to assess the effects of nutrient application on average soil P concentrations both annually and over the 14-year simulation period. Correlation analysis (SAS 9.3, PROC CORR) of annual changes in labile P concentrations indicated a highly significant relationship ($p<0.0001$) between each category of applied manure P and soil labile P concentrations for all land uses simulated (Table 4.2). The same results were found for fertilizer P applications with the exception of the PAS land use, likely due to the low rate of fertilizer P application attributed to the PAS land use. However, the direction of the correlation did not remain consistent across land uses. Specifically, solid manure sources were significantly and positively correlated ($r = 0.20$ to 0.79) with soil P concentrations for the HWM, LWM, and PAS land uses (Table 4.2). Conversely, liquid manure sources were significantly but weakly correlated ($r = -0.14$) for the HWM land use, but a positive correlation was exhibited between liquid manure application rate and soil labile P concentration for the LWM ($r = 0.08$) and PAS ($r = 0.87$) land uses (Table 4.2). The observed correlation relationships were attributed to the higher rate of application for solid manures as compared to the liquid manures across all land uses with the exception of the PAS land use.

Fertilizer applications also showed a significant but negative correlation ($r = -0.39$) for the HWM land use but a positive correlation for the LWM land use ($r = 0.32$). Results for the LWM and PAS land uses relate to the soil labile P concentration in the shallow surface soil layer and reflect the decreased rate of mixing between the surface layer and subsurface soil layer.

Another variable, crop P uptake, was found to be highly significant ($p<0.0001$) and negatively correlated with soil labile P concentrations for the HWM and LWM land uses (Table 4.2). The correlation was weak at $r = -0.22$, but was the expected inverse trend to soil P concentrations, i.e. as uptake increased, soil labile P concentrations decreased. No correlation between crop uptake and soil labile P concentration for the PAS land use was found (Table 4.2).This outcome is attributed to differences between the uptakes assumed for each land use. For example, the HWM and LWM land uses assumed commodity row crops with a higher level of annual crop P uptake (average 21 kg $ha^{-1}yr^{-1}$) compared to forage crop P uptake on the PAS land use(average 17 kg $ha^{-1}yr^{-1}$). Lastly, an unexpectedly significant but weak correlation ($p<0.0001$, $r = 0.15$) was observed between soil labile P concentration and soil clay content for the PAS land use (Table 4.2). Presumably, the absence of P mixing between soil layers combined with the high rates of manure P application resulted in shallow surface soil layers accumulating labile P relative to the surface soil clay content.

4.4.2 Tillage effects and land uses

Changes in soil labile P concentrations were further analyzed as a percent change in soil labile P concentration per county segment over the 14-year simulation period for each land use to investigate differences in soil labile P accumulation resulting from the assumed tillage practices (Table 4.3). For each land use simulated in APLE the parameters associated with tillage practices are defined, including the degree of soil mixing between layers, the percent incorporation into the soil, and the depth of incorporation that are associated with estimates of P losses from surface runoff. The degree of soil mixing between soil layers is a key variable to simulate distinctions between high-tillage, low-tillage or no-tillage practices (Table 4.4). Reduced or no-tillage practices result in P stratification and P accumulation in the surface soil layer (Abdi et al., 2014).

The APLE simulations indicated that the percent change in surface layer soil labile P concentrations increased dramatically as soil mixing decreased, over the range of mean manure and fertilizer P applications (2.69 to 19.4 kg TP ha^{-1} yr^{-1}) among the land uses. Specifically, a high rate of mixing was assumed for the HWM and NHI land uses with the result being a strong draw down in soil labile P concentrations (-9.64% mean reduction) over time. Comparatively, when soil mixing was decreased, as with the LWM land use, APLE simulations resulted in significant labile P accumulation in the shallow surface layer and only a modest 3.69% mean change in subsurface soil labile P concentration (Table 4.3). The median percent change in LWM subsurface soil labile P concentrations exhibited a drawdown over time, as the median subsurface soil labile P concentration decreased 7.95% (Table 4.3). APLE simulations of the PAS land use over the 14-year period resulted in substantial P accumulation in the subsurface soil layer with a mean percent change of 95.8% and a median subsurface soil P concentration increase of 60% (Table 4.3).

Significant increases in the soil surface layer's labile P concentration for LWM and PAS land uses are attributed to two factors: 1) nutrient application rates relative to crop P uptake in the WSM, and 2) simulated mixing between the soil layers. To the first factor, in APLE, applications of manure and fertilizer are handled by a mass balance approach such that when soil P is not removed by crop uptake, lost to dissolved P surface runoff, or eroded as sediment-bound P, the nutrient source P applications will be initially distributed into the soil labile P pool of the surface layer and trigger the sequence of relationships expressed in Eq. 4.6, Eq. 4.7, and Eq. 4.8. Under the APLE mass balance approach crop P uptake is often the largest annual mechanism for labile P removal from the soil. Accordingly, a steady increase in labile P concentration was observed when simulating the average nutrient applications assumed for each land use by the WSM (Figures 4.1 and 4.2) relative to the crop P uptake (average 21 kg $ha^{-1}yr^{-1}$ for LWM and 17 kg $ha^{-1}yr^{-1}$ for PAS). This high rate of presumed nutrient application relative to crop P uptake is coupled with the second factor of mixing between soil layers. APLE estimates of changes to soil labile P concentrations in the LWM, and PAS land uses, in which two distinct soil layers were simulated, compared to the single, highly-mixed soil layer simulated for the HWM land use, emphasized the increased likelihood of P-enriched surface soils (Table 4.3)

It should be clarified for the LWM and PAS land uses, soil labile P concentrations are based on the assumption that the initial soil-test P value derived for each land segment in year 1 of the simulation period (Appendix D) was assumed constant for both soil layers and, thus, assumed that P stratification did not exist in P-enriched soils in year 1. Rather, APLE simulations predicted how P accumulation would occur over the 14-year simulation timeframe.

4.4 3 State soil P trends and effects of intensive animal feeding operations

Per the WSM methodology, an annual mass balance approach to nutrient P application was assumed for all manure generated within a land segment with the exception of losses for volatilization or losses incurred during storage and handling, and in some limited instances manure transported to adjacent counties (USEPA, 2012). Additionally, manure application rates were assumed to be N-based nutrient management planning rates, such that modeled P application rates could exceed the agronomic recommended P application rate. The combination of these established WSM assumptions were most consequential in land segments with large volumes of organic nutrient sources, because the WSM methodology did not account for residual soil labile P remaining in the soil solution being available for crop uptake (USEPA, 2012). This shortcoming of the WSM protocol which allowed for repeated P applications beyond the agronomic recommendation rate, resulted in the attribution of higher manure application rates and greater accumulation of P-enriched soils in land segments containing intensive animal production operations. Figures 4.1 and 4.2 and Tables 4.5 through 4.10 summarize the results at the state scale to demonstrate the model outcome effect of higher nutrient application rates.

4.4 4 Sensitivity of soil P initial concentrations

Due to the degree of uncertainty concerning the soil test P values used for initializing year-1 of the 14 year APLE simulation, additional sensitivity testing was conducted. As reported, initial soil test P values for the early 1990's was obtained from historical county-level records from university soil testing laboratories from each Bay watershed state were converted to a single scale approximating Mehlich 3 soil test P.

To evaluate the sensitivity of APLE predictions of EOF P losses and changes to soil labile P concentrations, the soil test P value used to initialize the 14-year simulation was adjusted to +/- 10% of the original value (Appendix D) and APLE scenarios were -simulated again, keeping all other variables unchanged. No significant changes in predicted EOF P losses for either total P or total dissolved P were predicted when APLE employed initial soil test P values of +/- 10% of the original value. Additionally, no significant changes in APLE predictions of soil labile P concentrations over the 14-year time course and no significant changes in the correlations described between soil labile P concentrations and manure and tillage variables previously discussed resulted from +/- 10% adjustment of the initializing soil-test P values.

4.5 Conclusion

The objective of our study was to simulate the WSM's assumptions of nutrient application, transport, soil properties, and field practices with the alternative APLE model to evaluate changes in soil labile P concentrations. Nutrient applications were simulated in APLE by initially distributing soil P additions to the labile P pool in the surface soil layer, creating a disruption to the equilibrium between the four soil pools and triggering the response relationships to re-establish dynamic equilibrium. When these relationships were applied to the WSM assumptions, changes in the soil labile P concentrations trended primarily with the degree of mixing between soil layers and were dependent on the rate of nutrient P application and rate of crop P uptake. Rates of nutrient P application were imposed according to the WSM's assumptions, whereby many land segments received annual P applications of all nutrient sources: manure (mean range of 2.69 to 19.4 kg TP ha^{-1} yr^{-1}), directly excreted manure (18.4 kg TP ha^{-1} $yr^{-1,}$ for PAS only), and fertilizer (9.46 to 16.1 kg TP ha^{-1} yr^{-1}). Accordingly, correlation analysis (SAS 9.3, PROC CORR) revealed highly significant relationships ($p<0.0001$) between all manure categories and soil labile P concentrations for all land uses. The same finding was true for fertilizer P applications with the exception of the PAS land use, likely due to the low rate of fertilizer application required on the PAS land use due to the direct excretion. When considered at the state-scale, those states wither higher annual rates of nutrient application also coincided with the higher mean annual soil labile P concentrations.

The second finding of the APLE simulation was confirmation of the association of soil labile P concentrations with the degree of mixing between soil layers whereby reduced tillage resulted in increasingly stratified P-enriched surface soil layers for the LWM and PAS land uses, an average 61% and 490% increase in the surface soil layer labile P concentrations, respectively, over the simulation period, while an average 10% drawdown in soil labile P concentration was observed for the highly mixed HWM land use.

The APLE simulations suggest that for land segments and land uses where high assumed rates of annual total P application, often beyond crop P uptake, coincides with reduced tillage practices, soil labile P concentrations in the surface soil layer will increase dramatically, and consequently will contribute to increased soil dissolved P losses when hydrologic surface transport mechanisms exist.

Table 4.1: List of user inputs and data sources to APLE simulations

Category	Data Parameter	Units	Data Source
Soil Properties	Depth of soil layers	inches	Variable based on land use
	Mehlich-3 soil P	mg kg^{-1}	University labs
	Clay content	percent	Chesapeake Bay Program[1]
	Organic matter content	percent	Chesapeake Bay Program
Transport Factors	Annual rain	inches	Chesapeake Bay Program
	Annual runoff	inches	Chesapeake Bay Program
	Annual erosion rate	tons $acre^{-1}$	Chesapeake Bay Program
Field Properties	Field size *(pasture land use only)*	acres	Chesapeake Bay Program
	Annual crop P uptake	pounds $acre^{-1}$	Chesapeake Bay Program
	Manure application	kg ha^{-1}	Chesapeake Bay Program
	Manure solids	percent	Scholarly literature
	Manure WEP/TP	percent	Scholarly literature
	Manure incorporation	percent and inches	Variable based on land use
	Fertilizer application	kg ha^{-1}	Chesapeake Bay Program
	Fertilizer incorporation	percent and inches	Variable based on land use
	Degree soil mixing	percent	Variable based on land use

1. Chesapeake Bay Program Partnership, USEPA, state and agency partners. www.chesapeakebay.net

Table 4.2: Summary of correlation statistic (r) and significance (p) results for three simulated land use categories showing the relationship between soil P labile concentrations relative to independent variables in the APLE simulation (N=2881). Statistically significant relationships ($p < 0.05$) are bolded.

	Soil P labile concentration (mg kg^{-1}) HWM	Soil P labile concentration (mg kg^{-1}) LWM	Soil P labile concentration (mg kg^{-1}) PAS
Clay Percentage	r = 0.016 p = 0.390	r = -0.006 p = 0.73	r = 0.155 **p < 0.0001**
Degree of Soil Mixing	NA	r = -0.038 **p =0.04**	r = 0.018 p = 0.317
Total P application (kg ha^{-1} yr^{-1}) solid manures	r = 0.62 **p < 0.0001**	r = 0.794 **p < 0.0001**	r = 0.205 **p < 0.0001**
Total P application (kg ha^{-1} yr^{-1}) liquid manures	r = -0.135 **p < 0.0001**	r = 0.078 **p < 0.0001**	r = 0.873 **p < 0.0001**
Total P application (kg ha^{-1} yr^{-1}) directly excreted manures	NA	NA	r = 0.875 **p < 0.0001**
Total P application (kg ha^{-1} yr^{-1}) inorganic fertilizers	r = -0.387 **p <0.0001**	r = 0.317 **p < 0.0001**	r = 0.024 p =0.198
Crop P uptake	r = -0.224 **p < 0.0001**	r = -0.229 **p = 0.0008**	r = 0.017 p = 0.369

Table 4.3: Percent change in APLE predicted soil P labile concentrations in the soil layer(s) from year 1 to end of the 14-year simulation period (1992-2005) by land use category.

Land Use	HWM	LWM		PAS	
	Full soil layer	Surface layer	Subsurface layer	Surface layer	Subsurface layer
Mean Change (%)	-9.64	61.4	3.69	490	95.8
Median Change (%)	-20.0	38.7	-7.95	468	60.0
Range (%)	-56.6 to 322	34.9 to 577	-42.7 to 467	-16.7 to 5656	-58.8 to 1954
N	207	207	207	207	207

Table 4.4: Assumptions included in APLE to simulate tillage practices (if any) for the five land use categories. For parameters with a range specified, each annual occurrence was randomized within the given range.

Land Use	HWM and NHI	LWM and NLO	PAS
Degree soil mixing (%)	70-80	30-40	10-15
Incorporation (%)	70-80	30-40	0
Depth incorporation (mm.)	178	102	13

Table 4.5: Summary of APLE-estimated soil labile P concentrations for three land uses in the state of Delaware over the 14-year simulation period (N = 42).

	HWM	LWM		PAS	
	Full soil layer	Surface layer	Subsurface layer	Surface layer	Subsurface layer
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Mean	70.1	89.1	70.8	218	107
Standard Deviation	33.8	36.4	32.8	56.7	51.9
Median	66.9	92.5	69.1	220	100
Maximum	129	142	130	325	192
Minimum	28.6	39.4	32.1	69.2	34.6

Table 4.6: Summary of APLE-estimated soil labile P concentrations for three land uses in the state of Maryland over the 14-year simulation period (N = 335).

