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Ecosystem Services and Environmental Markets in Chesapeake Bay Restoration

Final Report

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NOTICE

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ABSTRACT

This report contains two separate analyses, both of which make use of an optimization framework previously developed to evaluate trade-offs in alternative restoration strategies to achieve the Chesapeake Bay Total Maximum Daily Load (TMDL). The first analysis expands on model applications that examine how incorporating selected co-benefits of nutrient reductions into the optimization framework alters the optimal distribution of nutrient reductions in the watershed (U.S. EPA, 2011). In previous applications, the analyzed co-benefits included carbon sequestration and recreational hunting benefits from certain agricultural best management practices (BMPs). In this report we expand the optimization framework to also include benefits from water quality improvements in freshwater river and streams. We find that these nontidal water quality co-benefits are larger than the other co-benefits combined and would result in greater nutrient control efforts in upstream portions of the watershed. Compared to cost-minimization results that do not account for co-benefits, including all co-benefits in the optimization would increase annual nutrient control costs by $16 million in the Susquehanna River Basin in Pennsylvania; however, the co-benefits would increase by $31 million, for a net gain of $15 million per year. In the James River Basin in Virginia, considering monetized co-benefits results in an estimated increase in nutrient control costs of $17 million but an increase in co-benefits of $42 million (net gain of $25 million per year).

The second analysis expands on previous applications of the optimization framework that have focused on the potential cost savings from allowing nutrient trading in the Chesapeake Bay watershed (Van Houtven et al., 2012). These applications do not include the co-benefit estimates. Instead they examine how the costs of achieving TMDL goals could be reduced under alternative trading scenarios. For this report, we apply the optimization framework to assess how nutrient trading may interact with other incentives for agricultural nutrient reductions, as well as how simplified crediting of nutrient reductions influences the nutrient control costs, load reductions, and participation in a nutrient trading market. We estimate that nutrient trading can act as an incentive for some agricultural entities to adopt nutrient controls and meet their load allocation under the TMDL. However, we also find that the incentive of nutrient trading alone would only support achieving 11 percent of the required agricultural nitrogen load reductions in the Susquehanna River Basin in Pennsylvania and 4 percent of the required agricultural phosphorus reductions. In the James River Basin in Virginia, we estimate nutrient trading would be a more effective incentive to achieve the required agricultural nutrient reductions, with 35 percent of the nitrogen reduction and 41 percent of the phosphorus reduction achieved through nutrient trading.

Finally, we estimate that simplified crediting of nutrient reductions results in higher costs (by 8 percent across the watershed) for achieving significant wastewater and industrial discharge nutrient reductions through nutrient trading because it discourages placement of nutrient controls where they would be most effective. In addition, simplified crediting of nutrient trading is estimated to result in failure to meet the load reduction requirements in 11 of the 14 basin-state combinations in the Chesapeake Bay watershed due to practices in certain agricultural areas receiving more credit for nutrient reductions than would be achieved.

SECTION 1
INTRODUCTION

In 2010, EPA established the Chesapeake Bay Total Maximum Daily Load (TMDL), which limits the amount of nutrients and sediment that enter the largest estuary in the United States. The goal of these nutrient and sediment limits is to achieve water quality standards for dissolved oxygen (DO), water clarity, submerged aquatic vegetation (SAV) and chlorophyll-*a*. To meet the TMDL, sufficient controls must be in place by 2025 to reduce nitrogen, phosphorus, and sediment by 25 percent, 24 percent, and 20 percent respectively relative to 2009 conditions (U.S. EPA, 2010).

In the development of the TMDL, five of the seven jurisdictions within the Bay watershed agreed to a methodology to allocate needed load reductions based on their controllable load and relative effectiveness of improving DO in the Bay's main channel. Jurisdictions then developed watershed implementation plans (WIPs) describing what nutrient and sediment controls will be implemented to achieve their load allocation. The WIPs also provide reasonable assurances that controls for sources not regulated under the Clean Water Act, including much of the agricultural sector, will be implemented through regulatory or voluntary programs.

While all of the source controls described in the WIPs will reduce nutrients and/or sediment, some source controls provide additional environmental co-benefits. For example, in addition to retaining nutrients and sediment, a restored wetland provides multiple ecosystem services including water retention that reduces downstream flood risk, wildlife habitat that can improve recreational experiences such as wildlife watching and waterfowl hunting, and carbon sequestration to reduce the risk of damages from climate change.

In 2011, EPA issued a report evaluating how considering these co-benefits might change the costs and benefits of implementing alternative combinations of source controls to achieve the TMDL (U.S. EPA, 2011). To examine tradeoffs among alternative implementation strategies, an optimization framework was developed to estimate the least-cost combination of source controls needed to meet the TMDL and the least-net-cost combination of source controls, which was defined as the difference between monetized co-benefits and costs.

More recently, the optimization framework has been applied by the Chesapeake Bay Commission to analyze potential market-based approaches to support more cost-effective Bay restoration, such as nutrient trading (Van Houtven et al., 2012). Nutrient trading is a mechanism through which entities with high costs of nutrient reduction can purchase nutrient reduction credits from other entities that reduce nutrient loads beyond their required level. Setting aside the co-benefit estimates, Van Houtven et al. (2012) used the cost-minimization features of the optimization framework to simulate conditions under alternative nutrient trading scenarios and to estimate potential cost savings from trading.

The purpose of this report is to build on these two previous applications of the optimization framework through two separate analyses. In both cases, the purpose is to evaluate the economic trade-offs and implications of alternative restoration strategies for achieving the Chesapeake Bay TMDL; however, the first analysis examines the implications of expanding the types of co-benefits included in the framework, and the second analysis focuses on the implications of alternative incentive-based approaches for achieving the TMDL.

The main objective of the first analysis is to expand on the approach of U.S. EPA (2011) to include co-benefits from water quality improvements upstream from the Bay itself. The earlier analysis primarily valued co-benefits that were independent of the nutrient and sediment reductions, and thus omitted the potentially important co-benefits derived from water quality improvements in the tributaries and main stem of the Bay. To address this shortcoming, the current analysis incorporates estimates of the nontidal (i.e., freshwater river and stream) water quality improvement co-benefits associated with the TMDL. It incorporates these estimates into the optimization framework to investigate how their inclusion influences the optimal selection (based on economic considerations) of available projects to reduce nutrient and sediment loads to the Chesapeake Bay.

The main objective of the second analysis is to expand on the application in Van Houtven et al. (2012) by assessing (1) how nutrient trading may interact with other incentives for agricultural nutrient reductions and (2) how simplified crediting of nutrient reductions influences the control costs, load reductions, and participation in a nutrient trading market. The current analysis explores how nutrient trading, which would allow agricultural entities to sell credits for any load reductions that *exceed* their required reductions, can also provide an incentive for these sources to *meet* their required reductions. In other words, to what extent can nutrient trading help to achieve the dual objectives of reducing TMDL costs and providing incentives to meet TMDL load reduction requirements? We also evaluate how other incentive-based approaches, such as public funding for agricultural nutrient reductions, may interact with a nutrient trading market to achieve these objectives. In addition, the analysis explores how alternative nutrient crediting methodologies aimed at reducing transaction costs of offset and trading programs, such as uniformly crediting practices throughout a river basin, may impact water quality and the potential cost savings from nutrient trading.

The document is organized as follows:

- Section 2 describes the general optimization framework and input data used for the subsequent analyses of alternative policies and approaches for meeting the TMDL.
- Section 3 details how including the upstream water quality co-benefits of nutrient reductions within the optimization framework changes the economically optimal distribution of nutrient reductions in the watershed.
- Section 4 discusses the analysis of alternative designs for incentive-based systems to meet TMDL goals.
- Section 5 summarizes the main findings and limitations of the analysis and discusses implications.

SECTION 2
OPTIMIZATION FRAMEWORK

2.1 Model Overview

As noted in Section 1, the main purpose of the optimization framework is to identify the combinations of available nutrient control projects in the Chesapeake Bay watershed that together achieve the TMDL goals at lowest overall cost (or costs net of co-benefits) to society. In other words, within this framework, the optimal solution is one that minimizes the cost (or net costs) of nutrient controls, subject to the load reduction constraints defined by the TMDL.

Figure 2-1 provides a graphical representation of the main components of the optimization framework and how they are connected.[1] In this framework the first step is to define the total load reduction targets, which represent the main model constraints. For each defined geographic area (i.e., major basin and/or jurisdiction), total nutrient load reductions must be at least as large as those required by the TMDL. As described in more detail below, we defined separate load reduction targets for the two main source sector categories included in this analysis—significant wastewater treatment and industrial facilities and agricultural nonpoint sources—across the main basins and jurisdictions. These sources contribute more than half of the annual nutrient loads to the Bay. As a simplification, the analysis does not include load reduction targets or nutrient control projects for urban stormwater sources, septic systems, concentrated animal feeding operations, or nurseries. The implications of including these sources are left for investigation in future analyses.

The next step is to create an inventory of potential nutrient control projects. These projects include discrete upgrades at significant wastewater treatment and industrial facilities, as well as agricultural best management practices (BMPs) that can be adopted across the landscape. Next, each project must be assigned estimates of its annual nutrient load reductions to the Bay and its annual costs. As feasible, we also develop and assign ecosystem service co-benefit estimates (in monetary terms) to the projects. In previous model applications, these monetized co-benefits were primarily comprised of carbon sequestration and recreational hunting benefits, which are described in more detail below and in Appendix A. Combining the previous steps, the net costs of each project are calculated as the difference between its annual costs and its annual monetized co-benefits.

[1] A more detailed and technical discussion of the optimization model and framework is provided by Van Houtven et al. (2012). Additional details, particularly regarding co-benefit estimates, are provided by U.S. EPA (2011).

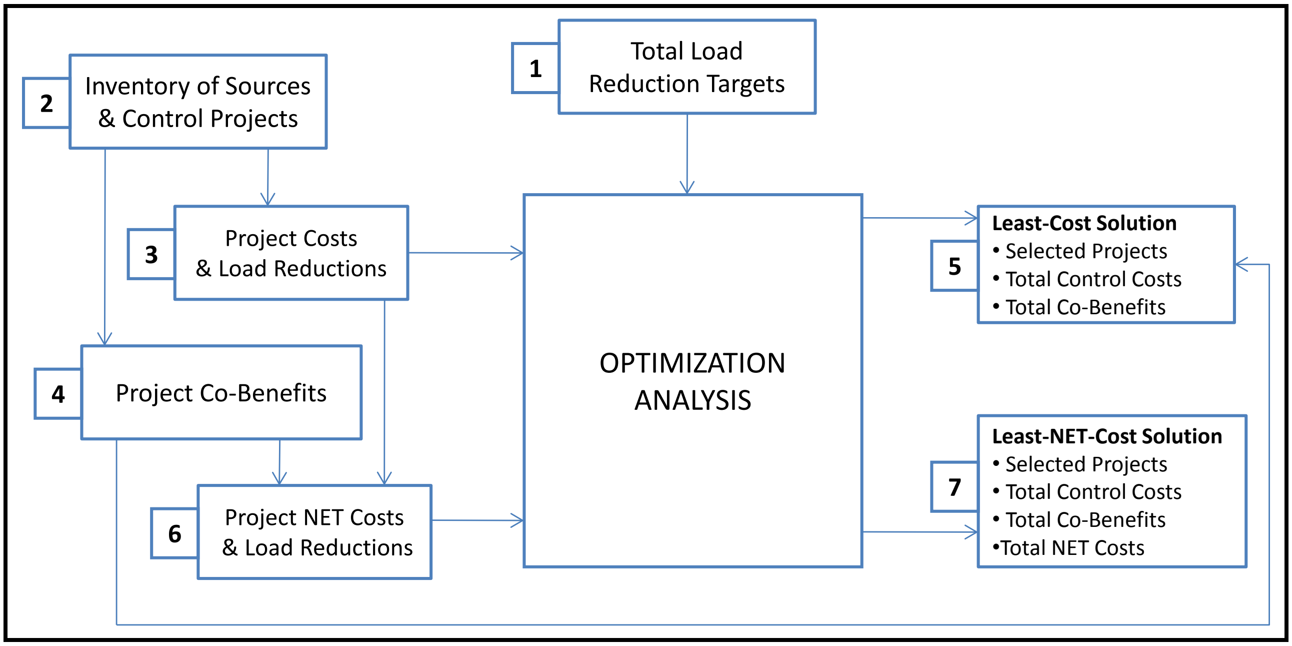

Figure 2-1. Overview of the Optimization Framework (Adapted from U.S.EPA, 2011)

The modeling framework for this analysis is built around and expands on the Chesapeake Bay Program's Phase 5.3.2 Watershed Model (CBWM).[1] CBWM subdivides the watershed into a network of 2,468 "land-river segments" and simulates the movement of nutrients through the network. Importantly, it serves as the main accounting framework for estimating compliance with the TMDL goals for nutrient and sediment loads.

The optimization component of our analytical framework is defined as a mixed integer linear programming (MILP) optimization problem. Because agricultural nonpoint source controls can be adopted continuously (i.e., on an acre-by-acre basis) across a landscape, these controls can be modeled using a linear programming approach. In contrast, upgrade options at significant wastewater treatment and industrial facilities are best characterized as discrete, all-or-nothing source control options; therefore, a binary integer decision variable is required to reflect their adoption. The MILP is solved using the CPLEX solver in the General Algebraic Modeling System.

As shown in Figure 2-1, the optimization model can be applied to solve for two general types of solutions—a least-cost solution or a least-net-cost solution. The only difference is that the least-cost solution does not consider co-benefits as part of the optimization process. They can be

[1] The 2011 analysis (U.S.EPA, 2011) was conducted using an earlier version of the CBWM.

estimated for the projects in the solution set, but they do not affect which projects are selected. In contrast, co-benefits are directly factored into the net costs which are minimized as part of the second type of solution.

For each solution type, the optimization model identifies (1) the optimal control technology for each point source and (2) the optimal number of additional acres of each BMP in each land-river segment. The resulting total costs, load reductions, and co-benefits for each area can then be calculated by summing across the selected projects.

Below we provide additional details about how the inputs for the optimization analysis are defined for point sources and agricultural nonpoint sources.

2.2 Significant Wastewater Treatment and Industrial Dischargers

2.2.1 Cost and Effectiveness of Nutrient Controls at Significant Wastewater Treatment and Industrial Dischargers

To represent point-source projects in our inventory, we developed estimates of nutrient reductions and annual costs of upgrades at significant wastewater treatment and industrial facilities (PS) within the Chesapeake Bay watershed based on data provided by EPA (CBPO, 2012). Available upgrades for each facility are represented by different combinations of nitrogen and phosphorus effluent concentration targets. For nitrogen, the upgrade options are below 8 mg/L, 5 mg/L, and 3 mg/L, and for phosphorus they are below 1 mg/L, 0.5 mg/L, and 0.1 mg/L. For each upgrade option at each facility, the annual end-of-pipe nutrient reductions were calculated as the difference between the current and new nutrient concentrations multiplied by the annual treated wastewater flow. These end-of-pipe nutrient reductions were further adjusted to account for instream attenuation estimated in the CBWM between the facility and the tidal waters of the Chesapeake Bay. In other words, using this attenuation adjustment, we estimated the reductions in "delivered" loads to the Bay for each upgrade option. Annual costs for each upgrade option were based on estimates of (1) annual operation and maintenance costs for the technology and (2) the one-time capital costs of the technology, annualized using a 20-year assumed lifetime and 7 percent discount rate.[1]

2.2.2 Aggregate Nutrient Reduction Targets for Significant Wastewater Treatment and Industrial Dischargers

We estimated the aggregate nutrient reduction targets for the significant wastewater and industrial sector consistent with EPA's 2012 draft cost analysis assumptions (CBPO, 2012).

[1] A detailed description of the cost estimates are available in Appendix A of Van Houtven et al. (2012).

Reduction targets are only included for facilities with 2010 nutrient discharges above their TMDL-assigned waste load allocation. For these facilities, their required nutrient reduction is equal to the difference between their 2010 and TMDL-assigned nutrient concentration (a) multiplied by the facilities' 2010 treated wastewater flow to estimate annual end-of-pipe loads, and (b) adjusted to reflect instream attenuation between the facility and the tidal waters of the Chesapeake Bay (Table 2-1). The aggregate nutrient reduction targets were defined and calculated as the sum of delivered load reductions across these facilities within a basin-state. These aggregate nutrient reduction targets are included as constraints in the optimization framework to ensure that the set of nutrient controls selected by the optimization routine combine across *both* sectors to meet the required reductions for the significant point-source sector. In other words the load reduction requirements for this sector can in part be met through reductions in the agricultural sector.