	HWM	LWM		PAS	
	Full soil layer	Surface layer	Subsurface layer	Surface layer	Subsurface layer
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Mean	54.8	73.9	56.3	201	76.1
Standard Deviation	25.8	41.3	24.6	91.2	40.0
Median	47.4	62.1	49.7	198	68.3
Maximum	189	310	187	636	303
Minimum	17.3	26.8	20.0	39.3	21.5

Table 4.7 Summary of APLE-estimated soil labile P concentrations for three land uses in the state of New York over the 14-year simulation period (N = 266).

	HWM	LWM		PAS	
	Full soil layer	Surface layer	Subsurface layer	Surface layer	Subsurface layer
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Mean	22.9	37.1	24.1	58.6	22.7
Standard Deviation	4.87	5.58	4.75	32.3	6.84
Median	22.2	36.8	23.9	48.6	21.1
Maximum	36.7	49.2	36.4	185	49.2
Minimum	14.9	22.9	17.8	22.3	11.8

Table 4.8: Summary of APLE-estimated soil labile P concentrations for three land uses in the state of Pennsylvania over the 14-year simulation period (N = 784).

	HWM	LWM		PAS	
	Full soil layer	Surface layer	Subsurface layer	Full soil layer	Surface layer
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Mean	34.6	49.0	36.0	188	65.6
Standard Deviation	10.8	14.0	10.9	79.2	51.7
Median	33.3	47.2	34.5	192	51.6
Maximum	73.0	142	74.7	464	484
Minimum	12.4	22.3	14.4	33.3	9.57

Table 4.9: Summary of APLE-estimated soil labile P concentrations for three land uses in the state of Virginia over the 14-year simulation period (N = 1244).

	HWM	LWM		PAS	
	Full soil layer	Surface layer	Subsurface layer	Surface layer	Full soil layer
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}		mg kg^{-1}
Mean	26.4	46.5	27.4	133	35.3
Standard Deviation	14.4	25.9	14.3	153	47.2
Median	21.8	39.7	22.9	119	28.7
Maximum	98.5	203	103	2917	811
Minimum	7.50	18.2	8.89	28.4	14.0

Table 4.10: Summary of APLE-estimated soil labile P concentrations for three land uses in the state of West Virginia over the 14-year simulation period (N = 210).

	HWM	LWM		PAS	
	Full soil layer	Surface layer	Subsurface layer	Full soil layer	Surface layer
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Mean	45.9	66.4	47.0	61.7	30.6
Standard Deviation	20.3	30.1	20.2	35.6	10.1
Median	40.4	59.7	41.6	49.3	30.7
Maximum	96.6	177	97.9	169	51.7
Minimum	11.1	23.0	13.1	19.4	8.03

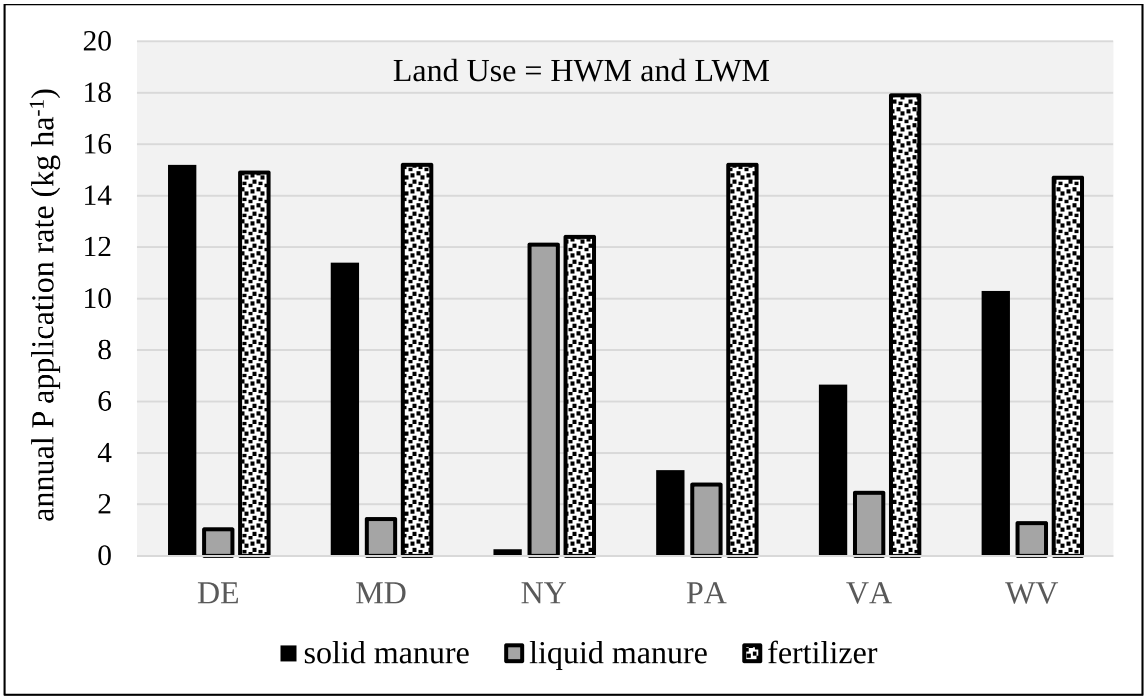

Figure 4.1: Mean annual P application rates of manure (solid and liquid) and fertilizer by state for the HWM and LWM land uses simulated by APLE. Annual P application rates for the HWM and LWM land uses were the same as for the WSM methodology.

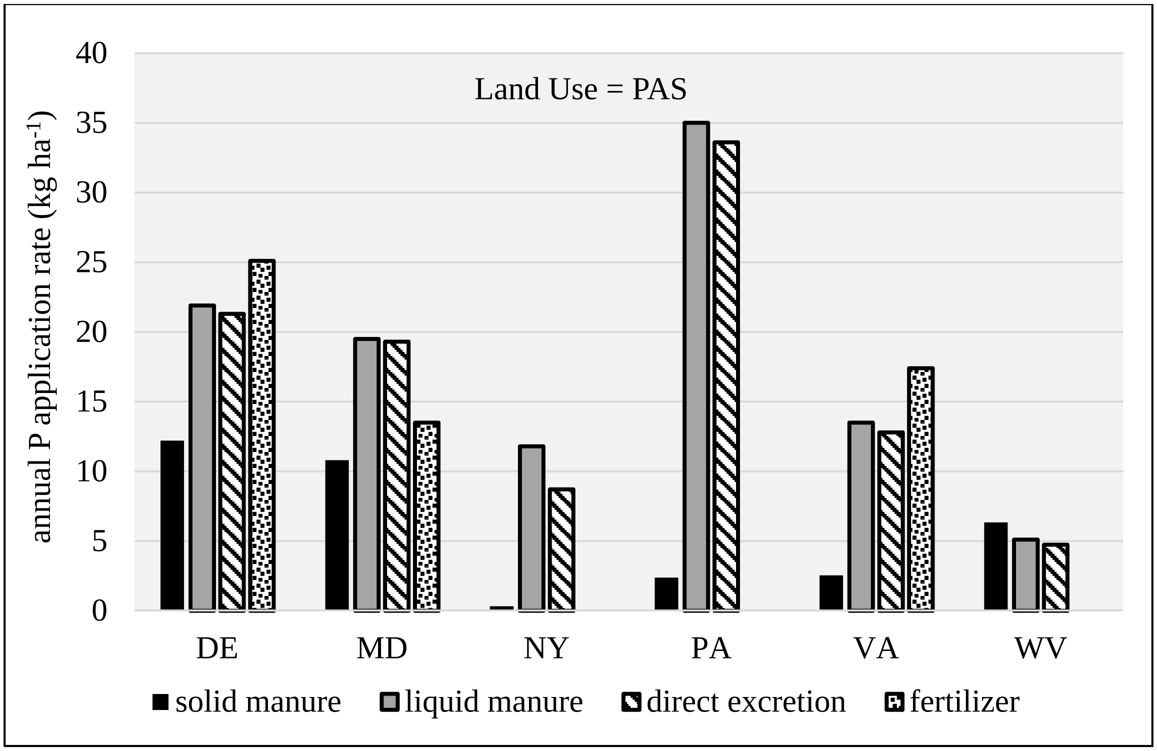

Figure 4.2: Mean annual P application rates of manure (solid and liquid), directly excreted manure, and fertilizer by state for the PASland use as simulated by APLE. Annual P application rates for the PAS land use were the same as for the WSM methodology.

Chapter 5: Cost effectiveness of BMPs with new total P loads

5.1 Abstract

The mandates of the Chesapeake Bay TMDL require a multi-jurisdictional approach in order to achieve the required nutrient and sediment load reductions to the Bay. As such, the jurisdictions are each pursuing an aggressive implementation schedule for best management practices (BMPs) to mitigate losses from the landscape and are evaluating methods to better prioritize the cost effectiveness of BMP implementation. Our study objective was to assess the effects of APLE-estimated EOF P losses on the cost effectiveness for an additional kilogram hectare^{-1} of annual P abatement. Results found prescribed grazing was the most cost effective BMP for pasture land uses, and continuous no-till was found to be the most cost effective BMP for high and low-till land uses.. Additionally, cost effectiveness estimates were highly variable due directly to the high variability of total P loss estimates from APLE; however, BMP implementation in a landscape is also highly variable by practice and costs so analyses must be re-evaluated as new data becomes available.

5.2 Introduction

To address water quality concerns in the Chesapeake Bay (Bay), a TMDL was mandated in December 2010 (USEPA, 2010a). Given the magnitude of the watershed, approximately 166,000 km^2 of drainage in the Mid-Atlantic region, and the multi-jurisdictional approach necessary to achieve the required pollutant load reductions, reliance on watershed modeling has become absolute. As the current suite of watershed models has proven to be extremely helpful in describing the complex Bay ecosystem due to their inherent time and cost effectiveness, enhanced scrutiny of model output has underscored the importance of updating model routines as new routines becomes available.

A critical limitation in the utilization of watershed models is the fact that agricultural Best Management Practices (BMPs) which are designed and implemented to reduce the edge-of-field (EOF) nutrient or sediment load losses are not simulated in most process models. Rather, BMP implementation is often tracked for post-processing as an efficiency or percentage reduction applied to the assumed EOF base load loss. In the case of the Chesapeake Bay Watershed Model (WSM), expert panels have been convened throughout the TMDL process to craft a methodology for BMP tracking and their associated efficiency credits for supplemental pollutant load reductions (Palace et al., 1998).

However, using a BMP accounting methodology that applies a simple percentage reduction to loads prior to stream delivery also dictates the course of action that state resource management agencies undertake in designing aggressive nutrient load reduction efforts within the Bay watershed. Consequently, in the absence of model simulation of BMPs, an alternative approach is to evaluate the cost effectiveness of BMPs as an annual cost per kilogram of P reduction in an effort to achieve a more targeted BMP implementation. Wieland et al. (2009) assessed 12 prevalent BMPs promoted in Maryland and found assigning a singular cost effectiveness to a BMP was complicated because of the many implementation scenarios that can exist on the landscape. In two other studies within the Bay watershed (Chesapeake Bay Commission, 2012; Wainger et al., 2013), when BMP options across the agricultural nonpoint source sector were compared to those in the urban stormwater sector and urban point source (e.g. waste water treatment plant) sector, both studies consistently found the least-cost options for achieving reduced P loads resulted from load reductions in the agricultural nonpoint source sector. Achieving the required P load reductions mandated by the Bay TMDL will require prioritizing BMP implementation based on costs during a period of frequently constrained state budgets. Additionally, as new data becomes available in the literature BMP cost effectiveness must be re-evaluated with new economic analysis (Wieland et al., 2009; Chesapeake Bay Commission, 2012; Wainger et al., 2013).

The objective of our study was to evaluate the presumed cost effectiveness of some commonly and widely implemented agricultural BMPs when alternative annual P loss estimates derived from Annual Phosphorus Loss Estimator (APLE) model simulations were incorporated at the land segment scale.

5.3 Materials and Methods

5.3.1 Overview

The Chesapeake Bay WSM methodology accounted for BMPs in three ways: 1) an efficiency credit applied to the estimated P load, 2) a land use change, or 3) a load reduction (USEPA, 2012). An example of the efficiency credit option is a fixed percentage P reduction applied to edge-of-stream loads when winter annual cover crops are grown. An example of a land use change is the planting of a riparian buffer on a formerly cultivated agricultural field that would change the land use designation from agricultural row cropland to forest. Since each land use category defined in the WSM has an assumed target base load, a change in the designated land use would equate to a new base load P loss assumption. Finally, an example of the load reduction option is the livestock precision feeding BMP wherein a change in the animal's diet would change the nutrient concentration of the generated manure and subsequently the nutrient application to the landscape within the land segment. The majority of agricultural BMPs accounted for in the WSM are adjusted by option 1, the efficiency credit, based on an adaptive management framework led by expert panels during the TMDL process (USEPA, 2012). Without the ability to directly simulate BMP implementation in the WSM, interpretive evaluation of field research studies and professional judgment continue to be the source for approximating BMP effectiveness through efficiency crediting. These BMP efficiency credits have been coupled with BMP implementation costs and estimates of total P loads from the WSM to determine BMP cost effectiveness. In this study, APLE estimates of total P load losses from three land uses – high-till with manure (HWM), low-till with manure (LWM), and pasture (PAS) – were used to calculate alternative BMP cost effectiveness for reducing P loss from agricultural landscapes.

5.3.2 Estimates of BMP costs and efficiency

Four BMPs were evaluated for each land use (Table 5.1). The selected BMPs were those considered most representative of in-field practices that an operator would implement to minimize off-site dissolved P or sediment P losses. Likewise, the selected BMPs best aligned with the APLE estimates of EOF total P losses. Five economic analysis studies were compared for estimates of annual unit costs (cost hectares^{-1} year^{-1}) for each BMP. In most cases, because the study objectives were specific to the Bay watershed, cost estimates were comparable between studies (Wieland et al., 2009; Chesapeake Assessment Scenario Tool, 2011; AREC-DNR; Chesapeake Bay Commission, 2012; Wainger et al., 2013). Annual unit costs represented the one-time installation and capital costs of a BMP and the annual maintenance costs in 2010 dollars. The efficiency assumptions for each BMP were the same as in the WSM with the following exceptions: 1) the same efficiency for continuous no-till (CNT) implementation was assumed for the HWM and LWM land use categories, because CNT implementation on HWM land was not included in the WSM. The assumed efficiency for CNT implies the conversion of a previously high-till (HWM) field to CNT, and not an annual change in tillage between high and reduced tillage; 2) a median efficiency of 30% was assumed for CNT implementation on HWM and LWM land use categories, because the WSM applies the efficiency based on the specific physiographic region of CNT implementation, and 3) the same efficiency for rye cover crop implementation was assumed for the HWM and LWM land use categories, because rye cover crop implementation on LWM land was not included in the WSM (Table 5.1). To calculate the cost effectiveness of each BMP, the cost per unit BMP was divided by the product of the annual total P load and the BMP efficiency.