2.3 Agricultural Nonpoint Sources

2.3.1 Costs, Effectiveness, and Co-benefits of Agricultural Nutrient Controls

The required data for representing agricultural nonpoint-source (AgNPS) controls were developed primarily using data from the CBWM, Scenario Builder, cost data provided by EPA (CBPO, 2012), and co-benefit estimates described in U.S. EPA (2011). Data from Scenario Builder describe the effectiveness of BMPs as well as their estimated level of implementation in the Bay watershed. Output data from the CBWM describe how 35 land uses and other sources from 2,468 modeled land-river segments contribute nutrients to the tidal waters of the Chesapeake Bay. Within the optimization framework, we aggregate the 14 agricultural land uses specified in the CBWM into 5 land use categories (Table 2-2). We include 14 agricultural BMPs in the optimization framework (Table 2-3). Ten of the BMPs and their possible combinations (52 possibilities) are eligible on cropland and 10 BMPs and their possible combinations, (57 possibilities) are eligible on pastureland. Nutrient reductions for some BMPs are based on an efficiency estimate, which specifies a percentage reduction in edge-of-stream loads when applied, while others are based on a conversion to a land use with lower nutrient contributions, such as replacing cropland with forest.

Table 2-1. Nutrient Load Reduction Targets by Major Basin, Jurisdiction, and Sector

		Significant Industrial and Wastewater Treatment Facilities		Agricultural Nonpoint Sources	
Major Basin	**Jurisdiction**	**N**	**P**	**N**	**P**
Eastern Shore	Delaware	16,265	0	603,721	0
Eastern Shore	Maryland	271,242	21,579	3,924,500	0
Eastern Shore	Pennsylvania	0	0	117,750	2,428
Eastern Shore	Virginia	198,746	2,274	551,696	23,199
James	Virginia	9,565,746	596,542	1,388,383	335,755
James	West Virginia	0	0	3,514	0
Patuxent	Maryland	47,552	15,571	296,692	24,055
Potomac	Washington D.C.	1,530,618	0	0	0
Potomac	Maryland	218,557	36,411	2,075,880	139,891
Potomac	Pennsylvania	55,572	32,775	917,129	72,068
Potomac	Virginia	716,517	112,135	1,834,727	309,500
Potomac	West Virginia	109,952	78,777	147,983	20,669
Rappahannock	Virginia	57,861	18,922	1,225,637	230,863
Susquehanna	Maryland	0	0	472,586	14,397
Susquehanna	New York	614,862	121,018	1,887,455	137,910
Susquehanna	Pennsylvania	4,725,252	256,906	20,667,783	491,755
Western Shore	Maryland	4,650,796	178,742	0	6,694
York	Virginia	502,936	19,790	749,634	50,652
Total		**23,282,474**	**1,491,442**	**36,865,070**	**1,859,836**

Table 2-2. Agricultural Land Uses in the Chesapeake Bay Watershed Model and Optimization Framework

Chesapeake Bay Watershed Model	Optimization Framework
Alfalfa	Hay
Alfalfa nutrient management	
Hay without nutrients	
Hay with nutrients	
Hay with nutrients nutrient management	
High-till without manure	High-Till
High-till with manure	
High-till with manure nutrient management	
High-till without manure nutrient management	
Low-till with manure	Low-Till
Low-till with manure nutrient management	
Degraded riparian pasture	Degraded Riparian Pasture
Pasture	Pasture
Pasture nutrient management	

Table 2-3. Agricultural Best Management Practices in the Chesapeake Bay Watershed Model and Optimization Framework

Agricultural Land Use	Eligible Best Management Practices
Hay, Cropland, and Pastureland	Conservation Plans
	Forest Buffers
	Grass Buffers
	Land Retirement
	Tree Planting
	Wetland Restoration
Hay and Cropland	Decision Agriculture
	Enhanced Nutrient Management
Cropland	Continuous No Till
	Cover Crops[a]
Pastureland	Pasture Alternative Watering
	Prescribed Grazing
	Precision Intensive Rotation Grazing
Degraded Riparian Pasture	Stream Access Control with Fencing

[a]While the CBWM includes multiple cover crop options, the optimization framework includes only Early Drilled Rye, which is the least costly and most effective (according to the CBWM framework, which must be used for TMDL accounting) cover crop option.

Two main categories of monetized co-benefits were included in the U.S. EPA (2011) report for selected AgNPS BMPs. The first category—carbon co-benefits—includes unit value (dollars per acre per year) estimates for changes in carbon sequestration and changes in greenhouse gas (GHG) emissions. Carbon sequestration benefits apply to BMPs involving land use conversion from crop or pastureland to forest, wetland, or grass cover. Changes in GHG emissions were also associated with land conversion BMPs and "working land" BMPs that reduce fertilizer application. The second category—recreational hunting co-benefits—includes unit value estimates associated with increases in wetland cover (waterfowl hunting benefits) and increases in forest cover (nonwaterfowl hunting benefits). In Section 3 of this report, we expand the list of co-benefits to also include values for improving water quality in rivers and streams.

To estimate nutrient reductions associated with a BMP or combination of BMPs, we first use data from CBWM to estimate the per-acre nutrient loads delivered to the Bay from each land use and existing BMP combination within each land-river segment. We then determine which BMP or BMP combinations could be added to these existing ones. For each potential option, we then apply BMP-specific load removal efficiencies and attenuation factors from CBWM to estimate reduced loads to the Bay. For example, if there are 100 acres of high till cropland in a land-river segment, none of which currently have BMPs in place, and they deliver 1,500 lbs of nitrogen to the Chesapeake Bay each year (15 lb N/acre), we would estimate that applying a BMP with a 40 percent removal efficiency to these acres would reduce loads by 600 lbs per year. Alternatively, if the 100 acres deliver 1,500 lbs per year, but half of those acres currently use a BMP with a 50 percent efficiency, then the 50 acres with BMPs would be estimated to deliver 10 lb N/acre, while the 50 acres without a BMP would deliver an estimated 20 lb N/acre. Adding the BMP with 40 percent efficiency to the area without BMPs would reduce loads by 400 lbs per year, whereas it would reduce loads by only 200 lbs per year if added to the area that already uses one BMP. In other words, nutrient reductions from implementing a specific BMP depend on the current estimated delivered loads per acre, which depends on the currently implemented BMPs.

2.3.2 Eligibility of Agricultural Nutrient Controls

Eligibility for BMP implementation on agricultural lands is determined by the type of land use (e.g., crop or pasture) and the set of BMPs currently implemented. For example, if cover crops are already implemented on a portion of cropland acres, cover crops would not be an eligible BMP in that same area. For some BMPs, eligible acres are identified by the following site-suitability criteria. Land retirement is restricted to occur only on highly erodible soils, wetland restoration is restricted to only hydric soils, and forest and grass buffers are restricted to only the riparian area remaining after subtracting the acres of forest and grass buffer implemented in the CBWM. This combination of land-river segment, land use, existing BMP combinations, and highly-erodible soils, hydric soils and riparian areas constitutes the unit of analysis for agricultural source controls in the optimization framework.

2.3.3 Nutrient Reduction Target for Agricultural Sources

We estimate the aggregate nutrient reduction targets for the agricultural nonpoint sector by taking the difference between the nutrient contribution of the included agricultural land uses in 2010 and with the TMDL[1] (Table 2-1). These nutrient reduction targets are included in the optimization framework as constraints to ensure that the set of selected nutrient controls combine to meet the agricultural nonpoint sector's required reductions. In addition to including the nutrient reduction target as a constraint, where specified, we impose a constraint on the conversion of agricultural land to other land uses.

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[1] The CBWM TMDL scenario used is based on the jurisdictions' Phase I WIPs.

SECTION 3

INCLUDING CO-BENEFITS FROM FRESHWATER QUALITY IMPROVEMENTS IN THE OPTIMIZATION FRAMEWORK FOR CHESAPEAKE BAY RESTORATION

3.1 Introduction

Nutrient control projects being implemented throughout the watershed in order to comply with the TMDL are addressing water quality concerns in the Chesapeake Bay estuary, but will simultaneously generate ancillary benefits in nontidal rivers and streams by improving water quality. Reduced nutrient loads to upstream catchments will improve conditions within those catchments, as well as along the downstream river network to the Bay. These improvements will enhance freshwater ecosystem services and benefit people living inside and outside the watershed.

In this section, we expand our optimization framework to include monetized estimates for these freshwater co-benefits. This effort supplements prior work (U.S. EPA, 2011) in which we monetized co-benefits from reductions in atmospheric GHGs and recreational hunting benefits from increased forest cover and wetland restoration. We found that the GHG co-benefits generated the vast majority of monetized co-benefits from nutrient source controls that achieve the TMDL. In this section, we examine how including freshwater co-benefits into the optimization framework influences the selection and geographic distribution of nutrient source controls.

3.2 Freshwater Quality Benefits of Reduced Nutrient Loads

To include these freshwater co-benefits in the optimization framework, we must estimate the value of reducing nitrogen and phosphorus loads to streams and rivers throughout the watershed. In particular, we must estimate *average per-pound values* for reduced edge-of-stream loads in each river segment. We do this using the following four steps.

1. For each nontidal river segment in the Bay watershed, we acquire water quality improvement estimates from the CBWM, based on a watershed-wide load reduction scenario.

2. We use economic benefit transfer methods to estimate the total annual value of these water quality improvements in each river segment.

3. We estimate the average per-pound value of reduced nutrient loads received by each segment. To calculate this value we divide the total value of water quality improvements in the segment (from the previous step) by the sum of reduced loads from nutrient sources in the segment itself *and from all upstream river segments* (accounting for instream attenuation of nutrients between segments).

4. We estimate the average per-pound value of reducing nutrient loads discharged to surface water in each river segment. Using results from the previous step, we take the segment's

per-pound value for received loads, and we add the per-pound values *for all downstream segments* (adjusting for instream attenuation between segments).[1]

Below, we describe each of these four steps in more detail.

For the first step, we use results from CBWM model runs to define water quality improvements for a specific watershed-wide load reduction scenario. The selected scenario compares estimated nutrient loads in 2009 (i.e., "without-TMDL") to load estimates based on implementation of the Phase II WIPs (i.e., "with-TMDL"). For both with- and without-TMDL load conditions, the CBWM provides daily average concentration estimates in each nontidal (i.e., freshwater) river segment for six main water quality parameters: total nitrogen (TN), total phosphorus (TP), DO, chlorophyll-*a*, total suspended solids (TSS), and 5-day biochemical oxygen demand (BOD5). For our analysis, we use these estimates to calculate average annual concentrations for each parameter within a river segment.[2]

For the second step, we apply a benefit transfer approach to approximate the total economic value (i.e., use plus non-use benefits) of changes in these water quality parameters resulting from the load reduction scenario. Our approach is similar to methods used by EPA's Office of Water to estimate the benefits of proposed and promulgated effluent guidelines (e.g., U.S. EPA, 2013). The approach requires several simplifications, recognizing that developing precise monetary estimates of the benefits that humans receive from water quality improvements is difficult because the links between changes in these water quality parameters and their effects on human well-being are complex and multifaceted. The benefits of water quality improvements are likely to include in situ uses (recreational activities and aesthetic appreciation), as well as non-use benefits for those who simply derive value from knowing that water resources are being protected.

The benefit transfer approach has three main components. First, for each nontidal (freshwater) river segment and load condition (i.e., with- or without-TMDL), we combine the multiple water quality parameter values from the CBWM into a single composite water quality index (WQI) (Figure 3-1). The WQI is a nonlinear combination of subindices for each of the six water quality parameters listed above. Each of the six subindices is derived from a separate nonlinear function, which translates concentrations of the water quality parameter to a 100-point scale.[3] The water quality improvement for each river segment due to the TMDL is therefore expressed as a change in the WQI

[1] The per-pound value for *received loads* approximates the value that each pound discharged in the segment has on water *in the segment*. The per-pound values for *downstream segments* approximates the value that each pound discharged in the segment has on water *in the downstream segments.*

[2] Water quality parameters for the tidal river segments closest to the Bay itself are not simulated in the CBWM.

[3] For some of these parameters, including TN and TP, the coefficients and upper and lower bounds of these functions vary across ecoregions in the United States (including within the Chesapeake Bay watershed).

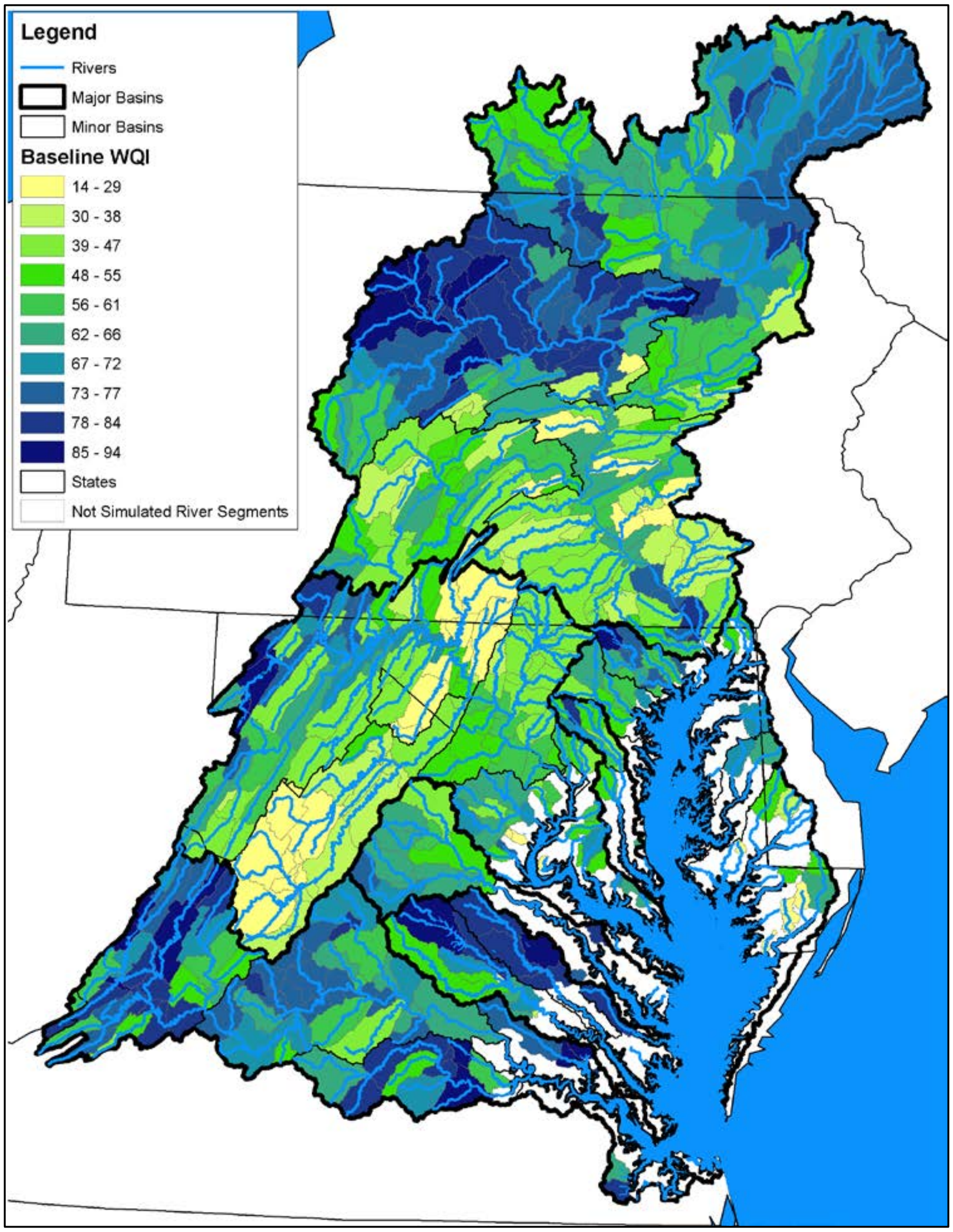


Figure 3-1. Without-TMDL Water Quality Index (100-point Scale) Values

between the without-TMDL and with-TMDL load conditions (Figure 3-2).[1] Second, to value these water quality improvements, we apply a benefit transfer function that translates changes in WQI into an average household-level willingness to pay (WTP) for the improvement. This benefit transfer function is reported in Van Houtven et al (2007) and is based on a meta-analysis of 131 WTP estimates from 18 water quality valuation studies conducted in the United States. Third, we apply

[1] For this analysis, we assume that this load reduction scenario (combining the with- and without-TMDL load scenarios) provides representative average per-pound values for load reductions in the watershed. Additional investigation will be needed to determine how much these average values would vary using alternative scenarios.

these WTP values to estimate aggregate state-level benefits for WQI improvements in each river segment. Specifically, we multiply the WTP values for each segment by (1) the total number of households residing in the state where the segment is located and (2) the percentage of each state's total river miles that are located in the segment.

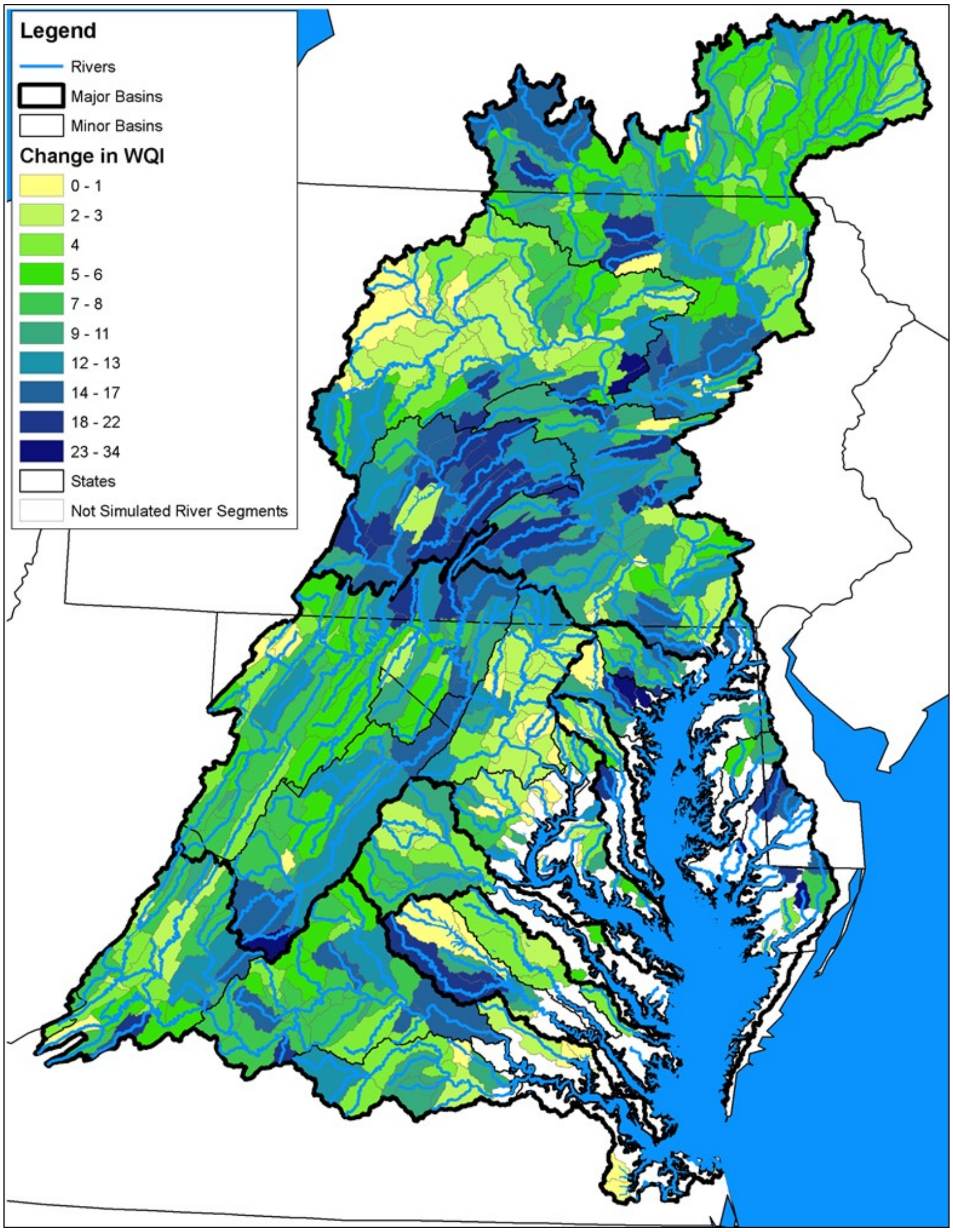


Figure 3-2. Change in Water Quality Index (100-point Scale) Values with the TMDL

For the remaining steps, the objective is to transform the values for *water quality improvements in* each segment to values for *load reductions from* each segment. That is, for each freshwater river segment, we need to estimate the average per-pound value of a load reduction, as it contributes to water quality improvements in its own segment, as well as in all downstream segments.

In the third step, we estimate the total amount of nutrients entering the river segment both from within the river segment and from all upstream river segments. To estimate the nutrient contribution from an upstream segment to a connected downstream river segment, we account for instream attenuation of nitrogen and phosphorus between the segments. These instream attenuation rates between segments are derived from the CBWM's attenuation factors, which are expressed as delivery ratios between each segment and the Bay tidal waters. We estimated attenuation between segments by dividing the delivery ratio of the upstream river segment by the delivery ratio of the receiving segment. For example, if every pound of nitrogen from an upstream river segment *A* contributed half a pound of nitrogen to the Bay (delivery ratio = 0.5) and every pound from a downstream segment *B* contributed three-fourths of a pound to the Bay (delivery ratio = 0.75), then we estimated the attenuation rate between river segment *A* and *B* to be two-thirds (0.667 = 0.5/0.75).

In addition, we need to account for the separate contributions of nitrogen loads and phosphorus loads to water quality. To simplify, we first created a nutrient load index (NLI) that combines both nutrients in proportion to their expected impact on water quality and can be interpreted as the total nutrient contribution in pounds of nitrogen equivalent. [1] The index formula is sum of the nitrogen load plus 12 times the phosphorus load (NLI = N +12*P). The selection of the number 12 to convert pounds of phosphorus into equivalent pounds of nitrogen is based on two main considerations. First, the TN and TP subindices of the WQI share the same functional form but have different coefficient values, which also differ across the 12 ecoregions in the Chesapeake Bay watershed (U.S. EPA, 2013). For each ecoregion, we estimated the nitrogen and phosphorus concentrations (mg/L) that would correspond with a subindex value of 75 (a midrange of modeled values for the watershed). Across ecoregions, the ratio of these values ranged from 4:1 to 39:1 with an average of 12:1. Second, after generating changes in the WQI for each river segment, we regressed these WQI changes on changes in nitrogen and phosphorus loads in each segment. In this regression, the ratio of the coefficients on nitrogen and phosphorus is 11.6, meaning that a 1 pound change in phosphorus will have on average the same impact on the WQI as roughly a 12 pound change in nitrogen.

To complete step three, we divide the valued water quality benefits in each river segment by the total nutrient load index delivered to the segment to estimate the value per pound of nitrogen equivalent in that river segment. These per-pound nitrogen equivalent values are, by necessity, linear approximations of complex and often nonlinear relationships between nutrient loads, instream

[1] To estimate a separate average value for nitrogen and phosphorus loads in a way that avoids double counting, we would need to divide and allocate the portion of WQI changes in each segment that are attributable to upstream loads of each pollutant. Using a combined nutrient index provides a simplified alternative for including both pollutants.

concentrations, and the WQI. Nutrient reductions in river segments selected by the optimization framework may be much higher or much lower than those observed in the scenarios used to calculate the per-pound value, and, as such, the per-pound value applied may be too high or too low. To reduce the chance that an overestimate of per-pound values drives the results in the optimization framework, we removed outliers by capping all values at the third quartile plus 1.5 times the interquartile range of estimated values, or $9.53 per pound of nitrogen equivalent. Within a river segment, the value per pound of nitrogen equivalent can now be used to estimate the value per pound of nitrogen and the value per pound of phosphorus.

In the fourth step, we estimate the average per-pound value of reduced nutrient loads to surface water in each river segment. This estimate is based on the value per pound of nitrogen equivalent within the river segment, plus the value per pound of nitrogen equivalent in all downstream river segments, after accounting for the attenuation rate between the river segment and all downstream river segments. This attenuation rate can differ for nitrogen and phosphorus. Using the hypothetical river segments *A* and *B* described above, if a pound of nitrogen reduced to river segment *A* is valued at $4/lb and a pound of nitrogen reduced to the downstream river segment *B* is valued at $2/lb, the pound reduced in river segment *A* is valued at $4 plus $2 times the attenuation rate between *A* and *B*, or $5.33 ($4 + $2 x 2/3 = $5.33).

The general pattern observed in the geographic distribution of values per pound of nutrient reductions is that the higher values per pound are for those farther upstream (Figure 3-3). This is due to two complementary factors. First, upstream river segments have relatively lower flow than downstream river segments. A pound of nutrients in a river with lower flow will have a larger impact on the nutrient concentration in the river, which is the basis for the WQI. Second, a pound reduced in an upstream river segment will improve water quality in all downstream river segments. Therefore, load reductions in segments located farther from the Bay have the capacity to improve water quality in a higher number of downstream segments, even when accounting for instream attenuation.

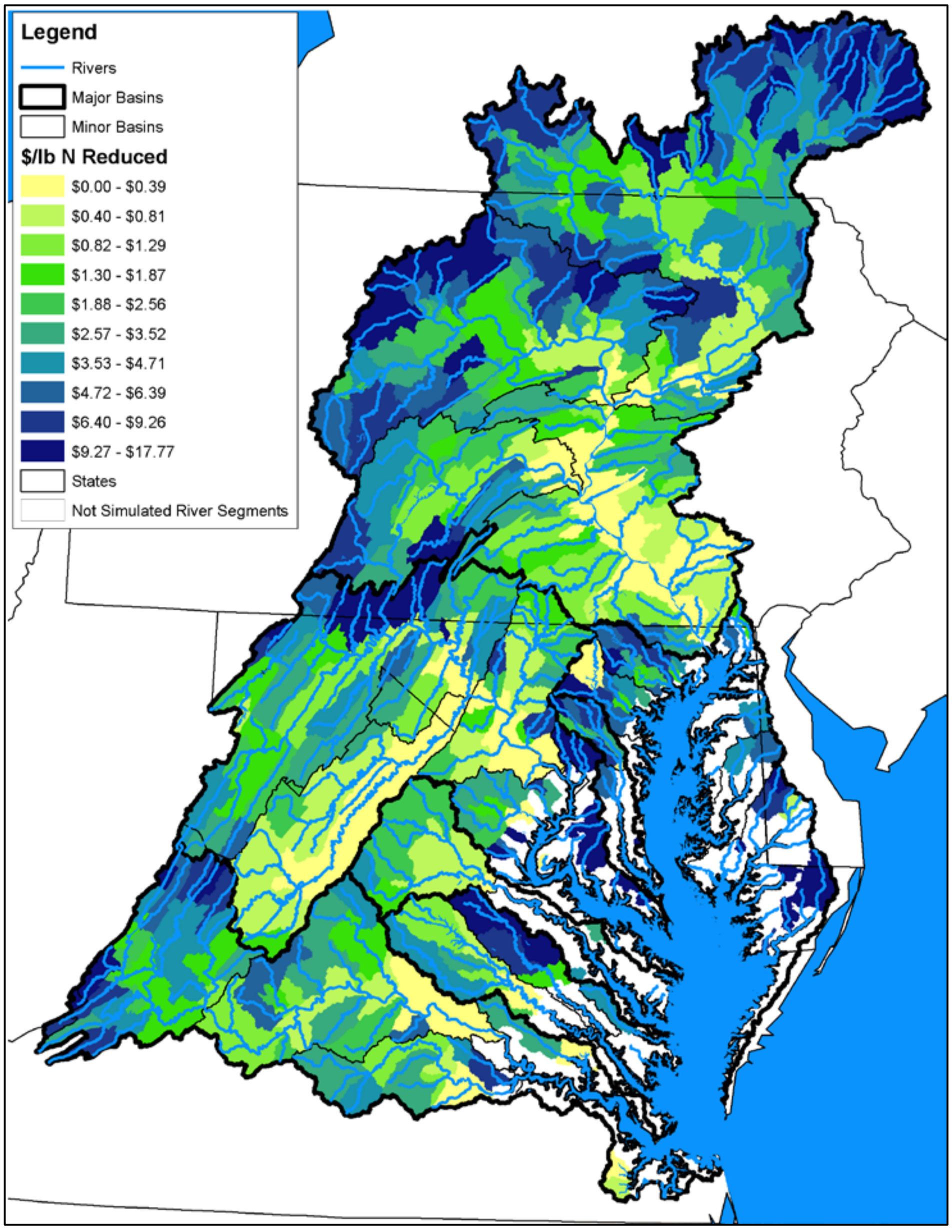


Figure 3-3. Value ($) per Pound of Nitrogen Reduced by River Segment

3.3 Scenario Analysis Results

The next steps in the analysis are to (1) incorporate these water quality co-benefit estimates into the optimization framework, (2) estimate the least-net-cost solution including these co-benefits, and (3) compare this solution to other load reduction scenarios that meet the TMDL requirements. The first load reduction scenario, which we refer to as "TMDL," represents the load reduction practices specified in the jurisdictions' Phase I WIPs. This scenario does not make use of the optimization framework.

The second scenario—the least-cost scenario—applies the optimization framework to achieve the same overall load reductions as the TMDL scenario. In both cases, the load reduction targets are defined as the combined targets for significant wastewater and industrial facilities (PS) and agricultural nonpoint sources (AgNPS) (see Table 2-1). This scenario optimizes over the costs and delivered load reductions from available nutrient source projects; however, it does not include co-benefits in the optimization.

The third scenario—the least-net-cost scenario—includes co-benefits in the optimization framework. These co-benefits include the monetized carbon and hunting benefits estimated in the (U.S. EPA, 2011) analysis, which only accrue to agricultural acres applying selected BMPs (see Appendix A). The co-benefits also include the freshwater quality benefits associated with edge-of-stream load reductions in each nontidal land-river segment. These water quality benefits accrue both to point sources and to agricultural nonpoint sources. The net-costs for each potential project are calculated by deducting all of the monetized co-benefits from the costs of the control project. The optimization solves for the approach that minimizes these net costs while still achieving the total load reduction targets.

To analyze the differences among these scenarios, we focused on two "basin-state" areas -- the Susquehanna River basin in Pennsylvania (Susquehanna-PA) and the James River basin in Virginia (James-VA). In all scenarios, agricultural land conversion is restricted to 25 percent. The results are shown in Table 3-1.

In the Susquehanna-PA, the TMDL scenario results in total annual costs of approximately $280 million to meet their required load reductions for PS and AgNPS. These costs are from the specific PS and AgNPS source controls included in the Phase I WIPs, which are the basis for this TMDL scenario. We estimate these nutrient controls generate $137 million in ecosystem services co-benefits, including $29 million of upstream water quality benefits from PS source controls.

By solving for the least-cost set of source controls, we estimate that these reductions could be achieved at 49 percent lower costs than specified by the Phase I WIPs; however, the co-benefits from upstream water quality improvements would also be lower (Table 3-1). In the least-cost scenario, much of the nutrient reduction occurs in southeastern Pennsylvania (Figures 3-4 and 3-5).