5.4 Results and Discussion

5.4.1 High-till with manure (HWM) land use

Bay-wide, the mean cost of abatement of EOF P loss for the Nutrient Management Planning (NMP) BMPs was \$24.37 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost of \$8.84 ha^{-1} y^{-1} kg^{-1} of P loss reduction. The mean cost effectiveness of EOF

P loss abatement for Soil and Water Conservation Planning (SWCP) BMPs was \$21.61 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost of \$7.36 ha^{-1} y^{-1} kg^{-1}total P reduction. The mean cost effectiveness of Ploss abatement for drill-seeded rye cover crops (CC), planted early was \$48.91 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost of \$16.65 ha^{-1} y^{-1} kg^{-1}total P reduction. Lastly, the mean cost effectiveness of P abatement for continuous no-till (CNT) was \$20.00 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost of \$6.81 ha^{-1} y^{-1} kg^{-1}total P reduction (Table 5.2). The discrepancies between the mean and median of each BMP demonstrates the wide range of annual cost effectiveness calculations for each land segment (i.e. county) that can be directly attributed to the wide range in APLE estimates of annual EOF total P loads. When evaluating the cost effectiveness of each BMP, the median cost effectiveness for P abatement was considered to be less influenced by outliers and consequently should be used to rank BMP cost effectiveness. According to BMP median P effectiveness cost ranking, CNT was identified as the most cost-effective BMP for total EOF P loss reduction, under the conditions simulated, followed by SWCP, NMP, and lastly, CC (Table 5.2). As previously stated, the implementation of CNT would represent the full conversion of a previously high-tilled field to a reduced tillage (CNT) practice.

When evaluated at the state scale, findings for cost effectiveness followed a similar trend with CNT being the most cost effective BMP for total P load reduction followed by SWCP, NMP, and CC (Table 5.2 and Figure 5.1). The estimated P abatement cost effectiveness for BMPs in Virginia was a notable exception to the Bay-wide findings. In Virginia, NMP was identified to be the most cost effective BMP for reducing EOF P loads, followed in order by CNT, SWCP and CC. The greater cost effectiveness of NMP in Virginia was attributed to the lower annual unit cost of the BMP, \$2.84hectare^{-1}, which is at least two-times lower than ascribed by other states (Table 5.1). Conversely, the state of West Virginia had the highest annual unit cost for NMP, \$10.53hectare^{-1}, resulting in the NMP BMP ranking last in cost effectiveness for this state (Figure 5.1).

5.4.2 Low-till with manure (LWM) land use

The Bay-wide mean cost effectiveness of EOF P loss abatement for NMP BMPs was \$87.84 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost of \$21.72 ha^{-1} y^{-1} kg^{-1}total P reduction. The mean cost effectiveness of P loss abatement for SWCP BMPs was \$244.24 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost effectiveness of \$52.64 ha^{-1} y^{-1} kg^{-1}total P reduction. The mean cost effectiveness of abatement for CC was \$184.27 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost effectiveness of \$39.71 ha^{-1} y^{-1} kg^{-1}total P reduction. The Bay-wide P loss cost effectiveness for CNT was \$75.35 ha^{-1} y^{-1} kg^{-1}total P reduction with a median cost effectiveness of \$16.24 ha^{-1} y^{-1} kg^{-1}total P reduction (Table 5.2). Bay-wide the median P loss cost effectiveness for the LWM land use were nearly 2.5 times the cost effectiveness of similar P abatement BMPs implemented on HWM land uses for three of the BMPs (CNT, NMP, and CC). The exception, SWCP, was attributed to the WSM assigning a substantially lower efficiency credit for LWM, i.e. 5%, as compared to the HWM land use, i.e. 15% (Table 5.1). As a result, SWCP was identified as the least cost effective BMP for the LWM land use under the conditions simulated.

The most cost effective BMP for the LWM land use was CNT, followed in order by NMP and CC, similar to the HWM results.

When evaluated at the state scale, BMP cost effectiveness followed a similar trend for LWM as was observed for HWM land uses with CNT typically being the most cost effective BMP for P loss abatement followed by NMP, CC and, lastly, SWCP (Tables 5.2). Again like the HWM land use, the exception was the state of Virginia where NMP was found to be the most cost effective BMP, followed by CNT, CC and SWCP (Figure 5.2), which was driven by the much lower annual unit cost attributed to the NMP BMP in Virginia (Table 5.1). Overall, findings of higher abatement costs for the LWM land use as compared to the HWM land use are attributed to lower overall APLE estimates of total P losses for the LWM land use prior to the BMP being implemented. Specifically, APLE estimates of total EOF P load losses were inversely proportional to the calculated P abatement cost effectiveness of a BMP whereby reduced-tillage practices (LWM) are analogous to reduced sediment disturbance and a lesser likelihood for overland total P transport to the EOF resulting in lesser cost effectiveness of the BMPs.

Additionally as was observed for the HWM land use, a large disparity between the mean and median P loss cost effectiveness estimates for each modeled BMP existed due to the highly variable APLE estimations of annual EOF P load losses. In all cases, it is important to consider that actual BMP implementation across the landscape is also highly variable in actual costs and effectiveness. While the data above represent mean and median P loss abatement costs by BMP, the variability of APLE P load estimates discussed in Chapter 3 affirms the subsequent non-homogeneity of cost effectiveness estimates. Likewise, the BMP cost effectiveness analysis found that CNT was the most cost effective P-loss reducing BMP for both the HWM and LWM land uses. While CNT decreases the likelihood for soil disturbance and subsequent soil erosion and sediment-bound P losses, it can also contribute significantly to soil P stratification and P over-enrichment of shallow surface soil, which implies the potential for increased surface soil labile P concentration and soluble P loss via surface runoff pathways. This potential result should not be inferred to discourage CNT as a BMP, but rather it highlights the complexity of prioritizing BMPs for cost effectiveness and impact on water quality preservation.

5.4.3 Pasture (PAS)

For the PAS land use, the Bay-wide mean cost effectiveness of EOF P loading abatement for the NMP BMP was \$208.91ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost effectiveness of \$36.89ha^{-1} y^{-1} kg^{-1}total P reduction. The mean cost effectiveness of P loss abatement for the SWCP BMP was \$222.91ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost effectiveness of \$33.97ha^{-1} y^{-1} kg^{-1}total P reduction. The mean cost effectiveness of EOF P load abatement for the alternate watering facilities without fencing (AW) BMP was \$42,961 ha^{-1} y^{-1} kg^{-1}total P reduction, and a median cost effectiveness of \$6,548 ha^{-1} y^{-1} kg^{-1}total P reduction. Lastly, the mean cost effectiveness of P abatement for the prescribed grazing (PG) BMP was \$161.24 ha^{-1} y^{-1} kg^{-1}total P reduction with a median cost effectiveness of \$19.95 ha^{-1} y^{-1} kg^{-1}total P reduction (Table 5.3).

When the applicable PAS land use BMPs were ranked relative to P loss abatement cost effectiveness, PG was identified as the most cost effective BMP, followed by SWCP and NMP, which were all much more cost effective than implementation of AW practices (Table 5.3). The superior cost effectiveness of PG was due to its greater efficiency credit, 20%, as compared to the other BMPs available for the PAS land use (Table 5.1). Interestingly, the P abatement cost effectiveness of SWCP for the PAS land use was better than SWCP cost effectiveness for the LWM land use due to the greater SWCP efficiency credit assumed for PAS, 10%, as compared to LWM, 5% (Table 5.1).

When evaluated at the state scale, PG was primarily the most cost effective BMP per kilogram of total P load reduction followed by SWCP, NMP, and, by a significant margin, AW (Tables 5.3 and Figure 5.3). Two exceptions were Delaware and Virginia. In Delaware, the estimated P abatement cost effectiveness of SWCP was better than PG (Table 5.3), due, in large part, to the higher annual unit cost of PG ascribed to Delaware, \$13.50hectare^{-1}, as compared to the other states (Table 5.1). As a result, SWCP was the most cost effective BMP for PAS land use in Delaware, followed by PG, NMP, and AW. Likewise in Virginia, the low ascribed cost for NMP (\$2.84 hectare^{-1}) and the higher ascribed cost for PG (\$10.45 hectare^{-1}) compared to other states resulted in NMP being the most cost effective BMP for PAS land uses, followed by PG, SWCP, and AW. For all states, the AW BMP was by far the least cost effective BMP for the PAS land use (Figure 5.3), due to the significantly high annual unit cost of the BMP, \$955hectare^{-1}, and the low efficiency credit assumed, 8% (Table 5.1).

5.5 Conclusion

Best management practices are encouraged and implemented across the diverse landscape of the Bay watershed in hopes of achieving further EOF P load reductions and ultimately attaining the water quality standards prescribed by the Bay TMDL. Many states in the Bay watershed are aggressively pursuing BMPs through cost-share and incentive based programs to further promote BMP adoption. Consequently, economic analyses that can prioritize BMP cost effectiveness by land segment or BMP type are essential. However, the reality is more complicated than a simple Bay-wide ranking because annual BMP unit costs vary by state; BMP efficiencies are not credited consistently across the Bay watershed; and estimates of total P losses can be highly variable spatially. Additionally, BMPs deemed most cost effective, such as continuous no-till, may efficiently mitigate against P loss by one pathway (sediment P loss) while inadvertently increasing the P loss potential by an alternative pathway (dissolved P loss). All such variables must continue to be studied and brought together cooperatively as new data becomes available. Iterative, revised analyses are warranted to support the Bay watershed states in attaining the aggressive P loading reductions required under the imposed Bay TMDL.

Table 5.1: Summary of BMP definitions, percent efficiency, and unit cost used in an analysis of BMP cost effectiveness for three land uses. Cost effectiveness is calculated as the BMP unit cost divided by the product of the annual total P load and the BMP efficiency, where estimates of annual total P loads were from APLE.

BMP	BMP Definition[1]	BMP efficiency per land use (%)			BMP cost hectare $^{-1}$(2010$)[6]
		HWM	LWM	PAS	
Nutrient Management Planning (NMP)	Comprehensive planning that describes the optimum use of nutrients to minimize nutrient loss while maintaining yield. A NMP details the type, rate, timing, and placement of nutrients for each crop	15	5	10	State specific[4]
Soil and Water Conservation Plans (SWCP)	Combination of agronomic, management and engineered practices that protect and improve soil productivity and water quality, and prevent deterioration of natural resources on all or part of a farm. All practices must meet a technical standard	10	10	8	$6.20
Cover Crops (CC)[2]	*P*lanting and growing of cereal crops (non-harvested) with minimal disturbance of the surface soil during the winter months	15	15	NA	$14.02
Continuous no-till (CNT)	Crop planting and management practice in which soil disturbance by plows, disk or other tillage equipment is eliminated on all crops in a multi-crop, multi-year rotation	30	30	NA	$11.47
Alternate Watering (AW)[3]	Use of permanent or portable livestock water troughs placed away from the stream corridor *to control livestock access*	NA	NA	8	$955
Prescribed grazing (PG)	*R*ange of pasture management and grazing techniques to improve the quality and quantity of the forages grown on pastures and reduce the impact of animal concentration areas or other degraded areas	NA	NA	20	State specific[5]

1. BMP definitions are those used in the Chesapeake Bay Program WSM (USEPA, 2012).
2. A variety of cover crop specie and planting methods are available for credit in the WSM, but for purposes of this study early-planted, drill-seeded rye was selected (USEPA, 2012).
3. Refers to alternate watering facilities without exclusion fencing.
4. Costs are DE = $6.14; MD = $8.85; NY = $7.23; PA = $6.22; VA = $2.84; WV = $10.53
5. Costs are DE = $13.50; MD = $5.94; NY = $5.40; PA = $6.48; VA = $10.45; WV = $3.67
6. Chesapeake Assessment Scenario Tool, 2011.

Table 5.2: Cost effectiveness ($ ha^{-1} y^{-1} kg^{-1}) for total P loss reduction for four BMPs implemented on two land uses Bay-wide and in each state of the Chesapeake Bay watershed. BMPs are: NMP=nutrient management planning; SWCP=soil and water conservation planning; CC=winter annual cover crop production; and CNT=continuous no-tillage management. Land uses are: HWM=high till with manure; LWM=low till with manure.

State	Land use			BMP			
				NMP	SWCP	CC	CNT
			N	$ ha^{-1} y^{-1} kg^{-1}			
Bay-wide	HWM	Mean	2350	24.37	21.61	48.91	20.00
		Median	2350	8.84	7.36	16.65	6.81
	LWM	Mean	2350	87.84	244.24	184.27	75.35
		Median	2350	21.72	52.64	39.71	16.24
DE	HWM	Mean	42	25.91	17.46	39.52	16.16
		Median	42	12.59	8.48	19.20	7.85
	LWM	Mean	42	119.56	241.64	182.31	74.54
		Median	42	39.66	80.15	60.47	24.73
MD	HWM	Mean	307	28.10	13.12	29.69	12.14
		Median	307	8.02	3.75	8.48	3.47
	LWM	Mean	307	108.97	152.61	115.13	47.08
		Median	307	21.91	30.69	23.15	9.47
NY	HWM	Mean	266	18.00	10.29	23.29	9.52
		Median	266	12.48	7.13	16.15	6.60
	LWM	Mean	266	46.78	80.23	60.53	24.75
		Median	266	28.03	48.07	36.27	14.83
PA	HWM	Mean	602	17.01	11.30	25.57	10.45
		Median	602	8.33	5.53	12.53	5.12
	LWM	Mean	602	52.96	105.51	79.60	32.55
		Median	602	21.85	43.52	32.84	13.43
VA	HWM	Mean	979	22.76	33.16	75.07	30.69
		Median	979	7.04	10.25	23.20	9.49
	LWM	Mean	979	91.65	400.63	302.26	123.59
		Median	979	16.03	70.09	52.88	21.62
WV	HWM	Mean	154	66.47	26.07	59.00	24.13
		Median	154	22.30	8.74	19.79	8.09
	LWM	Mean	154	220.20	259.05	195.45	79.92
		Median	154	53.65	63.12	47.62	19.47

Table 5.3: Cost effectiveness ($ ha^{-1} y^{-1} kg^{-1}) for total P loss reduction, for four BMPs implemented on the PAS land use Bay-wide and in each state of the Chesapeake Bay watershed. BMPs are: NMP=nutrient management planning; SWCP=soil and water conservation planning; AW=alternative watering; PG=prescribed grazing.

State			BMP			
			NMP	SWCP	AW	PG
		N	$ ha^{-1} y^{-1} kg^{-1}			
Bay-wide	Mean	2350	208.91	222.91	42,961	161.24
	Median	2350	36.89	33.97	6,548	19.95
DE	Mean	42	537.90	434.87	83,811	473.66
	Median	42	149.53	120.89	23,299	131.67
MD	Mean	307	184.82	103.53	19,954	49.63
	Median	307	60.71	34.01	6,555	16.30
NY	Mean	266	148.15	101.64	19,590	44.28
	Median	266	84.56	58.01	11,181	25.27
PA	Mean	601	94.69	75.46	14,543	39.45
	Median	601	34.53	27.52	5,304	14.39
VA	Mean	980	202.16	353.33	68,096	297.90
	Median	980	19.37	33.87	6,527	28.55
WV	Mean	154	760.87	358.06	69,008	106.13
	Median	154	52.34	24.63	4,747	7.30

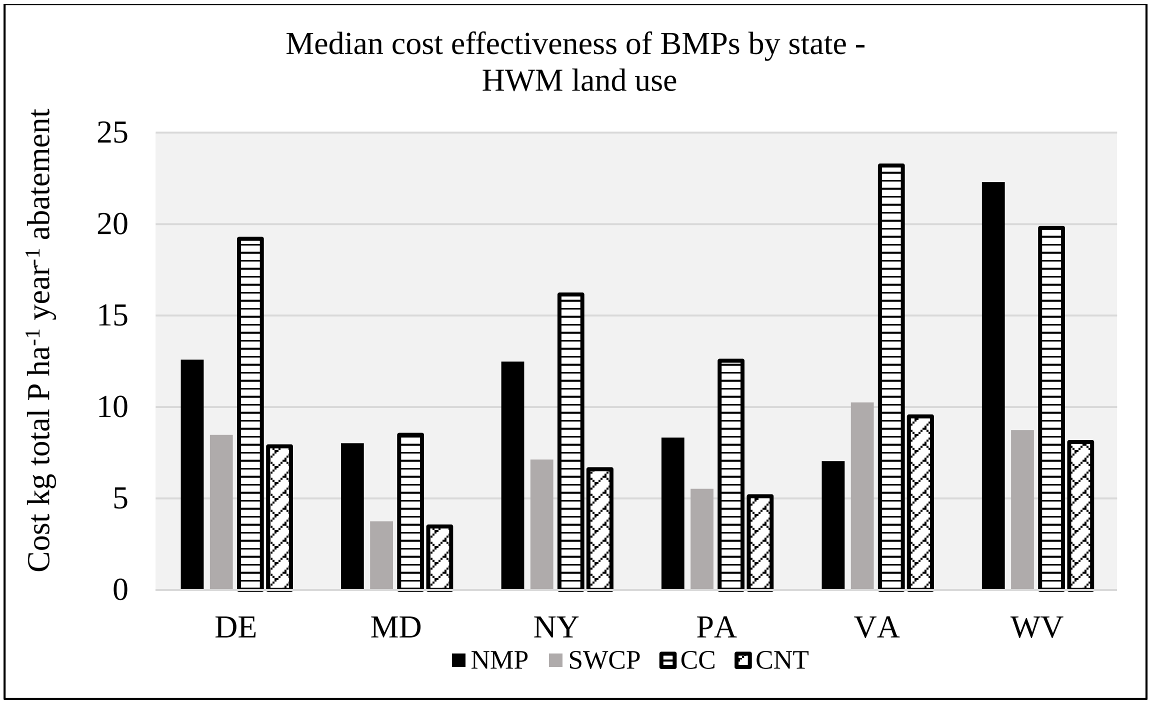

Figure 5.1: Median cost effectiveness ($ hectare^{-1} year^{-1}) for four BMPs implemented on the HWM land use. BMPs are NMP = nutrient management planning; SWCP = Soil and Water Conservation Plans; CC = rye cover crop; CNT = continuous no-till.

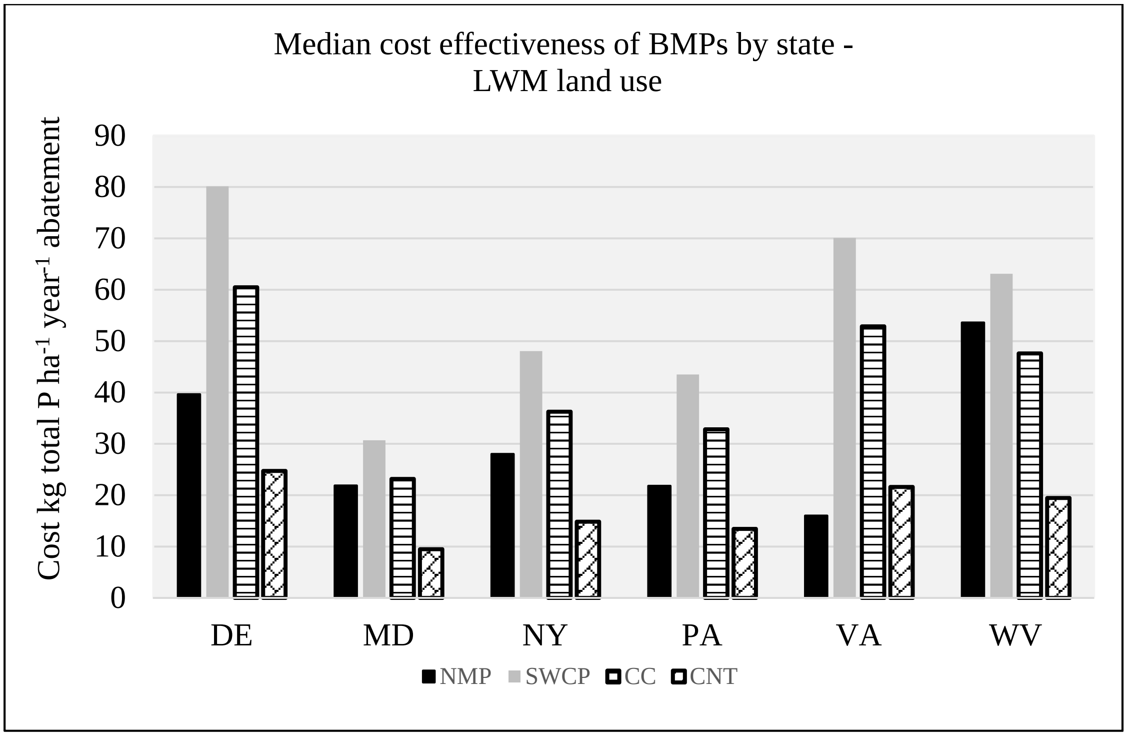

Figure 5.2: Median cost effectiveness (\$ hectare^{-1} year^{-1}) for four BMPs implemented on the LWM land use. BMPs are NMP = nutrient management planning; SWCP = Soil and Water Conservation Plans; CC = rye cover crop; CNT = continuous no-till.

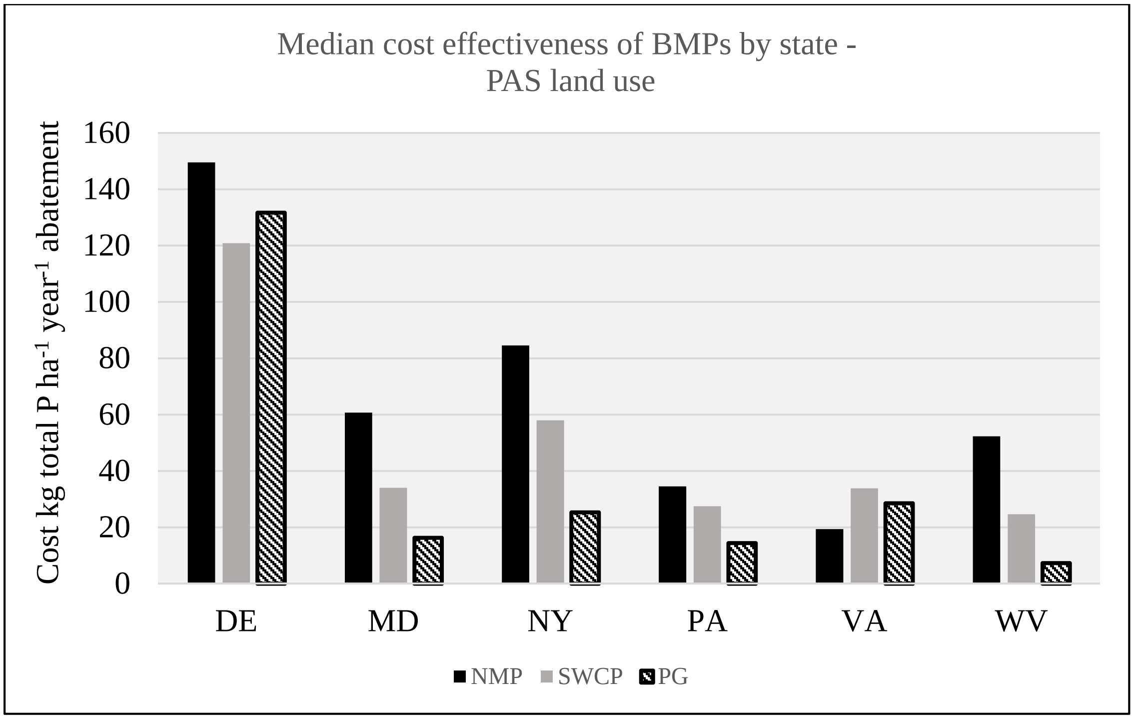

Figure 5.3: Median cost effectiveness (\$ hectare^{-1} year^{-1}) for three BMPs implemented on the PAS land use. BMPs are NMP = nutrient management planning; SWCP = Soil and Water Conservation Plans; PG = Prescribed Grazing. The cost effectiveness of alternate watering facilities without exclusion fencing was excluded from the graph due to significantly greater cost effectiveness estimates.

Appendices

APPENDIX A: List of land segment's unique alphanumeric code and location as simulated within the Chesapeake Bay Watershed Model. Land segments are generally modeled at the county scale; however, some counties are defined at a smaller scale due to physiographic differences within the county. In such cases, additional land segments are assigned with a B or C designation. Below is the list of land segments by state including their alphanumeric codes.

Delaware

A10001	Kent
A10003	New Castle
A10005	Sussex

Maryland

A24001	Allegany	A24031	Montgomery
A24003	Anne Arundel	A24033	Prince George
A24005	Baltimore	A24035	Queen Anne's
A24009	Calvert	A24037	St Mary's
A24011	Caroline	A24039	Somerset
A24013	Carroll	A24041	Talbot
A24015	Cecil	A24043	Washington
A24017	Charles	A24045	Wicomico
A24019	Dorchester	A24047	Worcester
A24021	Frederick	B24001	Allegany
A24023	Garrett	B24021	Frederick
A24025	Harford		
A24027	Howard		
A24029	Kent		

New York

A36003	Allegany	A36095	Schoharie
A36007	Broome	A36097	Schuyler
A36015	Chemung	A36101	Steuben
A36017	Chenango	A36107	Tioga
A36023	Cortland	A36109	Tompkins
A36025	Delaware	A36123	Yates
A36043	Herkimer		
A36051	Livingston		
A36053	Madison		
A36065	Oneida		
A36067	Onondaga		
A36069	Ontario		
A36077	Otsego		

Pennsylvania

A42001	Adams	A42099	Perry
A42009	Bedford	A42105	Potter
A42011	Berks	A42109	Snyder
A42013	Blair	A42111	Somerset
A42015	Bradford	A42107	Schuylkill
A42021	Cambria	A42113	Sullivan
A42023	Cameron	A42115	Susquehanna
A42025	Carbon	A42117	Tioga
A42027	Centre	A42119	Union
A42029	Chester	A42127	Wayne
A42033	Clearfield	A42131	Wyoming
A42035	Clinton	A42133	York
A42037	Columbia	B42001	Adams
A42041	Cumberland	B42009	Bedford
A42043	Dauphin	B42015	Bradford
A42047	Elk	B42027	Centre
A42055	Franklin	B42035	Clinton
A42057	Fulton	B42041	Cumberland
A42061	Huntingdon	B42043	Dauphin
A42063	Indiana	B42055	Franklin
A42065	Jefferson	B42079	Luzerne
A42067	Juniata	B42081	Lycoming
A42069	Lackawanna	B42119	Union
A42071	Lancaster	B42131	Wyoming
A42075	Lebanon	C42009	Bedford
A42079	Luzerne		
A42081	Lycoming		
A42083	McKean		
A42087	Mifflin		
A42093	Montour		
A42097	Northumberland		

Virginia

A51001	Accomack	A51161	Roanoke
A51003	Albemarle	A51163	Rockbridge
A51005	Alleghany	A51165	Rockingham
A51007	Amelia	A51179	Stafford
A51009	Amherst	A51181	Surry
A51011	Appomattox	A51187	Warren
A51013	Arlington	A51193	Westmoreland
A51015	Augusta	A51199	York
A51017	Bath	A51510	Alexandria (city)

A51019	Bedford	A51530	Buena Vista (city)
A51023	Botetourt	A51540	Charlottesville (city)
A51029	Buckingham	A51550	Chesapeake (city)
A51031	Campbell	A51570	Colonial Heights (city)
A51033	Caroline	A51580	Covington (city)
A51036	Charles City	A51600	Fairfax City
A51041	Chesterfield	A51610	Falls Church (city)
A51043	Clarke	A51630	Fredericksburg (city)
A51045	Craig	A51650	Hampton (city)
A51047	Culpepper	A51660	Harrisonburg (city)
A51049	Cumberland	A51670	Hopewell (city)
A51053	Dinwiddie	A51678	Lexington (city)
A51057	Essex	A51680	Lynchburg (city)
A51059	Fairfax	A51683	Manassas (city)
A51061	Fauquier	A51685	Manassas Park (city)
A51065	Fluvanna	A51700	Newport News (city)
A51069	Frederick	A51710	Norfolk (city)
A51071	Giles	A51730	Petersburg (city)
A51073	Gloucester	A51735	Poquoson (city)
A51075	Goochland	A51740	Portsmouth (city)
A51079	Greene	A51760	Richmond (city)
A51085	Hanover	A51790	Staunton (city)
A51087	Henrico	A51800	Suffolk (city)
A51091	Highland	A51810	Virginia Beach (city)
A51093	Isle Of Wight	A51820	Waynesboro (city)
A51095	James City	A51830	Williamsburg (city)
A51097	King & Queen	A51840	Winchester (city)
A51099	King George	B51003	Albemarle
A51101	King William	B51009	Amherst
A51103	Lancaster	B51015	Augusta
A51107	Loudoun	B51017	Bath
A51109	Louisa	B51019	Bedford
A51113	Madison	B51023	Botetourt
A51115	Mathews	B51061	Fauquier
A51119	Middlesex	B51071	Giles
A51121	Montgomery	B51079	Greene
A51125	Nelson	B51091	Highland
A51127	New Kent	B51113	Madison
A51131	Northampton	B51125	Nelson
A51133	Northumberland	B51139	Page
A51135	Nottoway	B51157	Rappahannock
A51137	Orange	B51161	Roanoke
A51139	Page	B51163	Rockbridge
A51145	Powhatan	B51165	Rockingham

A51147	Prince Edward	B51171	Shenandoah
A51149	Prince George	B51187	Warren
A51153	Prince William	C51015	Augusta
A51157	Rappahannock	C51071	Giles
A51159	Richmond	C51165	Rockingham

West Virginia

A54003	Berkeley
A54023	Grant
A54027	Hampshire
A54031	Hardy
A54037	Jefferson
A54057	Mineral
A54063	Monroe
A54065	Morgan
A54071	Pendleton
A54077	Preston
A54093	Tucker
B54023	Grant
B54031	Hardy
B54057	Mineral
B54071	Pendleton

Appendix B: List of agricultural BMPs simulated within the Chesapeake Bay WSM including BMP name and description.