Table 3-1. Costs and Benefits of Significant Wastewater Treatment and Industrial Facilities (PS) and Agricultural Nonpoint Source (AgNPS) Nutrient Reductions in the Susquehanna-PA (Million $)

Nutrient Load and Cost-Benefit Categories	Source	Scenario TMDL	Cost Minimizing	Net Cost Minimizing
N Load Reductions	**Total**	**25.4**	**25.4**	**25.4**
(mil. lbs/yr)	PS	4.7	3.6	3.2
	AgNPS	20.7	21.8	22.2
P Load Reductions	**Total**	**0.7**	**0.7**	**0.7**
(mil. lbs/yr)	PS	0.3	0.3	0.3
	AgNPS	0.5	0.5	0.5
Control Costs	**Total**	**$280.0**	**$142.7**	**$158.9**
($ mil./yr)	PS	$60.0	$24.0	$21.1
	AgNPS	$220.0	$118.7	$137.7
Co-Benefits ($ mil./yr)	**Total**	**$137.2**	**$103.9**	**$135.3**
Freshwater Benefits	PS	$29.0	$24.9	$26.2
Freshwater Benefits	AgNPS	$86.9	$62.1	$82.1
Carbon Benefits	AgNPS	$21.0	$16.7	$26.6
Hunting Benefits	AgNPS	$0.3	$0.2	$0.4
Net Costs	**Total**	**$142.8**	**$38.8**	**$23.6**
($ mil./yr)	PS	$31.0	($0.9)	($5.0)
	AgNPS	$111.8	$39.6	$28.6

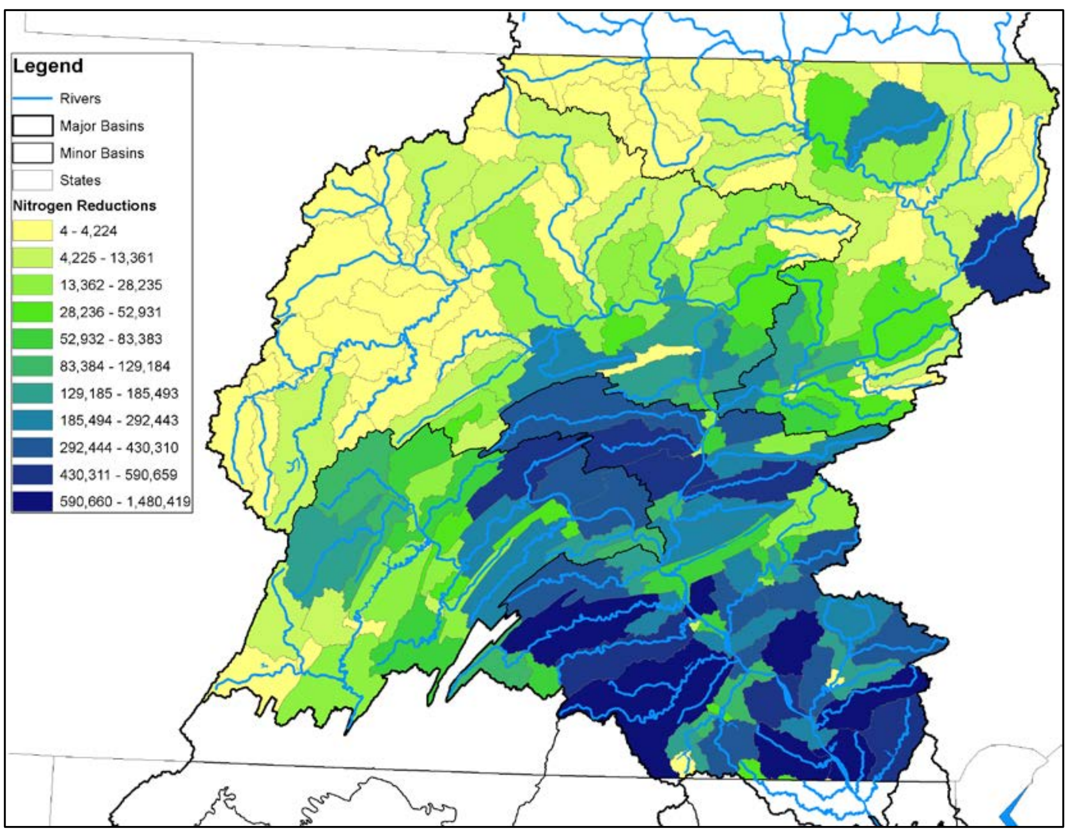


Figure 3-4. Least-Cost Scenario Nitrogen Load Reductions in the Susquehanna-PA by River Segment (Delivered Pounds per Year)

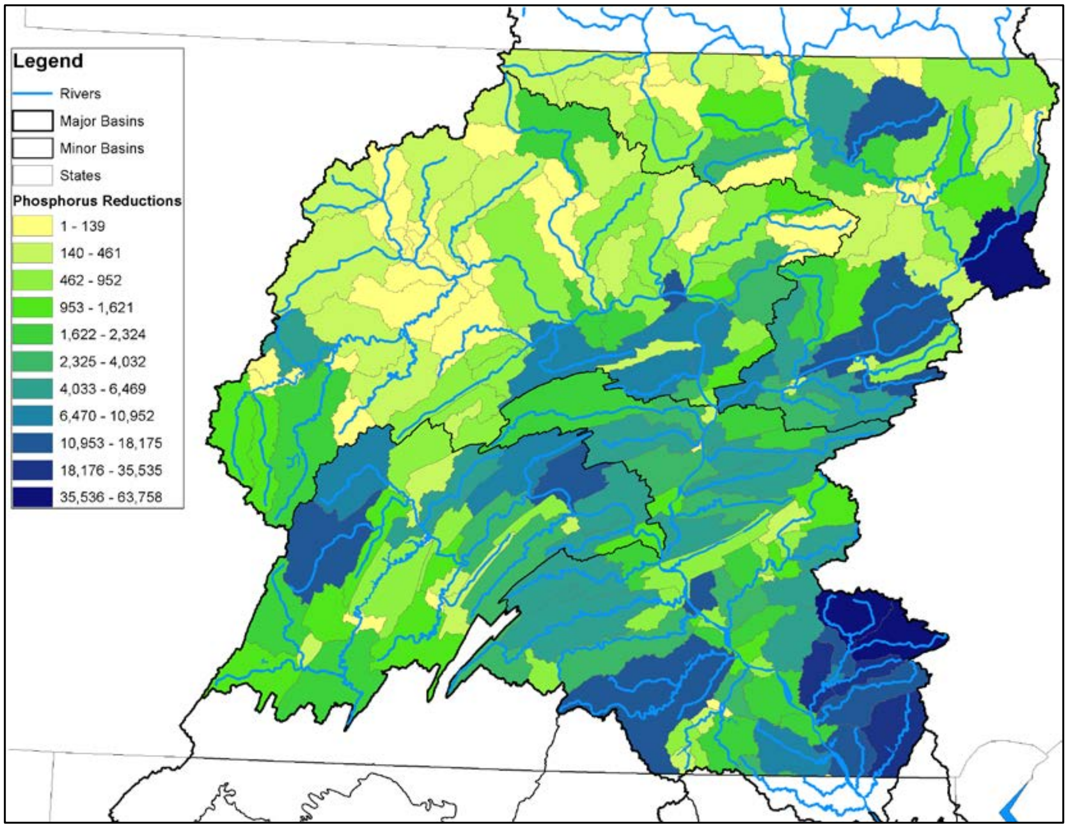
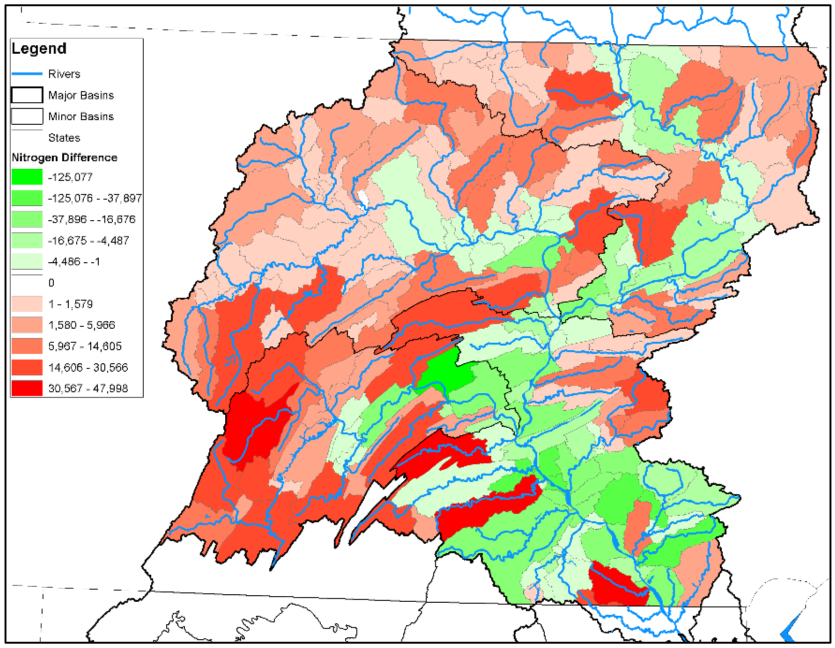


Figure 3-5. Least-Cost Scenario Phosphorus Reductions in the Susquehanna-PA by River Segment (Delivered Pounds per Year)

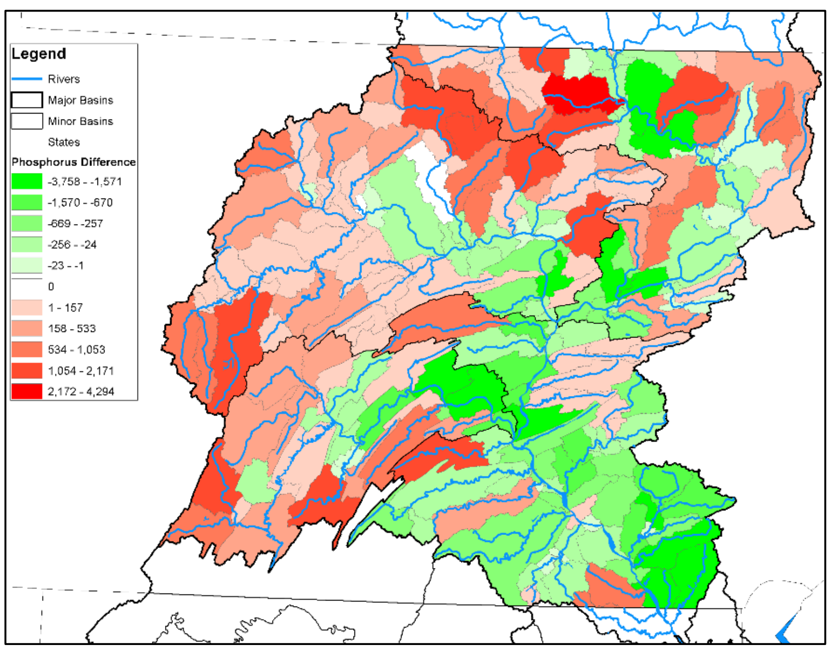
By including monetized ecosystem service co-benefits and solving for the least-net-cost set of source controls, the cost of source controls increases by $16 million relative to the least-cost scenario, and the monetized co-benefits increase by $31 million. Including the co-benefits encourages a shift towards AgNPS source controls. Interestingly, despite generating fewer nutrient reductions than in the least-cost solution, PS nutrient reductions generate greater upstream water quality benefits under the least-net-cost solution. Overall, we observe that more nutrient reductions occur in upstream tributaries to the main channel of the Susquehanna River under the least-net-cost scenario relative to the least-cost scenario (Figures 3-6 and 3-7).

In the James River-VA, the TMDL scenario costs are estimated to be $188 million per year. Compared to the Susquehanna case, a much larger portion of these costs (73%) and load reductions are associated with PS controls. Despite this difference, about 75 percent of $116 million in annual freshwater quality co-benefits come from AgNPS. The proximity of many PS sources to the Bay accounts for their relatively low contribution to freshwater benefits in this scenario.



Note: Positive values reflect river segments where more nutrient reductions occur in the least-net-cost scenario relative to the least-cost scenario.

Figure 3-6. Difference in Nitrogen Reductions from Least-Net-Cost Source Controls and Least-Cost Source Controls in the Susquehanna-PA by River Segment (Delivered Pounds per Year)



Note: Positive values reflect river segments where more nutrient reductions occur in the least-net-cost scenario relative to the least-cost scenario.

Figure 3-7. Difference in Phosphorus Reductions from Least-Net-Cost Source Controls and Least-Cost Source Controls in the Susquehanna-PA by River Segment (Delivered Pounds per Year)

As in the Susquehanna-PA, the results of the least-cost scenario in the James River basin indicate that there could be substantial (46%) cost savings relative to the scenario based on the Phase I WIPs. However, in the James River basin, the least-cost scenario also results in relatively larger freshwater quality co-benefits from AgNPS compared to the TMDL scenario. In the least-cost scenario, nitrogen reductions occur primarily around the tidal region of the James River (Figure 3-8), while phosphorus reductions primarily occur farther upstream (Figure 3-9).

Going from the least-cost scenario to least-net-cost scenario costs increase by $17 million, while benefits increase by $42 million. Therefore, the strategy that targets all benefits instead of just tidal water quality improvements increases net benefit (reduces net costs) by $25 million. The majority of the estimated co-benefits in the least-net-cost scenario are from freshwater quality improvements attributable to agricultural nonpoint source controls (Table 3-2), which were valued using the methods described in Section 3.2. Figures 3-10 and 3-11 show how the spatial pattern of nitrogen and phosphorus load reductions change between the least-cost and least-net-cost scenarios. When freshwater co-benefits are included in the optimization, the nutrient source control tends to shift away from the main stem and tidal areas of the James-VA to river segments further upstream.

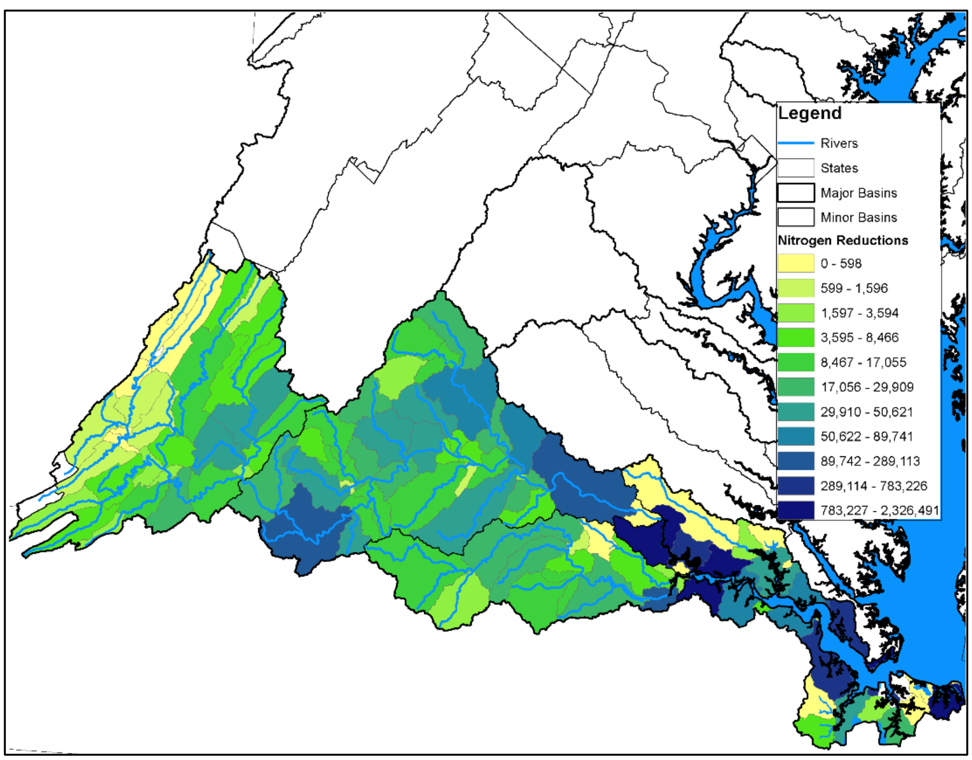


Figure 3-8. Least-Cost Scenario Nitrogen Reductions in the James-VA by River Segment (Delivered Pounds per Year)

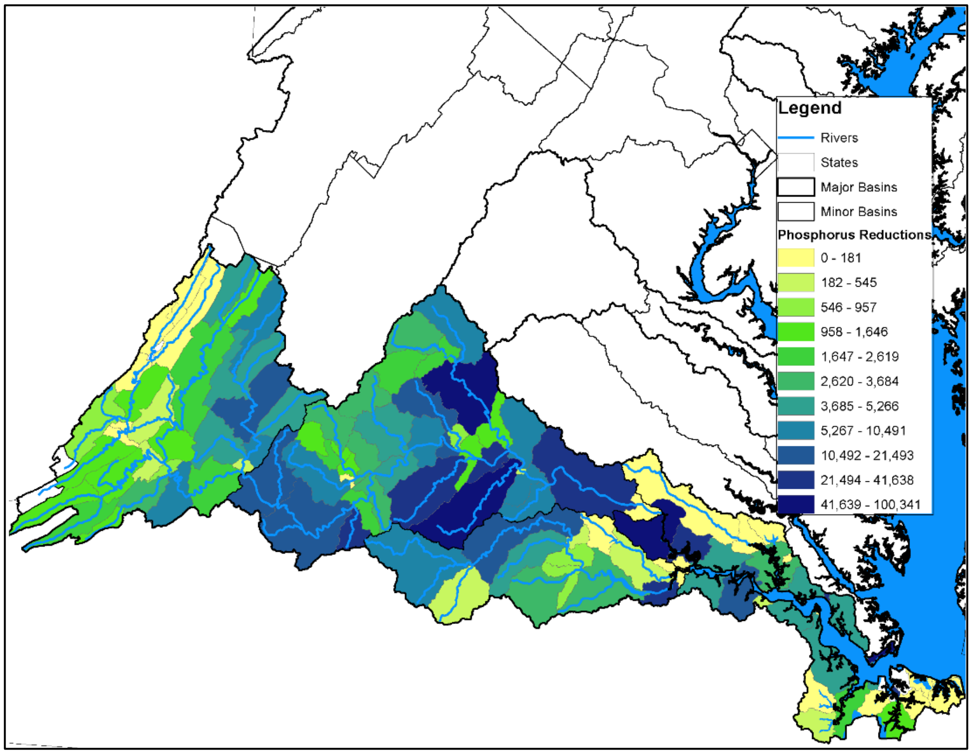


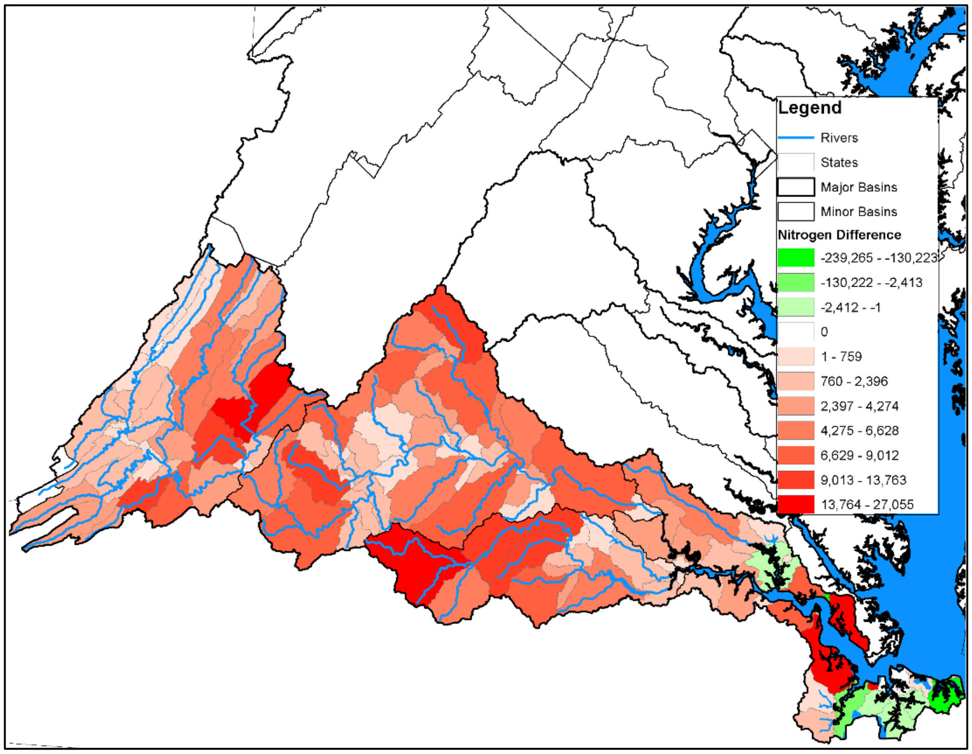
Figure 3-9. Least-Cost Scenario Phosphorus Reductions in the James-VA by River Segment (Delivered Pounds per Year)