BMP NAME	BMP DESCRIPTION
Agricultural Forest Buffers	Agricultural riparian forest buffers are linear wooded areas along rivers, stream and shorelines. Forest buffers help filter nutrients, sediments and other pollutants from runoff as well as remove nutrients from groundwater. The recommended buffer width for riparian forest buffers (agriculture) is 100 feet, with 35 feet minimum width required.
Agricultural Grass Buffers	Agricultural riparian grass buffers are linear strips of grass or other non- woody vegetation maintained between the edge of fields and streams, rivers or tidal waters that help filter nutrients, sediment, and other pollutants from runoff. The recommended buffer width for riparian grass buffers (agriculture) is 100 feet, with 35 feet minimum width required.
Agricultural Wetland Restoration	Agricultural wetland restoration activities reestablish the natural hydraulic condition in a field that existed before the installation of subsurface or surface drainage. Projects can include restoration, creation and enhancement acreage. Restored wetlands can be any wetland classification including forested, scrub-shrub or emergent marsh.
Ammonia Reduction	Litter amendments like alum suppress the formation of ammonia from ammonium in litter. Biofilters attached to animal enclosure ventilation systems detoxify ammonia. Geotextile manure covers reduce surface area and temperature of manure, therefore preventing ammonia volatilization.
Animal Waste Management Systems	Animal waste management systems are practices designed for proper handling, storage, and use of wastes generated from AFOs and includes a system of collecting, scraping, or washing wastes and contaminated runoff from confinement areas into appropriate waste storage structures. Lagoons, ponds, or steel or concrete tanks are used for treating or storing liquid wastes. Storage sheds or pits are common storage structures for solid wastes. Controlling runoff from roofs, feedlots and loafing areas are an integral part of such systems. Practices designed for proper handling, storage, and use of wastes generated from animal feeding operations.

Alternative Watering Facility without Exclusion Fencing	Alternative watering facilities typically involves the use of permanent or portable livestock water troughs placed away from the stream corridor. The source of water supplied to the facilities can be from any source including pipelines, spring developments, water wells, and ponds. In-stream watering facilities such as stream crossings or access points are not considered in this definition. The modeled benefits of alternative watering facilities can be applied to pasture acres in association with or without improved pasture management systems such as prescribed grazing. They can also be applied in conjunction with or without stream access control**.**
Barnyard Runoff Control	This practice includes the installation of practices to control runoff from barnyard areas. This includes practices such as roof runoff control, diversion of clean water from entering the barnyard and control of runoff from barnyard areas.
Conservation Tillage	Conservation tillage involves planting and growing crops with minimal disturbance of the surface soil. Conservation tillage requires two components, (a) a minimum 30% residue coverage at the time of planting and (b) a non-inversion tillage method. No-till farming is a form of conservation tillage in which the crop is seeded directly into vegetative cover or crop residue with little disturbance of the surface soil. Minimum tillage farming involves some disturbance of the soil, but uses tillage equipment that leaves much of the vegetation cover or crop residue on the surface.
Continuous No-till	The Continuous No-Till (CNT) BMP is a crop planting and management practice in which soil disturbance by plows, disk or other tillage equipment is eliminated. CNT involves no-till methods on all crops in a multi-crop, multi-year rotation. When an acre is reported under CNT, it will not be eligible for additional reductions from the implementation of other practices such as cover crops or nutrient management planning. Multi-crop, multi-year rotations on cropland are eligible. Crop residue should remain on the field. Planting of a cover crop might be needed to maintain residue levels. Producers must have and follow a current nutrient management plan. The system must be maintained for a minimum of five years.
Cover Crop	Cereal cover crops reduce erosion and the leaching of nutrients to groundwater by maintaining a vegetative cover on cropland and holding nutrients within the root zone. This practice involves the planting and growing of cereal crops (non-

	harvested) with minimal disturbance of the surface soil. The crop is seeded directly into vegetative cover or crop residue with little disturbance of the surface soil. These crops capture or "trap" nitrogen in their tissues as they grow. By timing the cover crop burn or plow-down in spring, the trapped nitrogen can be released and used by the following crop. Different species are accepted as well as, different times of planting (early, late and standard), and fertilizer application restrictions. Manure application on cover crops is not modeled and acres of cover crops that receive manure are not eligible. There is a sliding scale of efficiencies based on crop type and time of planting. <u>Commodity cover crops</u> differ from cereal cover crops in that they can be harvested for grain, hay, or silage and they might receive nutrient applications, but only after March 1 of the spring following their establishment. The intent of the practice is to modify normal small grain production practices by eliminating fall and winter fertilization so that crops function similarly to cover crops by scavenging available soil nitrogen for part of their production cycle.
Dairy Feed Management	Reduces the quantity of phosphorous and nitrogen fed to livestock by formulating diets within 110% of NRC recommended level to minimize the excretion of nutrients without negatively affecting milk production.
Decision Agriculture	A management system that is information and technology based, is site specific and uses one or more of the following sources of data: soils, crops, nutrients, pests, moisture, or yield for optimum profitability, sustainability, and protection of the environment.
Enhanced Nutrient Management	Based on research, the nutrient management rates of nitrogen application are set approximately 35% higher than what a crop needs to ensure nitrogen availability under optimal growing conditions. In a yield reserve program using enhanced nutrient management, the farmer would reduce the nitrogen application rate by 15%. An incentive or crop insurance is used to cover the risk of yield loss. This BMP effectiveness estimate is based on a reduction in nitrogen loss resulting from nutrient application to cropland 15% lower than the nutrient management recommendation.
Horse Pasture Management	Stabilizing overused small pasture containment areas (animal concentration area) adjacent to animal shelters or farmstead.

Land Retirement	Agricultural land retirement takes marginal and highly erosive cropland out of production by planting permanent vegetative cover such as shrubs, grasses, and/or trees. Agricultural agencies have a program to assist farmers in land retirement procedures.
Loafing Lot Management	The stabilization of areas frequently and intensively used by people, animals or vehicles by establishing vegetative cover, surfacing with suitable materials, and/or installing needed structures. This does not include poultry pad installation.
Manure Transport	Manure is transported by truck from the county of origin to another or out of the watershed. Manure transported to another county in the watershed results in increased manure mass in the receiving county. Excess manure is defined as manure nutrients produced within an area that exceeds the recommended application rates associated with the crops grown.
Nutrient Management Plans	Nutrient management plan (NMP) implementation is a comprehensive plan that describes the optimum use of nutrients to minimize nutrient loss while maintaining yield. An NMP details the type, rate, timing, and placement of nutrients for each crop. Soil, plant tissue, manure, or sludge tests are used to assure optimal application rates. Plans should be revised every 2 to 3 years.
Phytase	Phytase can be injected into poultry feeds by the integrator or other feed suppliers. Manure phosphorous reductions occur because less phosphorous needs to be blended into feed rations, resulting in a P source reduction
Precision and Prescribed Grazing	This practice utilizes a range of pasture management and grazing techniques to improve the quality and quantity of the forages grown on pastures and reduce the impact of animal travel lanes, animal concentration areas or other degraded areas. Pastures under the prescribed grazing systems are defined as having a vegetative cover of 60% or greater.
Soil and Water Conservation Plans	Farm conservation plans are a combination of agronomic, management and engineered practices that protect and improve soil productivity and water quality, and to prevent deterioration of natural resources on all or part of a farm. Plans can be prepared by staff working in conservation districts, natural resource conservation field offices or a certified private consultant. In all cases, the plan must meet technical standards.

Stream Access Control with Fencing	Stream access control with fencing involves excluding a strip of land with fencing along the stream corridor to provide protection from livestock. The fenced areas may be planted with trees or grass, or left to natural plant succession, and can be of various widths. To provide the modeled benefits of a functional riparian buffer, the width must be a minimum of 35 feet from top-of-bank to fence line. If an entity is installing a riparian buffer practice in conjunction with stream protection fencing, and can track and report these installations, additional upland benefits of those riparian buffers can be applied in the model. The implementation of stream fencing provides stream access control for livestock but does not necessarily exclude animals from entering the stream by incorporating limited and stabilized in-stream crossing or watering facilities. The modeled benefits of stream access control can be applied to degraded stream corridors in association with or without alternative watering facilities. They can also be applied in conjunction with or without pasture management systems.
Water Control Structure	Installing and managing boarded gate systems in agricultural land that contains surface drainage ditches.

Appendix C: Example output from APLE's estimated annual P loss for one land segment (Kent County, Delaware) for one land use (low-till with manure, LWM). Chart is divided into years' 1992 to 1999, and 2000 to 2005 for formatting purposes.

Year	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**
STP	136	136	136	136	136	136	136	136
Clay Percent	32.05	32.05	32.05	32.05	32.05	32.05	32.05	32.05
OM Percent	1.71	1.71	1.71	1.71	1.71	1.71	1.71	1.71
Degree Mix	38.00	39.00	36.00	32.00	30.00	36.00	38.00	34.00
Depth Incorp	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00
Depth1_IN	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Depth2_IN	7.00	7.00	7.00	7.00	7.00	7.00	7.00	7.00
Annual_ Rain_IN	39.33	42.40	47.93	41.12	61.06	41.61	41.40	46.82
Annual Runoff_IN	0.02	0.15	0.55	0.32	0.54	0.05	0.08	0.86
Sediment_ TONACRE	0.01	0.19	1.54	1.61	1.50	0.02	0.04	7.00
Liquid Manure_T P	1.27	1.30	1.34	1.37	1.41	1.44	1.39	1.34
Solid Manure_ TP	8.16	8.53	8.90	9.26	9.63	9.68	9.26	8.86
Fert_TP	16.04	15.85	15.67	15.49	15.31	15.21	15.14	15.05
Percent_ Incorp	40.00	30.00	38.00	33.00	40.00	39.00	32.00	37.00
Uptake_ LBAC	22.70	22.59	22.48	22.36	22.25	22.14	21.84	21.54

Year	1992	1993	1994	1995	1996	1997	1998	1999
Annual Rain_MM	999	1077	1218	1044	1551	1057	1052	1189
Annual Runoff_ MM	0.51	3.81	13.97	8.13	13.72	1.27	2.03	21.84
Sediment Loss	11.79	421.25	3441	3603	3362	41.10	99.91	15.7 E5
ER	4.87	1.99	1.18	1.16	1.19	3.56	2.85	1.00
Ratio	0.00	0.00	0.01	0.01	0.01	0.00	0.00	0.02
Manure_ PD	0.18	0.28	0.37	0.34	0.35	0.22	0.25	0.41
Fert_PD	0.03	0.03	0.04	0.03	0.04	0.03	0.03	0.04
Percent_ WEP_Solid	20.00	20.00	20.00	20.00	20.00	20.00	20.00	20.00
Percent WEP_ Liquid	42.00	42.00	42.00	42.00	42.00	42.00	42.00	42.00
Solid Manure_ WEP	1.63	1.71	1.78	1.85	1.93	1.94	1.85	1.77
Liquid Manure_ WEP	0.53	0.55	0.56	0.58	0.59	0.61	0.58	0.56
Solid Manure_ NonWEP	6.53	6.82	7.12	7.41	7.71	7.74	7.41	7.09

Year	1992	1993	1994	1995	1996	1997	1998	1999
Liquid Manure_ NonWEP	0.73	0.76	0.78	0.80	0.82	0.84	0.81	0.78
Percent Solids	86.00	86.00	86.00	86.00	86.00	86.00	86.00	86.00
Percent Liquids	14.00	14.00	14.00	14.00	14.00	14.00	14.00	14.00
Solid Manure WEP_Infiltrate	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Liquid ManureWEP_Infiltrate	0.32	0.33	0.34	0.35	0.35	0.36	0.35	0.34
Solid Manure NonWEP_ Infiltrate	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Liquid Manure NonWEP_ Infiltrate	0.44	0.45	0.47	0.48	0.49	0.50	0.48	0.47
Solid Slurry Factor	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00