Table 3-2. Costs and Benefits of Significant Wastewater Treatment and Industrial Facilities and Agricultural Nonpoint Source Nutrient Reductions in the James-VA (Million $)

Nutrient Load and Cost-Benefit Categories	Source	Scenario		
		TMDL	**Cost Minimizing**	**Net Cost Minimizing**
N Load Reductions	**Total**	**11**	**11**	**11**
(mil. lbs/yr)	PS	9.6	9.2	8.8
	AgNPS	1.4	1.8	2.1
P Load Reductions	**Total**	**0.9**	**0.9**	**1.1**
(mil. lbs/yr)	PS	0.6	0.4	0.3
	AgNPS	0.3	0.5	0.7
Control Costs	**Total**	**$188.0**	**$101.2**	**$118.5**
($ mil./yr)	PS	$138.0	$84.6	$75.7
	AgNPS	$50.0	$16.7	$42.8
Co-Benefits ($ mil./yr)	**Total**	**$41.8**	**$39.0**	**$81.1**
Freshwater Benefits	PS	$6.3	$3.5	$3.6
Freshwater Benefits	AgNPS	$24.4	$26.4	$45.4
Carbon Benefits	AgNPS	$11.1	$9.0	$31.8
Hunting Benefits	AgNPS	$0.1	$0.1	$0.2
Net Costs	**Total**	**$146.2**	**$62.2**	**$37.5**
($ mil./yr)	PS	$131.7	$81.1	$72.1
	AgNPS	$14.5	($18.8)	($34.6)

In summary, through this analysis we demonstrate an approach for including co-benefits from freshwater quality improvements into the optimization framework. In both of the river basins we examined and across all scenarios, we estimate values for freshwater quality co-benefits that are larger than the combined values for the other monetized co-benefits.

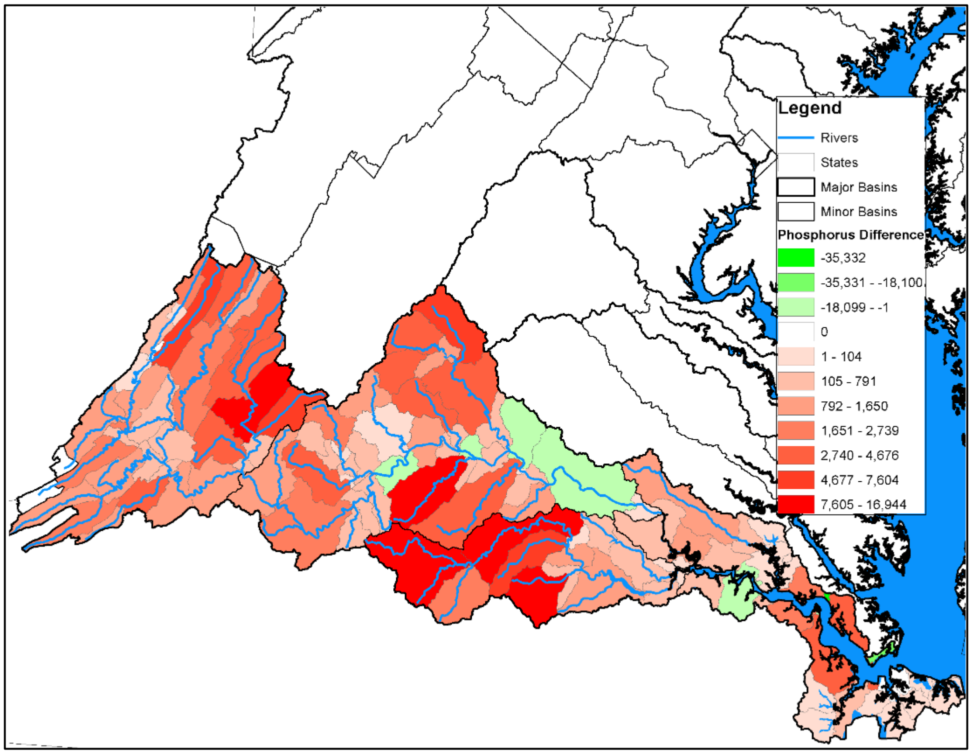
Applying these results in the optimization model, we are also able to improve our estimates of the efficiency gains that can be achieved by accounting for co-benefits. Compared to cost-minimization results that do not account for co-benefits, we find that including all co-benefits in the optimization would increase annual nutrient control costs by $16 million in the Susquehanna-PA; however, the co-benefits would increase by $31 million (net gain of $15 million per year). In the James-VA, considering monetized co-benefits results in an estimated increase in nutrient control costs of $17 million but an increase in co-benefits of $42 million (net gain of $25 million per year).



Note: Positive values reflect river segments where more nutrient reductions occur in the least-net-cost scenario relative to the least-cost scenario.

Figure 3-10. Difference in Nitrogen Reductions from Least-Net-Cost Source Controls and Least-Cost Source Controls in the James-VA by River Segment (Delivered Pounds per Year)

In addition, when including these freshwater co-benefits in the optimization analysis, we find that the least-net-cost scenario tends to shift load reductions (1) towards river segments located farther from the Bay and (2) from point sources to agricultural NPS (compared to the least-cost scenario). The spatial shift is primarily due to the higher per-pound values of load reduction co-benefits in more upstream areas. These higher per-pound values occur because there is less flow in the more upstream receiving waters (i.e., less dilution of loads) and because they affect more downstream river miles. The shift in load reduction away from point sources occurs in large part because point sources do not provide carbon sequestration benefits. The tendency for point sources to be located in the more downstream areas also contributes to this result.



Note: Positive values reflect river segments where more nutrient reductions occur in the least-net-cost scenario relative to the least-cost scenario.

Figure 3-11. Difference in Phosphorus Reductions from Least-Net-Cost Source Controls and Least-Cost Source Controls in the James-VA by River Segment (Delivered Pounds per Year)

SECTION 4
ALTERNATIVE MARKET-BASED INCENTIVE DESIGNS FOR CHESAPEAKE BAY RESTORATION

4.1 Introduction

The Chesapeake Bay TMDL is designed to reduce nutrient loads delivered to the Bay by roughly 25 percent compared to conditions in 2009 (U.S. EPA, 2010). To achieve this objective, the TMDL defines maximum allowable annual load allocations (to be achieved by 2025) for various point and nonpoint source categories. To achieve these load allocations, agricultural nonpoint sources are expected to contribute over 60 percent of the total annual nutrient load reductions.

How the TMDL load allocations for agricultural sources will be achieved remains a crucial question, since they are not federally regulated sources and, therefore, their reductions have historically been achieved largely through voluntary programs. Several states in the Bay watershed have developed trading programs to support more cost-effective nutrient reductions and to provide a potential additional incentive for agricultural sources to generate nutrient reductions beyond their allocation, or baseline, to sell. In addition to trading programs, more traditional agricultural cost-share and payment programs are being used to make progress towards reduction goals. The incentives offered by these public sector programs can be used to bring farms into compliance with baseline requirements for trading.

An efficient market offers the potential to achieve a more cost-effective implementation of the TMDL by allocating nutrient reductions to those with the lowest costs. The optimization framework provides insights into how an efficient nutrient trading market may function in the face of multiple incentives and how changes in market rules, such as trading baseline requirements, may influence the load reductions and potential cost savings of an efficient nutrient trading market.

The remainder of this section is organized as follows:

- Section 4.2 assesses the role nutrient trading can play in encouraging agricultural sources to meet their TMDL requirements, including interactions of markets with other incentive programs.
- Section 4.3 analyzes how alternative methodologies for crediting agricultural BMPs impact the load reductions and potential cost savings from nutrient trading.

4.2 Nutrient Credit Trading and Other Incentives to Meet Agricultural Baseline Requirements

In this section, we use the optimization framework to examine the potential implications of different types and combinations of incentive-based approaches for agricultural nonpoint sources. In particular, we address the following questions:

- To what extent can the incentives offered by nutrient credit trading encourage farmers to achieve their TMDL load reduction requirements?
- How are these conclusions affected by including additional public sector payments to farmers for nutrient controls?
- How does the interaction of trading, baseline requirements, and direct payments affect the total costs of nutrient control implementation resource costs and public sector budget requirements for funding nutrient controls?

Although our optimization approach provides a useful and tractable framework for simulating behaviors and policy outcomes, it also requires simplifying assumptions that are likely to overstate behavioral responses to a nutrient credit market and public sector incentives. Consequently, **the simulated results are best interpreted as upper bound estimates of policy-induced changes rather than as predictions of actual outcomes**. In particular, by using a cost-minimization approach, we are assuming that farmers and point-source operators are strictly motivated by a desire to maximize their profits. In practice, their behaviors are more complex and include several factors that we cannot formally observe or account for in our model. To address some of the potential transaction and information-related costs, such as the time required to understand the requirements for nutrient trading, finding a trading partner, and negotiating a contract, that can interfere with these behaviors, we have augmented the unit costs of nonpoint-source nutrient controls by a fixed percentage (38%) (McCann and Easter, 2000); however, this approach only provides a partial adjustment.[1] Although in practice there are differences in trading program requirements across jurisdictions in the watershed (which may lead to differences in transaction costs), for the purposes of this analysis we assume uniform trading requirements across the entire watershed.

4.2.1 Nutrient Credit Trading and the Role of Trading Baselines

The purpose of nutrient trading programs is primarily to provide regulated (i.e., "capped") sources with additional flexibility and the opportunity to incur lower costs for meeting their regulatory requirements. In particular, trading offers point source dischargers the ability to

[1] For example, some administrative costs of trading may be fixed and not vary directly with the number or size of control practices credited.

achieve compliance by purchasing load reductions (credits) from other sources, who are either unregulated or who reduce loads beyond their own regulatory requirements.

Trading programs often include baseline requirements for credit generation by agricultural nonpoint sources. This trading baseline is defined by a set of preconditions for controlling nutrient runoff that a farm must achieve before it is eligible to generate credits for additional (i.e., “beyond baseline”) nutrient reductions. Generally speaking, there are two main approaches for defining the trading baseline requirements:

- Practice-based approach, which typically defines the types or combinations of BMPs that must be in place; and
- Performance-based approach, which typically defines the load reductions or maximum allowable level of loads that must be achieved.

Currently, three Bay states—Maryland, Pennsylvania, and Virginia—have established credit trading programs, and each uses a different approach for defining agricultural trading baselines (Branosky et al., 2011). Virginia uses a practice-based approach, Maryland uses a performance-based approach, and Pennsylvania uses a combination.

Despite the differences across state programs, it is important to note that the trading baselines are all expected to be consistent with the TMDL load allocations. In other words, if all agricultural nonpoint sources were to meet the trading baseline requirements of their state’s trading programs, then they are expected collectively to be in compliance with the TMDL load allocations.

The inclusion of baseline requirements in these trading programs has potentially important implications for farmers’ incentives to generate credits and for the agricultural sector as a whole to meets its TMDL load allocations. In effect, the baseline requirements create a hurdle for farmers to participate in trading. If trading programs did not include this hurdle, farmers would have a greater incentive to reduce their loads and generate credits; however, reductions sold as credits cannot be counted as progress towards meeting agriculture’s load allocation. Instead, all of these load reduction must be transferred to the buyers (e.g., wastewater treatment plants) and credited towards achieving the buyers’ TMDL allocations.

In other words, without the baseline requirement, it would be easier and cheaper for farmers to generate load reductions that they can sell as credits. At the same time, farmers would have less of an incentive to generate load reductions that they would not sell and that would count towards their own TMDL requirements. While credit buyers would benefit by being able to meet their load reduction requirements at a lower cost, there would be less incentives for the agricultural sector to meet its TMDL requirements. Moreover, allowing farmers to sell their relatively low-cost load reductions to other sectors would make it more costly for the agricultural sector to achieve its own load allocation.

To examine the implications of baseline requirements, the first question then becomes:

- How large of a hurdle do the baseline requirements present for potential credit sellers?

For trading to be profitable for a farmer, the revenue from selling credits must exceed the full costs of reducing nutrient runoff. However, whereas costs must be incurred to both meet the baseline and generate additional reductions, only the nutrient reductions that surpass the trading baseline requirements are eligible to generate salable credits. Therefore, the profitability of credit trading for a farmer will depend in part on the costs she must incur to meet her trading baseline requirements.

If selling credits can in some cases be profitable despite the trading baseline hurdle, the next questions are:

- How large will the resulting load reductions from the agricultural sources be?
- How much will the portion of reductions below the trading baseline contribute towards agriculture's TMDL load reduction requirement?
- How will trading between point and nonpoint sources affect the overall costs of achieving the TMDL?
- How would results change if public sector incentives for nutrient reductions were also available?
- How do results change in response to more or less complex program rules?

To examine these questions, we restrict our analysis to the same two main geographic areas ("basin-states") that were are the focus of Section 3. The first is the portion of the Susquehanna-PA, which includes 76 percent of the Susquehanna River basin area. The second is the James-VA, which constitutes over 99 percent of the James River basin. Table 2-1 shows the total annual TMDL nutrient load reduction targets for significant wastewater treatment and industrial facilities (PS) and agricultural nonpoint sources (AgNPS) in each basin-state. These two basin-states were selected in part because they provide an interesting contrast. In the Susquehanna-PA, the TMDL requires that a majority of reductions come from AgNPS, whereas in the James-VA, the PS load-reduction requirements are larger.

Although our analysis focuses on these two basin-states, our intention is not to model the existing trading programs in these two areas. Rather, we examine the implications of an alternative policy framework applied in both areas. We apply the same optimization framework, trading assumptions, and policy scenarios in the two areas, and we examine how the results differ due to their other features and attributes, such as the relative contribution of AgNPS and PS sources in the basin-state.

To examine the effects of trading baseline requirements, we assume and apply a *performance-based approach* in both areas. Using estimates from CBWM, we define baseline load reduction requirements for each agricultural land use category within each land-river segment that are consistent with the aggregate TMDL load reduction targets shown in Table 2-1. In each land-river segment and 2010 land use category (cropland, hay, and pasture), we compute the baseline requirement as the per-acre reduction in delivered nutrient loads from 2010 to TMDL conditions in the land use.[1] For each eligible BMP or combinations of BMPs available to be selected in the optimization framework, only the load reductions beyond this baseline are eligible to generate credits. For areas that are estimated to already be in compliance with TMDL conditions in 2010, all nutrient reductions for newly implemented BMPs in these units are assumed to be eligible to generate credits.

4.2.2 Additional Incentives through Public Sector Subsidies or Credit Purchases

In addition to developing and supporting water quality trading programs, the public sector can also provide direct incentives to agricultural nonpoint sources for reducing nutrient loads. A number of incentive-based programs already exist for farmers, for example, through USDA agricultural cost share programs. Expanded and more targeted programs by federal and state agencies would be needed to make additional progress towards the TMDL goals.

For this analysis, we consider two types of government incentives:

- Direct dollar-per-pound payments to farmers for reductions in nitrogen and phosphorus loads delivered to the Bay. All reductions in a basin-state receive the same subsidy.

[1] This calculation includes areas prescribed to be converted to other land uses in the TMDL.

- Nutrient credit purchases from farmers through participation in an expanded credit market that uses current (2010) practices as the trading baseline.

In the first case, the government is assumed to set the per-pound subsidy rate and then compensate farmers for all of their nonpoint source reductions up to their basin-level load allocation (from current conditions) at that rate. In the second case, the government would be required to purchase credits from nonpoint sources, but at the same market-based price, estimated endogenously in the optimization framework, as point source credit buyers. In the second case, the total number of credits purchased by the government would have to be equal to the total AgNPS load reduction targets for the basin. In both cases, the public sector is assumed to pay a constant per-pound amount for load reductions, which does not vary across BMPs.

We also model programs that use a combination of incentive payments and trading. As a way to deal with the trading baseline hurdle, government policies allow farmers to use payments to meet the trading baseline but not to generate credits for sale.

4.2.3 Incentive Scenarios

To address the research questions posed above, we apply the optimization framework[1] to analyze the following scenarios:

4.2.3.1 Reference Scenario. PS-PS Trading + Fixed AgNPS Subsidy

The purpose of this scenario is to provide a reference point for examining how AgNPS trading, baseline requirements and public sector payments affect total nutrient control costs, load reductions, and public sector spending. In this case, the incentives for AgNPS and PS load reductions are kept completely separate.

As a reference point for a larger trading market, we assume in this case that, to meet the total PS load reduction targets, trading is only allowed between point sources. We simulate trading between point sources with our optimization framework by solving for the least-cost combination of point source control projects in each basin-state that together achieve the point source reduction targets shown in Table 2-1. The marginal conditions (i.e., marginal nutrient reduction costs) at the model solution are interpreted as the market price for credits.

As a reference point for AgNPS payments, we assume that the government uses dollar-per-pound payments to farmers to meet the AgNPS load reductions targets. We assume that the dollar-per-pound subsidy rate, which is the same for all AgNPS in the basin-state, is set by the

[1] No agricultural land conversion constraints are applied to these scenarios.

government to exactly achieve the load reduction target. We simulate this system with our optimization framework by solving for the least-cost combination of available AgNPS BMPs in 2010 that together meet the relevant total AgNPS reduction target shown in Table 2-1. The marginal conditions (i.e., marginal nutrient reduction costs) at the model solution are interpreted as the subsidy rate required to exactly achieve the target.

4.2.3.2 Policy Scenario 1. PS-AgNPS Trading + No Subsidy

In this scenario, trading is allowed between PS and AgNPS to meet the PS load reduction target. However, to participate in the trading market, AgNPS must meet their performance-based load reduction baseline requirements—that is, they can only sell credits for reductions beyond this baseline. No new subsidy payments are provided by the public sector to AgNPS.

For this scenario, we simulate trading between point and nonpoint sources by solving for the least-cost combination of PS and AgNPS source control projects in each basin-state that together achieve the point source reduction targets shown in Table 2-1. To account for baseline requirements for agricultural nonpoint source, the cost minimization model only credits BMP alternatives for load reductions beyond their baseline, but it includes all of the costs of implementing the BMP alternative.

4.2.3.3 Policy Scenario 2. PS-AgNPS Trading + Subsidy ($2/lb N to $10/lb N)

This scenario is equivalent to Scenario 1 except that the public sector also provides fixed per-pound subsidy payments to farmers for load reductions that are less than or equal to the baseline required reductions. Three sub-scenarios are investigated, with payments of $2, $5, and $10 per pound of nitrogen. There is no "double-dipping" of payments because only the government purchases reductions to meet the baseline and only point sources purchase reductions (credits) beyond the baseline.

We simulate this scenario using the same approach as for Scenario 1; however, we include the subsidies to offset some of the costs of implementing the BMP alternative.

4.2.3.4 Policy Scenario 3. PS & Public Demand for AgNPS Credits

We include this scenario to some extent as an alternative point of reference. Like the Reference Scenario, it includes incentives to ensure that both the PS and AgNPS targets are fully achieved; however, rather than separating the sources into two separate incentive systems, they are combined into one trading system. Under this scenario, point sources and the public sector both participate in the nutrient market as demanders of AgNPS credits. The government's objective is to achieve the AgNPS load allocation by purchasing credits from farmers. However, rather than setting a price for load reductions, as in the Reference Scenario and Scenario 2, the public sector must compete with point sources for AgNPS credits and pay the market-determined price. In this case, there is no baseline requirement for AgNPS credits because the public sector's involvement in the credit market ensures that the AgNPS target is met. In other words, for farmers, all load reductions from current conditions are eligible for generating credits. Requiring the government to pay market prices is consistent with using reverse auctions in the sense that it helps to ensure that credits purchased are cost-effective. However, market prices could be more than traditional payments which often have a cost-share requirement.

We simulate this scenario by solving for the least-cost combination of point and nonpoint source control projects in each basin-state that together achieve the combined PS and AgNPS reduction targets shown in Table 2-1. Once again, the marginal conditions (i.e., marginal nutrient reduction costs) at the model solution are interpreted as the market price for credits, which applied to both PS and public sector credit buyers.

4.2.4 Results

Model simulation results for the three scenarios are shown in Table 4-1. To simulate the Reference Scenario, we ran the optimization model separately for the AgNPS and PS load reduction targets. For the PS-PS trading, we solve for the least-cost combination of installed PS treatment technologies required to meet the PS load reduction targets (4.7 million pounds of nitrogen and 0.3 million pounds of phosphorus in Susquehanna-PA). The resulting annual control cost estimate for point sources is $40 million.

For AgNPS, we solve for the least-cost combination of new agricultural BMPs that are needed to meet the AgNPS target, and we use the marginal conditions of this solution to provide estimates of the subsidy rates (market prices) for nitrogen and phosphorus reductions. Under this solution, the annual cost to farmers for implementing these BMPs is $92.3 million. The public sector purchases all of the required AgNPS load reductions—20.7 million pounds of nitrogen and 0.5 million pounds of phosphorus—for $208 million per year. Although not shown in Table 4-1, the model-estimated prices for nitrogen and phosphorus are $6.03/lb and $169.27/lb respectively. Under this scenario, farmers earn annual profits of $115.6 million for reducing nutrient loads[1].

[1] Farm profits are estimated by subtracting the cost of the nutrient control project from the revenue received for nutrient reductions based on the estimated nutrient market prices.

Table 4-1. Policy Scenario Results for the Susquehanna River Basin in Pennsylvania

	Total Control Cost (mil. $)		Load Reduction (mil. lbs/yr)						Revenue to AgNPS (mil. $)	
			Nitrogen			Phosphorus				
	AgNPS	PS	AgNPS Below Trading Baseline[a]	AgNPS Sold to PS	PS	AgNPS Below Trading Baseline[a]	AgNPS Sold to PS	PS	Subsidy	Credits
REFERENCE										
PS-PS Trading + Fixed AgNPS Subsidy	92.33	40.06	20.67	0.00	4.73	0.49	0.00	0.26	207.95	0.00
SCENARIO 1										
PS-AgNPS Trading + No Subsidy	12.61	11.09	2.29	2.27	2.45	0.02	0.04	0.21	0.00	18.37
SCENARIO 2										
PS-AgNPS Trading + $2/lb N Subsidy	20.00	8.18	5.62	2.68	2.05	0.08	0.05	0.21	11.24	16.58
PS-AgNPS Trading + $5/lb N Subsidy	48.24	7.50	14.30	2.76	1.96	0.14	0.06	0.20	71.49	13.64
PS-AgNPS Trading + $10/lb N Subsidy	62.96	6.42	16.33	2.99	1.73	0.16	0.06	0.20	163.30	13.04
SCENARIO 3										
PS & Public Demand for AgNPS Credits	106.78	15.92	20.67	1.95	2.77	0.49	0.01	0.25	235.43	15.93

[a] AgNPS TMDL load reduction target and trading baseline is 20.67 million pounds of nitrogen and 0.49 million pounds of phosphorus. This column shows the portion of the target that is achieved under each scenario (the remainder is unmet).

To simulate Scenario 1, we solve for the least-cost combination of agricultural BMPs and PS treatment technologies needed to meet the PS load reduction target; however, the AgNPS controls only receive credit for nutrient reductions beyond their trading baseline. By applying BMPs that cost $12.6 million per year, the AgNPS generate 2.3 million pounds of nitrogen credits and 0.04 million pounds of phosphorus credits, which they sell to PS for $18.4 million. Because these reductions are sold to PS, they do not contribute to the AgNPS load reduction targets under the TMDL. The PS sector incurs $11 million in control costs and $18.4 million in credit purchase costs each year, which translates to $10.6 million in savings for the PS sector compared to the Reference Scenario.

In addition to the load reductions sold as credits to the PS sector, the agricultural BMPs in Scenario 1 generate another 2.3 million pounds of nitrogen and 0.02 million pounds of phosphorus reductions. These reductions cannot be sold to PS, but they do contribute to the TMDL load reduction target for AgNPS. Unfortunately, these contributions are a relatively small percentage of the reduction needed to meet nitrogen and phosphorus reduction targets—11 percent and 4 percent.

For Scenario 2, we expand Scenario 1 to include different fixed payment subsidies for nitrogen reductions. While such a policy is not currently in place in the Chesapeake Bay, this is analogous in some respects to Maryland's Chesapeake Bay Restoration Fee (also known as the "flush tax"), which is partly used to fund a payment per acre of implemented cover crops. If the state were to estimate the pounds of nutrients reduced from the cover crop implementation, the farmer could be compensated for the estimated performance of the practice instead. These payments to AgNPS only apply to nitrogen reductions below their trading baseline, and they are treated in the optimization model as reductions in BMP costs. As expected, these subsidies lead to larger AgNPS load reductions, which also increase as the subsidy is raised from $2/lb to $10/lb of nitrogen. However, even with a $10/lb subsidy and an annual cost to the public sector of $163 million, the portion of these load reductions that can be applied to baseline requirements 16.3 million pounds of nitrogen and 0.16 million pounds of phosphorus, which only achieves 79 percent of the nitrogen target and 33 percent of the phosphorus target. Meanwhile, the PS purchasers also benefit indirectly from the subsidies through lower credit prices of $3.80/lb N and $27.42/lb P compared to $17.95/lb N and $180.75/lb P under Scenario 1.

For Scenario 3, we simulate a market that combines public sector and PS demand for credits. We remove the trading baseline requirement for AgNPS, and we solve for the least-cost combination of agricultural BMPs and PS treatment technologies that meets the combined load reduction targets for AgNPS and PS. To meet the combined targets, this scenario results in the highest annual public sector spending for load reductions ($235.4 million) and highest annual cost for AgNPS controls ($106.8 million). However, compared to the Reference Scenario, which also meets the combined targets, it results in lower total control costs. The combined control cost for AgNPS and PS sources is $122.7 million for Scenario 3, compared to $132.4 million for the Reference Scenario. Through nutrient credit trading, the benefits of this $9.7 million reduction in total costs are spread between AgNPS and PS, with most (85%) going to PS.

Table 4-2 reports results of these same scenarios for the James River basin in Virginia. Overall, the results are qualitatively very similar to those from the Susquehanna River Basin in Pennsylvania. Without a subsidy (Scenario 1), trading between PS and AgNPS provides an incentive for AgNPS to partially achieve their TMDL reduction targets. When subsidies are added (Scenario 2), the AgNPS get closer but not completely to their targets, even with a $10 per pound subsidy. When subsidies are replaced with a combined market for credits (Scenario 3), it results in the higher spending by the public sector, but more overall savings in control costs compared to the Reference Scenario.

One of the main differences between the results for the two basins is that nutrient trading in the James-VA basin gets AgNPS closer to their load reduction targets than in the Susquehanna-PA basin. For example, even without a subsidy (Scenario 1), AgNPS achieve 35 percent of their nitrogen target and 41 percent of their phosphorus target in the James, compared to 11 percent and 4 percent, respectively, in the Susquehanna. One reason for this difference is the higher percentage of the total load reduction placed on PS in the James, which translates to a higher demand for credits from AgNPS compared to the AgNPS required reductions.

4.3 Impacts of Alternative Methodologies for Estimating Nutrient Reductions from Nonpoint Source Controls

A concern with trading is that nonpoint-source control practices are less reliable and measurable than point-source controls. Thus, the development of a nutrient trading market between point and nonpoint sources requires methods for measuring or estimating nutrient reductions from nonpoint sources. Unfortunately, monitoring the actual nutrient reductions from the implementation of a specific BMP through in field and/or in stream measurements is usually costly. As a result, methods to calculate nutrient credits from nonpoint sources generally rely on estimated nutrient reductions associated with specific BMP implementation. In this section, we use the optimization framework to analyze how alternative methodologies influence both the estimated environmental outcome and the potential cost savings from nutrient trading with nonpoint sources.

Table 4-2. Policy Scenario Results for the James River Basin in Virginia

	Total Control Cost (mil. $)		Load Reduction (mil. lbs/yr)						Revenue to AgNPS (mil. $)	
			Nitrogen			Phosphorus				
	AgNPS	PS	AgNPS Below Trading Baseline[a]	AgNPS Sold to PS	PS	AgNPS Below Trading Baseline[a]	AgNPS Sold to PS	PS	Subsidy	Credits
REFERENCE										
PS-PS Trading + Fixed AgNPS Subsidy	7.80	109.17	1.39	0.00	9.57	0.34	0.00	0.60	14.21	0.00
SCENARIO 1										
PS-AgNPS Trading + No Subsidy	10.42	75.70	0.49	0.75	8.82	0.14	0.24	0.36	0.00	18.10
SCENARIO 2										
PS-AgNPS Trading + $2/lb N Subsidy	10.54	75.70	0.58	0.75	8.82	0.17	0.24	0.36	1.16	17.03
PS-AgNPS Trading + $5/lb N Subsidy	15.45	71.79	0.87	0.94	8.62	0.23	0.24	0.36	4.35	23.50
PS-AgNPS Trading + $10/lb N Subsidy	16.19	71.79	0.98	0.94	8.62	0.26	0.24	0.36	9.76	22.22
SCENARIO 3										
PS & Public Demand for AgNPS Credits	20.28	75.57	1.39	0.75	8.82	0.34	0.25	0.34	33.57	18.50

[a] AgNPS TMDL load reduction target and trading baseline is 1.39 million pounds of nitrogen and 0.34 million pounds of phosphorus. This column shows the portion of the target that is achieved under each scenario (the remainder is unmet).

4.3.1 *Estimating Nutrient Reductions from Nonpoint Source Controls*

Nutrient reductions from implementing a BMP or set of BMPs on a farm depend on several site-specific factors, such as the type of soil present, the slope of the land, the timing and magnitude of rain events, and existing management practices (Sharpley et al, 2009; Simpson and Weammart, 2009). Methodologies used to estimate nutrient reductions for a nutrient trading market could take all relevant site-specific factors into consideration, allowing nutrient credits generated by a BMP to vary farm by farm.[1] Or, methodologies could rely on an average expected performance across a wider region, where a BMP would generate the same number of nutrient credits at every farm.

To estimate nutrient reductions towards meeting the TMDL, CBWM relies on average expected performance values based on numbers for correctly implemented BMPs in each land-river segment. Nutrient trading programs within the Chesapeake Bay watershed have adopted methods designed to be consistent with CBWM; however, they vary in the level of detail included to calculate the number of credits generated by a BMP. Crediting methodologies developed for Maryland, Pennsylvania, and West Virginia rely on BMP performance efficiencies from CBWM and also include site-specific information such as fertilizer and manure application data (Branosky et al., 2011). In contrast, Virginia has adopted a uniform method of crediting nutrient reductions. For every eligible BMP, such as early cover crops, the number of nutrient credits it generates within a river basin only depends on whether it is installed east or west of I-95 (Table 4-3), regardless of other local conditions that could be considered (VDEQ, 2008).

The methods used to calculate how many credits a BMP generates can influence both the environmental outcome and the potential cost savings from nutrient trading (Table 4-4). For example, if uniform regional crediting, as opposed to site-specific crediting, were to result in placing the majority of BMPs on areas that generate below-average nutrient reductions, then uniform crediting could result in a failure to achieve the required nutrient reductions. Alternatively, if it would result in BMPs being distributed evenly across the landscape, such that their average performance is equal to the uniform credited value, then nutrient targets would be met. The major advantage of using regional averages is that simpler calculations would be expected to lower costs of market transactions compared to systems that required lots of farm-specific information.