Year	1992	1993	1994	1995	1996	1997	1998	1999
Liquid SlurryFactor	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65
Solid Manure WEP Incorp	0.65	0.51	0.68	0.61	0.77	0.75	0.59	0.66
Liquid ManureWEP_Incorp	0.09	0.07	0.09	0.08	0.09	0.09	0.07	0.08
Solid ManureNonWEP_Incorp	2.61	2.05	2.70	2.45	3.08	3.02	2.37	2.62
Liquid ManureNonWEP_Incorp	0.12	0.09	0.12	0.11	0.13	0.13	0.10	0.11
Solid ManureWEP_Surface	0.98	1.19	1.10	1.24	1.16	1.18	1.26	1.12
Liquid ManureWEP_Surface	0.13	0.15	0.14	0.15	0.14	0.15	0.16	0.14
Solid Manure NonWEP_Surface	3.92	4.77	4.41	4.97	4.62	4.72	5.04	4.46
Liquid Manure NonWEP_Surface	0.18	0.21	0.19	0.21	0.20	0.20	0.22	0.20
Solid Manure NonWEP_Mineralize	0.59	0.72	0.66	0.74	0.69	0.71	0.76	0.67

Year	1992	1993	1994	1995	1996	1997	1998	1999
Liquid Manure NonWEP_ Mineralize	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
Solid Manure WEP_ Available	1.57	1.91	1.76	1.99	1.85	1.89	2.02	1.79
Liquid Manure WEP_ Available	0.10	0.12	0.11	0.12	0.11	0.12	0.12	0.11
Total Manure_ NonWEP_ Mineralize	0.61	0.75	0.69	0.78	0.72	0.74	0.79	0.70
TotalManure_WEP_ Available	1.67	2.03	1.87	2.11	1.96	2.00	2.14	1.90
Manure OrgAddt1	0.38	0.40	0.42	0.45	0.47	0.46	0.43	0.42
Manure OrgAddt2	0.09	0.10	0.09	0.09	0.08	0.10	0.10	0.09
Fert Rate	16.04	15.85	15.67	15.49	15.31	15.21	15.14	15.05
FertAddt1	12.99	12.76	12.85	13.01	13.02	12.47	12.26	12.49
FertAddt2	3.05	3.09	2.82	2.48	2.30	2.74	2.88	2.56
Slope1	64.08	64.08	64.08	64.08	64.08	64.08	64.08	64.08
Slope2	64.08	64.08	64.08	64.08	64.08	64.08	64.08	64.08
Intercept1	294	294	294	294	294	294	294	294

Year	1992	1993	1994	1995	1996	1997	1998	1999
Soil_OC	0.99	0.99	0.99	0.99	0.99	0.99	0.99	0.99
Mixing	0.38	0.39	0.36	0.32	0.30	0.36	0.38	0.34
Initial Labile1_ MGKG	68.00	76.16	81.14	84.76	88.77	92.38	93.64	93.70
Initial Labile2_ MGKG	68.00	67.28	67.14	67.14	67.16	67.28	68.01	68.89
Soil_PSP1	0.28	0.28	0.29	0.29	0.30	0.30	0.30	0.30
Soil_PSP2	0.28	0.27	0.27	0.27	0.27	0.27	0.28	0.28
StableRatio 1	0.14	0.14	0.14	0.14	0.13	0.13	0.13	0.13
StableRatio 2	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14
Labile Ratio1	0.19	0.19	0.20	0.20	0.20	0.21	0.21	0.21
Labile Ratio2	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19
Labile Initial1	20.40	22.85	24.34	25.43	26.63	27.71	28.09	28.11
Labile Initial2	122.4	121.1	120.9	120.9	120.9	121.1	122.4	124.0
Active Initial1	53.66	60.34	64.32	67.07	69.90	72.20	72.57	71.92
Active Initial2	321.98	322.67	324.65	325.82	325.18	322.73	321.31	320.06
Stable Initial1	214.65	191.11	155.81	121.13	93.21	71.20	50.88	35.68
Stable Initial2	1288	852.59	530.68	294.55	136	51.76	26.52	17.91
Org Initial1	26.60	26.83	26.94	26.83	26.78	26.76	26.95	27.03
Org Initial2	159.6	159.36	159.22	159.02	158.8	158.5	158.5	158.5

Year	1992	1993	1994	1995	1996	1997	1998	1999
Uptake2_ Final	21.80	21.29	20.96	20.71	20.44	20.19	19.91	19.68
Uptake1_ Final	3.63	4.02	4.22	4.36	4.50	4.62	4.57	4.46
Labile_19	77.29	85.75	91.15	95.24	99.60	102.94	103.91	104.00
Labile_20	68.36	67.66	67.49	67.45	67.45	67.64	68.38	69.21
Labile_21	23.19	25.72	27.35	28.57	29.88	30.88	31.17	31.20
Labile_22	123.05	121.79	121.48	121.41	121.41	121.74	123.08	124.59
Leach_23	0.06	0.08	0.09	0.09	0.10	0.11	0.12	0.12
LeachFinal 1	0.06	0.08	0.09	0.09	0.10	0.11	0.12	0.12
Leach_24	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05
LeachFinal 2	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05
DRP_D25	0.37	0.49	0.62	0.59	0.97	0.72	0.73	0.81
DRP_E25	0.36	0.47	0.60	0.57	0.94	0.69	0.71	0.79
DRP_D26	0.23	0.25	0.28	0.24	0.35	0.24	0.25	0.28
TotalP_ KGHA_ Initial1	315.3	301.13	271.41	240.47	216.5	197.9	178.5	162.7
TotalP_ KGHA_ Initial2	1892	1455.73	1135.40	900.24	740.8	654.1	628.7	620.5

Year	1992	1993	1994	1995	1996	1997	1998	1999
TotalP_MGKG_Initial 1	1051.	1003.77	904.69	801.56	721.8	659.6	595	542.5
TotalP_MGKG_Initial 2	1051	808.74	630.78	500.13	411.6	363.4	349.3	344.7
Sediment_PLoss_KGHA	0.06	0.84	3.67	3.36	2.88	0.10	0.17	8.51
SoilDissolve_PLoss_KGHA	0.00	0.01	0.06	0.03	0.06	0.01	0.01	0.10
Manure_Disolve_PLoss_KGHA	0.00	0.00	0.01	0.01	0.01	0.00	0.00	0.01
Fertilizer_DissolvePLoss_KGHA	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
TotalDissolve_PLoss_KGHA	0.00	0.02	0.07	0.04	0.07	0.01	0.01	0.12
Total_PLoss_KGHA	0.06	0.86	3.74	3.41	2.95	0.10	0.18	8.64
NetInorg_Addt1	16.23	15.75	15.91	16.51	16.39	15.79	15.15	15.14
NetInorg_Addt2	-16.9	-16.17	-16.14	-16.32	-16.05	-15.11	-14.66	-15.09
Labile Erosion	0.00	0.06	0.33	0.36	0.35	0.01	0.03	1.47
Active Erosion	0.01	0.17	0.87	0.94	0.93	0.04	0.07	3.76
Stable Erosion	0.04	0.53	2.11	1.69	1.24	0.03	0.05	1.87
Organic Erosion	0.01	0.08	0.36	0.38	0.36	0.01	0.03	1.41

Year	1992	1993	1994	1995	1996	1997	1998	1999
Org Mineralize1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Org Mineralize2	0.47	0.45	0.45	0.46	0.45	0.42	0.41	0.42
Labile Final 1	24.25	26.64	27.98	29.25	30.48	31.80	32.02	30.59
Labile Final 2	120	118.55	118.30	118.27	118.3	118.7	120.1	121.6
Labile_MGKG_Final1	80.83	88.81	93.26	97.49	101.6	106.0	106.7	102.0
Labile_MGKG_Final2	66.51	65.86	65.72	65.70	65.75	65.95	66.72	67.56
Final Active1	63.79	69.91	73.23	76.24	78.96	81.74	81.67	77.32
Active Final2	319.2	319.06	319.67	318.85	316.0	312.1	310.3	308.5
Stable Final1	216.9	192.73	155.87	121.67	94.18	73.27	52.85	35.83
Stable Final2	826.8	493.76	259.82	107.51	28.78	4.12	0.74	0.12
Organic Final1	26.98	27.15	26.99	26.90	26.89	27.21	27.35	26.04
Organic Final2	159.2	159.00	158.86	158.65	158.4	158.2	158.2	158.2
TotalP_KGHA_Final1	331.9	316.44	284.07	254.06	230.5	214.0	193.9	169.8
TotalP_KGHA_Final2	1425	1090.37	856.64	703.28	621.5	593.2	589.3	588.3
TotalP_MGKG_Final1	1106	1054.79	946.89	846.88	768.4	713.4	646.3	565.9
TotalP_MGKG_Final2	791.7	605.76	475.91	390.71	345.3	329.6	327.4	326.8

Year	1992	1993	1994	1995	1996	1997	1998	1999
Final Labile1	22.85	24.34	25.43	26.63	27.71	28.09	28.11	27.58
Final Labile2	121.1	120.85	120.85	120.88	121.1	122.4	124.0	124.6
Final Active1	60.34	64.32	67.07	69.90	72.20	72.57	71.92	69.77
Final Active2	322.7	324.65	325.82	325.18	322.7	321.3	320.1	316.0
Final Stable1	191.1	155.81	121.13	93.21	71.20	50.88	35.68	25.40
Final Stable2	853.0	530.68	294.55	135.97	51.76	26.52	17.91	10.56
Final Organic1	26.83	26.94	26.83	26.78	26.76	26.95	27.03	26.13
Final Organic2	159.4	159.22	159.02	158.78	158.5	158.5	158.5	158.1
Final_SoilP1_KGHA	301.1	271.41	240.47	216.53	197.9	178.5	162.7	148.9
Final_SoilP2_KGHA	1456	1135.40	900.24	740.81	654.1	628.7	620.5	609.2
Final_SoilP1_MGKG	1004.	904.69	801.56	721.76	659.6	595.0	542.5	496.3
Final_SoilP2_MGKG	808.7	630.78	500.13	411.56	363.4	349.3	344.7	338.5
Final_Labile1_MGKG	76.16	81.14	84.76	88.77	92.38	93.64	93.70	91.93
Final_Labile2_MGKG	67.28	67.14	67.14	67.16	67.28	68.01	68.89	69.23

Year	2000	2001	2002	2003	2004	2005
STP	136	136	136	136	136	136
Clay Percent	32.05	32.05	32.05	32.05	32.05	32.05
OM Percent	1.71	1.71	1.71	1.71	1.71	1.71
Degree Mix	35.00	32.00	35.00	40.00	32.00	34.00
Depth Incorp	4.00	4.00	4.00	4.00	4.00	4.00
Depth1_IN	1.00	1.00	1.00	1.00	1.00	1.00
Depth2_IN	7.00	7.00	7.00	7.00	7.00	7.00
Annual_ Rain_IN	49.41	40.71	50.43	60.56	44.81	48.03
Annual Runoff_IN	1.34	0.26	0.16	1.27	0.32	0.14
Sediment_ TONACRE	6.71	0.79	0.30	0.79	1.27	0.13
Liquid Manure_ TP	1.29	1.23	1.19	1.14	1.09	1.03
Solid Manure_ TP	8.45	8.06	7.77	8.28	8.81	9.37
Fert_TP	14.96	14.86	14.76	14.87	15.00	15.13
Percent_ Incorp	34.00	37.00	33.00	30.00	31.00	31.00
Uptake_ LBAC	21.24	20.94	20.64	20.87	21.11	21.34

Year	2000	2001	2002	2003	2004	2005
Annual Rain_MM	1255	1034	1281	1538	1138	1220
Annual Runoff_ MM	34.04	6.60	4.06	32.26	8.13	3.56
Sediment Loss	15028	1775	678	1778	2839	296
ER	1.00	1.39	1.77	1.39	1.24	2.18
Ratio	0.03	0.01	0.00	0.02	0.01	0.00
Manure_ PD	0.44	0.32	0.27	0.42	0.33	0.27
Fert_PD	0.04	0.03	0.03	0.04	0.03	0.03
Percent_ WEP_Solid	20	20	20	20	20	20
Percent WEP_ Liquid	42	42	42	42	42	42
Solid Manure_ WEP	1.69	1.61	1.55	1.66	1.76	1.87
Liquid Manure_ WEP	0.54	0.52	0.50	0.48	0.46	0.43
Solid Manure_ NonWEP	6.76	6.44	6.22	6.62	7.05	7.49

Year	2000	2001	2002	2003	2004	2005
Liquid Manure_ NonWEP	0.75	0.71	0.69	0.66	0.63	0.60
Percent Solids	86	86	86	86	86	86
Percent Liquids	14	14	14	14	14	14
Solid Manure WEP_Infilt rate	0	0	0	0	0	0
Liquid Manure WEP_Infilt rate	0.32	0.31	0.30	0.29	0.27	0.26
Solid Manure NonWEP_ Infiltrate	0	0	0	0	0	0
Liquid Manure NonWEP_ Infiltrate	0.45	0.43	0.41	0.40	0.38	0.36
Solid Slurry Factor	1	1	1	1	1	1

Year	2000	2001	2002	2003	2004	2005
Liquid Slurry Factor	0.65	0.65	0.65	0.65	0.65	0.65
Solid Manure WEP Incorp	0.57	0.60	0.51	0.50	0.55	0.58
Liquid ManureWEP_Incorp	0.07	0.08	0.07	0.06	0.06	0.05
Solid ManureNonWEP_Incorp	2.30	2.38	2.05	1.99	2.18	2.32
Liquid ManureNonWEP_Incorp	0.10	0.11	0.09	0.08	0.08	0.07
Solid ManureWEP_Surface	1.12	1.01	1.04	1.16	1.22	1.29
Liquid ManureWEP_Surface	0.14	0.13	0.13	0.13	0.13	0.12
Solid Manure NonWEP_Surface	4.46	4.06	4.17	4.64	4.86	5.17
Liquid Manure NonWEP_Surface	0.20	0.18	0.18	0.19	0.17	0.17
Solid Manure NonWEP_Mineralize	0.67	0.61	0.63	0.70	0.73	0.78

Year	2000	2001	2002	2003	2004	2005
Liquid Manure NonWEP_ Mineralize	0.03	0.03	0.03	0.03	0.03	0.02
Solid Manure WEP_ Available	1.79	1.62	1.67	1.85	1.94	2.07
Liquid Manure WEP_ Available	0.11	0.10	0.11	0.11	0.10	0.09
Total Manure_ NonWEP_ Mineralize	0.70	0.64	0.65	0.72	0.76	0.80
TotalManure_WEP_ Available	1.90	1.73	1.77	1.96	2.04	2.16
Manure OrgAddt1	0.40	0.39	0.37	0.38	0.42	0.43
Manure OrgAddt2	0.09	0.07	0.08	0.09	0.08	0.09
Fert Rate	14.96	14.86	14.76	14.87	15.00	15.13
FertAddt1	12.34	12.48	12.17	11.89	12.60	12.56
FertAddt2	2.62	2.38	2.58	2.97	2.40	2.57
Slope1	64.08	64.08	64.08	64.08	64.08	64.08
Slope2	64.08	64.08	64.08	64.08	64.08	64.08
Intercept	293.90	293.90	293.90	293.90	293.90	293.90

Year	2000	2001	2002	2003	2004	2005
Soil_OC	0.99	0.99	0.99	0.99	0.99	0.99
Mixing	0.35	0.32	0.35	0.40	0.32	0.34
Initial Labile1_ MGKG	91.93	90.37	93.11	94.02	92.71	95.35
Initial Labile2_ MGKG	69.23	69.51	69.73	70.28	71.10	71.37
Soil_PSP1	0.30	0.30	0.30	0.30	0.30	0.30
Soil_PSP2	0.28	0.28	0.28	0.28	0.28	0.28
StableRatio 1	0.13	0.13	0.13	0.13	0.13	0.13
StableRatio 2	0.14	0.14	0.14	0.14	0.14	0.14
Labile Ratio1	0.21	0.20	0.21	0.21	0.21	0.21
Labile Ratio2	0.19	0.19	0.19	0.19	0.19	0.19
Labile Initial1	27.58	27.11	27.93	28.21	27.81	28.61
Labile Initial2	124.61	125.11	125.52	126.51	127.99	128.46
Active Initial1	69.77	67.83	69.14	68.96	67.07	68.17
Active Initial2	316.00	311.59	306.85	303.86	302.30	297.71
Stable Initial1	25.40	18.26	14.61	11.50	8.69	7.73
Stable Initial2	10.56	7.83	5.52	4.93	4.53	2.92
Org Initial1	26.13	25.54	25.85	26.11	26.20	26.22
Org Initial2	158.06	157.40	156.93	156.60	156.38	156.11

Year	2000	2001	2002	2003	2004	2005
Uptake2_ Final	19.49	19.29	18.92	19.13	19.43	19.56
Uptake1_ Final	4.31	4.18	4.21	4.26	4.22	4.35
Labile_19	101.90	100.24	102.72	103.59	102.97	105.83
Labile_20	69.55	69.80	70.05	70.65	71.41	71.70
Labile_21	30.57	30.07	30.82	31.08	30.89	31.75
Labile_22	125.19	125.64	126.08	127.17	128.53	129.05
Leach_23	0.11	0.11	0.11	0.12	0.11	0.12
LeachFinal 1	0.11	0.11	0.11	0.12	0.11	0.12
Leach_24	0.05	0.05	0.05	0.05	0.05	0.05
LeachFinal 2	0.05	0.05	0.05	0.05	0.05	0.05
DRP_D25	0.81	0.66	0.86	1.04	0.77	0.89
DRP_E25	0.78	0.63	0.84	1.01	0.74	0.86
DRP_D26	0.29	0.25	0.31	0.37	0.28	0.31
TotalP_ KGHA_ Initial1	148.88	138.75	137.53	134.78	129.77	130.73
TotalP_ KGHA_ Initial2	609.22	601.92	594.81	591.91	591.20	585.20

Year	2000	2001	2002	2003	2004	2005
TotalP_MGKG_Initial1	496.25	462.49	458.43	449.26	432.55	435.78
TotalP_MGKG_Initial2	338.46	334.40	330.45	328.84	328.44	325.11
Sediment_PLoss_KGHA	7.46	1.14	0.55	1.11	1.52	0.28
SoilDissolve_PLoss_KGHA	0.16	0.03	0.02	0.15	0.04	0.02
Manure_Disolve_PLoss_KGHA	0.02	0.00	0.00	0.02	0.00	0.00
Fertilizer_DissolvePLoss_KGHA	0.01	0.00	0.00	0.01	0.00	0.00
TotalDissolve_PLoss_KGHA	0.19	0.04	0.02	0.18	0.05	0.02
Total_PLoss_KGHA	7.65	1.18	0.57	1.29	1.56	0.30
NetInorg_Addt1	14.66	15.02	14.10	13.57	15.46	15.50
NetInorg_Addt2	-14.95	-15.15	-14.34	-13.91	-15.11	-14.78
Labile Erosion	1.38	0.22	0.11	0.23	0.33	0.06
Active Erosion	3.49	0.56	0.28	0.57	0.78	0.15
Stable Erosion	1.27	0.15	0.06	0.09	0.10	0.02
Organic Erosion	1.31	0.21	0.10	0.22	0.31	0.06

Year	2000	2001	2002	2003	2004	2005
Org Mineralize1	0.00	0.00	0.00	0.00	0.00	0.00
Org Mineralize2	0.42	0.43	0.40	0.39	0.43	0.42
Labile Final 1	30.00	30.76	31.49	31.52	31.51	32.61
Labile Final 2	122.22	122.69	123.23	124.28	125.56	126.08
Labile_MGKG_Final1	100.00	102.54	104.98	105.06	105.03	108.70
Labile_MGKG_Final2	67.90	68.16	68.46	69.05	69.75	70.04
Final Active1	75.17	76.40	77.41	76.60	75.66	77.39
Active Final2	304.25	299.59	295.41	292.76	290.23	285.85
Stable Final1	26.09	20.13	16.43	13.22	10.65	9.78
Stable Final2	0.00	0.00	0.00	0.00	0.00	0.00
Organic Final1	25.23	25.72	26.11	26.27	26.30	26.60
Organic Final2	157.72	157.05	156.60	156.30	156.03	155.78
TotalP_KGHA_Final1	156.48	153.01	151.45	147.61	144.12	146.38
TotalP_KGHA_Final2	584.19	579.33	575.23	573.35	571.81	567.71
TotalP_MGKG_Final1	521.61	510.05	504.84	492.05	480.41	487.94
TotalP_MGKG_Final2	324.55	321.85	319.57	318.53	317.67	315.40

Year	2000	2001	2002	2003	2004	2005
Final Labile1	27.11	27.93	28.21	27.81	28.61	29.23
Final Labile2	125.11	125.52	126.51	127.99	128.46	129.46
Final Active1	67.83	69.14	68.96	67.07	68.17	68.72
Final Active2	311.59	306.85	303.86	302.30	297.71	294.52
Final Stable1	18.26	14.61	11.50	8.69	7.73	6.93
Final Stable2	7.83	5.52	4.93	4.53	2.92	2.85
Final Organic1	25.54	25.85	26.11	26.20	26.22	26.41
Final Organic2	157.40	156.93	156.60	156.38	156.11	155.97
Final_SoilP1_KGHA	138.75	137.53	134.78	129.77	130.73	131.30
Final_SoilP2_KGHA	601.92	594.81	591.91	591.20	585.20	582.80
Final_SoilP1_MGKG	462.49	458.43	449.26	432.55	435.78	437.65
Final_SoilP2_MGKG	334.40	330.45	328.84	328.44	325.11	323.78
Final_Labile1_MGKG	90.37	93.11	94.02	92.71	95.35	97.44
Final_Labile2_MGKG	69.51	69.73	70.28	71.10	71.37	71.92

Appendix D: Soil test P values (Mehlich 3-P equivalent, mg kg^{-1}) assumed for each simulated land segment as inputs for Year 1 of the APLE simulation. Method of deriving the values is described in Chapter 4, 4.3.2.

Delaware

A10001	Kent	136
A10003	New Castle	70
A10005	Sussex	186

Maryland

A24001	Allegany	87
A24003	Anne Arundel	140
A24005	Baltimore	104
A24009	Calvert	150
A24011	Caroline	153
A24013	Carroll	98
A24015	Cecil	94
A24017	Charles	138
A24019	Dorchester	133
A24021	Frederick	90
A24023	Garrett	67
A24025	Harford	116
A24027	Howard	45
A24029	Kent	124
A24031	Montgomery	86
A24033	Prince George	123
A24035	Queen Anne's	101
A24037	St Mary's	116
A24039	Somerset	164
A24041	Talbot	96
A24043	Washington	111
A24045	Wicomico	164
A24047	Worcester	183

New York

A36003	Allegany	42
A36007	Broome	46
A36015	Chemung	55
A36017	Chenango	56
A36023	Cortland	52
A36025	Delaware	52

A36043	Herkimer	41
A36051	Livingston	66
A36053	Madison	47
A36065	Oneida	41
A36067	Onondaga	46
A36069	Ontario	74
A36077	Otsego	39
A36095	Schoharie	45
A36097	Schuyler	39
A36101	Steuben	44
A36107	Tioga	51
A36109	Tompkins	63
A36123	Yates	53

Pennsylvania

A42001	Adams	70
A42009	Bedford	69
A42011	Berks	96
A42013	Blair	76
A42015	Bradford	80
A42021	Cambria	58
A42023	Cameron	20
A42025	Carbon	74
A42027	Centre	78
A42029	Chester	68
A42033	Clearfield	47
A42035	Clinton	45
A42037	Columbia	90
A42041	Cumberland	94
A42043	Dauphin	86
A42047	Elk	39
A42055	Franklin	71
A42057	Fulton	68
A42061	Huntingdon	63
A42063	Indiana	45
A42065	Jefferson	38
A42067	Juniata	71
A42069	Lackawanna	51
A42071	Lancaster	144
A42075	Lebanon	82
A42079	Luzerne	106

A42081	Lycoming	72
A42083	McKean	54
A42087	Mifflin	86
A42093	Montour	88
A42097	Northumberland	94
A42099	Perry	72
A42105	Potter	71
A42107	Schuylkill	120
A42109	Snyder	81
A42111	Somerset	54
A42113	Sullivan	69
A42115	Susquehanna	86
A42117	Tioga	61
A42119	Union	74
A42127	Wayne	131
A42131	Wyoming	106
A42133	York	77

Virginia

A51001	Accomack	113
A51003	Albemarle	34
A51005	Alleghany	42
A51007	Amelia	55
A51009	Amherst	31
A51011	Appomattox	33
A51015	Augusta	58
A51017	Bath	42
A51019	Bedford	35
A51023	Botetourt	47
A51029	Buckingham	41
A51031	Campbell	44
A51033	Caroline	64
A51036	Charles City	43
A51041	Chesterfield	91
A51043	Clarke	42
A51045	Craig	43
A51047	Culpepper	38
A51049	Cumberland	59
A51053	Dinwiddie	78
A51057	Essex	48

A51059	Fairfax	56
A51061	Fauquier	49
A51065	Fluvanna	48
A51069	Frederick	44
A51071	Giles	47
A51073	Gloucester	57
A51075	Goochland	43
A51079	Greene	43
A51085	Hanover	58
A51087	Henrico	101
A51091	Highland	36
A51093	Isle Of Wight	72
A51095	James City	66
A51097	King & Queen	50
A51099	King George	41
A51101	King William	64
A51103	Lancaster	50
A51107	Loudoun	39
A51109	Louisa	35
A51113	Madison	43
A51115	Mathews	58
A51119	Middlesex	54
A51121	Montgomery	46
A51125	Nelson	36
A51127	New Kent	60
A51131	Northampton	136
A51133	Northumberland	52
A51135	Nottoway	71
A51137	Orange	35
A51139	Page	72
A51145	Powhatan	48
A51147	Prince Edward	46
A51149	Prince George	62
A51153	Prince William	41
A51157	Rappahannock	35
A51159	Richmond	60
A51161	Roanoke	51
A51163	Rockbridge	44
A51165	Rockingham	77
A51171	Shenandoah	66

A51177	Spotsylvania	39
A51179	Stafford	40
A51181	Surry	59
A51187	Warren	39
A51193	Westmoreland	57
A51199	York	60
A51550	Chesapeake (city)	84
A51800	Suffolk (city)	72
A51810	Virginia Beach (city)	93

West Virginia

A54003	Berkeley	75
A54023	Grant	86
A54027	Hampshire	50
A54031	Hardy	138
A54037	Jefferson	102
A54057	Mineral	83
A54063	Monroe	98
A54065	Morgan	45
A54071	Pendleton	155
A54077	Preston	86
A54093	Tucker	50

Bibliography

Abdi, D., BJ Cade-Menun, N. Ziadi, and L. Parent. 2014. Long-term impact of tillage practices and phosphorus fertilization on soil phosphorus forms as determined by ^{31}P nuclear magnetic resonance spectroscopy. Journal of Environmental Quality 2014 43: 4: 1431-1441

Andraski, T. and L. Bundy. 2003. Relationships between phosphorus levels in soil and in runoff from corn production systems. Journal Environmental Quality 32: 310-316.

Arnold, J. G., D.N. Moriasi, P.W. Gassman, K.C. Abbaspour, M.J. White, R.Srinivasan, C. Santhi, R.D. Harmel, A. van Griensven, M.W. Van Liew, N. Kannan, and M.K. Jha. 2012 SWAT: model use, calibration, and validation. Trans. ASABE. 55: 1491-1508.

American Society of Agricultural Engineers. 2003. Manure Production and Characteristics. ASAE D384.1 FEB03

Ator, S.W., J.W. Brakebill, and J.D. Blomquist. 2011. Sources, fate, and transport of nitrogen and phosphorus in the Chesapeake Bay watershed: An empirical model. U.S. Geological Survey Scientific Investigations Report 2011-5167.

Beaulac, M.N., and K.H. Reckhow. 1982. An examination of land use – nutrient export relationships. Water Resources Bulletin 18:1013-1024.

Bicknell, B.R., J.C. Imhoff, J.L. Kittle, Jr., T.H. Jobes, and A.S. Donigian, Jr. 2001. Hydrological Simulation Program - Fortran (HSPF): User's manual for release 12. U.S. Environmental Protection Agency. Athens, GA.

Boes, D. C., F.A. Graybill, and A.M. Mood. 1974. Introduction to the Theory of Statistics, 3rd ed. New York: McGraw-Hill.

Boesch, D.F., R.B. Brinsfield and R.E. Magnien. 2001. Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration, and challenges for agriculture. Journal of Environmental Quality 30(2): 303-320.

Boynton, W.R., J.H. Garber, R. Summers, and W.M. Kemp. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. Estuaries 18: 285–314.

Brady, N.C.and R.R. Weil. 2002. The Nature and Properties of Soil: 13th edition. Prentice Hall, Upper Saddle River, NJ.