[1] Selective monitoring of farms would also help validate BMP performance more generally.

Table 4-3. Nutrient Credits Generated by Best Management Practices in the James River Basin in Virginia

	West of I-95		East of I-95	
Best Management Practice	**Nitrogen Credited (lbs/acre)**	**Phosphorus Credited (lbs/acre)**	**Nitrogen Credited (lbs/acre)**	**Phosphorus Credited (lbs/acre)**
Early Planted Cover Crops	0.54	0.00	0.91	0.00
Enhanced Nutrient Management	1.75	0.00	3.70	0.00
Continuous No-Till	1.05	0.49	1.13	0.19
Tree Planting on Cropland	5.48	1.22	9.34	0.93
Cropland Retirement	3.44	0.33	3.08	0.00

Source: VDEQ, 2008

Table 4-4. Potential Impacts of Uniform Crediting on Environmental Outcomes and Cost of Source Controls under Nutrient Trading

BMP Placement under Uniform Crediting	Environmental Outcome	Cost of Source Controls
Areas with Above Average Nutrient Reductions	Above Nutrient Target	Increase Relative to Site-Specific Crediting
Areas with Below Average Nutrient Reductions	Below Nutrient Target	Decrease or Increase Relative to Site-Specific Crediting

4.3.2 Alternative Crediting Scenarios

To estimate the impacts of alternative methodologies of estimating nutrient reductions, we use the optimization framework to analyze the following scenarios:

4.3.2.1 Scenario A. PS-AgNPS Trading with Site-Specific Crediting

In this scenario, nutrient reductions from PS upgrades and AgNPS source controls beyond baseline (the same AgNPS inputs described in Section 4.2) can contribute to meeting the PS nutrient reduction requirement. We apply the optimization framework to identify the least-cost solution for meeting the PS load reduction targets in each basin-state.[1] To represent site-specific crediting for agricultural BMPs, we use the nutrient load data and BMP performance assumptions from CBWM. In other words, we use the same approach described in previous

[1] In addition, agricultural land conversion is restricted to 25 percent to represent policies used to prevent loss of agricultural lands. (Wainger et al., 2013).

sections of this report to estimate load reductions for AgNPS projects. CBWM uses a modeling approach to calculate load reductions; therefore, it involves simplifications and does not provide the same level of site-specific information that might be achieved with direct monitoring. Nevertheless, it does account for differing conditions across land use categories and land-river segments, which contribute to variation in BMP load reductions.

4.3.2.2 Scenario B. PS-AgNPS Trading with Uniform Crediting

For this scenario, we again use the optimization framework to identify the least-cost solution for meeting the PS load reduction targets in each basin-state. In contrast to Scenario A, nutrient reductions for AgNPS BMPs are estimated to be uniform within a basin-state.[1] This uniform value is set at the mean value from CBWM for each basin-state. For example, each acre of enhanced nutrient management is assigned the same nutrient reductions throughout the Susquehanna River Basin in Pennsylvania.

4.3.3 Results

Applying the optimization framework across the 14 basin-states in the watershed, overall we find that uniform crediting results in fewer AgNPS credits being generated compared to site-specific crediting. As shown in Table 4-5, the uniform crediting scenario results in 3.1 million pounds of nitrogen credits and 295 thousand pounds of phosphorus credits being sold to PS buyers per year, compared to 4.1 million and 227 thousand pounds, respectively, under site-specific crediting.

Table 4-5 also shows how, under a uniform crediting approach, credited and “actual” load reductions may differ from each other. In this application, “actual” load reductions are estimated using the CBWM method (i.e., the same approach used for site-specific crediting). With this approach, we find that, across all basin-states, actual reductions are 8 percent lower than credited reductions for nitrogen and 23 percent lower for phosphorus. However, these results vary across basin-states. In several cases, especially for phosphorus, we find that actual reductions are higher than credited, which means that in these areas BMPs are placed such that their average load reductions are higher than the uniform rate for the basin-state.

Overall, we find that uniform crediting results in lower actual nutrient load reductions from AgNPS than site-specific crediting (by 30 percent for nitrogen and 34 percent for phosphorus) and higher costs for achieving these reductions (by 8%). This result occurs because site-specific crediting encourages BMP placement for nutrient trading in areas where they produce relatively

[1] The one exception to this is for practices, such as forest buffers, where effectiveness varies by hydrogeomorphic region. For these BMPs, the effectiveness varied by hydrogeomorphic region.

high nutrient reductions and are therefore more cost-effective (assuming costs are not positively correlated with removal efficiencies). In contrast, uniform crediting does not provide this type of incentive. It should also be noted that, although uniform crediting results in higher costs in every basin-state, in a few cases it also results in greater load reductions; however, even in these cases the overall cost-effectiveness (load reduction per dollar) is higher under site-specific crediting.

Table 4-5. Actual[a] vs. Credited AgNPS Load Reductions and Control Costs under Alternative Crediting Approaches

		Nitrogen (1,000 pounds)			Phosphorus (1,000 pounds)				
		Uniform Crediting		Site-Specific Crediting	Uniform Crediting		Site-Specific Crediting		
Major Basin	Jurisdiction	Credited Load Reductions	Actual Load Reductions	Credited and Actual Reductions	Credited Load Reductions	Actual Load Reductions	Credited and Actual Reductions	Uniform Crediting Scenario Cost (million $)	Site-Specific Crediting Scenario Cost (million $)
Eastern Shore	Delaware	16	17	16	1	1	1	$0.1	$0.1
Eastern Shore	Maryland	74	87	271	2	3	15	$5.7	$2.8
Eastern Shore	Virginia	15	14	15	2	2	2	$0.3	$0.3
James	Virginia	379	359	618	126	69	164	$93.9	$90.1
Patuxent	Maryland	1	1	0	0	0	0	$0.3	$0.3
Potomac	Maryland	33	23	33	5	6	6	$2.7	$2.1
Potomac	Pennsylvania	29	23	56	2	1	2	$1.0	$0.8
Potomac	Virginia	571	467	710	82	67	64	$6.0	$4.2
Potomac	West Virginia	110	77	110	11	9	11	$1.0	$0.9
Rappahannock	Virginia	55	47	55	10	13	12	$0.6	$0.4
Susquehanna	New York	203	204	263	20	21	20	$7.7	$6.7
Susquehanna	Pennsylvania	1,514	1,456	1,815	22	25	32	$31.1	$27.3
Western Shore	Maryland	1	0	1	0	0	0	$40.1	$40.0
York	Virginia	132	100	132	13	11	12	$3.1	$2.9
Total		**3,132**	**2,876**	**4,094**	**295**	**227**	**342**	**$193.6**	**$179.0**

[a] "Actual" load reductions are modeled (based on the CBWM methods) rather than monitored values.

SECTION 5
CONCLUSIONS

In this report we adapt and apply an economic optimization framework to analyze strategies for achieving the goals of the Chesapeake Bay TMDL. With this framework, we conduct two main analyses.

The purpose of the first analysis is to expand the existing framework, which includes costs and selected co-benefits (i.e., carbon sequestration and hunting recreation benefits) of nutrient control practices, to include monetized benefit estimates for improvements in freshwater quality in the watershed. Using a benefit transfer approach, we first develop estimates of the average (per-pound) value of reducing edge-of-stream nitrogen and phosphorus loads in each nontidal river segment of the watershed. These values represent approximations of households' average willingness to pay for the resulting freshwater quality improvements in their own state. We find that these per-pound values are generally higher in the more upstream sections of the watershed, which reflect the relatively low water flow in these segment and the relatively high number of downstream segments affected.

With this expanded framework, we then analyze and compare three scenarios for achieving the TMDL load reduction scenarios in selected basins: (1) a TMDL scenario based on the states' WIPSs (i.e., no optimization), (2) a least-cost optimization scenario, and (2) a least-net-cost optimization scenario. In all cases, we find that the benefits from improving freshwater quality in the watershed (separate from the water quality benefits for the Bay itself) are greater than the carbon and hunting co-benefits combined. Comparing the least-cost and least-net-cost scenarios, we also find the latter scenario results in greater nutrient control efforts in the more upstream portions of the watershed, which is consistent with the higher per-pound values for freshwater benefits.

The results from this first analysis indicate that, although the purpose of the TMDL is to improve water quality in the Bay estuary, many measures to achieve this goal will also provide significant upstream water quality benefits. Therefore, **providing additional incentives for delivered load reductions that originate farther upstream may improve the overall efficiency (in a net-cost sense) of meeting TMDL goals**. However, it must be emphasized that the per-pound value estimates for upstream load reductions are based on a linear approximation derived from a single watershed-wide load reduction scenario. Additional sensitivity analyses, including the use of alternative load reduction scenarios to generate the per-pound values, will be needed to determine the robustness of these estimates. In addition, this analysis does not include nutrient controls from other sectors, in particular urban stormwater BMPs. Although those BMPs tend to be less cost-effective and agricultural BMPs, they also offer distinct co-benefits (e.g., flood control and air quality improvements). Future analyses could examine how including these sources and BMPs alters our findings.

For the second analysis, we use the optimization framework to analyze the implications of different nutrient trading and incentive-based approaches. In particular, we investigate (1) how nutrient trading may interact with other incentives for agricultural nutrient reductions and (2) how simplified crediting of nutrient reductions influences the control costs, load reductions, and participation in a nutrient trading market.

We find that, although nutrient trading can act as an incentive for some agricultural entities to adopt nutrient controls and meet their load allocations (i.e., trading baseline) under the TMDL, these incentives would only support a portion of the required agricultural load reductions. In other words, **these results indicate that nutrient trading is not a particularly effective mechanism for encouraging the agricultural sector to meet its TMDL goals**. In the Susquehanna-PA, trading would only incentivize agricultural NPS to achieve 11 percent or less of required reductions. In contrast, in the James-VA, we estimate nutrient trading would be more effective, with 35 percent of the nitrogen reduction and 41 percent of the phosphorus reduction achieved through nutrient trading. One reason for this difference is the higher percentage of the total load reduction placed on PS in the James. This difference translates to a relatively high demand for credits from AgNPS.

Given this gap between achieved and required load reductions with trading alone, we examine how additional incentives from public subsidies could alter these outcomes. We find that per-pound subsidies could help to narrow this gap, but at a relatively high budgetary cost for the public sector. A “combined market,” where the public sector competes with PS for credits, would be the most economically efficient approach for achieving both PS and AgNPS targets, but the budgetary costs of this approach are likely to be prohibitive.

Finally, we explore how simplified crediting approaches for nutrient reductions would affect trading outcomes. In the case examined, we estimate that a simplified approach results in higher costs (by 8 percent across the watershed) of achieving significant wastewater and industrial discharges nutrient reductions. Unlike the site-specific approach the simplified uniform crediting approach does not encourage placement of nutrient controls where they would be most cost-effective for reducing nutrients. In addition, simplified crediting of nutrient trading is estimated to result in failure to meet the load reduction requirements in 11 of the 14 basin-state combinations in the Chesapeake Bay watershed. The shortfall occurs because, in these cases, the simplified approach results in certain agricultural areas receiving more credit for nutrient reductions than are actually achieved.

While these findings provide potentially important insights for designing and evaluating incentive-based approaches for achieving the TMDL, it is important to interpret them with certain caveats in mind. Most importantly, even with adjustments for transaction costs, the optimization framework offers a somewhat idealized representation of credit markets. Due to uncertainties and real world market frictions, in practice credit buyers and sellers are unlikely to take advantage of all the cost saving opportunities available. Therefore, the cost estimates generated with the optimization framework should be interpreted as lower bound values. The framework also provides a simplified representation of the load reduction options in the watershed. For example, it does not include all of the possible agricultural and urban stormwater BMPs that can be used to achieve the TMDL goals. Future analyses would benefit from an expanded framework that includes a larger set of BMP options.

APPENDIX A.
CO-BENEFITS FROM AGRICULTURAL BEST MANAGEMENT PRACTICES

This appendix describes the methods used to estimate specific ancillary benefits resulting from implementing agricultural best management practices (BMPs) to meet the Total Maximum Daily Load (TMDL). The methods described in this section are based, to a large extent, on those developed and applied by EPA's Office of Research and Development to quantify the ecosystem services from these practices (U.S. EPA, 2011).

A.1 Carbon Sequestration and Changes in Greenhouse Gas Emissions

To predict the carbon-related benefits of agricultural BMPs, it is necessary to calculate both the change in the total amount of greenhouse gases (GHGs) emitted and the change in amount of carbon sequestered.

A.1.1 Estimation of GHG Emissions

We identified three main types of GHGs whose emissions can be estimated for selected land use/land cover categories. Expressed in the common unit of carbon dioxide (CO_2) equivalents using the most recent Intergovernmental Panel on Climate Change (IPCC) (2013) estimates of global warming potential, they are the following:

CO_2 = 1 CO_2 equivalent

CH_4 = 28 CO_2 equivalents (global warming potential for 100 years)

N_2O = 265 CO_2 equivalents (global warming potential for 100 years).

CO_2 emissions occur as a result of decomposition and aerobic degradation and can be temporarily accelerated following conversion of lands to wetlands. Methane (CH_4), a product of anaerobic degradation, also commonly occurs in wetlands because of the low oxygen availability with a high water table. Nitrous oxide (N_2O) emissions are most common with croplands, with higher emissions associated with crops such as corn that require nitrogen fertilization, unlike nitrogen-fixing crops such as soybeans.

Two main reference sources were used to identify GHG emission rates for this exercise. First, the Forest and Agricultural Sector Optimization Model (FASOM) (Adams et al., 1996) was used for crop and pasture N_2O emission rates. FASOM was initially developed to evaluate welfare and market impacts of alternative policies for sequestering carbon in trees, but also has been applied to a wider range of forest and agricultural sector policy scenarios (http://www.treesearch.fs.fed.us/pubs/viewpub.jsp?index=2876). N_2O emission rates were identified in FASOM's March 2010 version.

Second, the IPCC *2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 4—Agriculture, Forestry and Other Land Use* (henceforth referred to as the IPCC 2006

Guidelines) was referenced to identify CO_2 and CH_4 emission rates, where available, for wetlands.

Table A-1 lists the GHG emission rates used in this analysis.

Table A-1. Assumed GHG Emission Factors for Selected Land Uses

Land Use	CO_2	N_2O	CH_4
Cropland	0 kg C/ha-yr[a]	[b]	0[c]
Pasture	0[f]	[g]	0[h]
Wetland	13.55 lb/ac-day (15.2 kg CO_2/ha-day[d])	0	0.54 lb/ac-day (0.061 kg CH_4/ha-day[e])
Forest[i]	0	0	0

[a] Assumes no crop burning (negligible; EPA GHG Inventory report [U.S. EPA, 2010] assumes only 3% of crops in the United States are burned).

[b] Crop-specific N_2O emission factors reported in U.S. EPA(2011).

[c] Assumes no rice grown (only crop that emits CH_4) and no crop burning (negligible; EPA *GHG Inventory* report [U.S. EPA, 2010] assumes only 3% of crops in the United States are burned).

[d] Source: IPCC, 2006 v.4 , App 2, Table 2A.2.

[e] Source: IPCC, 2006 v.4, App 3, Table 3A.2.

[f] Source: IPCC, 2006 v.4.

[g] Crop-specific N_2O emission factors reported in U.S. EPA(2011).

[h] Assumes zero CH_4 emissions for this analysis (pasture emissions from enteric fermentation depend on herd size).

[i] Assumes no thinning or harvesting for this analysis.

A.1.1.1 Cropland and Pastureland Emissions

To estimate crop-based GHG emissions, county-based predominant crop types can be identified using U.S. Department of Agriculture (USDA) National Agriculture Statistics Service (NASS) data. These data can be combined with FASOM's N_2O emission rates, which are reported by crop and by state. These rates range from 0.002 to 0.009 ton N_2O/ha-year.

The emission sources included in the N_2O emission estimates include:

- nitrogen fertilizer application practices under managed soil categories under AgSoilMgmt,
- emissions from nitrogen-fixing crops,
- emissions from crop residue retention,
- indirect soils volatilization, and
- indirect soils leaching runoff.

FASOM also includes N_2O emission rates for pasture, which were used in this analysis.

A.1.1.2 Wetland Emissions

For this analysis, we used the IPCC emission estimates for flooded lands (IPCC, 2006) to estimate CH_4, CO_2, and N_2O emissions associated with wetland restoration or construction in the Chesapeake Bay,[1] as described below.

- **Methane—**A CH_4 emission rate of 0.54 lb/ac-day (0.061 kg CH_4/ha-day) represents the median diffusive emission rate of CH_4 for flooded land located in a cold temperate, moist climate (IPCC, 2006, Appendix 3). When using this emission rate, expressed in kilograms of CH_4 per hectare per day, annual emissions should exclude days with ice cover, because CH_4 emissions are reduced dramatically when wetland waters are frozen. The number of ice-free days can be determined by the mean number of days with minimum temperatures 32° F or less for cities within the Chesapeake Bay watershed. This value is 257 days based on 37 to 73 years of data from seven cities (NOAA, 2010).