Brandt, R.C., H.A. Elliott, and G. O’Connor. 2004. Water extractable phosphorus in biosolids: Implications for land-based recycling. Water Environment Research 76(2): 121-129.

Buda, A.R., P.J.A. Kleinman, R.B. Bryant, G.W. Feyereisen and M.S. Srinivasan. 2009. Effects of hydrology and field management on phosphorus transport in surface runoff. Journal of Environmental Quality 38: 2273-2284.

Butler and Coale. 2005. Phosphorus leaching in manure amended Atlantic Coastal Plain soils. J. Environ. Qual. 34: 370-381.

Chesapeake Bay Commission. 2012. Nutrient credit trading for the Chesapeake Bay: An economic study.

Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology - Progress Series 210: 223-253

Coale, F.J. 2001. Converting among soil test analyses frequently used in Maryland. Soil Fertility Management Information, SFM-4. University of Maryland Extension, College Park, MD.

Cox, F. and S. Hendricks. 2000. Soil test phosphorus and clay content effects on runoff water quality. Journal Environmental Quality 29: 1582-1586.

Curtis, J.L., W.L. Kingery, M.S. Cox and Z. Liu. 2010. Distribution and dynamics of phosphorus in an agricultural watershed with a long-term history of poultry waste application. Communications in Soil Science and Plant Analysis 41: 2057-2074.

Devereux, O.H. and J.R. Rigelman. 2011. Chesapeake Assessment Scenario Tool (CAST). Devereux Environmental Consulting, Inc. and J7, LLC for U.S. Environmental Protection Agency, Region 3. accessed August 2014, at URL http://casttool.org/

Donigian, A.S., B.R. Bicknell, A.S. Patwardham, L.C. Linker, C. Chang and R. Reynolds. 1994. Chesapeake Bay Program: Watershed model application to calculate bay nutrient loadings.U.S. EPA-903-R-94–042. Chesapeake Bay Program Office: Annapolis, Maryland

Ekholm, P., E. Turtola, J. Gronroos, P. Seuri, and K. Ylivaino. 2005. Phosphorus loss from different farming systems estimated from soil surface phosphorus balance. Agriculture, Ecosystems, and Environment 110: 266-278.

Elliott, H.A., R.C. Brandt, and G. O'Connor. 2005. Runoff phosphorus losses from surface-applied biosolids. Journal Environmental Quality 34:1632-1639.

Executive Order No. 13508. 74 FR 23099, May 15, 2009.

Gartley, K.L., J.T. Sims, C.T. Olsen, and P. Chu. 2002. Comparison of soil test extractants used in mid-Atlanitc United States. Communications in Soil Science and Plant Analysis 33(5-6): 873-895.

Harmel, D., S. Potter, P. Casebolt, K. Reckhow, C. Green, and R. Haney. 2006. Compilation of measured nutrient load data for agricultural land uses in the United States. J. Amer. Water Resources Association. 42(5):1163-1178.

He, Z.L., M. K. Zhang, P. J. Stoffella, X. E. Yang 2006. Phosphorus concentrations and loads in runoff water under crop production. SSSA 70: 1807-1816..

Hecky, R.E. and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. Limnology and Oceanography 33: 796-822.

Jones, C.A., C.V. Cole, A. Sharpley, and J.R. Williams. 1984. A simplified soil and plant phosphorus model: Documentation. SSSA 48: 800-805.

Kleinman, P.J., A.N. Sharpley, B.G. Moyer and G.F. Elwinger. 2002. Effect of mineral and manure phosphorus sources on runoff phosphorus. Journal of Environmental Quality 31: 2026-2033.

Kleinman, P.J., A. Wolf, A. Sharpley, D. Beegle, and L. Saporito. 2005. Survey of water extractable phosphorus in livestock manures. SSSA 69: 701-708.

Kleinman, P.J.A., A.L. Allen, B.A. Needelman, A.N. Sharpley, P.A. Vadas, L.S. Saporito, G.J. Folmar, and R.B. Bryant. 2007. Dynamics of phosphorus transfers from heavily manured Coastal Plain soils to drainage ditches. Journal of Soil and Water Conservation 62(4): 225-234.

Langland, M., and T. Cronin. 2003. A summary report of sediment processes in the Chesapeake Bay and watershed. Water-Resources Investigations Report 03-4123. U.S. Geological Survey, New Cumberland, PA.

Lookman, R., D. Freese, R. Merckx, K. Vlassak, and W. Van Riemsdijk. 1995. Long-term kinetics of phosphate release from soil. Environmental Science and Technology 29:1569-1575.

Maguire, R. and J. Sims. 2002. Measuring agronomic and environmental soil phosphorus saturation and predicting phosphorus leaching with Mehlich 3.SSSA 66: 2033-2039.

Martucci, S.K., J.L. Krstolic, J.P. Raffensperger, and K.J. Hopkins. 2006. Development of land segmentation, stream-reach network, and watersheds in support of Hydrologic Simulation Program-Fortran (HSPF) modeling, Chesapeake Bay watershed, and adjacent parts of Maryland, Delaware, and Virginia. U.S. Geological Survey, Reston, VA.

McDowell, R., A.N. Sharpley, and G.J. Folmar. 2001. Phosphorus export from an agricultural watershed: linking source and transport mechanisms. Journal of Environmental Quality 30: 1587–1595.

McDowell, R.W. and A. N. Sharpley. 2002. Phosphorus transport in overland flow in response to position of manure application. Journal of Environmental Quality 31: 217–227.

McDowell, R.W.and A.N. Sharpley. 2003. Uptake and release of phosphorus from overland flow in a stream environment. Journal of Environmental Quality 32: 937–948.

McDowell, R., A.N. Sharpley, and G.J. Folmar. 2001. Phosphorus export from an agricultural watershed: linking source and transport mechanisms. Journal of Environmental Quality 30: 1587–1595.

McDowell, R.W., B.J.F. Biggs, A.N. Sharpley and L. Nguyen. 2004. Connecting phosphorus loss from agricultural landscapes to surface water quality. Chemistry and Ecology 20(1): 1-40.

McGrath, J.M., J.T. Sims, R.O. Maguire, W.W. Saylor, C.R.Angel, and B. Turner. 2005. Broiler diet modification and litter storage: Impacts on phosphorus in litters, soils, and runoff. Journal Environmental Quality 34: 1896-1909.

NRCS. 2006. Conservation Resource Brief: Soil Erosion, number 0602. Accessed November 13, 2014.

Owens, L.B. and M. J. Shipitalo. 2006. Surface and subsurface phosphorus losses from fertilized pasture systems in Ohio. Journal Environmental Quality 35:1101 – 1109.

Pagliari, P. and C. Laboski. 2012. Investigation of the inorganic and organic phosphorus forms in animal manure. Journal Environmental Quality 41: 901-910..

Palace, M.W., J.E. Hannawald, L.C. Linker, G.W. Shenk, J.M. Storrick, and M.L. Clipper. 1998. Chesapeake Bay Watershed Model Application and Calculation of Nutrient and Sediment Loadings. Appendix H: Tracking Best Management Practice Nutrient Reductions in the Chesapeake Bay Program. EPA 903-R-98-009, CBP/TRS 201/98. U.S. Environmental Protection Agency Chesapeake Bay Program, Annapolis, MD.

Penn, C. and J. Sims. 2002. Phosphorus forms in biosolids amended soils and losses in runoff: Effects of wastewater treatment process. Journal Environmental Quality 31: 1349-1361.

Phillips, S.W., editor. 2007. Synthesis of U.S. Geological Survey science for the Chesapeake Bay ecosystem and implications for environmental management. U.S. Geological Survey Circular 1316.

Pionke, H.B., W.J.Gburek, and A.N. Sharpley. 2000. Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. Ecological Engineering 14(4): 325-335.

Public law 92-500. 1972. Federal Water Pollution Control Act Amendment.

Radcliffe, D. J. Freer, and O. Schoumans. 2009. Diffuse phosphorus models in the United States and Europe: Their usages, scales, and uncertainties. Journal Environmental Quality 38: 1956-1967.

Sharpley, A., C. Jones, C. Gray, and C. Cole. 1984. A simplified soil and plant phosphorus model II: Prediction of labile, organic, and sorbed phosphorus. SSSA 48: 805-809.

Sharpley, A.N. 1995. Dependence of runoff phosphorus on extractable soil phosphorus. Journal Environmental Quality 24: 920-926.

Sharpley, A.N. and B. Moyer. 2000. Phosphorus forms in manure and compost and their release during simulated rainfall. Journal of Environmental Quality 29: 1462-1469.

Sharpley, A.N., P.J. Kleinman, R.W. McDowell, M. Gitau, and R.B. Bryant. 2002. Modeling phosphorus transport in agricultural watersheds" Processes and possibilities. Journal Soil and Water Conservation 57(6): 425-439.

Sharpley, A.N, editor. 2000. Agriculture and Phosphorus Management: the Chesapeake Bay. CRC Press, Boca Raton, FL.

Sharpley, A., P. Kleinman and R. McDowell. 2001. Innovative management of agricultural phosphorus to protect soil and water resources.. Communications in Soil Science and Plant Analysis 32: 7-8.

Sims, J.T. 1998. Phosphorus soil testing: Innovations for water quality protection. Communications Soil Science and Plant Analysis. 29(11-14): 1471-1489.

Sims, J.T. 1989. Comparison of Mehlich 1 and Mehlich 3 extractants for P, K, Ca, Mg, Mn, Cu, and Zn in Atlantic Coastal Plain soils. Communications Soil Science and Plant Analysis 20: 1707–1726.

Soileau, J.M., J.T. Touchton, B.F. Hajek, and K.H. Yoo. 1994. Sediment, nitrogen, and phosphorus runoff with conventional-and conservation-tillage cotton in a small watershed. Journal Soil and Water Conservation 49(1) 82-89.

Stevenson, F.J., and Cole, M.A. 1999. Cycles of soils: carbon, nitorgen, phosphorus, sulfur, micronutrients. John Wiley & Sons, Canada.

Torbert, H., T. Daniel, J. Lemunyon, and R. Jones. 2002. Realtionship of soil test phosphorus and sampling depth to runoff phosphorus in calcareous and noncalcareous soils. Journal Environmental Quality 31: 1380-1387.

USEPA. 1999. Protocol for developing nutrient TMDLs. EPA 841-B-99-007. Office of Water (4503F), Washington D.C.

USEPA. 2010a. Final Chesapeake Bay Total Maximum Daily Load. Accessed on web September 2, 2014 at http://www.epa.gov/reg3wapd/tmdl/ChesapeakeBay/tmdlexec.html

USEPA. 2010b. Chapter 2: agriculture. *In* Guidance for Federal land management in the Chesapeake Bay watershed. EPA 841-R-10-002. Office of Wetlands, Oceans and Watersheds, Washington D.C. 247 pages.

USEPA. 2010c. Chesapeake Bay phase 5.3 community watershed model. EPA 903S10002 - CBP/TRS-303-10. Chesapeake Bay Program Office, Annapolis MD.

USEPA. 2012. Estimates of county level nitrogen and phosphorus data for use in modeling pollutant reduction: Documentation for Scenario Builder, version 2.4. Chesapeake Bay Program Office, Annapolis MD.

Vadas, P., T. Krogstad, and A. Sharpley. 2006. Modeling phosphorus transfer between labile and nonlabile soil pools: Updating the EPIC model. SSSA 70: 736-743.

Vadas, P.A. and M.J. White. 2010. Validating soil phosphorus routines in the SWAT model. American Society of Agricultural and Biological Engineers 53(5): 1469-1476.

Vadas, P. P. Kleinman, A. Sharpley, and B. Turner. 2005. Relating soil phosphorus to dissolved phosphorus in runoff: A single extraction coefficient for water quality modeling.Journal Environmental Qaulity 34: 572-580.

Vadas, P., B. Haggard, and W. Gburek. 2005b. Predicting dissolved phosphorus in runoff from manured field plots. Journal Environmental Qaulity 34: 1347-1353.

Vadas, P.A., L.W. Good, P.A. Moore, and N. Widman. 2009. Estimating phosphorus loss in runoff from manure and fertilizer for a phosphorus loss quantification tool. Journal of Environmental Quality 38: 1645-1653.

Vadas, P., L. Owens, and A. Sharpley. 2008. An empirical model for dissolved phosphorus in runoff from surface applied fertilizers. Agriculture, Ecosystems, and Environment 127: 59-65.

Vadas, P., P. Kleinman, and A. Sharpley. 2004. A simple method to predict dissolved phosphorus in runoff from surface applied manures.Journal Environmental Qaulity 33: 749-756.

Vadas, P.A., W.J. Gburek, A.N. Sharpley, P.J. Kleinman, P.A. Moore, Jr., M.L. Cabrera, and R.D. Harmel. 2007. A model for phosphorus transformation and runoff loss for surface-applied manures. Journal of Environmental Quality 36: 324-332.

Vadas, P., C. Bolster, and L. Good. 2013. Critical evaluation of models used to study agricultural phosphorus and water quality. Soil Use and Management 29: 36-44.

Vadas, P. 2013. Annual Phosphorus Loss Estimator, version 2.4 Theoretical Documentation.

Wainger, L., G. Van Houtven, R. Loomis, J. Messer, R. Beach, and M. Deerhake. 2013. Tradeoffs among ecosystem services, performance certainty, and cost-efficiency in implementation of the Chesapeake Bay TMDL. Agricultural and Resource Economics Review 42: 196-224.

White, J.W., F.J. Coale, J.T. Sims, and A.L. Shober. 2010. Phosphorus runoff from waste water treatment biosolids and poultry litter applied to agricultural soils. Journal of Environmental Quality 39:314–323.

Whithers, P.J.A., S.D. Clay, and V.G. Breeze. 2001. Phosphorus transfer in runoff following application of fertilizer, manure, and sewage sludge. Journal of Environmental Quality 30: 180–188.

Wieland, R., D. parker, W. Gans, and A. Martin. 2009. Costs and cost efficiencies for some nutrient reduction practices in Maryland. Prepared for NOAA, Chesapeake Bay Program Office, and Maryland Department Natural Resources.

Williams, J.R., C.A. Jones, and P.T. Dyke. 1990. The EPIC model. *In* A.N. Sharpley and J.R. Williams (ed.) EPIC—Erosion/Productivity Impact Calculator 1. Model Documentation. U.S. Department of Agriculture, Agricultural Research Service, Technical Bulletin, pp. 3–92.