- **Carbon Dioxide—**A CO_2 emission rate of 13.55 lb/ac-day (15.2 kg CO_2/ha-day) was selected by the IPCC to represent the median diffusive emission rate of CO_2 for flooded land located in a cold temperate, moist climate (IPCC, 2006, Appendix 2). This daily emission rate should only be applied for the first 10 years after flooding (i.e., the first 10 years following conversion to wetland), and the annual emission estimate should exclude days of the year with ice cover.

A.1.1.3 Changes in GHG Emissions from Agricultural BMPs

For agricultural BMPs involving land conversion away from cropland or pastureland, we assumed that GHG emissions are *reduced* according to the rates reported in Table A-1. For the wetland conversion BMP, we also estimated an offsetting *increase* in GHG emissions, based on the wetland emission rates in Table A-1.

In addition, we estimated a reduction in GHG emission for “working land” BMPs that reduce fertilizer application. For the decision agriculture and enhanced nutrient management BMP, the Chesapeake Bay Watershed Model (CBWM) assumed a reduction in fertilizer application of 7.5 percent and 15 percent, respectively. For these BMPs, we therefore also assumed a reduction in N_2O emissions by 7.5 percent and 15 percent, respectively.

A.1.2 Estimation of Carbon Sequestration

For this analysis, we also estimated carbon sequestration for the BMPs involving land conversion to forests, wetlands, or grasslands. Conversion to forests will result in accumulation or sequestration of carbon in aboveground and belowground vegetation, as well as soil pools

[1] The IPCC chose **not** to recommend emission rates specifically for wetlands because of a lack of wetlands research at the time of publication. The IPCC’s *2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4—Agriculture, Forestry and Other Land Use* reports “Some uses of wetlands are not covered in the report because adequate methodologies are not available. These include ‘rewetting of previously drained wetlands’ and ‘wetland restoration’” (Section 7.3.2.1). However, wetlands can be significant sources of GHGs; therefore, we included them in the analysis as described in this section.

during stand development. Conversion to wetlands will sequester carbon in vegetation and soils, with a large amount of carbon accumulating in the soils because of higher water tables and anoxic conditions, which slows decomposition. Conversion to grasslands also will result in carbon sequestration, mostly below ground.

The following five-step process was used to estimate sequestered carbon:

- **Step 1: Determine predominant forest type by ecoregion by county.** County-level forest cover within the Chesapeake Bay watershed was calculated with the U.S. Forest Service (USFS) National Forest Type Dataset and Omernik ecoregions (Omernik, 1987). A total of eight Omernik ecoregions overlap the CBW.

- **Step 2: Select tree species.** For crop or pastureland converted to forest, we assumed that the land would be planted with the main tree species found in the dominant forest type of each ecoregion. Conversion to wetlands was assumed to involve planting of the wetland area with a bald cypress/water tupelo forest type (Neely, 2008). For land retirement, land would naturally regenerate to an even mixture of all forest types found within the ecoregion.

- **Step 3: Obtain carbon sequestration rates by tree species and ecoregion.** The National Council for Air and Stream Improvement/USFS Carbon On-Line Estimate (COLE) was used to calculate total carbon stocks. Estimates were made for the forest types assigned in Step 2. The total no-soil carbon storage values reported for 5- to 10-year increments during years 0 to 90 were combined with the total soil carbon values to produce "total carbon sequestered."

- **Steps 4 and 5: Create tables of sequestered carbon by county and land-use categories, and apply estimates to modeled scenarios.** Applying Steps 1 through 3 described above and assigning counties to their respective main ecoregions, we calculated carbon sequestration rates by county for the land-use conversion from cropland and pastureland to (1) forest, (2) wetlands, (3) natural revegetation, and (4) grassland. The carbon estimates produced for each land-use conversion scenario were compiled by county as 5-year sequestration rates (tons of carbon per acre per 5-year period) over a 90-year term.

In addition to the agricultural BMPs involving land conversion described above, other agricultural BMPs included in the model also have an effect on carbon sequestration. Below we describe how these effects are included (or not included).

Management practices that implement varying levels of tillage are expected to impact soil carbon pools. Full tillage reduces soil carbon, whereas the absence of tillage increases carbon sequestration (Ogle et al., 2005). Therefore, changes in soil carbon were estimated using three (low, high, and no) tillage levels and the methodology outlined in IPCC (2006). It was assumed that all cropland in the modeled areas of the Chesapeake Bay watershed would be planted with perennial crops; the Bay is subject to a moist, temperate climate regime; and the soils would consist of high and low activity clays (equal amounts of each).

Based on these conditions, the native soil carbon pools in the Bay soils were estimated to be 75.5 metric tons C/ha in the top 30 cm. It was also assumed that all modeled agricultural lands have been subject to long-term cultivation. Full (1.0), reduced (1.08), and no-till (1.15) relative stock change factors were used to determine the influence of different tillage levels on soil carbon over a 20-year period. The impacts of different levels of residue return or input on soil tillage carbon sequestration were not considered or included in the carbon sequestration estimates.

A.1.3 Valuation of Carbon Sequestration and Reduced GHG Emissions

The ecosystem services associated with carbon sequestration and avoided GHG emissions can be valued using estimates of the average avoided damages that would otherwise result from a release of 1 metric ton of carbon to the atmosphere (also referred to as the social cost of carbon [SCC]). For the benefits of the Chesapeake Bay TMDL analysis, we relied on a recommended mean SCC for 2010 using a 3 percent discount rate (Interagency Working Group on Social Cost of Carbon, 2013). From this, we assumed a value of $34.71 (2010$) per metric ton of CO_2 equivalent emissions reduced, or $127.18 per metric ton of C.

Applying this estimate of SCC to the estimated time paths of carbon flux reported**,** we calculated the present value of carbon storage associated with each land-use conversion category using a 3 percent discount rate and an annualized value of carbon storage for each acre of land conversion determined. These estimates are reported in Table A-2.

Table A-2. Per-Acre Value of Carbon Sequestration Services from Land-Use Conversion ($/ac)

	Present Value[a]				Annualized Value[a]			
Ecoregion	To Forest	To Wetland	To Grass Buffer	Land Retirement	To Forest	To Wetland	To Grass Buffer	Land Retirement
Per Acre Value of Carbon Sequestration Services from Cropland Conversion ($/ac)								
Central Appalachians	$3,021	$3,841	$308	$2,539	$97	$124	$10	$82
Middle Atlantic Coastal Plain	$3,700	$3,841	$308	$3,116	$119	$124	$10	$101
North Central Appalachians	$3,021	$3,841	$308	$2,539	$97	$124	$10	$82
Northern Appalachian Plateau and Uplands	$3,259	$3,841	$308	$2,749	$105	$124	$10	$89
Northern Piedmont	$4,123	$3,841	$308	$2,950	$133	$124	$10	$95
Piedmont	$4,101	$3,841	$308	$3,109	$132	$124	$10	$100
Ridge and Valley	$2,802	$3,841	$308	$2,386	$90	$124	$10	$77
Southeastern Plains	$4,185	$3,841	$308	$3,205	$135	$124	$10	$103

[a] 90-year period; 3% discount rate.

Table A-2. Per-Acre Value of Carbon Sequestration Services from Land-Use Conversion ($/ac) (continued)

	Present Value[a]				Annualized Value[a]			
Ecoregion	To Forest	To Wetland	To Grass Buffer	Land Retirement	To Forest	To Wetland	To Grass Buffer	Land Retirement
Per Acre Value of Carbon Sequestration Services from Pastureland Conversion ($/ac)								
Central Appalachians	$2,747	$3,841	$0	$2,698	$89	$124	$0	$87
Middle Atlantic Coastal Plain	$3,403	$3,841	$0	$3,378	$110	$124	$0	$109
North Central Appalachians	$2,747	$3,841	$0	$2,698	$89	$124	$0	$87
Northern Appalachian Plateau and Uplands	$2,985	$3,841	$0	$2,934	$96	$124	$0	$95
Northern Piedmont	$3,836	$3,841	$0	$3,200	$124	$124	$0	$103
Piedmont	$3,815	$3,841	$0	$3,382	$123	$124	$0	$109
Ridge and Valley	$2,601	$3,841	$0	$2,529	$84	$124	$0	$82
Southeastern Plains	$3,897	$3,841	$0	$3,492	$126	$124	$0	$113

[a] 90-year period; 3% discount rate.

A.2 Waterfowl Hunting Services from Wetland Restoration

For this analysis, we used a methodology adapted from Murray et al. (2009), who estimated the effects of wetland restoration in the Mississippi Alluvial Valley on waterfowl hunting services.

The first step is to develop a model for estimating energetic carrying capacity of the CBW for ducks. To accomplish this, we applied a "duck energy day" (DED) model. DEDs are the number of ducks that can meet their daily energy requirements from an area of foraging habitat for a single day (Lower Mississippi Valley Joint Venture Management Board [LMVJV] Waterfowl Working Group, 2007). The first step is to calculate DEDs per acre provided by a specific habitat (i.e., land use) using the following equation:

$$\frac{(\text{Food density kg/ac} - 20.24\text{ kg/ac})*(1{,}000\text{ g/kg})*\text{TME kcal/g}}{\text{DER (294.35 kcal/day)}} \tag{1.1}$$

where

food density = The food available in kilograms per acre in a given foraging habitat; the value 20.24 kg/ac is subtracted from the food available because ducks do not forage in habitats where finding food becomes difficult

TME = true metabolizable energy of waterfowl foods in kilocalories per gram

DER = daily energy requirement per duck, assumed to be 294.35 kilocalories per day for a dabbling duck

Based on a review of the literature, we selected the parameter values reported in Table A-3 for estimating DEDs from various land-cover types in the Chesapeake Bay watershed. Food density estimates for wetland habitats were derived from a study of American black duck carrying capacity in the Chesapeake Bay of Virginia (Eichholz and Yerkes, 2008). The resulting average DEDs per acre vary from 34 for soybean cropland to 1,098 for tidal wetlands.

Table A-3. Duck Energy Days per Acre for Selected Land Cover Types

Land Cover	Food Available (kg/ac)	TME[a] (kcal/g)	DER[a] (k/cal)	DEDs/ac
Cropland				
Corn	61[a]	3.67	294.35	508
Soybean	24[a]	2.65	294.35	34
Wetland				
Freshwater	107[b]	2.47	294.35	482
Tidal	194[b]	2.47	294.35	1,206

[a] LMVJV (2007).

[b] Eichholz and Yerkes (2008) combines food from seeds and invertebrates and, for tidal wetlands, is an average value for the brackish water, salt marsh, and mudflat categories.

The second step is to estimate baseline DEDs in the Chesapeake Bay watershed. We accomplished this step by multiplying the number of acres in each land-cover category by the corresponding DED/acre estimates. The third step is to estimate the baseline value of duck hunting services, multiplying the total number of duck hunting days by state (based on Richkus et al., 2008) by the regional average consumer surplus value of a duck hunting day (Rosenberger and Loomis, 2001).

The final step is to estimate the increase in the value of duck hunting services associated with each acre converted from cropland or pastureland to freshwater or tidal wetland. For this step, we assumed that the aggregate value of duck hunting in each state increases in direct proportion to the increase in total DEDs. The results of this step are reported in Table A-4.

Table A-4. Incremental Annual Value of Duck Hunting Services per Acre of Wetland Restoration (2010 $)

	Type of Land-Use Conversion			
State	Cropland to Tidal Wetland	Cropland to Freshwater Wetland	Pastureland to Tidal Wetland	Pasture to Freshwater Wetland
DE	$7.19	$3.07	$8.28	$4.16
MD	$7.65	$3.38	$8.60	$4.33
NY	NA	$3.16	NA	$6.85
PA	NA	$2.27	NA	$4.06
VA	$3.83	$1.71	$4.26	$2.14
WV	NA	$0.95	NA	$1.61

A.3 Nonwaterfowl Hunting Services from Increases in Forest Cover

To estimate the effects of land-use/land-cover change on other hunting services, a hedonic price study of hunting leases by Shrestha and Alavalapati (2004) was applied to the Bay. Although this study was conducted in central Florida, it is geographically the closest study that has estimated the effect of different types of land cover on hunting values. Using 2002 data for 74 parcels, the study found that the percentage of land under forest (i.e., tree and vegetation) cover had a positive and statistically significant effect on the value of leases. In particular, they estimated a price elasticity of 0.132 with respect to forest cover.

To apply the Shrestha and Alavalapati (2004) results to the Bay watershed, their estimated price elasticity was assumed to also reflect the incremental contribution of forest cover to nonwaterfowl hunting values in the Bay watershed. Specifically, the following relationship was used to estimate the increase in hunting services associated with increases in forest cover:

$$\Delta HS_i = D_i \times V_i \times \alpha \times (100 \times \Delta F_i / Li) \tag{1.2}$$

where

ΔHS_i = increase in the aggregate value of hunting services in the Chesapeake Bay watershed from state i in 2010 dollars

D_i = annual number of nonwaterfowl hunting days in the watershed in state i in 2008

V_i = average value (consumer surplus) of a nonwaterfowl hunting day in state i in 2010 dollars

α = elasticity of hunting value with respect to forest cover

ΔF_i = increase in acres of forest land cover in the watershed in state i due to land-use conversion

L_i = acres of land in the watershed in state i

To estimate D_i, we used data from Ribaudo et al. (2008) and the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (FHWAR) (U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, U.S. Census Bureau, 2002; 2007). The duck hunting day estimates were deducted from these estimates to get nonwaterfowl hunting days by state.

Average nonwaterfowl hunting day values (V_i) are based on the average of estimates for small- and big-game hunting reported in the Rosenberger and Loomis (2001) meta-analysis. Converted to 2010 dollars using the Consumer Price Index, these estimates are \$45.19 per day in the Southeast (Virginia and West Virginia) and \$52.32 per day in the other states.

The model summarized in equation 1.2 uses estimates of the percentage point change in forest cover per state in the Chesapeake Bay watershed to estimate the increase in value of hunting land (i.e., public and private land used for hunting). By setting ΔF equal to 1 acre in equation 1.2, the incremental annual value of nonwaterfowl hunting per acre of additional forest cover can be estimated. These resulting estimates are reported in Table A-5.

Table A-5. Annual Value of Nonwaterfowl Hunting Services in the Chesapeake Bay Watershed (2008 $)

State	(D_i) Estimated Nonwaterfowl Hunting Days in 2008 (000s)	(V_i) Per-Day Value of Nonwaterfowl Hunting (2010 $)	Aggregate Annual Value of Nonwaterfowl Hunting Days (2010 $)	Incremental Value of Nonwaterfowl Hunting per Additional Forest Acre (2010 $)
DE	212	$52.32	$11,092,566	$1.15
MD	1,825	$52.32	$95,502,031	$1.98
NY	3,114	$52.32	$162,908,344	$4.84
PA	6,835	$52.32	$357,579,328	$3.40
VA	3,698	$45.19	$167,089,839	$1.28
WV	649	$45.19	$29,348,078	$1.94

APPENDIX B.
DATA QUALITY ASSURANCE

None of the analyses conducted for this report required or involved primary data collection, either from environmental media (such as water quality sampling or monitoring) or from human subjects (such as through household surveys). Instead, the analysis relied on secondary data sources, with the main ones being datasets created by the EPA's Chesapeake Bay Program Office (CBPO), either as output or input to Phase 5.3.2 Chesapeake Bay Watershed Model (CBWM). These data have either been posted on the Program's public ftp site (ftp://ftp.chesapeakebay.net/Modeling/) or provided by CBPO staff and have therefore been developed in compliance with CBPO's data quality assurance procedures. These data include CBWM estimates of nutrient loads by land-river segment and land use category and policy scenario runs, water quality by land-river segment, nutrient attenuation factors, best management practice (BMP) application rates, point source loads and treatment technology options and costs, BMP effectiveness rates, and BMP unit costs. Other secondary sources of data are reported in the reference sections of this report and include published federal government reports and peer-reviewed journal publications.

The datasets created as part of the analyses discussed in this report were all generated using established software programs – Microsoft Excel, SAS, and GAMS. Following QA procedures established within RTI, the batch program files are created and documented using a template format that requires the program author to specify (1) the filename and server location, (2) the RTI project number, (3) the author name, (4) the dates of the initial program and most recent update, (5) the purpose of the program, and (6) the names and server location for input and output files. In addition, subroutines within the program are commented to describe each step in plain language. Intermediate and all final datasets generated with these programs were routinely verified and validated by reviewing summary statistics and conducting consistency checks, and they were stored in a project file on a secure RTI server accessible to team members.

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